FRAMEWORK FOR ENVIRONMENTAL ACCOUNTS FOR AGRICULTURE

Final Report

submitted to

Department for Environment, Food and Rural Affairs (UK)
Department of Agriculture and Rural Development (Northern Ireland)
Scottish Executive
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Executive Summary

E.1 Introduction

This study ‘A Framework for Environmental Accounts for Agriculture’ examines the potential application of monetised environmental accounting to the UK agricultural sector. Undertaken for Defra, DARDNI, Scottish Executive and the Welsh Assembly, the research responds to terms of reference that call for “a study to identify data sources for the environmental impacts of agriculture and to develop methodologies that would enable us to produce an account to give an adjustment to the aggregate agricultural accounts showing this impact”.

The main driver for creating environmental accounts is the recognition that the current national accounting system does not reflect the full costs and benefits to society of economic activities, and, therefore, is an inadequate indicator of well being or true economic progress. Given the primary importance of traditional accounting indicators such as Gross Domestic Product (GDP) and Net Domestic Product (NDP) in public policy making, adjustments of these measures for environmental outcomes of economic activities are a step towards a better understanding of the sustainability (or otherwise) of economic development.

The work undertaken for this study draws from the large body of literature on green accounting which is grounded in the concept of sustainability, and which has sought to identify greener measures of national wealth and income\(^1\). Practical attempts to operationalise these concepts include those undertaken by the United Nations’ Statistical Office (UNSTAT) and its System of Environmental and Economic Accounts (SEEA), as well as the World Bank’s annual cross-country estimates of genuine (or adjusted net) saving and its components. While many countries have satellite environmental accounts that record environmental inputs and outputs, as with the Environmental Chapter of the UK Agricultural Accounts, none have yet attempted to integrate these within the final accounts, although the potential benefits are well recognised.

Once fully developed, a monetised environmental account for agriculture would be capable of providing:

- an economic measure of the sustainability of agriculture and a truer measure of the quality of life;
- an indication of the extent to which agriculture is a net contributor to the nation’s wellbeing as well as how it affects the welfare generated by other sectors;
- information that can be used for priority setting within agricultural policy; and
- inputs to cost benefit analysis for agricultural and related environmental policies.

As this report shows, current limitations on data and incomplete understanding of the linkages between agricultural practices and inputs, and environmental and economic outcomes mean that these remain goals, rather than reality, for the moment.

Past attempts at monetary Environmental Accounts of UK agriculture and recent attempts to place monetary values on the environmental impacts of UK agriculture provide a useful starting point for this exercise\(^2\). However, the study expands on previous research in a number of respects: (i) by focusing on positive as well as negative impacts of agriculture; (ii) by taking a systematic approach to identifying the accounting framework and how this would apply to a particular sector; (iii) by undertaking a wide review of the economic valuation literature and (iv) by presenting recommendations for future research.

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\(^1\) Examples of such measures include green Net National Product (gNNP) and genuine savings indicators.

E.2 Study Methodology

A four step methodology is employed, where each of the steps is covered in a separate section of the main of the report, as indicated in brackets:

Step 1: **Accounting framework** (Section 2) - identifying the accounting adjustments that would be required to reflect the changes in income and wealth resulting from agriculture's impacts on the environment;

Step 2: **Environmental impacts** (Section 3) - identifying the environmental impacts of agriculture and reviewing the physical data sources to select the appropriate data for environmental accounting;

Step 3: **Economic value of environmental impacts** (Section 4) - establishing the economic outcomes of environmental impacts and reviewing the economic valuation literature to select the appropriate monetary values for the accounting framework; and

Step 4: **Calculation of accounting adjustments** (Section 5) - combining physical and economic data for the most recent year to arrive at adjustments in income that could be applied to the existing accounts.

Finally, in Section 6, the gaps in the data are identified and recommendations are made to fill these gaps in recognition of the infancy of accounting exercises of this kind. The report also contains six Appendixes presenting supplementary information and data.

E.3 Accounting Framework

The framework for environmental accounts for agriculture starts by recognising the three main roles of environmental assets as serving a:

- **resource function**, whereby the environment provides the raw materials that are transformed by the economy to produce goods and services;
- **sink function** whereby pollution generated by production and consumption is assimilated by the environmental media of air, water or land; and
- **service function**, which provides both survival functions and amenity functions, such as recreation.

The resource functions (or raw materials) are generally provided as market goods that are paid for in the economy and, are therefore, already included in the accounts; whereas, the sink and service functions are not marketed, not priced and, are therefore, absent from the current set of accounts. Thus environmental accounts for agriculture should aim to value the sink and service functions provided by the environment. In its simplest form the value of these services or functions is the product of society’s willingness to pay for a unit of these services and their current level of provision.

For the accounting framework, environmental service and sink functions impacted upon by agriculture were considered systematically according to key impact headings (see Table E.1). From the table, agricultural activities either maintain or enhance environmental assets (indicated by positive sign in the table, e.g. through maintenance of landscapes) or serve to reduce them (indicated by negative sign in the table, e.g. by reducing air quality), where assets are water, air or land-based. Impact categories such as waste and nuisance are not linked to any one asset, but can impact upon a range of asset functions.

A key distinction to make when building environmental accounts for agriculture is between those **assets that are under agricultural control** (or can be attributed to agriculture) and those that are not. Land-based assets managed by agriculture can be thought of as assets of the agricultural sector. This is particularly the case in the UK where landscapes, habitats and species are often intimately linked to past and current agricultural activity. Thus, it follows that the positive environmental services that flow from these assets should be attributed as additional income to the sector and, that this income is reported net of any losses. It also follows that the positive service flows from other assets not attributable to agriculture are not recorded as income to the sector (such as service and sink functions provided by the aquatic environment). In fact, it is generally the case that agriculture has the effect of reducing these sink and service functions through release of pollutants to the environment, with the result of reducing society’s welfare and/or the productivity of other sectors, as explored below.
Table E.1: Categorisation of Environmental Impacts of Agriculture

<table>
<thead>
<tr>
<th>Main categories</th>
<th>Environmental outcomes associated with agriculture</th>
</tr>
</thead>
<tbody>
<tr>
<td>I. Water</td>
<td>i. change in water quality (+/-)</td>
</tr>
<tr>
<td></td>
<td>ii. change in water availability (+/-)</td>
</tr>
<tr>
<td>II. Air</td>
<td>i. change in air quality - local and global impacts (+/-)</td>
</tr>
<tr>
<td></td>
<td>ii. dust and allergens (-)</td>
</tr>
<tr>
<td>III. Soil</td>
<td>i. change in soil composition and attributes (+/-)</td>
</tr>
<tr>
<td></td>
<td>ii. soil loss or gain (-/+))</td>
</tr>
<tr>
<td>IV Landscape</td>
<td>i. change in landscape (+/-)</td>
</tr>
<tr>
<td></td>
<td>ii. maintenance of landscape (+)</td>
</tr>
<tr>
<td>V. Habitats and species</td>
<td>i. change in biodiversity (+/-)</td>
</tr>
<tr>
<td></td>
<td>ii. maintenance of biodiversity (+)</td>
</tr>
<tr>
<td>VI. Waste</td>
<td>i. generation of waste (-)</td>
</tr>
<tr>
<td></td>
<td>ii. disposal of waste (-/+))</td>
</tr>
<tr>
<td>VII. Nuisance</td>
<td>i. odour (-)</td>
</tr>
<tr>
<td></td>
<td>ii. noise (-)</td>
</tr>
<tr>
<td>VIII. Resource use</td>
<td>i. depletion of non renewable resources (-)</td>
</tr>
<tr>
<td></td>
<td>ii. provision of alternative resources (e.g. renewable energy) (+)</td>
</tr>
</tbody>
</table>

This gives rise to two types of accounting adjustment: one for service flows, i.e. for services that emanate from the land based stock of assets, and one for changes to stocks, i.e. where agriculture reduces (enhances) the ability of assets to provide environmental services either by impacting reducing (enhancing) the quality of the asset or by changing the quantity of the asset; the latter applies particularly in the case of land and non-renewable resources.

Another distinction to make concerns who the costs and benefits of environmental outcomes accrue to. Whether positive or negative, impacts of agriculture on the environment can affect either (i) society in general, resulting in an increase/decrease social welfare or (ii) other sectors, resulting in gains/losses to the productivity of those sectors.

Finally, in arriving at a set of environmental adjustments for the accounts consideration is made about whether the accounting adjustment is a measure of wealth or income. The only measure of wealth that could be attributed to agriculture would be for changes in the quantity of land-based stock, where the change in wealth is measured as the loss of future earnings from the lost stock. For all other impact categories only income measures are relevant to agriculture; either 'negative income' that arises through reducing service flows from natural assets or 'positive income' through provision of services from agriculturally managed environmental assets.

Table E.2 presents the set of adjustments that would need to be calculated in order to integrate the environmental impacts of agriculture into the current set of accounts. These adjustments could be made to the balance sheet, the production accounts or any other account to give a truer picture of income and wealth in the sector.

Section 2 of the main report treats each of these accounting adjustments in turn tackling issues such as the relevant baseline against which each should be measured and the treatment of subsidies and taxes within this framework, providing insights from the literature.
Table E.2: Adjustments to the Agricultural Accounts

<table>
<thead>
<tr>
<th>Adjustments for Welfare Impacts on Society:</th>
<th>Service flow</th>
<th>Stock change (quantity and quality)</th>
</tr>
</thead>
<tbody>
<tr>
<td>I. Water</td>
<td>Flow not attributable to agriculture</td>
<td>Value of water pollution arising from agricultural production</td>
</tr>
<tr>
<td></td>
<td>Flow not attributable to agriculture</td>
<td>Value of agricultural water abstraction</td>
</tr>
<tr>
<td>II. Air</td>
<td>Flow not attributable to agriculture</td>
<td>Value of air pollution arising from agricultural production</td>
</tr>
<tr>
<td>III. Soil</td>
<td>Impact of (net) soil erosion on-farm on current yields is already accounted for</td>
<td>Value of (net) soil erosion on-farm on future yields</td>
</tr>
<tr>
<td>IV. Landscape</td>
<td>Value of landscape amenity services by the current provision of landscapes (within the agricultural sector)</td>
<td>Value of (net) change in landscape amenities</td>
</tr>
<tr>
<td>V. Habitats and Species</td>
<td>Value of habitat and species protection services provided by current land-use (within the agricultural sector)</td>
<td>Value of (net) change in habitats and species</td>
</tr>
<tr>
<td>VI. Waste</td>
<td>Value of waste pollution and disamenity arising from agricultural production</td>
<td></td>
</tr>
<tr>
<td>VII. Nuisance</td>
<td>Value of noise and odour disamenity arising from agricultural production</td>
<td></td>
</tr>
</tbody>
</table>

Adjustments for Impacts on Other Sectors (productivity gain or loss):

| I. Water                                   | (-) Cost of clean up for water pollution |
|                                            | (-) Costs of flooding |
| II. Air                                    | Included in value of stock change above |
| III. Soil                                  | Impact of (net) soil erosion on-farm on current yields is already in existing accounts |
|                                            | (-) Cost of off-site soil erosion (cost of dredging streams, etc) |
| IV. Landscape                              | (+) e.g. Value of landscape to tourism |
| V. Habitats and Species                    | (+) e.g. Value of habitats and species to tourism |
| VI. Waste                                  | None |
| VII. Nuisance                              | (-) e.g. cost of dealing with nuisance complaints |

E.4  Environmental Impacts

Agriculture's role in the UK economy is small, accounting for only 0.8% of the Gross Domestic Product in 2000. But in terms of natural resource use its role is significant, accounting for a high percentage of land use (e.g. 74% of land in the UK). While agricultural productivity has improved substantially, the introduction of new farming methods has often led to environmental degradation such as soil erosion and water pollution. However, agriculture also has positive impacts including helping to create and manage landscaped and wildlife habitats; and maintaining soils under agriculture which can, amongst other things, act as a sink for greenhouse gases and provide water filtering and retention functions.

In examining interactions across a broad front, several attributes of the agriculture-environment relationship, which make it more difficult to attribute a precise environmental impact to agriculture than is the case for most other economic activities, need to be considered:

- **Agriculture is relatively diverse**; there are more than 200,000 holdings in the UK containing a range of different enterprise types adapted to different environmental and economic conditions;

- **Farming forms a part of an ecosystem** rather than being external to it, unlike most other economic activities;
The spatial distribution of agriculture is such that similar farming types will occur in a range of different environmental conditions, and so a specific practice in one location can have significantly different environmental outcomes in another;

- The timescale over which environmental impacts become apparent also varies greatly and consequently it is difficult to be precise about the impact of farming in any given year;
- Some environmental impacts are not confined to one particular medium - it is often necessary to make links between different impacts and there may be multiple references to a single input such as herbicide;
- Agricultural impacts on the environment are mainly diffuse - data are relatively expensive and reliable time series information relatively hard to come by; and
- In seeking to quantify impacts it may be necessary to apportion a share of the final outcome to agriculture but in few cases can this be very accurate.

It is therefore necessary to make a considerable number of judgments, estimates, approximations and assumptions in order to provide some basis for the quantification of environmental impacts.

The framework employed in this study for classifying the complex interactions between agriculture and the environment is the DPSIR model (Driving forces, Pressures, State of the Environment, Impact on final end points and policy Responses) shown in Figure E.1 below, which is widely used by the European Environment Agency 3 and the European Commission, in characterising environmental impacts. Understanding and mapping environmental pathways is key to the interpretation of existing data sources and important for directing future research.

![DPSIR Model](image)

Essentially what the DPSIR framework provides is a map of the environmental pathways that demonstrates how, for example, a change in water quality may be linked to changes in agricultural practices, such as the application of fertilisers. In the framework, what has until now been called an environmental impact is termed a change in the state of the environment (S). An impact, in turn, is referred to as the resultant change in human well-being or welfare - in other words, the economic impact of the environmental change. Data relating to the physical environment are, therefore, limited to D, P and S, as reported in Section 3 (and also includes response data, i.e. the number of hectares of agri-environment schemes). The review of the ‘Impact’ data within the DPSIR framework is discussed in Section 4, which shows that the challenge is finding economic data that match the physical data.

For the most part, the DPSIR framework can only point to associations in environmental pathways. An exception is the case of air pollution, where decades of research with the aid of air pollution models, have produced dose-response functions that provide a reasonably accurate measure of the environmental and human health impacts that are causally linked to concentrations of, or exposure to, some pollutants. Since agriculture is only one of the many sources of environmental impacts, in addition to understanding the agriculture-environment relationship, we need to also know the share of agricultural factors causing a

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3 “Particularly useful for policy-makers, DPSIR builds on the existing OECD model and offers a basis for analysing the inter-related factors that impact on the environment.” - from the EEA website: http://org.eea.eu.int/documents/brochure/brochure_reason.html
particular impact. Such data are reported in Section 3, and are termed linking data. An example of linking data is a research result that points to agriculture as contributing to 43% of pesticides in the water environment.

Ideally, it would be possible to map the DPSIR model for each agricultural practice or sub-sector (dairy, maize etc.), as this would link environmental accounts directly with main sector accounts. However, there are a great many unknowns, and for the most part, it is only possible to draw associations between drivers and emissions, drivers and changes in the state of the environment, or drivers and final impacts. Thus, in the absence of clear links between pressures, state and impact, a practical approach to understand the impacts of agriculture on the environment is to focus on the classification of the impacts themselves, as presented in Table E.1 above.

An array of primary data was collected for the study and selection criteria applied in order to arrive at the set of data presented in full in Annex 4 of the main report. From this initial set it was necessary to select data that could be matched to the economic data collected in the next step. The data were selected if it met one or more of the following data criteria: (i) that the data were from a reliable source; (ii) that the data matched with the economic data in terms of units and changes in the environment being valued; (iii) that the data were updated on a regular basis, or part of a survey; (iv) that the data provide a convenient accounting metric; (v) that the data were available across all countries in the UK; and (vi) that the data were already presented in the environmental chapter of the UK agricultural accounts.

Section 3 of the main report is organised by impact category, under which the interactions between agriculture and the environment are discussed and data selected for use in the accounts presented.

E.5 Economic Value of Environmental Impacts

The economic value of an environmental impact is estimated using data on individuals’ (or households’) preferences for that impact. Preferences, in turn, are measured by people’s Willingness to Pay (WTP) for environmental goods and services. Similarly, preferences can be measured by people’s Willingness to Accept Compensation (WTA) to tolerate a negative environmental impact or to forgo a positive one.

The first source of economic data on people’s preferences is actual market data for those environmental impacts that are traded or reflected in actual markets. These ‘marketed’ services of the environment include the supply of drinking water and formal recreation or tourism. In fact such market data (costs and prices) are akin to WTP or WTA of individuals in the sense that market prices reflect the WTP of buyers and WTA of sellers. Generally an individual will only consume a good or a service when its price is less than or equal to his/her WTP. That is, a given good is purchased if they perceive that the benefit yielded by its consumption (as measured by their maximum WTP for that unit of consumption) exceeds the cost of consumption (as measured by the price of the unit of consumption). When the price is less than WTP, consumers receive a surplus of benefit. This surplus is known as ‘consumer surplus’, which is equal to WTP minus market price. Therefore, in most cases, the market price paid for a good is only a lower-bound estimate of an individual’s WTP for a good.

Non-market values, i.e. the values for environmental goods and services that are currently external to the market (provided for free), can be estimated using one of two techniques: revealed or stated preference techniques, which measure economic welfare and produce economic values as opposed to market prices. These techniques aim to measure individuals’ preferences directly and capture consumer surplus as well as market price based expenditure when the latter exists. Revealed preference techniques use existing markets (e.g. housing or travel) as surrogates for estimating the economic values for the environment. Stated preference techniques create hypothetical, or simulated, markets by way of surveys to elicit society’s valuations for environmental changes. For instance, an example might be asking households to state their willingness to pay increased water bills for improvements in local river water quality.

Since it is not feasible to undertake an original economic valuation exercise for every policy question and project that arises in relation to non-market goods, a procedure called benefits transfer has been developed to enable appropriate, transparent and consistent use of economic value estimates from the literature. Benefits transfer is the approach employed in this study and essentially involves borrowing the estimates of non-market values from previous studies and applying them to a new, but similar, context.

4 In economic terminology ‘price’ is not necessarily equal to ‘value’. Valuation techniques seek to provide recognised economic measures of value such as consumer surplus.
Section 4 of the report reviews the literature of economic valuation studies selecting from Annex 5 where over 100 studies are summarised. For each impact category studies and estimates are selected for application to the accounting framework using the following selection criteria:

- **Study subject** - the focus of the review is studies that consider either positive or negative impacts of agricultural activity, or alternatively, assess impacts generated by other sectors that are similar to those generated by agriculture;
- **Study context** - it is important to match the valuation context and the changes in provision of the quantity or quality of a good in an original study to the new valuation context;
- **Study origin** - typically it is recommended that benefits transfer exercises in a UK context use studies relating to the UK;
- **Study methodology** - studies should be grounded in the theory of welfare economics and use robust valuation methodologies; and
- **Study age** - the date of the original study is recent since the design and implementation of economic valuation techniques have developed over the past 15 years or so.

In a number of cases adjustments to these estimates are required in order to provide a match with the units in which the physical data are presented, and these are detailed in Section 4 of the report. The data selected are both market and non-market data. For the most part non-market data only are applied to the valuation of impacts on society (or welfare changes), while market data are used only for the valuation of impacts on other sectors.

### E.6 Calculation of Accounting Adjustments

The calculations detailed Section 5 of the main report provide monetary estimates of the environmental impacts of agriculture (subject to data availability) that can be used in an income accounting exercise, even though no effort is made here to include these in the accounts or to calculate any measures of adjusted income. The calculations show the economic value of the positive environmental services provided by agriculture as well as the negative service flows that result from depreciation of natural assets such as air and water. This is manifested by the impacts on the welfare of society in general and productivity gains and losses to other sectors as shown in Table E.3 below. In practical terms, this means that the calculations presented here are in £ per year terms (representing service flows or changes to income) and no present value calculations are undertaken to account for changes in wealth due to a change in a given stock of environmental capital.

In addition to this definition of the environmental and accounting boundaries of the calculations, we also need to establish the relevant affected population. For most impacts, the entire UK population is taken as the relevant population. Exemptions include the larger (global or European) populations impacted by global and regional air pollution from the UK. These population estimates are embedded in the unit economic costs used here.

In line with the objectives of the study, country-level breakdown of the calculations is presented in Table E.3. This breakdown has not been possible for all impacts due to the physical data not being collected or presented at the country level. Where country level physical data are available, it is sometimes the case that economic data for the UK as a whole or from a particular location are applied uniformly to all countries.

As with all exercises that entail any empirical or statistical procedures, sensitivity analysis should be conducted in order to reflect areas of uncertainty. Therefore, where possible, from the available literature, a range of economic estimates have been used to calculate lower and upper bound values as well as a central one. Expert judgement has been employed to try and narrow these ranges as much as possible, and all summary data are presented in using mid points or central estimates.

Table E.3 presents the monetary value estimates that reflect the interactions of agriculture with the environment. The table is complete only so far as the available physical and economic data allow. Where cells are shaded grey this indicates missing country level data, and where cells are merged this indicates that the data cover more than one country (e.g. it is often the case that only UK level data are available). Notes to the table indicate some overlap in values calculated for habitats and species. Elsewhere in the table distinctions are made between coastal and inland water pollution, local and global air pollution, and

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5 Note that the only relevant calculations here would be changes in wealth to agriculture from changes to land-based assets.
between sectors upon which water clean up costs are imposed. Note that monetary values presented in the table have been rounded up or down for presentation.

### Table E.3: Estimated Monetary Adjustments to Agricultural Accounts (£million 2003, central estimates)

<table>
<thead>
<tr>
<th>Impact Category</th>
<th>Accounting Adjustment</th>
<th>England (E)</th>
<th>Wales (W)</th>
<th>Scotland (Sc)</th>
<th>N. Ireland (NI)</th>
<th>UK</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Adjustments for Welfare Impacts on Society:</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>I. Water</td>
<td>Value of water pollution arising from agricultural production</td>
<td>-£48</td>
<td>-£1</td>
<td>-£14</td>
<td>-£7</td>
<td>-£71</td>
</tr>
<tr>
<td></td>
<td>Inland</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Coastal</td>
<td>-£3</td>
<td></td>
<td></td>
<td></td>
<td>E &amp; W only</td>
</tr>
<tr>
<td></td>
<td>Value of agricultural water abstraction</td>
<td></td>
<td></td>
<td></td>
<td>-£36</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Global</td>
<td>-£543</td>
<td>-£109</td>
<td>-£143</td>
<td>-£95</td>
<td>-£889</td>
</tr>
<tr>
<td></td>
<td>Regional/Local</td>
<td>-£43</td>
<td>-£7</td>
<td>-£10</td>
<td>-£7</td>
<td>-£67</td>
</tr>
<tr>
<td>II. Air</td>
<td>Value of air pollution arising from agricultural production</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Global</td>
<td>-£543</td>
<td>-£109</td>
<td>-£143</td>
<td>-£95</td>
<td>-£889</td>
</tr>
<tr>
<td></td>
<td>Regional/Local</td>
<td>-£43</td>
<td>-£7</td>
<td>-£10</td>
<td>-£7</td>
<td>-£67</td>
</tr>
<tr>
<td>III. Soil</td>
<td>Value of (net) soil erosion on-farm on future yields</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>n/e</td>
</tr>
<tr>
<td>IV. Landscape*</td>
<td>Value of landscape amenity services by the current provision of landscapes (within the agricultural sector)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>普托</td>
<td>+£124</td>
<td>+£321</td>
<td>+£45</td>
<td></td>
<td>+£488</td>
</tr>
<tr>
<td>V. Habitats and Species*</td>
<td>Value of habitat and species protection services provided by current land-use (within the agricultural sector)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Habitats</td>
<td>+£225</td>
<td></td>
<td></td>
<td></td>
<td>E only</td>
</tr>
<tr>
<td></td>
<td>Species</td>
<td></td>
<td></td>
<td></td>
<td>+£307</td>
<td></td>
</tr>
<tr>
<td>VI. Waste</td>
<td>Value of waste pollution and disamenity arising from agricultural production</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>-£15</td>
</tr>
<tr>
<td>VII. Nuisance</td>
<td>Value of noise and odour disamenity arising from agricultural production</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>n/e</td>
</tr>
<tr>
<td><strong>Adjustments for Impacts on Other Sectors:</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>I. Water</td>
<td>Cost of water pollution clean up costs</td>
<td>-£2</td>
<td>-£0.1</td>
<td></td>
<td></td>
<td>E, W &amp; Sc only</td>
</tr>
<tr>
<td></td>
<td>Costs of flooding</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>-£181</td>
</tr>
<tr>
<td>II. Air</td>
<td>Included in measure above</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>III. Soil</td>
<td>Cost of off-site soil erosion</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>-£9</td>
</tr>
<tr>
<td>IV. Landscape</td>
<td>(+) e.g. Value of landscape to tourism</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>n/e</td>
</tr>
<tr>
<td>V. Habitats and Species</td>
<td>(+) e.g. Value of habitats and species to tourism</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>n/e</td>
</tr>
<tr>
<td>VI. Waste</td>
<td>None</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>VII. Nuisance</td>
<td>(-) e.g. cost of dealing with nuisance complaints</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>n/e</td>
</tr>
</tbody>
</table>

* These two categories overlap, which means that some element of habitat and species will be captured in the landscape valuations, and some element of landscape value will be captured in the habitat valuations.

n/e: not estimated due to lack of physical and/or economic data.

**Water:**

Table E.3 shows that the largest cost of agriculture in terms of water quality and quantity is its share in the cost of flooding. This is also the least certain of the impact categories depending entirely on linking data that merit further research. Another interesting observation of the table is that the productivity impacts on other sectors (using market data) are higher than welfare impacts on society (using non-market data). This is partly because of the incompleteness of reliable economic data on the impact of agriculture on marine ecology (currently the adjustment for coastal pollution reflects only impacts on human health and wellbeing from faecal contamination). Nonetheless, the results point to significant gains in terms of both improvements to social welfare and economic gains to other sectors if water pollution from agriculture can be further reduced. In terms of abstraction costs, agriculture accounts for 14% of total water resource use by industry, which provides some perspective. The monetary estimates come from a study that was specific to South East England, and, hence, could be over or underestimating the total costs when these are applied across the whole UK.
Air:

The range of air pollution costs presented here is high compared with £585 million estimated by Hartridge and Pearce (2001) and £393 million estimated by Pretty et al (2000). This increase is partly due to updated unit damage cost estimates and partly due to a larger set of pollutants and impacts being valued. Among the pollutants, the greatest impact in monetary terms appears to arise from nitrous oxide emissions due to its high damage costs. The second largest contribution to the total cost comes from methane, again, due to the high economic cost, level of emissions and global warming potential. In terms of the monetary value of impacts, carbon and ammonia have fairly similar aggregate values for the UK. The costs of all greenhouse gas emissions are affected by the cost of carbon, for which the current official Defra estimates have been used in this study.

As Table E.3 shows the majority of the cost of air pollution reflects the global damages of greenhouse gas emissions. While £889 million represents a sizeable cost, it is worth keeping in mind that agriculture accounts for only 7% of the UK's greenhouse gas emissions. Thus, if the same exercise is undertaken for national accounts, £889 million would be dwarfed by total costs. If all sectors undertook accounting exercises of this type then it would be useful to compare how the costs of agricultural greenhouse gas emissions compared with other sectors. Essentially, if the benefit cost ratio of reducing emissions in agriculture is lower than in other sectors, some action may be warranted.

While these costs appear large, they are still likely to be underestimates, as a number of the impacts of air pollution are not included in the economic data applied here due to a lack of research. These impacts include: impacts on ecosystems, damage to cultural heritage, effects of ozone on materials and additional health effects. Another missing element from the air impact category is the economic value of the impacts of dust and allergens on human health. Without economic data, or adequate physical data, it is hard to say what the extent of this gap is. Therefore, further research is warranted.

Soil:

The cost figure presented in the table refers to the off-farm impacts of soil erosion on other sectors. The on-farm impacts of soil erosion on the current yields are already reflected within income in the existing sector accounts, while the impacts of soil erosion on future yields are not captured in the existing accounts and not possible to quantify here due to a paucity of economic data and research in this area.

Landscape, habitats and species:

Large positive service flows are attributed to agriculture through the provision of landscapes, habitats and species. The approach employed reflects the extensive coverage of agriculturally managed land in the UK (comprising 74% of all land) and the reliance of many habitats and species on the continuation of certain agricultural management practices. Changes in these practices have led to the decline of farm species (such as birds) and the degradation of semi-natural habitats. These negative impacts would appear in the accounts with the availability of time-series data which would show falling incomes from these environmental services over time.

The scale of the benefit accrued from agricultural practices that maintain landscapes, habitats and species valued by society, can be compared with the payments made to farmers to provide these services. While the data only allow a crude comparison, total payments to agri-environment schemes totalled £387 million in 2002 (Defra et al, 2004), which is far outweighed by the magnitude benefits recorded in Table E.3. Currently the data on the physical environment is the most limiting factor in achieving time series comparison of such costs and benefits.

Waste:

Waste from agriculture amounted to approximately 22% of total waste produced in the UK (1998/99 figures). Of the agriculture’s share, 92.5 million tonnes is animal and vegetable waste and 1 million tonnes is general waste. For better coverage of the waste generated and its environmental impacts, further research is needed. As the adjustments presented here only account for the general waste category, the total disamenity value of waste produced by agriculture is likely to be underestimated. While the landfill tax will ‘internalise’ to some extent the environmental costs of waste disposal, this does not change the fact that the environmental costs per tonne of waste remain the same regardless of whether they are paid for. Therefore, despite the existence of a policy instrument to deal with the environmental impacts of waste disposal, the economic costs are included in the accounting adjustments.
E7.0 Gap Analysis

Section 6 of the main report presents gaps in the physical and economic data under each impact heading, and also makes some general points about the data, which are summarised here. Four main types of gaps present themselves from the analysis undertaken for this study. Some examples of what these entail and the challenges presented is provided here:

- **the scope of the analysis**: extending the scope of analysis, which is currently limited to within the farm gate, might see impacts such as the environmental impacts of the transport of farm goods included;

- **the geographical and sub-sector disaggregation of the accounts**: this task is complicated by the local nature of many impacts and the difficulty of making regional or national generalisations for particular farm practices and sectors, even though work on agricultural indicators (see OECD, 2000) could be explored further with a view towards filling this gap;

- **the annual nature of the accounts**: as data are often difficult and expensive to collect on a regular basis, the time period between accounting periods for some impacts can be over 10 years. While it may be unrealistic to expect annual data, especially for impacts that have long lag times, more regular survey work and data collection should be a priority; and

- **physical and economic data**: there are numerous impacts for which physical and economic data are lacking. These gaps on the whole arise as a result of the following reasons:

  - There may be no physical or economic data about an environmental impact. For example, there is no information about the effect of current on-farm soil erosion on future crop yields. This is the type of gap that requires the most effort to fill since the gap is the greatest. On the positive side, the efforts on filling this type of gap may benefit other policy analysis as well as the impacts of agriculture.
  - There may be physical data about an environmental impact but the share of agriculture as one of the sources of that impact is not known. In other words, the linking data that exist for some impacts do not exist for others. For example, it may be argued that agricultural landscapes have a positive impact on tourism sector. While there are market data about the economic value of the tourism sector, it is not known what contribution agriculture makes to this.
  - There may be physical data about an environmental impact and the share of agriculture as one of the sources of that impact may also be known but there may be no economic valuation data. In fact, as For example, intensive agricultural land has any economic value. This type of gap sends signals to economists in planning and undertaking future research.

These data gaps are pervasive throughout all impact categories, and also have implications for the ability of the data to provide annual estimates, geographical disaggregation of estimates, and so on. The following provides a few areas that are prioritised for future research. Further treatment of these is provided in Section 6 of the main report.

- Create designated contact points within each major body which collects and collates data that would be able to assist with the identification of the most up-to-date and appropriate data sets, and coordinate to collect data more regularly and consistently to avoid current inconsistencies and any future duplication of effort.

- Include in the Countryside Survey questions on whether the different habitats surveyed were under agricultural management and, ideally, attempt to measure qualitative change for these habitats in terms of biodiversity (habitat and species) that could be linked to land and landscape types. These measurements could learn from OECD work on indicators (OECD, 2000), notably the Natural Capital Index.

- More research is required to establish the nature of the relationships between agricultural practice, non-farming factors and the population dynamics of different species, in order to augment current data on keystone species, such as farmland birds.

- New work in establishing transferable economic values for changes in river water quality should employ the Environment Agency’s GQA index as the measure of quality as it is a good accounting
measure and groups different environmental effects together. Ideally a benefits transfer model similar to that developed for the ELF model (IERM and SAC, 2001) for landscapes could be developed for the water environment, which would allow sensitivity for different regions to be incorporated.

- Additional economic research linking agriculture’s contribution to the overall water quality, and share of the economic costs, is also required to improve calculations of accounting aggregates. Considering that the results of such a study could also be used in future for the Periodic Review process that water companies go through every five years, as well as feeding into the Water Framework Directive requirements, this new research becomes an important priority.

- More research is required to fill the gap in knowledge about the economic value of ecological impacts on the marine environment such as changes in marine flora and fauna, and specifically agriculture’s contribution to these changes.

- Research is required to fill gaps in data about the potential social value of intensive agrarian landscapes. The current literature concentrates on extensive agriculture such as the semi-natural habitats and the designated areas. Further research on this area could make use of computer generated visual aids to allow a differentiation between different landscapes, including future possible landscapes. Such work could augment the currently employed benefits transfer model, ELF and benefit from technology employed by the ZICER Institute at the University of East Anglia.

These are only a select few of the suggestions for future research from the main report, but could provide a good starting point. A full set of environmental accounts for agriculture, complete with an understanding of the economic value of the environmental impacts would be an invaluable resource for the UK. Already, the data selected for this study should provide a useful reference point, but overall we are only at the beginning of a longer process and much data and coordination are required to see the full benefits of a monetised environmental account. Moreover, the benefits of such an analysis will not be limited to agriculture as other sectors and government departments are sure to see the benefits made to collect more environmental and economic data to improve our understanding of these complex agriculture-environment-economy interactions.
Table of Contents

1  INTRODUCTION
   1.1 ENVIRONMENTAL ACCOUNTING – AN OVERVIEW AND PROGRESS TO DATE .................................................. 2
   1.2 THE NEED FOR ENVIRONMENTAL ACCOUNTS FOR AGRICULTURE ....................................................... 3
   1.3 STUDY OBJECTIVES AND SCOPE ........................................................................................................... 5
   1.4 OVERVIEW OF STUDY METHODOLOGY AND REPORT STRUCTURE ........................................................... 6

2  ACCOUNTING FRAMEWORK ............................................................................................................................ 11
   2.1 BASIC ACCOUNTING FRAMEWORK FOR THE NATIONAL ECONOMY AND ITS IMPLICATIONS FOR AGRICULTURE ..........11
   2.2 BASIC FRAMEWORK FOR ENVIRONMENTAL ACCOUNTS FOR AGRICULTURE .............................................. 13
   2.3 ACCOUNTING FOR THE ENVIRONMENTAL IMPACTS OF AGRICULTURE ...................................................... 15
      2.3.1 Water ............................................................................................................................................. 17
      2.3.2 Air ............................................................................................................................................. 18
      2.3.3 Soil .......................................................................................................................................... 18
      2.3.4 Landscape ................................................................................................................................. 19
      2.3.5 Habitats and species ................................................................................................................... 19
      2.3.6 Waste ......................................................................................................................................... 20
      2.3.7 Nuisance ................................................................................................................................... 20
      2.3.8 Resource use ............................................................................................................................... 20
      2.3.9 Summary of accounting adjustments ......................................................................................... 201

   2.4 ESTABLISHING BOUNDARIES FOR THE ENVIRONMENTAL ACCOUNTS FOR AGRICULTURE ....................... 22
      2.4.1 How do we account for subsidies and taxes? ............................................................................. 22
      2.4.2 What is the appropriate baseline or the implied counterfactual? ............................................. 23
      2.4.3 Statutory requirements – who provides the benefits? ................................................................. 24
      2.4.4 What about other sectors' impact on agriculture? .................................................................. 25
      2.4.5 Should we be using marginal or average economic values? .................................................... 25

3  ENVIRONMENTAL IMPACTS OF AGRICULTURE ................................................................................................. 26
   3.1 OVERVIEW .............................................................................................................................................. 26
      3.1.1 Establishing Environmental Impacts ............................................................................................ 26
      3.1.2 Collecting Physical Data ............................................................................................................. 30
   3.2 SUMMARY OF ENVIRONMENTAL IMPACTS ............................................................................................. 32
      3.2.1 Water ......................................................................................................................................... 32
      3.2.2 Air pollution ............................................................................................................................... 36
      3.2.3 Soil .......................................................................................................................................... 38
      3.2.4 Landscape ................................................................................................................................. 40
      3.2.5 Habitats and species ................................................................................................................... 44
      3.2.6 Waste ......................................................................................................................................... 47
      3.2.7 Nuisance ................................................................................................................................... 49
      3.2.8 Resource use ............................................................................................................................... 49

4  ECONOMIC VALUE OF ENVIRONMENTAL IMPACTS ........................................................................................... 51
   4.1 ECONOMIC VALUATION ............................................................................................................................. 51
      4.1.1 Total Economic Value ................................................................................................................ 51
      4.1.2 Economic Valuation ................................................................................................................... 52
      4.1.3 Market versus Non-market Economic Data ............................................................................. 52

   4.2 THE USE OF BENEFITS TRANSFER FOR ENVIRONMENTAL ACCOUNTS FOR AGRICULTURE .................... 53
   4.3 REVIEW OF THE ECONOMIC DATA ......................................................................................................... 56
4.3.1 Water ................................................................. 56
4.3.2 Air ................................................................. 60
4.3.3 Soil ............................................................... 62
4.3.4 Landscape, Habitats and Species ......................... 62
4.3.5 Waste ............................................................ 68
4.3.6 Nuisance ........................................................ 68
4.3.7 Cultural heritage and archaeology ......................... 68

5 MONETARY ENVIRONMENTAL ACCOUNTS FOR AGRICULTURE .................................................. 70
5.1 APPROACH .......................................................................................................................... 70

5.2 Monetary Estimates of Environmental Impacts ................................................................. 71
  5.2.1 Water ......................................................................................................................... 71
  5.2.2 Air ............................................................................................................................. 75
  5.2.3 Soil .............................................................................................................................. 77
  5.2.4 Landscape, habitats and species ................................................................................ 77
  5.2.5 Waste ....................................................................................................................... 82

5.3 AGGREGATING ECONOMIC VALUES OF ENVIRONMENTAL IMPACTS ........................................ 82

6 GAP ANALYSIS ..................................................................................................................... 86
6.1 OBJECTIVES AND OVERVIEW OF GAP ANALYSIS  .................................................................. 86

6.2 GAPS IN THE PHYSICAL DATA ................................................................. 88
  6.2.1 Water quality and availability ............................................................... 88
  6.2.2 Air ............................................................................................................................. 89
  6.2.3 Soil .............................................................................................................................. 89
  6.2.4 Landscape, habitats and species ................................................................................ 89
  6.2.5 Waste ....................................................................................................................... 90

6.3 GAPS IN THE ECONOMIC DATA ................................................................. 90
  6.3.1 Water quality and availability ............................................................... 91
  6.3.2 Air ............................................................................................................................. 92
  6.3.3 Soil .............................................................................................................................. 92
  6.3.4 Landscape, habitats and species ................................................................................ 92
  6.3.5 Waste ....................................................................................................................... 93
  6.3.6 Nuisance ................................................................................................................ 93

REFERENCES ............................................................................................................................ 94

Annex 1: Review of Studies on Environmental Accounts for UK Agriculture
Annex 2: Environmental Accounting Issues
Annex 3: Classification of Environmental Impacts from Agriculture
Annex 4: Environment Data Tables
Annex 5: Review of Economic Studies
Annex 6: Calculations for Accounting Adjustments
1 Introduction

The guiding principle of agricultural, and other sectoral, policies is to maximise the net benefits of the sector to society and thereby contribute towards sustainable development. Central to this principle is a better understanding of the significant and complex interactions between the sector and the environment.

