Appendix B. Analysis of environmental impacts and their integration and communication

Final Report for Defra Project FO0419
Effective approaches to environmental labelling of food products

Agriculture and Environment Research Unit
Science and Technology Research Institute
University of Hertfordshire
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Executive Summary

The objective of Approach 2 was to identify the key issues and challenges associated with assessing, integrating and communicating environmental impacts within the context of a product label. This involved identifying environmental impact categories, sub-impacts and effects; the techniques available to measure and assess the impacts; the types of data and information available; the techniques available to communicate the impact and any meaningful baselines to place impacts in context.

A key theme throughout this analysis was the aggregation of information and data. Environmental assessments inherently involve large amounts of data that needs to be analysed, assessed and presented in a form to aid the original purpose of the assessment, which is usually to support decision making in one form or another. This typically involves taking all the information and data associated with the environmental effects of a product over its lifecycle and then through a process of analysis and aggregation distilling the findings into a small set of impact areas or categories.

A full description of the work undertaken and its findings together with the associated bibliography is included in this appendix. For completeness a brief description of the work is given below together with a summary of the key findings.

A key aspect of any environmental label is to determine which impacts should or could be covered. Some of the key issues such a label should include are information on greenhouse gases, energy use, waste, air and water quality impacts and water use. These are some of the key areas but there are potentially many more. The first section of this appendix introduces the range of impact areas which may need to be considered by an environmental label and different impact assessment techniques and approaches each which has their own process steps and phases. However, generally, any impact assessment process can split into the following sections or stages:

1. Stressors: These are the actual recorded, observed, calculated or modelled effects associated, in this instance, with the full life-cycle of a food product. They can include quantitative data such as emissions of pollutants and toxic substances, which would form part of the inventory phase of an LCA (LCI results), but can also include qualitative information.
2. Effects or mid-point impact categories: These are identifiable, and usually quantifiable, effects associated with the measurements (from stage 1) and usually involves some aggregation. For example, converting emissions of different greenhouse gases into a common measure of Global Warming Potential (GWP) using carbon dioxide equivalents. This often referred to as a process of impact characterisation.
3. Damage or endpoint impact categories: These are the end consequences of the effects identified in Stage 2 and this process aims to place the effect on a scale of damage caused, for example in terms of biodiversity this might be the Potentially Disappeared Fraction of a species or group of species as a consequence of an ecotoxicity effect identified in Stage 2, due to an emission of a toxic substance identified in Stage 1. This process may involve some further aggregation, for example combining all biodiversity impacts in stage 2 (e.g. ecotoxicity, eutrophication, acidification, etc.) in a single damage category of ecosystem health. This is often referred to as a process of damage characterisation.
4. Normalisation: This stage aims to place the impacts in context with respect to the contribution that the impact of the product has in relation the entire impact being observed. Normalisation can be applied to the outputs of either Stages 2 or 3. For example, emissions of greenhouse gases for a product could be compared to the total emissions of greenhouse gases for a region, e.g. Europe, and/or can be further normalised by examining this data on a per head of...
population – so in terms of a food product its GWP could be expressed as a percentage of the total GWP for an average person in Europe per year.

Many environmental assessments do not go beyond Stage 2. This is because there are inherent difficulties associated with damage characterisation and normalisation. There is a lack of established techniques and data which in combination with additional aggregation can make the assessment less transparent and more uncertain. However, being able to place any impacts in context, e.g. with meaningful baselines, and communicating those impacts, are important aspects within the context of an environmental label for food.

Section 2 of this appendix covers the common Stages of 1 and 2 and explores the measurement, assessment and characterisation techniques available. This following impact areas are considered in the report:

- Air quality
- Landscape and heritage
- Stratospheric ozone depletion
- Soil quality
- Biodiversity
- Noise, odours and dust
- Waste management
- Climate change
- Resource depletion
- Water quality

For each of these impact areas the background to the issues is summarised and the stressors and impact categories identified. The techniques available to measure and assess the stressors and impacts are then explored. This includes identifying the source of data that is used to describe the effects of a product during its lifecycle.

Section 3 of this appendix considers how environmental damage may be assessed and explores the scientific credibility of some of the more common approaches such as the ‘distance to target’ approach, the ‘damage function’ approach, environmental valuation techniques, panel and scoring methods and also discusses the metrics and aggregation techniques employed within the different approaches such as the Disability Adjusted Life Years (DALYs), Toxicity Equivalent Potentials (TEP) and Human Toxicity Potential (HTP) that are both used to communicate human health related impacts. Other aggregation techniques discussed include methods for evaluating and communicating ecological risks such as Toxicity Exposure Ratios, Potentially Disappeared Fraction (PDFs) and Potentially Affected Fraction (PAFs) and, with respect to Natural Resources how availability, scarcity and depletion may be measured and communicated. Section 3 also discusses issues relating to normalisation, data uncertainty and transparency. Finally a number of example techniques and models are considered including the Ecoscarcity method, the Dutch and Swedish Environmental Theme (ET) approach, the Environmental Priority Strategy technique, the EcoIndicator99 model and the USETox model.

The work undertaken in Approach 2 has shown that despite extensive research, development and application, and its importance to both government and industry, both the mid-term impact assessment stages and the final damage assessment and/or normalisation processes undertaken in Life Cycle Assessment have a great number of both practical and theoretical problems still to be resolved if they are to be successfully used in the context of an environmental label.

- With the exception of greenhouse gas emissions (BSI, 2008) there is a severe lack of scientifically credible techniques that have been tested, standardised and accepted by the scientific community.
• Methods that are in common use for damage assessment, characterisation and normalisation are prone to subjectivity, uncertainty and unjustified assumptions.
• The work undertaken has concluded that the use of LCA techniques within the context of an environmental label should be limited to the mid-point phase as, although desirable, damage characterisation is currently undeveloped.
• Communication of multiple environmental impacts that are scientifically complex on an environmental label would be very difficult and require aggregation and considerable simplification. Aggregation techniques themselves are highly complex and whilst they have been used in academic applications they are insufficiently tested and validated and have not been widely accepted by the scientific community to enable them to be used with confidence on an environmental label. Composite impacts are not transparent and there is the risk of losing data resolution.
• There is a lack of appropriate data available. In many instances the data necessary to measure and/or calculate mid-point impacts and/or damage cannot be measured easily, reliably or cost effectively. It is also subject to large uncertainties.
• There are issues of spatial and temporal resolutions to be resolved. Data sets are not harmonised in terms of their spatial or temporal characteristics. Environmental issues significant in one location may not be so important in a different location.

In conclusion, in the context of environmental labelling our opinion is that a considerable amount of scientific development and debate towards achieving standardised techniques for measuring and assessing environmental impacts is required before a true omni-label for food could be a reality, particularly if it was to be used for a statutory application. However, that is not to say that it is not possible to begin with some simple measures for a voluntary application particularly if the scheme and its associated label were designed to be flexible and expandable.
1.0. Introduction

1.1. Objectives

The objective of approach 2 is to identify the key issues and challenges associated with assessing, integrating and communicating environmental impacts within the context of a product label. This involves identifying environmental impact categories, sub-impacts and effects; the techniques available to measure and assess the impacts; the types of data and information available; the techniques available to communicate the impact including any scientifically sound and meaningful baselines in use that can help to place environmental impacts into context.

A key theme throughout this analysis is aggregation of information and data. Environmental assessments inherently involve large amounts of data. This data needs to be analysed, assessed and presented in a form to aid the original purpose of the assessment, which is usually to support decision making in one form or another. This typically involves taking all the information data associated with the environmental effects of a product over its lifecycle and then through a process of analysis and aggregation distilling the findings into a small set of impact areas or categories.

1.2. Impact areas

A key aspect of any environmental label is to determine which impacts should or could be covered. The Defra project specification document states that where possible an environmental label should include information on "greenhouse gases, energy use, waste, air and water quality impacts and water use". These are some of the key areas but there are potentially many more.

In determining impact categories it is important to understand and acknowledge the chain of linkages between causes and end impacts. This is often expressed using different terminology in different disciplines for example source-pathway-receptor (DETR, 2000; Environment Agency, 2004), stressors-midpoints-endpoints-damages (Bare & Gloria, 2008) or activity-effect-outcome (Tzilivakis et al., 2009a). There is a hierarchy of impact areas, impact categories and indicators and measures – a pyramid whereby information and data are collected, analysed and aggregated. In the context of an eco-label the impact areas are what is likely to be displayed on the product label or packaging and not necessarily specific, individual impact categories. For example, photo-oxidant formation (an impact category commonly used in LCIA) may fall under the Air quality on an environmental label, rather than be presented in its own right. Similarly, in many instances, impact categories will have sub-impacts or indicators. For example, under climate change there are greenhouse gas (GHG) emissions and carbon sequestration, and under GHGs there are specific emissions of CO₂, N₂O and CH₄, each of which can be used as a means of judging impact, but which are combined under the heading of climate change. It is also important to acknowledge that some commonly recognised impact categories are actually ‘midpoints’ (using LCIA terminology) rather than actual endpoint impacts. For example, there are many consequences of climate change including impacts on biodiversity, agricultural production and human health, but emissions of greenhouse gases is generally used as the measure of assessing that impact.

Given the complexities and amount of data involved, it is important to maintain transparency in any environmental assessment. There needs to be clear distinctions between different assessment phases and a clear understanding of the effects of aggregation, especially for communication purposes, and any uncertainty that this introduces.

Different impact assessment techniques and approaches have their own steps and phases. However, generally, any impact assessment process can split into the following sections or stages:
1. Stressors: These are the actual recorded, observed, calculated or modelled effects associated, in this instance, with the full life-cycle of a food product. They can include quantitative data such as emissions of pollutants and toxic substances, which would form part of the inventory phase of an LCA (LCI results), but can also include qualitative information.

2. Effects or mid-point impact categories: These are identifiable, and usually quantifiable, effects associated with the measurements (from stage 1) and usually involves some aggregation. For example, converting emissions of different greenhouse gases into a common measure of Global Warming Potential (GWP) using carbon dioxide equivalents (see Section 2.1). This often referred to as a process of impact characterisation.

3. Damage or endpoint impact categories: These are the end consequences of the effects identified in Stage 2 and this process aims to place the effect on a scale of damage caused, for example in terms of biodiversity this might be the Potentially Disappeared Fraction of a species or group of species as a consequence of an ecotoxicity effect identified in Stage 2, due to an emission of a toxic substance identified in Stage 1. This process may involve some further aggregation, for example combining all biodiversity impacts in stage 2 (e.g. ecotoxicity, eutrophication, acidification, etc.) in a single damage category of ecosystem health. This is often referred to as damage characterisation.

4. Normalisation: This stage aims to place the impacts in context with respect to the contribution that the impact of the product has in relation the entire impact being observed. Normalisation can be applied to the outputs of either Stages 2 or 3. For example, emissions of greenhouse gases for a product could be compared to the total emissions of greenhouse gases for a region, e.g. Europe, and/or can be further normalised by examining this data on a per head of population – so in terms of a food product its GWP could be expressed as a percentage of the total GWP for an average person in Europe per year.

Many environmental assessments do not go beyond Stage 2. This is because there are inherent difficulties associated with damage characterisation and normalisation (Bare et al., 2000; Jolliet et al., 2004; Pennington et al., 2004; Sleeswijk et al., 2008; UNEP, 2003). There is a lack of established techniques and data which in combination with additional aggregation can make the assessment less transparent and more uncertain. However, being able to place any impacts in context, e.g. with meaningful baselines, and communicating those impacts, are important aspects within the context of an environmental label for food.

Section 2 covers the common Stages of 1 and 2 and explores the measurement, assessment and characterisation techniques available. This section has been structured based on the following impact areas:

- Air quality
- Biodiversity
- Climate change
- Landscape and heritage
- Noise, Odour and Dusts
- Water quality
- Resource depletion
- Soil quality
- Stratospheric ozone depletion
- Waste and recycling

For each of these impact areas the background to the issues is summarised and the stressors and impact categories identified. The techniques available to measure and assess the stressors and impacts are then explored. This includes identifying the source of data that is used to describe the effects of a product during its lifecycle.
Section 3 discusses how mid-point impacts that may be determined using the techniques discussed in Section 2 could be communicated to the consumer as environmental damage in the context of an environmental label e.g. displaying the amount of carbon (gCO$_2$e) that has been emitted by a product. Other than being a number by which to make a judgement, this is not very meaningful in terms of either the significance of the environmental impacts or its environmental performance (i.e. is it good, poor or average). Section 3 will explore some of the techniques that are available that attempt to address this issue and examine their practicality within the context of an environmental label for food products.
2.0. Stressors and mid-point impact categories: measurement, assessment and characterisation

2.1. Climate change

Background

This is one of the more established impact categories based on the relative radiative forcing of emissions of different greenhouse gas emissions. The theory is that heat becomes trapped in the atmosphere by the infrared adsorption of reflected sunlight (in a spectral window of ~10-15 μm) and changes in this adsorption capacity can result in climatic changes. Anthropogenic emissions contributing significantly to this capacity include carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O). As previously mentioned, climate change (and enhanced radiative forcing) are midpoint impact categories/effects. The consequences (and impacts) of climate change are potentially very severe and may include significant detrimental impacts on human health, crop production and ecosystems and biodiversity. However, characterising these impacts (in terms of their scale and magnitude) within an environmental assessment framework (let alone an eco-labelling scheme) is far more difficult and uncertain. Thus radiative forcing remains the primary means by which to judge impact.

GHG emissions are one part of the ‘equation’ with respect to climate change. There is also carbon sequestration which is the process of removing carbon dioxide from the atmosphere. Activities that increase carbon sequestration have the potential to ‘offset’ emissions. For example, some products may claim to be carbon neutral. This does not necessarily mean that there were no GHG emissions associated with their production as in many instances the production is accompanied with activities to sequester carbon that are equal or greater than any GHG emissions, so it is the net carbon balance that has been calculated for the product as illustrated in Figure 1.

![Figure 1: Determining net carbon](image)

There are a number of caveats associated with deducting carbon equivalents due to sequestration from the GHG balance of a product. The accumulation of carbon after e.g. a change of land use to woodland does not continue indefinitely. Eventually the trees reach maturity and no longer accumulate biomass and carbon within that biomass. The carbon within soils will also reach equilibrium although different soil types and land uses have the potential to have accumulated greater quantities. If the change in land use arose from conversion of productive agricultural land there is a risk that the marketable crop component originally grown on that land will be replaced by crops grown and imported from outside the UK or Europe. In such a scenario no net reduction in emissions will result.
It should also be noted that PAS2050 does not permit the use of carbon offsetting in the calculation of GHG emissions associated with a product. PAS2050 states that ‘GHG emissions offset mechanisms, including but not limited to, voluntary offset schemes or nationally or internationally recognized offset mechanisms, shall not be used at any point in the life cycle of the product in order to claim reduction in the emissions associated with the product.’ It is the intention that PAS2050 reflects the GHG intensity of the production process prior to the implementation of external measures to offset emissions. The use of an energy source that results in lower GHG emissions to the atmosphere and therefore achieves a lower emission factor, such as renewable electricity or conventional thermal generation with carbon capture and storage, is not a form of offsetting.

**Available techniques**

The relative contribution each gas makes towards climate change is commonly calculated in LCIA using Global Warming Potential (GWP), where GWPs are referenced to as 1 mass unit of carbon dioxide and expressed as CO$_2$e. GWP for a gas is calculated by taking into account its estimated duration in the atmosphere and its ability to absorb outgoing infrared radiation. Consequently there can be different values for each gas for different time horizons, as shown in Table 1.

**Table 1: GWP values and lifetimes**

<p>| Gas                         | Lifetime (years) | GWP time horizon |</p>
<table>
<thead>
<tr>
<th></th>
<th></th>
<th>20 years</th>
<th>100 years</th>
<th>500 years</th>
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<tbody>
<tr>
<td>Methane</td>
<td>12</td>
<td>72</td>
<td>25</td>
<td>7.6</td>
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<tr>
<td>Nitrous Oxide</td>
<td>114</td>
<td>310</td>
<td>298</td>
<td>153</td>
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<tr>
<td>HFC-23 (hydrofluorocarbon)</td>
<td>270</td>
<td>1200</td>
<td>14800</td>
<td>12200</td>
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<tr>
<td>HFC-134a (hydrofluorocarbon)</td>
<td>14</td>
<td>3830</td>
<td>1430</td>
<td>435</td>
</tr>
<tr>
<td>Sulphur Hexafluoride</td>
<td>3200</td>
<td>16300</td>
<td>22800</td>
<td>32600</td>
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Table 1 shows how the GWP for methane, for example, decreases from 72 to 25 as we go from a 20 year to 100 year time horizon because methane has an average lifetime of 12 years, and as the time horizon increases beyond its average lifetime, an increasing amount of methane is destroyed by interacting with other gases. In contrast, sulphur hexafluoride has a much longer lifetime of 3200 years on average, and consequently its GWP increases as the time horizon increases from 20, 100 to 500 years (over a 500 year period, the emission of 1 ton of sulphur hexafluoride is the same as the emission of 32600 tons of carbon dioxide). Therefore, even a tiny amount of sulphur hexafluoride emissions can make a large contribution to climate change.

The Intergovernmental Panel for Climate Change (IPCC) propose a time horizon of 100 years, GWP$_{100}$. The GWPs are revised occasionally by the IPCC, but the current GWP$_{100}$ values for methane and nitrous oxide, for example, are 25 and 298 respectively.

The calculation of GWP is a well established technique, but it does have some uncertainties attached to it. For example, it is difficult to predict the lifetime of a gas that is highly reactive to other gases in the atmosphere, because the gas’s lifetime is likely to be hard to quantify and have a changing lifetime, over the time period of measurement. Also, because current measurement tools are imprecise, it is difficult to measure the composition of the atmosphere accurately as it changes. Finally, as substances interact and change in the atmosphere, their ability to absorb outgoing radiation can also change. As a result, its ability to absorb outgoing infrared radiation may also change over time.
The accurate calculation of the GWP for a product or process is dependent on being able to calculate the GHG emissions that arise from different stages of the life cycle. There are a number of issues to consider including:

- The source of emissions data – is it actually measured or based on standard emissions factors, and if the latter what is their origin and reliability.
- The allocation of emissions to specific products and process and the boundaries of the production system. For example if a production process results in two or more products, how are the emissions associated with that process allocated to just one product. Similarly, the raw materials and energy that are inputs into a process all have associated emissions (and other impacts) and may also have other inputs themselves – so it’s a question of where do you draw the line with respect to the boundary of the system being assessed.

Standards and guidelines do now exist, such as PAS 2050 (BSI, 2008), to help harmonise the approach to tackling these issues. However, there are still many grey areas and uncertainties, especially with respect to primary production, such as agriculture, which of course is a major component in food production. Such uncertainties, and associated variability in data, can be quite critical in tipping the balance of production in favour or one system or another. The defined scope of the assessment must include all aspects of the supply chain. An assessment limited to, for example, the distance travelled by produce (‘food miles’) favours locally grown produce. Where additional heating is needed to grow such produce locally but not overseas, fewer emissions overall (the net GHG balance) may result from imported produce (e.g. Defra, 2008). A standardised set of emission factors are also required. Direct comparison of two assessments that have used different data sources must be treated with caution. This is of particular relevance to comparing assessments of origin in two different countries, and a problem encountered by Foster et al., 2006.

Data availability

There are two aspects to consider here:

- Do reliable emission factors exist for stages of the production chain?
- Do the data exist to feed those emission factors?

Agricultural (pre-farm gate) GHG emissions are mainly N$_2$O, CH$_4$ and CO$_2$. An agricultural GHG balance (t CO$_2$e ha$^{-1}$ year$^{-1}$) may be calculated using the following equation:

$$GHG\text{balance} = (m + d + i + Ns + Gs + Cs + Ni + Gf + Gl) – (\text{Seq}_{SOC} + \text{Seq}_{biomass})$$

where:

- $m =$ indirect emissions from the manufacture of agro-chemicals (fertilisers, pesticides)
- $d =$ direct emissions from the operation of machinery (application of agro-chemicals, tillage operations, harvest, drying)
- $i =$ indirect emissions associated with the manufacture of machinery taking account of machinery depreciation
- $Ns =$ emission of N$_2$O from soils
- $Gs =$ emission of CH$_4$ from soils
- $Cs =$ emission of CO$_2$ from soil
- $Ni =$ emission of N$_2$O from manures
- $Gf =$ emission of CH$_4$ from enteric fermentation
- $Gl =$ emission of CH$_4$ from manures
- $\text{Seq}_{SOC} =$ C sequestered in soil during year $n$
- $\text{Seq}_{biomass} =$ C sequestered in plant biomass during year $n$
The subtraction of sequestered carbon is not used in the PAS2050 method but has been included for illustrative purposes.

Fossil fuels (e.g. DERV, diesel) are used to power farm machinery to undertake operations such as soil tillage and application of agro-chemicals (Williams et al., 2006; Hülsbergen & Kalk, 2001; Donaldson et al., 1994; Hunt 1995). Carbon dioxide is emitted during the combustion of fossil fuels. Fuels are also consumed when agro-chemicals and farm machinery are manufactured and for their transportation to the farm. Emission of CO₂ during crop production is largely dependent on the number of farm operations and the quantity of agro-chemicals applied. This depends on the crop grown. The quantification of GHG emissions from fossil fuels from crop production typically include:

1. Product (pesticides and fertilisers) manufacture, packaging, storage and transport (to farm) (Brentrup & Pallière, 2008; Green, 1987; Hülsbergen & Kalk, 2001; Jenessen & Kongshaug, 2003; Tzilivakis et al., 2005a,b; Williams et al., 2006).
2. Application (spraying or spreading) of agrochemicals, tillage (in relation to soil type and depth, derived e.g. from regression equations) and drilling (Dalgaard et al., 2001; Donaldson et al., 1994; Hunt, 1995; Kalk & Hülsbergen, 1999; Williams et al., 2006).
3. Manufacture of farm machinery as a function of depreciation per operation (Hülsbergen & Kalk, 2001; Jenessen & Kongshaug, 2003; Williams et al., 2006).

The GWP₉₀₀ of N₂O is 298 times greater than CO₂ (Table 1) therefore small amounts have potential to contribute significantly to the GHG balance of a crop. Losses of N₂O from agriculture are mainly either from soils (in response to fertiliser application) or from livestock manures and grazing deposition (Jackson et al., 2009; Williams et al., 2006).

The emission of nitrous oxide from soils results largely from microbial nitrification and denitrification and is highly variable temporally and spatially because of the impact of soil type, water content and temperature (Machefer et al., 2002). On farm practices such as the amount of nitrogen fertiliser applied, incorporation of crop residues and clover, and use of irrigation also impact the emission of soil N₂O (Tzilivakis, 2005a; Bouwman, 1996). It is therefore difficult to predict and extremely site-specific. The IPCC (2006) methodology uses default values to calculate N₂O emissions from the application of nitrogen fertilisers and manures (Tier 1 approach) on which the UK GHG inventory approach (Jackson et al., 2009) is largely based. It does not account for differences in soil type and regional climate (specifically rainfall) and has been subject to criticism for the over-estimation of soil N₂O emissions when applied to UK arable conditions (Brown et al., 2001; 2002). The Tier 3 approach allows calculation of N₂O emissions using models such as SUNDIAL (Smith et al., 1996) or MANNER (Chambers et al., 1999) that calculate the nitrogen emitted via the nitrification and denitrification pathways or from nitrate leaching and ammonia volatilisation. The fraction of N released as N₂O-N from these processes may be calculated from emission factors provided in the published literature (e.g. DeVries et al., 2003; Jackson et al., 2009).

Nitrous oxide emissions associated with livestock are influenced by the quantity of nitrogen excreted as grazing deposition and the amount stored as manure. Grazing deposition is dependent on the type of stock and the proportion of the year that the animal remains outside. Emissions from the handling and storage of manures is dependent upon the animal type, the length and method of storage, and the N content of the diet and the efficiency with which is utilised (Jackson et al., 2009; Williams et al., 2006; Moorby et al., 2007; Freibauer, 2003). The crude protein content of feeds have been documented by Thomas (2006) and allow calculation of the total N contained within the diet of the animal allowing the proportion of N excreted to be calculated for a given livestock type. The IPCC (2006) provides guidance for the calculation of N₂O from different types of manure storage systems.
but more accurate country specific emission factors are given for the UK by e.g. Chadwick et al. (1999).

Agricultural methane emissions result mainly from ruminant enteric fermentation and livestock manures (Moorby et al., 2007; Williams et al., 2006). Mean annual UK enteric methane emissions (country specific Tier 2 approach) are available for each livestock category from the UK GHG inventory (Jackson et al., 2009). More accurate diet specific calculations are possible based on the proportion of forage and concentrates within the diet (Williams et al., 2006; 2009). The influence of the quantity of starch, a potential inhibitor of enteric methane production, in forage and concentrates may be calculated using decision support tools such as DAIRYWISE (Schils et al., 2007).

Manures produced by livestock may also contribute significantly to agricultural methane emissions. Emissions occur during storage of both liquid and solid manure although emissions are greater in liquid systems and increase with an increase in temperature (IPCC, 2006; Monteny et al., 2006; Sommer et al., 2007). They may be reduced by the cooling of slurry during storage or by covering lagoons. The UK GHG inventory provides country specific mean emission factors that account for the proportion of manures stored by each method. More specific emission factors may be calculated using Williams et al. (2009) or Chadwick et al. (1999).

Emission of methane from soils is predominantly from water-logged soils, an attribute of rice production. Losses of CH$_4$ from arable systems and woodland are considered negligible (Smith et al., 2000a, b & c; Falloon et al., 2004; Freibauer, 2003) in comparison to other agricultural land use.

Post farm gate emissions tend to result from fuel consumed during produce transportation, and the electricity utilised during processing and in particular, refrigeration. Emissions of CO$_2$ per km t transported are provided for a number of vehicles by Defra and DECC (2009). Emissions associated with controlled atmosphere post-harvest storage are provided for fruit by Mila-i-Canals (2007).

Carbon may be stored or sequestered within soils or plant biomass but as discussed previously, accumulation does not increase indefinitely because an equilibrium is eventually reached. The time to reach a new equilibrium (when carbon accumulation ceases) may be calculated as follows (Dyson et al., 2009):

$$T = \frac{SOCeqb_{(new)} - SOCeqb_{(baseline)}}{R_{(SOC)}}$$

where: $T =$ Time to establish new SOC equilibrium
$SOCeqb_{(new)} =$ potential SOC at equilibrium (t CO$_2$e ha$^{-1}$) of the new land use
$SOCeqb_{(baseline)} =$ SOC at equilibrium (t CO$_2$e ha$^{-1}$) of the baseline scenario (current land use)
$R_{(SOC)} =$ SOC accumulation rate (t CO$_2$e/ha/year) for a given change in land management

Carbon contained within soils is subject to spatial and temporal variation but is dependent primarily on the land use and the amount of soil disturbance. Frequently disturbed arable land tends to contain smaller quantities of SOC at equilibrium than e.g. permanent grassland or woodland (Bradley et al., 2005; Dyson et al., 2009). The carbon storage potential of plant biomass is dependent on factors such as the dominant vegetation (e.g. trees or grass) and intensity of grazing (Smith et al., 2000a & b; Falloon et al., 2004). Sequestered carbon is not a permanent removal of carbon since it may be re-emitted if there is a change in land management or death of the plant biomass in which it is contained.


2.2. Air quality

Background

Elevated levels of smoke and sulphur dioxide due to the combustion of sulphur-containing fossil fuels (e.g. from domestic and industrial use) were previously the main cause of declines in air quality. At present problems with air quality result mainly from emissions from road vehicles. Pollutants are dominated by carbon monoxide (CO) but also include oxides of nitrogen (NO$_x$), volatile organic compounds (VOCs) and particulates (PM$_{10}$). Photo-chemical reactions catalysed by sunlight on nitrogen dioxide (NO$_2$) and VOCs forms ozone, a secondary long-range pollutant able to disperse great distance from the original source of emission. Emissions of NO$_x$ and its subsequent dispersal over large distances results in ‘acid rain’. Air pollution caused by vehicle emissions is continuing to increase on a global scale.

The main air pollutants and associated impacts are outlined below:

- **Sulphur dioxide (SO$_2$):** is produced by the combustion of a sulphur containing material. Most atmospheric SO$_2$ globally is natural in origin, however in the UK electricity generation using fossil fuels (mainly coal and heavy oils) is the major source. Domestic coal use may result in localised high concentrations of SO$_2$ that at moderate concentrations has potentially detrimental impacts on lung function in asthmatics such that medical assistance is needed. Sulphur dioxide is more harmful as a pollutant in the presence of high concentrations of other pollutants.

- **Nitrogen oxides:** Nitrogen dioxide (NO$_2$) and nitric oxide (NO) are both oxides of nitrogen and referred to as nitrogen oxides (NO$_x$). Nitric oxide results mainly from road vehicle emissions and fossil fuels combusted during electricity generation. Nitric oxide is not considered harmful to health but the NO$_2$ that it forms rapidly upon release to the atmosphere is. It may cause irritation of the lungs and increase susceptibility to respiratory infections (e.g. influenza). Continued or frequent exposure to high concentrations may increase the incidence of child acute respiratory illness.

- **Fine Particles (PM10, PM2.5 and PM1):** Fine Particles consist of particles from fuel combustion (mainly from road vehicles), secondary particles (such as sulphate and nitrate that are readily dispersed and transported across national boundaries) and coarse particles (dust, sea salt, biological particles i.e. pollen and particles from construction). Particles are defined according to size (mean aerodynamic diameter) and PM10 is the main focus of air quality monitoring, but the finer fractions such as PM2.5 and PM1 are becoming of increasing interest in terms of health effects. The smaller particles (PM2.5 and PM1) may penetrate deep into the lungs and worsen heart and lung disease or transport surface-absorbed carcinogenic compounds.

- **Ozone:** Ozone (O$_3$) is formed primarily formed by the action of sunlight on volatile organic compounds (VOCs) in the presence of NO$_x$. Its formation occurs over a period of hours or days and can, therefore occur a large distance away from the source of its precursor molecules, usually downwind. Ozone causes irritation of the airways and agitates the symptoms of asthma and lung disease.

- **Volatile Organic Compounds (VOCs):** Hydrocarbon VOCs are usually grouped into methane and other non-methane VOCs (NMVOCs). Methane is an important VOC due to its contribution to global warming and to the production of ground level or lower atmosphere ozone. Most methane is released to the atmosphere via the leakage of natural gas from distribution systems. Benzene is a VOC present in petrol. The majority of benzene emissions (a NMVOC) arise from the use of petrol-fuelled vehicles. Consequently, benzene concentrations are higher in urban areas, particularly along roadsides, than in rural areas (Friedrich & Obermeier, 1999). Impacts on health include cancer, disorders of the central nervous system, damage to liver and kidneys,
reproductive disorders, and birth defects. 1,3-Butadiene a VOC emitted mainly from combustion of petrol and diesel but also used in the manufacture of synthetic rubber. The health impacts are similar to those of benzene. With respect to agriculture there are several sources of VOC emissions from crop and livestock sources and some NMVOCs also occur from damaged plants (e.g. during harvesting and mowing operations). Many VOCs are quite harmful to human health (IGER, 2003).

- **Toxic Organic Micro-Pollutants (TOMPs):** TOMPs arise from the incomplete combustion of fuels, but their constituents although emitted in minute quantities are highly toxic or carcinogenic. No threshold levels exist. Such compounds include: PAHs (Poly Aromatic Hydrocarbons); PCBs (Poly Chlorinated Biphenyls); Dioxins; Furans. TOMPS may be carcinogenic, result in reduced nervous system disorder immunity or interfere with child development.

- **Carbon monoxide:** is produced by incomplete or inefficient combustion. Its main source is road vehicles, especially petrol-engines. It prevents transport of oxygen in the blood and a significant reduction of oxygen supplied to the heart.

- **Lead and Heavy Metals:** The main sources of lead at present are secondary non-ferrous metal smelters. Small quantities are harmful, especially to unborn and young children where exposure is linked to impaired mental function, visual-motor performance, neurological damage, memory and attention span.

**Available techniques**

Of the above issues that one that has received most attention under the area of air quality is ozone and Volatile Organic Chemicals in the context of photo-oxidant formation (or photochemical smog).

As illustrated in Table 2 individual air pollutants (see above) may be handled under other impact categories such as human toxicological effects, ecotoxicological effects, eutrophication and acidification. Table 2 provides values from the University of Leiden LCA characterisation factors database (University of Leiden, 2009) for human toxicity, respiratory effects on humans caused by organic or inorganic substances, carcinogenic effects on humans caused by organic or inorganic substances, freshwater aquatic ecotoxicity, marine aquatic ecotoxicity, terrestrial ecotoxicity, eutrophication and acidification (incl. fate, average Europe). Blank cells indicate no value is currently present in the database.

**Table 2: Characterisation factors**

<table>
<thead>
<tr>
<th></th>
<th>Human toxicity</th>
<th>Respiratory effects humans</th>
<th>Carcinogenic effects humans</th>
<th>Freshwater aquatic ecotox</th>
<th>Marine aquatic ecotox</th>
<th>Terrestrial ecotox</th>
<th>Eutrophication</th>
<th>Acidification</th>
</tr>
</thead>
<tbody>
<tr>
<td>SO₂</td>
<td>0.096</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1.2</td>
<td></td>
</tr>
<tr>
<td>NO</td>
<td></td>
<td>1.37E-04</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.2</td>
<td>0.76</td>
</tr>
<tr>
<td>NO₂</td>
<td>1.2</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>PM₁₀</td>
<td>0.82</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.13</td>
<td>0.5</td>
</tr>
<tr>
<td>VOCs</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>PAH</td>
<td></td>
<td>1.7E-04</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>PCBs</td>
<td></td>
<td></td>
<td></td>
<td>0.00197</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Dioxins</td>
<td>1.9E+09</td>
<td></td>
<td></td>
<td>2.1E+06</td>
<td>3.0E+08</td>
<td></td>
<td>1.2E+04</td>
<td></td>
</tr>
<tr>
<td>Benzene</td>
<td>1830.8</td>
<td></td>
<td></td>
<td>4.12E-06</td>
<td>0.0914</td>
<td></td>
<td>0.0027</td>
<td>1.4E-05</td>
</tr>
<tr>
<td>1,3-Butadiene</td>
<td></td>
<td></td>
<td></td>
<td>3.3E-07</td>
<td>2.7E-06</td>
<td></td>
<td>2.3E-08</td>
<td></td>
</tr>
</tbody>
</table>
Photo-oxidants include ozone that can be created in the troposphere via photochemical oxidation of VOCs and carbon monoxide in the presence of NOx, catalysed by ultraviolet light. Smaller quantities of natural background ozone are transported downwards from the stratosphere.

Each individual VOC has its own unique reaction path although some may be similar. Ozone production is dependent on the chemical and meteorological conditions where the VOC is emitted. A particular VOC may produce high levels of ozone where high NOx concentrations exist, and low quantities where the availability of NOx is limited. The VOCs that produce radicals during photolytic degradation increase oxidation of any other VOC present and as a consequence, ozone production. A greater intensity of radiation increases the efficiency at which VOCs are able to produce ozone. The ozone creation potential of each VOC is subject to significant spatial and temporal variability (Labouze et al., 2004).

Despite these complexities some relatively simple measures exist to assess photo-oxidant formation. In Northern Europe the assessment is based on the calculated photochemical ozone creation potential (POCP) which is expressed in units of ethylene (in a similar way to equivalent units of CO2 for GWP). An example of some POCP factors is given in Table 3 (Derwent et al., 1998).

<table>
<thead>
<tr>
<th>Pollutant</th>
<th>POCP (kg ethylene eq.)</th>
</tr>
</thead>
<tbody>
<tr>
<td>SO2</td>
<td>0.048</td>
</tr>
<tr>
<td>NO</td>
<td>-0.427</td>
</tr>
<tr>
<td>NO2</td>
<td>0.028</td>
</tr>
<tr>
<td>Benzene</td>
<td>0.218</td>
</tr>
<tr>
<td>1,3-Butadiene</td>
<td>0.851</td>
</tr>
</tbody>
</table>

The POCPs given by Derwent et al. (1998) (e.g. Table 3) have been widely used as a means to assess and aggregate the emissions for the impact category photo-oxidant formation using the following equation:

\[
\text{Photo-oxidant formation} = \text{POCP}_x \times m_x
\]

Where:
- \( m_x \) = the mass of substance X released (kg)
- \( \text{POCP}_x \) = the photochemical ozone creation potential of the substance

The POCPs are calculated based on different scenarios in Northern Europe. In the United States, photochemical ozone production is estimated based on the Maximum Incremental Reactivity (MIR), and is measured in units of O3. MIR is based on laboratory measurements of the maximum amount of ozone that can be produced by given VOC's in an artificial atmosphere that represents the weighted average of US cities. The POCP approach has the advantage that it provides different scenarios, and the disadvantage that it does not evaluate non-Northern European situations. The MIR approach is simpler to use, but its results have not been verified outside of North America.