One way to contribute to this improved understanding is to revise national agricultural accounts in a way that reflects the full economic implications of the sector’s impacts on the environment. This report presents such an accounting framework and monetary estimates of the environmental costs and benefits of agriculture as far as the available data and literature allow.

This Section introduces the notion of environmental accounting, including a summary of efforts to date (Section 1.1) and then provides key arguments for and implications of undertaking such an exercise (Section 1.2). The Section continues with a presentation of the objectives and scope of the research (Section 1.3) and concludes with an overview of the methodology employed in the study (Section 1.4). In addition, Annex 1 reviews some of the environmental accounting studies for UK agriculture.

1.1 Environmental Accounting - An Overview and Progress to Date

As a result of the Brundtland Commission in 1987 and the United Nations Conference on Environment and Development in 1992, most governments world-wide have adopted sustainability as a national goal. While discussion continues regarding what it is that policy-makers have committed themselves to, Pezzey (1989) offers a widely accepted definition that a development path is sustainable if wellbeing per capita does not decline along that path. Achieving sustainability, in turn, has been equated with propositions regarding how an economy should manage its wealth over time. Guiding principles in this respect include that of weak sustainability, which emphasises changes in the real value of wealth in the aggregate, and strong sustainability, which (typically) also emphasises the conservation of critical natural capital, i.e. critically important resources for which there are essentially no substitutes.

Emerging interest in the measurement of sustainability has resulted in a great deal of progress in constructing indicators. A fundamental element of these efforts has been proposals for the greening of national accounts. This work arose from a concern that economic indicators, such as Gross National Product (GNP)\(^6\), do not reflect the depletion and degradation of the environment and so may lead to incorrect (and unsustainable) development decisions, in much the same way that cost benefit analyses that do not include people’s preferences for associated environmental changes may yield poor investment decisions. Hence, the early 1990s saw a growing recognition that the construction of indicators or green accounts to monitor progress towards sustainability or environmental goals was an essential element of any programme to integrate sustainable development into policy-thinking. However, few practical indicators or applications existed for this purpose. Those few contributions which were available (e.g. Repetto et al 1989) presented a novel and informative picture of development but arguably raised as many (if not more) questions than they answered.

More than 10 years on, it is little exaggeration to say that the landscape has been almost entirely transformed. Many of the pieces of the puzzle required to understand, in theory, adjustments to accounting aggregates and the implications (and caveats) of not saving enough to sustain (total or per capita) welfare are now known. In addition, a wide range of data on national-level resource depletion and environmental degradation now exist (e.g. World Bank, 2003). While a number of important debates continue, these should not obscure the conclusion that significant progress has been made (See Annex 2 for international experience with theoretical and practical developments in this field).

“Official” international experience (e.g. within governmental or international organisations) has been split between efforts to construct sustainability or environmental indicators and efforts to construct environmental and resource accounts. A crucial distinction between these two approaches is that the former (typically carried out within Environmental Ministries) tends to focus on headline indicators while the latter (typically carried out in statistical offices) tends to focus on highly detailed tables of environmental and natural resource data (by economic sector). Thus, on the one hand, indicators tend to appeal to a wider audience and so their primary purpose is in raising the public profile of a particular

\(^6\) Or Gross Domestic Product (GDP); the distinction is not crucial here.
environmental issue. On the other hand, environmental and resource accounts while appealing to a more narrow audience (potentially) give rise to a greater range of detailed policy applications.

The United Nations’ Statistical Office (UNSTAT) proposes its System of Environmental and Economic Accounts (SEEA) as an *adjunct* to the conventional System of National Accounts (SNA) (United Nations, 1993, 2003). Thus the SEEA provides the potential for international comparability in green national accounting (just as the SNA traditionally has). Nevertheless, the SEEA embraces a wide range of activities and methods. Broadly speaking, these include natural resource balances, emissions or wastes accounts, environmental expenditure accounts and green national accounting aggregates. The most prominent exponent of approaches based on green national accounting aggregates is the World Bank which publishes annual cross-country estimates of genuine (or adjusted net) saving and its components and has periodically published information on wealth accounting and changes in per capita wealth.

In practice, almost no country has officially sought to tackle the latter activity for a variety of (explicit and implicit) reasons (Fraser and Harris, 2002). Within the US there has been recent scrutiny of the robustness of green national accounting proposals. This external investigation was ordered by US Congress and carried out under the auspices of the National Academy of Sciences and, specifically, a “Blue Ribbon” Panel on Integrated Environmental and Economic Accounting. Interestingly, this Panel concluded that:

“...the development of environmental and natural-resource accounts is an essential investment for the nation. It would be even more valuable to develop a comprehensive set of environmental and non-market accounts. The panel emphasises, however, that environmental accounts must not come at the expense of maintaining and improving the current core national accounts, which are a precious national asset.”

(Nordhaus and Kokkelenberg, 1999, p9).

Given that the Panel’s findings were based on arguably the most significant reflection on the merits of green national accounting, this clean bill of health is a forceful vindication of these efforts. There is, however, a caveat to be added here. Some commentators have queried to what extent these new accounts have actually been used in policy analysis and formulation. It certainly seems to have been the case that most official green accounting programmes were initiated with very little discussion of end-uses. For example, a survey highlighting this point by Hamilton et al (1994) emphasised an important relationship between use of green accounts to inform policy and institutional links between data providers (e.g. statistical offices) and policy-makers (e.g. environment ministries). These authors claimed that these links were lacking in all of the countries that they studied (with the exception of Norway). It is difficult to say in general whether or not the use of environmental and resource accounts by governments has increased dramatically as there is no up-to-date study comparable to that of Hamilton et al (1994).

However, UN (2003) provides a section on applications and policy uses of the SEEA which offers some encouraging signs. Section 1.2 also provides some strong arguments for undertaking green accounting exercises, based on the value of policy uses of the information provided.

Past attempts at monetary Environmental Accounts for UK agriculture include work undertaken by Hartridge and Pearce (2001), Pretty et al (2000, 2002), the Environment Agency (2000) and Adger and Whitby (1991, 1993 and 1996). This report builds on these approaches and expands previous research in a number of respects, as described in section 1.3 on objectives and scope of the study. These efforts are reviewed in Annex 1, with reference to the approaches used in this study and how they differ. On the whole, however, this study focuses on positive as well as negative impacts; takes a systematic approach to identifying the accounting framework and the accounting entities that require measurement; undertakes a wider review of the economic valuation literature and presents the gaps in physical and economic data than previous attempts.

### 1.2 The Need for Environmental Accounts for Agriculture

The main driver for creating environmental accounts is the recognition that the current national accounting system does not reflect the full costs and benefits to society of economic activities, and, therefore, is an inadequate indicator of well being or true economic progress. Given the primary importance of traditional accounting indicators such as Gross Domestic Product (GDP) and Net Domestic Product (NDP) in public policy making, adjustments of these measures for environmental outcomes of economic activities are a step towards a better understanding of the sustainability (or otherwise) of economic development.

The content and extent of such adjustments depend on the policy questions posed. Historically, for example, national accounts were not set up to measure human wellbeing nor to determine sustainability.
Rather, they were designed to measure the size of the economy in market and public sector terms, and to chart changes in the economy over time. Gradually, national accounts have been increasingly interpreted as saying something about the quality of life of a nation's citizens. But if that is to be one of the uses of the accounts, then it is important to have a much wider notion of what constitutes 'goods and services'. In the current case, we know that environmental quality affects human health and welfare, so some accounting of those changes needs to take place alongside the changes recorded in the conventional accounts.

The practice of providing this wider form of accounting has generally been to present satellite environmental accounts, as with the current ONS environmental accounts (ONS, 2004). While providing a host of useful information, the lack of a common unit or relative weights that would allow aggregation of impacts expressed in physical terms is an important shortcoming. What the weights do is to assign a level of importance to the individual physical impacts, based on evidence of the economic impacts resulting from changes in the physical environment.

This discussion enables us to determine the first reason for moving towards a set of comprehensive monetised accounts that include environmental costs and benefits: if we are to have a true measure of the quality of life that is integrated with the existing national accounts, monetised environmental accounts are necessary.

It is possible to go beyond this modification and seek a set of comprehensive accounts that tell us something about the sustainability or otherwise of the current quality of life. In concise terms, the quality of life is sustainable at its current or higher level if, and only if, economic activity and government policy do not result in a reduction of the capital asset base that generates that quality of life. The notion is very familiar in common language: do not 'mine the capital base' or do not 'sell the family silver'. The accounts can be modified to cast light on sustainability by shifting the focus towards the change in the overall asset base. If it is rising (in per capita terms) the prospects are potentially good for sustainability. If it is falling, the prospects are bad. This means that changes in the capital asset base, or the change in ‘wealth’, must be measured. In turn, we know that this wealth comprises not only the stock of all produced (or man-made) assets, but also environmental assets, social assets and human assets (skills, knowledge).

The second reason for measuring environmental asset change, and the stock of assets, is to cast light on the extent to which current activity levels are sustainable. For the national accounts, this goal is met by moving towards measures of notions such as genuine savings (see Annex 2). For sectoral accounts, this interpretation is more complex. But, regardless of the meaning of sectoral sustainability, measures of overall national sustainability require that we produce comprehensive sectoral accounts.

A variation on this second reason for producing accounts lies in the distinction between weak and strong sustainability. In the former, all capital assets are assumed to be substitutable at the margin so long as overall wealth is non-declining over time. Strong sustainability, on the other hand, accepts weak sustainability but adds the caveat that the stock of natural capital should not decline. Again, it is easy to understand this double condition for a set of national accounts, but not so easy at the sectoral level. As with weak sustainability, where it might be quite legitimate to ‘run down’ capital in one sector whilst building it up in a more modern sector, so with strong sustainability: it may be that society wishes to contract an environmental asset of one kind only to build up another kind. But whatever the goal - weak or strong sustainability - the starting point has to be a set of monetised accounts. Annex 2 explains why even the strong sustainability approach requires monetary estimates.

The third reason follows directly from the previous discussion. If market principles govern the working of an economic sector, that sector would not survive for long if it fails to add value to the nation’s market wealth. For sectors where, for various reasons, public monies are a significant fraction of net income, it is possible that the sector survives but makes little real contribution to overall wealth. If there is a risk that this is so, society will want to be clear what it is getting back for the public monies it provides. If it gets back little, the continued subsidisation of the sector might not be justified. If it gets back a lot (e.g. in the form of biodiversity and landscape services in the case of agriculture), those subsidies may be very justified. Alternatively, if it is clear that the subsidies should be changed so as to maximise the social return to them that would be an important policy message.

The third reason for comprehensive accounts, then, is to determine the extent to which agriculture is a net contributor to the nation’s wellbeing.
Assigning monetary valuations to environmental impacts increases the usefulness of the physical data for policy making, providing answers to a range of questions, such as ‘how much harm does this impact do, or how much benefit does it generate?’ Of particular importance are the questions: ‘how important is a given impact compared to other impacts?’ and ‘how much is it worth to spend to reduce the negative impacts?’ For impacts to be compared they must be in the same units and monetised environmental accounts fulfil that requirement. One immediate use of such measures is then priority setting. Thus, the fourth reason for constructing accounts is priority setting.

If agriculture is shown to have its biggest negative environmental impacts in one area rather than another, then, other things being equal, that impact should be the subject of priority policy attention. A great deal more information is needed to say that a high environmental cost is a high priority, but the cost information is essential. To find out if the policy is worthwhile, of course, it is necessary to compare costs and benefits. It is not essential to have a set of accounts to do this cost-benefit exercise, but the accounts help to systematise this cost and benefit information and to ensure that like is being compared with like. The fifth reason is to facilitate cost benefit analysis while addressing the sector or national priorities.

Ideally, a set of monetised accounts would be able to serve all of the above purposes. However, as this report shows, this is not always possible due to incomplete scientific understanding of the environmental impacts of agriculture and the resultant economic effects, as well as lack of reliable data on both. This report therefore stops short of completing the accounting exercise and does not incorporate aggregate economic results for identified impacts into the main accounts or use these data to derive sustainability indicators.

However, this lack of data also points to another reason for constructing accounts: that they can eventually tell us what we do not know. It is vitally important to understand the nature of the current exercise. This report does not generate any new economic values for agricultural impacts (positive or negative). It simply assembles what we know and puts this into an accounting framework (See Section 1.3 for scope). This procedure is known as benefits transfer. There are two weakness of such an exercise, both of which can be turned into a strength. The first weakness is that some of the value estimates will themselves have low degrees of reliability. The reality is that, for quite a few areas of environmental impact, there is only one or a few studies. That is not an alterable state of affairs in the short run. What it tells us is that we need more studies in those areas to generate more estimates which can then be compared. In the extreme, there may be no studies at all that can be used. If so, the urgency of new work is all the greater. The second risk is inherent in benefits transfer anyway. If we ‘borrow’ numbers from the literature and put them in the accounting framework, how do we know they are true? In many cases the answer is we do not know and no amount of finessing of the figures will alter that situation in the short run. A lot of effort is currently going into the criteria for valid benefits transfer and into the estimation of the degrees of error involved. It will be a while before we can pronounce on the validity of some of the estimates.

The final reason for monetised accounts is, then, to identify impacts where we have no information and impacts where we have a limited basis for benefits transfer. We seek to identify the information gaps to facilitate updating the environmental accounts over the years (See Section 6).

1.3 Study Objectives and Scope

The main aim of this study, as set out in the terms of reference, is to identify data sources for the environmental impacts of agriculture and to develop methodologies that would enable the Government to produce adjustments to the aggregate agricultural accounts showing this impact. The more specific study objectives are to:

- Identify a fully comprehensive list of the positive and negative impacts of agriculture on the environment;
- Identify available data sources, appraise the quality and note where and how these data could be improved or updated;
- Identify gaps in the data and recommend potential sources or surveys that will be required to fill them;
- Develop methodologies for valuations of these impacts that will be sensitive enough to reflect changes in farm practice methods and noting where any double counting might occur with the aggregate agricultural accounts; and
Prepare an accounting framework for the UK showing a full breakdown for Scotland, Wales, Northern Ireland and England that best shows the revenue and capital aspects of the environmental impacts of agriculture and can also be used to develop a time series. If possible and if the methodology allows, a sectoral or regional breakdown could be considered.

In line with these objectives, the scope of the study can be described in terms of (i) the boundaries of the accounting framework; (ii) the list of environmental impacts; (iii) the geographical coverage; and (iv) sub-sector disaggregation.

The accounting framework reflects a weak sustainability approach, rather than a strong sustainability approach. This is discussed further in Section 2 and in Annex 2.

The report covers environmental impacts that are linked to primary agriculture and does not address those related to the agro-food chain (e.g. pesticide manufacturing, food processing, and human health implications of consumption of agricultural outputs) or the environmental impacts of other sectors on agriculture. The spatial limits of the analysis stay within the farm gate, and thus do not extend to the analysis of environmental impacts of transport between farms and other points along the production chain.

It is well known that the dominant factor in the damages from some environmental impacts (especially air pollution) is health damage. In part this may reflect the current bias in the ‘state of the art’ towards measuring health damages. But it also reflects the high priority that society gives to damage to human health and which shows up in the high unit values for premature deaths and some forms of morbidity. Since health impairment is a major cause of the loss of human wellbeing, the inclusion of health impacts in environmental accounts for agriculture is both logical and policy-relevant. Therefore, where agriculture impacts on human health through environmental pathways (e.g. through air or water pollution), these impacts are included in the analysis. However, the study does not include human health impacts that arise directly from agricultural production, such as deaths or injuries from operation of on-farm vehicles as these are regarded as occupational risks and not as environmental costs. Neither does the analysis include chemical residues in food and the impacts of BSE or foot and mouth disease.

Finally, dust and allergens generated by agriculture also impact human health negatively. Given the environmental pathway for these effects, these appear to be closer to health effects that could be regarded as ‘environmental’. Accordingly, an entry for such impacts is included in the accounts but not quantified since at the time of writing we are not aware of any assessment of the share of agriculture in this impact and what the economic cost of these impacts is.

The geographical scope of the study is the UK. However the availability of data limits the regional analysis. While the research has aimed to identify country-level primary data sources, in many instances it has only been possible to identify data at the level of the United Kingdom or Great Britain (or for one or two countries). While the lack of geographically disaggregated data is a gap for some impacts, for others UK level data are, in fact, more accurate (e.g. emissions to air).

The available physical impact data are also rarely disaggregated at the sub-sector (e.g. arable land versus livestock or even individual crop or animal type). Again this is a product of the current monitoring effort and provision of such disaggregated data does not have high priority in terms of the more strategic objectives of the environmental accounts for agriculture.

1.4 Overview of Study Methodology and Report Structure

The methodology employed in this study can be broken down into four discrete steps which correspond to the study objectives and are addressed in each of the sections that follow:

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7 The costs of dealing with BSE are already documented in the agricultural accounts. While BSE is implicated in human deaths from CJD, so that lost life values become relevant, recording human deaths against the output of the years in question would raise difficult issues. First, since CJD operates with significant time lags, it is unclear what lost life should be debited. Second, the debits would apply to a few years of the accounts only, distorting the longer-run picture. Third, it is unclear how far the problem can be regarded as an environmental one. In light of these considerations, we have excluded CJD deaths from the analysis, but note that it is an issue of concern that should be kept under review.
Step 1: **Accounting framework** (Section 2) - identifying the accounting adjustments that would be required to reflect the changes in income and wealth resulting from agriculture’s impacts on the environment;

Step 2: **Environmental impacts** (Section 3) - identifying the environmental impacts of agriculture and reviewing the physical data sources to select the appropriate data for environmental accounting;

Step 3: **Economic value of environmental impacts** (Section 4) - establishing the economic outcomes of environmental impacts and reviewing the economic valuation literature to select the appropriate monetary values for the accounting framework; and

Step 4: **Calculation of accounting adjustments** (Section 5) - combining physical and economic data for the most recent year to arrive at adjustments in income that could be applied to the existing accounts.

Finally, in Section 6, the gaps in the data are recognised and recommendations are made to fill these gaps.

**Step 1** examines the distinction between natural capital or assets of air, water and land, the services that flow from these and the extent to which positive and negative changes in these (see Table 1.1) can be attributed to agriculture based on green accounting theory. The services or functions of natural assets can be classified as follows:

- **a resource function**, whereby the environment provides the raw materials that are transformed by the economy to produce goods and services;
- **a sink function** whereby pollution generated by production and consumption is assimilated by the environmental media of air, water or land; and
- **an service function**, which provides both survival functions and amenity functions, such as recreation.

The resource functions (or raw materials) are generally provided as market goods that are paid for in the economy, and are, therefore, already included in the accounts, whereas the sink and service functions are not marketed, not priced and, therefore, absent from the current set of accounts. Thus environmental accounts for agriculture should aim to value the sink and service functions provided by the environment (United Nations et al, 2003). In its simplest form the value of these services or functions is the product of society’s willingness to pay for a unit of these services and their current level of provision.

In the context of this study, agricultural practices have impacts on all three functions of the environment. To the extent that agricultural activity uses resources or produces pollution that exceed the regenerative and assimilative capacity of the environment, respectively, there will be negative impacts on the environment and also on human health and welfare. On the other hand, agriculture also helps to maintain, or even at times improve, the quality of some services leading to positive impacts on the environment.

Whether positive or negative, these impacts of agriculture on the environment can affect:

- **other sectors** (referred to, in this report, as productivity impacts): Impacts on other economic sectors represent losses (or gains) in productivity of those sectors as a result of either damage to the environment arising from agricultural production (or environmental services maintained by agriculture). An example that is also quantified in this report is the increased drinking water treatment costs incurred by water companies due to agricultural sources of water pollution; and

- **society in general** (referred to, in this report, as welfare impacts): Impacts on human health and welfare result from agriculture’s impact on the environment’s sink and service functions. This impact can be through the assets managed by agriculture (e.g. through the amenity value of landscapes that are supported by agricultural activities) or through assets that are affected by agricultural process (e.g. air and water pollution). Such impacts could result from a quantitative effect (e.g. loss of hectares of habitat) or from a qualitative effect (e.g. the ability of a given size of habitat to support more species enabled through better agricultural practices).

Adjustments to the accounts therefore will show additional income that reflects, for example, agriculture’s maintenance of landscapes, habitats and species, as well as losses in income that reflect the degradation of natural assets lying outside of but influenced by agriculture such as air and water. These effects can be measured in both terms of changes in wealth, and some could also be interpreted as liabilities.
Section 2 provides an overview of the theory of green accounting and builds a framework for environmental accounts for agriculture that addresses the issues that are specific to constructing sectoral accounts. Sustainability indicators and accounting aggregates are also addressed, with particular focus on how to measure genuine net value added in a future set of environmental accounts for agriculture.

Step 2 in building environmental accounts for agriculture is to establish how the environment and economy interactions manifest themselves in physical terms and to select the appropriate data to describe these manifestations. The framework employed in this study for classifying these interactions is the DPSIR model (Driving forces, Pressures, State of the environment, Impact on final end points and policy Responses) shown in Figure 1.1 below, which is widely used by the European Environment Agency\(^8\) and the European Commission (see Pearce and Howarth, 1998) in characterising environmental impacts. Understanding and mapping environmental pathways is key to the interpretation of existing data sources and important for directing future research.

![Figure 1.1: DPSIR Model - example of agriculture](chart.png)

Driving forces, such as agricultural intensification, result in emissions of pollutants and other pressures which affect the state of the environment and, in turn, may impact on human health and wellbeing. Responses may address the driving forces as well as seek to reduce pressures, state of the environment or alleviate impacts on humans. In this example the response is addressing the pressure, by aiming to reduce nitrate emissions.

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\(^8\) “Particularly useful for policy-makers, DPSIR builds on the existing OECD model and offers a basis for analysing the inter-related factors that impact on the environment.” - from the EEA website: [http://org.eea.eu.int/documents/brochure/brochure_reason.html](http://org.eea.eu.int/documents/brochure/brochure_reason.html)
Ideally, it would be possible to map the DPSIR model for each agricultural practice or sub-sector (dairy, maize etc.), as this would link environmental accounts directly with main sector accounts. However, there are a great many unknowns and for the most part it is only possible to draw associations between drivers and emissions, drivers and changes in the state of the environment, or drivers and final impacts. Thus, in the absence of clear links between pressures, state and impact, a practical approach to understand the impacts of agriculture on the environment is to focus on the classification of the impacts themselves. Table 1.1 below presents such a classification, underlying which is an understanding of the links between agriculture and environmental impacts. The environmental outcomes within each main impact category show that agriculture has both positive and negative impacts on the environment. In other words, agricultural activities either maintain or enhance environmental functions (indicated by positive sign in the table), or serve to reduce them (indicated by negative sign in the table).

<table>
<thead>
<tr>
<th>Main categories</th>
<th>Environmental outcomes associated with agriculture</th>
</tr>
</thead>
<tbody>
<tr>
<td>I. Water</td>
<td>i. change in water quality (+/-)</td>
</tr>
<tr>
<td></td>
<td>ii. change in water availability (+/-)</td>
</tr>
<tr>
<td>II. Air</td>
<td>i. change in air quality - local and global impacts (+/-)</td>
</tr>
<tr>
<td></td>
<td>ii. dust and allergens (-)</td>
</tr>
<tr>
<td>III. Soil</td>
<td>i. change in soil composition and attributes (+/-)</td>
</tr>
<tr>
<td></td>
<td>ii. soil loss or gain (-/+)</td>
</tr>
<tr>
<td>IV Landscape</td>
<td>i. change in landscape (+/-)</td>
</tr>
<tr>
<td></td>
<td>ii. maintenance of landscape (+)</td>
</tr>
<tr>
<td>V. Habitats and species</td>
<td>i. change in biodiversity (+/-)</td>
</tr>
<tr>
<td></td>
<td>ii. maintenance of biodiversity (+)</td>
</tr>
<tr>
<td>VI. Waste</td>
<td>i. generation of waste (-)</td>
</tr>
<tr>
<td></td>
<td>ii. disposal of waste (-/+))</td>
</tr>
<tr>
<td>VII. Nuisance</td>
<td>i. odour (-)</td>
</tr>
<tr>
<td></td>
<td>ii. noise (-)</td>
</tr>
<tr>
<td>VIII. Resource use</td>
<td>i. depletion of non renewable resources (-)</td>
</tr>
<tr>
<td></td>
<td>ii. provision of alternative resources (e.g. renewable energy) (+)</td>
</tr>
</tbody>
</table>

Section 3 provides a detailed account of the environmental impacts of agriculture and selects data for the accounts using selection criteria applied to the complete set of data collected and presented in Annex 4. Section 2 explains which of the impacts presented in Table 1.1 should and should not be included in the accounting framework, and details the measures of economic impact that should be accounted for and how.

Step 3 in building the environmental accounts for agriculture is to understand the economic costs and benefits of the environmental impacts of agriculture. In practical terms, given that primary research is not possible within this study, this means selecting relevant and robust cost and benefit estimates from the available sources and matching these to the physical data from Step 2.

Economic costs and benefits of an environmental impact are estimated using data on individuals’ (or households’) preferences for that impact. Preferences, in turn, are measured by people’s Willingness to Pay (WTP) to maintain the current quality and quantity of environmental assets, to avoid a negative environmental impact or to secure a positive one. Similarly, preferences can be measured by people’s Willingness to Accept Compensation (WTA) to tolerate a negative environmental impact or to forgo a positive one. Most of the studies relevant here use the WTP measure.

Studies and theoretical work to date show that people have preferences for environmental assets because (i) they may make direct use of them (use values or the resource and service functions in the terminology above); (ii) they may indirectly benefit from the assets (indirect use values or sink and service functions); (iii) they may make use of them in future (option value) and (iv) they may value the assets irrespective of their current or future use but for the use others make now (altruistic value), for future generations.
(bequest value) or for the sake of the resource itself (existence value). The sum of all these values or all WTP and/or WTA for a given environmental resource is known as the Total Economic Value. What the economic data used in this framework actually show is the change in this Total Economic Value of the environmental assets affected by agriculture.

The first source of economic data on people’s preferences is market data for those environmental impacts that are traded or reflected in actual markets. These ‘marketed’ services of the environment include supply of drinking water, formal recreation or tourism and so on. In fact such market data (costs and prices) are akin to WTP or WTA of individuals in that market prices reflect the WTP of buyers and WTA of sellers. Generally an individual will only consume a good or a service when its price is less than or equal to his/her WTP. That is, a given good is purchased if they perceive that the benefit yielded by its consumption (as measured by their maximum WTP for that unit of consumption) exceeds the cost of consumption (as measured by the price of the unit of consumption). When the price is less than WTP, consumers receive a surplus of benefit. This surplus is known as ‘consumer surplus’, which is equal to WTP minus market price. Therefore, in most cases, the market price paid for a good is only a lower-bound estimate of an individual’s WTP for a good.

The second source of economic data on people’s preferences requires the use of economic valuation techniques. In the absence of market data (or where goods and services are provided at zero price), where WTP is comprised wholly of consumer surplus, economic valuation techniques have been developed to estimate people’s WTP or WTA. These techniques use either surrogate markets that reflect the preferences for some aspects of the environment (e.g. the impact of surrounding landscape on house prices) or hypothetical markets that elicit people’s preferences for changes in quality and quantity of environmental assets. The first group of techniques is known as revealed preference techniques and undertake econometric analysis of market data, while the second is known as stated preference techniques and implement carefully structured questionnaires to affected populations.

Market data are used in this framework to reflect the costs imposed by agriculture on other sectors including water companies as mentioned above and the public sector (e.g. Environment Agency’s spending for replenishing the affected fish populations following pollution incidents). Non-market data, on the other hand, are used for the impacts of agriculture on society. A more detailed look at the types of data used for this purpose is given in Table 2.2 and further discussion on how the literature is reviewed and relevant studies are selected for this study can be found in Section 4. Section 4 also provides a fuller account of economic valuation theory and methodology, including a discussion of the issues and caveats of employing benefits transfer as a technique.

Step 4 is the culmination of Steps 1-3, whereby the economic data and physical data are aggregated to arrive at measures of environmental wealth and income that are necessary to creating an environmentally adjusted account. In this study only calculations for income measures are calculated in the final step. Section 5 provides details of each of the calculations for each of the environmental accounting adjustments identified in Step 1, and provides an overview of the results of this exercise with some policy interpretations.

Finally, in Section 6, an assessment of the gaps in physical and economic data on the environment is presented, and recommendations made for how these might be addressed.
2 Accounting Framework

The main purpose of this Section is to address the key theoretical issues in the application of environmental accounting models to UK agriculture and to translate these models of accounting into an operational framework. Section 2.1 briefly outlines the basic framework for national environmental accounts and the relationship of this framework to the concept of sustainable development. Section 2.2 discusses the application of this framework to the specific case of agriculture and provides an introduction to various environmental accounting aggregates. Section 2.3 reviews environmental (and other) impacts in the context of this framework (as listed in Table 1.1) and summarises the accounting adjustments that would be required for an environmental accounting exercise. Section 2.4 addresses a range of questions that define the boundaries of the accounting exercise and provides methodological details. In addition, Annex 2 contains some theoretical discussions underpinning the accounting framework presented here.

2.1 Basic Accounting Framework for the National Economy and its Implications for Agriculture

Before proceeding to the application of environmental accounting in the agricultural sector, it is useful to outline the theory of sustainability and green or environmental national accounting in a more general context, i.e. for the economy in the aggregate. This discussion follows on from the progress to date briefly reported in Section 1.1.

There are a wide range of proposals in the literature which, broadly speaking, have sought to provide greener measures of income and national welfare as well as more comprehensive measures of total wealth and its accumulation (see, Atkinson and Dietz, 2003, for a recent review). Much of the theoretical literature builds on important contributions by Weitzman (1976), Hartwick (1990) and Mäler (1991). The focus in most contributions is typically on accounting for the value of changes in total wealth in national income. National income is typically defined along the (optimal) path of a simple economy with stocks of goods (including natural resources used in production, i.e. the resource function of the environment) and bads (including the degradation of environmental stocks such as clean air, that negatively affect wellbeing or sink and service functions). A general expression for a (net) national income aggregate or green Net National Product ($g\text{NNP}$) is:

$$g\text{NNP} = C + \sum p_i \dot{X}_i = C + S_G$$

where $g\text{NNP}$ is equivalent to the money value of consumption ($C$) plus the sum of net changes in $i$ assets ($\dot{X}_i$, where the dot refers to a change) each valued at its shadow price ($p_i$). Assets here refer to all forms of capital including human made or produced, natural or environmental, human and social. Alternatively, this can be written as consumption plus adjusted net or genuine saving ($S_G$): that is:

$$\sum p_i \dot{X}_i = S_G$$

where the changes in assets might refer to net investments in produced, human and natural capital.

With regards to signalling prospects about (weak) sustainable development, an interpretation of $g\text{NNP}$ is that it measures extended Hicksian income: that is, the maximum amount of produced output that could be consumed at a point in time while leaving wealth constant (Pemberton and Ulph, 2001). Given an interpretation of (weak) sustainability is that the change in the (real) value of total wealth should not be negative in the aggregate, this definition of income suggests that our focus should be on the genuine saving or $S_G$ component of the expression for $g\text{NNP}$ above. The reason for this is that $S_G$ tells us about the (net) change in total wealth, i.e. it can be shown that:

$$\dot{W} = 0 \text{ if } S_G = 0.$$
That is, the change in total wealth ($\dot{W}$) is zero if genuine saving is zero. More specifically, the key finding in this literature is that a point measure of negative genuine saving ($S_G < 0$) means that a development path is unsustainable (Hamilton and Clemens, 1999). Negative genuine saving implies that the level of wellbeing over some interval of time in the future must be less than current wellbeing - development is not sustained, to use Pezzey's (1997) terminology. Interestingly, the proposition that negative genuine saving is unsustainable holds for (characterisations of) non-optimal development paths (Dasgupta and Mäler, 2000) and other extensions such as time-varying discount rates and exogenous technological change (Hamilton, 2002a, and Weitzman and Löfgren, 1997). In addition, Hamilton (2002a) shows that if $S_G$ is persistently greater than zero - arguably a sensible policy target in a risky world - then not only is wealth increasing but development can, in certain circumstances, also be said to be sustained.  

The key conclusion, then, is that (persistent) negative genuine savings is a sure sign of unsustainability, i.e. of declining aggregate wealth. This is consistent with more popular notions of ‘not eating into one’s capital’ or ‘not selling the family silver’ as also mentioned in Section 1.2. Persistent positive genuine savings is fairly indicative of sustainability, although the conclusion in this respect is less certain than for negative genuine savings. 

In general, little attention has been given in the literature to extending green accounting principles to either geographical regions or economic sectors. Exceptions in the case of the former include Vincent (1997) and in the case of the latter include Atkinson (2000). It might be argued that there is no inherent reason why any sector within a national economy should be sustainable. For example, in the context of the forestry sector Vincent and Hartwick (1997) caution against a blinkered emphasis on sustainability in any one sector and argue that priority should be the sustainability of the whole economy. Indeed, for some sectors (such as mining) this may not even be feasible even if it were desirable.

In fact, a casual glance at the history of the British economy indicates changing fortunes of various sectors in the context of overall long-term growth in the (real) economy. Put another way, one would expect the process of economic development to be characterised by net wealth destruction in ‘old’ sectors which give way to net wealth creation in ‘new’ sectors. This process of destruction and creation will be driven by major forces such as technological change and international comparative advantage. At the very least, issues arising from concerns about (long-term) sustainability need to be disentangled from (long-term) processes of structural change that occur within an economy.

Is agriculture a special sector that should be sustained or sustainable? It seems highly desirable that the global agricultural system is sustainable (in the sense of fulfilling the nutritional needs of the world’s population now and in the future). In fact, the literature on the meaning of sustainable agriculture or the properties of sustainable agricultural systems has sought to examine such issues (see, for example, Cobb et al, 1999). The question here relates to the sustainability of the sector within a national economy. Certain individual countries may well place a premium on food security and this might motivate such concerns.

However, in the UK, the share of the agricultural sector in total output has fallen considerably over time (as it has in most developed economies). Therefore, it would be desirable to seek more general arguments and, in particular, those arguments which motivate the need for green accounting in the agricultural sector. In addition to those given in Section 1.2, there are at least three reasons why a green accounting approach is useful for understanding the notion of sustainability in agricultural sector:

- it would be worthwhile knowing whether current levels of agricultural production are being financed by farming practices that, other things equal, diminish future income generating potential within the sector (e.g. by leading to erosion of soil productivity);
- perhaps more important are those wider concerns about the (net) impact of agriculture and farming practices on society’s welfare or wellbeing and the impacts on other sectors. These external impacts too fall within the ambit of a notion of sustainability in the agricultural sector not least in determining the (net) contribution of the sector towards the sustainability of the wider economy, and
- measuring changes in net wealth in a sector helps to identify which industries are taking on the mantle of ‘old’ and which the mantle of ‘new’.

\[\text{10}^{\text{10}}\] Specifically, this requires that the growth rate in genuine saving does not exceed the interest rate.

\[\text{11}^{\text{11}}\] In other words, $S_G$ is strictly speaking a one-sided indicator of sustainability (Asheim, 1994; Pezzey and Withagen, 1998).
2.2 Basic Framework for Environmental Accounts for Agriculture

International attempts at creating environmental accounting frameworks are a useful starting point to inform this study. In the current context, the SEEA (UN, 2003) discussion of land makes explicit reference to the multiple goods and services provided by this asset in its different uses. However, there is less explicit guidance within this framework with regards to how to arrive at a monetary counterpart to physical land accounts apart from the discussions of the market-oriented values such as future losses in agricultural productivity because of (on-site) soil erosion. Thus, while the SEEA 2003 devotes an entire chapter to non-market valuation; the context for that discussion is largely the consideration of economic valuation methods and their application to the impacts of air and water pollution rather than to valuing environmental (or non-market) services provided by, say, woodland. Because of this, we do not pursue the framework outlined in UN (2003) further in this report. However, we note that there will be similarities, both in spirit and in detail, between that approach and the approach taken here. Further details on the international attempts to environmental accounts can be found in Annex 2.

As mentioned in Section 1.1, there is a small literature which has sought, within a national accounting framework, to place money values on either the environmental impacts of the agricultural sector or (more generally) the contribution of the land-use sector to human well-being. The focus in most of these studies has been on estimating an environmentally adjusted value-added or welfare measure. In Adger and Whitby (1993) and Hartridge and Pearce (2001), this (sector-level) accounting aggregate for the UK provides adjustments for: (i) the value of the change in non-market assets (such as land-use change, land degradation or changes in environmental stocks as a result of farming activity); and (ii) flows of non-market services which arise from land in its existing use. Hrubovcak et al (2000) estimate adjustments to net valued-added of the US agricultural sector in the context of (i) only. A slightly different means of presenting adjusted accounting data is that of Whitby and Adger (1996) who estimate the value of asset changes in the UK agricultural sector in order to discuss the contribution of agriculture to net wealth creation.

Following the above, a natural starting point for an adjusted or environmental account, in money values, for UK agriculture is the existing economic accounts for the sector: ‘Agriculture in the United Kingdom: 2003’ (DEFRA et al, 2004). Chapter 8 of that report consists of: (i) a summary production account, i.e. the (gross and net) output of the sector; (ii) a capital account describing net accumulation of produced capital (structures and machinery) and livestock; and (iii) a balance sheet describing opening and closing stocks for the agricultural sector. In principle, environmental accounts for agriculture could entail augmenting any or each of the above accounts. With regards to the alternative accounting aggregates that such adjustments imply, the following are worth noting:

- **An environmentally-adjusted measure of the contribution of agricultural activity to net value added:**

  This entails an adjustment to the production account. Net value-added is important as an indicator of a sector’s contribution to the overall economy after the (real) value of net changes (positive and negative) in net wealth or asset accumulation in that sector has been taken into account. This net change could refer to changes in produced, human and natural assets that occur as a result of economic activity within the sector. This indicates the extent to which a sector adds to the value of the nation’s wealth.

- **A comprehensive measure of the contribution of agricultural activity to net wealth accumulation:**

  This entails an adjustment and extension to the capital account. Sectors do not “save” in the sense of the simple calculation of income minus consumption that is used to estimate (gross) saving for the national economy. However, investment expenditures, i.e. gross capital formation, are ascribed to each sector. While (adjusted) net-value added already embodies information about a sector’s net contribution to wealth, defining a more comprehensive measure of (net) capital formation would be a more direct means of signaling the contribution of a sector to the value of the nation’s wealth.

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12 Nevertheless the theoretical model that the authors outline suggests, in principle, an adjustment for the current flow of services enjoyed from undegraded surface water.
o **A comprehensive measure of the contribution of agricultural activity to net wealth:**

This entails an adjustment and extension to the balance sheet. The link between the balance sheet and the foregoing discussion about a more comprehensive measure of net wealth accumulation is that the difference between opening and closing stocks in the balance sheet is determined by net asset accumulation during the (intervening) accounting period (excluding capital gains and losses). In other words, if this net accumulation is greater (less) than zero then the real value of total assets is increasing (decreasing).

o **An environmentally-adjusted measure of the contribution of agricultural activity to (economic) welfare:**

This again entails an adjustment to the production account. As mentioned in Section 1.4, for example, land managed within the agricultural sector plausibly provides a number of non-market outputs (e.g. the flow of amenity enjoyed as a result of the provision of the existing ‘stock’ of woodland areas) in addition to more tangible produced outputs such as crops and so on. In accounting terms, the sum of non-market services can be thought of as an extended notion of consumption. By definition, taking account of the positive values of services consumed from certain natural assets provides a boost to the measures of the contribution of agriculture to the nation’s wellbeing or economic welfare.

In practical terms, the issue of how to construct a more comprehensive balance sheet, production or capital account (in monetary terms) raises complex questions which are worthy of lengthy consideration in their own right. For example, which additional assets fall within the ambit of a balance sheet for agriculture in addition to the conventional focus on produced capital and livestock? It seems straightforward to propose that the non-market asset value of land managed within the sector should be included within an extended balance sheet.

 Clearly, however, agriculture affects a range of other environmental and natural resources which comprise natural assets but do not recognisably fall within the province of an agricultural balance sheet. One illustration might be climate stability which is affected by (net) emissions of greenhouses gases from the agricultural sector, i.e. emissions net of carbon sequestration. A solution to this problem might be to include, in this example, the sector’s climate liability as an item in the balance sheet. That is, the cumulative “environmental debt” represented by the money value of damage caused by net emissions of greenhouse gases within the sector. While the agricultural balance sheet is not adjusted in this study, the (change in the) climate liability of agriculture, i.e. the economic value of damage caused by greenhouse gas emissions from the sector, is estimated in Section 5.

Finally, agriculture, through its impacts on the environment, may also impose costs on other sectors that result in losses in productivity to those sectors. The accounts may wish to reflect the fact that farming raises or lowers output in another productive sector, even though the overall effect on GNP remains the same.

The next sub-section presents the adjustments that would be required to calculate the accounting aggregates presented above (or make additions to the balance sheet or production and capital accounts), under the impact category headings set out in Table 1.1. A number of important issues are examined using examples from each impact type, including those raised above. However, these adjustments are not taken forward to estimating the above accounting aggregates for two reasons:

- While the valuation of the environmental impacts of agriculture may be a useful starting point, the uncertainties inherent in at least some of the data are likely to suggest that it may be premature to use them to adjust national accounting aggregates.

- It arguably makes little sense to provide a ‘green’ account for agriculture that includes environmental impacts, when the output measure itself rests on distorted, i.e. subsidised, agricultural product prices. Ideally, a ‘green’ account for agriculture should use shadow pricing, i.e. prices net of subsidies and taxes, of produced outputs. However, while this clearly would be important for enhancing the policy usefulness of the accounts, it raises many complex issues and is beyond the scope of the current study (this issue is discussed further below).
2.3 Accounting for the Environmental Impacts of Agriculture

The discussion in this sub-section is couched in terms of the environmental impacts of agriculture. Broadly speaking, each of these impact categories either corresponds to a change in a stock of an asset (e.g. land type) or a flow of current non-market services from that asset (e.g. sink and amenity functions of land). Table 2.1 summarises each of the impact categories referred to previously in terms of whether it is a change in a stock or a flow of a non-market service. Moreover, the table distinguishes between (i) whether the underlying asset is land-based and therefore attributable to agriculture, or whether it is some other environmental asset (such as a watershed or airshed etc.), and hence lies outside the province of agriculture; (ii) whether the change in stock is in terms of a change in quantity (such as number of hectares) or a change in quality (e.g. air quality) and (iii) whether the stock itself or the flow of non-market services from it are affected (e.g. changes in landscape amenity).

<table>
<thead>
<tr>
<th>Table 2.1: Environmental Impacts and Accounting Terminology</th>
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<tbody>
<tr>
<td>Asset Category</td>
</tr>
<tr>
<td>Land asset</td>
</tr>
<tr>
<td>Other environmental assets</td>
</tr>
<tr>
<td>Change in stock (change in quantity of underlying asset)</td>
</tr>
<tr>
<td>○ Soil erosion/ enhancement</td>
</tr>
<tr>
<td>○ Loss/ gain of habitats/species</td>
</tr>
<tr>
<td>Change in stock (change in quality of asset)</td>
</tr>
<tr>
<td>○ Degradation (loss)/ improvement (gain) of landscape amenities</td>
</tr>
<tr>
<td>○ Air pollution (including dust and allergens)</td>
</tr>
<tr>
<td>○ Water pollution</td>
</tr>
<tr>
<td>Flow of a non-market environmental services</td>
</tr>
<tr>
<td>○ Consumption/ enjoyment of landscape amenity services</td>
</tr>
<tr>
<td>○ Consumption/ enjoyment of land-based biodiversity protection services</td>
</tr>
<tr>
<td>○ Nuisance (noise/odour)</td>
</tr>
<tr>
<td>○ Waste generation</td>
</tr>
</tbody>
</table>

The flows of services are distinct, in accounting terms, from changes in the underlying natural asset that (wholly or partly) generates these services. In the case of a change in the value of an underlying natural asset, it is the ability to generate future (non-produced) services which is the issue. Whether it is the quantity of the stock that changes, e.g. the quantity of heather moorland, or the quality of the stock that is affected, e.g. underlying water quality, both are measured in the income accounts in terms of the annual flow of non-market services that the stock can produce in its current state (i.e. at its current stock amount or quality).

To elaborate, in one accounting period, a reduction in quality or quantity of stock essentially translates into a loss in the functions of the asset, and shows up in the accounts as a reduced flow of environmental services from that stock. Landscape or habitat-specific examples include the degradation or restoration of land and changes in land-use. Similarly, in the case of water, it is the ability of the water environment to produce amenity services that is eroded through water pollution and abstraction that are considered as changes (or depreciations) in water environmental assets. Where the impact is negative, the accounting adjustment equates to an amount of the negative service flow that arises from reduced asset quality relative to a higher baseline. This baseline is discussed again later in this section, and also demonstrated in Section 4 for the calculation of negative water service flows.
Valuing non-market services is, in principle, just the same as putting a value on market goods and services: that is, it consists of a price and a quantity. However, in the case of environmental services the latter might refer to (an index of) the quality as well as quantity of the non-market output. Hamilton (1996) shows that the value of environmental services is the product of marginal willingness to pay for a unit of these services and the current level of these services. In turn, the latter term, i.e. the current provision of services (or non-market services) is itself related to the current level of the environmental asset, or stock of environmental capital, which provides those services (Hrubovcak et al, 2000).

What are the implications of these conceptual findings for measuring the non-market services of habitats and landscapes managed within the agricultural sector? Atkinson and Hamilton (2003) examine the case of environmental services provided by a standing forest where: (i) the value of current environmental services provided by a hectare of forestland is equivalent to marginal willingness to pay to conserve that hectare; and (ii) if this unit of land is (permanently) cleared, then the resulting change in the stock of forestland is given by marginal willingness to pay divided by the (social) discount rate, i.e. the present value of the lost environmental services on this land over perpetuity.

There are complications in measuring losses in value as a result of land use changes (See also Section 2.4.2 on the baseline). Firstly, the alternative land may itself provide environmental services which need to be taken in account. Secondly, land-use may change (as in the case of clearing forest to make way for agriculture) but the land asset in its current use is degraded in some way, i.e. the quality of services is degraded. That is, in such cases, it is the quality of the land that changes and conceivably this land provides fewer environmental services. This seems to be the approach of Adger and Whitby (1993) who estimate degradation of e.g. SSSIs caused by farming practices.

For the ‘straightforward’ case where a land-based asset is neither reduced in terms of its area or quality over an accounting period, then this non-produced output has a value equal to willingness to pay per hectare multiplied by the hectares of land in that particular use for that particular period, i.e. one year in this case. See Section 5 for how this case is treated in this report.

Table 2.2 deliberately present the changes in stocks alone, and does not introduce an accounting term that refers to the total value of stocks. This is simply a reflection that, unlike man-made stocks, environmental stocks cannot disappear from the balance sheet (one can’t imagine the world without air or water). The phenomena witnessed is one of change from one state of the environment to the next (e.g. from one habitat or land type to another) where each provides a different flow of environmental services, not one of complete loss. An exception might be the loss of a species, however, this could also be considered a change in the stock of all species.

In classifying environmental adjustments to the accounts, it is also important to differentiate between who benefits or bears cost. Within an accounting framework for a single sector, it is necessary to identify how the environmental impacts resulting from that sector’s activities affect both (i) the performance of other sectors and (ii) the welfare of society. Where environmental impacts affect the performance of the agricultural sector itself, these are assumed to be evident already in the sectoral accounts (e.g. through reduced productivity). Table 2.2 below outlines this particular point with some examples for agriculture.

This distinction between the impacts of agriculture on other sectors and society is continued below in the discussion under each impact heading.
Table 2.2: Environmental Impacts of Agriculture and Relevant Economic Data for Accounting

<table>
<thead>
<tr>
<th>Productivity impacts on other sectors a</th>
<th>Welfare impacts to society</th>
</tr>
</thead>
<tbody>
<tr>
<td>(market data)</td>
<td>(mostly non-market data)</td>
</tr>
<tr>
<td><strong>Productivity gains</strong> provided by agriculture that are enjoyed by other sectors</td>
<td><strong>Welfare gains</strong> provided by agriculture which are enjoyed by the general public</td>
</tr>
<tr>
<td>- e.g. landscape and recreational benefits that impact positively on the tourism industry</td>
<td>- e.g. wildlife and landscape maintenance the benefits of which are enjoyed free of charge by ramblers and local residents.</td>
</tr>
<tr>
<td><strong>Productivity losses</strong> created by agriculture and borne by other sectors</td>
<td><strong>Welfare losses</strong> created by agriculture which are borne by the general public</td>
</tr>
<tr>
<td>- e.g. costs to water companies of removing nitrate and pesticide pollution from drinking water</td>
<td>- e.g. loss of ecological services (such as river quality) and recreational opportunities through agricultural pollution.</td>
</tr>
</tbody>
</table>

a: As mentioned above, these are ‘transfers’ in that accounting for these impacts does not alter the national accounts (e.g. GNP) but rather the measured allocation of value-added between economic sectors.

2.3.1 Water

**Water quality**

Hrubovcak et al (2000) examine the case of ground water pollution due to agriculture, within an extended or green national accounting framework. Within this framework, the impact of the decrease in the quality of water resources has two distinct impacts. Firstly, water pollution affects productivity in other sectors of the economy. For example, water pollution might conceivably reduce the output of sectors that use clean water as an input to production (e.g. by increasing the costs of treatment). The conventional national accounts will reflect these impacts: that is, GNP will be lower because of the negative effect that economic activity in the agricultural sector has on other productive sectors. In other words, while it might be interesting to know the magnitude of this lost output, it is already reflected in gNNP for the aggregate economy.

However, from the perspective of measuring ‘true’ income in the agricultural sector and in a productive sector whose output is adversely affected by agricultural water pollution it is arguable that: (i) the ‘true’ income in agriculture should be lower reflecting the fact that farming lowers output in another productive sector; and, (ii) the ‘true’ income of the affected sector should be correspondingly higher. This re-attribution of pollution costs between polluter and victim leaves national income (e.g. gNNP) unaffected.

Secondly, water pollution from agriculture also affects the wellbeing of society. That is, households dislike water pollution for reasons over and above the fact that it lowers produced output in the economy. Hrubovcak et al (2000) show that the value of these water pollution impacts is estimated by multiplying the (net) reduction in quality by the value of change in water quality (e.g. willingness to pay for clean up).

Thus two accounting measures, productivity effects on other sectors and welfare effects on society should be employed to measure the economic value of depreciation of water assets. As this report goes on to show, the available data allow monetary estimates to be calculated for productivity losses to the water sector (for drinking water treatment) and welfare losses to society, for some impacts on water quality, but not all (e.g. pollution of inland waters and faecal contamination of coastal waters).

**Water availability**

Hrubovcak et al (2000) examine the case in which agriculture abstracts ground water. Formally, this is akin to the extraction of a natural resource. The stock of water recharges at some rate (through
precipitation and imports from elsewhere) and is discharged (through evapotranspiration, abstraction and consumption etc.) If discharges exceed recharge then the stock of available water resources declines in quantity. The relevant question is the proportion of this physical depreciation that is attributable to consumption and specifically abstraction by agriculture. Current data do not allow estimation of this effect, however, as is argued for the extraction of non-renewable resources (Section 2.3.8), the effect of reducing the availability of this resource would be reflected in market prices and hence accounted for in the existing national and sectoral accounts.

In addition to changing the stock of water available in the ground, abstraction of water will have welfare impacts on society in a similar way to reductions in water quality. Reducing water in the environment generally has negative impacts on water ecology and amenity value. In accounting terms this is referred to as depreciation of the water asset and is included in Section 5 as such. The affect of agriculture on the risk of flooding in certain areas should also be included in accounts and Section 5 illustrates how, given the available data, this could be done.

2.3.2 Air

An interesting aspect of accounting for depreciation of the stock of clean air is that emissions of air-borne pollutants will have local, regional and global impacts. While local impacts have clear bearing on the GNP of the UK, in the case of regional and global emissions from the UK the majority of this cost will be borne elsewhere.

Hamilton (1996) outlines the theory of green national accounting in the presence of air pollution, while Hamilton and Atkinson (1996) discuss the rationale for deducting transboundary impacts (on wellbeing outside of a study country) from an estimate of a study country’s gNNP. For cumulative pollutants (such as greenhouse gases etc.), Hamilton (1996) shows that the value of pollution damage is akin to a stock change which should be deducted from greener measures of income and saving. Accounting for UK depreciation of the regional and global air quality is in line with the polluter pays principle and is supported by UK’s ratification of regional and global treaties and protocols pledging to address regional and global air pollution. Therefore, in this study, the nature of the depreciation, i.e. local, regional or global, has been made clear to allow more obvious policy interpretation of the results and the damages from air pollution emitted by agriculture are estimated as depreciation of the stock of clean air.

**Dust and allergens** generated by and dispersed in the air through agriculture contribute to impacts on human health such as hay fever. While acknowledged as an impact of agriculture and as a part of the framework for environmental accounts, this impact is not further analysed in this report essentially due to a lack of data in both physical and economic terms.

2.3.3 Soil

Soil impacts can be looked at in terms of soil erosion (both on and off site or farm) and soil productivity. The current yields reductions and/or expenditures to compensate for the loss of (on-site) soil quality are assumed to be already in the existing sector accounts. However, a ‘truer’ measure of income in the sector should reflect the impact of soil erosion on future income-earning prospects. This is akin to deducting the value of depletion of a non-renewable resource (such as oil) from gNNP. Conversely, expenditures on improving soil fertility (such as spending on fertilisers) represent an investment in soil productivity and thereby count positively towards the sector’s income (Hartwick, 1992, 1993 and Hrubovcak et al, 2000). What this (net) soil erosion expression is intended to capture is the change in the contribution of farmland to future agricultural output (Hrubovcak et al, 2000). It is important to note that these future impacts (whether positive or negative) are not included in the conventional income and production accounts for the agricultural sector. Given the lack of adequate scientific data to allow the net change in soil productivity to be calculated, this measure is also not included in this report.

The off-site impacts of soil erosion are akin to the effects of water pollution caused by agriculture on other sectors. The impacts include the costs imposed on local government of dredging stream channels and clearing roads of soil washed off from farmland during storm events. In addition there are also costs to society, which include the welfare loss through reductions in ecological or service functions associated with excess soil in water courses.

There are nuances in the above that are worth noting. Investments in soil productivity (e.g. fertilisers) might have adverse impacts on the environment as well as positive impacts on future agricultural output. In principle, this observation creates no problems. Assuming that these negative impacts can be
quantified, it simply means that the shadow prices of certain investments in soil productivity are lower than relevant market prices would indicate (e.g. market prices net of the economic value of damage caused by fertilisers). In practice, it is more complicated to measure the environmental impact associated with particular soil productivity investments in this way.

2.3.4 Landscape

From an accounting perspective, there are two terms which are of interest with regards to landscape: landscape amenity services and landscape assets.

Firstly, landscape amenity is a non-market service ‘consumed’ by households. But little if any of the value of this consumption is reflected in the conventional accounts. Hence, it is important to take account of the annual flow of environmental services or amenities provided by land in its current use. This provides a potentially significant positive impact that is attributable to current agricultural activities.

Secondly, if landscape is altered (perhaps through land-use change) then there is a change in the asset value of this stock of land (Hartwick, 1992, 1993). This is because some landscapes are more highly prized than others in the sense of the level of, say, visual amenity that they provide to households. Altering a given landscape could conceivably result in either less or more amenity being enjoyed over time. In other words, the asset value of the land will have either declined or increased. In either case, this change should be reflected in a greener measure of income for agriculture.

A change in asset value (or change in the stock of land) would appear in the agricultural accounts as the present value of the loss of gain in future flows of environmental services from the previous land use to the current land use. Note that this change in asset value of stock will also be evident by the increased or decreased annual flow of environmental services from one accounting period to the net. For the purposes of the accounting exercise carried out in the remainder of this report, present value calculations are not conducted. This type of calculation would only be applied to changes in stocks of land and would appear as a change in wealth. For this, data on land uses in the previous period are required, whereas the data presented in this report relates only to the current accounting period.

In the latest version of the System of Environmental and Economic Accounting (SEEA) (United Nations et al, 2003) a detailed land-use and land quality accounting framework is outlined. Using data on broad habitat classifications from the Countryside Survey 2000 as reported in Haines-Young et al (2000), this accounting framework includes land cover and land use measured in hectares where ‘land’ covers a diverse range of categories relevant in the current context including agricultural land, and other land (which itself includes, for example, types of grassland). These accounts describe a ‘snap-shot’ of, say, current land-use as well as changes in the composition of land-use between periods. It is clear that this framework constitutes a very useful means of organising heterogeneous data about land in a manner that is potentially amenable to policy analysis, and hence the approach has been replicated in a similar way in calculating accounting adjustments for land.

The nature of land is such that the landscape, habitats and species provided on a particular hectare are inextricably linked, which makes accounting for them separately a potentially impossible, and even unnecessary, task. In addition, willingness to pay estimates from the literature for the non-market benefits provided by particular types of land often encapsulate all of these elements of value in one £/hectare estimate. The approach used in this report is unavoidably data driven and thus seeks to estimate service flows from the types of land reported in Haines and Young (2000), applying data that encapsulates elements of habitats and species in addition to landscape benefits.

2.3.5 Habitats and species

Habitats and species are affected (positively or negatively) by agriculture through inputs to agriculture such as pesticides and changes in agricultural management practices. For example, from an accounting perspective, pesticides have an adverse affect on biological assets and the ‘true’ income of the agricultural sector should reflect this cost. If the application of these inputs were to decline from one period to another then the magnitude of this biodiversity cost should be correspondingly lower (even though this may happen after a time lag).

In addition, changes in the use of land managed by agriculture may entail a loss of (or gain in) biodiversity services. In the framework by Hartwick (1992 and 1993), land is the asset and (one constituent of) its value is defined in terms of the biodiversity protection services it provides. Thus, when a unit of land is
switched from, say, intensive farming to a more environmentally friendly management practice (e.g. an agri-environment scheme) then one expectation is, typically, that biodiversity protection services are enhanced. Put another way, other things being equal, there is a (positive) change in the asset value of land.

Agricultural land in the UK, covering as it does over 70% of the total surface area is home to the majority of habitats and species. These have developed over centuries of agricultural change and many are dependent on the continuation of certain agricultural practices for survival. Thus, the starting point for accounting for habitats and species, should be the current healthy stock of these under agricultural management. The accounts should reflect both the non-market services (e.g. birdwatching) arising from the stock of habitats and species and the changes in that stock (e.g. sum of lost values from the lost hectares).

Agriculture may have other positive impacts on biodiversity not mentioned thus far in this section. For example, to the extent that the protection of rare breeds (of cattle and so on) confers benefits (in terms of retaining diversity) not adequately reflected in market values then green accounts should be adjusted for these. These adjustments might include: (i) the (positive) current services provided by protection of rare farm animal breeds; and, (ii) the full value of any (net) investment in stocks of rare breeds.

### 2.3.6 Waste

From an accounting perspective, issues related to the impacts arising from the generation (and disposal of) wastes by agriculture are analytically similar to the cases for water and air pollution. Indeed, impact categories related to waste disposal ultimately relate to emissions or release to water or air, if waste is stored or disposed of on-farm. Therefore, one practical issue is that care must be taken not to double-count in the case, for example, of slurry contamination of a watercourse which might also be counted as part of the water pollution category. Therefore, only the ‘general waste’ category of waste generated by agriculture is included in the adjustments in Section 5.

General waste that is sent to a landfill is subject to a landfill tax. If farmers pay this tax directly or through their council tax and if it is assumed that landfill tax reflects the non-market costs of landfilling waste, we can say that the related environmental impacts are internalised and already included in farmers’ bottom line and hence the sector accounts. However, in order to reflect the actual impacts of landfilling, regardless of what portion of them is internalised by the landfill tax, the monetary expression of these impacts are included in the accounts (see also the discussion in Section 2.4.1).

For waste that is disposed of through incineration, environmental impacts also need to be determined or estimates of economic costs in terms of £/tonne of waste incinerated can be used. Therefore, the adjustments in Section 5 use a weighted average external cost for waste that is landfilled and incinerated.

### 2.3.7 Nuisance

Odour and noise affect the quality of the amenity services provided by the environment (e.g. fresh air or peace and quiet) and would appear as debits in an environmental account for agriculture. However, it has not been possible to quantify these in physical or monetary terms due to lack of data (beyond the data on the number of noise and odour complaints received).

### 2.3.8 Resource use

It is common in life cycle analysis to include resource depletion as a negative impact of economic activity for a particular sector. In terms of economic accounts the issue is more complex. It is a basic proposal, in the green national accounting literature, that an economy which is extracting a non-renewable resource should deduct the value of this resource depletion from gNNP. In economic language, there is a ‘user cost’ which reflects the fact that resources used up today are not available for future use.

It is clearly the case that agriculture depends on non-renewable resources in a number of ways. For example, it uses intermediate goods which are either based on non-renewable resources (e.g. petrol) or have these resources embodied within them (e.g. energy or most manufactured inputs). Thus, physical shortages of these resources or proposed taxes on their carbon content will have implications for agricultural production. Therefore, resource use issues are appropriate for analysis within input-output accounts and policy modelling exercises.
However, it is not clear whether this practice also applies to a green account for the agricultural sector. Arguably, resource depletion (reflecting the scarcity of a non-renewable resource) should appear as a deduction from the true income of the primary sectors such as mining sector. If impacts of resource scarcity are priced properly in markets, then they should not be counted again in an environmental account for agriculture. In any event, purchases of non-renewable resource based products such as petroleum are intermediate expenditures. These outlays are not included in final output or value-added for the agricultural sector and so do not require further action. A caveat to this argument reflecting concerns about changing terms-of-trade (e.g. for non-renewable resources) is explored in Annex 2. Therefore, while this impact is described in Section 3, it is not taken any further in this analysis.

2.3.9 Summary of accounting adjustments

Adjusting any of the accounts or the balance sheet thus entails making a succession of additions and subtractions, as outlined above. Table 2.3 and Table 2.4 outline such adjustments for the welfare impacts of agriculture on society and its productivity impacts on other sectors, respectively. The tables cover the same impacts listed in Table 1.1 with the exception of resource use for the reasons presented above. The adjustments given in Tables 2.3 and 2.4 are examined throughout the remainder of the report, providing a framework for presenting physical data (Section 3), economic data (Section 4) and for aggregating total values for these adjustments (Section 5).

The only environmental asset from which positive service flows can be attributed is land, as this is considered under agricultural control. Changes to the quality of water or air are debited as depreciation on the stock of these assets. The positive services from air and water are not attributed to agriculture, whereas the positive service flows from land are attributed to agriculture. In Table 2.3, the columns for waste and noise are merged as both are negative service flows which are not linked to any particular stock (unlike the case with water or air).

Table 2.4 shows a range of possible accounting adjustments for impacts of agriculture on the productivity of other sectors - both negatively and positively. These costs and benefits would be subsumed within the accounts of other sectors (or within a set of national accounts), and are thus similar to transfer payments. Attributing these costs and benefits to agriculture provides a complete picture of the monetary value of agriculture’s environmental impacts.

| Table 2.3: Adjustments to the Agricultural Accounts for Welfare Impacts on Society |
|-------------------------------------------------|-------------------------------------------------|
| Service flow | Stock change (quantity and quality) |
| **I. Water** | | |
| Flow not attributable to agriculture | Value of water pollution arising from agricultural production |
| Flow not attributable to agriculture | Value of agricultural water abstraction |
| **II. Air** | | |
| Value of the stock of clean air is outside the province of agriculture but the cost of dust and allergens should be accounted for | Value of air pollution arising from agricultural production |
| **III. Soil** | | |
| Impact of (net) soil erosion on-farm on current yields is already accounted for | Value of (net) soil erosion on-farm on future yields |
| **IV. Landscape** | | |
| Value of landscape amenity services by the current provision of landscapes (within the agricultural sector) | Value of (net) change in landscape amenities |
| **V. Habitats and Species** | | |
| Value of habitat and species protection services provided by current land-use (within the agricultural sector) | Value of (net) change in habitats and species |
| **VI. Waste** | | |
| Value of waste pollution and disamenity arising from agricultural production | |
Table 2.4: Adjustments to the Agricultural Accounts for Impacts on Other Sectors

<table>
<thead>
<tr>
<th>Productivity Gain or Loss</th>
</tr>
</thead>
<tbody>
<tr>
<td>I. Water</td>
</tr>
<tr>
<td>(-) Cost of water pollution clean up costs</td>
</tr>
<tr>
<td>(-) Costs of flooding</td>
</tr>
<tr>
<td>II. Air</td>
</tr>
<tr>
<td>Included in value of stock change in Table 2.3</td>
</tr>
<tr>
<td>III. Soil</td>
</tr>
<tr>
<td>Impact of (net) soil erosion on-farm on current yields is already accounted for</td>
</tr>
<tr>
<td>(-) Cost of off-site soil erosion (cost of dredging streams, etc)</td>
</tr>
<tr>
<td>IV. Landscape</td>
</tr>
<tr>
<td>(+) e.g. Value of landscape to tourism</td>
</tr>
<tr>
<td>V. Habitats and Species</td>
</tr>
<tr>
<td>(+) e.g. Value of habitats and species to tourism</td>
</tr>
<tr>
<td>VI. Waste</td>
</tr>
<tr>
<td>None</td>
</tr>
<tr>
<td>VII. Nuisance</td>
</tr>
<tr>
<td>(-) e.g. cost of dealing with nuisance complaints</td>
</tr>
</tbody>
</table>

2.4 Establishing Boundaries for the Environmental Accounts for Agriculture

This section provides further discussion on a number of issues that affect the environmental accounting framework for agriculture and its practical application. These include treatment of subsidies and taxes (2.4.1); measuring non-produced (non-market) outputs and environmental services against a baseline or relative to a counterfactual (2.4.2); treatment of statutory provision of environmental or cultural assets (2.4.3); agricultural land as a sink for pollutants from other sectors (2.4.4); and the implications of using marginal or average economic value estimates (2.4.5).

2.4.1 How do we account for subsidies and taxes?

One purpose of a set of environmental accounts is to record the value-added by the sector in question and to add in or net out any positive and negative non-market impacts (or externalities). The correct procedure is for the accounts to record these impacts regardless of the fact that they may be ‘internalised’ through policy measures. Consider the example of waste that goes to landfill. Such waste should be debited with the external costs of transportation of the waste to a landfill site and the relevant site externalities (disamenity, greenhouse gas emissions etc.). In the UK, waste sent to landfill is subject to a landfill tax. Does the existence of such a policy instrument remove the need to account for the adverse impacts of waste that still ends up in landfills? The simple answer, from an accounting perspective, is arguably no. That is, a primary rationale for environmental accounts is to account more comprehensively for changes in the quantity and quality of natural assets regardless of the policy measures in place.

The question is, then, how the adverse impacts of waste going to landfills should be valued. If, for example, it can be assumed that the Landfill Tax is ‘optimal’ (in the sense of internalising the level of externality consistent with an economic optimum) then the correct deduction would be the unit tax rate multiplied by the quantity of waste going to landfill. It is not currently clear if the Landfill Tax ‘over’ or ‘under’-regulates landfilled waste from the standpoint of economic analysis. Current levels of the tax are, for example, designed to help achieve the goals of the EU Landfill Directive which itself may or may not pass a cost-benefit test. Efforts to debit, in this case, agriculture with the ‘optimal’ level of externality would be beset by all kinds of difficulties. In addition, as a practical matter, it is more straightforward to debit the entire externality, optimal or otherwise, to the externality-creating activity using a ‘direct’ estimate of marginal damage. And, as mentioned above, this is what is done in Section 5.

While this procedure is the correct one, it does mean that the resulting accounts need to be used with care. For example, if landfill externalities appear to be ‘large’ it would be tempting to conclude that they should be addressed by some policy measure aimed at reducing waste from agriculture. Unless the resulting externality is non-optimally controlled, i.e. less control is exerted than should be the case, then no such policy implication follows from the accounts. Put another way, the policy implications of the

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13 Economic theory requires that only the ‘non-optimal’ part of the externality should be internalised, leaving an ‘optimal’ level remaining. Essentially, non-optimal externality is the damage that costs less to abate than the benefit of abatement.
accounts can only be determined by securing additional information about existing policy measures and the extent to which they optimally internalise the externality. However, in principle, deductions from the accounts typically mean that savings should be higher to offset any losses in future wellbeing as a result of, say, waste disposal to landfill today.

2.4.2 What is the appropriate baseline or the implied counterfactual?

A particular challenge in building a framework for environmental accounts is the treatment of the counterfactual or the baseline. As is further discussed in Annex 2, the framework here assumes a baseline of no agriculture. This translates into no agricultural activity or zero water or air emissions from agriculture or zero provision of landscapes, habitats and species associated with agriculture.

This baseline assumption might appear at odds with reality. For policy purposes ‘zero agriculture’ is not credible since it implies that we would have to stop eating beef and milking livestock or else import all agricultural produce from abroad. Assuming a world without agriculture also begs the question: if there was no agriculture what would be the alternative land use? Clearly, this is an insoluble question: that is, presumably some land would revert to a rather different natural or semi-natural state while other land might be developed in some way, but what would happen, on average, would not be known with any precision. Much would depend ultimately on ‘unknowns’ such as future public policy. However, this is immaterial to the accounting exercise. National accounts are intended to provide a ‘snap-shot’ of the world as it is now. This suggests that analytical anxiety about the future in the context of large-scale future reform of the agricultural sector, while important from a policy perspective, is not the domain of national or environmental accounts as the discussion below attempts to demonstrate.

First, a distinction needs to be made between the choice of baseline for policy purposes and the choice for accounting purposes. Accounting procedures work on the basis of a ‘with and without’ the level of agricultural activity that prevails. Assuming a world without agriculture also begs the question: if there was no agriculture what would be the alternative land use? Clearly, this is an insoluble question: that is, presumably some land would revert to a rather different natural or semi-natural state while other land might be developed in some way, but what would happen, on average, would not be known with any precision. Much would depend ultimately on ‘unknowns’ such as future public policy. However, this is immaterial to the accounting exercise. National accounts are intended to provide a ‘snap-shot’ of the world as it is now. This suggests that analytical anxiety about the future in the context of large-scale future reform of the agricultural sector, while important from a policy perspective, is not the domain of national or environmental accounts as the discussion below attempts to demonstrate.

Secondly, zero agricultural activity is exactly the same baseline as that which is used by the conventional sectoral accounts. We measure UK agricultural output as the total market value of output (due allowance being made for subsidies), and that total market value has a baseline of zero agricultural activity. In other words, total market value approximates what the UK would have to pay for imports if there was zero agricultural activity in the UK (again with due allowance for price distortions such as subsidies). Since the implied baseline for measures of conventional output is zero, the same baseline is required for the environmental impacts.

This baseline assumption means that all of the positive impacts of agriculture (or environmental services provided by it) in a given accounting period should be quantified. The same holds for all of the negative impacts of agriculture in a given period. For the purposes of this framework, the monetary expressions of environmental impacts are the product of the unit economic value of the impact and the physical quantity of the impact as shown in Section 5. It is in the calculation of environmental impacts, specifically in the application of valuation data, that the treatment of the baseline becomes transparent. The theory underpinning the use of valuation data and the zero agriculture baseline is further explored in Annex 2, while an overview of key differences in practice in approaching the baseline is presented here for each impact category.

- **Water**: calculations for water quality assume a baseline of zero water pollution from agriculture. The negative service flow created by agricultural water pollution is the welfare value of moving from poor water quality (that results from agricultural pollution) to good water quality. This baseline implies that the public has a right to good water quality. For water availability, the baseline is zero abstraction by agriculture or that the amount currently abstracted by agriculture is left in the environment. Thus, the environmental cost of abstraction is imposed by the entire quantity of water abstracted by agriculture (net of recharge).

- **Air**: the environmental damage from air pollution uses a zero emissions baseline in that it presents the economic cost of all emissions from agriculture (net of sequestration). While the unit damage estimates for some pollutants (or rather the epidemiological evidence underlying these) may be based on some ‘safe minimum’ standards, on the whole, the zero emission approach is adopted.
o Soil: again, the baseline of no off-farm soil erosion from agriculture is assumed. The possibility that alternative land uses may cause more or less off-farm soil erosion than currently caused by agriculture does not enter into the baseline discussion. As discussed above, on-farm soil erosion is assumed to be already included in the sector accounts terms of effects on current period crop yields but the effect on future yields is not possible to quantify at present.

o Landscape, habitat and species: Since all these three impacts are related to land use, they can be looked at together in the context of the zero baseline. In fact, they are valued as a bundle in most of the economic valuation studies as they are delivered jointly. Of interest in respect of the accounts, are the value of current environmental services generated by existing land-use and changes in asset value that occur when actual land-use is altered (or degraded).

Matters, in principle, are relatively straightforward for the case where land is being switched from one use to another, i.e. estimating the change in land asset value that arises because of the change in land-use. Vincent (1999) and Atkinson and Hamilton (2003) show that the (net) change in the value of the land asset has two components. In the case, say, of a switch in land-use to an agri-environment scheme from a less ‘environmentally-friendly’ form of farming on that land, these components are: (i) the present value of the unit of land under the agri-environment scheme; and, (ii) the present value of land under the previous agricultural use. The net change in the value of the land asset from making this land-use switch is (i) minus (ii). Hence, this calculation takes account of the ‘counterfactual’ but, clearly, this is because in this case the alternative land-use is observable.

Quantifying the counterfactual is more difficult in the calculation of current environmental services from an existing (e.g. unaltered) land use as is the requirement here. However, as discussed above, what would happen in absence of agriculture may be interesting in a policy context but is not the domain of national or environmental account. The aggregate value of the environmental services is the product of the unit economic value differentiated by land, habitat and species type and the physical quantity of these assets - as far as available economic and physical data allow.

o Waste: a baseline of zero waste generation by agriculture is adopted and the entire quantity of the general waste category produced by the sector is valued using the unit economic damage estimate.

The final point about the baseline concerns the economic data. Some valuation studies in the context of landscape, habitats and species elicit WTP for ‘maintaining’ the current environmental assets (akin to zero emission or activity baseline), while others specify an alternative resource use or provision. The former, i.e. WTP to maintain the current assets and services, are the most relevant for the context of environmental accounts. This is explored in more detail in Annex 2. Study selection criteria employed in Section 4 include this context issue but the selection is ultimate limited by available information.

2.4.3 Statutory requirements - who provides the benefits?

Another issue concerns the assets provided by agriculture in compliance with statutory requirements. The methodological issue here is the extent to which services provided are attributable to the agricultural or government accounts. Examples include cultural assets such as listed buildings or protection of certain types of land use either by legal requirement or in exchange for monetary compensation as in the case of agri-environmental and protected area schemes. In so far as the responsibility for their value is concerned, then it is arguable that (a fraction of) this value arises from government regulation and hence the value should be assigned to government accounts. Alternatively, in so far as these assets are provided on land managed by agriculture, then it could equally be argued that they should appear as part of the agricultural accounts. For example, farmers will typically bear a cost for honouring any statutory obligation to protect land or sites and so it could be argued that the asset is ‘produced’ by agriculture.

The resolution of this issue is clearly important if the objective is to derive a consistent set of ‘green’ accounts across all economic sectors of interest. However, as the objective of the current study is to account for the environmental impacts of agriculture rather than to determine, say, the ‘true’ level of agricultural net value-added, a definitive resolution is not needed. That is, it simply could be noted that some of the categories of value that are accounted for might plausibly represent ‘transfers’ to another (e.g. the government) sector.

Nevertheless, it is of more than theoretical interest to know unequivocally which sector or accounts should get the credit for providing these mandatory assets. It is not surprising that determining the ownership of such assets is not straightforward. What the government is, in effect, doing is finding some
means to require a farmer to provide a (quasi) public good. While farmers may provide these in some degree without the legislative encouragement, there would be no reason to expect that they would do so in optimal or socially desirable amounts. It is through regulation that the government acts as intermediary between prospective managers of these assets, i.e. farmers, and those who are willing to pay for their protection, i.e. households. This suggests that a national accounting matrix approach such as that extensively investigated by Hartwick (2000) might be a fruitful way of analysing these linkages. For example, a variety of implicit and explicit transactions could be described within an accounting framework, whereby government sector pays farmers to protect land which is, in turn, financed by, say, taxation of households. Farmers incur costs (including opportunity costs) in providing protection services which are enjoyed ultimately by those in the household sector either through the generation of use (e.g. recreational values) or non-use values. The calculations shown in Section 5 assume that all statutory requirements are provided by agriculture and hence the benefits should be accredited to agriculture in the sectoral accounts.

2.4.4 What about other sectors’ impact on agriculture?

Pollution (e.g. ground-level ozone) from other sectors of the economy can lower productivity, and therefore output, in the agricultural sector. While it might be interesting to know the magnitude of this lost output, the conventional national accounts will reflect these impacts. That is, GNP will be lower because of the negative effect other sectors have on the agricultural sector.

However, from the perspective of measuring the ‘true’ income in the agricultural sector and the polluting sector which adversely affects farming output it is arguable that: (a) the ‘true’ income in the agricultural sector should be higher (as it would have been in the absence of pollution impacts from the other productive sector); and (b) the ‘true’ income of the polluting sector should be correspondingly lower. This re-attribution of pollution costs between victim and polluter leaves national income (e.g. gNNP unaffected). This is similar to treatment of the impacts of agriculture on other sectors but is outside the scope this accounting exercise.

2.4.5 Should be we using marginal or average economic values?

The economic valuation literature contains both average and marginal estimates for environmental costs and benefits. For example, air pollution damages (£ per tonne of pollutant) are generally averages, while some ecological damage estimates use measures like ‘x’ hectares loss of a particular habitat where ‘x’ is relatively small and hence the WTP measure is marginal.

In the case of the average impact estimates, it is far from certain that one additional (marginal) unit of, say, airborne emission will cause the same (average) amount of damage. In fact, the marginal damage may be higher or lower than the average. A typical example of increasing marginal costs may be observed in the cases of certain pollutants and damage to (increasingly) rare species and ecosystems. Destroying the first animal of a common species may cause only limited damage, but killing the last animal of this species will be ‘unaffordable’. Another peculiar situation occurs there is threshold level, below which no discernable impact on the population of the species occurs.

The opposite relationships exist for noise for which the marginal damage costs are generally lower than average. This may be encountered in the case of the noise produced by one additional truck adds less than proportionally to the noise nuisance caused by all trucks which are already on the road. A similar reasoning may be applied in the case of landfill sites. It is mainly the landfill itself which causes environmental harm and nuisance. One additional load of waste deposited on an existing landfill adds relatively little to the total damage. However, if landfill capacity is exhausted and a new landfill site has to be created for the additional amount of waste, the marginal damage is higher than average.

These examples show that using average instead of marginal values may lead to underestimates as well as overestimates and that the choice is also relevant to the baseline issue. Where the literature provides a marginal measure (and this usually the case for land based assets and services such as landscape and so on), this has been used in the calculations in Section 5. Otherwise, average estimates (e.g. for air pollution) are used. The choice is, effectively, made by the availability of data.
3 Environmental Impacts of Agriculture

This Section covers the main environmental impacts identified in Table 1.1, providing an overview of the environmental impacts of agriculture in each ‘impact category’, introducing the areas where positive and negative interactions between agriculture and the environment occur and the processes that lead to such results. The aim is that these will provide a basic understanding of how the raft of agricultural inputs affects different environmental endpoints.

This section also presents the data selected from the literature as the most appropriate to use in the accounting framework. There are two Annexes that accompany the information in this Section. Annex 3 illustrates agricultural inputs, environmental pathways and impacts and how the available data cover these. Annex 4 presents those data that were selected as a result of the review of data sources, but cannot be used in the environmental accounts for a variety of reasons discussed below under each impact heading.