These approaches are not without problems. For example, neither approach includes the impact of NOx and this often accounts for more than 50% of the total ozone formation impact. On a more basic level, it is not clear that measuring or estimating the ozone in smog is the best indicator of the overall environmental effects. For example, peroxyacetyl nitrate and other photochemically produced...
substances may cause damage to human health and the environment, and so the impacts of smog might be better covered under human health and ecotoxicity impact categories (Jolliet et al., 2003a).

Whilst the term NMVOC is used specifically to identify a group of chemicals that exclude methane it has also been used as for quantifying total emissions, where all NMVOC emissions are summed on a per weight basis to a single figure. In absence of more detailed data, this could be used as a very coarse parameter for pollution, e.g. for summer smog or indoor air pollution. Regarding NMVOC emissions, the large number and diversity of compounds is cited by Van Aardenne and Gros (2005) as limiting a complete inventory of these substances. The authors overcome this to a certain degree by aggregating NMVOCs into 25 groups (Table 4). However, even if grouped NMVOC data is available and expressed as the proportion of the total emitted there is usually insufficient data resolution to enable modelling and assessment of damage (Van Aardenne & Gros, 2005).

### Table 4: Aggregate NMVOC groups

<table>
<thead>
<tr>
<th>Group</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>a01</td>
<td>alkanols (alcohols)</td>
</tr>
<tr>
<td>a02</td>
<td>ethane</td>
</tr>
<tr>
<td>a03</td>
<td>propane</td>
</tr>
<tr>
<td>a04</td>
<td>butanes</td>
</tr>
<tr>
<td>a05</td>
<td>pentanes</td>
</tr>
<tr>
<td>a06</td>
<td>hexanes and higher alkanes</td>
</tr>
<tr>
<td>a07</td>
<td>ethene</td>
</tr>
<tr>
<td>a08</td>
<td>propene</td>
</tr>
<tr>
<td>a09</td>
<td>ethyne</td>
</tr>
<tr>
<td>a12</td>
<td>other alk(adi)enes and alkynes</td>
</tr>
<tr>
<td>a13</td>
<td>benzene</td>
</tr>
<tr>
<td>a14</td>
<td>toluene</td>
</tr>
<tr>
<td>a15</td>
<td>xylenes</td>
</tr>
<tr>
<td>a16</td>
<td>trimethylbenzene</td>
</tr>
<tr>
<td>a17</td>
<td>other aromatics</td>
</tr>
<tr>
<td>a18</td>
<td>esters</td>
</tr>
<tr>
<td>a19</td>
<td>alkoxy alkanes (ethers)</td>
</tr>
<tr>
<td>a20</td>
<td>chlorinated HCs</td>
</tr>
<tr>
<td>a21</td>
<td>methanal</td>
</tr>
<tr>
<td>a22</td>
<td>other alkanals</td>
</tr>
<tr>
<td>a23</td>
<td>alkanones (ketones)</td>
</tr>
<tr>
<td>a24</td>
<td>(alkanoic) acids</td>
</tr>
<tr>
<td>a25</td>
<td>other NMVOC (HCFCs, nitriles, etc.*)</td>
</tr>
</tbody>
</table>

### Data availability

In order to calculate the potential for photo-oxidant formation during production of a product, the emissions of different VOCs need to be known. There are a few emission factors available for a few basic processes as shown in Table 5 (taken from Caserini et al., 2010). However, from the literature review undertaken the available data is severely limited.

### Table 5: Average Emission factors for biomass and fossil fuel combustion for estimating photo-oxidant formation

<table>
<thead>
<tr>
<th>Process</th>
<th>PM10</th>
<th>NOx</th>
<th>VOC</th>
<th>SO2</th>
<th>CO</th>
<th>PAH</th>
<th>PCDD/F</th>
</tr>
</thead>
<tbody>
<tr>
<td>Units</td>
<td>gGJ(^{-1})</td>
<td>gGJ(^{-1})</td>
<td>gGJ(^{-1})</td>
<td>gGJ(^{-1})</td>
<td>gGJ(^{-1})</td>
<td>mgTEFGJ(^{-1})</td>
<td>ngTEFGJ(^{-1})</td>
</tr>
<tr>
<td>Open fireplace</td>
<td>500</td>
<td>70</td>
<td>5650</td>
<td>13</td>
<td>5650</td>
<td>280</td>
<td>500</td>
</tr>
<tr>
<td>Conventional wood stove</td>
<td>250</td>
<td>70</td>
<td>1130</td>
<td>13</td>
<td>5650</td>
<td>280</td>
<td>400</td>
</tr>
<tr>
<td>Low emission wood log stove</td>
<td>150</td>
<td>60</td>
<td>560</td>
<td>13</td>
<td>2260</td>
<td>280</td>
<td>120</td>
</tr>
<tr>
<td>Pellet stove BAT</td>
<td>30</td>
<td>60</td>
<td>60</td>
<td>13</td>
<td>620</td>
<td>0.1</td>
<td>5</td>
</tr>
<tr>
<td>CHP plant 8 MW</td>
<td>2.2</td>
<td>57</td>
<td>2.7</td>
<td>2.2</td>
<td>8</td>
<td>0.003</td>
<td>0.13</td>
</tr>
<tr>
<td>CHP plant 100 MW</td>
<td>0.2</td>
<td>34</td>
<td>No data</td>
<td>7.2</td>
<td>12</td>
<td>0.003</td>
<td>0.6</td>
</tr>
<tr>
<td>Natural gas domestic boiler</td>
<td>0.2</td>
<td>50</td>
<td>5</td>
<td>0.5</td>
<td>25</td>
<td>-</td>
<td>1.7</td>
</tr>
<tr>
<td>Oil domestic boiler</td>
<td>40</td>
<td>150</td>
<td>10</td>
<td>150</td>
<td>16</td>
<td>-</td>
<td>1.7</td>
</tr>
<tr>
<td>Diesel domestic boiler</td>
<td>5</td>
<td>50</td>
<td>3</td>
<td>100</td>
<td>20</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>
2.3. Soil quality

Background

Soils are poorly understood in comparison with other environmental media, but their quality/health is becoming of increasing interest to both farmers concerned with the sustainability of agricultural production, and a wider community of environmentalists, regulators and other organisations (Warwick HRI, 2005). Consequently the need to be able to define soil quality has never been greater, although our ability to do so is subject to significant limitations.

Soil quality is difficult to define, not least because even in a ‘pristine’ state, they vary enormously in their chemical, physical and biological make-up. As a result setting pre-defined standards against which to judge quality is difficult, since these too must vary considerably to reflect site specific circumstances. Most studies therefore accept that concepts of soil quality need to be defined within the context of the different functions those soils perform, such that soil quality can be defined as:

“...the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation” - Karlen et al. (1997).

However, these functions are wide ranging, and include those of environmental interaction, food and fibre production, provision of a building platform, support for ecological habitat and biodiversity, the provision of raw materials and the protection of cultural heritage, each of which will have its own requirements. Nevertheless, this concept has formed the basis for the work that has been undertaken to derive soil quality indicators (e.g. Loveland et al., 2002; Merrington et al., 2006), and there is now a plethora of potential indicators available (see Table 6), although some are of more relevance to the production of food than others. Within the context of an environmental label for food production, soils clearly need to be able to perform the function of food and fibre production; but there are inevitably overlaps with ecological, biodiversity and environmental interaction functions. Agricultural soils not only have to be productive in terms of food and fibre, but they can be a source and/or sink of greenhouse gases, a buffer/filter for other pollutants and provide a habitat for biodiversity.

Relatively little work has been done on the use of indicators to quantify ‘impact’ (as opposed to current status), although Lilburne et al. (2004) examined the development of criteria based on production and environmental characteristics, by which to judge quality. Since as they point out “there seems little point in proposing any soil property as a soil quality measure if we cannot provide for its interpretation”. However, in studies looking at the potential for the use of soil quality indicators by land managers to support sustainable land management (Wander & Drinkwater, 2000; Andrews et al., 2003), it has been concluded that there are significant barriers to their widespread adoption, not least the fact that many of the systems currently available are too expensive and/or complex for on-farm use (Herrick, 2000). Nevertheless, the increasing emphasis on including measures of soil quality in sustainability assessment continues to provide a driver for this sort of work.

Available techniques

Due to the multifunctional nature of soils and their varied properties, there is no single tool for ‘measuring’ quality; there are however, a wide range of indicators available covering physical, chemical and biological properties of the medium. The most appropriate indicator (or more likely,
suite of indicators) is dependent on the purpose for which ‘quality’ is being assessed (i.e. the needs of the construction industry will tend to focus on physical characteristics, whilst for agriculture all three classes may be important). In addition, as discussed above, the ‘target’ values of the selected indicators which will tend to vary depending on the type of soil and the particular function being assessed. Table 6 shows a number of potential soil quality indicators, with ticks marking those seen as being particularly relevant to the present study due to their relevance to food and fibre production, interpretability and scientific robustness (Loveland et al., 2002; Merrington et al., 2006). This list is by no means exhaustive, but illustrates that the number of potential measures is significant and assessing even just those marked would present a significant challenge within the context an environmental assessment or environmental labelling scheme.

Table 6: Indicators of soil quality

<table>
<thead>
<tr>
<th>Indicators</th>
<th>Selected</th>
<th>Indicators</th>
<th>Selected</th>
</tr>
</thead>
<tbody>
<tr>
<td>Above ground biomass</td>
<td></td>
<td>Extractable K</td>
<td></td>
</tr>
<tr>
<td>Soil water content at 1 m</td>
<td>✓</td>
<td>Extractable Mg</td>
<td>✓</td>
</tr>
<tr>
<td>Soil wetness characterisation</td>
<td></td>
<td>Extractable S</td>
<td></td>
</tr>
<tr>
<td>Soil storage capacity</td>
<td></td>
<td>Extractable Ca</td>
<td></td>
</tr>
<tr>
<td>Top soil aggregate stability</td>
<td>✓</td>
<td>pH</td>
<td>✓</td>
</tr>
<tr>
<td>Bulk density</td>
<td>✓</td>
<td>Total Zn, Cu, Ni, Cd</td>
<td>✓</td>
</tr>
<tr>
<td>Aeration</td>
<td></td>
<td>Total Pb</td>
<td></td>
</tr>
<tr>
<td>Organic carbon</td>
<td>✓</td>
<td>Extractable B, Cu, Mn, Se</td>
<td></td>
</tr>
<tr>
<td>Macroporosity</td>
<td></td>
<td>Biomass indicator</td>
<td></td>
</tr>
<tr>
<td>Soil (horizon) depth</td>
<td></td>
<td>Earthworms (total number)</td>
<td></td>
</tr>
<tr>
<td>Root penetration</td>
<td></td>
<td>Soil borne diseases</td>
<td></td>
</tr>
<tr>
<td>Total N</td>
<td>✓</td>
<td>POPs</td>
<td></td>
</tr>
<tr>
<td>Wind throw</td>
<td></td>
<td>ALC/Land capability</td>
<td></td>
</tr>
<tr>
<td>Depth to water logged layer</td>
<td></td>
<td>Salinity (EC)/Sodicity</td>
<td></td>
</tr>
<tr>
<td>PMN</td>
<td></td>
<td>Seed bank</td>
<td></td>
</tr>
<tr>
<td>Extractable P</td>
<td>✓</td>
<td>Plastic glass/extraneous material</td>
<td></td>
</tr>
</tbody>
</table>

**Quality indices:** As intimated above, to obtain a holistic impression of soil quality, multiple indicators are required, which has led a number of authors to propose the use of indices (e.g. Parr et al., 1992; Doran & Parkin, 1994) of quality based on combinations of indicators and/or function characteristics. One of the first was that by Parr et al. (1992), which can be stated as:

\[
SQ = f (SP, P E, H, ER, BD, FQ, MI)
\]

Where:

- \(SP\) = soil properties
- \(H = \) human/animal health
- \(FQ = \) food quality/safety
- \(P\) = potential productivity
- \(ER = \) erodability
- \(MI = \) management inputs
- \(E = \) environmental factors
- \(BD = \) biological diversity

However, the authors made no attempt to define either how the factors could be calculated or how they should be integrated (Warwick HRI, 2007), a problem which to a lesser or greater extent dogs many such systems, particularly when applied to a wide range of soil types and/or production systems. Although Karlen et al. (2001) suggest that this may be less of a problem if more narrowly applied.
Faced with this complexity, impact assessments have typically therefore tended to use the simplest and/or single measures of impact, for example soil loss/depletion, soil organic matter and soil compaction, each of which is discussed in more detail below.

**Soil loss/depletion:** At a basic level, the loss of soil (e.g. through erosion) at a rate in excess of replenishment, is unsustainable and is a depletion of a resource regardless of what functions that soil is performing. There are also secondary consequences, in that the lost soil and its constituents (organic matter, nutrients, pesticides, heavy metals, etc.) will result in impacts such as sedimentation in rivers (impacting on flooding and fish spawning grounds), eutrophication, and ecotoxicity; although, in the context of this study, these secondary impacts should be accounted for within other impact categories (e.g. water quality, biodiversity, etc.). It is worth considering however, that except where soil loss is a major problem, such as in areas with considerable visible soil erosion, there is unlikely to be a great deal of data on actual losses, as it isn’t something that is generally measured on farm. There are however, a number of methods for estimating losses, although each has its limitations in the context of an environmental labelling scheme. These include the Universal Soil Loss Equation (Wischmeier & Smith, 1978) and the Morgan, Morgan and Finney model (Morgan et al., 1984), although this shouldn’t be taken as an exhaustive list.

The Universal Soil Loss Equation (USLE) model (Wischmeier & Smith, 1978) and its derivative, the Revised Universal Soil Loss Equation (RUSLE) model (Renard et al., 1998), are amongst the most established techniques for assessing soil loss. Five component factors (R, K, LS, C, and P) are multiplied together to compute the average annual sheet and rill erosion per unit area (the main differences being in the determination of factors).

\[
A = R \times K \times LS \times C \times P
\]

Where:

- **R** = annual erosivity
- **K** = base soil erodibility
- **P** = support practices factor
- **C** = cover-management
- **LS** = slope length, steepness, and shape

The USLE/RUSLE is relatively simple and so lends itself well to being used in environmental assessment. However, it does have a number of limitations and shortcomings. Firstly, although it has been used throughout the world, the model was initially designed for use in the USA east of the Rockies. As a result, it may not be truly applicable in some UK situations. In addition, it only gives a very crude estimate of long-term expected soil loss; gully erosion (where this occurs) is not taken into account; the effect of stones and rock fragments in the soil is also not included; and there are uncertainties in the rainfall erosivity and soil erodibility factors.

The Morgan, Morgan and Finney model (Morgan et al., 1984) was designed for use in UK situations, and unlike the USLE is a process based model of erosion. It calculates the ability of rainfall to detach sediments (based on the kinetic energy available in rainfall, and the detachability of the soils), and the ability of overland flow to transport those sediments (based on runoff volumes, crop cover, management and slope). However, being a more process based model, it is perhaps more appropriate for modelling a specific set of circumstances than the more generalised circumstances that might be appropriate for an environmental labelling scheme.

**Soil Organic Matter (SOM) or Soil Organic Carbon (SOC):** SOM/SOC is widely recognised as the best stand-alone indicator for soil quality even though it does not fully consider all aspects of soil
functioning (Milà i Canals et al., 2007), and has been described as the most promising indicator for use in situations in which the information will feed into management decisions (Wander & Drinkwater, 2000). In addition, the organic matter content of agricultural top soils is one of the government’s indicators of agricultural sustainability (together with the accumulation of heavy metals (MAFF, 2000). Changes in SOM/SOC as a result of land use practices can impact upon soil functions including agricultural fertility, buffering and filtering capacity and soil carbon (GHG emissions). In the context of impact assessment SOM/SOC can be accounted in a number of different ways. For example Cowell and Clift (2000) propose simply that additions of organic matter to the soil be used as the indicator of impact, whereas Brandão et al. (2010) and Milà i Canals et al. (2007) use the change in SOM/SOC between the start and end of the land use activity, and also take account of the time it takes for the soil to return to its original SOM/SOC value following that land use (relaxation time). Sparling et al. (2003) went on to use modelling to determine ideal ranges of SOM, which is attractive in that it allows for the setting of targets; however, it requires a detailed knowledge of the soils in question, which means it is only really applicable to soils that have been modelled.

**Soil compaction:** Compaction of soil is an issue of particular relevance for the sustainability of agricultural production because it both affects yields of future crops, and is a reflection of the production system (amongst other factors). In addition, subsoil compaction in particular is extremely difficult to treat once it has taken place, making its avoidance of even greater importance. Cowell and Clift (2000) proposed a Soil Compaction Indicator (SCI) based on Field Load Index (FLI - Kuipers and van de Zande, 1994), as a means of assessing soil compaction. This involves taking the weight of vehicles and implements for each operation and multiplying by the time (hours/ha) taken to undertake the operation and the area (ha) on which the operation(s) is/are carried out. In the context of the present study, this has the advantage of being directly linked to the process of crop production, and to data that is recordable on-farm. Although clearly actual soil compaction will be dependent on site specific soil characteristics, relatively simple data could be used to provide an indicator of the potential for compaction.

**Qualitative techniques:** In addition to a host of quantitative systems for assessing soil quality, a number of qualitative systems have been developed, including the Visual Soil Assessment developed by the Soil Management Initiative and Väderstad (2005) for use in the UK, from that produced by Landcare Research for use in New Zealand (Shepherd, 2000; Shepherd et al., 2000 a, b & c). These techniques are based on the sort of assessments it is possible for farmers to do on their own farms (i.e. they require no lab analysis), and are therefore predominantly related to soil physical properties (e.g. structure, porosity, colour, the presence of pans etc.), each of which is evaluated against a set of pre-defined photos, although there is the capability to include some biodiversity indicators (earthworm counts), and plant indicators are also used (crop height, weed infestation, yields). However, although such assessments are good management tools for identifying problems, in the context of this study, they may be difficult to attribute to crop production, since the results may be related to historic processes, and therefore not reflective of current activities.

Alternatively, there are a number of systems available that relate soil type, agricultural production processes, climate and other factors to the potential for soil damage and/or loss. This includes the ‘Soil Erosion Risk’ and ‘Soil Damage Risk’ tools within the Environmental Management for Agriculture (EMA) software produced by the University of Hertfordshire (Lewis & Bardon, 1998), as well as the Soil Protection Review (SPR (Defra, 2009)) now required under Cross Compliance. In each case, there is only limited attempt made to associate ‘actual’ soil impacts to agricultural processes (although the SPR does for example require farmers to identify problems on their land), rather they identify activities that contribute to the risk of damage/loss (or conversely the actions taken to reduce risks). This means that it is possible (in some cases at least) to associate risk to the production of particular
process and perhaps more importantly (as far as this study is concerned), crops and therefore food products.