3.1 Overview

3.1.1 Establishing Environmental Impacts

Agriculture's role in the UK economy is small, accounting for only 0.8% of the Gross Domestic Product in 2000 (National Statistics online). But in terms of natural resource use its role is significant, accounting for a high percentage of land use (e.g. 74% of land in the UK). While agricultural productivity has improved substantially, particularly in recent decades, it has often led to environmental degradation such as soil erosion and water pollution. However, agriculture also has positive impacts including helping to create and manage landscaped and wildlife habitats; and maintaining soils under agriculture which can, amongst other things, act as a sink for greenhouse gases and provide water filtering and retention functions.

In examining interactions across a broad front, several attributes of the agriculture - environment relationship need to be considered:

- **Agriculture is relatively diverse.** There are more than 200,000 holdings in the UK containing a range of different enterprise types adapted to different environmental and economic conditions. Overall there is a trend towards specialisation and increases in scale of farms. However, there are large numbers of farms of different sizes with some supporting a mix of arable, livestock and more specialist forms of production.

- **The interactions between agriculture and natural environmental processes are distinctive because farming forms a part of an ecosystem rather than being external to it,** unlike most other economic activities. Agri-environmental relationships are often complex, site specific and non linear reflecting a wide range of biophysical conditions (e.g. variations in climate, soils, availability of water resources and land use patterns). Farming manipulates the natural environment to produce agricultural commodities, through a range of different practices, i.e. drainage of land, tilling of soil, diverting of natural water sources, using irrigation, and applying nutrients/pesticides. The agro-ecosystem, like a natural one, is dynamic with a constant cyclical flow of inputs entering the system and outputs leaving. At the same time natural cycles (e.g. carbon and nutrient cycles) affect agriculture and are altered by farming activities.

- **The spatial distribution of agriculture is such that similar farming types will occur in a range of different environmental conditions.** A specific practice in one location can have significantly different environmental outcomes in another. In this analysis, the focus is on the relationship between farming practice and the environment (e.g. the application of nutrients and the enrichment of fresh water), since this is more precise than associating a general type of farming with an environmental impact. Nonetheless, the variations are such that it is difficult to extrapolate from the local to the national level. Many estimates of the impact of farming on a national scale rely on scaling up evidence gathered in a relatively limited area, sometimes over a limited period of time. This can lead to distortions and a considerable degree of caution is required in utilising some of the figures quoted in the literature.

- **The timescale over which impacts become apparent also varies greatly.** Some environmental changes, such as the removal of stone walls, are easily visible even though they may take place but over an extended period (e.g. when walls are neglected and gradually collapse). Other changes, such
as the contamination of groundwater, may occur over a much longer period of time and may not be detected until many years after the critical farming activities have taken place. Consequently, it is difficult to be precise about the impact of farming in any given year and the relevant evidence may not be available for many years after the event. In this study it has been necessary to use environmental data from a range of different years because of their availability. In some cases, this is not a major problem because the impacts appear to be relatively consistent over time or can be directly related to an easily identifiable variable (e.g. the total population of certain livestock will have a close bearing on their methane production). In others, variations between years, uncertainties over the direction of change since the year in question, or other factors mean that the earlier data are likely to be much less satisfactory as a guide to impacts in a more recent year.

- Some environmental impacts are not confined to one particular medium. For example, the use of herbicides may have a direct impact on the target plants, cause some contamination of ground water, affect the feeding and breeding habits of certain wildlife species and alter the appearance of the landscape. In analysis presented here, it is often necessary to make links between different impacts and there may be multiple references to a single input such as herbicide. See Figure 3.2 for an illustration of the multi causal nature of one type of farm input, pesticides. Further links are shown in data tables in the rest of the section following the DPSIR framework.

- Agricultural impacts on the environment are mainly diffuse. One consequence of this is that the collection of data is relatively expensive and reliable time series information relatively hard to come by. In addition, there are a number of areas where agriculture is not solely responsible but does contribute to an environmental outcome. In seeking to quantify impacts it may be necessary to apportion a share of the final outcome to agriculture but in few cases can this be very accurate.

- There is incomplete knowledge or data to establish clearly the precise environmental impact of several agricultural activities. Knowledge is constantly improving but the full impact of particular farming practices, say, on biodiversity is a long way from being fully understood.

These characteristics make it more difficult to attribute a precise environmental impact to agriculture than is the case for most other economic activities. It is necessary to make a considerable number of judgments, estimates, approximations and assumptions in order to provide some basis for the quantification of environmental impacts.

In order to illustrate the complexities of the interactions between agriculture and the environmental impacts that result Figures 3.1 and 3.2 have been included. The first of these shows the interactions and inputs at a generic level; demonstrating some of the input and output linkages, with agricultural management practices as the central variable. An arable farm has been taken as an example.

Figure 3.2 shows the impacts potentially arising from a single measurable input, pesticides, and how these impacts produce a more complex pattern of interactions than may, at first, be apparent. Secondary and tertiary impacts may be as important as primary ones, but are more difficult to quantify in relation to the level or nature of the input use.
Figure 3.1: Farm management choices and outcome on an arable farm - a simplified view

Agricultural Practices on Farm
- seed growing, plough, buy
- tree seedling, plant, spray, fertilise (manage hedge), harvest, dry, store, sell

Agricultural Outputs
- eg. Wheat, barley, set-aside (obligatory via policy), break crops such as OSR.

Non-Agricultural Outputs
- eg. fellow, hedges, etc., required partly for farming, other 'private' goods such as game birds farmed, other 'public' goods eg landscape - some 'provided voluntarily, others under contract.

Environmental Outcomes
- Subject to natural forces eg weather
- Mainly derive from combination of farm capital and practice. Differences between outputs within arable spectrum will exist (eg, more pesticides for some crops) but often minor.
- Some derive directly from policy (eg set-aside)
- Specific practices have different results depending on place and circumstances.
- Often time lags and uncertainties between practice and outcome.

Off-farm Input
- eg. fertiliser, fuel, contractors

Farm Capital Stock
- eg land, buildings, enhancement of landscape features
Figure 3.2: Environmental Pathways and Impacts - the example of pesticides
3.1.2 Collecting Physical Data

An array of primary data were collected to which economic valuation data could potentially be applied. This review of data sources was conducted systematically based on the impacts identified at the scoping stage of the study, in consultation with Defra.

A set of selection criteria was applied to arrive at the data presented in Annex 4. In an attempt to ensure accuracy and creditability, official statistics quoted by governmental departments were preferentially used, i.e. those available from the devolved administrations, Defra and DARD. When data from such sources were unavailable academic papers, independent studies and industry statistics were used. When using independent studies and papers preference was given to those quoted in government produced or respected studies. Studies and papers were also assessed in order to identify how robust they were in terms of methodology. Once it had been decided to use information from a study, the limitations of it were assessed and the potential for overlaps with other data sets highlighted. Where a range of data was considered to be accurate the most recently collected statistics were used.

From the set of data presented in Annex 4, data were selected for input to the environmental accounting framework. The reason for selecting data from this set was that they met some or all of the following criteria:

- that the data were from a reliable source;
- that they match with the economic data - in terms of units and changes in the environment being valued;
- that the data were updated on a regular basis, or part of a survey;
- that the data provide a convenient accounting metric;
- that it is available across all countries in the UK; and
- that it is already presented in the environmental accounts: Agriculture in the UK (Defra et al, 2004).

In order to provide a framework for examining the data on these complex interactions, the DPSIR framework is applied. Figure 3.3 uses the DPSIR framework to show the associations that exist between driving forces within agriculture and final impacts on the environment. The figure provides a number of examples, including the use of fertilisers, pesticides and energy and specific farming practices. It does not attempt to provide a complete picture of all the driving forces and associated impacts, but rather serves to illustrate a framework for assessing and categorising the data presented in this section.

Consistent with the DPSIR framework, the data presented in Annex 4, and in this Section, have been labelled as fitting into the D, P, S, I or R categories. For example, there are data on the tonnes of pesticide used per year (a driving force indicator); the concentration of pesticide in coastal waters (a state of the environment indicator); the number of people who become ill from bathing in contaminated waters (an impact indicator); and the uptake of agri-environment schemes (a response indicator). The usefulness of each type of data for the environmental accounts will depend on both the ‘closeness’ of the data to estimating the final impact and on the availability of economic data which match the environmental data. By illustrating from which stage of the impact pathway data are derived double-counting can be avoided in the final aggregation stage.

The data in Annex 4 are also presented in terms of the country they pertain to, the data source and the most recent year of data collection. Grey shaded cells indicate that data were not found for a particular country.
### Figure 3.3: Example of DPSIR Impact Pathway Categorisation for Data

<table>
<thead>
<tr>
<th>D (Driving Forces)</th>
<th>P (Pressures)</th>
<th>S (State of the Env.)</th>
<th>I (Impacts)</th>
<th>R (Response)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Application of pesticides</td>
<td>Leaching and runoff of pesticides to water courses</td>
<td>Concentration of pesticides in water, food and on land</td>
<td>Damage to fish nursery grounds</td>
<td>EU Drinking Water Directive</td>
</tr>
<tr>
<td></td>
<td>Spray drift of pesticides, dry deposition</td>
<td></td>
<td></td>
<td>Various regulations for pesticides content and use</td>
</tr>
<tr>
<td></td>
<td>Emissions of local, regional and global air pollutants</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Ingestion, inhalation through food, direct contact or drinking water</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Application of fertiliser and animal wastes</td>
<td>Nitrate loss through leaching</td>
<td>Eutrophication of surface waters and toxic algal blooms</td>
<td>Human health - blue baby syndrome, risk factor in gastric cancer</td>
<td>EU Drinking Water Directive</td>
</tr>
<tr>
<td></td>
<td>Ammonia volatilisation</td>
<td>Increased concentration of nitrogen in water and soils</td>
<td>Climate change</td>
<td>Regulations for good management in Nitrate Sensitive Areas and NVZs</td>
</tr>
<tr>
<td></td>
<td>Denitrification</td>
<td>Increased concentration of N₂O in the atmosphere</td>
<td>Deaths of animals from toxic-blooms</td>
<td>Nitrates Directive</td>
</tr>
<tr>
<td></td>
<td>Mineralisation of soil nitrogen</td>
<td></td>
<td>Long-term disruption of ecosystems</td>
<td>Codes of good agricultural practice</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Species diversity reduction in grasslands</td>
<td>Water Framework Directive</td>
</tr>
<tr>
<td>Agricultural practices that lead to soil erosion</td>
<td>Loss of soil to other sites, to water courses, etc (by wind, frost, water, animals) - especially loss of fine soil particles</td>
<td>On-farm: reduced soil storage capacity; loss of nutrients; deterioration of soil structure</td>
<td>Deterioration in aquatic habitat</td>
<td>Codes of good agricultural practice such as minimal tillage and cultivation across slopes</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Off-farm: flooding of properties; roads and drains blocked with soil; Sedimentation in rivers</td>
<td>Blocking of waterways, roads, and so on</td>
<td></td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>Reduced productivity of soils</td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td>Loss of housing stock</td>
<td></td>
</tr>
<tr>
<td>Energy use</td>
<td>Emissions of local, regional and global air pollutants</td>
<td>Change in atmospheric concentration of pollutants and changes to atmospheric chemistry</td>
<td>Climate change - damages to human health, infrastructure, livelihoods, loss of habitat and species</td>
<td>Climate change levy</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Acidic deposition</td>
<td>Costs to human health from local air pollution</td>
<td>Other air pollution and climate change regulations and emission reduction targets</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Reduced productivity of crops</td>
<td></td>
</tr>
<tr>
<td>Waste generation</td>
<td>Tonnes of waste to landfill and incineration contribute to emissions of pollutants to land and air</td>
<td>Impacts on atmospheric chemistry, on water ecology</td>
<td>Disamenity effects of incinerators</td>
<td>Landfill tax</td>
</tr>
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<td></td>
<td></td>
<td></td>
<td>Health costs</td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td>Loss of recreational value</td>
<td></td>
</tr>
</tbody>
</table>
3.2 Summary of Environmental Impacts

Each impact category is addressed below, following the order of Table 1.1, and data selected for the accounting framework is presented at the end of each sub-section. Annex 4 provides the complete and extensive set of data reviewed for the study.

3.2.1 Water

Agriculture has a significant role in determining the quality of fresh water in large parts of the country and also influences water levels and the management of water resources because of its own water requirements. Clearly there are relationships between water quality and water quantity issues but they can be conveniently separated into two categories to simplify discussion: water quality (change in chemical and biological status) and water availability (change in flow patterns).

Water quality

Description of Impacts

Farming is one of four major sources of water pollution in the UK. A distinction can be made between diffuse pollution, arising from the spreading of nutrients on the land, for example, and point sources that include run off from livestock buildings. Key areas of concern in relation to water quality are nitrate pollution in surface and groundwater, phosphorus levels in surface water, contamination by pesticides and harmful effects of soil sediments and mineral salts. A decline in water quality can result in impaired drinking water quality, resulting in potential health implications, as well as environmental problems.

Agricultural nutrients, primarily nitrogen and phosphate are probably the most diffuse and important source of water pollution. Excessive levels of nitrogen and phosphate in the environment arise from the application of both manures and inorganic fertiliser. It is estimated that between 1995 and 1997, 47% of nitrogen inputs to farmland were from inorganic fertilisers, 28% from manures and 25% from other sources (Environment Agency, 1998a). As a rule of thumb it has been estimated that about two thirds of nitrogen emissions to surface and marine waters and one third of phosphorus are present as a result of agricultural activities (Environment Agency, 1998a). Phosphate in surface water also arises from urban and domestic sewage where levels are declining. Agriculture appears to represent a growing proportion of a phosphate pollution load that is falling overall.

Increased resource efficiency has resulted in a decline in the use of several agricultural inputs, including inorganic fertilisers, in the UK in recent years and this has had a beneficial effect on the pollution burden. For example OECD (2001) suggest that in 1995-1997 the UK had a positive nitrogen balance of 86kg/ha (equivalent to an efficiency ratio of 48 per cent) having been reduced from 107 kg/ha measured in 1985-87. These figures illustrate the continued potential for relatively high levels of nutrient pollution, even though this is not a direct measure of the actual pollution load and does not take into consideration the spatial variability in nitrogen inputs and hence impacts. Pesticide use, as measured by the weight of active ingredient applied by farmers, is also declining. It fell from 40,826 tonnes of active ingredient in 1985 to 35,432 tonnes in 1997 (MAFF 2000) and will have fallen further since. However, it is argued by some that the effectiveness of pesticides as toxins has been rising, so newer compounds have had to be applied in smaller quantities than previously, although not necessarily less frequently. Consequently the environmental burden will not have fallen in parallel with the reduction in applications. Standards for the maximum admissible concentration of pesticides in drinking water are relatively stringent. Despite the fall in fertiliser use and the low concentration of pesticides in surface and ground waters, the UK water industry is still reported to invest large sums of money to remove nitrates, phosphates and pesticide residues from water to comply with EU drinking water standards (Pretty 2000).

There are several different pathways whereby pollutants reach water bodies (e.g. dissolved in runoff, attached to soil particles, and atmospheric deposition). Pollution from agriculture occurs in both surface and
ground water and in some marine waters, particularly shallow estuarine areas but also in larger bodies of water such as the North Sea. Nutrients may leach through the soil into groundwater or may run off the surface of fields into streams and ponds. Similarly, other inputs, including pesticides, veterinary products and trace elements in fertilisers can either leach or run off, with the level of contamination potentially varying with weather conditions, farm practice and local geology. Phosphates can be carried into water courses dissolved in surface runoff or attached to soil particles eroded from farmland.

Other sources of water pollution include heavy metals, which may enter the environment as a result of their presence in inorganic fertilisers, manure, sewage sludge and pesticides or as a consequence of aerial deposition onto land. Cadmium, which is associated with the use of phosphate fertilisers and copper found in pesticide compounds are of particular concern. Biological contaminants include pathogens which can be transferred from manure and sewage sludge into water bodies or be traced back to the disposal of animal carcasses on farms - a practice now prohibited.

Point sources of pollution are associated with housed livestock and other facilities such as slurry stores, silage making clamps, pesticide stores, crop treatment processes and so on. Dairy farms are one of the main sources of surface water pollution, often associated with the run off of slurry, yard washings, silage effluent and milk in some cases. Serious pollution incidents often involve sharp increases in Biological Oxygen Demand (BOD) and fish kills, causing considerable damage to aquatic ecosystems as well as contributing to nutrient enrichment in most cases.

An adequate supply of nutrients is essential for plant growth and hence agricultural production. But a surplus of nutrients, in excess of immediate crop needs, can be a source of potential environmental damage to surface and ground waters (eutrophication). Eutrophication can affect both fresh and marine waters via a process of rapid algal growth resulting in reduced levels of photosynthesis in other aquatic flora and an eventual drop in the availability of oxygen for other forms of aquatic life. Nutrient enrichment affects the ecological functioning of a wide range of habitats and in nearly all cases reduces their ability to house species of conservation concern. Nitrate in drinking water is subject to strict limits on the level of contamination under EU legislation primarily because of the link to blue baby syndrome. This requires the removal of nitrate from drinking water, with resultant costs.

**Data on Impacts**

As Annex 4 shows there is a sizeable quantity and variety of data on the water environment and on agriculture’s inputs to the water environment. These are described below briefly according to the DPSIR framework. Similar explanations for the components of the DPSIR framework apply equally to other impacts, where data allow. Further details on the data, such as for which countries they are available, the data source and the most recent year, are included in the Annex.

- **D:** In terms of data on driving forces or agricultural practices, this includes data on the overall consumption and application of fertilisers and pesticides in agriculture, (e.g. to crops and grasslands);
- **P (pressures):** The data also report on the number of different types of water pollution incidents (e.g. due to piggeries, cattle, sheep dip, pesticides, vegetable washings and so on);
- **S:** In terms of state of the environment, the data reports concentration of pesticides in waters (including of specific chemicals), which will partly reflect the driving forces and pressures from agriculture above and will partly reflect other sources. Other such data include nitrate concentrations in rivers, % of rivers classified as being in chemically and/or biologically good status, and number of eutrophied water bodies; and
- **Linking data:** These data provide a link between D-P-S and agriculture. Linking data on the water environment includes: that 43% of all phosphates entering surface waters are from agriculture, that agriculture uses 89% of all pesticides, and so on.

Potentially the most useful set of data for the agricultural accounts for measuring the impact on surface (inland) waters is data on the chemical and biological quality of water. River water quality is one of the UK Government’s 15 headline indicators for sustainable development. While there is a host of other water data

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applicable to agriculture, as presented in the Annex, this data on river water quality provides the best fit with the way in which economic valuation studies are undertaken, namely focusing on perceived water quality. Changes in river quality are measured in England and Wales by the Environment Agency, in Scotland by the SEPA and in Northern Ireland by EHS. Both chemical and biological water quality data are presented annually. Table 3.1 presents the most recent figures (2002) for these measures of the current state of the water environment (S, in DPSIR), and thus reflect contributions from a variety of polluting sources, not just agriculture.

<table>
<thead>
<tr>
<th>Table 3.1: River Water Quality (State of the Environment) Data, 2002</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>BIOLOGICAL</strong></td>
</tr>
<tr>
<td>England</td>
</tr>
<tr>
<td>Wales</td>
</tr>
<tr>
<td>Northern Ireland</td>
</tr>
<tr>
<td>Scotland</td>
</tr>
<tr>
<td><strong>CHEMICAL</strong></td>
</tr>
<tr>
<td>England</td>
</tr>
<tr>
<td>Wales</td>
</tr>
<tr>
<td>Northern Ireland</td>
</tr>
<tr>
<td>Scotland</td>
</tr>
</tbody>
</table>

The advantages of these data are: (i) that they are regularly collected by reliable sources; (ii) that the units of measurement fit well with economic valuation data which elicits willingness to pay for different levels of water quality, and/or changes from one level to the next; and (iii) the data are available for all countries. The main disadvantage or drawback of using this data in the accounting framework is that they do not provide a link with agriculture. In order to attribute the benefits of good water quality or the costs of poor water quality to agriculture we have to rely on, the so-called ‘linking’ data. This is provided EA (2002) which reports from WRC (1999) that 70 per cent of all nitrogen inputs to inland waters are from agriculture.

For the impacts on the marine water assets, the data selected for the agricultural accounts relate to ‘linking data’ which describe the contribution of agriculture to faecal contamination of coastal waters as 5% (Environment Agency, 2002). This data were selected principally because of the existence of matching economic data on the value of clean bathing waters and the welfare impacts of faecal contamination in UK coastal waters.

Data selected for estimation of costs to other sectors are summarised in Table 3.2 below, for England and Wales only. This data on number of pollution incidents from agriculture and linking data on the contribution of agriculture to water pollution issues, is matched with economic data from Section 4 to arrive at the aggregate values presented in Section 5.
Table 3.2: Water Quality Data

<table>
<thead>
<tr>
<th>Water body</th>
<th>Data description</th>
<th>DPSIR</th>
<th>Country</th>
<th>Data</th>
<th>Year</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>marine</td>
<td>Faecal contamination of coastal waters due to farming</td>
<td>Linking</td>
<td>England and Wales</td>
<td>5%</td>
<td>2002</td>
<td>Environment Agency, 2002</td>
</tr>
<tr>
<td>surface waters</td>
<td>number of category 1 pollution incidents from agriculture (i.e. the most significant pollution events most likely to cause water pollution)</td>
<td>P</td>
<td>England and Wales</td>
<td>13 incidents</td>
<td>2002</td>
<td>Environment Agency online (1), 2002</td>
</tr>
<tr>
<td>surface waters</td>
<td>number of significant pollution events most likely to cause water pollution</td>
<td>P</td>
<td>Scotland</td>
<td>56 incidents</td>
<td>2002</td>
<td>SEPA online (1)</td>
</tr>
<tr>
<td>surface waters</td>
<td>number of category 2 pollution incidents from agriculture</td>
<td>P</td>
<td>England and Wales</td>
<td>137 incidents</td>
<td>2002</td>
<td>Environment Agency online (1), 2002</td>
</tr>
<tr>
<td>surface waters</td>
<td>% nitrates in surface waters due to farming</td>
<td>Linking</td>
<td>EU</td>
<td>70%</td>
<td>1992-1997</td>
<td>WRC 1999</td>
</tr>
<tr>
<td>surface waters</td>
<td>% phosphates in surface waters due to farming</td>
<td>Linking</td>
<td>England and Wales</td>
<td>43%</td>
<td>1993</td>
<td>Morse et al, 1993</td>
</tr>
<tr>
<td>surface waters</td>
<td>% pesticides in surface waters due to farming</td>
<td>Linking</td>
<td>England and Wales</td>
<td>89%</td>
<td>2002</td>
<td>Environment Agency, 2002</td>
</tr>
</tbody>
</table>

Water availability

Description of Impacts

Agricultural production affects the quantity of water available for other human consumption and natural processes in a variety of ways. One of these is the alteration of water flows, at a local level, arising from changes in land use. Converting natural vegetation to farmland increases the speed of overland flow rates, allowing less time for infiltration, hence reducing the recharge speed of aquifers while increasing the speed at which water reaches surface bodies and ultimately potentially increasing flooding. The increase in water flow rate can also contribute to the deterioration of water quality through mobilisation of salts and nutrients. The type of water flow and speed can also affect the erosion of top soil with factors such as soil cover, water retaining capacity and speed of over land flow affecting the quantity of soil eroded. Conversion of more natural vegetation to farmland now occurs rather infrequently and is subject to environmental impact assessment regulations. However, flooding is a topic of increasing concern and more attention is being given to changing farm management, including the provision of water storage on farms to deal with peak flows.

There is much concern about the potential link between agricultural practices, runoff processes and flooding, however, the nature of this link remains uncertain. The Environment Agency (EA, 2002) attempts to draw a link based on its own flood event data and finds that 25% of all flooding events in the 1980s and 90s were hillslope events. On the basis that 57% of hillslope events were caused by erosion and deposition the report concludes that at least 14% of all flood events, and flood event costs, should be attributed to agriculture. The EA purports that this estimate is likely to be conservative due to under-reporting of local agricultural related flooding events and also lack of data linking agriculture’s contribution to fluvial flooding.

A more direct method by which agriculture can affect water availability is through abstraction and the diversion of watercourses for irrigation. Irrigation is used to extend the level of agricultural production where the natural rainfall pattern is at variance with crop needs. The use of water for agriculture and food production can conflict with other human and ecosystem needs (e.g. over-abstraction is leading to the decline of aquatic habitats in some areas). Over-pumping of groundwater resources can have serious implications for base flow levels of surface water bodies in all areas. However, in coastal areas there is the added dimension that high pumping levels can allow seawater to penetrate into freshwater aquifers.
Agricultural water use does not represent a large proportion of water abstracted being about 1 per cent of the total in water use in the late 1990s. However, the extent of irrigation and abstraction for agricultural purposes has been increasing. Between the late 1980s and late 1990s water abstracted for agricultural use in England and Wales increased from 82 to 184 million cubic metres with irrigated land area increasing by 50% over the same period (OECD, 2001). Irrigation accounts for approx 150,000ha of land and 160,000 ML/day of water in a dry year (Defra e-digest of environmental statistics, 2003(20)). By 2021, the irrigated area is predicted to increase by 14% and the volume applied by 50% (Environment Agency, 2001b). Water use statistics are likely to be underestimates of agriculture’s contribution to water use. This is because the returns of water to the original source expected after agricultural use are far less, when compared to other high consumers (e.g. power generators).

In recent years farm storage of water has increased considerably in dryer parts of the country where abstraction is a particular concern. This shift has resulted in both positive and negative impacts. On the positive side the reservoirs represent a new habitat with associated biodiversity benefits and also benefit water management. However, they can also have a negative impact on the local landscape and valued landscape features.

Data on Impacts

Data on water availability presented in Annex 4 relates to the number of abstraction licenses granted to agriculture and the corresponding number of megalitres abstracted. Data selected for use in the accounts are the number of megalitres abstracted per year by agriculture in England and Wales (similar data were not available for Northern Ireland and Scotland). This data also provide a good match for economic data, presented in Section 4, which value the loss of non-market value of the water ecosystem from reduced water availability in the environment, and thus the depreciation on water natural capital.

Data on flooding, as described above, are limited to linking data that 14% of all flood events can be linked to agriculture. On this basis, the economic data presented in Section 4 on the total costs of flooding is applied to the accounts. The disadvantage of this data is that it relies on historical evidence and linking data, rather than an annual or more recent analysis.

| Table 3.3: Water Availability Data |
|-----------------------------------|-------------------|----------------|-------|------------------|
| Data description                  | DPSIR         | Country          | Data   | Year   | Source                |

3.2.2 Air pollution

Description of Impacts

Agriculture’s contribution to the level of atmospheric pollution is two fold. Depending on the type and scale of production methods, land managed for agriculture can act as a sink for pollutants already in the atmosphere, or emissions from manures and the soil itself can increase pollution levels. The main gas fluxes (in volume terms) from agriculture are ammonia, carbon dioxide, methane and nitrous oxide. However, the levels of specific volatile organic compounds such as hexachlorobenzene and methyl bromide are also affected. The processes that result in agriculture influencing the flux of gases and the effects these additions and removals have are both highly varied and compound specific.

Ammonia emissions in the UK are predominantly from agriculture and are associated with the production and use of nitrogen rich products particularly from cattle, poultry and pigs. The use of inorganic fertiliser is the second major agricultural source of ammonia, produced as nitrogen in fertiliser reacts with compounds in air.
and soil (Defra, 2002). Ammonia is a soluble and reactive, nitrogen based gas, and these characteristics influence its impacts on the environment. The effects of ammonia are generally local, although depending on meteorological conditions UK emissions could result in pollution in other countries. Ammonia is of concern due to its contribution to acidification and enrichment. When dissolved in water and deposited in the environment it reduces the pH of surrounding media. This, in turn, leads to increased pressure on species poorly adapted to low pH and a potential change to biodiversity quality. This is of particular concern as the current deposition of ammonia from the atmosphere is thought to be above the critical load - load above which harmful effects are thought to occur - for a number of semi natural habitats. In addition ammonia can also contribute to nutrient enrichment of soils and eutrophication in water courses which also affects the competitiveness of species and biodiversity. Other nitrogen based air emissions, i.e. nitrogen oxides, also have the potential to cause acidification and enrichment.

Emissions of carbon dioxide, methane and nitrous oxide are mainly of concern as they are all greenhouse gases. Agriculture is the main source in the UK of the two latter gases (UNECE, 2000). Carbon dioxide emissions from agriculture are mainly the result of direct and indirect fossil fuel use and the release of organic carbon contained in soils. This happens to some extent in all soils, but can be increased on agricultural land through activities such as ploughing of fields especially in previously unploughed areas. Agriculture also contributes to carbon dioxide emissions as a consequence of its use of electricity generated by burning fossil fuels and the use of materials such as fertilisers the manufacture of which results in significant carbon dioxide emissions.

Methane, in contrast, is predominantly generated as a result of enteric fermentation in ruminant animals (e.g. cattle and sheep). Nitrous oxide as well as other oxides of nitrogen are predominantly produced as a consequence of the reduction of nitrate in soils, hence linked to fertiliser use. Measurements of nitrous oxide in the Lothians (equivalent of 381,000kgN/yr) and the Ayshire Basin (equivalent of 794,000kgN/yr) have shown that gas emission levels vary according to levels of soil moisture and the type of landuse - being higher from wetter soils and grazed pastures (Lilly, 2003). In addition, nitrous oxide can be produced when waters are enriched with nitrogen. It has been calculated that an annual flux of 0.04kgN₂O/ha is emitted from major UK aquifers (Hiscock et al, 2003).

Volatile organic compounds (VOCs) emissions from agriculture are a consequence of the use of agro-chemicals. One of the most notorious of these is methyl bromide, which is to be banned, except for critical uses (amounting to an allowance of 129 tonnes in the UK for 2005) in all developed countries from 2005 (UNEP, 2004). This compound contributes to the breakdown of stratospheric ozone. VOCs in general can also contribute to the generation of low level ozone. While the impacts of low level ozone are not accurately identifiable, it has been implicated in crop damage and in increases in respiratory diseases.

As mentioned above, agriculture also contributes to the gas fluxes through the direct and indirect combustion of fossil fuels. As discussed in Section 3.2.8 agriculture contributes to the use of non-renewable fuels, firstly through combustion of petrol and diesel in motor vehicles both on and off-farm. This results in the same raft of pollutants generally associated with space heating and motor vehicles including carbon dioxide, nitrogen oxides, volatile organic compounds and particulates. Agriculture as a sector also consumes a considerable quantity of electricity, which if produced using fossil fuels results in emissions of various pollutants.

Although soil processes are instrumental in the emission of many airborne pollutants, they can also act as sinks or catalyse the change of a compound to a less reactive one, (e.g. soils can break down methane to carbon dioxide). An area of increasing interest is the ability of soil to act as a sink for carbon dioxide, even though figures highlighting the extent of this potential are in some cases contradictory. However, what has been established is that specific farm practices can be used to realise this potential. There are numerous factors that can affect the ability of soil to absorb carbon (sequestration) including the type of planting, fertiliser application, tillage, grazing, soil type, temperature and moisture.
Data on impacts

All of the air pollution data used in this study are provided by Netcen and present the emissions by country as much as possible (See Table 3.4). These data are also reported in ‘Agriculture in the UK’ (DEFRA et al, 2004) and are used in the ONS Environmental Accounts. The data is regularly updated and reliable, and also provides a good link with the economic data, which estimate the damages associated with air pollution.

The most recent year for available annual emissions data is 2001, the 2002 data will not be completed until September 2004. Two main categories of emissions data are available for air pollution attributable to the agricultural sector: (i) agriculture is the primary source of emissions and (ii) agriculture is classed as an end-user or secondary source of emissions. The end user categories of activity include agricultural road freight and electricity used by the sector. Given the scope of this study, which is confined to the ‘farm gate’, the latter emissions category is not considered here.

In the primary source category emissions are simply defined by the activity generating the emission. Data on carbon emissions are net emissions (gross emissions minus the carbon sequestration by soils and forestry). Primary source is further divided into:

- combustion: which includes emissions from cultures with and without fertiliser application, field burning, manure management, enteric fermentation and use of pesticides and limestone; and
- non-combustion activities: emissions related to fuel consumption from mobile machinery and stationery sources.

3.2.3 Soil

Description of Impacts

Soil is one of the fundamental assets of most farms, as well as a major component of the environment (Defra, 2004). Soil is also vulnerable, it takes hundreds of years or more to develop, but mismanagement can result its loss and degradation, sometimes quite rapidly with the greatest deterioration taking place on the most vulnerable land. Many farming operations involve working the soil, hence altering its chemical composition and structure. Exposing it to the elements and inappropriate management can lead to both on and off-farm problems discussed below. Soil on agricultural land can be used as a resource for purposes beyond agriculture, with soil processes allowing the sequestration of carbon (given the correct conditions) and the break down of other potentially harmful compounds such as methane. Soil is also fundamental to the use of manures and the reuse of waste generated by other industries (e.g. sewage sludge).

Changes in chemical, physical and structural soil quality over time will influence a farm’s productive potential as well as its environmental value. Changes to soil quality can be as a result of deliberate farm management such as spreading lime or as a consequence of external pressures such as flooding, runoff and air borne emissions which can affect soils on land both farmed and unfarmed. Deliberate alterations to agricultural soils include the addition of fertiliser, manure and sludge to improve nutrient content, lime to reduce pH, drainage and change in soil structure as a consequence of ploughing and cultivation. Most British soils in cultivation are reasonably robust but some are vulnerable for example peaty soils in part of the fens due to oxidation.

Less welcome impacts arising from farming operations occur as a result of overgrazing and poaching by livestock around feeding and watering points and winter harvesting of crops. Changes in soil structure may need to be remedied over several years. Other impacts on soil, which will have an influence both on and off-farmland, include enrichment as a consequence of runoff and air emissions and pesticide contamination.
### Table 3.4: Annual emissions of air pollutants from agriculture as a source (kT)\

<table>
<thead>
<tr>
<th>Pollutant</th>
<th>Source</th>
<th>England</th>
<th>Scotland</th>
<th>Wales</th>
<th>N. Ireland</th>
<th>UK</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>NON-COMBUSTION</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>CH4 - Total Agriculture</td>
<td>510</td>
<td>150</td>
<td>135</td>
<td>119</td>
<td>914</td>
<td></td>
</tr>
<tr>
<td>CH4 - Enteric Fermentation</td>
<td>444</td>
<td>137</td>
<td>125</td>
<td>107</td>
<td>813</td>
<td></td>
</tr>
<tr>
<td>CH4 - Manure Management</td>
<td>65</td>
<td>13</td>
<td>10</td>
<td>12</td>
<td>101</td>
<td></td>
</tr>
<tr>
<td>CH4 - Field Burning of Agricultural residues</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>N2O - Total Agriculture</td>
<td>57</td>
<td>14</td>
<td>9</td>
<td>7.9</td>
<td>88</td>
<td></td>
</tr>
<tr>
<td>N2O - Enteric Fermentation</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>N2O - Manure Management</td>
<td>2.8</td>
<td>0.67</td>
<td>0.43</td>
<td>0.57</td>
<td>4</td>
<td></td>
</tr>
<tr>
<td>N2O - Agricultural Soils</td>
<td>53.7</td>
<td>13.3</td>
<td>8.8</td>
<td>7.35</td>
<td>83</td>
<td></td>
</tr>
<tr>
<td>N2O - Field Burning of Agricultural residues</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>NH3 - Total Agriculture</td>
<td>172</td>
<td>41</td>
<td>28</td>
<td>31</td>
<td>272</td>
<td></td>
</tr>
<tr>
<td>NH3 - Cattle farming</td>
<td>120</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NH3 - Pig farming</td>
<td>25</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NH3 - Poultry farming</td>
<td>38</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lindane (gamma-HCH): Agriculture (non-combustion)</td>
<td>6</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Agricultural pesticides*</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hexachlorobenzene (HCB): Total Agriculture</td>
<td>549a</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>HCB: Agricultural pesticide (chlorothalonil)</td>
<td>470b</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>HCB: Agricultural pesticide (chlorthal-dimethyl)</td>
<td>77a</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>HCB: Agricultural pesticide (quintozine)</td>
<td>2a</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>COMBUSTION</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NOx: Total Agriculture</td>
<td>12.6</td>
<td>2.35</td>
<td>1.94</td>
<td>1.78</td>
<td>18.67</td>
<td></td>
</tr>
<tr>
<td>NOx: Agriculture (stationary combustion)</td>
<td>0.68</td>
<td>0.13</td>
<td>0.10</td>
<td>0.07</td>
<td>0.98</td>
<td></td>
</tr>
<tr>
<td>NOx: Agriculture (mobile machinery)</td>
<td>11.92</td>
<td>2.22</td>
<td>1.84</td>
<td>1.71</td>
<td>17.69</td>
<td></td>
</tr>
<tr>
<td>Non-Methane VOC: Total Agriculture</td>
<td>3.07</td>
<td>0.57</td>
<td>0.47</td>
<td>0.44</td>
<td>4.18</td>
<td></td>
</tr>
<tr>
<td>Non-Methane VOC: Agriculture (stationary combustion)</td>
<td>1.18</td>
<td>0.22</td>
<td>0.18</td>
<td>0.17</td>
<td>1.18</td>
<td></td>
</tr>
<tr>
<td>Non-Methane VOC: Agriculture (mobile machinery)</td>
<td>1.89</td>
<td>0.35</td>
<td>0.29</td>
<td>0.27</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td>SO2: Total Agriculture</td>
<td>0.99</td>
<td>0.19</td>
<td>0.15</td>
<td>0.15</td>
<td>1.47</td>
<td></td>
</tr>
<tr>
<td>SO2: Agriculture (stationary combustion)</td>
<td>0.46</td>
<td>0.09</td>
<td>0.07</td>
<td>0.07</td>
<td>0.68</td>
<td></td>
</tr>
<tr>
<td>SO2: Agriculture (mobile machinery)</td>
<td>0.53</td>
<td>0.10</td>
<td>0.08</td>
<td>0.08</td>
<td>0.79</td>
<td></td>
</tr>
<tr>
<td>CO: Total Agriculture</td>
<td>13.86</td>
<td>2.59</td>
<td>2.15</td>
<td>1.99</td>
<td>20.57</td>
<td></td>
</tr>
<tr>
<td>CO: Agriculture (stationary combustion)</td>
<td>9.69</td>
<td>1.81</td>
<td>1.50</td>
<td>1.39</td>
<td>14.38</td>
<td></td>
</tr>
<tr>
<td>CO: Agriculture (mobile machinery)</td>
<td>4.17</td>
<td>0.78</td>
<td>0.65</td>
<td>0.60</td>
<td>6.19</td>
<td></td>
</tr>
<tr>
<td>Carbon (from CO2): Total Agriculture</td>
<td>325</td>
<td>60</td>
<td>50</td>
<td>39</td>
<td>475</td>
<td></td>
</tr>
<tr>
<td>Carbon: Agriculture (stationary combustion)</td>
<td>135</td>
<td>25</td>
<td>21</td>
<td>12</td>
<td>193</td>
<td></td>
</tr>
<tr>
<td>Carbon: Agriculture (mobile machinery)</td>
<td>190</td>
<td>35</td>
<td>29</td>
<td>27</td>
<td>282</td>
<td></td>
</tr>
</tbody>
</table>

**Note:**
- a: measurements for HCB are in kilograms not in kT as for other emissions.
- b: NH3 emissions from different agricultural sectors come from DEFRA (2002) and correspond to 44%, 9% and 14% of total UK emissions, respectively.
- Source: data provided by NETCEN and also used in Defra et al (2004).
Perhaps the most prominent concern, in relation to the impacts of agriculture on soil, arises from soil erosion. Soil erosion in the UK is caused by the action of water and wind, the former being more widespread. Soil may be lost entirely from a field or redeposited within it depending on topographic features. The coarse fraction of the soil, i.e. sand and stones, is transported only a short distance whereas the finer silt, clay and organic matter can be moved well away from the site. Rates of soil erosion vary greatly not only between soils but between years, being influenced by soil type, slope, topography, rainfall and farming practices. Some crops, including roots and maize, raise particular soil management issues.

On-farm, many of the impacts of soil erosion are as a consequence of particles becoming progressively coarser grained. The loss of fine grains results in reduced water storage capacity of the soil, loss of nutrients or fertility and deterioration of soil structure (Skinner, 1997). In the 2001 farm practices survey for England, farmers assessed the extent of soil erosion on their farms. In all 65% of farmers stated that they had experienced some indicator of soil erosion on their land - indicators included discoloured runoff entering ditches and water courses, sediment deposited in ditches and water courses, sediment deposited on roads and formation of gullies and rills. On-farm impacts of soil erosion occur over long time periods and are therefore difficult to assess, suggesting that these figures and others are generally underestimates of the actual situation. Another major issue in relation to the quality of agricultural soils is loss of organic matter. It is estimated that 18% of the soil organic carbon present in arable topsoils in 1980 had been lost by 1995 (Defra, 2004). This reduces the usefulness of soil to the farmer, but also its ability to perform other functions.

There are also off-farm impacts of soil erosion, which include damage to property and accidents (if deposited on other land or roads), flooding (if deposited in rivers, ditches and drains). In addition, there are also negative impacts on aquatic habitats as a consequence of increased turbidity and pollution of water as a result of soil erosion (much of the phosphorus and pesticide losses from farm land to surface water are bound to soil particles). Changes in the quality of water courses will also impact on recreational activities (e.g. fishing) and increase the cost of treating water for other uses.

Data on impacts

Data on soil presented in Annex 4 include indicators of soil improvements such as the area of land that receives organic manures, potential changes in soil pH from the application of lime, and indicators of losses in soil fertility. Data employed in the accounting framework in Section 5 are linking data that state that agriculture causes 95% of all soil erosion. This link is used to estimate the percent of total off-site soil erosion costs that are attributable to agriculture. These costs provide environmental accounting adjustments for productivity losses to other sectors from soil erosion. Any current productivity gains or losses to agriculture from changes in soil characteristics will already be incorporated within the existing sector accounts through gains or losses in incomes to farming. The potential effect of decline in soil quality on future yields is not included in the existing accounts and cannot yet be quantified here.

No data were found which would allow the welfare losses and gains from improvements or degradations to soil as a natural asset.

3.2.4 Landscape

Description of impacts

Agricultural land occupies most of the landscape, covering approximately 74% of the total land area in the UK - equating to 71 per cent in England, 77% in Wales, 76% in Scotland and 82% in Northern Ireland (Defra e-digest of environmental statistics online, 2003 (10)). There is scarcely any wilderness in the UK outside coastal areas and higher mountains, so the managed semi natural environment, dominated by agricultural use, is of particular importance. The management of this land historically has played a crucial role in the formation of the landscape, which is now familiar to us. Without agriculture the great majority of land would be occupied by woodland, some stretches of wetland and built structures. The practice of agriculture is associated with various forms of land management such as:

- arable land, much of which is subject to crop rotation and includes a proportion of set-aside land;
temporary grass, this is ploughed and reseeded from time to time, much of it is rather intensively managed;

permanent grassland and moorland (also known as rough grazing), particularly widespread in the hills and uplands and includes some common land;

more specialised land uses, including horticulture and orchards;

patches of trees and woodland, subject to varying degrees of active management by the farmer;

a range of features particularly associated with agriculture, such as ditches, hedges, walls, fences and other field boundaries, ponds and reservoirs; and

buildings and structures, varying from the traditional farmyard to contemporary grain silos and intensive livestock buildings.

Agriculture plays a key role in shaping the quality of the national ‘stock’ of landscape; indeed agricultural landscapes are the visible outcome of the interactions between agriculture, natural resources and the environment and encompass amenity, cultural and recreational values. Landscapes can be considered as composed of three key elements: (i) landscape structures, including appearance and environmental features (e.g. habitats), (ii) land use types (e.g. crops), and (iii) man-made structures (e.g. stone walls).

Perceptions of landscape are rooted in history and local, regional and national cultures and usually vary over time for the viewer and between different users of the landscape, i.e. farmers, environmentalists and urban dwellers.

In so far as the farmed landscape is valued by society, the activities that maintain it generate a positive environmental externality. The relevant ‘output’ is the difference between the landscape with and without (the counterfactual) agriculture. While without agriculture typically natural succession results in scrub and then woodland replacing open land, the alternative to agriculture depends on the local circumstances. In some locations the most likely alternative to agriculture would be woodland, in others it might be grazing for horses and other leisure activities or urban development. Maintaining the landscape implies both the continuity of broad landuse and the upkeep of certain elements within it. Maintenance requires active management, both to prevent neglect and to renew features that have decayed. To a degree the act of farming itself forms part of the landscape: the presence of farm animals and also machinery can be both striking and distinctive. Thus, farming maintains a cultural rather than a natural landscape.

Landscape change is continuous, although not always reported in detail. Usually it is measured through GIS and ground level surveys and shows evidence of both deterioration and improvement as a result of agricultural activity. Pressures on the landscape include the neglect of established landscape elements, removal of traditional features, field enlargement, new buildings, farm infrastructure such as irrigation, the intrusion of unfamiliar crops and new farm roads. While some features, such as hedges, are subject to removal they are also being supplemented by newly planted hedges and field boundaries. In ecological and in cultural terms older features tend to be of greater value but over time new features can win increasing acceptance.

Data about landscape features are not easy to collect. While some linear features, such as hedges, are relatively easily distinguished from other landscape features, even they can be difficult to survey and quantify as they generally occur as continuous networks, rather than discrete habitat patches and their character is highly variable. They can be found singly or in combination forming multi element features. For example, hedges may be comprised of either a single species of recent origin or ancient assemblage of high species diversity. Walls can equally vary from traditional dry stone through to new brick built and concrete walls.

Also within the data covered in the landscape category are impacts of farming on the UK’s cultural assets such as listed buildings and archaeological features. These form part of the landscape to some extent and are also a feature of agri-environment schemes. Rather than create a separate category, impacts on the historical, built environment are included within landscape even though these cannot be expressed in monetary terms.

Current agri-environmental schemes active in the UK include: Environmentally Sensitive Areas (ESAs) and Countryside Stewardship in England, Tir Gofal in Wales, the Rural Stewardship Scheme in Scotland, ESAs and the Countryside Management Scheme in Northern Ireland. Other schemes include assistance for organic farming.
The **ESA scheme** was originally a major component of agri-environmental programmes in all four countries of the UK. It was established by the Agriculture Act 1986 which empowered Ministers to designate environmentally sensitive areas and within them to enter into agreements with farmers to manage the land in return for certain payments. Areas can be designated by virtue of their landscape, wildlife or historical importance. In practice, the environmental interest of many of the areas selected has been vulnerable to changes in farming practice, such as the intensification of production. Each designating order identifies the boundary of the ESA, sets the objectives, itemises the management prescriptions which will apply to farmers joining the scheme and sets the rates of payment, which can be adjusted. There is considerable variation between ESAs with regard to such matters as precise objectives and management prescriptions. Farmers are offered a five-year agreement, which is entirely voluntary.

**Countryside Stewardship** was originally launched in 1991. Its purpose is the conservation and enhancement of key English landscapes, features and habitats outside ESAs. Those targeted by the schemes include uplands, chalk and limestone grassland, waterside land, lowland heath, coastland, old meadows and pastures, historic landscapes and field margins. Priority land types and features are agreed for each county or group of countries through an annual process of consultation at local and regional level. Agreements are discretionary and run for ten years.

In 1999, an integrated **Countryside Management Scheme** was launched in Northern Ireland. This broadly follows the Stewardship model and complements the Northern Irish ESAs. Also in 1999, a new all-Wales agri-environment scheme, **Tir Gofal**, was launched, combining the former Tir Cymen pilot scheme, and the Welsh ESAs, Moorland and Habitat Schemes into one comprehensive, menu-based scheme.

Following the Welsh example, in Scotland in 2000, the Scottish Executive launched the **Rural Stewardship Scheme** as a new all-Scotland agri-environment scheme, combining the former Countryside Premium Scheme and Scottish ESAs into a single menu-based framework. One notable feature of the Scottish scheme is special provisions for crofting.

One result of the relatively rapid development and consolidation of agri-environmental schemes in the UK has been the creation of a large number of schemes, which are now closed to new applicants but under which some farmers in each country will continue to have agreements until their terms end. These include: Nitrate Sensitive Areas (Great Britain only); the Habitat Scheme (Great Britain only); the Moorland Scheme; Tir Cymen (Wales); and the Countryside Access Scheme.

**Data on impacts**

Data on the agricultural landscapes are summarised in Annex 4. The data reflect agriculture’s contribution to the maintenance of landscape features as well as loss or degradation of landscape features, quality or character. Detailed data on the length of linear landscape features were last collected in 1998 covering dry-stone walls, hedgerows and so on. On a more general level, an indicator of agricultural landscape quality is provided by the types of farming taking place on agricultural land. This is also relevant to the economic valuation of landscape, as the public may enjoy the aesthetics of certain activities such as sheep farming more than others such as intensively farmed mono-crops. The Annex presents data on the coverage of the agri-environmental schemes which support agricultural extensification, landscape and habitat maintenance since this is an important response activity and some valuation data are available. Finally, general statistics on type of farming are also presented. Data on the negative aspects of the agricultural landscape, such as a stock of or growth in ‘ugly buildings’, have not been found, and some of the data provide an incomplete picture (e.g. on degradation of archaeological features). Grey shaded cells in the Annex tables indicate that no data have yet been found on that impact type.

There is some overlap between landscape and the habitats and species categories, both in terms of the collection of physical data and the application of economic data. In calculating monetary estimates, care has been taken not to double count these categories. As mentioned above, the landscape category also includes the impacts of farming on the UK’s cultural assets such as listed buildings and archaeological features in the agricultural area.
There are two different types of landscape datasets, which could potentially be used for environmental accounting for agriculture: agriculture land use and habitat based. Both sets of data are presented in Annex 4.

The agricultural land use datasets are mainly based on the Agricultural Census completed in June each year by DEFRA, DARNI, the Scottish Executive and the National Assembly for Wales. These data sets are based on a survey of farmers who will fill in the area in use under each category, i.e. the area of different arable crops, the area of set aside, and the area grazed. This makes them a purely land use based sample (e.g. the area of land used for grazing rather than the area of grassland grazed). Thus, use of these data in environmental accounts presents difficulty as there is little matching data from the economic valuation literature, which tends to focus on particular habitats or features of land. For example, there is no identification of exactly what habitats fall within, say, ‘rough grazing’.

A significant amount of the rough grazing category will be heather moorland in a biological sense but this is one of the few ‘habitats’ in the farm data which correspond broadly to some of the Biodiversity Action Plans (BAP) and broad habitat classification categories (see below). For other categories it is less clear which habitats are represented. Breaking down these agricultural figures further to identify the exact habitats each of the categories may correspond to, is not possible as the necessary data are not being collected as part of the Agricultural Census. Using the census data in the accounts also causes potential difficulties in relation to double counting as it is difficult to identify the exact extent of the overlaps between the census data categories. For this reason the census results, while presented in the Annex, are not used to calculate environmental accounting adjustments.

Recent habitat based data is mainly from the Countryside Survey 2000 (Haines-Young et al, 2000) and the UK BAP, the use of which as reliable datasets has been confirmed with English Nature. The first of these describes broad habitat types in Great Britain. This data set is solely based on the biological make up of the land cover rather than the land use, i.e. it measures different types of grassland habitat rather than the grazed area. The second data set provides detailed information about specific habitats. These represent smaller areas than the broad stock habitats, based on very specific species assemblages.

In many cases these habitats are used for agricultural purposes, predominantly grazing. However, there is no generalised data concerning exactly what proportion of these habitats is used for or managed by agriculture. In some very specific instances information on individual habitat types within the BAP series includes areas under agri-environment schemes. However, this is not a full representation of the total extent of agriculture occurring on this habitat type, as these schemes are voluntary and therefore do not include all the land in a particular class.

Considerable efforts were made to further link habitats to agricultural use but it was not possible to identify a reliable and complete data set. Thus, in the absence of this information, it was assumed that those broad habitat types listed in Table 3.5 below (from the Countryside 2000 survey) are agricultural – or under agricultural management. The total land coverage represented by these habitat types is just under 74% of total UK land coverage and is, therefore, considered to adequately represent land under agricultural management. The advantages of using these data to value landscape benefits is that (i) it provides a reasonably good match to the economic data and that (ii) it can also be used to calculate habitat benefits in England through SSSI data which uses the same classification system – thereby avoiding double counting between landscape and habitat categories. The disadvantages of the data are that the extensive work involved in producing these data sets for specific habitats means that these are not regularly updated, the latest data set being already 6 years old. Consequently it would be difficult to alter these figures accurately for yearly accounts.

Data on landscape features are presented in Table 3.6 and come from the Countryside Survey also. These data can be matched with economic data on landscape features to provide a separate environmental adjustment to the account. Grey areas show gaps in data.
### Table 3.5: Data on Agricultural Landscapes Associated with Land Types

<table>
<thead>
<tr>
<th>Resource</th>
<th>Data description</th>
<th>Data (ha)</th>
<th>Source [ref] (year of data)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Improved Grassland</td>
<td></td>
<td>4,431,000</td>
<td>1,051,000</td>
</tr>
<tr>
<td>Arable and Horticultural</td>
<td></td>
<td>4,609,000</td>
<td>639,000</td>
</tr>
<tr>
<td>Neutral Grassland</td>
<td></td>
<td>461,600</td>
<td>168,000</td>
</tr>
<tr>
<td>Bog</td>
<td></td>
<td>186,000</td>
<td>2,038,000</td>
</tr>
<tr>
<td>Dwarf Shrub Heath</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Acid Grassland</td>
<td></td>
<td>563,900</td>
<td>748,000</td>
</tr>
<tr>
<td>Fen, Marsh and Swamp</td>
<td></td>
<td>240,000</td>
<td>337,000</td>
</tr>
<tr>
<td>Calcareous Grassland</td>
<td></td>
<td>54,900</td>
<td>27,000</td>
</tr>
</tbody>
</table>

### Table 3.6: Stock of Linear landscape features or field boundaries (km)

<table>
<thead>
<tr>
<th>Resource</th>
<th>DPSIR</th>
<th>England and Wales</th>
<th>Scotland</th>
<th>N.Ireland</th>
<th>Source (year of data)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hedge</td>
<td>S</td>
<td>449,300</td>
<td></td>
<td>19,000</td>
<td>233,000</td>
</tr>
<tr>
<td>Remnant hedge</td>
<td>S</td>
<td>52,300</td>
<td></td>
<td>5,300</td>
<td></td>
</tr>
<tr>
<td>Wall</td>
<td>S</td>
<td>105,800</td>
<td></td>
<td>87,100</td>
<td></td>
</tr>
<tr>
<td>Dry Stone Walls</td>
<td>S</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ruined dry stone walls</td>
<td>S</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mortared Walls</td>
<td>S</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Line of trees/shrubs/relict hedge and fence</td>
<td>S</td>
<td>70,000</td>
<td></td>
<td>11,100</td>
<td></td>
</tr>
<tr>
<td>Line of tress/shrubs/relict hedge</td>
<td>S</td>
<td>83,400</td>
<td></td>
<td>13,300</td>
<td></td>
</tr>
<tr>
<td>Bank/grass strip</td>
<td>S</td>
<td>70,000</td>
<td></td>
<td>12,400</td>
<td></td>
</tr>
<tr>
<td>Earth banks</td>
<td>S</td>
<td></td>
<td></td>
<td></td>
<td>41,000</td>
</tr>
<tr>
<td>Fence</td>
<td>S</td>
<td>432,200</td>
<td></td>
<td>233,700</td>
<td>55,000</td>
</tr>
</tbody>
</table>

### 3.2.5 Habitats and species

**Description of impacts**

Agriculture’s significance as a land use and its influence on the non-farmed environment gives it a particularly strong link to biodiversity. Agricultural practices interact with natural factors such as soil type, climatic conditions and existing populations of flora and fauna to create semi natural habitats. The quality of these habitats and their suitability for natural flora and fauna depends on the management of the land use, of inputs and other factors such as recreational activities.

The impacts of agriculture on habitats and species, hence biodiversity, are complex and diverse. This is partly because biodiversity encompasses such a vast array of different elements, i.e. genetic variability within species, the variety of species and the variety of ecosystems. In the face of such complexity and in order to make the impact assessment within this study manageable, changes in habitats and species are taken as proxy...
indicators of changes in biodiversity. A narrower focus on habitats and species is more manageable but still ambitious.

In addition to being of value on their own, biological resources and their variety, such as the case for soil microbes, also impact on the productivity of agriculture (Defra, 2004). This is because some flora and fauna provide ecosystem services. These useful communities are affected by the inputs from agriculture in a similar way to other species, giving a further dimension to the relationship.

**Change in habitats and species**

Agricultural activities can both augment and erode the integrity of habitats and species. **Biodiversity loss** is generally as a result of the conversion of natural or semi natural habitats to agricultural land or use of inappropriate farm management practices. These can result in rapid stock changes in species or habitat and long term decline. In the UK it is now the latter process that is more relevant. Triggers for this process include the input of nutrients, leading to enrichment of soil and waters; change in water availability due to abstraction; changes in overland flow patterns; cropping and mowing practices; over and under grazing; introduction of pesticides and other pollutants; change in soil pH (e.g. acidification due to ammonia deposition or fertiliser use, and reduction of pH due to liming); reduction in target food species or groups of species (e.g. due to targeting of invertebrates using pesticides); neglect of traditional management; habitat fragmentation; increased disturbance; introduction of an invasive non native species; and loss of soil due to erosion.

The magnitude of the influence of agriculture on biodiversity in the UK is illustrated by the **Biodiversity Action Plans** prepared for both habitats and species. Of the 391 Species Action Plans (SAPs), 174 (44%) made one or more references to agriculture as a threat and an additional 89 (23%) identified it as an area where action should be taken to address concerns. This coverage of agriculture in a total of 263 plans (67%) is greater than that for any other economic sector. This trend is repeated when the same criteria in the Habitat Action Plans (HAPs) are considered, with 29 out of 45 (64%) highlighting agriculture as an issue associated with threat to habitats. This compares with only 7 HAPs and 62 SAPs that are exclusively associated with agricultural land; underlying the impacts of agriculture beyond farm boundaries.

Between taxonomic groups the influence of agriculture varies with agriculture listed as important in 92% of SAPs for ants, bees and wasps, 88% for birds and 87% for butterflies and moths. In total, agriculture is listed in over 50% of all taxonomic groups. The most frequent agriculture related problems mentioned in the action plans are associated with inputs of fertiliser or resultant eutrophication, herbicides and pesticides (95 plans) and over/under or inappropriate grazing (36 plans) (Countryside Agency, 2000).

Broader environmental changes associated with agriculture can affect more widespread as well as rarer species and habitats. For example, nutrient enrichment can occur in water and soils leading to eutrophication and loss of aquatic life (see Section 3.2.1) or can change the balance between species. The addition of nutrients means that the conditions for plant growth are changed and a limiting factor to growth, i.e. lack of nutrients, is removed. Typically enrichment results in an increase in fast growing more competitive species, which then squeeze out others. This results in a more uniform plant community and has been observed to be happening on unimproved grassland, one of the most diverse habitats in Britain. Countryside 2000 shows a significant loss of plant diversity in less improved grasslands and a general increase in nutrients since 1990. A study in 1999 looking at vascular plants showed that on cropland there were on average six different species within a 200m$^2$ random plot, while for fertile grassland there were 11 and for unimproved grassland there were 19 (Haines-Young et al, 2000).

One of the best-known examples of agriculture affecting biodiversity is the decline of farmland birds. Now an official UK Government indicator for sustainable development, the number of farmland birds has decreased by 40% since the mid-1970s (Gregory et al, 2003). Reasons for this decline are diverse but two of the main factors include loss of food sources and reduction in habitat quality including loss of nest sites and change in the mosaic of habitats.
Maintenance of Habitats and Species

Agriculture can contribute to the maintenance of biodiversity through appropriate management, within or outside formal production based activities. For example, routine grazing can benefit biodiversity by maintaining a sward in which species of conservation concern can survive while more competitive species are controlled, preventing them from dominating the grassland habitat.

The majority of the UK is covered by semi-natural habitats and the importance of agriculture in relation to the creation and maintenance of these valued resources should not be understated. Many semi-natural habitats were created by traditional extensive farming methods and their maintenance usually requires the continuation of agriculture. Semi-natural grassland is the most widespread of habitats although wetlands, moorland, wood pasture, salt marshes and other habitats are also affected. As mentioned above, the semi-natural habitats, arising from historical agricultural practices often have a higher biodiversity value than the type of habitat that would result if management were removed - normally scrub followed by woodland. However, both scrub and woodland are of greater biodiversity value than most land intensively managed for agriculture and in many cases the most appropriate scenarios would be a mosaic of habitats. Therefore, there are considerable areas where the right form of agriculture maintains valued habitats, hence biodiversity, by preventing the natural progression towards less diverse habitats.

As mentioned in Section 3.2.4, there have been considerable efforts in the UK to encourage farmers to manage the land for wildlife by adhering to a variety of measures affecting both the principal farmed area and the mosaic of accompanying habitats, including boundary features such as field margins and hedgerows. Improved education and information, assurance schemes and private initiatives by farmers and food-sellers have been launched alongside incentive schemes. Voluntary agri-environmental schemes include Countryside Stewardship, Environmentally Sensitive Areas, the Countryside Premium Scheme, Tir Cymen and Tir Gofal. An example of a success, as a consequence of such a scheme, is the Corncrake. This globally threatened species hit a low of just 479 calling males in 1993 as a result of loss of vegetation cover and grassland management. The Corncrake initiative set up to help farmers and crofters manage the land for the species has resulted in an increase to 622 calling males in 2000 - although the future of the species still remains delicately balanced (Haines-Young et al, 2000).

In addition to agriculture's impacts on wild species, there is also the issue of maintenance of crop and livestock diversity. Within the farming sector there is a huge variation within species, the great array of different crop varieties for instance, or the range of cattle breeds. Often these varieties and breeds are particular to certain locations or environments, having developed particular characteristics that make them well adapted to local conditions and climates. Over the last century there has been a trend away from using this variety (in tandem with increasing agricultural productivity) in favour of a small number of breeds that perform more uniformly (SEERAD, 2002). However, there are now efforts to halt the decline. For example, the Rare Breeds Survival Trust lists 14 breeds of native cattle whose numbers have seriously declined over the last century, but whose decline has now been stabilised (RBST, 2002).

The maintenance of traditional breeds is important for the diversity of farmed species, and to some extent for the maintenance of biodiversity more generally. As mentioned above certain livestock breeds have evolved based on environmental conditions in certain areas. This means that they are of great use for the grazing of conservation areas or the maintenance of particular priority habitats. There is no economic data on the value of these breeds nor any analysis of economic benefits that has been undertaken, therefore, accounting adjustments for traditional breeds cannot be presented in the report.

Data on Impacts

Establishing the impacts of agriculture on habitats and species requires an understanding of the environmental pathways of agricultural pollution and the effects of land-use change due to agriculture. As a result, there is little robust data in this category and a number of data gaps are shown in Annex 4. What is reported is known declines in farmland bird species thought to be associated with agriculture (also reported in AUK: Defra et al, 2003) (Table 3.7), and data on agriculture’s impact on SSSIs (Table 3.8).
Table 3.7: Data on Farmland Birds

<table>
<thead>
<tr>
<th>Resource</th>
<th>DPSIR</th>
<th>England and Wales</th>
<th>Scotland</th>
<th>N.Ireland</th>
<th>Year of data</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>UK farmland bird population index, total of 19 species</td>
<td>S</td>
<td>Base year for index is population of birds in 1970 = 100%</td>
<td>Total farmland species 2002 = 58%</td>
<td>2002</td>
<td>Defra/RSPB/BTO Wild Bird Indicator (Gregory et al, 2003)</td>
<td></td>
</tr>
</tbody>
</table>

Data presented in Table 3.8 on certain protected habitat types are clearly also relevant to landscape. This overlap is addressed further in Sections 4 and 5, when the task of aggregating economic and environmental data is explained and presented. In addition, data from English Nature on the state of SSSIs in England (English Nature, 2003) provides a valuable additional source of information on habitats of particular conservation importance. Assessments of the number of hectares of habitat (using the broad habitat classification system) in favourable or unfavourable condition due to particular agricultural practices (or the lack thereof) are presented in Table 3.8. This information was gleaned from the information presented in the report and in consultation with English Nature staff.

Table 3.8: SSSIs and Agriculture in England

<table>
<thead>
<tr>
<th>Resource</th>
<th>Data description</th>
<th>DPSIR</th>
<th>Data (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>SSSI</td>
<td>SSSI - favourable condition</td>
<td>SSSI - unfavourable condition</td>
<td></td>
</tr>
<tr>
<td>Improved Grassland</td>
<td>$</td>
<td>none</td>
<td>none</td>
</tr>
<tr>
<td>Arable and Horticultural</td>
<td>$</td>
<td>13,818</td>
<td>282</td>
</tr>
<tr>
<td>Neutral Grassland</td>
<td>$</td>
<td>36,351</td>
<td>14,249</td>
</tr>
<tr>
<td>Bog</td>
<td>$</td>
<td>86,304</td>
<td>99,696</td>
</tr>
<tr>
<td>Dwarf Shrub Heath</td>
<td>$</td>
<td>86,330</td>
<td>125,569</td>
</tr>
<tr>
<td>Acid Grassland</td>
<td>$</td>
<td>25,307</td>
<td>16,593</td>
</tr>
<tr>
<td>Fen, Marsh and Swamp</td>
<td>$</td>
<td>24,750</td>
<td>5,250</td>
</tr>
<tr>
<td>Calcareous Grassland</td>
<td>$</td>
<td>41,000</td>
<td>13,900</td>
</tr>
</tbody>
</table>

3.2.6 Waste

**Description of Impacts**

Waste is defined as ‘any substance or object….which the producer or the person in possession of it discards or intends or is required to discard’ (Environment Agency, 2001). Like any other production based industry by-products result from agricultural processes and some require disposal, whether to landfill or incineration, or can be used as a raw material in another process. Agricultural waste, i.e. waste produced on premises used for agriculture, has traditionally been excluded from waste legislation. However, in 2004 this will change as a consequence of the EU Waste Framework Directive now being applied to this waste stream. This will have implications for the way waste is managed by farmers, including changes to the transfer of farm waste, the need to use licensed contractors and application of the duty of care for waste (Environment Agency, 2003).

**Generation of waste**

Agriculture generates a considerable array of waste products ranging from typical waste materials such as packaging to agriculture specific wastes such as pesticide washings. The wastes produced in the greatest quantities by agriculture are organic, for example, silage effluent, dirty water, milk, blood, vegetable washings, animal carcasses and crop waste cuttings, with agriculture producing an estimated 90% of all organic waste (Environment Agency, 2002). In addition to organic wastes, there are also inorganic wastes amounting to approx 0.5 million tonnes per year for England and Wales (Environment Agency, 2001). These
include agrochemical washings and concentrates, and wastes relating to animal health waste such as unused medicines, syringes and building wastes.

As with the majority of other agricultural issues the impacts of waste depend on the farming practice employed (e.g. slurry will only arise when animals are housed rather than grazed). The generation of waste is only one aspect; the practices followed on farms after generation, i.e. storage and disposal, also affect the degree of waste impacts. In the case of storage, the quality and management of facilities greatly affect the impact of the stored waste. For example, inappropriate storage can lead to water pollution as a consequence of spillage of agrochemicals or slurry, gaseous emissions or nuisance in the form of odour or an eyesore. These problems can lead to a cascade of other environmental impacts such as decline in biodiversity, reduced air and water quality. Storage of unwanted material can also result in the reduction in the perceived quality of the landscape - over 70% of farmers in Great Britain are storing some wastes on their farm with no intent to dispose of it (e.g. scrap metal, old machinery and tyres) (Environment Agency, 2003).

**Disposal and use of waste**

Issues relating to waste disposal and agriculture can be divided into two aspects, the first is disposal and reuse of waste produced by farms and the second is disposal of wastes from other sectors on farms.

The issue of agricultural waste disposal is complex especially in light of imminent changes to legislation. Currently farmers use a variety of recovery and disposal methods including reuse on farms, take back by suppliers, inclusion with household waste, stockpiling, burial and burning. However, 90% of farmers are currently disposing of some wastes using practices that may not remain legal once the new legislation comes into force (Environment Agency, 2003).

A large proportion of waste arising from agriculture is organic, and this is largely reused as a raw material in other processes (see Section 3.2.8). There are also by-products of agriculture that cannot be reused and must be disposed of responsibly. Out of this, some wastes will be incinerated (e.g. animal carcasses), while others may be landfilled. The presence of incinerators and landfills are controversial and generally negatively received by local residents. In addition, they both have associated environmental impacts such as emissions to air including greenhouse gases, and, in the case of landfilling, the use of considerable areas of land.

Farmers may also bring waste products from other activities on to agricultural land as a raw material. Some food wastes are spread on farmland as a disposal route. One of the major examples of this is the use of sewage sludge to enrich farmland soils. This is an effective way of using this waste, meaning it no longer requires incinerating etc. However, there are also impacts from use. Sludge may have high levels of heavy metal containments and high nutrient content, which contributes to enrichment of soils and water supplies.

**Data on Impacts**

The waste impact category is not an environmental resource or service, but rather a pressure created by agriculture on the environment. Waste outputs from agriculture will impact air, water, land quality and possibly on habitats and species as well. Waste has been designated as a separate category partly because of the way data are presented and the way this potentially links up with economic valuation data, which are often available in terms of tonne of waste output.

Annex 4 provides an overview of data on waste and presents gaps as well as existing data. The recent ‘Agricultural Waste Survey 2003’ produced by the Environment Agency (Environment Agency, 2003) provides detailed breakdowns of different waste streams and waste management practices in England and Wales. It is only possible to employ data on general waste from agriculture, in line with what could reasonably be used in the accounts and applied to economic data. Agricultural Accounts in the United Kingdom (Defra et al, 2004) presents data on general waste arisings, which were 1 million tonnes in 1999.
3.2.7 Nuisance

Description of impacts

Odour

Some agricultural operations result in odour either occasionally or on a more frequent basis. While the main sources are intensive livestock farms, particularly poultry and pig units, it may also arise from manure heaps, spreading of manure and sewage sludge on land and from their storage. The type of housing, management and equipment installed will influence the extent of odour. The degree of nuisance depends on the proximity of properties, topography, prevailing wind direction and extent of outdoor recreation as well as the level of odour itself. It is not possible to quantify (in physical or monetary terms) the impact on nearby properties and recreational activities based on available information. However, in 1989-1990 there were 3700 complaints about odour from farms in the UK, 40% of which were considered justifiable (Skinner et al, 1997). Although even more difficult to quantify, it is also necessary to mention the positive externalities generated by agriculture in the form of pleasant smells: freshly cut hay would be an example.

Noise

A variety of agricultural activities give rise to noise. In some circumstances this impinges on those living in or otherwise using neighbouring properties as well as those employed on farms or visiting them. There is noise associated with mobile machinery, such as tractors, combine harvesters and chain saws, all of which may be operated for significant periods of time. Some stationary equipment, such as feed mills on livestock farms, can also be a source of fairly continuous noise.

Nearly all livestock are responsible for some noise but usually this is only intrusive or potentially classified as a nuisance where large numbers of animals are kept together. Some intensive livestock units are particularly noisy and intrusive for the neighbouring community. There is some survey data indicating that agriculture is seen as a significant source of noise in parts of England but only a small percentage of people interviewed appear to find it annoying, see results below. It’s also worth noting that agriculture can contribute to the peacefulness of the countryside, although quantification of this effect is very difficult.

Data on impacts

The available data on noise impacts come from the Defra on-line e-digest of environmental statistics. The data result from a survey of the UK population and seem to suggest that while 11-32% of the population reported hearing noise from farming and agriculture, only 1-7% of them actually find it moderately annoying - and none find it very or extremely annoying. This suggests that the noise impacts of agriculture are slight and are considered for the remainder of the report to be negligible.

Neither the data on noise or odour could be linked with economic data and thus they are not presented in the final accounting adjustments.

3.2.8 Resource use

Agriculture is responsible both for the use of non renewable resources and the production and use of renewable resources. With regard to non-renewable resources, like other sectors of the economy, agriculture uses considerable amounts of fossil fuels. These are used by both mobile and stationary machinery associated with both field operations, such as ploughing, and more stationary processes such as crop drying. The annual consumption of fossil fuels in UK agriculture is 1.5 million litres of oil equivalent (National Statistics, 2003). Agriculture also uses a considerable quantity of electricity predominantly produced from fossil fuels. There are also agriculture specific issues of resource consumption, with questions over how far other assets, such as phosphates and soils, are renewable.
Agriculture is responsible for the creation of non food and fibre resources, in addition to those intended, i.e. food crops and livestock. Traditionally, organic bi-products emanating from farming have been used as raw materials for other activities (e.g. manure and slurry are used to fertilise land while straw is used as animal bedding). There are also cases of manure being used in energy production (e.g. there are instances of poultry manure being incinerated to produce energy in the UK).

In addition to resources that are essentially bi-products of the farming process agriculture is responsible for the production of a small but potentially growing volume of energy sources, consisting of both biofuels and by products such as straw which have energy value. While these products have a market value, they also make a contribution to the wider role of reducing reliance on non renewable resources and environmental pollution associated with their use such as emissions to air.
4 Economic Value of Environmental Impacts

This Section starts with a brief overview of the background of total economic value and economic valuation. Discussion and issues concerning the use of benefits transfer for providing inputs into the environmental accounts for agriculture are provided in Section 4.2. A review of economic valuation studies and further details of the studies and estimates selected for use in this study are presented in Section 4.3. Further detail of all studies reviewed can be found in Annex 5.

4.1 Economic Valuation

Section 1.4 provided an overview of the concepts of economic value and economic valuation. This section repeats some of this introductory material, providing further detail particularly on the valuation technique employed in this study - benefits transfer.

4.1.1 Total Economic Value

There are a number of motivations behind people's preferences for environmental assets and the services these resources provide. They can be grouped as those that are related to the actual or future use of natural assets and their services (known as use values) and those that are not related to any use (passive use or non-use value). These categories of motivations or preferences or value are collectively known as total economic value (TEV). Table 4.1 maps out total economic value using examples of different resources.

Table 4.1: Total Economic Value of a Resource

<table>
<thead>
<tr>
<th>Use values</th>
<th>Non-use values</th>
</tr>
</thead>
<tbody>
<tr>
<td>Direct Use Values</td>
<td>Indirect Use Values</td>
</tr>
<tr>
<td>Products for direct use or consumption, including commercial uses, e.g.:</td>
<td>Ecosystem services, e.g.:</td>
</tr>
<tr>
<td>o recreation</td>
<td>o flood control</td>
</tr>
<tr>
<td>o fish / meat</td>
<td>o storm protection</td>
</tr>
<tr>
<td>o paper / wood</td>
<td>o carbon sequestration</td>
</tr>
<tr>
<td>o water abstraction</td>
<td>o habitat provision</td>
</tr>
</tbody>
</table>

More specifically, direct use values are derived from the actual use of a resource by an individual in either a consumptive way (e.g. harvesting forest timber and abstraction of water for drinking or commercial use) or a non-consumptive way (e.g. bird watching and trekking). Indirect use values arise where society benefits from ecosystem functions (e.g. watershed protection or carbon sequestration by forests) whilst option values exist where individuals are willing to pay for the option of using a resource in the future (e.g. future visits to a wilderness area, or possible future pharmaceutical uses of biological resources).

Non-use values are typically classified in terms of existence values, which reflect willingness to pay to keep a resource in existence even when the individual expressing the value has no actual or planned use for his/herself or for anyone else; altruistic values, which might arise when the individual is concerned that the resource in question should be available to others in the current generation; and bequest values, which reflect concerns that the next and future generations should have the option to make use of the resource.

While some use values might be captured in markets (e.g. the market for timber and fish), many environmental goods and services are external to markets or what economists call ‘non-market’, i.e. they are provided free of charge.
4.1.2 Economic Valuation

The economic value of an environmental impact is estimated using data on individuals’ (or households’) preferences for that impact. Preferences, in turn, are measured by people’s Willingness to Pay (WTP) to maintain the current quality and quantity of environmental assets, to avoid a negative environmental impact or to secure a positive one. Similarly, preferences can be measured by people’s Willingness to Accept Compensation (WTA) to tolerate a negative environmental impact or to forgo a positive one. Most of the studies relevant here use the WTP measure.

The first source of economic data on people’s preferences is the market data for those environmental impacts that are traded or reflected in actual markets. These ‘marketed’ services of the environment include supply of drinking water, formal recreation or tourism and so on. In fact such market data (costs and prices) are akin to WTP or WTA of individuals in that market prices reflect the WTP of buyers and WTA of sellers. Generally an individual will only consume a good or a service when its price is less than or equal to his/her WTP. That is, a given good is purchased if they perceive that the benefit yielded by its consumption (as measured by their maximum WTP for that unit of consumption) exceeds the cost of consumption (as measured by the price of the unit of consumption). When the price is less than WTP, consumers receive a surplus of benefit. This surplus is known as ‘consumer surplus’, which is equal to WTP minus market price. Therefore, in most cases, the market price paid for a good is only a lower-bound estimate of an individual’s WTP for a good.

The two main approaches to quantifying the non-market values of TEV: revealed preference techniques and stated preference techniques, measure economic welfare and produce economic values as opposed to market prices\(^\text{15}\). These techniques aim to measure individuals’ preferences directly and capture consumer surplus as well as market price based expenditure when the latter exists. Revealed preference techniques use existing markets as surrogates for estimating the economic values for the environment. For example housing market data are used to estimate the value to households’ of cleaner air and reductions in traffic noise, by holding all other factors constant and observing changes in property value. Stated preference techniques create hypothetical, or simulated, markets by way of surveys to elicit society’s valuations for environmental changes. For instance, an example might be asking households to state their willingness to pay increased water bills for improvements in local river water quality.

Since it is not feasible to undertake an original economic valuation exercise for every policy question and project that arises in relation to non-market goods, a procedure called benefits transfer has been developed to enable appropriate, transparent and consistent use of economic value estimates from the literature. Benefits transfer is the approach employed in this study and essentially involves borrowing estimates of non-market values from previous studies and applying them to a new, but similar, context. Further detail about how benefits transfer is used in this study is given in Section 4.2.

4.1.3 Market versus Non-market Economic Data

Prior to discussing in more detail the issues surrounding the use of benefits transfer in this report, it is important to further highlight the distinction between market and non-market economic value estimates. Section 2 addressed the question as to which market price data should be included in the accounting framework and which should not be. To supplement this discussion, Table 4.2 below presents a summary of economic value estimates that fall under the market and non-market category headings.

As introduced in Table 2.4, and indicated by Table 4.2, some environmental impacts of agriculture have economic impacts that can be measured through market prices. For example the costs to water companies of removing pesticides from water or the cost of restocking fish population that declines due to pollution incidents. In addition, as per discussion in Section 2 some impacts of agriculture affect the productivity of the agricultural sector itself, such as is the case with on-site soil erosion which may reduce current crop yields. If such an impact takes place, the reduction in yield is already captured by the existing accounts since the crop revenue (yield multiplied by the market price for the crop) would be less than it would have been if there was no soil erosion. Some elements of market cost data are also included in the costs of air pollution, through costs to property and to the health service of air pollution damage on buildings and human health.

\(^{15}\) In economic terminology ‘price’ is not necessarily equal to ‘value’. Valuation techniques seek to provide recognised economic measures of value such as consumer surplus.
As already stated, market costs do not provide an estimate of the true welfare losses and gains arising from changes in the quantity and quality of environmental assets. Use of market prices can capture some elements of total economic value such as direct and indirect use values. However, aspects of welfare losses and gains arising from non-use and option values can only be captured through stated preference techniques. Non-market data are employed in this study for the majority of environmental impacts that affect the welfare of society (e.g. water and air pollution, landscape and habitat provision), while market economic data are employed to value losses/gains in productivity to other sectors resulting from agriculture’s impact on the environment.