**Data availability**

As discussed above, many of the systems currently available are intended for the assessment of current soil quality (and the problems that may be associated with it), although they are used for monitoring medium to long-term changes in quality, particularly on a relatively large spatial scales (e.g. national). For this purpose, there are a number of data collection programmes in operation in the UK. These include the Defra funded Representative Soil Sampling Scheme (ADAS and Rothamsted Research), the National Soils Resources Institute’s Land Information System (LandIS), the Countryside Survey, and work carried out by the UK Environment al Change Network, amongst others. However, these national sources of soils data, although potentially providing data to feed into the process, are unlikely to be of sufficient resolution or frequency to provide a basis for the development of an environmental label, neither are they easily associated with specific food products. Equally, the farm-based evaluations of actual soil quality (including visual and lab based assessments), are more appropriate for assisting on-farm management by identifying problems in need of address, than for the incorporation into an ‘environmental label’ for the food produced. Not least because they rarely attempt to attribute to the observed patterns to specific, short-term production processes (and therefore crops).

It is also important to remember that the above indicators of soil quality (and any associated impacts) could be considered to be only preliminary impacts, since any degradation in soil quality will go on to have a number of secondary impacts that may be assessed in other ways. There is therefore a need to ensure that there is no ‘double accounting’ in any performance rating system. For example, Cowell and Clift (2000) propose that soil loss/depletion should be part of the abiotic resource depletion (see Section 2.7) impact category in Life Cycle Assessment. It could also be argued that although in this instance changes in SOM/SOC and soil compaction are being used to reflect impacts on soil quality, they are not actual end impacts in themselves. SOM/SOC is a particularly complex issue because, for example, although it is generally seen as desirable to increase SOM/SOC, under some circumstances this can result in greater nitrogen mineralisation and increased greenhouse gas emissions (Bending & Turner, 2009). This highlights again the issue of soils being multi-functional and thus the difficulty in defining soil quality and measures to assess impacts on soils.

Therefore, within the context of this study, it is perhaps most appropriate to consider a relatively limited set of quality criteria (for example compaction), which can be associated with particular production processes, crops and therefore food products. This is particularly the case since assessments of soil quality generally require farm level data, either to describe impacts directly or to allow them to be evaluated from management practices (e.g. compaction from vehicle passes – see above), consequently if a system is to be widely adopted, then the required data must be either already collected on farm or relatively easy to collect. Particular attention therefore needs to be given to the use of pre-existing data collection such as that needed for the Soil Protection Review (Defra, 2009), or that can be recorded during production (e.g. the field operations performed, the vehicles used, etc.). Therefore, it is unlikely that a label can reflect ‘actual’ impact on soil quality without becoming overly burdensome in both data collection and label differentiation. It is however perfectly plausible to reflect the ‘potential’ impact that food production has on a few key factors (especially those that cannot be reflected in other indicators), and given the importance of soil as a medium, it is perhaps vital that this is done.
2.4. Water quality

Background

The aquatic environment has long been seen as central to environmental impact assessment in general, and agri-environmental impact assessment in particular, since the industry has a major impact on the quality of surface and ground waters (particularly now many industrial sources of pollution have been addressed), with knock-on effects on biodiversity and water resources. In an industry so intrinsically linked to its operating environment, this can happen in a number of ways, including (amongst others):

- **Use of water resources**: Farms make considerable use of water resources (for irrigation, farm yard cleaning, watering cattle, washing produce, etc.), and although much of this will be considered in terms of resource use (see Section 2.4), it also has the potential to impact on environmental water quality. Of particular significance is the impact of water abstracted from surface waters, which although controlled through the licensing process, can impact on river chemistry and biodiversity, particularly during periods of unusually low flow (i.e. when being used for summer irrigation). Of course this can also be related to other forms of water use (e.g. mains water); however, being less direct, it is less easy to relate to an environmental impact.

- **Loss of nutrients**: Nutrients from inorganic fertiliser and livestock manures and slurries, that can have a significant impact on surface and groundwaters, have been the focus of considerable legislation and best practice advice in recent years. Nutrients are applied to agricultural land as a means to maintaining soil fertility, but can move into the wider environment in a number of ways. Run-off, sub-surface flow and field drains can move both dissolved and absorbed nutrients into surface waters, and leaching and percolation can move them (mainly dissolved pollutants) into groundwaters. Of particular importance are nitrogen (which tends to move in solution) and phosphate (the movement of which is generally associated with sediments – soil erosion). Both can be associated with eutrophication and the knock-on effects it has on biodiversity (via out competition of other species and oxygen depletion – due to the BOD of the organic material and increased demand from algal species), and nitrogen in particular (as nitrate) can cause human health problems (e.g. methemoglobinemia). In recent years, nitrogen in particular has been the focus of considerable legislation (e.g. the Nitrates Directive, and the Nitrate Pollution Prevention Regulations 2008) and associated best practice guidance (e.g. within the context of the COGAPS (Defra, 2009a) and various NVZ guidance).

- **Pathogen contamination**: Microbiological contaminants can be a significant issue when slurries and manures contaminate surface waters and sometimes groundwaters. Such problems can occur via run-off and sub-surface flow into surface water, or via percolation into groundwater with consequent impacts for drinking water quality, aquatic ecosystems and biodiversity. Key issues include the spreading of manures/slurries close to surface waters, or in unsuitable weather/soil conditions, or as a result of leakage from storage systems.

- **Pesticide contamination**: Pesticides applied to land/crops can enter surface waters via run-off, drift, sub-surface flow and field drains (either in solution or adsorbed to soil particles), and groundwaters by leaching and percolation, with consequent impacts on drinking water resources, aquatic ecosystems and biodiversity. They can cause significant problems for water companies in terms of ensuring that water supplies meet the drinking water limit of 0.1μg l⁻¹ (water companies spend millions on treatment), and can be toxic in aquatic ecosystems (Skinner et al., 1997). As such there is considerable pressure for the agricultural industry to manage its use of pesticides so as to minimise the impact on the wider environment (including water quality), with a pesticide tax never being too far from the agenda and the Voluntary Initiative (VI)
producing a considerable volume of guidance in an attempt to avoid this (e.g. Voluntary Initiative, 2002).

- **Erosion of soil:** This leads both to the transport of pollutants into receiving waters (see above – nutrients and pesticides) and (in surface waters) to sedimentation, which can have a significant impact on aquatic ecosystems. This is particularly the case in rivers in which gravel beds are the norm, since these can become clogged with finer material damaging fish spawning grounds and damaging the habitat of a number of benthic species. In extreme cases however, the increased presence of fine sediment within the water column, can also impact on aquatic biodiversity.

For these reasons (and others), controlling agricultural water pollution is fundamental to EU (the Water Framework Directive) and national (e.g. River Basin Management Planning (Environment Agency, 2006); Catchment Sensitive Farming (Environment Agency & English Nature, 2005); Nitrate Vulnerable Zones (Environment Agency, 2003) environmental policy and legislation. Indeed, until replaced at the top of many people’s agendas by climate change, it was the single most important environmental issue facing the industry.

Some (if not all) of these impacts on water quality could however, be considered only midpoint impacts which are of relevance due to the knock-on effects they have (e.g. on biodiversity, human health, etc.). For example:

- Water use is covered under resource use, and hence its impact on flows may constitute double accounting.
- The impact of nitrate and pesticides on groundwater is a water resources issue, and may be covered in relation to human toxicity.
- Nutrients in surface water could be covered under eutrophication and/or ecotoxicity (and theoretically at least human health).
- Pesticides in surface water impact on ecotoxicity and potentially human health.

In each case therefore, it is vital to ensure that the categories adopted in relation to an eco-label for food, avoid duplication.

### Available techniques

There are a number of techniques available for assessing the impact of agriculture on the quality of surface and groundwaters, albeit with problems associated with them. These include direct methods of measuring water quality (although there are a number of ways of defining quality – chemical, biological, micro-biological), as well as indirect methods focusing on the adoption of minimisation measures and best practice.

**Direct monitoring:** There are a number of techniques available for assessing impact on water quality, although there is at present little done to assess the impact of specific farms (let alone specific products) unless there is a known problem being investigated. In general, these can be chemical (including nutrient), biological and/or micro-biological, or often a combination thereof. The key methodologies used in this country are those adopted by the Environment Agency.

Until recently the Environment Agency has been using the GQA (General Quality Assessment) methodology to determine the quality of rivers and estuaries and identify changes in quality (Green & Faulkner, 2000), for 40,000km of rivers and canals and 2,800km of estuaries. This involved monitoring at over 7,000 sites across England and Wales, and was based on their chemical (BOD,
dissolved oxygen and ammoniacal-N), biological and nutrient status. This is clearly a limited set of determinants, which was selected to facilitate the identification of temporal trends, but which may fail to pick up some problems associated with agricultural pollution. Although BOD, dissolved oxygen and ammoniacal-N are all related to contamination by organic materials and/or nutrients, in theory at least it is possible to use changes in ecosystem structure to pick other changes. In practice however, the BMWP (Biological Monitoring Working Party) based system used was originally intended to pick up levels of organic contamination as well, so in effect the system was setup to monitor relatively few types of problem. This regular (monthly) monitoring strategy was supplemented by specific monitoring/assessment work intended to address indentified problems.

Under the Water Framework Directive, a new, tougher methodology is being introduced. This system of monitoring (classification) is risk-based and focuses on locations at which there is likely to be a problem. It also adopts the so called ‘one out, all out’ principle, which means that the poorest individual result drives the overall assessment. Critically, it is also based on a far wider range of assessments than GQA classification, and reports on over 30 measures, grouped into ecological status (including biology and ‘elements’ such as phosphorus and pH) and chemical status (‘priority substances’ - European Communities, 2001). The WFD also covers coastal waters, groundwater and lakes as well as rivers and estuaries.

These chemical/biological systems are in theory the best way of assessing impact, in that they assess that impact directly. However, in practice it is still difficult to attribute the observed patterns to specific farms or products due to the interconnectivity of catchments. Regular monitoring networks are also relatively sparse and generally placed at the downstream end of catchments (or sub-catchments), meaning that unless further investigation is carried out, it is difficult to be specific about the location of the cause of an identified impact.

**Indirect approaches:** Alternatively, there are qualitative systems for assessing the impact of agriculture on water quality, that consider the management protocols implemented to determine the site-specific steps taken to minimise the occurrence of problems. Being a major agr-environmental issue over the past few decades, there has been considerable emphasis on the adoption of best practice so as to avoid water quality problems. Nutrients (especially nitrogen) in a variety of forms (organic and inorganic) and pesticides have come in for the most attention, although other issues (including sediment issues, although these may be better covered in relation to soil quality) are also covered. In particular, there is a considerable volume of best practice advice relating to NVZs (e.g. Defra, 2009b-k), as well as advice from the VI amongst others relating to minimisation of pesticide problems. In addition, there are a number of Environmental Stewardship options that are specifically intended to reduce water quality problems (e.g. ELS options E1-E10 amongst others (Natural England, 2010)), as well as more general advice in the Code of Good Agricultural Practice (Defra, 2009a). Take up of these options and compliance with the appropriate regulations can be taken as an indirect indication of the likelihood of water quality problems occurring (or more accurately not occurring).

**Data availability**

As discussed in detail above, the main source of regular, quantitative water quality monitoring data is the Environment Agency though it’s GQA and now WFD monitoring programmes. This data is relatively good at its primary task, which is the identification of temporal trends, but is less good at identifying specific causes or in some cases specific pollutants. The use of a restricted set of determinants in the GQA system means that some problems may be missed, particularly those relating to pesticides for which all too often, chemical specific lab testing is required. Although their knock-on effect on biodiversity may be picked up by biological sampling, this is in general aimed at
picking up organic pollution, and other contaminants can skew the results without the root-cause being identified. These issues may be addressed (to some extent at least) by the updated WFD monitoring strategy, but this too has limitations in term of use within an eco-label. Firstly, it is if anything even more restricted in its spatial coverage due to being risk based and therefore focusing on areas in which problems are envisaged. This may be fine for the purpose for which it is intended, but if food produced on farms with a broad coverage is to be encompassed by the label, data on impacts over a wide area is also required.

The alternative is more generally available farm data, such as that on selected options in Environmental Stewardship or that required to comply with the requirements of legislation (or indeed assurance schemes). This data is held on-farm, and would therefore require limited additional effort on the part of the farmers involved. The disadvantage is that it is not a direct measure of problems occurring (or conversely being avoided), but rather just an evaluation of farm management processes which if adopted, current wisdom tells us will minimise the risk to surface and or groundwaters. In addition, there is data collated for the Defra Farm Practices Survey (e.g. Defra, 2005; Defra, 2006; Defra, 2007), which examines the percentage of sampled farms on which particular water quality issues are a problem, and what measures are being taken to prevent water pollution. The problems evaluated are however, those that are easy to see visually (e.g. discoloured runoff entering surface waters, and sediment deposited in watercourses) which is a very restricted set, that ignores a lot the most serious problems facing the industry. Nevertheless, the availability of the ‘practices’ data collated for this survey means that farmers are likely to have available, or relatively easily put together, information of this sort that could be used in an eco-labelling system if deemed appropriate.

2.5. Biodiversity

Background

The scientific use of the term biodiversity is quite specific, being defined, for example, as ‘the diversity of plant and animal life within a particular habitat’, or ‘the totality of genes, species and ecosystems of a specific region’. There is scope for impacts on biodiversity throughout the entire lifecycle of a product, but it could be argued that the greatest potential exists at the farm level due to the integrated nature of agroecosystems. Such impacts can be very broad and varied, and include, for example, the physical disturbance or degradation of habitats and ecosystems or the ecotoxicological effects that occur as a consequence of exposure to certain substances such as pesticides or pathogens.

The impacts of agricultural activities on biodiversity are well established (e.g. Henle et al., 2008; Natural England, 2010; Stoate et al., 2009) and include damage to aquatic and terrestrial ecosystems, invertebrates, plants, birds and mammals. Many declines, such as farmland bird numbers are well documented and recorded (Defra, 2009), however actually quantifying impacts on biodiversity is much more difficult, especially within the context of an environmental label for a food product.

‘Observed’ declines (or increases) in populations of species are derived from national or regional monitoring programmes and are therefore not directly associated with any particular farm, let alone food products. Some farms do monitor/survey wildlife on their farms, e.g. those which have a personal interest in conservation or in some instances as part of a scheme (e.g. Jordan’s Conservation Grade, which has a habitat assessment every 5 years to ensure the delivery of high quality habitats), but such farms are in the minority. Consequently, assessing impacts on biodiversity tend to be based
on activities associated with wildlife conservation or be based on potential impacts associated with the emission of certain pollutants.

**Available techniques**

In the absence of actual data on populations of different species, there are number of different techniques to help assess potential impacts. These include:

- Ecotoxicity
- Eutrophication (aquatic & terrestrial)
- Acidification
- Qualitative scoring systems, e.g. AMY (the Agrobiodiversity Management Yardstick)

**Ecotoxicity:** An ecotoxicological effect is an adverse change in the structure, or function, of a species as a result of exposure to a chemical. It is a complex area and involves understanding the exposure of different organisms to different substances and the hazard such an exposure presents to that organism. A range of parameters exist. Aquatic organisms such as fish use the LC$_{50}$, which refers to the concentration of a substance that is lethal to 50% of the fish within a specific time period, usually 96 hours. Similarly for terrestrial organisms such as mammals the acute oral LD$_{50}$ is the dose (mg per kg of body weight) that is lethal to 50% of the population. The EC$_{50}$ is the concentration of a chemical that can be expected to cause a defined non-lethal effect in 50% of the tested population, and the NOEL/NOEC which is the greatest concentration or level of a chemical, found by observation or experiment, which causes no detectable effect. Given these parameters, it is then possible to estimate what the risk might be for a number of different species in different environments for a given concentration of a chemical. However, determining emissions of substances can be very difficult, let alone their consequent concentration in the environment. In many production processes, especially those that are relatively closed and contained, the 'end-of-pipe' emissions are likely to be known. However, in more open systems, such as agricultural production, emissions to the environment tend to be more diffuse and highly variable due to site specific factors (e.g. climate, topography, soil type, temperature, farming practices, etc.), so amounts lost to different environments are more difficult to calculate. There are some quite sophisticated mathematical models available to help determine the fate of different substances in the environment and thus determine their concentration. For example, dispersion models used in air quality assessments (e.g. CAR-FMI (Härkönen et al., 1995); UDM-FMI (Karppinen et al., 2000); FLEXPART (Stohl et al., 1998)), or fate and transport models for pesticides applied to field crops (Green et al., 2009; Bergström & Jarvis, 1994; Dubus et al., 2009).

Modelling the fate and environmental concentration of every substance used in the production of a product is an onerous task. In many cases it is simply not practical or cost effective to do so or reliable models do not exist. Consequently a lot of effort has gone into developing characterisation factors for a range of substances, to simplify the ecotoxicological impact assessment process. A number of approaches have been developed to do this including, for example, expressing emissions of toxic substances as equivalents of 1,4-dichlorobenzene (Huijbregts, 2001) or triethylene glycol (Jolliet et al., 2003b) in a similar fashion to CO2 equivalents used for GWP.

There have been efforts to gain consensus on a more common approach (Hauschild et al., 2008) and these have resulted in the USETox™ model and database (Rosenbaum et al., 2008). However, the approach in USETox™ utilises Potentially Affected Fractions (PAFs) (see Section 3.2.5) as the characterisation factor. This approach aims to reflect the toxic stress on an ecosystem, so in theory it incorporates a degree of species sensitivity and exposure into the characterisation factor. However,
PAFs are a measure of damage and so combining them into the impact characterisation factor blurs the boundaries between different stages of the impact assessment making the process less transparent. PAF’s are discussed further in Section 3.2.5.

**Eutrophication (aquatic & terrestrial):** Eutrophication can be split into aquatic and terrestrial. Aquatic eutrophication is the result of nutrient enrichment (N or P) in aquatic environments. Under natural conditions, the supply of nutrients to water is in balance with the growth of biomass. Anthropogenic nutrient inputs can disturb this balance, leading to increases in algal growth that make the water turbid and decrease the level of oxygen content. This then leads, for example, to an increase in fish mortality and ultimately loss of fauna (Kristensen & Hansen, 1994). Increased loads of P are mainly responsible for eutrophication of freshwater. In saline estuarine habitats it tends to be nitrate. Terrestrial eutrophication is the nutrient enrichment of soils. Exposure of nitrogen-limited ecosystems to increased nitrogen loads often increases the competitive advantage of previously nitrogen limited plant species at the expense of those species adapted to low nitrogen containing soils.