### Table 4.2: Market and Non-Market Economic Data for Environmental Impacts

<table>
<thead>
<tr>
<th>MARKET ECONOMIC VALUES</th>
<th>NON-MARKET ECONOMIC VALUES</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Costs to water companies of removing pesticides from water</td>
<td>• Welfare losses from degradation of environmental assets</td>
</tr>
<tr>
<td>• Monitoring costs (e.g. to the Environment Agency) [not used in this study]</td>
<td>• Welfare losses from damages to human health</td>
</tr>
<tr>
<td>• NHS costs for treating human health impacts [included in the costs of air pollution]</td>
<td>• Welfare losses from degradation of animal and plant species</td>
</tr>
<tr>
<td>• Costs of restocking fish stocks lost due to pollution incidents</td>
<td>• Welfare gains from environmental services provided by the current stock of environmental assets [included in this study for land-based assets under agricultural control]</td>
</tr>
</tbody>
</table>

### 4.2 The Use of Benefits Transfer for Environmental Accounts for Agriculture

In implementing benefits transfer, the three most common procedures are to (i) transfer an average WTP estimate from an original valuation study, (ii) transfer WTP estimates from meta-analyses\(^{16}\), or (iii) transfer a WTP function. If there is an a priori reason to expect a difference between the original study information and the new policy context; WTP values (and function coefficients) may be adjusted to reflect this expectation.

For a benefit transfer exercise to be ‘valid’ certain conditions should be met (Boyle and Bergstrom, 1992; Desvousges et al, 1992). These conditions are widely recognised to be the following:

- the studies included in the analysis must be ‘sound’;
- the studies should include a WTP bid function (regression analysis showing the influence of explanatory variables on WTP);
- the ‘study site’ (the location of the original study) and ‘policy site’ (the new location) must be sufficiently similar in terms of population and site characteristics, or differences in characteristics must be adjusted for;
- the change in the environment asset and its services being valued at two sites should be similar; and
- property rights should be the same across the sites.

These conditions may also serve as ‘health warnings’ which recognise the limitation of the benefits transfer approach. Specifically, individuals at two different locations may value environmental goods and services differently due to differences in: (i) socio economic characteristics; (ii) the physical characteristics of a particular site; (iii) the proposed changes in the environmental goods at each site; and (iv) the availability of substitutes for a particular site (Bateman et al, 2000). Moreover, the opportunities to experience the environmental benefit between two sets of individuals may be different (Bergland et al, 1995) and so may the underlying preferences of individuals. Hence WTP in the original study may not provide an accurate estimate of WTP in the new study context.

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\(^{16}\) Meta analysis is typically defined as the statistical analysis of the results of a number of empirical studies. It is an attempt to derive valid generalisations by identifying consistency of results across different studies (Brouwer, 2000).
Whilst recognising these limitations, and in particular the potential for a large margin of error, it is generally accepted that the benefits transfer technique provides useful advice in terms of the likely magnitude of environmental costs and benefits and is endorsed for policy analysis by the HM Treasury ‘Green Book’ (HM Treasury, 2003). Moreover, without the use of benefits transfer, it would be very difficult (if not impossible) to give a monetary account of both the positive and negative non-market aspects of UK agriculture.

For the purposes of environmental accounts for agriculture what is required are essentially estimates of the current value of environmental services provided by or affected positively or negatively by agriculture. The literature review undertaken for this report (Section 4.3 and Annex 5) seeks to identify estimated values from existing economic valuation studies that can be used as proxies for the various categories of impact identified in the accounting methodology. This approach implicitly assumes that the preferences of the average individual captured in the original study are a suitable estimate of the preferences of the average individual in the context of this current study.

This study employs two benefits transfer approaches, mean WTP transfer and WTP function transfer (for landscape and habitats impacts). WTP values typically need to be adjusted into current monetary values (using a price inflator) and also converted to appropriate units (e.g. transforming WTP per household per year for a stated environmental change into WTP per hectare per year, WTP per km of river per year, per bird species per year and so on). All adjustments to the data that were undertaken are detailed in this Section and also in Annex 6.

The review of economic literature is also useful in identifying where gaps in the analysis exist (e.g. where there is no economic research on a particular impact or where economic research results do not match physical data). There are also a large number of studies that cover the same or similar environmental impacts as caused by agriculture but are not suitable for use in the context of the accounts for a number of reasons. The studies that are ultimately used are the ones that pass all or a sufficient number of the following selection criteria:

1. **Study subject** - the focus of the review are studies that consider either positive or negative impacts of agricultural activity, or alternatively, assess impacts generated by other sectors that are similar to those generated by agriculture. For instance, emissions of methane from agriculture can be valued using studies that assess the impacts of methane from any other source since the source of emissions does not alter the damage assessment.

2. **Study context** - it is important to match the valuation context and the changes in provision of the quantity or quality of a good in an original study to the new valuation context. This refers to aspects of the context of valuation which could be (i) maintaining the status quo, (ii) avoiding a decrease in quantity/quality, and (iii) attaining an improvement in quantity/quality. In the latter two contexts, the nature of the alternative will also be an important factor in determining the magnitude of WTP (e.g. the implied value of a hectare of moorland will be greater if WTP was elicited as avoiding conversion to housing development rather than avoiding conversion to a degraded state). In most cases it will not be possible to find an exact match between contexts in which case a judgement needs to be made about the transferability of results to an accounting context, where ideally WTP contexts are couched in terms of maintaining the status quo (thereby providing a ‘snap shot’ of economic value provided in the current period). In some instances in this study, avoiding a decrease and attaining an improvement are used interchangeably with maintaining the current situation in order to make maximum use of the available data.

3. **Study origin** - typically it is recommended that benefits transfer exercises in a UK context use studies relating to the UK. However, it is sometimes inevitable that studies from countries with similar socio-economic characteristics to the UK are considered. In this study, all economic data estimates come from the UK with the exception of the estimates for some damage categories within air pollution, which cannot be disaggregated. In addition, the scope of this study includes the remit to breakdown and assess the impacts of agriculture by regions in the UK. The extent to which country level distinctions can be made depends not only on the physical impact data reported in earlier sections but also on the available

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17 Note that the mean WTP transferred could be for an average change in the quality and quantity of environmental resources or a marginal change.
economic literature. For the most part the insufficient breadth in the economic research requires that regional or site specific economic data are applied to all areas of the UK or national economic data are assumed to apply to individual countries as well.

4. **Study methodology** - studies should be grounded in the theory of welfare economics and use robust valuation methodologies. Ideally, studies considered for benefits transfer should demonstrate that valuations given can be assessed for their reliability and validity by providing statistical results from the analysis undertaken. Although it is rare for studies to have undertaken tests for validity in benefits transfer, there are some examples of such studies employed in this report.

5. **Study age** - the date of the original study is recent since the design and implementation of economic valuation techniques have developed over the past 15 years or so. Generally, the more recent the study the more preferred the results will be, as they are more likely to reflect prevailing conditions in the environment and society, and are more likely to have benefited from application of the latest academic developments to the techniques employed.

Where possible lower and upper estimates as well as central estimates are reported here and in Annex 5, and used in Section 5. The ranges reflect statistical confidence intervals, results from different techniques to measure the same good or estimates of similar, or different goods measured by the same technique (e.g. different SSSI sites). Where possible, expert judgement has been employed to narrow the range presented and applied.

Having identified suitable unit estimates in monetary terms for the impacts of agriculture the next step is aggregation of the results. Such calculations are sensitive to the relevant population over which the estimated impacts are aggregated. Generally there are three possible choices of population: (i) residents (in the vicinity of the asset being valued), (ii) visitors (to the same location), and (iii) the general population. Typically residents and visitors are considered ‘users’ of a resource whilst the general population includes both ‘users’ and ‘non-users’ and should be considered where non-use values are deemed significant.

Hence, in some cases, aggregating by the rural population alone (as in Hartridge and Pearce, 2001)\(^\text{18}\) may underestimate the overall benefit or costs imposed by agriculture. For example, multiplying the estimated benefit of the agrarian landscape by the rural population ignores the value placed on that landscape by the urban population. On the other hand, it should also be recognised that non-use values are often observed to decline as geographical distance from a resource increases. This ‘distance decay’ is likely to be an important factor in estimating some of the impacts of agriculture. Moreover, for other impacts the affected resource and the scale of the impact will determine the relevant affected population (e.g. water pollution).

The choice of relevant population will depend on the type of resource being valued, i.e. whether of local, regional or global importance, the population surveyed in the original valuation study, and the nature of the physical data, i.e. if physical data are only available at UK level, then the UK population - either total or rural - is relevant. All these considerations are taken into account in Section 5.

Finally, it is pertinent to consider the use of WTP specifically in the context of environmental accounts for agriculture. To expound, a study that estimates WTP per household per year to preserve a given resource (e.g. from a permanent loss) is in fact estimating the annual service flow accruing from that resource (e.g. the annual benefit received by households). If this resource is permanently lost then the loss to households is estimated as the present value of WTP per household per year over perpetuity (see also Section 2). Shorter periods such as 25 to 30 years could also be used based on the argument that we cannot make a judgment about what the flow of services is likely to be over perpetuity. Thus annual household WTP data can be used to calculate the monetary value of both:

- **Annual flows of (dis)amenity**: (i) positive service flows arising from natural assets under agricultural control or (ii) negative flows of ‘disamenity’ that arise from depreciation on other natural assets due to agricultural activities; and

\(^{18}\) In aggregating the benefits of landscape and biodiversity resources provided by UK agriculture, Hartridge and Pearce (2001) multiply by the rural population and confine their analysis to use values only. Justification for doing so is based on the difficulty and uncertainty surrounding the estimation and application of non-use values, i.e. a lack of empirical studies that estimate distance decay relationships for non-use values of environmental assets.
Changes in stock: changes in asset value that arise from changes in the quantity of stock (such as the permanent loss of a land type under agricultural control) - measured in terms of the present value of all future losses in flow of service.

In this study, only the first type of environmental adjustment has been carried through using economic data, as shown in Section 5. This adjustment relates to data in the current period only, while the second adjustment - not calculated here - requires data for the previous accounting period also.

4.3 Review of the Economic Data

This section summarises the review of economic data sources undertaken for this study. Where appropriate, the distinction is made between market and non-market economic value estimates, and between those estimates that are applied to value welfare changes and those that are applied to value productivity impacts on other sectors. In Section 5, estimated values from the studies reviewed here are applied to physical data to calculate ‘indicative’ monetary values for the economic effects of environmental impacts of UK agriculture for application in the environmental accounts.

Separate sub-sections are presented below for different environmental impact categories associated with agriculture in the same order as Sections 2 and 3. Over 100 studies were reviewed for this project, and as a result of applying the selection criteria, only the most suitable were selected. In particular, the relevance and applicability of the various studies to the context of this study was considered, as well as the applicability of the economic data to the physical data. The body of literature under each environmental impact category is briefly reviewed and at the end of each sub-section the selected estimates are presented, along with details of any adjustments that were required to provide a match for the units of the selected physical data from Section 3. In addition, a brief discussion is provided for impact categories for which no valuation literature has been identified.

Greater consideration of each of the studies considered in the review of literature is provided in Annex 5. This annex provides a tabular summary of the economic valuation studies that relate to each of the impact categories. Studies are described in terms of the ‘environmental aspects’ considered, i.e. the focus of the study, the ‘stressor’ (which indicates, where relevant, the process or feature that is generating the change to be valued), the context of the change valued (e.g. prevention of loss or maintenance of a resource), the methodology employed (e.g. contingent valuation or travel cost method), the economic valuation results that are presented and the study location. Studies and the estimates that are used in Section 5 are highlighted in grey. A list of abbreviations is given at the end of the table. The ‘Economic Value’ column reports the each study’s main results. In most cases these are given in Sterling (£) along with the relevant year.

4.3.1 Water

Review of the Literature

Valuations of the welfare impact of changes in water quality can be sought via a variety of different approaches. Studies on inland waters such as Green and Tunstall (1991) and Georgiou et al (2000) ask WTP for improvements in water quality using a stated preference survey, whilst Johnstone (2003) estimates the consumer surplus per trip from recreational use of rivers with varying levels of water quality, using revealed preference-travel cost method. Spurgeon et al (2001) value improvements in water quality through estimating users’ (anglers) WTP per trip to rivers with different fish populations and characteristics. In terms of valuing the impact of agriculture on water quality, the use of the majority of recreation studies (Peirson et al, 2001; Spurgeon et al, 2001; Davis and O’Neil, 1992) would require an understanding of the extent to which agriculture is responsible for the environmental change being valued. Essentially what is missing to allow many of these studies to be applied in the framework are dose-response functions (as exist for air pollution) to link physical pressures on the state of the environment with final impact. In other words a better understanding of how agricultural inputs to the water environment affect the water environment’s ability to provide the sink and service functions that are valued by society.

Coastal water quality has typically been assessed in terms of WTP for improvements in bathing water quality, focusing on the use value of marine waters, in terms of human health benefits. Eftec (2002) assesses the
effect of pollutants in coastal waters through WTP for reductions in the risk of suffering a stomach upset. Non-UK studies by Le Goffe (1995) and Zyclicz et al (1995) value improvements in water quality that arise from a reduction of the effects of eutrophication. Again in order to link agriculture to the above impacts, it is important to point out that the actual quantitative estimate of agriculture’s contribution may be lacking.

Non-market values for welfare changes associated with changes in water availability and low flow rivers are provided by several studies. WTP estimates are expressed in terms of increases in water level (eftec and CSERGE, 1998), improved flow levels (Garrod and Willis, 1996) and informal recreation (ERM, 1998; Jacobs Gibb Ltd, 2002). In seeking data appropriate for environmental accounting, these studies do not provide the means to estimate the welfare losses associated with water abstraction from agriculture, essentially because no data is available to link agriculture’s contribution to the environmental changes being valued, and data are not provided in matching units. A study conducted by eftec (2003) for Southern Water examined the marginal social benefit of reducing the demand for water, per m³, through an analysis of the environmental benefits of increasing the availability of water in the environment. The ability of this research to provide values per m³, allow it to be employed in the accounting framework. The results are discussed further below.

As described in Section 3, diffuse water pollution from agriculture reduces the productivity of other sectors that bear the burden of clean up costs. The market costs of restocking of fish following agricultural pollution incidents are presented in Pretty et al (2000) and are subsequently employed in estimating accounting adjustments in Section 5. Market cost data are also available for other impacts to water arising from agricultural activities. Specifically, costs are incurred by water companies in meeting the drinking water quality standards. The analysis of the costs of additional water treatment, due in part to agricultural and other diffuse pollution, has focused on the cost of removing excess nitrogen, phosphates and pesticides residues from raw water to meet drinking water standards. This analysis, which is the latest available at the time of writing, was carried out in the mid 1990s and mainly focused on additional capital investment with some review of the additional operating costs. Recent updates were not available as cost data are typically reported by class of capital asset rather than by the function provided by capital assets (e.g. removal of nutrients from drinking water).

Finally, market data on the annual costs of flooding in the UK are provided by the Foresight Report (Evans et al, 2004), reported below, which allow agriculture’s contribution to this cost to be calculated.

Estimates Selected for Accounting

Application of the study selection criteria to the studies summarised above and in Annex 5, selects for the following studies, summarised in turn below. Details of any adjustments required prior to aggregating the results with physical data in Section 5 are also presented here.

Georgiou et al (2000) is the only study undertaken that has linked WTP for improvements in water quality indices. And for this reason has been selected for the accounts. The study reports WTP per household for different schemes to improve water quality in the River Tame. The schemes are defined in terms of achieving improvements in the Resources for the Future (RFF) Water Quality Index (see Vaughan, 1981). The RFF index corresponds to the Environment Agency’s General Quality Assessment (GQA) classifications for river water quality through physical parameters for total ammonia, biochemical oxygen demand (BOD) and dissolved oxygen.

In Georgiou et al (2000) WTP is sought for three schemes: (i) a ‘large’ improvement in water quality level; (ii) a ‘medium’ improvement in water quality level, and; (iii) a ‘small’ improvement in water quality level. Table 4.3 presents the changes in each of the water quality indices that these three schemes would entail and according to each type of index.
Table 4.3: Improvements in Water Quality in Georgiou et al (2000)

<table>
<thead>
<tr>
<th>Water Quality Improvement Scheme presented by Georgiou et al (2000)</th>
<th>Change in Water Quality</th>
<th>Index</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>From</td>
<td>To</td>
</tr>
<tr>
<td>Large Improvement</td>
<td>0.8</td>
<td>7</td>
</tr>
<tr>
<td></td>
<td>RE5</td>
<td>RE1</td>
</tr>
<tr>
<td></td>
<td>E</td>
<td>A</td>
</tr>
<tr>
<td>Medium Improvement</td>
<td>0.8</td>
<td>5</td>
</tr>
<tr>
<td></td>
<td>RE5</td>
<td>RE2/RE3</td>
</tr>
<tr>
<td></td>
<td>E</td>
<td>B/C</td>
</tr>
<tr>
<td>Small Improvement</td>
<td>0.8</td>
<td>2.5</td>
</tr>
<tr>
<td></td>
<td>RE5</td>
<td>RE4</td>
</tr>
<tr>
<td></td>
<td>E</td>
<td>D</td>
</tr>
</tbody>
</table>

In order to apply this study to environmental data presented on water quality in Section 3, further analysis and interpretation of the results were required. This process involved consultation with the study authors and with Environment Agency Guidance on benefits transfer for the water environment. Taking the definitions for the three improvement schemes in Georgiou et al (2000) and the data for chemical and biological indicators, Figure 4.1 illustrates this correspondence on the GQA scale.

In Figure 4.1, foremost is the issue of the baseline. In the River Tame study, this was the state of the river in 1999; that of ‘poor’ quality (‘E’ grade). The small improvement scheme raises water quality to the ‘fair’ standard (‘D’ grade) whilst the medium improvement scheme raises quality to between ‘fairly good’ and ‘good’ (C/B grade). Note that from Table 4.3 this arises from the use of the RFF scale in the Georgiou et al paper. Finally, the large improvement raises water quality to ‘very good’ (A grade), a level that corresponds to pre-industrialisation levels (Georgiou et al, 2000).

The Environmental Accounts baseline is taken to be that of a ‘good’ water quality level. It is implicitly assumed that property rights lie with society to have river water quality of a good level. Hence rivers that are of a good water quality level are assumed to be unaffected by agricultural water pollution. What is then sought is the monetary valuation for negative deviations from this baseline due to the pollution of rivers from agricultural sources.

In order to employ the results from Georgiou et al, it is necessary to assume that WTP for an improvement in environmental quality from state A to state B is equal to WTP to prevent a change in the opposite direction, i.e. from the higher quality of state B to state A. As can be seen from Table 4.3 and Figure 4.1, WTP for a medium improvement best approximates a reduction in water quality from ‘good’ quality to ‘fair’ quality (B to D). WTP for a medium improvement would appear also to be the best approximation for a reduction in water quality from ‘good’ to ‘poor’ (B to E). The nature of the approximation would suggest that using WTP for a medium improvement to value the difference between good and fair quality rivers would be an over-estimate, but using the same WTP amount to value the difference between good and poor quality rivers would likely result in an under-estimation. Overall, application of WTP for a medium improvement in water quality from Georgiou et al (2000) can offer an indication of the magnitude of the non-market cost of reductions in water quality.

Georgiou et al (2000) present estimates of WTP for improvements in water quality in per household terms. WTP for a medium improvement in water quality is estimated to be about £16 per household per year (mean WTP, in 2003 prices calculated from all positive and non-protest zero WTP results, pers. comm. Georgiou July 2004). Assuming this value applies to all rivers and multiplying it by UK population (24.7 million households) gives a total WTP of approximately £391 million per year. Finally dividing by 70,330 km of rivers (total length of river classified for water quality chemical indicator in Section 3) gives a value of £5,560 per km per year, equivalent to the welfare loss to society from the degraded state of inland waters. What proportion of this should be attributed to agriculture is explained in Section 5.
In *eftec* (2002), individuals were willing to pay between about £1 and £2 per household per year for a 1% reduction in the risk of suffering a stomach upset as a result of bathing in marine waters (2003 prices). From this study, total WTP for England and Wales was estimated to be about £62 million per year, a value which can be used in the Environmental Accounts in conjunction with estimates of agriculture’s contribution to faecal pathogens in marine waters. See Section 5 for this calculation.

From *Pretty et al* (2000) the cost to the Environment Agency of pollution incidents (in terms of re-stocking fish) are between £2,155 - £5,388 for category 1 incidents and £1,078 - £2,694 for category 2 incidents (£, 2003). The costs to water companies of providing clean drinking water, are £22.5 million per year for removal of nitrates, £80.2 million for the removal of phosphates, and £147.4 million for the removal of pesticides.

*Eftec* (2003) is selected for estimating the welfare value of impacts of water abstraction on the environment. The assumption in applying these values to the abstraction of water by agriculture for irrigation is that the environmental costs of decreased availability of water in the environment are equal to the lost benefits of leaving this water in the environment. This is a simplifying assumption, but is the best that the literature can currently offer for this impact. The calculations in the report estimated an external benefit per water zone in Southern Water’s area of operation, and provided an average for the South West. Thus, the application of this result to other regions will be less robust, and may under or overestimate the impact of water use by agriculture. The results of the study are an average social benefit of increased water in the environment of £0.27 per m³ per day.
The Foresight Future Flooding report (Evans et al, 2004) provides estimates for the present day annual economic damage to the UK from flooding of £1.09 billion (in 2003). A proportion of this cost is attributed to agriculture in calculations presented in Section 5.

4.3.2 Air

This section summarises the relevant economic valuation literature for air pollution separately at the (i) local and regional and (ii) global level.

Local and Regional Air Pollution

The study titled 'An Evaluation of the Air Quality Strategy' funded by Defra and DfT (AEA Technology, 2004) provides up-to-date damage cost estimates for CO, NOx and SO2 emissions released in the UK. The study evaluates air quality policies that have been implemented over the past decade in the transport and electricity supply industry (ESI) to help provide information on which policies have been successful and which have not, including a cost benefit assessment of the policies.

Annex 5 provides details of the results of this study with estimates for the electricity generation sector, and the transport sector. The key difference between the two relates to (i) the dispersion and deposition of emissions (e.g. emissions from a stack in an electricity generation plant will travel further distances from source those from a car exhaust, but modelling will also take into account that vehicles are mobile sources and power plants are stationary sources) and (ii) the stock at risk, i.e. the human population, environmental assets, buildings, and so on that come in contact with the emissions (e.g. this is likely to be higher for transport emissions in cities that for power generation plants in rural areas).

The central low and central high values represent a restricted range based around the valuation of deaths brought forward and the quantification and valuation of chronic mortality. The implicit value of a statistical life in the AEA Technology estimates (for acute mortality from air pollution) are £3,100 and £110,000, which gives the lower and upper range values. These values only include impacts that occur in the UK, i.e. they exclude impacts that occur from trans-boundary pollution in Europe.

The numbers also include the effects of primary and secondary pollutants. This includes sulphates (from SO2), nitrogen oxides (from NOx) and ozone (from VOC emissions). However, the values for NOx exclude the potential effects on ozone formation (both at a local and regional scale). The estimates do not include a number of effects, and are, thus, likely to underestimate the total economic damage costs of these pollutants. Excluded from the analysis are:

- Impacts on ecosystems;
- Additional health effects, for example, on morbidity and mortality from chronic (long-term) exposure to ozone, and chronic morbidity effects from PM10;
- Damage to cultural heritage, such as cathedrals and other fine buildings, statues, etc.;
- Change in visibility (visual range) as a function of particle and NO2 concentration;
- Effects of ozone on materials, particularly rubber;
- Non-ozone effects on agriculture (e.g. through acid deposition, nutrient deposition, etc.);
- Macroeconomic effects of reduced crop yield and damage to building materials; and
- Altruistic values associated with health impacts.

For SO2 emissions the range employed for the accounts is £744 - £2,296 per tonne SO2 (central low and central high estimates), and for CO emissions a central estimate of £1.4 per tonne CO is used. For NOx emissions the full range from transport and electricity generation sector has been adopted to reflect the variety of locations of NOx emissions from agriculture, thus giving estimates of £190 to £897 per tonne of NOx. The estimate of £1,367 per tonne for Non-Methane VOCs is taken from Pearce et al (1998) and

19 The approach to calculating VOSL is based on the estimation of the WTP for a small reduction in the risk or probability of death, where people are asked to ‘buy’ or ‘sell’ small variations of the probabilities associated with death by various causes. The aggregation of individual values over a whole population leads to the VOSL (or the value of a statistical death avoided).
updated to 2003 prices. Holland et al (1999) provide estimates for NH$_3$ emissions of between £87 and £270 per tonne of NH$_3$. These studies are all summarised in the tables in Annex 5.

Global Air Emissions

A significant number of studies have been conducted on the economic costs of global warming expressed as cost per tonne of carbon dioxide or carbon utilising integrated assessment models. The UK Government has an on-going review of the economic literature about the costs of global warming. Up to 2003 it had adopted a value of £70 tC based on its own review of the available studies (Clarkson and Deyes, 2002) and this figure was released as official guidance (Defra, 2002). The report arrived at this estimate by adopting the values in Eyre et al (1997), which it judged to have used the most sophisticated methods to date. Several new studies on the social costs of carbon have been completed since that date, including a meta-analysis by Tol (2003), which suggests a lower value than £70 per tonne of carbon.

Tol’s conclusion is that, using standard assumptions about discounting and aggregation, the marginal costs of carbon dioxide emissions are unlikely to exceed $50/tC (around £30/tC), and are probably much smaller. Pearce (2003), which is also a review of other studies, concludes that the likely range is £2-15/tC. In both reviews, equity weighting is adopted, i.e. climate damages to poor countries are weighted upwards by a factor based on estimates of the marginal utility of income. Tol’s survey has the advantage of being a meta-analysis, i.e. a statistical analysis of 22 studies and over 80 estimates.

Coverage of the Clarkson-Deyes study is narrower than that of Tol (2003) and Pearce (2003) and, like the latter, suffers from adopting judged ‘best’ studies, whereas Tol’s analysis is a full meta analysis. Arguably, none of the studies adequately accounts for factors that may increase or decrease the damage costs. Increases in the cost may be justified by adding in an allowance for ‘catastrophes’, whilst reductions may be justified by greater allowance for enhanced amenity effects. A further reason for increasing the values is the use of time-declining discount rates - see the review by Groom et al (2003). Pearce (2003) suggests that time-declining discount rates would approximately double the estimates, making his own range £4-30/tC. Tol (2003) does not discuss time-varying discount rates, nor was the issue considered in Clarkson and Deyes (2002).

The Clarkson and Deyes review was taken forward in July 2003 by a DEFRA chaired inter-departmental group on the social cost of carbon (IGSCC), which is due to report findings in the summer of 2004. The group commissioned two research consortia to improve the current Government estimates, provide a better understanding of the uncertainty that surrounds them and to explore how to apply them to policy assessment. In the meantime, the group recommends that the above-mentioned range be employed in any cost-benefit analysis, applying caution where the balance may be tipped by the use of the upper or lower bound estimates. Thus, the Clarkson and Deyes (2002) estimate of £70 per tonne of carbon (with £35-£140 per tonne C as lower and upper bound estimates, or £9.5 - £38/tCO$_2$) is used in this study.

The conventional approach to calculating the social cost of other greenhouse gases (GHGs) is to adopt global warming potentials (GWPs) and simply convert £X/tC into £aX/tCequ (Carbon equivalent) for other GHGs where ‘a’ is the relevant GWP. However, using GWPs does not produce the correct ratios for the different GHGs in terms of the marginal damage done. This is largely due to the use of discounting in marginal damage estimates, whereas GWPs do not allow for discounting. In the current context it is marginal damage that matters. Despite this, the Kyoto Protocol requires that Parties use pre-defined GWPs to secure carbon-equivalents for the other GHGs.

The approach adopted here for arriving at a recommended value for methane and N$_2$O emissions is to adopt the same methodology employed by the Clarkson and Deyes study for recommending the Government’s value of the social cost of carbon. This essentially involves adapting the values for the social costs of methane and N$_2$O reported in the Eyre et al (1997) study, and converting these to current UK prices, using the same conversion factors as Clarkson and Deyes$^{20}$, which gives £400 per tonne of CH$_4$ and £5,588 per tonne of N$_2$O as central estimates. In line with the Government’s recommended approach to using these estimates in policy

\[20\] The Eyre et al (1997) figures were converted to 2000 prices using an inflation rate of 1.35 and from a dollar value using an exchange rate of $1=£0.56$. The final figure is the average of the values suggested by the two models.
evaluation, a range is used for each, where the lower bound is half the central estimate and the upper bound is twice the central estimate.

4.3.3 Soil

Studies that estimate the non-market value of depreciation in soil quality (composition and attributes) or the value of soil losses in terms of its effects on future productivity of land were not found in the literature. The study by Lohr and Park (1995) estimates the benefit to farmers of reducing on-site soil erosion. However, the on-site soil erosion effects on reduced crop yield would already be accounted for in the value of agricultural output.

Some of the off-site costs would already be incorporated in the values of damage to water, insofar as soil erosion impacts upon water quality. Other impacts, such as the effect of soil erosion on property damage and traffic accidents have been reported by Evans (1996). Specifically Evans estimated the off-site cost of soil erosion to be approximately £9.2 million (mostly incurred through dredging of stream channels). A proportion of this cost is attributed to agriculture in Section 5.

4.3.4 Landscape, Habitats and Species

Review of the Literature

Agriculture provides a bundle of non-market goods and services that includes the varied scenery and farmed landscape of the UK (e.g. the ‘traditional patchwork quilt’ notion of the countryside), recreation opportunities through footpaths and farm woodlands, as well as habitats for common and rare or endangered species of flora and fauna. Individuals have preferences for this bundle of goods and thus they derive utility from its provision and hence society benefits from this aspect of agriculture. Such benefits are essentially a jointly produced output of agricultural production, yet their external (or non-market) nature entails that conventional markets do not account for their existence.

The review of studies relating to landscape, habitats and species reveals a great deal of overlap between each sub-category. Therefore, as well as considering the landscape and wildlife amenity provided by agriculture this section also includes aspects of amenity at least partially related to agricultural activity such as the provision of woodlands, recreation, habitats and species. The broad coverage of this category highlights the difficulty of valuing the bundle of non-market goods provided by agriculture. Several studies focus on the benefits of specific aspects of UK agriculture (e.g. the benefits provided by policy designations such as Environmentally Sensitive Areas) whilst some seek to value specific environmental features of the agrarian landscape (e.g. marginal increases in hedgerows or footpaths).

Hence difficulty exists in determining whether a study that produces a value for the non-market benefit of the overall agrarian landscape also incorporates the preferences of individuals for the habitats and species supported by that landscape. Issues of double counting abound, and there are inherent limitations to the use of both physical data relating to landscapes, habitats and species and also the application of estimated economic values.

In Annex 5 the summary table for landscape, habitats and species has been sub-divided on the basis of the subject of study. The following in turn provides a brief overview of the valuation studies relating to these study subjects.

Environmentally Sensitive Areas

A relatively large body of literature reviews the non-market benefits provided by Environmentally Sensitive Areas (ESAs). In total, seven ESAs have been the subject of economic valuation studies, resulting in a total ‘public benefit’ economic value for each. Two studies have also elicited willingness to pay for all ESAs in England and Scotland.

Alvarez et al (1999) report WTP to enhance the Machair ESA in Scotland, which is approximately 15,000 hectares in a predominately coastal plain area. The agricultural land in the area is typically rough pasture
with features such as dunes that support rare bird and plant species. Different landscape and environmental features that are contributed to by agriculture are assessed by Bateman et al. (1994) who compare WTP estimates for the preservation of the Norfolk Broads ESA and WTP for the Yorkshire Dales national park landscape (Willis, 1994). In addition, non-users’ WTP for the preservation of the Norfolk Broads is estimated by Bateman and Langford (1997).

Bullock and Kay (1997) focus on grazing land in the Central Southern Uplands of Scotland and the landscape changes that would result with and without the ESA policy. In contrast, Garrod et al. (1998) consider the benefit derived from marginal increases in the provision of countryside access though footpaths and bridleways provided by agri-environment schemes in England. However, the reported marginal WTP values are based on a linear model and cannot be robustly extrapolated to value the total stock of access provision to the UK countryside.

Garrod and Willis (1995) consider WTP to maintain traditional farming practices in the South Downs. This landscape is mainly chalk down land with rough grazing with a variety of wildlife species and features of archaeological and historical interest. The study also elicits respondents’ WTP for the all English ESAs. The Breadalbane ESA in Scotland is 179,284 hectares of mostly grassland, heather moorland and some woodland. Hanley et al. (1998) elicit household WTP per year to maintain the attractive landscape of the Breadalbane ESA, which includes upkeep of stone walls, protection of rare species of flora and fauna and protection of archaeological sites.

A study by Moss and Chiltern (1997) assesses WTP for environmental improvements to the Slieve Croob ESA in Ireland. Improvements include maintenance of stone walls, replanting and maintenance of hedges, repairs to farm buildings and protection of rough land. Finally, Willis et al. (1995) report WTP estimates to maintain the agricultural landscape of the Somerset Levels and Moors ESA. This area is low-lying with ditches, peat and meadows and the ESA designation is intended to prevent conversion to arable land. The study also reports estimates of WTP for all ESAs.

In summary, it is apparent that ESA schemes embody a vast range of landscape, ecological, heritage and recreational features provided by agriculture in the UK. Typically, valuation studies focus on the benefit provided by extensive agricultural land and prevention of intensification of this land. In general, individuals tend to state a preference for extensive agricultural land over intensive agricultural land (Willis et al, 1995; Bateman et al, 1993). In the context of green accounting, the relatively specific nature of the individual ESAs studied makes it difficult to apply estimated values. In particular, the Norfolk Broads, the Machair, Breadalbane, and the Somerset Levels and Moors are fairly atypical of the average hectare of UK agricultural land.

Thus, a potential approach for using the information provided by the above ESA studies would be to ‘match’ estimated values to all the other ESAs in the UK. However, for the reasons outlined in Section 3, ESA data has not been selected to account for habitats, landscape or species provision by agriculture, and thus valuation studies specific to ESAs are not selected (although their results have been subsumed into the benefits transfer model provided by IERM and SAC, 1999 and 2001, which is used in this study). For one, ESA designated agricultural land accounts for only a relatively small proportion of total agricultural land. In particular, it would be necessary to also estimate the value of the remaining non-designated land. Note also that ESAs are only one agri-environment schemes, for the details of both the ESAs and other schemes, see Section 3.
Landscape and landscape features

Whereas valuation studies that focus on ESAs attempt to value the bundle of non-market goods provided by these areas, there are also other studies that attempt to value specific features of the landscape such as different land types, hedgerows, and traditional stone walls. The most comprehensive study in this respect is IERM and SAC (1999 and 2001). This report, to the then MAFF, combines benefits transfer (through the Environmental Landscape Feature (ELF) model) and original valuation surveys.

Specifically, the ELF model uses the function transfer approach and predicts WTP for seven landscape features: hay meadows, heather moorland, rough grazing land, woodland, arable headland, hedgerows and wetlands. The studies used as an input to the ELF model were classified in terms of the environmental feature for which a WTP amount was elicited. For each feature, two WTP functions are derived from the reported regression coefficients of previous studies (one function with upper bound coefficient estimates the other one with lower bound coefficient estimates). The independent variables included relate to socio-economic, demographic and use factors. This study is selected for use in the accounting framework and is further detailed below. It is worth noting that many of the studies described below have been used in this model.

In a similar context, Hanley et al (2001) use stated preference surveys to value individuals’ preferences for the preservation of certain landscape types. In particular, WTP values are elicited for preventing the conversion of heather moorland to productive grassland or forestry and preventing the conversion of rough grassland to productive grassland or forestry. The land types covered by this study could also be considered as relevant to the ‘habitats’ section of this category, hence demonstrating the overlap between valuation studies in considering these aspects of agriculture.

Two studies, Willis and Whitby (1985) and Hanley and Knight (1992), seek to value the non-market benefits accruing from land that is generally classified as either ‘greenbelt’ or ‘green field’. Use of this land in the urban fringe is varied and typically includes natural and semi-natural land, forested land and both intensive and, to a lesser extent, extensive agricultural land (eftec and Entec, 2002).

Also summarised in the landscape and landscape features section of Annex 5 is Willis and Garrod (1993), a study that focuses on determining preferences for alternative landscape scenarios in the Yorkshire Dales. In part, these results are also summarised in Bateman et al (1994) in the ESA section above, where WTP values to preserve the present Norfolk Broads landscape are compared to WTP values to preserve the landscape of the Yorkshire Dales.

Several non-UK studies exist that consider the non-market benefits of landscapes. Bonnieux and Le Goffe (1997) evaluate WTP for the bundle of non-market services provide by the ‘bocage’ rural landscape of Normandy, France. Bowker and Diychuck (1994) assess the preferences of individuals in Canada for the retention of agricultural land under threat from urban development. In a similar context, Drake (1992) focuses on WTP to prevent the conversion of agricultural land to forest in Sweden.

Finally, Kline and Wichelns (1998) develop a model of preferences towards the preservation of farmland and attributes such as scenic views, open space and ecological functions based on respondents’ attitudes towards the aesthetics, the environment and agriculture in Rhode Island, USA. This study is not in fact a valuation study, but does demonstrate how other areas of economic analysis can also be applied to the assessment of individuals’ preferences towards the non-market services provided by agriculture.

Woodland

Annex 5 includes a broad range of studies that consider various economic value aspects of woodland, such as preferences for a forested landscape, recreation opportunities, and/or habitat and biodiversity functions of forests. Agriculture both ‘supplies’ woodland because some is on farms and managed (to varying degrees) by farmers and is in competition with woodlands as the principal alternative land use. Therefore, the most
relevant to environmental accounts are those studies that consider the benefits of farm woodlands (e.g. those that consider aspects of the ‘Community Woodlands Scheme’ scheme)\(^{21}\).

Bateman et al (1996) estimate preferences of individuals for the conversion of farmland to woodland and the recreation opportunities that would arise. Similarly, Bishop (1992) assesses the recreation opportunities of community woodlands in the urban fringe as well as incorporating preferences towards wildlife and the landscape. In a less relevant context, Maxwell (1994) considers WTP for improvements to a community forest such as tree planting, enhancement of views and water features. In addition, IERM and SAC (2001) also provide per hectare estimates for farm woodland in different regions and also in a national average context.

A slightly different valuation context is assessed by Bennett et al (1995) who assess WTP for different levels of public access to Windsor Forest. Other studies summarised in Annex 5, such as Benson and Willis (1990) focus on larger scale commercial forests and provide potentially less appropriate estimated economic values for the application of the green accounts.

Habitats

In Annex 5 there is a certain degree of overlap between studies that have been classified in this review as either ‘landscape/features’ or ‘ESAs’\(^{22}\) with those classified as ‘habitats’.

Two studies, Willis and Benson (1988) and Willis (1990), consider WTP for wildlife conservation in three Sites of Special Scientific Interest (SSSIs). In both studies, intensification of agricultural processes is the ‘stressor’ on these key conservation sites, giving a valuation context that is potentially suitable to valuing losses of significant habitats and associated species in the environmental accounts. In a similar valuation context, Willis et al (1996) assess the benefits of a wildlife enhancement scheme in the Pevensey Levels SSSI where previous farming pressures have threatened wetlands that support a diverse range of wildlife.

Hanley and Spash (1993) report the results from two separate stated preference surveys. The first presents WTP for the preservation of lowland heath whilst the second assesses WTP to prevent the loss of ancient woodland habitat to a road development. In Hanley and Craig (1991) respondents’ WTP to protect peat bogs in Northern Scotland was elicited.