In the context of an environmental assessment, and in the absence of any actual data of observed eutrophication, we are reliant on assessing likely impact based on emissions associated with particular processes or practices. In LCIA emission of nutrients can be converted to kg PO₄-eq and kg NOₓ-eq for aquatic and terrestrial eutrophication respectively. Table 7 shows the method applicable to aquatic eutrophication. Agricultural sources of significance include nitrate (NO₃) from inorganic and organic fertiliser use and phosphate (PO₄) from surface run-off and soil erosion. Emission of ammonia (NH₃) to air results from urea based fertilisers and livestock manures, liquid manures in particular. The magnitude is dependent on the method of storage, and timing and method of application (e.g. surface or injection) of liquid manures to agricultural land.

**Table 7: Aquatic eutrophication impact sub-category: characterisation factors (aquatic eutrophication potentials) for N and P emissions** (from Brentrup et al., 2004)

<table>
<thead>
<tr>
<th>Substance (in kg)</th>
<th>Aquatic Eutrophication Potential (in kg PO₄ equivalents per kg emission)</th>
</tr>
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<tbody>
<tr>
<td>N</td>
<td>0.42</td>
</tr>
<tr>
<td>NH₃</td>
<td>0.35</td>
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<tr>
<td>NH₄</td>
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</tbody>
</table>

A comprehensive database of mean Europe eutrophication factors is provided by the University of Leiden (2009). Brentrup et al. (2004) also provide regionalised characterisation factors for terrestrial eutrophication. For example, in the UK 1 kg NOₓ emitted = 0.76 kg NOₓ-e and 1 kg of NH₃ emitted = 1.70 kg NOₓ-e (the reference region is Switzerland, where the factors are 1.00 and 5.00 respectively).

Brentrup et al. (2004) used normalisation values to place the impacts in context to better understand the magnitude for each indicator result of the product system under study. They use values based on Europe and they have been calculated per person.
**Acidification:** Acidification refers to an increase in acidity, the hydrogen ion concentration, in water and soil systems. Acidification is principally caused by emissions of sulphur dioxide (SO$_2$), nitrogen oxides (NO$_x$) and ammonia (NH$_3$). SO$_2$ mainly arises from combustion of sulphur-containing coal and oil, NO$_x$ from combustion processes in vehicles, and NH$_3$ predominantly arises from animal husbandry and can also arise from volatilisation during and after application of urea and ammonium-containing fertilisers.

Acid deposition has negative impacts on terrestrial and aquatic ecosystems including primary production, fish production capacity, base cation capacity of soils, and contribution to species extinction. The actual impact will vary from one region to another according to the deposition pattern and the susceptibility of the receiving area to acidification.

In LCIA emissions of SO$_2$, NO$_x$ and NH$_3$ can be converted to SO$_2$ equivalents. Similar to terrestrial eutrophication, different characterisation factors exist for different regions to reflect the different deposition pattern and the susceptibilities to acidification. For example in the UK 1 kg of SO$_2$ = 0.86 kg SO$_2$e, 1 kg of NO$_x$ = 0.43 kg SO$_2$e and 1 kg of NH$_3$ = 1.5 kg SO$_2$e (Brentrup et al., 2004).

**Qualitative scoring systems:** The impact categories above assess the impact of emissions of substances on the environment, and generally negative impacts. However, there can be a significant impact on biodiversity (positive and negative) from managing the actual environment at the farm level, both in terms of cropped and non-cropped areas. However, quantifying any potential impact is not simple in the absence of monitored impacts on biodiversity. One approach to overcome this is to create a scoring system for various activities and practices that should (or can) result in positive (or negative) impacts with respect to biodiversity.

An example is the Agrobiodiversity Management Yardstick (AMY) developed in the Netherlands (van Amstel et al., 2007a & 2007b). AMY uses a ladder of abstraction (Sartori, 1991) to create links between agrobiodiversity policy goals and the concrete level of management measures on a farm. Figure 2 shows four levels of abstraction, with the 5th level containing around 140 on-farm management measures that positively affect agrobiodiversity. A group of experts are used to construct this abstraction and score each management practice for its efficacy in relation to the positive impact on agrobiodiversity and the extent to which it contributes to conservation and sustainable use of agrobiodiversity. The AMY approach tends to focus on positive impacts and therefore does not necessarily reflect any potential impact of agronomic practices on biodiversity. A similar scoring approach was developed by Lewis et al. (1997) to score and rank farmland conservation activities. This approach used an eco-rating system where the normalised score could range from -250 to +250, thus both negative and positive management practices could be captured.

**Data availability**

With regard to ecotoxicology, eutrophication and acidification a range of data would be needed. With respect to agricultural production quantification of inputs such as pesticides and nutrients would be required. This is very field specific but might be available, at least in the UK, as this is usually recorded by the grower. However, accessing that information may not be easy or practical. For eutrophication and acidification characterisation factors are available in the various databases mentioned previously. With respect to ecotoxicity data regarding toxicological endpoints (e.g. LD50, LC50, NOEC etc.) would be required for toxic substances such as pesticides and there are various excellent sources of this information particularly with respect to pesticides authorised for use in Europe (e.g. EU 91/414 Conclusion documents (EFSA, Undated) and the Pesticide Properties Database (Green et al., 2009)).
However, even if the data can be obtained to help quantify or predict ecotoxicology, eutrophication and acidification the question still remains on how this is related back to biodiversity and the impacts assessed.

Figure 2: The AMY, the first four levels of abstraction (from van Amstel et al., 2007a)


2.6. Stratospheric ozone depletion

Background

Stratospheric ozone helps to filter out/absorb ultraviolet light (wavelength ~300 nm), preventing it from reaching the Earth’s surface. Depletion of stratospheric ozone results in greater exposure to UV radiation. The most established impacts of this increased exposure are associated with human health, especially increases in skin cancer and cataracts. Other potential impacts include crop damage, immune system suppression, damage to materials like plastics, and marine life damage (Jolliet et al., 2004; UNEP, 2003).

Depletion of stratospheric ozone has been observed since the 1970’s, with the most noticeable effects in the southern hemisphere and over Antarctica. Emissions of chlorine and bromine-containing substances (halocarbons) such as chlorofluorocarbons (CFCs), halons and hydrochlorofluorocarbons (HCFCs) have been identified as being responsible for stratospheric ozone depletion along with methyl chloroform, carbon tetrachloride and methyl bromide. The most severe ozone depleting chemicals are now mostly banned as a consequence of the Protocol of Montreal in 1987. The importance of this impact category may therefore be diminishing (Pennington et al., 2004).

Available techniques

The midpoint impact category for stratospheric ozone depletion is Ozone Depletion Potential (ODP). Similar to other midpoint impact categories this is expressed in relation to a reference substance, CFC-11. Table 8 presents a list of substances and their ODP in kg CFC-11 equivalents.
Table 8: Ozone Depletion Potentials (IEC, 2008)

<table>
<thead>
<tr>
<th>Species</th>
<th>Formula</th>
<th>ODP [kg CFC-11 eq./kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bromo-methane</td>
<td>CH₃Br</td>
<td>2.30</td>
</tr>
<tr>
<td>CFC-11 (Trichlorofluoromethane)</td>
<td>CFCI₃</td>
<td>1</td>
</tr>
<tr>
<td>CFC-113</td>
<td>C₂F₃Cl₁</td>
<td>0.59</td>
</tr>
<tr>
<td>HALON-1211</td>
<td>CClF₂Br</td>
<td>9</td>
</tr>
<tr>
<td>HALON-1301</td>
<td>CF₂Br</td>
<td>10.50</td>
</tr>
<tr>
<td>HALON-2402</td>
<td>C₃F₃Br₂</td>
<td>11</td>
</tr>
<tr>
<td>HCFC-123</td>
<td>CHCl₂CF₃</td>
<td>0.08</td>
</tr>
<tr>
<td>HCFC-124</td>
<td>CHFCICF₃</td>
<td>0.08</td>
</tr>
<tr>
<td>HCFC-141b</td>
<td>CH₃CFCI₂</td>
<td>0.33</td>
</tr>
<tr>
<td>HCFC-142b</td>
<td>CH₃CF₂Cl</td>
<td>0.14</td>
</tr>
<tr>
<td>HCFC-22</td>
<td>CHF₂Cl</td>
<td>0.14</td>
</tr>
<tr>
<td>HCFC-225ca</td>
<td>C₃HCl₂F₅</td>
<td>0.10</td>
</tr>
<tr>
<td>HCFC-225cb</td>
<td>C₃HCl₂F₅</td>
<td>0.11</td>
</tr>
<tr>
<td>Tetrachloromethane</td>
<td>CCl₄</td>
<td>1.23</td>
</tr>
<tr>
<td>Trichloroethane</td>
<td>CH₃CCl₃</td>
<td>0.45</td>
</tr>
</tbody>
</table>

Data availability

In order to be able calculate this impact, emissions of any ozone depleting substances arising from the life cycle of the product need to be known. Some emission factors are available in the University of Leiden database (2009). For specific processes quantities used of particular ODP may be available or measurable. Detailed life cycle assessments usually include the ODP of a product, for example crop production (Williams et al., 2009) or specific products such as polyethylene (Bousted, 2003). For agricultural products the main contributors were previously soil fumigants that have been either banned (e.g. methyl bromide) or are being phased out (e.g. chloropicrin). Refrigerants are also potential sources during post harvest storage and processing.

2.7. Resource depletion

Background

An increasing global population needs ever more resources. Generally, resources can be classified into those that are biotic and those that are abiotic. However, classifying resources in this manner is not as simple as may first appear and, as well as their origin, their potential for regeneration must also be considered.

At a basic level biotic resources are the living resources. There are both renewable and non-renewable biotic resources. The renewables include forests, plants and animals. However, resources such as sunlight, wind and air also fall into the biotic category. Fossil fuels such as coal, natural gas and petroleum are also biotic as they are formed from decayed organic matter. However, due to the time it takes for such resources to form (i.e. long geological periods, often millions of years) they cannot be replenished once they become depleted and so are normally considered to be non-renewable. The abiotic resources are non-living resources such as minerals and metals. These resources cannot regenerate by themselves and so are often called non-renewable resources.

Water and land are clearly resources but their classification within LCA studies and how they are handled varies significantly and usually depends on the individual study and its purpose. In many
instances, water and land are considered to be biotic. However, the definition of the term biotic may include reference to their ‘living status’ and using this definition all waters and land would be classified as abiotic as they themselves are not living but a support media for living things. Strictly speaking and from a global perspective, water is renewable although it can become scarce in some parts of the world, and consequently overexploitation of water resources in those areas becomes a more significant issue than in wetter areas. There is also the issue of certain geological water bodies, for example fossil waters, (also known as paleowater and are underground water reservoirs which have been geologically sealed) that cannot be replenished once used and so should be considered as non-renewable.

To confuse the issue further the distinction between renewable and non-renewable resources is also unclear. For example, sand would normally be considered to be abiotic and non-renewable. However, there are examples when this definition does not hold true. For example, sand extracted from a river bed can be replenished quite rapidly by the action of water and weather on rocks. Consequently, many classification systems define non-renewable resources to be those that cannot be replenished within a given time horizon e.g. 500 years (van Oers et al., 2002)

Overexploitation of renewable resources can have an impact on their future availability and also on ecosystem functioning, such as population decrease and species extinction. These issues are usually addressed in LCA studies under the biodiversity impact category. It is, therefore, the biotic and abiotic non-renewable resources that tend to be addressed in the LCA resource category. The potential scarcity of the non-renewable resources and competition over their use has caused controversy for more than a century. Easy access to these resources is often seen as a precondition for economic development. According to Steen (2006), with the exception of climate change, the depletion of non-renewable resources poses the largest environmental threat to our daily lives. Abiotic resource depletion is one of the LCA impact categories Guinée et al. (2002) considered important to any environmental impact assessment.

In some instances the depletion of non-renewable resources is a global issue, for example, fossil fuels. In other instances, the impacts caused by resource depletion can be very local. As the global population increases, the pressure on both renewable and non-renewable resources is increasing, so using them efficiently is vital for sustainable consumption.

Available techniques and data availability

Non-renewable resource depletion is one of the most widely discussed and published impact categories (vanOers et al., 2002; Guinée & Heijungs, 1995; Lindeijet et al., 2002) and, consequently, there are many techniques available for characterisation. Quantification, i.e. the mid-point impact, is rarely an issue as it may simply be a measure of how much of a specific input is used. The problem lies in the desire to aggregate the different resources on to a common scale rather than simply listing the consumption of every resource used in the life cycle of a product and how to relate the quantities used to the rate of depletion and environmental damage.

One approach, the CML method (Guinée, 2001), aggregates different resources using their Abiotic Resource Depletion Potential (ADP), where antimony is used as a reference substance (ADP is expressed in kg antimony-equivalents), based on the scarcity of reserves. However, Brentrup et al. (2002) highlight that this neglects to consider that many resources are used for different purposes and are not equivalent to each other. Therefore, the depletion of reserves of functionally non-equivalent resources should be treated as separate environmental problems. Brentrup et al. (2002) develop the concept of grouping resources based on their function, e.g. the use of oil, natural gas
and coal as energy sources, and then expressing use of those resources in MJ, as a means of aggregating the impacts.

Steen (2006) drawing on the research of others discusses four main methods of characterising the rate of depletion based upon:

- Energy or mass – simply the total quantity used. However this suggests all resources are equal and exchangeable.
- The relationship between the rate of use and available deposits – Flava et al. (1993) discuss this approach in detail. In summary, characterisation factors or indicators are used to represent the rate of use and the mass of known or anticipated deposits. Some authors (e.g. Guinée & Heijungs, 1995) also suggest considering the economics of extraction or the total amount of the specific resource in the earth’s crust. Brentrup et al. (2002) proposed a distance to target approach (see Section 3.2) that determines a target rate of consumption based an estimate of total resource reserves and the need to make the reserve last a certain period of time e.g. 100 or 1000 years.
- The future consequences for extraction – this approach assumes that unsustainable consumption now will create greater problems for the environment and world economics in future generations. Various methods of quantifying this have been proposed including assessing the impacts of over extraction in other resource categories (Weidema, 2000), considering impacts based on the increased energy requirements of future generations (Müller-Wenk, 1998).
- Exergy consumption or entropy production. Processing abiotic resources requires energy and Finnveden (1994) proposed a method based upon exergy used when producing raw materials assuming that exergy is the ultimate limiting resource as production will not be undertaken when the cost of energy becomes too high.

It is clear that these types of approaches are subject to huge uncertainties and inaccuracies, and lack in transparency and scientific credibility. The importance of a specific resource is something not resolved full in the scientific debate. This is significant as the quantities of resources used and even improvements to resource use efficiency do not necessarily reflect the impacts of resource use, especially with respect to the use of significantly depleted reserves or where there are local issues, such as scarce water resources. In these instances production can still have high efficiency, but still be drawing upon resources in an unsustainable fashion.

Another important aspect rarely considered is resource use efficiency. By presenting the resources used per unit of product, e.g. per tonne of wheat, production techniques can be pursued which result in greater output per unit of resources input, e.g. increasing outputs using the same or less resources and/or reduce resource inputs while maintaining or increasing output.

### 2.8. Waste and recycling

**Background**

Waste and recycling are issues that are often seen as key for many environmental assessments but in many cases they may not be impacts per-se. The production of ‘waste’ could instead be treated as either an emission (the impacts of which would generally be considered under other categories such as air pollution resulting from landfill sites or human health in the case of some hazardous wastes) or a by-product. Recycling is an activity to handle any such emissions or by-products and which is intended to place less demand on ‘virgin’ resources. This would be addressed and encompassed within the resource use impact category.
Equally of course, the concept of ‘waste’ is less clearly defined within an agricultural situation, with some wastes performing a dual role as resource. Manures and slurries, for example, can be viewed as either a waste requiring disposal, or an invaluable resource for the maintenance of soil fertility, depending on the specific circumstances of a particular farm business. Nevertheless, agriculture (like any other form of production) does produce some wastes for which other forms of accounting may not be appropriate, including (but by no means restricted to):

- empty pesticide containers and fertiliser bags;
- old silage wrap and other farm plastics;
- out of date pesticides, livestock medicines and anthelmintics;
- used tyres, obsolete machinery / equipment; and
- surplus milk.

For many years agriculture was, in the main, beyond the scope of waste legislation, but since 2006 it has been subject to the same controls as other industries, which has in particular put an end to the uncontrolled burning or tipping of wastes on farm land. There are of course some exemptions to this (generally those considered to be of low risk, which can still be carried out once a suitable permit has been issued (Environment Agency, 2009)); but nevertheless, the legislation means that farmers have a duty of care to manage their waste in a responsible way, which includes storing it securely, ensuring it is passed on to a registered waste ‘handler’, and perhaps most importantly (in terms of the present study at least), to maintain suitable documentation relating to waste movements (transfer notes for non-hazardous waste and consignment notes for hazardous waste).

**Available techniques**

The legal requirement to maintain records of waste movements, discussed above, means that, potentially at least, farm businesses do have reasonably detailed records of many of the wastes they produce, minimising the need to maintain specific records for eco-labelling purposes. However, the extent to which these can be attributed to individual crops is debatable, and there is little if any literature available on this subject, since there has, to date, been little incentive to do so. Wastes clearly resulting from the production of specific crops (e.g. pesticide containers) may be relatively straightforward to apportion in theory at least (although the extent to which sufficiently detailed data is maintained is questionable), but those that result from multiple activities (such as those associated with the use of agricultural vehicles) may be more difficult to apportion. Nevertheless, if waste disposal quantities can be apportioned to farm yields some estimate of their environmental burden (per unit for example) may be possible.

The process is however, complicated still further by the diverse nature of the wastes concerned. The actual, environmental impact of waste generation as a whole is highly variable, being dependent on both the make-up of that waste, and the disposal methods adopted (e.g. landfill, incineration, recycling). Since it is highly probable that even individual businesses will produce a variety of wastes and utilise several ‘disposal’ techniques, it is unlikely that waste could be integrated into a labelling system in anything other than a generalised form. Although this may be adequate for distinguishing between final products.

There are very simple techniques for assessing and communicating waste production and these include, for example, reporting it on weight (or volume) per unit of production or per unit consumed. In order for this to be meaningful it may be necessary to characterise wastes into different types. This could be as simple as hazardous / non-hazardous wastes, or controlled / uncontrolled. Other
options include classifying them on the basis of their recycling or re-use potential. There are also more complex options such as classifying wastes into more distinct categories such as paper, glass, etc. or with respect to hazardous wastes according to their properties (see Table 9). This last option would then identify a priority for further assessment.