Species

Few studies look at changes in or maintenance of a particular species or biodiversity in relation to farming practices. In addition, as opposed to, say, pollution, the impacts of agriculture on biodiversity can be perceived to be either positive or negative and typically depend on the preferences of individuals and also location (Macmillan, 2001) (this point may also be valid for preferences individuals hold for the agrarian landscape). Moreover, agriculture’s impact on biodiversity is complex. While the status of some species is declining due to agricultural practices, several depend on certain types of farm management for their survival. Whilst physical impact assessments identify such cases, economic valuation studies are limited.

One study which does consider in more detail the links between species and agriculture is Macmillan et al (2002). This study shows that individuals typically value endangered species of wild geese more so than non-endangered species. Two studies reported in Annex 5 consider the issue of species decline. Foster et al (1998) estimate WTP for reductions in the long term decline of bird species as a result of pesticide use. O’Neill (2001) considers bird species and the loss of habitats through WTP to maintain the quality of various sites for bird watching.

Christie et al (2004) seeks to elicit WTP, in Cambridgeshire and Northumberland, for a number of policies that would avoid or reduce biodiversity decline, where one of the policies is agri-environment schemes. The study

\(^{21}\) The Community Woodlands Scheme was introduced in 1991 by the Forestry Commission with the aim to provide woodland close to areas where woodland recreation opportunities were limited. Farmers were eligible for financial support to establish recreational woodland on their land (Bateman et al, 1996).

\(^{22}\) For example, Hanley et al (1998) considers the preservation of heather moorland and rare plant species and could be classified under the ‘habitats’ heading. In classifying valuation studies of this sort, one determining factor was whether specific reference is made to agriculture. Studies considering the benefits of ESA designations or the agrarian landscape have been classified appropriately so.
also elicits WTP for ‘rare’ and ‘familiar’ species and for habitat restoration and creation generally. The results of the study could be used to derive a national WTP for biodiversity provision by agri-environment schemes. However, the interpretation of these results is that agri-environment schemes are the vehicle for avoiding a decline in biodiversity. Thus, the values could be applied if this was believed to be the case. Further work on interpreting the results could prove more fruitful in developing values for environmental accounting in agriculture, but is beyond the scope of this study.

Estimates Selected for Accounting

On the basis that the IERM and SAC (2001) study’s ELF model provides a validity tested23 benefits transfer model based on the majority of the existing literature on UK landscapes and habitats presented above, the study’s estimates are used in Section 5 to calculate the aggregate value of agriculture’s provision of landscape and wildlife habitats. This provides, by far, the most robust approach to accounting for services provided by the stock of land assets held by agriculture.

In the ELF model willingness to pay per household is predicted for different regions using the value of the explanatory variables for each region. WTP per hectare is then calculated on the basis of two assumptions: (i) that WTP in a given region is distributed normally, and (ii) that in the original studies, the respondents are expressing their WTP for the total regional abundance of the feature in question. Hence, the ELF model estimates the benefits of a feature in a given region that accrues only to the population of that region. Therefore, the benefit of a particular feature to those living outside the region in which it is found and also to non-resident visitors to that region is not estimated. Hence the predicted values are likely to serve as lower bound estimates. It should also be noted that non-resident valuations can be a significant component of total benefit (CRER and CJC Consulting, 2002). In order to account for these aspects, distance decay relationships would need to be estimated, which are likely to be both feature and region specific.

However, the ELF model does enable the calculation of demand curves that estimate the rate of diminishing marginal utility and an indication of WTP for different abundances of each feature. This aspect of the model is particularly appropriate for the assessment of policy measures which seek to increase the abundance of different environmental features. In addition, national average WTP per hectare for each feature is calculated by multiplying mean WTP for each feature by the whole population and then by dividing by national abundance of that feature.

The ELF model was designed to provide economic values for marginal changes in the provision of environmental features of agri-environment schemes. It was not designed to provide values for the current provision of the whole range of agricultural land types; thus, some adaptations of the model were required to apply the results to the environmental accounting framework.

In selecting values from the ELF model, the approach was to take those values that represent WTP for 100% provision of current land-based environmental services (rather than for, say, a 10% or 50% change). Regional estimates for 100% provision of each landscape type or feature were extracted from the model and used to arrive at a national average value based on the regional abundance of each land type reported in the study. These average values are presented in Table 4.4, and are subsequently employed in Section 5. Ideally, as the regional abundance of these features change they should be fed back into the model to re-estimate WTP. Currently, this level of sophistication is not possible within the model. In order to be able to apply the ELF model to a wider set of agricultural land types, a matching exercise was undertaken to pair the values for the land type categories employed in ELF to the more encompassing broad habitat classification system used in both the Countryside Survey, the Biodiversity Action Plans and in classifying SSSIs (see Section 3). This mapping exercise is also demonstrated in Table 4.4, which also shows in shaded cells where it was not possible to provide a suitable approximate match. Only those broad habitat types considered to be under agricultural management are included in the analysis. This already accounts for 74% of land in the UK.

---

23 In a validity test the model was used to predict the WTP results of a new valuation study. The model’s prediction was statistically the same as the study’s result (with overlapping 95% confidence intervals).
Table 4.4: Matching the ELF model to Broad Habitat Types

<table>
<thead>
<tr>
<th>Broad Habitat Type</th>
<th>Feature (from ELF)</th>
<th>National average WTP per ha per year (£, 2003)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Improved Grassland</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Arable and Horticultural</td>
<td>Hay Meadow</td>
<td>131</td>
</tr>
<tr>
<td></td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Dwarf shrub heath</td>
<td>Heather Moorland or Heathland</td>
<td>23</td>
</tr>
<tr>
<td>Neutral, acid and calcareous grassland</td>
<td>Rough Grazing</td>
<td>84</td>
</tr>
<tr>
<td>Woodland</td>
<td>Woodland</td>
<td>135</td>
</tr>
<tr>
<td>Arable Headlands</td>
<td>Hedgerows</td>
<td>21</td>
</tr>
<tr>
<td>Linear features a</td>
<td>Wetland</td>
<td>92</td>
</tr>
<tr>
<td>Fen, Marsh, Swamp and Bog</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

a: Although not a broad habitat type this data is also collected by the Countryside 2000 Survey.

Considerable difficulty is encountered in attempting to coherently link physical data currently collected with economic values in order to give a complete picture of agriculture’s role in maintaining or degrading the biodiversity of natural habitats. The studies by Foster et al (1998) and Willis (1990) are applied in Section 5 as a first step.

Willis (1990) reports WTP per hectare for conservation of three SSSI sites. The highest value reported is that for Skipwith Common (£3,484 per hectare, 2003 prices) whilst Derwent Ings (£767 per hectare) and Upper Teesdale (£669 per hectare) are significantly lower WTP values. Given the characteristics of the Skipwith Common site, and particularly the fact that the small land area (315 hectares of wet heath, poor fen and woodland) is a significant factor in determining the large magnitude of the per hectare value, this value is not applied in the Environmental Accounts. A more conservative approach is adopted and a midpoint between the values for the Derwent Ings site (783 hectares, mostly floodplain land) and the Upper Teesdale SSSI (9,200 hectares of valley meadows and uplands), which gives £718 per hectare of SSSI designated land.

Although the approach to valuing SSSIs is somewhat crude, compared to the approach applied through the use of the ELF model, having the two sets of data is useful. The advantage of adopting a separate estimate for SSSI land is that the accounts can differentiate between the value of landscape, habitats and species provided on non-SSSI land and that provided by SSSI land. The value of the latter being higher, as would be expected. Nonetheless, more research to develop a more refined approach is desirable.

Foster et al (1998) report the results of a choice modelling study that estimates the value of human health and biodiversity impacts associated with pesticide use. The basic idea behind the study was to ask consumers to rank a conventional loaf of bread against a number of alternative loaves, differentiated in terms of price and eco-labels which identified the environmental impacts generated by the underlying production processes. In terms of biodiversity, the impact was measured as ‘the number of farmland bird species in a state of serious long-term decline as a result of pesticide use in cereal cultivation’, where the baseline was nine species in decline. The values produced for potential application to this accounting exercise are expressed in units of £ per bird species in decline (i.e. household willingness to pay to avoid the loss of a bird species). When aggregated across the UK population, the total value of preventing the decline of nine species was estimated to be £246 million per year (£,1997), based on the number of loaves sold in the UK.

In order to calculate the value to society of all farmland bird populations this result needs to be interpreted in order that it fit with RSPB data on farmland birds (Gregory et al, 2003). These data, presented in Section 3, use the 1970 bird population level as a baseline to measure current bird populations. In 2003, these were at 60% of 1970 levels. The annual welfare value of avoiding the decline to zero of nine farmland species, of £246 million, is adjusted to provide a value to society ‘per species per population %’, as £5.12 million/yr/species/population % (=£246 million per year / (100-0) / 9 species).

Gaps in the economic literature have meant that it was not possible to account for the visual (dis)amenity of intensive agricultural land use (following the broad habitat types, this refers to improved grassland, and
non-SSSI designated arable land) since the only study on the issue (Bowker and Dyychuck, 1994) is in the context of urban development and in Canada and hence not relevant here. Similarly, studies on the value of greenbelt land value agricultural landscape against the threat of development and are thus reflect both the disamenity of housing development and the value of the landscape without housing. An agricultural policy context would be more appropriate for the accounts. This gap in the literature is picked up again in Section 6.

4.3.5 Waste

Several studies have attempted to estimate all externalities of solid waste going to landfill and incineration, the most up to date of which is COWI (2000) (given this, Annex 5 reports only data from this study). DEFRA will soon publish a report which collates and updates this information Enviros and etec (forthcoming), however, the results are not available in time for the completion of this study, and in any case, will not be presented in terms of £/tonne of waste. Thus, the COWI (2000) figures are reported here.

COWI (2000) provides net external cost estimates per kilogram of waste to incineration and landfill for the EU, based on a complete review of the economic valuation literature for each of the environmental impacts from these waste management options. Several scenarios are explored for each option, and these are summarised in Annex 5. Impacts covered in the external costs of incineration and landfill are global warming, damage from regional and local air pollution, damage from leachate to water and disamenity costs.

If it is assumed that all landfills in the UK comply with current legislation, which they are required to do in order to receive a permit for operation, then the average external cost is £12 per tonne of waste to landfill based on the COWI (2000) review, just below the current tax rate.

All UK incinerators need to meet current standards in order to operate, however the number of incinerators meeting the proposed directive standards is unknown. The Environment Agency reports on its website that in 2000/2001, about 8% of municipal solid waste was incinerated with energy recovery in England and Wales. As energy recovery accounts for the majority of the benefit reported for the incineration scenario in COWI (2000) (see Annex 5), we can assume that 8% of incinerators in the UK adhere to the proposed standards. This gives a weighted external cost of £32 per tonne of waste to incineration It should be noted that this value is calculated on the basis of municipal solid waste, but is a ‘best estimate’ in the context of the green accounts since no analysis has been carried out in relation to agricultural waste.

The weighted average of all waste is derived from estimates from the Environment Agency that 77% of all waste is sent to landfill, 9% to incineration, and 13% recycled or composed. If we assume that the external costs of recycling are zero (which is an underestimate) then this gives a weighted average external cost of £15 per tonne of waste, which is actually equal to the current landfill tax for non-inert waste.

4.3.6 Nuisance

No economic valuation studies were found that consider nuisance aspects of agriculture such as odour and noise. Studies that consider noise pollution typically concentrate on that which arises from the transport sector. Estimates of the disamenity associated with odour exist in the literature, but only in the context of landfills, and, therefore, not suitable for application to the agricultural context. Nor is it possible to separate out values for odour and noise disamenity from the estimates of total disamenity from landfill valuation studies.

4.3.7 Cultural heritage and archaeology

While buildings of cultural heritage and archaeological importance have a market value which will be included in the conventional agricultural accounts, what is not known is the willingness to pay of society for their conservation. This willingness to pay is likely to be highest for buildings that the public can see or visit. Alternatively, the values, particularly, of farm buildings, may be subsumed in the wider value of ‘the countryside’ if we knew what that was. The example underscores the accounting problems that arise from sheer lack of information about economic values.
As mentioned in Section 3, there is a difficult issue of deciding whether these buildings have a value that can be ascribed to agriculture. If they are listed then, technically, they cannot be destroyed and yet some of their value arises because of listing (although listing works both ways: by restricting conversion and adaptation, it can lower financial value; by giving a public status it can raise the financial value). The question is then how much of the willingness to pay of the public arises because of agricultural ownership and how much because of government regulation? There are two arguments here. In so far as the buildings are on agricultural land, then they should appear as part of the agricultural accounts. In so far as the responsibility for their value is concerned, then perhaps a fraction of this value arises from government regulation and the value should be assigned to government accounts.

The problem of ascription arises more with other assets where the degree of government regulation is greater, e.g. archaeological sites. If the site is registered and must be protected then the conservation of the site would appear to be a benefit that should be ascribed to government rather than agriculture. In so far as farmers bear a cost on honouring any statutory obligation to protect such sites, then we can argue that the sites are ‘produced’ by the agricultural sector.

From the perspective of the review of economic studies, no literature has been identified that specifically addresses the role of agriculture and cultural heritage and archaeological sites. Several of the ESAs studies reported in Section 4.3.4 do include such elements in the ‘bundle’ of non-market goods and services provided by agricultural for which a valuation is sought. However, it is generally difficult to extract a WTP value for such sites. Only Hanley et al (1998) provide a WTP value for archaeological features though the choice experiment aspect of the study. Moreover, Table A4.4 in Annex 4 reveals a gap in the physical data for the maintenance of historic farm buildings and historic and archaeological sites.
5 Monetary Environmental Accounts for Agriculture

This Section presents the estimates of environmental costs and benefits of agriculture as much as the available data allow. After the overview of the approach in Section 5.1, specific calculations for each impact are presented in Section 5.2. Where physical data exist and can be matched with economic data, the economic values of the environmental impacts are calculated. More detailed calculations can be found in Annex 6.

Section 5.3 presents a summary table and interpretation that points to where there are gaps in either the physical data or the economic data, this calculation has not been possible. Section 6 addresses these gaps in terms of future research in the context of building the environmental accounts for agriculture.

5.1 Approach

The environmental accounts and hence the calculations outlined in this Section incorporate the DPSIR model to analyse the linkages between the environment and economy using the impact categories in Table 1.1. As detailed in Section 4, the economic value estimates come from a literature review and are adjusted for the agricultural accounting context following the benefits transfer approach.

The calculations in this Section provide monetary estimates of the environmental impacts of agriculture (subject to data availability) that can be used in an income accounting exercise. They do not cover wealth accounting since it is more meaningful to quantify the services provided or impacted upon by agriculture than to quantify the value of the entire set of environmental assets most of which are beyond the province of agriculture and hence are not attributable to agricultural accounts.

This distinction between income and wealth accounting means that the calculations show the economic value of the positive environmental services provided by agriculture as well as the negative service flows that result from depreciation of natural assets such as air and water. This is manifested by the impacts on the welfare of society in general and productivity gains and losses to other sectors as shown in Tables 2.3 and 2.4 as part of the accounting framework. In practical terms, this means that the calculations presented here are in £ per year terms (representing service flows or reductions in income) and no present value calculations are undertaken to account for changes in wealth resulting in change to a given stock of environmental capital.

Thus, the time period for the calculations is one year, in line with standard practice. Therefore in order to produce a yearly account of such impacts, one essential component of the environmental accounts is annual data for the physical impacts (e.g. the state of water quality in rivers each year, or the number of hectares of land providing landscape benefits). However, as discussed in Sections 3 and 6, some difficulty is encountered in that physical data relating to various aspects of, and impacts from, agriculture is not collected or collated on an annual basis.

In addition to this definition of the environmental and accounting boundaries of the calculations, we also need to establish the relevant affected population. For most impacts, the entire UK population is taken as the relevant population (a more sophisticated approach is used in the valuation of landscape benefits which relate WTP to regional populations and characteristics). Exemptions include the larger (global or European) populations impacted by global and regional air pollution from the UK. These population estimates are embedded in the unit economic costs used here.

In line with the objectives of the study, country-level breakdowns of the calculations are presented in the results tables in this Section. This breakdown has not been possible for all impacts due to the physical data not being collected or presented at the country level. Where country level physical data are available, it is

Note that the only relevant calculations here would be changes in wealth to agriculture from changes to land-based assets.
sometimes the case that economic data for the UK as a whole or from a particular location are applied uniformly to all countries. Further detail on this can be found in Annex 6.

As with all exercises that entail economic valuation, and, indeed, other empirical or statistical procedures, sensitivity analysis should be conducted in order to reflect areas of uncertainty. Therefore, where possible, from the available literature, a range of economic estimates have been used to calculate lower and upper bound values as well as a central one. On the flip side, expert judgement has been employed to try and narrow these ranges as much as is possible, and all summary data is presented in using mid points or central estimates.

5.2 Monetary Estimates of Environmental Impacts

5.2.1 Water

This section presents the calculations for water quality and quantity impacts of agriculture as manifested by welfare impacts on society and productivity impacts on other economic sectors. The text outlines the sources of economic and physical data and illustrates some of the adjustments and calculations. The full set of calculations is presented in Tables 5.1 and 5.2, for welfare and productivity impacts, respectively. Note that the adjustment factor presented in the Tables referred to the linking data presented in Section 3. Detailed calculations can be found in Annex 6 (worksheet A6.1 Water).

**Adjustments for Welfare Impacts on Society**

**Water quality:**

For the purposes of calculating welfare impacts to society from depreciation on inland (or surface) waters the results from Georgiou et al (2000) (see Section 4) can be applied to the kilometres of river that are classified as either ‘fair’ or ‘poor’ quality (See Table 3.1). Applying these willingness to pay (WTP) estimates to all of the rivers provides a measure of the annual welfare impact on society from loss of amenity associated with rivers classified as ‘fair’ or ‘poor’ (or the depreciation of water as a natural asset) of £5,560/km of river. However, not all of the pollution causing this impact is from agriculture and hence not all of this cost can be attributed to it.

Linking data mentioned in Section 3 are required to assess the share of agriculture in this cost. Such data are provided by Environment Agency (2002) which suggests that 70% of nitrogen entering surface waters is from agriculture (on the basis of WRc 1999). Hence this proportion is attributed to chemical water quality in this report, where chemical water quality is total ammonia, biochemical oxygen demand and dissolved oxygen. For the UK, this results in a cost of around £50 million per year (=£5,560/km x 12,951km x 0.7).

Similarly, linking data are used to estimates the share of disamenity attributable to agriculture for poor quality in bathing waters. Environment Agency (2002) estimates that 5% of faecal pathogens to bathing waters is accounted for by agriculture. The welfare cost of this impact is calculated using eftec (2002) which estimated WTP for avoiding stomach upsets associated with bathing in faecally contaminated bathing water in British beaches to be £61.8 million when aggregated across England and Wales. For the UK, this results in a cost of around £3 million per year (=£61.8 million/year x 0.05).

**Water availability:**

Section 3 reports the amount of water abstraction by agriculture (in England and Wales) in terms of mega litres per day. Eftec (2003) gives a value of the benefit to society from reducing abstraction and leaving more water in the environment. This benefit estimate, when applied to the amount of water abstracted by agriculture, gives the cost of abstraction (or avoided benefits) to society. In order to match the units of economic data (£ per m$^3$ per day) to physical data (ML), the economic cost estimate was multiplied by 365 (days in year) and by 1000 (number of cubic meters in a mega litre). For England and Wales, for which
abstraction data (excluding spray irrigation) are available, this means a cost of around £10 million per year (=£0.27 per m$^3$ per day x 365 x 1000). Both this and the cost for spray irrigation are presented in Table 5.1.

**Adjustments for impacts on other sectors**

**Water quality:**

Two impacts under this heading can be expressed in monetary units: the cost of restocking fish populations in rivers after pollution incidents and the cost of treating drinking water for nitrates, phosphates and pesticides.

Section 3 shows that there are 13 Category 1 and 137 Category 2 pollution incidents in England and Wales (Environment Agency Pollution Incidents Database, 2002) and 56 ‘significant’ pollution incidents in Scotland. Following Pretty et al (2000), it is assumed that only Category 1 and 2 incidents are severe enough to result in fish kills and that ‘significant’ incidents in Scotland are deemed to be sufficiently similar to Categories 1 - 3 in England and Wales. The cost of restocking fish was reported in Pretty et al (2000) and these values updated to 2003 prices are used here. The Category 2 costs are used for ‘significant’ pollution incidents in Scotland. For England and Wales, this results in a cost (central estimate) of about £0.4 million per year for Category 1 (=£3,772 per incident x 118 incidents) and £1.6 million per year for Category 2 (=£1,886 per incident x 860 incidents).

Market cost data are also used to estimate the costs of removing nitrates, phosphates and pesticides from drinking water. Following Pretty et al (2000) capital and operating costs of water treatment per year for the period 1992 to 1997 are used in conjunction with estimates of agriculture’s contribution to nitrates, phosphates and pesticides in the water supply. Since these cost estimates have not been updated by the water sector (see discussion in Section 6.3.1), they are used here as updated to 2003 prices. The linking data are used to estimate the share of agriculture in nitrates (70%), phosphates (43%) and pesticides (89%) in drinking water and assumed to apply the whole country (See Section 3.2.1 for data sources). For the UK, this means a total cost of about £181 million per year (=£22.5 million x 0.7 + £80.2 million x 0.43 + £147.4 million x 0.89).

**Water availability (Flooding):**

There is much concern about the potential link between agricultural practices, runoff processes and flooding, however, the nature of this link remains uncertain. The Environment Agency (EA, 2002) attempts to draw a link based on its own flood event data and finds that 25% of all flooding events in the 1980s and 90s were hillslope events. On the basis that 57% of hillslope events were caused by erosion and deposition, the report concludes that at least 14% of all flood events, and flood event costs, should be attributed to agriculture. The EA purports that this estimate is likely to be conservative due to under-reporting of local agriculture related flooding events and also lack of data linking agriculture’s contribution to fluvial flooding.

The Foresight Future Flooding report (Evans et al, 2004) provides estimates for the present day annual economic damage to the UK from flooding of £1.09 billion (in 2003). If the EA approach is adopted, then the cost of flooding that could be attributed to agriculture would be about £153 million per year (=£1.09 billion x 0.14). While the report notes significant evidence of current land-management practices leading to increased surface runoff at the local scale, it finds a general absence or uncertainty of evidence of the impacts of land management practices at the catchment or national scale and recommends further research in this area.
### Table 5.1: Water - Welfare Impacts on Society

<table>
<thead>
<tr>
<th>Impact from Agriculture</th>
<th>Physical Data ((\alpha))</th>
<th>Adjustment factor ((p))</th>
<th>£ per unit or total (£, 2003)</th>
<th>Total (£ million)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Lower bound ((\beta_L))</td>
<td>Central ((\beta_C))</td>
</tr>
<tr>
<td><strong>Value of water pollution arising from agricultural production</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pollution of rivers: length of rivers classified as ‘fair’</td>
<td>E: 10,133 km</td>
<td>0.7</td>
<td>(70% due to agriculture)</td>
<td>-</td>
</tr>
<tr>
<td>W: 274 km</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>S: 2,544 km</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NI: 1,735 km</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>UK: 12,951 km</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pollution of rivers: length of rivers classified as ‘poor’</td>
<td>E: 2,171 km</td>
<td>0.7</td>
<td>(70% due to agriculture)</td>
<td>-</td>
</tr>
<tr>
<td>W: 91 km</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>S: 1,018 km</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NI: 165 km</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>UK: 3,280 km</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Faecal contamination of marine waters</td>
<td>-</td>
<td>0.05</td>
<td>(5% due to agriculture in the UK)</td>
<td>-</td>
</tr>
<tr>
<td><strong>Value of agricultural water availability (in excess of rates of recharge)</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Agricultural abstraction (excl. spray irrigation)</td>
<td>108 ML/day (E&amp;W)</td>
<td>(365 × 1000)</td>
<td>-</td>
<td>0.27 (per m(^3))</td>
</tr>
<tr>
<td>Spray irrigation</td>
<td>259 ML/day (E&amp;W)</td>
<td>(365 × 1000)</td>
<td>-</td>
<td>0.27 (per m(^3))</td>
</tr>
</tbody>
</table>

E: England; W: Wales; S: Scotland; NI: N. Ireland; UK: United Kingdom
Table 5.2: Water - Productivity Losses in Other Sectors

<table>
<thead>
<tr>
<th>Impact from Agriculture</th>
<th>Physical Data (α)</th>
<th>Adjustment factor (p)</th>
<th>£ per unit (£, 2003)</th>
<th>Total (£ million)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Lower bound (β_L)</td>
<td>Central (β_C)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>5</td>
<td>5</td>
</tr>
<tr>
<td>Market cost of water pollution</td>
<td></td>
<td></td>
<td>5</td>
<td>5</td>
</tr>
<tr>
<td>Number of Category 1 pollution incidents</td>
<td>13 (E&amp;W)</td>
<td>-</td>
<td>2,155</td>
<td>3,772</td>
</tr>
<tr>
<td>Number of Category 2 pollution incidents</td>
<td>137 (E&amp;W)</td>
<td>-</td>
<td>1,078</td>
<td>1,886</td>
</tr>
<tr>
<td>Number of 'significant' pollution incidents</td>
<td>56 (S)</td>
<td>-</td>
<td>1,078</td>
<td>1,886</td>
</tr>
<tr>
<td>Nitrates in drinking water</td>
<td>-</td>
<td>0.7 (nitrates from agric) (UK)</td>
<td>-</td>
<td>22.5m</td>
</tr>
<tr>
<td>Phosphate in drinking water</td>
<td>-</td>
<td>0.43 (phosphate from agric) (UK)</td>
<td>-</td>
<td>80.2m</td>
</tr>
<tr>
<td>Pesticides in drinking water</td>
<td>-</td>
<td>0.89 (pesticides from agric) (UK)</td>
<td>-</td>
<td>147.4m</td>
</tr>
<tr>
<td>Costs of flooding</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Flooding attributed agriculture</td>
<td>-</td>
<td>0.14 (agric. contribution)</td>
<td>-</td>
<td>£1090.0m</td>
</tr>
</tbody>
</table>

E: England; W: Wales; S: Scotland; NI: N. Ireland; UK: United Kingdom

Notes:
* In the absence of central estimates of economic values, these values are calculated as: (lower bound estimate + upper bound estimate)/2.
5.2.2 Air

Section 3 presents the physical data on air-borne emissions from agriculture provided by Netcen for this study. These are matched with the unit damage cost for the pollutants covered, namely, methane and nitrous oxide (from Eyre et al 1997), ammonia (Holland et al, 1999), nitrogen oxide, sulphur dioxide and carbon monoxide (AEA Technology, 2004), carbon (Clarkson and Deyes, 2002) and non-methane volatile organic compounds (Pearce et al, 1998). In some instances the literature does not provide a central estimate (e.g. AEA Technology, 2004) does not calculate central estimates for the cost per tonne of NOx and SO2). In these cases, aggregate central estimates are simple averages of lower and upper bounds. The impact of those air-borne pollutants covered in this analysis on the welfare of the society results in a total cost (central estimate) of £956 million per year. Table 5.3 shows the calculations behind this figure for each pollutant for each country. The table presents costs as welfare impacts on the society. While some of these costs are due to productivity losses (e.g. damage to materials or medical costs incurred due to impacts on health), these are not possible to disaggregate within the scope of this study. Further detail is given in Annex 6 (worksheet A6.2 - Air).
Table 5.3: Air - Welfare Impacts on Society

<table>
<thead>
<tr>
<th>Impact from Agriculture</th>
<th>Physical Data (tonnes)</th>
<th>Adjustment factor (p)</th>
<th>£ per unit (£,2003)</th>
<th>Total (£ million)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Lower bound (β_L)</td>
<td>Central (β_C)</td>
</tr>
<tr>
<td>Value of air pollution arising from agricultural production</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>CH₄ E: 509,838</td>
<td>-</td>
<td>200</td>
<td>400</td>
<td>800</td>
</tr>
<tr>
<td>W: 134,847</td>
<td></td>
<td></td>
<td>102.0</td>
<td>204.0</td>
</tr>
<tr>
<td>S: 150,479</td>
<td></td>
<td></td>
<td>30.1</td>
<td>60.2</td>
</tr>
<tr>
<td>NI: 118,911</td>
<td></td>
<td></td>
<td>23.8</td>
<td>47.6</td>
</tr>
<tr>
<td>UK: 914,075</td>
<td></td>
<td></td>
<td>182.8</td>
<td>365.6</td>
</tr>
<tr>
<td>N₂O E: 56,520</td>
<td>-</td>
<td>2,794</td>
<td>5,588</td>
<td>11,176</td>
</tr>
<tr>
<td>W: 9,228</td>
<td></td>
<td></td>
<td>158.0</td>
<td>315.8</td>
</tr>
<tr>
<td>S: 14,023</td>
<td></td>
<td></td>
<td>25.8</td>
<td>51.6</td>
</tr>
<tr>
<td>NI: 7,924</td>
<td></td>
<td></td>
<td>39.2</td>
<td>78.4</td>
</tr>
<tr>
<td>UK: 87,695</td>
<td></td>
<td></td>
<td>22.1</td>
<td>44.3</td>
</tr>
<tr>
<td>NH₃ E: 172,099</td>
<td>-</td>
<td>87</td>
<td>-</td>
<td>270</td>
</tr>
<tr>
<td>W: 27,791</td>
<td></td>
<td></td>
<td>15.0</td>
<td>30.7</td>
</tr>
<tr>
<td>S: 41,067</td>
<td></td>
<td></td>
<td>2.4</td>
<td>5.0</td>
</tr>
<tr>
<td>NI: 31,317</td>
<td></td>
<td></td>
<td>3.6</td>
<td>7.3</td>
</tr>
<tr>
<td>UK: 277,274</td>
<td></td>
<td></td>
<td>2.7</td>
<td>5.6</td>
</tr>
<tr>
<td>NOx E: 12,593</td>
<td>-</td>
<td>190</td>
<td>-</td>
<td>897</td>
</tr>
<tr>
<td>W: 1,948</td>
<td></td>
<td></td>
<td>2.4</td>
<td>6.8</td>
</tr>
<tr>
<td>S: 2,346</td>
<td></td>
<td></td>
<td>0.4</td>
<td>1.0</td>
</tr>
<tr>
<td>NI: 1,780</td>
<td></td>
<td></td>
<td>2.4</td>
<td>6.8</td>
</tr>
<tr>
<td>UK: 18,667</td>
<td></td>
<td></td>
<td>0.3</td>
<td>1.0</td>
</tr>
<tr>
<td>NMVOC E: 3,063</td>
<td>-</td>
<td>-</td>
<td>1,367</td>
<td>-</td>
</tr>
<tr>
<td>W: 474</td>
<td></td>
<td></td>
<td>3.5</td>
<td>10.1</td>
</tr>
<tr>
<td>S: 571</td>
<td></td>
<td></td>
<td>-</td>
<td>0.6</td>
</tr>
<tr>
<td>NI: 437</td>
<td></td>
<td></td>
<td>-</td>
<td>0.6</td>
</tr>
<tr>
<td>UK: 4,544</td>
<td></td>
<td></td>
<td>-</td>
<td>6.2</td>
</tr>
<tr>
<td>SO₂ E: 989</td>
<td>-</td>
<td>744</td>
<td>-</td>
<td>2,296</td>
</tr>
<tr>
<td>W: 153</td>
<td></td>
<td></td>
<td>0.7</td>
<td>1.5</td>
</tr>
<tr>
<td>S: 184</td>
<td></td>
<td></td>
<td>0.1</td>
<td>0.2</td>
</tr>
<tr>
<td>NI: 142</td>
<td></td>
<td></td>
<td>0.1</td>
<td>0.2</td>
</tr>
<tr>
<td>UK: 1,468</td>
<td></td>
<td></td>
<td>1.1</td>
<td>2.2</td>
</tr>
<tr>
<td>CO E: 13,863</td>
<td>-</td>
<td>-</td>
<td>1.4</td>
<td>-</td>
</tr>
<tr>
<td>W: 2,144</td>
<td></td>
<td></td>
<td>-</td>
<td>0.019</td>
</tr>
<tr>
<td>S: 2,582</td>
<td></td>
<td></td>
<td>-</td>
<td>0.003</td>
</tr>
<tr>
<td>NI: 1,984</td>
<td></td>
<td></td>
<td>-</td>
<td>0.003</td>
</tr>
<tr>
<td>UK: 20,574</td>
<td></td>
<td></td>
<td>-</td>
<td>0.029</td>
</tr>
<tr>
<td>Carbon equivalent (for CO₂) E: 325,228</td>
<td>-</td>
<td>35</td>
<td>70</td>
<td>140</td>
</tr>
<tr>
<td>W: 50,307</td>
<td></td>
<td></td>
<td>11.4</td>
<td>22.8</td>
</tr>
<tr>
<td>S: 60,584</td>
<td></td>
<td></td>
<td>1.8</td>
<td>3.5</td>
</tr>
<tr>
<td>NI: 39,322</td>
<td></td>
<td></td>
<td>2.1</td>
<td>4.2</td>
</tr>
<tr>
<td>UK: 475,441</td>
<td></td>
<td></td>
<td>1.4</td>
<td>2.8</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>16.6</td>
<td>33.2</td>
</tr>
</tbody>
</table>

E: England; W: Wales; S: Scotland; NI: N. Ireland; UK: United Kingdom

Notes:
* Includes emissions from non-agricultural horses. ^ Central estimate is the average of lower and upper bounds.
5.2.3 Soil

As reported in Pretty et al (2000) and EA (2002), Evans (1996) estimates on the basis of data from local authorities, that annual damage in the UK from off-site soil erosion is approximately £9.2 million (updated to 2003 prices). This figure includes damage to roads, footpaths, property, and mostly significantly channel degradation from soil erosion of all causes. Using the linking data from the Environment Agency (2002) which suggests that 95% of all off-site soil erosion is caused by agriculture, we can attribute 95% of that cost to agriculture. Table 5.4 shows this calculation to result in a cost of £8.8 million per year (=0.95 x £9.2 million). Detailed calculations can be found in Annex 6 (worksheet A6.3 - Soil). Clearly this is an average estimate, as erosion levels will vary considerably between years.

<table>
<thead>
<tr>
<th>Impact from Agriculture</th>
<th>Physical Data (α)</th>
<th>Adjustment factor (p)</th>
<th>£ per unit (£,2003)</th>
<th>Total (£ million)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cost of off-site soil erosion (cost of dredging streams, etc)</td>
<td>0.95 (95% damage due to agric.) (UK)</td>
<td>-</td>
<td>9.2m</td>
<td>8.8</td>
</tr>
</tbody>
</table>

5.2.4 Landscape, habitats and species

The stock of land held by agriculture represents the majority of land in the UK (Haines-Young et al, 2000). This environmental asset provides a flow of services in the form of landscape amenity and habitat and species provision. Certain agricultural practices are essential for the survival of designated UK habitats and species, while other practices can be damaging. In accounting for the services provided by the land managed or impacted by (and hence attributable to) agriculture in any one year, four sets of physical data are used:

- **Stock of broad habitat types** considered to be managed by agriculture (e.g. arable lands, heathland, grassland, etc), differentiated by:
  - Land designated as Site of Special Scientific Interest (SSSI) (data available for England only) and
  - Land not designated as such (Non-SSSI) (data available for England only)
- **Stock of linear features** managed by agriculture (e.g. hedgerows, stone walls);
- **Farm woodland**
- **Populations of keystone species** related to agriculture (in this case, farmland bird population data are used).

The use of the above reflects the availability of physical and economic data, and the potential to match the two. While the ELF model (IERM and SAC, 2001) provides estimates per hectare that can be applied to broad habitat classification type, allowing the landscape value of most of the land area under agricultural management to be accounted for. These values are summarised in Table 4.4 and repeated in Table 5.5 below as part of the calculations. Per hectare values range from £21 to £135 per hectare, depending on habitat type. For the total annual value of landscape services from each of these habitat types see Table 5.5.

Separate economic research on the value of SSSIs (Willis et al, 1990) provides monetary data for ‘special’ sites within this stock of land (£718 per hectare as reported in Section 4). The data provided on the state of SSSIs in England by English Nature (2003) allow these economic value estimates to be employed to provide a fuller picture of environmental value and agriculture’s contribution in England. The use of Willis et al (1990) is
interpreted as providing an estimate for the provision of biodiversity services (or habitats and species services in the language used in the accounting framework). While the ELF model is the source used for values for landscape amenity there is likely to be an element of habitat value encapsulated within any valuation of landscape.

In the interpretation of physical data from the English Nature report, it has been assumed that the proportion of SSSIs under agricultural management which are in unfavourable condition provide no biodiversity services, but provide landscape services, and thus these hectares are valued using the ELF study estimates (see estimates in Table 4.3). The biodiversity services provided by those in favourable condition are attributed to agriculture. These total £230 million to England and Wales in 2003. Note that while accounting for habitats in this way will result in a positive adjustment to the income accounts, any losses in hectares of SSSIs or degradation in quality of SSSIs will become evident when comparing the income from these assets in the current accounting period with those in the previous accounting period (conversely, any gains will clearly be evident in this way as well). While this provides a manageable approach and a useful starting point for the framework environmental accounts, a better handle on the true extent to which agriculture is the provider of all of these benefits is required for these calculations to better reflect reality.

The ELF model results are also used to value the sum of linear features and the value of woodlands thought to be under agricultural management. Data on linear features comes from the Countryside Survey 2000 (as does the data on broad habitat type). The total value of linear features in the UK amounts to a benefit of about £2 million per year (=£21 per ha x 21,340 hectares).

However the data on woodlands under farm management are taken from the farm woodland scheme data due to the difficulty of establishing the extent to which all woodlands under the broad habitat classification were on agricultural land. It is likely that the data on woodlands is an underestimate of total area of woodland managed as an inherent part of agricultural enterprises.

Economic data provided by Foster et al (1998) are used to calculate the economic importance of farmland birds, and agriculture’s role in providing habitats for them. The health of farmland bird populations is an important indicator of overall farmland biodiversity and hence the economic quantification of this indicator is a useful addition to the accounts. As with the other adjustments calculated here, the value of farmland birds will appear as a positive income adjustment to the accounts. This reflects the baseline of ‘no agriculture’ and the role that agricultural systems have in providing for these birds. However, if the accounting exercise were repeated for the year 1970, then the comparison of the service flows provided by this adjustment in the two periods would reveal a steep drop in economic value of this asset (and by inference a loss in wealth and income to the sector).

Table 5.5 illustrates the calculations used to arrive at aggregate estimates of the value of land-based environmental service flows attributable to agriculture. More detailed calculations can be found in Annex 6 (worksheet A6.4 - Land, habitats & species).