**Table 9: Classification of hazardous wastes based on their properties**

<table>
<thead>
<tr>
<th>Property Code</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>H1</td>
<td>Explosive substances and preparations</td>
</tr>
<tr>
<td>H2</td>
<td>Oxidising substances and preparations</td>
</tr>
<tr>
<td>H3</td>
<td>Highly flammable substances and preparations</td>
</tr>
<tr>
<td>H4</td>
<td>Irritant non-corrosive substances and preparations</td>
</tr>
<tr>
<td>H5</td>
<td>Harmful substances and preparations</td>
</tr>
<tr>
<td>H6</td>
<td>Toxic substances and preparations</td>
</tr>
<tr>
<td>H7</td>
<td>Carcinogenic substances and preparations</td>
</tr>
<tr>
<td>H8</td>
<td>Corrosive substances and preparations</td>
</tr>
<tr>
<td>H9</td>
<td>Infectious substances containing viable micro-organisms or their toxins which are known or reliably believed to cause disease in man or other living organisms</td>
</tr>
<tr>
<td>H10</td>
<td>Teratogenic substances and preparations</td>
</tr>
<tr>
<td>H11</td>
<td>Mutagenic substances and preparations</td>
</tr>
<tr>
<td>H12</td>
<td>Substances and preparations which release toxic or very toxic gases in contact with water, air or an acid</td>
</tr>
<tr>
<td>H13</td>
<td>Substances and preparations capable by any means, after disposal, of yielding another which possesses any characteristics listed above</td>
</tr>
<tr>
<td>H14</td>
<td>Ecotoxic substances and preparations which present or may present immediate or delayed risks for one or more sectors of the environment.</td>
</tr>
</tbody>
</table>

Alongside the list of hazardous properties there is the European Lists of Wastes (formerly known as the European Waste Catalogue (EWC)) that lists all wastes grouped together according to their generic industry or process. It is a hierarchical list of waste descriptions established by Commission Decision 2000/532/EC (EC, 2000). It is unlikely that a classification this detailed would be necessary.

**Data availability**

Nationally, data on both waste disposal methods used for a number of types of farm waste (Defra, 2001, 2004, & 2008a) and specific data on some elements of recycling (e.g. the recycling of non-packaging plastic products (Defra, 2009II)) is collected for the Defra Farm Practices Survey. However, although this data is differentiated in terms of farm type and size, and region, it is based on the percentage of holdings using particular disposal methods, rather than the amounts of generated by those farm types. Consequently, this survey data may be of limited use in providing all but a general overview of this type of impact. However, the fact that it is possible to publish this data, means that more accurate data should be obtainable if required for the eco-labelling of food products, particularly if provided by individual farms.

As discussed above farm businesses are now legally required to maintain records of waste movements from their land, and as such data clearly does exist on a per business basis for many types of waste. As a result, there would be only limited additional administrative burden in making this data available to an eco-labelling scheme. However, this does not cover all forms of waste generated by a business (for example those disposed of on farm under exemptions from the
legislation), and there is as yet no clear method for apportioning many forms of waste to production, but potential for development exists (see above).

2.9. Landscape and heritage

Background

Landscape and heritage impacts cover aesthetic and archaeological/historic issues respectively, and as such many impacts of this type are highly site specific. They also engender a high degree of interest within the general public, as evidenced by the number of local and national pressure groups relating to the aesthetic quality of the rural landscape (e.g. the Campaign to Protect Rural England). The importance of landscape and heritage is also reflected in EU and UK agricultural policy; for example, a number of options within Environmental Stewardship and some of the regulations in cross compliance are, directly (e.g. ELS options ED1-ED5 (Natural England, 2010a); GAEC 7 (RPA & Defra, 2008) or indirectly (e.g. ELS options for boundary features (Natural England, 2010a)), designed to bring about benefits with respect to the enhancement and protection of landscapes and preservation of archaeological features.

They have been grouped together here, as a reflection of the fact that although they are both important in terms of public interest (and the reputation of the agricultural industry), neither can necessarily be directly related to ecosystem functioning. In some instances a diverse and scenic landscape can also have benefits for biodiversity and vice versa. For example, the loss of rural hedgerows and stonewalls during the 20th century has had impacts on both biodiversity and the landscape. This however, cannot be assumed to be the case, since a visual degradation may in fact have little or no impact on biodiversity. Nevertheless, Haas et al. (2000), point out that the OECD (1997) identified ‘landscape’ as a key agri-environmental issue and they suggest that ‘landscape image (aesthetics)’ should be included in any agricultural life cycle assessment (LCA).

Available techniques

Impacts of this sort are extremely difficult to quantify, particularly within the framework of a life cycle assessment for a product, not least because many of the relevant elements are both qualitative in nature and site specific (the definition of an appropriate landscape will vary depending on geographical location, and historic features may be highly localised). It is also very subjective and many may also be heavily influenced by personal preference. Nevertheless, there have been attempts to incorporate landscape elements into the assessment process, usually using some form of scoring system. Haas et al. (2000), for example, selected suitable indicators for the impact category ‘landscape image’ based on the interests of people in the region (Allgäu, southern Germany), such as those expressed as the main local characteristics in tourist information booklets; something that may be difficult to do on a national scale.

Haas et al. (2000) also raised the issue of the ‘functional unit’, and considered that with respect to ‘landscape image’ (amongst other impacts), the only meaningful functional unit was the ‘whole farm’ rather than the unit of production (e.g. on a per hectare basis). If this argument is accepted, then it could have serious implications for the integration of landscape features into an environmental food label, for which impacts need to be related to the product. Unless that is, the label only reflects the overall performance of the farm from which the produce originates.

Data availability
Landscape issues being highly subjective, do not lend themselves to quantitative data collation. As a result, it is likely that if such issues were to be integrated onto an eco-label for food, then this could only be done based on indirect measures of performance. In particular, this may be related to the adoption of those Environmental Stewardship options (or equivalent) that relate to the maintenance of aesthetic properties. This is far from ideal, since a relatively large number of options could be argued to fall into this category, despite the fact that their primary aim may be related to other issues, for which they may be evaluated under other LCA categories. It also raises the issue of how farmers are assessed who may not be carrying out relevant work under an official scheme (but are doing it anyway), and whether farmers within particularly ‘aesthetically pleasing’ areas are at an advantage, since others may not be in a position to score highly in this area.

There are records of archaeological features maintained by each County/Unitary Authority in the UK, in the form of their ‘Sites and Monuments Record (SMR)’, even though (in England and Wales) there is no legal requirement for such records to be maintained (they are however mentioned in guidance - e.g. Planning Policy Guidance 16: Archaeology and Planning (ODPM, 1990)). Additionally, many such records are being extended to cover a broader range of sites in what are termed ‘Historic Environment Records (HERs)’. However, this data only identifies the location of important or significant sites, it does not reveal how they are being managed. It may be that the only realistic way of including this form of impact (if it is deemed necessary to do so at all) is to do so in a general (whole farm) way based on take up of Environmental Stewardship options (or similar), particularly those options which area specifically intended of benefit to archaeological features, (e.g. ELS options ED1-ED5 and HLS options HD6-HD11 (Natural England, 2010a,b)). However, many of the problems discussed above in relation to landscape can also be related to archaeological and historical sites, in that it may be inappropriate to ‘reward’ produce that is produced on farms on which there are sites of historical interest, when produce from farms with no such sites cannot score well in this sense.

2.10. Noise, dusts and odours

Background

Noise (as an environmental pollutant as opposed to a health and safety issue), dusts and odour are elements commonly included within the context of an environmental impact assessment (EIA), particularly for projects relating to construction and/or engineering, and are generally dealt with under the over-arching heading of nuisance. This corresponds to their legislative standing under the Environmental Protection Act 1990, Part III Section 79 of which defines as a statutory nuisance (amongst other things):

- ‘any dust, steam, smell or other effluvia arising on industrial, trade or business premises and being prejudicial to health or a nuisance’; and
- ‘noise emitted from premises so as to be prejudicial to health or a nuisance’.

In relation to the on-farm elements of the food production life cycle, there is clear scope for the generation of these impacts, particularly in relation to the more intensive forms of production.

With respect to noise, the intensive pig and poultry sectors have come in for particular attention, and ‘emissions’ generated by sufficiently large establishments in these sectors are covered by the Integrated Pollution Prevention and Control (IPPC) system (Defra, 2005a). However, the potential to cause ‘noise’ problems can be related to all agricultural businesses, as even relatively extensive farming activities often occur in the open environment, meaning that neighbouring residents and/or businesses are exposed to them (e.g. use of agricultural machinery, irrigation systems, etc.). Legally,
it is a requirement not to cause a nuisance to those ‘neighbours’ (although in most cases there is no clear definition of what constitutes a nuisance), and it is the responsibility of local authorities to enforce those regulations.

With respect to dust, exposure can result in severe health effects such as impairment to respiratory function, chronic bronchitis, occupational asthma, Toxic Organic Dust Syndrome, and Extrinsic Allergic Alveolitis. According to the HSE these conditions are not uncommon to many agricultural workers and they report that a review of ill health in agriculture identified a high prevalence of respiratory symptoms in poultry farm workers, who can be exposed to significant amounts of airborne dust (poultry dust) generated during poultry work activities (HSE, 2009). Dust also contributes toward air quality issues.

The emission of offensive odours is related to the issue of air quality (see Section 2.2), since some of the gasses which result in odour problems (such as ammonia and hydrogen sulphide - both of which can be generated from livestock manures) are toxic and/or harmful to the environment. However, there are both odourless harmful gasses and sources of odour problems that result in no other significant issues, so the two cannot quite be covered under the same impact category. Key sources of odours include (BS5502: Part 33 (BSI, 1991)):

- spreading of livestock wastes to land,
- spreading of sewage sludge to land,
- manure/slurry/waste storage,
- livestock buildings,
- production of animal feed,
- storage of silage, and
- the composting of food waste (although this can reduce the emission of odours on subsequent spreading – Defra, 2009a).

It could be argued (and has been) that ‘farm smells’ are nothing more than a normal part of rural life, and there is some validity in this argument. However, although it is no doubt the case that the movement of people between the urban and rural environment has led to some additional complaints, the increased intensity of UK agriculture (and the drive for increased composting) has also caused greater problems in some areas in recent decades.

**Available techniques**

Haas et al. (2000) argue that the impact category of nuisance (particularly noise and odour) should be excluded from agricultural LCAs, on the grounds that they are too subjective, and that as they say "agricultural smells and sounds are part of the rural image in rural areas and perceived indifferently by the people". However, this is somewhat inconsistent, in that the same authors consider that ‘landscape’ issues should be included even though they too are subjective. Other authors consider that such issues should be incorporated into LCA, including Udo de Haes et al. (1999) who suggest that they should be considered under the impact category of human toxicity, and Zobel et al. (2002). However, these authors do not propose techniques for doing this.

Specific noise impacts have received attention in previous LCA studies, such as that relating to the impact of road traffic noise on human health (Mueller-Wenk, 2002), but Jolliet et al. (2004) point out that as yet LCA inventories don’t contain data on noise emissions. Quantified noise limits are mentioned guidance (e.g. daytime LAeq <50dB and a night time LAeq <45dB is required to avoid annoyance), but what is or is not acceptable will in part be dependent on background levels (>10dB...
above background being likely to result in complaints). Consequently, in environmental terms at least, it is difficult to set hard and fast limits (health and safety limits can be set, but these are considerably higher). Jolliet et al. (2003a) propose that traffic and/or industrial noise could be expressed as a local impact and measured as an equivalent intensity of noise over a certain threshold (in dB), per individual; but this may be less applicable in agricultural situations, in which some loud noises may be intermittent, and (being in rural areas) may affect relatively few people. The IPCC regulations state that the number of people affected should not be a relevant factor, since rural populations are as deserving of protection as those in more densely populated areas.

There have been attempts to set quantitative odour thresholds based on the responses of ‘odour panellists’ (such as the Dutch malodourous air thresholds from 1989 (Pennington et al., 2004)), and it has been proposed that such systems be used as the basis for impact factors (Guinée, 2002). However, it is unclear to what extent this could be done for odours of uncertain make-up, since our understanding of what constitutes an appropriate level, is somewhat subjective (being heavily dependent on personal preferences). To a considerable degree the decision as to whether a given level constitutes a ‘nuisance’ is down to the regulators (as it is for other sources of ‘odour nuisance’ (Defra, 2009b)). However, factors that may be taken into account in the decision process include whether or not an obvious change in behaviour is caused (e.g. avoiding garden use, closing windows) as well as the proportion of the affected population making complaints, and their ‘odour diaries’. Like for noise, the decision will have to take into account the frequency and duration of odour episodes or events, as well as the characteristics of the odour and the numbers of people affected.

Such subjective assessments of course offer little in terms of a methodology for carrying out general assessments of the environmental impact of food (or other) products, although on a more limited basis it may be possible to consider noise and odour within an environmental impact assessment, although probably in qualitative terms.

Assessing dust is a very local issue as releases are often linked to local practices, the condition and operation of machinery. Whilst not as subjective as odour and noise, there are no widely accepted techniques identified for assessing dust that could be used on a product label. Some companies that specialise in dust consultancy and offer monitoring services do use two parameters:

1) Absolute Area Coverage: this indicates the magnitude and significance of dust sources using defined threshold levels.
2) Effective Area Coverage: this is a measure of the nuisance potential of the dust.

No examples of these types of techniques being used within an LCA type study have been identified. No scientific evaluation of such an approach has been identified.

Emissions could, of course, simply be measured and compared with maximum exposure limits, where these exist. However, how these measures could be related to impacts on human health and air quality is unresolved.

**Data availability**

It would appear that there is little data available on the agricultural emission of noise, dust or odour nuisance, as none are routinely assessed on-farm or easily quantified. Local authorities (and potentially farm businesses themselves) may have records of any complaints made and/or upheld against specific businesses. This however, cannot be taken as a quantitative assessment of the environmental impact of a business, as many aspects of the procedure leading up to a complaint are
subjective in nature (e.g. the personal tolerances of the complainant) and/or related to the site specific characteristics of the receiving population (e.g. size of local population).

Alternatively, farms (particularly in the intensive poultry and livestock sectors), may take a number of steps to comply with legislation and planning constraints, in relation to the control these nuisances from their businesses. For noise this may include the use of suitable, well maintained equipment and/or acoustic barriers (Environment Agency, 2005; Metcalfe, 1999a,b), and for odour it may involve covering or allowing crusts to form on slurry stores, keeping poultry manure dry, and ensuring high standards of hygiene and cleanliness, and injecting slurry rather than spreading it (Defra, 2009a). However, complying with such legislative and/or best practice guidance provided little basis on which to distinguish between businesses, since in general all should be complying with these requirements, and may offer little or nor benefit beyond that associated with traditional assurance schemes.
3.0. Assessing environmental damage

3.1. Introduction

The information provided in Section 2.0 described how the environmental effects of a product over its entire life-cycle may be measured. It also describes the limitations of these techniques and the methods available to aggregate effects into midpoint impact categories. Whilst many consider that mid-point impacts are just an interim stage of LCA and should be followed with an assessment of environmental damage, the characterisation of mid-point impacts is valuable as it provides an overview of the environmental profile of a product that allows it to be compared with other similar products - if the study boundaries, methods, data and assumptions are also comparable. With respect to environmental product labelling many schemes (e.g. some carbon labelling schemes that just state the level of CO₂ emissions per product unit) stop at this point and do not attempt to interpret damage or provide guidance for consumers. The reasons for this are varied but are often related to the lack of scientifically credibility and/or standard techniques for damage interpretation and communication.

The literature review carried out in Approach 1 and the findings from the consumer and industry workshop clearly showed that an environmental label needs to be simple to understand, scientifically credible and effective. However, evidence also suggests that satisfying all three criteria is exceedingly difficult.

If just mid-point impact data is provided to the consumer then the evidence base assembled in previous stages of this project indicate the ability of the label to enable consumers to make informed purchasing choices may be compromised. In the majority of situations such choices can only be made by comparing one product with another like product. Where a cross product comparison cannot be made the consumer would need to decide if the impact level was acceptable or not. To do this they (ideally) also need to know:

• How an impact relates to environmental damage;
• What the assessment criteria is; and
• What levels are tolerable before unacceptable or irreversible levels of damage are occurred.

Data interpretation can be further complicated if it refers to scientifically complex impacts (e.g. eutrophication or photochemical smog) which the lay person may not fully understand, if the label contains information on more than one impact or if there is aggregation or if values are highly variable across like products.

Some labelling systems do seek to provide mid-point impact interpretations and use communication techniques to aid consumer decision making. Most of these utilise techniques that have been developed for Life Cycle Assessment (LCA) studies. There are a number of such techniques each having their own advantages and limitations. And is arguably a very challenging of process. It can also be the least scientifically sound due to the lack of hard data, large areas of uncertainty and problems associated with interpretation and weighting (Powell et al., 2008).

Some techniques just seek to place mid-point impacts on a scale or compare it to a reference point (i.e. normalisation) whilst others introduce a further step which attempts to predict environmental damage from mid-point impacts (e.g. predict the incidence of cancer from exposure to a carcinogenic chemical or the percentage of fish population killed by a chemical concentration in a water body) in order to identify the significance of the impact before then placing the damage on a scale. Others use
valuation techniques (e.g. cleanup costs or willingness to pay) to convey meaning to a mid-point impact or environmental damage.

There are several common approaches which have been ‘packaged up’ into ‘named’ techniques and models. The common approaches are related to scaling and aggregation and these are discussed in Section 3.2. Some examples of different ‘named’ techniques and models that may use one or more of these common approaches are discussed in Section 3.3.

### 3.2. Common Approaches

There are several approaches that can be used to interpret mid-point impacts in order to express them as environmental damage such that they become more meaningful to consumers and laypersons. These include techniques for converting impact data into some other format that the consumer may better relate to and methods for aggregating impact into ‘damage categories’ to reduce the data presented to consumers and aid interpretation (e.g. combining various air pollutants, photochemical smog etc. into a single ‘air quality’ category or combining disease incidence, respiratory effects etc. into a ‘human health’ category). Some of the more common approaches and their advantages and disadvantages are discussed below.

#### 3.2.1. Distance To Target approach

In the Distance-to-Target approach (Seppälä1 & Hämaläinen, 2001) individual environmental impact data (mid-points or endpoints) are evaluated depending upon the distance between the current level and a future target level. The significance of the impact is described on the basis of the extent to which actual environmental performance deviates from some goal or standard. As described by Powell et al. (1997), the method ranks impacts as being more important the further away society is from achieving the desired standard for the pollutant and, thus, creates a weighting system. Soares et al. (2006) define the Distance-to-Target approach as the ratio between the actual level of an emission within a specified geographical area and the level considered to be critical (target value) for this emission. The method defines the weighting factor ($W_i$) as:

$$W_i = \frac{A_i - T_i}{T_i} \quad \text{Or} \quad W_i = \frac{A_i}{T_i}$$

Where:

- $A_i = \text{current value of impact category } i$
- $T_i = \text{target reference of impact category } i$

For example, if a pollutant concentration in a water body is 1.1 mg/l and the goal is 1.0 mg/l then the weighting associated with this environmental effect would be 0.1 or 10% meaning that the current concentration is 10% away from the target.

The main advantages of the Distance-to-Target approach are its simplicity and ease at which it can be communicated. However, many authors are strongly critical and suggest that it should be used with caution. The most common criticisms focus on the setting of the target position concluding that these are often set according to policy or political factors or even in an arbitrary fashion and not on what can be scientifically achieved or on what is desirable from an environmental quality perspective (Hird, 1994; Powell et al., 1997). Finnveden (1996) also highlights problems associated with spatial
and temporal aspects of setting targets. For example, standards may be different in different countries if based on national policy or on specific national environmental problems or could have different time-frames attached to reaching the target.

There is also the issue that whilst setting targets may be helpful for measuring progress, this approach ignores environmental effects that may be occurring at below target levels (Powell et al., 1997). Lindfors et al. (1995) and Finnveden (1999) highlight the issue that the Distance-to-Target approach assumes that all targets are of equal importance and that this critical assumption has never been scientifically justified. Similarly in order to interpret the result it is assumed that progress from the current situation to the target is linear which may not be the case.

Lee (1999) discusses the issues relating to the fact that the ratio only reflects the level of severity of a specified impact category and does not provide any information as to the relative meaning between the different impact categories. This can lead to, for example, global impacts being considered less important than regional/local ones. In addition, this approach does not consider the effects and relationships between impacts such as synergies and trade-offs.

Some authors suggest that Distance to Target approaches should not be seen as damage indicators but are more a process of normalisation (Finnveden, 1999; Finnveden & Lindfors, 1997).

3.2.2. Damage Function approach

One of the most important and difficult problems of understanding mid-point impacts is the relationship between them and actual environmental damage (Itsubu, 2000), for example, relating emissions of sulphur dioxide (mid-point impact) that may cause acid precipitation to the loss of forestry and biodiversity (end point damage) in a particular region. Another example is the risk of developing a specific disease or health effect following exposure to a particular chemical. The relationship is referred to as the ‘damage function’. To obtain reliable and scientifically credible damage functions, the cause-effect mechanism of the environmental problem must be fully understood. In reality, such relationships are highly complex, dependent upon many variables and often not fully comprehended.

Damage function modelling is an approach becoming more popular to help the interpretation of mid-point impact data (Hertwich & Hammitt, 2001). For example, a toxicity-exposure ratio (TER) which is defined as the ratio of the toxicity effects (expressed as a toxicity threshold such as the LD$_{50}$, LC$_{50}$, NOEC for a particular taxonomic group/species) to the estimated exposure may be used when converting environmental emission data to environmental damage. In this example, mathematical modelling techniques using mid-point data and other parameters describing the local environment (e.g. climate, topography, soil, etc.) would be used to predict environmental concentrations within a particular environmental compartment (e.g. waterbody, field margin, soil etc.) and then the ratio of this concentration to an ecotoxicological threshold is calculated. EU and national thresholds of TER acceptability are then used to interpret the TER values (e.g. Brown et al., 2003; Hart et al., 2003).

Another example of a damage function is a dose-response curve which plots the relationship between dose and the proportion of a population responding with a defined biological effect. This approach is used in a variety of applications. For example in medicine looking at the response a diseased human population have to a specific drug, the control effect a pesticide might have on a damaging insect or the toxicity effect of a chemical on a particular species (e.g. Swenberg et al., 1999; Reffstrup et al., 2010).
According to Hertwich & Hammitt (2001) damage modelling offers two main advantages. Firstly, unlike the Distance-to-Target approach it does consider actual environmental damage rather than just the environmental release and it allows the damage for different impact categories to be compared. However, regarding this last point, this will only be the case if the metrics used and spatial and temporal aspects are equivalent. Another benefit is that consumers and other end users may be more likely to relate to damage (e.g. human health effects) than to mid-term impacts (e.g. quantities of pollutant).

However, the approach does have its limitations. Firstly, the science is not fully developed. Cause-effect chains (Tzilivakis et al., 2009a) are often not fully understood, mathematical models each have their own limitations, for example they may not be fully validated and/or are data hungry, there are also issues relating to uncertainty in the data and the models themselves. The quality of the damage-function will determine the quality of the end-result. As Bare et al. (2000) point out the more complex the approach or the underpinning model the harder it is to maintain transparency as it is not always possible for the user of the information to understand what has been considered within the approach or what assumptions have been made. There still needs to be a sufficient level of consensus within the scientific community that the approach is acceptable.

### 3.2.3. Environmental Valuation

Environmental valuation is based on the assumption that consumers would be willing to pay for environmental benefits and, conversely, are willing to accept compensation for some environmental losses. The technique originates from the Contingent Valuation approach which was first introduced during the early 1940’s as a method for eliciting market valuation of a non-market good and then applied a few years later to attempt to place a value on the prevention of soil erosion (Ciriacy-Wantrup, 1947; Frykblom, 1997; Mitchell & Carson, 1989). The first practical application of the technique was in 1963 when surveys were used to estimate the value hunters and tourists placed on a particular wilderness area.

Within the environmental labelling context, the consumer makes purchasing choices that demonstrate their preferences, which, in turn, place values on environmental resources. Undoubtedly, society values environmental resources but actually placing a monetary value on environmental assets such as coastal areas, air quality of biodiversity is far more complex and very subjective (Gowdy & Mayumi, 2001). Thus, the more significant the monetary value of an impact, the greater significance it will take on. Environmental economists have developed a number of market and non-market-based techniques to value the environment. For example, the Environmental Control Costs (Tellus Institute, 1992) approach places values on impacts based upon the expenditure required to control environmental damage (Powell et al., 1997). For example, if it costs £100 to control one unit of impact A and £50 to control one unit of impact B then impact A would have twice the weighting of impact B. The control level is usually taken as the cost of achieving a regulatory environmental standard and so this approach has some similarity and some of the disadvantages as the Distance-to-Target approach. The rationale here, according to Powell et al. (1997), is that society has expressed its willingness to pay via the national acceptance of the Standard on to the statute book.

The Environmental Damage Costs method is another example of an environmental valuation approach. The weighting values for environmental impacts are derived from the publics’ willingness to pay to avoid the impacts identified in the LCA. The costs of avoidance in this approach would be likely to be different from the costs of control as in the previous example. In this approach no reference to any environmental standard or goal is required (Powell et al., 1997).
There are two main approaches to assigning weightings to impacts: ‘Revealed Preference techniques’ and ‘Stated Preference techniques’. Revealed Preference techniques infer preferences for specific environmental goods through observing the purchasing behaviour of consumers. For example, with respect to the housing market property prices may be determined not only by the physical characteristics of the property but also by the quality of the surrounding area such as the existence of parks or woodland close by. Stated Preference techniques seek to measure individuals’ preferences. This might be done by surveys or questionnaires that ask how much they would be willing to pay to secure the benefit of specific environmental goods or services. Another approach may be to ask participants to rank a number of alternative environmental goods or services in order of preference (EFTEC, 2006).

One of the main criticisms of this approach is its subjectivity and the fact that the economic value may not be in any way related to environmental impact significance. In a review of the technique Venkatachalam (2002) concludes that the technique is subject to severe criticism. The main area of criticism relates to the validity and the reliability of the results and both Smith (1993) and Freeman (1993) question the accuracy, consistency and reproducibility of the results. Randal et al. (1974) conclude that with many applications the valuation process is practically challenging due to the lack of information, pricing and accounting problems inherent in the analysis which makes it difficult to put the method into wide use. For example, trying to place a value on aesthetic improvements would be difficult, since the cost of aesthetic damage is not explicitly reflected in the market (Randal et al., 1974). These problems have never been effectively overcome and some authors are still reporting problems associated with the lack of scientific information. For example, Aizaki et al. (2006) who used the ‘willingness to pay’ technique to evaluate the multifunctionality of agriculture and rural areas, encountered a lack of scientific information and various empirical problems hindering their research. Another area of concern highlighted frequently by the scientific community is that different valuation techniques applied to identical case studies can give very different results. There are several reasons for this. It is partly due to the subjectivity of these types of techniques and also because different monetisation methods cover different types of values (e.g. Finnveden, 1999; Baumann & Rydberg, 1994; Lindfors et al., 1995; Guinée, 1995; Notarnicola et al., 1998).

### 3.2.4. Panel and scoring methods

These techniques essentially use expert judgement to assign weightings or scores to either mid-term impacts or to damage and this is often controversial due to its subjectivity (Powell et al., 1997). Sometimes this approach is based upon the judgements of a panel of experts which are then combined to provide a common decision (Ciroth et al., 2003). Such panels are usually comprised of a range of different stakeholders that will have different viewpoints (e.g. consumers, industry, scientific, environment and/or general public). Ensuring a range of stakeholders helps to reduce issues relating to subjectivity. However, sometimes scores are assigned by one particular stakeholder group (e.g. a particular organisation or action group) and used to define the values of that group.

The scoring or weights assigned are usually based upon a cost-benefit analysis, based on, for example, legislative standards, policy targets, market research or monetary values for remediation or mitigation work (Schmidt & Sullivan, 2002). There are a number of Decision Theory techniques within this general approach that can be used to enable a consensus on weighting to be reached. One of the most popular methods is the Delphi Approach (Thangaratinam & Redman, 2005; Hsu, 2007). This seeks to obtain the most reliable consensus of opinion of a group of experts using an iterative and anonymous questionnaire with controlled feedback. Thus it avoids counterproductive group dynamics where individuals may seek to persuade or intimidate fellow panellists. Thangaratinam & Redman (2005) suggest that the approach is best used when there is little scientific or measured
evidence available and expert opinion is the only way forward. Miller & Cuff (2005) propose other applications such as solving environmental disputes where there is no single ideological solution.

There are several advantages to the Delphi technique. The most significant being its versatility and cost of implementation. The technique also protects participants’ anonymity and so can be popular with stakeholders. However it does have its drawbacks. For example, group dynamics can be advantageous rather than problematic as the development of solutions via group thinking can lead to more pragmatic and insightful solutions. It can also be a lengthy and tedious procedure to undertake leading to high dropout rates of panel members (Hsu, 2007).

A modification of the Delphi Approach is the Policy Delphi (Needham & de Loë, 2008) which attempts to generate the strongest possible opposing views on the potential resolutions to a problem. It is based on the premise that the person (or people) making the ultimate decision in the process does not want the group to generate the decision but rather to identify the full range of options.

3.2.5. Metrics and aggregation

One of the issues relating to interpretation and communication of environmental information in the context of an environmental food label is the need to present the information to the end user (consumer) in a way that they will understand and can respond to by adjusting their purchasing habits. An LCA may identify a large number of different impacts and some could be highly technical in nature and so there may also be a need to aggregate these into a smaller number of impact/damage categories thus creating composite indices. For example, with respect to mid-point impacts there may be several different pollutants that could contribute towards air quality problems (e.g. sulphur dioxide, ozone, particulate matter) or, with respect to environmental damage, there may be several environmental emissions that could damage human health causing, for example, cancer or respiratory problems etc.

Aggregating a number of impacts into a single, coherent indictor is extremely challenging as identified by a number of researchers (e.g. Blanc et al., 2008; Booysen, 2002; Salzman, 2003). However, there are several well known examples of where this has been achieved relatively successfully. For example, the Ecological Footprint (WWF, 2004) which is the amount of natural resources consumed in a given year expressed in global hectares (gha); the Sustainability Index (ESI, 2005; Esty et al., 2005) that aggregates 76 different environmental sustainability measures using a scoring system; and the indictor developed by the Institute of Sustainable Development (ISSI) in Italy (Ronchi et al., 2002) which also aggregates a large number of impact measures uses a scoring approach. Other researchers have grouped large numbers of indicators into methodological frameworks and the DPSIR approach (Berger-Schmitt & Noll, 2000) is probably the most successful but it is not appropriate in the context of an environmental label. These approaches do have severe limitations with respect to, for example, the range of impacts that are included (e.g. the Ecological Footprint does not include impacts for which no regenerative capacity exists such as waste generation, toxicity, eutrophication). In addition, there are concerns over the loss of transparency and the subjectivity of the scoring approaches (see also Section 3.2.4 above). There are, however, techniques that are used to aggregated impacts and indicators into less broad damage categories which may have more resonance with consumers. For example there are many approaches where the three broad damage categories of human health, ecosystem quality and resource depletion are used (e.g. Goedkoop & Spriensma, 2000; Steen & Ryding, 1991; Steen, 1999; Lindfors et al., 1995).

In all situations where aggregation is required impacts must be placed on a common scale and/or need to have equivalent spatial and temporal reference points. There are a number of techniques that may be used to achieve this.
Health Related Impacts: Disability Adjusted Life Years (DALYs)

The impact of emissions, other environmental pressures and resource competition on human health is an important area of concern for individuals and is likely to be an important aspect of a product label. Developed by the World Bank, the DALY measures the overall burden of disease and is a time-based measure that combines years of life lost due to premature mortality and years of life lost due to time lived in states of less than full health. In so doing, mortality and morbidity are combined into a single, common metric. One DALY can be thought of as one lost year of "healthy" life and the sum of DALYs across the population can be thought of as a measurement of the gap between current health status and an ideal health situation where the entire population lives to an advanced age, free of disease and disability (WHO, 2010; Murray & Lopez, 1996; Jamison et al., 1993).

DALYs are calculated as the sum of the years of life lost due to premature mortality (YLL) in the population and the equivalent ‘healthy’ years lost due to disability or ill-health (YLD) for incident cases of the health condition:

\[
\text{DALY} = \text{YLL} + \text{YLD}
\]

Where YLL = number of deaths x standard life expectancy at age of death and YLD = incidence x duration x severity weighting. More details are given by Murray and Lopez (1996).

The duration of time lost due to premature mortality is calculated using standard expected years of life lost with model life-tables available in the literature. The reduction in physical capacity due to morbidity is measured using disability weights as given by Murray and Lopez (1996). The value of time lived at different ages is usually calculated using an exponential function which reflects the dependence of the young and the elderly on adults within a defined population (Murray and Lopez, 1996).

The DALY is used in some LCA techniques to convert mid-point impacts into a measure of damage to human health (i.e. an outcome indicator) and it is gaining in popularity. However it is not without its critics. Anand and Hanson (1997) argue that the conceptual and technical basis (e.g. quality of the data) for DALYs is flawed, and its assumptions and value judgements are often open to serious question. Arneson and Nord (1999) also highlight the fact that the DALY approach presupposes that life years of disabled people are worth less than life years of people without disabilities and implies that disabled people are less entitled to scarce health resources for interventions that would extend their lives. Fox-Rushby and Hanson (2001) provide several examples where incorrect life-expectancy values have been used, where DALYs have been calculated using different assumptions and state that there are very few examples that demonstrate the precise method of calculation leaving the technique prone to misuse.

Health Related Impacts: Toxicity Equivalent Potentials (TEP) and Human Toxicity Potential (HTP)

The Toxicity Equivalence Potential (TEP) risk scoring system was developed due to the need for comparing releases of toxic chemicals into different environmental media. It considers potential impacts from chemicals into different environmental compartments using a simplified representation of actual processes (McKone & Hertwich, 2001). It is an order of magnitude indicator where the toxicity of a particular chemical is compared with the toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) which is considered to be the most toxic of all polyhalogenated aromatic hydrocarbons. The
technique is used by the US SCORECARD service that was launched in 1998 as a free public-information service (Scorecard, Undated). Neumann (1999) highlights some limitations of the approach, particularly that TEPs were established to identify difference of magnitudes in toxicity between different chemicals and should not be used to identify precise differences in toxicity or to suggest greater process sophistication. Whilst this is useful in many applications in the context of an environmental product label it may not be useful when differences between different products may be relatively small. Neumann (1999) also highlights the fact that the approach ignores some important biological processes such as bioavailability and pharmacokinetics which are critical factors in assessing toxicity. Consequently the scientific credibility of the technique can be questioned.

The Human Toxicity Potential (HTP) is related to the TEP and reflects the potential damage a unit quantity of a chemical may cause when released into the environment (Guineé & Heijungs, 1993). It is a characterisation technique used to weight mid-point impacts (emissions), such as those generated for life-cycle and toxic release inventories (Rosenbaum et al., 2008; McKone & Hertwich, 2001). It accounts for both toxicity and the potential to result in exposure, however, it does not consider regional differences or variations in population density. Exposure data is estimated by dividing estimated cumulative dose by a toxicity benchmark which may be a reference concentration (RFC) or the Allowable Daily Intake (ADI) or No-Observable-Effect-Levels (NOELs). As such it often uses toxicity end-point parameters which themselves are subject to data uncertainty and variability errors (McKone & Hertwich, 2001; Bare & Gloria, 2006).

**Evaluating Ecological Risks - Toxicity Exposure Ratios, Potentially Disappeared Fraction (PDFs) and Potentially Affected Fraction (PAFs)**

The need to compare ecological risks and to develop criteria for assessing quality of ecological resources are common problems in environmental management and environmental policy development. A frequently used method of doing this is by using Toxicity: Exposure Ratios (TER). The toxicity measure is usually taken as a standard toxicity endpoint such as the LD50, LC50 or NOEC. These endpoints indicate the response of a population exposed to a toxic chemical (see also Section 3.2.2). In many instances exposure is estimated or predicted for a particular environmental compartment using a suitable mathematical model (FOOTPRINT, p-EMA etc.). TER’s can be interpreted using regulatory and internationally accepted thresholds and guidelines (e.g. EPPO/CoE, 1994a,b; Hart et al., 2003; Brown et al., 2003; De Lang et al., 2009).

\[
\text{TER} = \frac{\text{Toxicity (e.g. LD50, LC50, NOEC)}}{\text{PEC}}
\]

Where the PEC is the predicted environmental concentration.

TERs are extensively used for risk assessments and within various regulatory approval processes for authorising the use of specific chemicals such as drugs and pesticides (e.g. the approach set down in European Directive 91/414/EEC and its Annexes) based on their acceptability with respect to risks to health and the environment.

Some LCA techniques express potential damage to eco-system quality as the percentage of species that have disappeared within a certain area, due to the environmental pollution load and this is known as the ‘Potentially Disappeared Fraction’ (PDF) (ott et al., 2006; Life Cycle Strategies, 1998). It
is an indicator which can be considered to be the fraction of a particular species that has a high probability of no occurrence in a region, due to unfavourable conditions. All species are considered to be targets and it is usually used to address ecological damage arising from, for example, issues such as land use, acidification and eutrophication.

\[
\text{PDF} = \frac{\text{species diversity}_{\text{reference}} - \text{species diversity}_{\text{use}}}{\text{species diversity}_{\text{reference}}}
\]

The PDF’s are measured in \( m^2 \) per year per kg of each emission in various environmental compartments (soil, water, field margin etc.). The values are then totalled and multiplied by the area size and the time period to obtain the damage (Hamers et al., 1996).