---

25 As mentioned previously in this report, when applying economic valuation data to land based amenities, it is usually the case that all services generated by one type of land (or one specific site) are valued as a bundle of goods - so that WTP per hectare reflects all these aspects of the environment. Thus, results from the ELF model, when applied to broad habitat types will typically include valuations for both use (recreational) and non-use (existence value) of the flora and fauna found on those sites, as well as for the landscape amenity.
<table>
<thead>
<tr>
<th>Impact from Agriculture</th>
<th>Physical Data (hectares) (α)</th>
<th>Adjustment factor (p)</th>
<th>£ per unit (£, 2003)</th>
<th>Total (£ million)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Lower bound (β_L)</td>
<td>Central (β_C)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Value of landscape amenity services by the current provision of landscapes (within the agricultural sector)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Linear features (incl. hedgerows, field margins, banks) a</td>
<td>E&amp;W: 7,000</td>
<td>-</td>
<td>-</td>
<td>21</td>
</tr>
<tr>
<td></td>
<td>S: 1,240</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>NI: 4,100</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>UK: 12,340</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

Value of habitat and species protection services provided by current land-use (within the agricultural sector) - non SSSI b

<table>
<thead>
<tr>
<th>Impact from Agriculture</th>
<th>Physical Data (hectares) (α)</th>
<th>Adjustment factor (p)</th>
<th>£ per unit (£, 2003)</th>
<th>Total (£ million)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Lower bound (β_L)</td>
<td>Central (β_C)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Neutral Grassland</td>
<td>E&amp;W: 411,000</td>
<td>-</td>
<td>-</td>
<td>84</td>
</tr>
<tr>
<td></td>
<td>S: 168,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>NI: 254,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>UK: 833,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Bog</td>
<td>E&amp;W: (see SSSIs)</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>S: 2,038,000</td>
<td>-</td>
<td>-</td>
<td>92</td>
</tr>
<tr>
<td></td>
<td>NI: 148,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>UK: 2,186,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Dwarf Shrub Heath</td>
<td>E&amp;W: 399,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>S: 1,002,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>NI: 13,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>UK: 1,414,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Acid Grassland</td>
<td>E&amp;W: 522,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>S: 748,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>NI: 28,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>UK: 1,298,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

E: England; W: Wales; S: Scotland; NI: N. Ireland; UK: United Kingdom
### Table 5.5 continued: Landscape, Habitats and Species - Welfare Impacts on Society

<table>
<thead>
<tr>
<th>Impact from Agriculture</th>
<th>Physical Data (hectares)</th>
<th>Adjustment factor (p)</th>
<th>£ per unit (£, 2003)</th>
<th>Total (£ million)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Lower bound (β&lt;sub&gt;L&lt;/sub&gt;)</td>
<td>Central (β&lt;sub&gt;C&lt;/sub&gt;)</td>
<td>Upper bound (β&lt;sub&gt;U&lt;/sub&gt;)</td>
</tr>
<tr>
<td>Fen, Marsh and Swamp</td>
<td>E&amp;W: 210,000</td>
<td>-</td>
<td>-</td>
<td>92</td>
</tr>
<tr>
<td></td>
<td>S: 337,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>NI: 53,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>UK: 600,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Calcareous Grassland</td>
<td>E&amp;W: (see SSSIs)</td>
<td>-</td>
<td>-</td>
<td>84</td>
</tr>
<tr>
<td></td>
<td>S: 27,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>NI: 1,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>UK: 28,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Farm Woodland</td>
<td>516,000 (UK)</td>
<td>-</td>
<td>-</td>
<td>135</td>
</tr>
</tbody>
</table>

#### Value of habitat and species protection services provided by current land-use (within the agricultural sector) - SSSIs in favourable condition

<table>
<thead>
<tr>
<th>Impact from Agriculture</th>
<th>Physical Data (hectares)</th>
<th>Adjustment factor (p)</th>
<th>£ per unit (£, 2003)</th>
<th>Total (£ million)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Lower bound (β&lt;sub&gt;L&lt;/sub&gt;)</td>
<td>Central (β&lt;sub&gt;C&lt;/sub&gt;)</td>
<td>Upper bound (β&lt;sub&gt;U&lt;/sub&gt;)</td>
</tr>
<tr>
<td>Arable</td>
<td>13,818 (E&amp;W)</td>
<td>-</td>
<td>-</td>
<td>718</td>
</tr>
<tr>
<td>Neutral Grassland</td>
<td>36,351 (E&amp;W)</td>
<td>-</td>
<td>-</td>
<td>718</td>
</tr>
<tr>
<td>Bog</td>
<td>86,304 (E&amp;W)</td>
<td>-</td>
<td>-</td>
<td>718</td>
</tr>
<tr>
<td>Dwarf Shrub Heath</td>
<td>86,331 (E&amp;W)</td>
<td>-</td>
<td>-</td>
<td>718</td>
</tr>
<tr>
<td>Acid Grassland</td>
<td>25,307 (E&amp;W)</td>
<td>-</td>
<td>-</td>
<td>718</td>
</tr>
<tr>
<td>Fen, Marsh and Swamp</td>
<td>24,750 (E&amp;W)</td>
<td>-</td>
<td>-</td>
<td>718</td>
</tr>
<tr>
<td>Calcareous Grassland</td>
<td>41,000 (E&amp;W)</td>
<td>-</td>
<td>-</td>
<td>718</td>
</tr>
</tbody>
</table>

#### Value of habitat and species protection services provided by current land-use (within the agricultural sector) - SSSIs in unfavourable condition

<table>
<thead>
<tr>
<th>Impact from Agriculture</th>
<th>Physical Data (hectares)</th>
<th>Adjustment factor (p)</th>
<th>£ per unit (£, 2003)</th>
<th>Total (£ million)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Lower bound (β&lt;sub&gt;L&lt;/sub&gt;)</td>
<td>Central (β&lt;sub&gt;C&lt;/sub&gt;)</td>
<td>Upper bound (β&lt;sub&gt;U&lt;/sub&gt;)</td>
</tr>
<tr>
<td>Neutral Grassland</td>
<td>14,249 (E&amp;W)</td>
<td>-</td>
<td>-</td>
<td>84</td>
</tr>
<tr>
<td>Bog</td>
<td>99,696 (E&amp;W)</td>
<td>-</td>
<td>-</td>
<td>92</td>
</tr>
<tr>
<td>Dwarf Shrub Heath</td>
<td>125,569 (E&amp;W)</td>
<td>-</td>
<td>-</td>
<td>23</td>
</tr>
<tr>
<td>Acid Grassland</td>
<td>16,593 (E&amp;W)</td>
<td>-</td>
<td>-</td>
<td>84</td>
</tr>
<tr>
<td>Fen, Marsh and Swamp</td>
<td>5,250 (E&amp;W)</td>
<td>-</td>
<td>-</td>
<td>92</td>
</tr>
<tr>
<td>Calcareous Grassland</td>
<td>13,900 (E&amp;W)</td>
<td>-</td>
<td>-</td>
<td>84</td>
</tr>
</tbody>
</table>
### Table 5.5 continued: Landscape, Habitats and Species - Welfare Impacts on Society

<table>
<thead>
<tr>
<th>Impact from Agriculture</th>
<th>Physical Data (α)</th>
<th>Adjustment factor (p)</th>
<th>£ per unit (£,2003)</th>
<th>Total (£ million)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Lower bound (βₗ)</td>
<td>Central (βₖ)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Lower bound</td>
<td>Central</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>5.1 m²</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>307.3</td>
<td>-</td>
</tr>
<tr>
<td>Farmland bird population index</td>
<td>60 (current index value) (UK)</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

**Value of habitat and species protection services provided by current land-use (within the agricultural sector)**

- **Species**

**Notes:**

- Total of stock of linear features reported in Table 3.6 converted to per hectares terms (on basis that 1 ha = 1 km², and assuming that hedgerows are on average 1 m in width). Total stock is calculated from hedges, remnant hedges, line of trees/shrubs/relict hedge, bank/grass strip. See Annex 6.3 for a breakdown by each type of linear feature.

- Broad habitat types as reported in Table 4.4. Note that data for improved grassland and arable and horticultural land types are not presented since no monetary value is applied to these land types, except where SSSIs fall within these land types.

- Value per farmland bird population index point, based on analysis of Foster et al (1998). See Annex 6.3 (Box 1) for derivation of this value.
5.2.5 Waste

Table 5.6 presents the calculation for aggregating the external cost of waste from agriculture. As outlined in Section 4, an estimated cost of £15 per tonne of waste is used in this report (derived from COWI, 2000). From Section 3, a total of 1 million tonnes of general waste from agriculture are identified as being appropriate for inclusion in the accounts. This results in a cost for agriculture of £15 million per year (=£15 per tonne x 1 million tonnes). This calculation can also be seen in Annex 6 (worksheet A6.5 - Waste).

Note this will underestimate the environmental costs of waste, as physical data on organic wastes is not included here.

<table>
<thead>
<tr>
<th>Impact from Agriculture</th>
<th>Physical Data (tonnes)</th>
<th>Adjustment factor (p)</th>
<th>£ per unit (£, 2003)</th>
<th>Total (£ million)</th>
</tr>
</thead>
<tbody>
<tr>
<td>General waste</td>
<td>1,000,000 (UK)</td>
<td>-</td>
<td>15</td>
<td>15</td>
</tr>
</tbody>
</table>

5.3 Aggregating Economic Values of Environmental Impacts

Table 5.7 presents estimates of the total costs and benefits of the environmental impacts of agriculture. The table is complete only so far as the available physical and economic data allow. Where cells are shaded grey this indicates missing country level data, and where cells are merged this indicates that the data cover more than one country (e.g. it is often the case that only UK level data are available). Notes to the table indicate some overlap in values calculated for habitats and species. Elsewhere in the table distinctions are made between coastal and inland water pollution, local and global air pollution, and between sectors upon which water clean up costs are imposed (i.e. water companies for drinking water clean up, and the government - through the Environment Agency - for restocking of fish following pollution incidents). Note that monetary values presented in the table have been rounded up or down, which might mean that the totals of the rounded estimates do not equal the rounded totals.

Interpretation of the impact-specific results following the table points to the reasons behind the numbers including what is not in these numbers. These gaps are further discussed in Section 6.
Table 5.7: Estimated Monetary Adjustments to Agricultural Accounts (£million 2003, central estimates)

<table>
<thead>
<tr>
<th>Impact Category</th>
<th>Accounting Adjustment</th>
<th>England (E)</th>
<th>Wales (W)</th>
<th>Scotland (Sc)</th>
<th>N. Ireland (NI)</th>
<th>UK</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Adjustments for Welfare Impacts on Society:</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>I. Water</td>
<td>Value of water pollution arising from agricultural production</td>
<td>Inland: £48</td>
<td>£1</td>
<td>£14</td>
<td>£7</td>
<td>£71</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Coastal: -£3</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Value of agricultural water abstraction</td>
<td>Global: -£543</td>
<td>-£109</td>
<td>-£143</td>
<td>-£95</td>
<td>-£889</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Regional/Local: -£43</td>
<td>-£7</td>
<td>-£10</td>
<td>-£7</td>
<td>-£67</td>
</tr>
<tr>
<td>II. Air</td>
<td>Value of air pollution arising from agricultural production</td>
<td>Habits: +£225</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Species: +£307</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>III. Soil</td>
<td>Value of (net) soil erosion on-farm on future yields</td>
<td>n/e</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>IV. Landscape*</td>
<td>Value of landscape amenity services by the current provision of landscapes (within the agricultural sector)</td>
<td>Gov’t: -£2</td>
<td>-£0.1</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Water Company: -£181</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Costs of flooding</td>
<td>-£153</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Included in measure above</td>
<td>-</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Cost of off-site soil erosion</td>
<td>-£9</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>V. Habitats and Species*</td>
<td>Value of habitat and species protection services provided by current land-use (within the agricultural sector)</td>
<td>(+) e.g. Value of landscapes to tourism</td>
<td>n/e</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>VI. Waste</td>
<td>Value of waste pollution and disamenity arising from agricultural production</td>
<td>None</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>VII. Nuisance</td>
<td>Value of noise and odour disamenity arising from agricultural production*</td>
<td>n/e</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

* These two categories overlap, which means that some element of habitat and species will be captured in the Landscape valuations, and some element of landscape value will be captured in the habitat valuations.  

n/e: not estimated due to lack of physical and/or economic data  

Water: 

Table 5.7 shows that the largest cost of agriculture in terms of water quality and quantity is its share in the cost of flooding. This is also the least certain of the impact categories depending entirely on linking data that merits further research. Another interesting observation of the table is that the productivity impacts on other sectors (using market data) are higher than welfare impacts on society (using non-market data). This is partly because of the incompleteness of reliable economic data on the impact of agriculture on marine ecology (currently the adjustment for coastal pollution reflects only impacts on human health and wellbeing from faecal contamination). Nonetheless, the results point to significant gains in terms of both improvements to social welfare and economic gains to other sectors if water pollution from agriculture can be further reduced.  

In terms of abstraction costs, agriculture accounts for 14% of total water resource use by industry, which provides perspective on these costs also. The economic data employed uses expert judgement to assess the environmental impacts of abstraction for different changes in flow to apply benefits transfer. The
environmental impacts were specific to South East England, and thus could be over or underestimating the total costs when these are applied across the whole UK.

Air:

The range of air pollution costs presented here is high compared with £585 million estimated by Hartridge and Pearce (2001) and £393 million estimated by Pretty et al (2000). This increase is partly due to updated unit damage cost estimates and partly due to a larger set of pollutants and impacts being valued. Among the pollutants, the greatest impact in monetary terms appears to arise from nitrous oxide emissions. This result is driven by the relatively high unit costs for N₂O used in the analysis. The second largest contribution to the total cost for the UK comes from methane again due to the high economic cost, level of emissions and global warming potential. In terms of the monetary value of impacts, carbon and ammonia have fairly similar aggregate value for the UK.

As Table 5.7 shows the majority of the cost of air pollution reflects the global damages of greenhouse gas emissions. While £889 million represents a sizeable cost it is worth keeping in mind that agriculture accounts for only 7% of the UK’s greenhouse gas emissions. Thus if the same exercise is undertaken for national accounts, £889 million would be dwarfed by total costs. If all sectors undertook accounting exercises of this type then it would be useful to compare how the costs of agricultural greenhouse gas emissions compared with other sectors. By introducing information on the relative cost of reducing these emissions between sectors, the cost created by agriculture can be further put into perspective. Essentially, if the benefit cost ratio of reducing emissions in agriculture is lower than in other sectors, some action may be warranted.

While these costs appear large, it is worth pointing out that they are still likely to be underestimates, as a number of the impacts of air pollution are not included in the economic data applied here due to a lack of research. As described in Section 4 these impacts include: impacts on ecosystems, damage to cultural heritage, effects of ozone on materials and additional health effects.

Another missing element from the air impact category is the economic value of the impacts of dust and allergens on human health. Without economic data, or adequate physical data, it is hard to say what the extent of this gap is. Therefore, further research is warranted.

Soil:

The cost figure presented in Table 5.7 refers to the off-farm impacts of soil erosion on other sectors. The on-farm impacts of soil erosion on the current yields are already reflected within income in the existing sector accounts, while the impacts of soil erosion on future yields are not captured in the existing accounts and not possible to quantify here.

Landscape, habitats and species:

Large positive service flows are attributed to agriculture through the provision of landscapes, habitats and species. The approach employed reflects the extensive coverage of agriculturally managed land in the UK (comprising 74% of all land) and the reliance of many habitats and species on the continuation of certain agricultural management practices. Changes in these practices have led to the decline of farm species (such as birds) and the degradation of semi-natural habitats. These negative impacts would appear in the accounts with the availability of time-series data which would show falling incomes from these environmental services over time.

The scale of the benefit accrued from agricultural practices that maintain landscapes, habitats and species valued by society, can be compared with the payments made to farmers to provide these services. While the data only allows a crude comparison, total payments to agri-environment schemes totalled £387 million in 2002 (Defra et al, 2004), which is far outweighed by the magnitude benefits recorded in Table 5.7. This points to the potential usefulness of collecting annual data on benefits to compare with annual payments. Currently the data on the physical environment is the most limiting factor in achieving this objective.
Waste:

Waste from agriculture amounted to approximately 22% of total waste produced in the UK (1998/99 figures). Of the agriculture's share, 92.5 million tonnes is animal and vegetable waste and 1 million tonnes is general waste. As the adjustments presented here only account for the general waste category, the total disamenity value of waste produced by agriculture is likely to be underestimated to a great extent.

As Section 2 points out, the landfill tax will ‘internalise’ to some extent the environmental costs of waste disposal. This doesn’t change the fact that the environmental costs per tonne of waste remain the same regardless of whether they are paid for. However, if the tax is set at the right level then the amount of waste produced may be ‘socially optimal’. This is unlikely to be the case, but this discussion illustrates the point made throughout this report, namely that the role of the accounts is not to record externalities in reference to any policy, but to record the value of current environmental impacts. Divorcing the value of the impact from any predetermined policy optimum increases the transparency of the accounts and the policy usefulness of the data. On the latter point, information on the costs of environmental impacts of waste disposal, in the presence of a tax, can be used to evaluate whether the policy itself is socially optimal. In addition it means that the original goals of environmental accounting can be met, i.e. to give more accurate measures of income and wealth.
6 Gap Analysis

This Section starts with presenting the objectives and overview of gap analysis. It then pulls together the lessons learnt from the study in terms of physical (Section 6.2) and economic (Section 6.3) data gaps together with ongoing work and future research proposals and priorities. Further detail on the physical data availability and the coverage of the economic literature can be found in Annex 3 and 4 and Annex 5, respectively. The ideas for future research presented here are pointers only and do not fully develop technical specifications for potential new work.

6.1 Objectives and Overview of Gap Analysis

The relationship between agriculture and the environment is complex and this complexity is reflected in the large number of different indicators (both physical and economic) required to understand it. Therefore, this study is the first attempt of this scale to bring together all the relevant physical and economic data about the impacts of agriculture on the environment. In fact, including analysis of data gaps as one of the objectives of the study acknowledges this.

Six types of gaps present themselves from the analysis undertaken for this study. These are gaps related to: (i) the accounting framework and sustainability indicators; (ii) the scope of the analysis; (iii) the geographical disaggregation of the accounts; (iv) the sub-sector disaggregation of the accounts; (v) the annual nature of the accounts and (vi) data, both physical and economic.

The accounting framework presented in Section 2 and Annex 2 does not suffer from any gaps in theoretical development. The gaps relate mostly to the challenges faced in putting that framework into practice. While Section 5 does not attempt to produce any accounting aggregate (e.g. green Net National Product for agriculture) or sustainability indicator (e.g. genuine savings), this omission is due to incomplete coverage of physical and economic data rather than the shortcomings of the framework. In fact, the estimates of the individual environmental impacts presented in Section 5 can be used to adjust income accounts for the sector, if it is so desired. While the coverage of environmental impacts may be limited, even this limited analysis is likely to be an improvement over the current presentation of some of the physical data alone.

The scope of this study, in terms of the definition of the environmental impacts, is limited to the farm gate. Pretty et al (forthcoming) looks at the impacts outside the farm gate, such as the environmental impacts of transport of goods. Those impacts should be analysed further in terms of their suitability for inclusion in the environmental accounts for agriculture. On the other side of the coin, there are the impacts other sectors may have on the productivity of agriculture. For example, potential impacts of air pollution on crop yields which are already included in the sector accounts, but not explicitly separated.

The scope of this accounting exercise is the agricultural activities that take place within the UK and their local, national or global environmental impacts. One policy interpretation of this could be to reduce, say, the number of cattle raised in the UK and increase the imports of the related products in order to reduce the emissions of UK greenhouse gases. While this may appear as a reduction of emissions in the UK accounts, the overall global impact would not be reduced. In order to see this effect, an accounting exercise linked to consumption rather than production would be required. This is currently considered beyond the scope of any national or sectoral accounting exercise. However, further work on the environmental impacts that are associated with the consumption (including import) of agricultural products could be of interest for other policy areas.

The objectives of the study require geographical disaggregation (at the country level) of the environmental impacts of agriculture. This has been done in Section 5 as much as the available data allow. The main limiting factor here is the absence of physical data collected at or broken down by region or country. Therefore, this issue is further discussed in Section 6.2 on gaps in the physical data.

The objectives of the study also require that the study explore the sensitivity of the analysis to different agricultural practices and changes to these. The sensitivity to different practices and related to this the sub-sector disaggregation of the accounts are again limited by the availability of physical data and further discussed in Section 6.2. In fact, OECD (2001) also comments that we need a greater
understanding of the links and trade-offs between the changes in agricultural practices, farm input use and environmental impacts.

Currently both the main sectoral accounts for agriculture and the overview of its environmental accounts are prepared and published on an annual basis. The intention has been that the environmental accounts in monetary units would also be repeated on an annual basis. The main limiting factors here are the complexities of the environmental impacts and the availability of physical data. Some environmental impacts such as air-borne emissions can easily be measured on an annual basis, while others, such as pollution of surface and groundwater and impacts on species of flora and fauna may take many years to be detected even if annual measurements are taken. For other impacts, such as linear landscape features, the necessary surveys are simply not undertaken on an annual basis but can be approximated to intervening years. However, annual environmental accounts are still possible so long as the year of the data is made clear, and understood.

As highlighted earlier in the report, there are numerous impacts for which physical and economic data are lacking. These gaps on the whole arise as a result of the following reasons:

- There may be no physical or economic data about an environmental impact. For example, there is no information about the effect of current on-farm soil erosion on future crop yields. This is the type of gap that requires the most effort to fill, as it typically requires long term research to establish environmental pathways and processes with any certainty. On the positive side, the efforts on filling this type of gap may benefit other policy analyses as well that of agriculture.

- There may be physical data about an environmental impact but the share of agriculture as one of the sources of that impact is not known. In other words, the linking data that exist for some impacts do not exist for others. For example, we may argue that agricultural landscapes have a positive impact on tourism sector. While there are market data about the economic value of the tourism sector, we do not know the contribution agriculture makes to this. However, we should also be aware of the limits to how specific such linking data could be. While there may exist a better understanding of some linkages at the local level, these may not be possible to extrapolate to the national level, which is the appropriate level of analysis for the purposes of the environmental accounts.

- There may be physical data about an environmental impact and the share of agriculture as one of the sources of that impact may also be known but there may be no economic valuation data. In fact, as Annexes 3 and 4 show there is a potentially large list of physical data that could not be used in the study since the corresponding economic data were lacking. For example, intensive agricultural land hasn’t any economic value currently assigned to it, but may provide benefits although they are likely to be smaller than for extensive landscapes. This type of gap sends signals to economists in planning and undertaking future research.

While it is relatively easy to produce a long list of missing data, it is more useful to prioritise these. Since priorities depend largely on the priority criteria applied, these criteria should be selected carefully. In the context of this study, the relevant criteria might be the purposes for which environmental accounts for agriculture are required (see Section 1). Among these, the assessment of the extent to which agriculture contributes to society’s wellbeing and welfare, is the most relevant in terms of setting priorities for future research. This, in turn, points to gaps that could be filled in the first instance, i.e. where physical impact and economic value data exist but agriculture’s share in the impact (positive or negative) is not known. Although it may be that economic data is easier to gain than new physical data or greater scientific understanding.

Another approach to prioritisation would be on the basis of the (economic or other) importance of the environmental impact. For example, a negative impact on biodiversity (especially at an irreversible level) can be deemed to be more important than nuisance impacts from agricultural processes that are short-term and reversible. This also implies that a higher effort should be expanded to fill gaps in biodiversity impacts. This can then be used as an input for policy design and appraisal. However, caution should be exercised in determining importance for impacts for which we lack scientific understanding and physical and economic data since gaps and uncertainties in this case amount to unknowns. It may be best for such impacts to be priority simply because we do not know anything about them, even if this requires a long term further research programme. The next two sections are prepared with these concerns in mind.
6.2 Gaps in the Physical Data

Among the shortcomings of physical data we can count the following: (i) the time period of data; (ii) geographical disaggregation of data and (iii) sectoral disaggregation of data.

Time period of data: Agricultural accounts are produced on an annual basis but it is less clear how far environmental change attributed to agriculture can be measured with sufficient reliability on an annual basis at reasonable cost. Whilst pressures do change over time, some such as the neglect of landscape features or the decline of soil organic content might be recorded more usefully at longer intervals (e.g. three, five or ten years). Some of the variations which occur between years are attributable to the weather, rather than to any change in the underlying agricultural activity as when the timing and extent of rainfall alters the impact of cultivation on soil structure or the effects of irrigation on aquifer depletion. Consequently, impacts and costs may fluctuate around a trend which may not need to be measured annually. In any case, the majority of data sets are not collected on a regular basis meaning that it will be difficult to complete a yearly account without further commitments to data collection.

Geographical disaggregation: Some of the data are collected at a number of localities and then, if possible, aggregated to the national level. This does not always generate nationally representative datasets. On the other hand, other data are collected at the national level but cannot be disaggregated even at the country level. The importance of having geographically disaggregated data shows itself more in designing and implementing country level agricultural policies and for comparative analysis of the countries of the UK. Data for some impacts such as air emissions, however, are likely to be more accurate at the national level.

An issue is that there are also some inconsistencies between the way physical data are collected in different parts of the UK and across time. For example, data for Scotland does not necessarily match the categories used in, say, England. This means that it becomes difficult to aggregate information. There is also the issue of one off reports, i.e. where there is no consistent programme of work to collect data regularly on a UK-wide basis. This can mean that information may be available for, say, England and Wales in the form of Environment Agency reports, but this is not replicated for Scotland and Northern Ireland.

One recommendation for future updates would be to have designated contact points within each major body which collates data, i.e. Defra, SEPA, the Environment Agency, Welsh Assembly and DARNI who would be able to assist with the identification of the most up to date and appropriate data sets. This may also assist with improving consistency between data produced at the country level. It would also enable those updating the information to identify when new data sets are likely to be released, if these will enable gaps to be filled and avoid duplication of effort.

The rest of this section reviews the physical data gaps for each impact.

6.2.1 Water quality and availability

A variety of physical data is collected on diffuse water pollution from agriculture, including a programme of sampling pesticide concentrations in ground water. However, it is difficult to capture the overall level of diffuse water pollution in any one year, taking account of the different pollutants and pathways involved. In other words, more could be done to improve the reliability of linking data used especially for water quality. Environment Agency (2002) also mentions that there is a lack of information about the contribution of agricultural diffuse pollution to a range of negative environmental impacts. Information collected for the GQA databases of the Agency could also be reviewed to include suspended solids and some chemicals.

As the EU Water Framework Directive is implemented over the coming decade, the extent of information on the ecological quality of river basins will increase. But, this alone is unlikely to be sufficient to allow us to isolate the role of agriculture from other sources of diffuse pollution across the spectrum. To address this it might be useful to have an annual report on agricultural pollution of both fresh and marine water, bringing together both existing data collected by the Environment Agency and others and further sampling to complete a systematic view of diffuse pollution.
Some estimates of the contribution of agriculture to phosphate loading in water bodies is available but this is not sufficiently precise to allow us to calculate changes on an annual basis or to be clear about the exact proportion of phosphate attributable to farming practice in sensitive areas.

In terms of water availability, the quantity of water used by agriculture is well documented. However, what this means for surface and groundwater quantity and quality and associated impacts at the local or catchment level is not known. Considering that the relevant impacts are highly localised, the benefits transfer exercise that had to be undertaken in the absence of the relevant physical impact data is likely to be weak.

Finally, agriculture’s share in the economic cost of flooding is also based on a national level analysis using linking data. More up-to-date information on agriculture related flood events could be developed through the Environment Agency flooding database. In addition, the ongoing research on linking land use and catchment behaviour by the Environment Agency, Defra and University of Newcastle could provide more information on agriculture’s contribution to flooding. The extension of the physical impact analysis to local and catchment level would possibly benefit flood management policies and practice even more so than it would environmental accounts for agriculture.

6.2.2 Air

The data provided by Necten capture all emissions from agriculture, but further refinement of the measurement of greenhouse gas emissions and sinks would improve the accuracy of this data. Agriculture’s role as a sink should especially be further researched, given the potential for trading in carbon credits as well.

The main gap for this impact category refers to dust and allergens both in terms of the overall quantity and agriculture’s share in this total. Linking data could be used, though it is difficult to see what other physical indicator could be a surrogate for dust and allergens.

6.2.3 Soil

Various estimates are made of the severity of soil erosion and its off-farm impacts but a more regular survey would need to be undertaken in order to achieve a more reliable measure. Any survey would be constrained by costs and would need to be based on a sampling procedure, potentially focusing on those areas of known vulnerability to water or wind erosion.

Similarly there is a lack of reliable information about changes in soil composition over time. Two current concerns are the loss of organic content and the role of soils as carbon sinks. The latter is captured in the carbon emissions data provided by Necten as the emissions are reported net of sequestration but the assessments could be refined with regards to methodological issues as well as lack of monitoring.

Finally, while the on-farm impacts of soil erosion on the current yields can be measured and compared to a baseline of better soil quality, there are many unknowns about the implications on future yields. OECD (2001) states that little is known about how agricultural activities change soil’s biological processes and notes that there is a need to understand what determines the ‘sustainable’ use of soil resources by agriculture.

6.2.4 Landscape, habitats and species

Measurement of changes in habitat quality over time also presents methodological difficulties and issues of cost. However, it is an issue of growing importance not only on SSSIs but on other sites of conservation value. It is difficult to measure the extent of improvement and deterioration although both are occurring at any one time. One small change which would be helpful is to include in the Countryside Survey some questions on whether the different habitats surveyed were under agricultural management and, ideally, to attempt to measure qualitative change for these habitats.

Changes in agricultural land use, including transfers between arable and grass, could become more frequent following the Mid Term Review of the CAP and the new policy framework from January 2005. It is likely that a system for tracking changes in permanent pasture will have to be established as part of the cross-compliance requirements on the UK. This will provide some data relevant to the agricultural
accounts. However, the classification of agricultural crop cover used in the annual census is not ideal for measuring the contribution of farming to landscape maintenance. If the objective is to measure changes in broad landscape types over time some system of classification would be useful so that change can be measured at a broader scale in relation to agriculture in particular.

In order to get a better match between land use and landscape types and agricultural land use types, additional questions can be added to the annual agricultural census. These questions could relate to habitat types or simpler proxies that would allow this comparison.

The complexities of the relationships between agricultural practice, non-farming factors and the population dynamics of different species makes this a difficult area to establish definitive data. At present there is considerable reliance on the farmland bird indicators but these represent a relatively small proportion of the species found on farms. A more balanced picture could be developed if a wider range of species were subject to the same level of monitoring and reporting. Without underestimating the challenges involved there is a strong case for considering enhanced monitoring of a group of invertebrates and a group of plant species as well. These would lead to a better understanding of the impacts of agriculture on the diversity of species rather than on individual species. This could also mean that better use can be made of recent, and no doubt, future economic studies on the diversity of species. There have been some attempts to develop a Natural Capital Index which is calculated as the product of the quantity of the ecosystem multiplied by the quality of the ecosystem, i.e. the average of changes in wild species numbers from a baseline period (OECD, 2001).

Finally, all new work addressing these impacts should try to link habitats and species impacts to landscape types will simplify data collection and make a better match to economic analysis which considers these impacts as a bundle.

6.2.5 Waste

Data on waste arisings and disposal on agricultural holdings are not sufficient to allow a reasonable estimate of the scale of the problem or to measure changes over time. Although a large survey was recently conducted, the data are not presented in a format that can be applied to economic data. Given changes in the legislation a relatively authoritative survey of the problem based on a reasonably large sample would be extremely timely and could create a baseline from which further change could be measured.

In addition, positive environmental impacts of agricultural wastes used for energy generation (in terms of displaced pollution) could also be explored further.

6.3 Gaps in the Economic Data

As Section 4 summarises and Annex 5 details, there is a considerable economic literature in the UK (and in other countries) that consists of studies that could potentially be used for the purposes of environmental accounting for agriculture. However, a more detailed look at this literature shows that most of these studies are not relevant when measured against the study selection criteria listed in Section 4.

Due to the cost of original economic valuation studies and the broad range of impacts, benefits transfer is likely to remain the main economic valuation technique for environmental accounting for agriculture. The ELF benefits transfer model (IERM and SAC, 1999) could be taken as a template in determining the needs for new economic valuation studies. While the model is not comprehensive, it allows demand curves for the impacts covered to be drawn, while single economic valuation studies are likely to give point estimates. The advantage of having a demand curve is that it allows for estimating both the flow of services from a given level of environmental capital and the changes in the quality and quantity of this stock. Therefore, further economic research commissioned to fill the gaps in the economic data should be designed with the requirements of a benefits transfer model in mind.

Possibly the most important of these requirements is the context of the economic valuation studies. The policy or decision making contexts that are studied in most of the literature do not correspond to the policy context that is of concern here. Many of the existing studies are undertaken to address disparate research objectives rather than as part of a coordinated research effort to aid environmental, agricultural
Another issue involves for what the economic values are measured, i.e. maintaining the current services provided by the environmental assets, avoiding a negative change or securing a positive one. Most of the studies in the literature use a context of change and present an alternative scenario to the current situation. However, for the purposes of environmental accounting, the most relevant context is maintaining the current level of services and any future research should take this into account.

One way to formulate further valuation studies to fill the gaps is to address each gap or impact at a time. This has the advantage that the studies could be relatively small in scale and also of potential use to other sectoral or national policy developments. On the other hand, this would mean that, to some extent, the disparate nature of the current literature would continue. The problem with combining a large number of economic valuation studies on individual impacts of agriculture is that we may find that the sum of the individual impact cost or benefit estimates is greater than the total. This outcome was found in an experimental study in the context of air pollution (Bateman et al, 2002).

Therefore, the alternative approach would be a sector-wide economic valuation study that includes all (or at least all non-market) environmental impacts of agriculture. Both stated preference and revealed preference techniques could be used for this approach. For example, as part of the former, a choice experiment study could define agriculture as a bundle of goods and bads including crop and livestock production, environmental benefits and environmental costs. However, generating country level estimates, if this is desired, within a valuation study that also covers a large number of environmental impacts of agriculture could amount to an impossible (or impossibly expensive) task and hence more thought is required to develop this line of potential research.

Other impacts such as air pollution, carbon sequestration by soils and similar impacts that could be deemed too technical for a stated or revealed preference study will have to be studied separately.

Finally, most of the above discussion about the context also applies to market data. Gaps exist in terms of agriculture’s impacts on other economic sectors such as tourism and fisheries. Economic value-added and input-output studies can be considered to gather more information about these impacts. In the case of the interaction between agriculture and tourism, these can be augmented by surveys of visitors that explore the importance of agriculture (e.g. agricultural landscapes) in recreational decisions. Ongoing surveys such as the UK Leisure Day Visit survey could be augmented with questions of this kind. The resulting data could help generate market data as well as feed into a benefits transfer model.

6.3.1 Water quality and availability

This impact is the subject of the largest section of the currently available economic valuation literature. The main problem in quantifying these impacts in monetary terms is the lack of correspondence between the economic and physical data both in terms of the overall impact and agriculture’s share in this. For example, despite the large number of studies analysing the economic value of different river water quality levels, their use in the accounts has been limited since the quality levels covered in the economic studies do not provide an exact match for the river water quality index used by the Environment Agency. Any new work in this area, therefore, should use this index as the measures of quality as it is a good accounting measures and groups different environmental effects together even though additional information linking agriculture’s contribution to the overall quality is still required. Considering that the results of such a study could also be used in future for the Periodic Review process that water companies go through every five years, this new research becomes an important priority.

In terms of the environmental impacts of agriculture on marine waters, the analysis in this study is limited to the human health effects (and within that to mild stomach upset alone, even though this is the most significant impact). This scope is determined by the revisions to the Bathing Water Directive which are limited to the indicators for human health impacts. Therefore, it would be worth exploring the economic value of ecological impacts such as changes in marine flora and fauna.

In terms of the market costs, the costs imposed by agriculture on water companies require updating and further analysis. Companies may be required to meet standards stricter than the normal requirements of the Urban Wastewater Treatment Directive because of the poor nutrient status of receiving waters. For example, if these waters are classified as sensitive areas or if the catchment has been designated a
nitrate vulnerable zone, others in the catchment, particularly the agricultural sector may be responsible for excessive nutrient run-off and pollution of the waters. However, the operators of sewerage treatment works and their customers bear the burden of this outcome through the increase costs of effluent treatment. This problem could become more apparent if there is deterioration in nutrient status in rivers and more sensitive areas are designated. Understanding the relative contribution of different sectors to the nutrient load in waters and the associated costs of alleviating pollution should be examined for a variety of purposes, and could shed further light on the environmental accounting exercise.

6.3.2 Air

The existing economic valuation literature for air pollutants has developed over a substantial period of time and involved in-depth scientific and economic studies. Amongst the areas requiring further research are the impacts that are currently not included in the unit cost estimates (e.g. impacts on ecosystems, damage to cultural heritage, effects of ozone on materials and additional health effects).

There are no economic damage estimates for HCBs. This mainly reflects a lack of impact data or lack of quantifiable associations or causal links (e.g. through dose-response functions). However, given the small dosages of this pollutant it is not considered a high priority for research.

On the other hand, the economic cost of dust and allergens could possibly be estimated using estimates for medical expenditure, reduced productivity and WTP to avoid suffering. This would be a research project in itself.

6.3.3 Soil

The change in soil quantity and quality affects both the current and future use values. The current use values are easier to estimate using market price data for changes in crop yield or costs (or avoided costs) of off-farm soil erosion impacts. However, the change in future use values (option values) for soil quality and quantity is much more difficult to study and requires further thinking before it can be tackled through economic valuation. In fact, OECD (2001) also expresses the need to establish valuations of the benefits of higher farm productivity and benefits to the community generally of soil quality.

6.3.4 Landscape, habitats and species

There is a considerable literature on different landscape and habitats and some species. However, this literature, like many others, also suffers from the disparate nature of the economic research to date. One of the two major gaps has been the economic value of diversity of species. This gap is partially filled by the recent research for Defra (Christie et al, 2004). However, further analysis is needed to establish whether the results of that study could be used within the environmental accounts. The primary valuation research undertaken in this study could also be repeated for other parts of the country.

The other gap concerns the welfare impacts of the intensive agrarian landscape. The current literature concentrates on extensive agriculture such as the broad habitat types and the designated areas. Further research in this area could make use of carefully developed visual aids to allow a differentiation between different landscapes. Such visual aids were used by Willis (1994) for the Yorkshire Dales. The study showed respondents different landscapes and asked them to select their most preferred, after which their WTP to achieve and/or maintain that landscape was elicited. Extension of this study would be useful and could be used to update the ELF model.

A more recent attempt to generate visual aids related to the agricultural landscape is the work being undertaken by the Tyndall Centre at the University of East Anglia called ‘Developing and Visualising Landscape-scale Scenarios of Potential Climate Change Impacts’. The study also studies the landscape effects of different Common Agricultural Policy outcomes. These could be used to explore preferences for current landscapes as well as future ones. If a new study links landscapes to biodiversity then there would be economic data estimates that cover landscape, habitats and species altogether. The key in such new research would be to produce different value estimates for different levels of quality within each land type of landscape so that as landscapes, habitats and species quality and quantity change, the accounts could reflect these. How this could be undertaken within the accounting framework is already illustrated by the calculations for favourable and non-favourable SSSIs in Section 5.
The question of whether the benefits cultural and archaeological sites on agricultural land can be ascribed to agriculture (or to government policy) requires further thought since the answer depends on the characteristics of the site and the policy measure that is implemented. In any case, the literature on cultural heritage currently concentrates on unique sites and usually in a context of threat from air pollution, visitor damage and so on. Studies on more ‘common’ sites would be of useful both for heritage management policy and environmental accounts for agriculture. However, how important this impact as a priority for agricultural policy is currently not clear.

6.3.5 Waste

The environmental impacts of waste disposal have been studied in some depth and this research is currently been updated for Defra (Enviros and eftec, 2004 forthcoming). The same can also be said for the disamenity impacts of landfills. However, there is a gap in the UK literature about the disamenity impact of incinerators. This is partly due to the relatively small number of incinerators in the UK. However, this situation is likely to change in future and further work about the disamenity impacts of incinerators would benefit waste management policies as well as the preparation of environmental accounts for agriculture.

New research is required to cover the economic costs/benefits of disposing of organic wastes, which make up the majority of waste from agriculture.

6.3.6 Nuisance

Nuisance impacts include odour and noise. Odour has been studied in the context of landfill but it is bundled together with other nuisance (or disamenity) impacts such as litter, noise and so on. Noise has also been studied but mainly in the context of transport noise (e.g. rail, motorways and airports). Thus, the context captured in the currently available information does not match that of agricultural impacts.

The hedonic pricing approach can be used to value the benefit or cost of living near agricultural land. However, this would be likely to generate estimates for the bundle of nuisance impacts rather than individual ones. Therefore, surveys of the impact should use indicators that match what could be used in an economic study. The current impact data of number of complaints do not indicate the level of welfare impact since even those who make a complaint may not be disturbed sufficiently to express a positive willingness to pay to avoid the impact.

On the flip side, research could be undertaken on the economic value of peace and quiet provided by agriculture. The tranquility of the countryside, combined with the cleaner/er local air quality, open space and so on, have a measurable effect on human health which is evident in the increased life expectancy of those who live in rural areas. Expressing this increase in monetary terms would then be a straightforward task considering that the economic data on value of statistical life (or life year) exist.
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