The Potentially Affected Fraction (PAF) (van der Meent & Klepper, 1997; Klepper et al., 1998; Traas et al., 1998) is an indicator which corresponds to the fraction of a species exposed to a concentration equal to or higher than the No-Observed-Effect-Concentration (NOEC) for the species in question. It is a measure of toxic stress and not specifically environmental damage. It is calculated from a Species Sensitivity Distribution (SSD) curve (see Figure 4) and defined as the probability or percentage (y) of species that is affected at a given concentration (x) of a chemical. Species sensitivity distributions are used in ecological risk assessment to derive maximum acceptable concentrations of toxicants in the environment from a limited set of laboratory obtained ecotoxicity data (Posthuma et al., 2001; Kooijman, 1987).

![Species Sensitivity Distribution (SSD)](image)

**Figure 4: Example of a SSD Curve and determining PAFs**

As would be expected there are pros and cons to all of these techniques. Both Klepper et al. (1998) argue that the use of PAF’s is a more meaningful method of comparing ecological risks than TERs
because the TER is difficult to interpret even with guidance. For example a TER < 1 implies negligible damage or risk but if two substances have similar TERs their environmental impacts may be quite different. However, as both PAFs and PDFs are indicators the ecological implications of any given value is not obvious. For example, with respect to PAFs, should a NOEC be exceeded in the field we would expect to see some toxicological impacts such as a reduction in reproduction or decreased growth but this may not result in serious long term damage Klepper et al. (1998). Vighi et al. (2006) point out that the standard toxicological endpoint data is usually measured under laboratory conditions and consequently the reproducibility and comparability is high in this situation. However, due to the variability of natural systems compared to standard laboratory conditions, using this data to predict effects on natural populations is highly questionable.

An issue affecting many of the approaches to assessing ecological quality is the need for sound ecotoxicity data. However, there are very few toxicity studies available for fauna and flora beyond the usual indicator species (e.g. rats, birds, fish, algae etc.). For other species, such as butterflies, moths, dragonflies and reptiles etc., little if any data is available (Traas et al., 1998; US EPA website). Yet different species can react very differently to toxins (Schafer & Brunton, 1979; De Lang et al., 2010). If the range of species is limited or restricted when determining these metrics then caution in interpreting the results is required.

Natural Resources: Measuring availability, scarcity and depletion

The global supply of food, water, energy, land and materials is limited. The availability of natural resources relative to population can considered to be a fundamental indicator of social welfare and has crucial implications for economic growth (MacKellar & Vining, 1987). However, whilst it is relatively easy to measure the amount of waste or CO₂ produced by a company or during the manufacture of a product, measuring resource use and estimating environmental damage is far from simple. Environmental resources can be classified into two main types: those that are living (biotic) and those that are not (abiotic). The latter type includes water and land use.

Biotic resources can be considered from two key perspectives: their availability for future generations and their contributions to ecosystem functioning and biodiversity. Biotic resources are renewable but over exploitation can lead to damages such as population decrease and eventually species extinction. The approaches taken to the measurement of the scarcity or availability of natural resources depends upon the perspective.

Measuring their availability for future generations has been discussed by Guinée (2002) where it is proposed that damage can be described as the annual extraction rate minus the annual replenishment rate divided by the square of the current stock.

A number of different techniques are available for impacts on the ecosystem functioning and biodiversity. Both Müller-Wenk (2002a) and Steen (1999) propose using the difference between annual (global) extraction levels and the level at which there is no longer a threat of extinction is proposed as an indicator basis, using IUCN (2001) red-listed species data. Sas et al. (1997) expressed the risk of extinction in terms of reproduction time and the current amount of biomass.

Abiotic resources are often expressed in terms of their availability or scarcity. In terms of demand competition, the SETAC-Europe working group (Lindeijer et al., 2002) considered this to be covered through pricing. In terms of depletion, available impact methods are often based on the amounts of the deposits (their abundance), on the extraction rates (Wenzel et al., 1997; Guinée, 2002), on modelling (interventions due to) future ore extractions (Goedkoop & Spiersma, 2000; Müller-Wenk, 1998; Steen, 1999), or on energy consumption (Finnveden & Östlund, 1997).
Evaluation of resource deposits that are being irreversibly depleted (such as fossil oil, gas and coal) and so unavailable to future generations is very uncertain. Modelling and other forms of prediction of future extraction rates are similarly uncertain and scientific credibility of such assessments is questionable. Substitutability and accumulation in the economy are not currently taken into account by most techniques that seek to evaluation resource depletion (Lindeijer et al., 2002). However, Steen (1999) modelled abiotic resource indicators in terms of the impacts for acquisition of a “near-sustainable alternative”. For fossil oil, gas and coal, these alternatives are rapeseed oil, biogas, and charcoal, respectively.

3.3. Other Issues

3.3.1. Normalisation

Normalisation is an optional step in LCA that is used to better understand the relative importance and magnitude of the mid-point impacts. It is used to interpret the relative size of an environmental effect by comparing it with reference data such as a particular area or for a specific time period. There are several aims of the normalisation process. Firstly it provides a sound basis for comparison between impact categories. It also facilitates the interpretation of the characterisation results by expressing the impact as a proportion of the total environmental problem within a referenced time period. As a consequence it aids the judgement of the importance between the various environmental issues (Heijungs et al., 2006; van Oers & Huppes, 2001).

Many authors have highlighted the important of spatial factors (Bare et al., 1999; Hauschild & Wenzel, 2000; Reap et al., 2008) when evaluating the environmental impact of a product. Emissions occurring throughout a product lifetime can occur at a wide variety of different locations. For example, raw materials may be sourced from abroad, manufacturer may occur in one place but the product exported and then consumed somewhere entirely different. Some emissions can be considered in the global context such as global warming but for those that affect the local and region environment spatial referencing will be required in order to appropriately evaluate the impact. It may also be necessary to consider the environmental uniqueness of the area affected. A local environment will be uniquely sensitive to the stressors placed on it. For example, parameters such as local weather, geology, topography and hydrology will affect the intensity of a particular impact. Research into techniques that will allow these factors to be included into an LCA study are ongoing (Reap et al., 2008) and yet to be standardised and accepted by the scientific community.

Temporal issues are another problem that can affect LCA studies. Appropriate time horizons may differ across impact categories. Effects on climate change for example may take decades, it may take years for pollutants to reach some groundwaters but impacts on biodiversity may occur immediately. There is also the issue of recovery and reversibility and the related financial costs of these to be considered.

3.3.2. Uncertainty

An issue recognised by many authors (e.g. Finnvenden et al., 2009; Pennington et al., 2004; Reap et al., 2008) is the effect uncertainties in data and methods can have on the analysis of environmental impacts especially when using an LCA approach and using normalisation and weighting procedures. Uncertainty can arise in a number of ways including:
• Discrepancies between measured / calculated data and the actual value. For example, limitations or errors in the measurement, analytical or calculation technique;
• Natural variations in the impact. For example, some environmental impacts may be variable according to temperature, moisture levels and site specific details such as soil types. Other examples include variations in performance and emissions across individual processes and even items (e.g. two cars of the same age, make and model may have different emissions);
• Limitations to or failures in the assumptions made. For example a linear relationship between impact and damage may be assumed but this may not reflect the true relationship. Models may be over simplified or used inappropriately;
• Data, processes or models may not accurately represent the full spatial and temporal scope of the assessment;
• Limitations and simplifications inherent within the interpretation process.
• Loss of accuracy from aggregation and data transformation on to common scales;
• Inconsistencies in the way data is handled. For example round-off errors can be huge if subjected to scaling and normalization processes e.g. if 0.111 is rounded to 0.1 we have an error of 11%;
• Missing and incomplete data, proxy data, estimates and ‘expert’ judgments';
• Erroneous data being used. This may be due to typos, errors in reported units, misinterpretation etc.;

LCA and other forms of environmental impact assessment are very data intensive and many of the data sources will have elements of uncertainty and so it is important that an analysis of these and their sensitivity regarding the study’s conclusions is undertaken in parallel to the main study (Finnvenden et al., 2009). The more complex the assessment process is (for example, if it includes mathematical modelling) the greater the amount of data required and the more uncertainty will be imbedded in the process. Consequently, uncertainty can escalate along the LCA path. There are trade-offs between complexity and uncertainty that needs to be understood (Pennington et al., 2004) but these often remain unclear and unquantified as there is a distinct lack of methods to identify and evaluate them. Reap et al. (2008) review several techniques that have been proposed for assessing uncertainty but concludes that none are clearly appropriate for all causes of uncertainty and that this is an unresolved problem for LCA studies. Some authors suggest that the uncertainties are so high for the assessment of damage, normalisation and interpretation that it may often be appropriate to stop at the mid-point assessment stage (Bare et al., 2000).

3.3.3. Transparency

In order to be credible and to identify the potential for improvements, any environmental assessment method should be transparent. Stakeholders should be able to understand the principles of the calculation of the final results, if not the fine detail, and should be able to identify the most relevant environmental issues from the results. The more complex the assessment process the harder it becomes to maintain transparency. Some LCA practitioners have suggested that the more transparent the data and the methods used the lower the quality of the assessment (Frischknecht, 2004). Frischknecht (2004) highlights the requirements of the ISO standards in LCA (Guinée, 2002). The results of two LCAs on the same subject may differ according to the objectives, processes, quality of the data, and the impact assessment methods used. Therefore it is important that assumptions, processes and data are transparent to enable study duplication.

The lack of transparency in a study can arise from three main issues:
1) Complexities in the modeling systems: without hefty documentation, scientific knowledge and a detailed understanding of how a model works it is not always easy to understand what is included in the study, to what extent and the credibility of the model;
2) Issues relating to the confidentiality of information and data that may circumvent the ISO requirements and lead to studies not being reproducible. Consequently there may be doubts regarding the credibility of the findings;
3) A lack of information on parameter sensitivity and other uncertainties can also reduce the transparency of a study.

3.4. Example Techniques and Models

A number of ‘named’ techniques and models have been developed that combine approaches discussed in Section 3.2. A number of the more common ones are discussed below together with their strengths and weaknesses

3.4.1. The Ecoscarcity method

The EcoScarcity technique (Brand et al. 1998; Frischknecht et al., 2006), which is also known as the Environmental Scarcity, Ecological Scarcity and Swiss EcoPoints method (or model), was first published in Switzerland (Ahbe et al., 1990) and applied to the Swiss national situation. It is based upon the derivation of a comparative weighting and aggregation of various environmental impact categories using ‘eco-factors’. The model supplies these weighting factors for different emissions into air, water and top-soil/groundwater as well as for the use of energy resources. Eventually the weighting factors are combined to determine the overall impact value.

The eco-factors are based on the annual actual flows (current flows – i.e. the current situation) and on the annual flow considered as critical (critical flows – i.e. the target) in a defined area (country or region), as such the technique does not conduct a standard impact analysis as defined by ISO14042 but uses a Distance-to-Target approach (see Section 3.2). Generally, data describing current flows are taken from statistical data and critical flow data are based upon national (or regional) environmental policy. The technique has also been applied to projects in other countries such as Sweden (Baumann et al., 1993), Belgium and Japan (Siegenthaler et al., 2002) and sets of eco-factors developed for these countries.

The approach has been developed based upon the assumption that an established environmental policy framework, including international treaties and agreements, may be used as a reference for improving individual products and processes (Bare et al., 2006).

3.4.2. Dutch and Swedish Environmental Theme (ET) method

This method was developed in Netherlands and described by Heijungs (1992). Whilst not strictly an endpoint valuation technique as it uses mid-point data directly, the method is similar to the Ecocscarcity method and uses the Distance-to-Target approach, based on weighting and reduction factors from Dutch environmental policy. The actual facts are not directly evaluated, instead the facts are aligned in two stages with an estimation of the effect, and the resulting data is then weighted. Effectively, data on emissions and use of resources are calculated as contributions to environmental problems. These environmental problems are then weighted against each other according to political goals (McKinsey & Company, 1991; Baumann & Rydberg, 1994).
3.4.3. Environmental Priority Strategy (EPS) method

The Environmental Priority Strategies is a valuation model using a Distance-to-Target technique that was developed in Sweden (Steen & Ryding, 1991; Steen, 1999; Lindfors et al., 1995) to meet the requirements of an everyday product development process, where the environmental concern is just one among several others (e.g. durability, costs etc.). In this method, the damage towards protected targets within a specific area (human health, eco-system productivity, biotic and abiotic resources; aesthetic and cultural values) is evaluated based upon the costs of restoring them to full health. Costs are calculated for the area based upon actual restoration expenditure (current situation position) or on a contingent valuation i.e. a willingness to pay. The target position is usually taken as no expenditure on restoration is required. The EPS method does seek to quantify damage using a scientifically sound methodology such as basing disease incidence on dose-response functions and so this method could also be classified as a damage-function approach (see Section 3.2).

Finnveden (1999) highlights a number of failing of the model. One is that environmental impacts occurring after a specific time period are not considered – so any climate change damage occurring after, say, 100 years would be ignored. Other criticisms relate to the lack of transparency, logic and computational errors and data gaps. Some of Finnveden's (1999) criticisms have been addressed in the more recent version of the EPS method. EPS2000 (Steen, 2002) has, for example, fewer computational errors. However, the basic approach is the same for both the 1996 and the 2000 versions (Steen, 1999; Bare et al., 2006).

3.4.4. Eco-indicator99

The Eco-Indicator99 methodology (Goedkoop & Spriensma, 2000) is one of the more common techniques in use. The technique was developed by an international group of LCA and environmental experts as a 'damage function oriented' impact assessment method (Bare et al., 2006). The aim was to develop a technique to express the total environmental burden of a product as a single score (Goedkoop et al., 1998) by combining different impact categories into three damage endpoint areas (Human health, Ecosystem quality and Natural Resource depletion). This technique is based upon cause-effect relationships and follows a stepwise approach (Figure 5) which seeks to link environmental emissions, resource and land use to potential environmental damage.

Using mathematical modelling and other natural science prediction techniques the environmental concentrations (including concentrations in food and drinking water) of the pollutants are estimated. This data is then used to quantify the actual end environmental effects such as contribution to
climate change and ozone depletion and uses (where possible) standard literature data. End effects are aggregated into three damage categories using common damage category units (see Section 3.2).

The use of Disability Adjusted Life Years (DALYs) (see Section 3.2) are used for communicating human health impacts whilst Potentially Affected Fractions (PAF, see Section 3.2) are used for Ecosystem quality. With respect to resource depletion Eco-Indicator99 only considers mineral resources and fossil fuels and these are measured as ‘surplus energy’ in MJ per kg extracted material.

The interpretation stage (step 4) involves weighting the three damage categories (using one of three default weighting sets), aggregating them and converting into a single score or Eco-Indicator value. The absolute value of the score is irrelevant as it is mainly used to compare products or processes. However, the developers do suggest that a score of 1 is approximately equivalent to one thousandths of the yearly environmental load of one average European inhabitant.

The Eco-indicator99 developers have recently been involved in a related method called ReCiPe in collaboration with Leiden University – the developers of the CML model (see Section 3.3.4). ReCiPe is combines a mid-point approach as used by the CML model with the damage approach used by Eco-Indicator99.

3.4.5. UBA Method

This is a technique that was piloted on a German packaging system in 1996 (Schmitz et al., 1996; Plinke et al., 2000). It uses a modification of a panel and scoring system approach to provide an ordinal ranking system for impacts. Mid-point impacts and damage impact categories are assessed separately and the results of each are then ranked according to their overall importance (Ciroth et al., 2003). It diverges from traditional panel and scoring methods as a semi-quantitative technique is used to merge mid-point impacts and damage. Scores on a 5 point scale from low to high are assigned by experts to each impact category based on five criteria – hazard potential, temporal aspects, spatial coverage of the effect, public preferences and relationships to previous and existing environmental damage (Ciroth et al., 2003).

One serious criticism of the approach is that these five criteria are neither independent nor measureable (Ciroth et al., 2003). Scores are partly assigned based on published scientific findings and, in the case of public preferences, established using social science techniques however there is still a strong element of subjectivity.

3.4.6. The USETox Model

The USEtox model is an environmental model for the characterisation of human and ecotoxicological impacts in Life Cycle Impact Assessment and for comparative assessment and ranking of chemicals according to their inherent hazard characteristics. It has been developed by researchers working under the UNEP-SETAC Life Cycle Initiative (Jolliet et al., 2006).

The drivers behind the initiative were problems associated with the need to include within the LCA process the impacts associated with the full range of chemicals released into the environment. For many manufacturing processes this could amount to thousands of different substances. However, environmental impact and toxicity data is often not available and, where it is, is often of doubtful quality. Therefore many existing models and techniques for damage characterisation only cover a very limited range of substances. The UNEP-SETAC Life Cycle Initiative aimed to compare damage and characterisation models and techniques to identify how and why results are different and to
reach scientific consensus on good practice. It also sought to develop a scientific consensus model based upon the findings of the comparison exercise. The outcome of this was the USETox model.

Within USETox damage is characterised for human toxicity and for freshwater ecotoxicity. Toxicological effects resulting from the release of a chemical into the environment are assessed via evaluation of the cause-effect chain. This evaluation comprises of three steps: environmental fate predictions, exposure evaluation and identification of environmental effects.

### 3.4.6. CML method

This model was developed by Leiden University Institute of Environmental Sciences (CML) and is presented as the Dutch Guide to LCA (Heijungs et al., 2003). It provides a comprehensive step-by-step approach for conducting an LCA study based upon the ISO guidelines (Guinée, 2001). It is predominately a mid-point method that uses normalisation to contextualise the inventory data. Impact categories have been selected by the developers based depending upon their being practical and acceptable characterisation methods according to published review information (Bare et al., 2006).
4.0. Discussion and Conclusions

This study has shown that despite extensive research, development and application, and its importance to both government and industry, both the mid-term impact assessment stages and the final damage assessment and/or normalisation processes undertaken in Life Cycle Assessment have a great number of both practical and theoretical problems still to be resolved if they are to be successfully used in the context of an environmental label.

- With the exception of greenhouse gas emissions (BSI, 2008) there is a severe lack of scientifically credible techniques that have been tested, standardised and accepted by the scientific community. The findings of this study in this respect have been summarised in Table 10.
- Methods that are in common use for damage assessment, characterisation and normalisation are prone to subjectivity, uncertainty and unjustified assumptions. The findings of this study in this respect have been summarised in Table 11. The work undertaken has concluded that the use of LCA techniques within the context of an environmental label should be limited to the mid-point phase as, although desirable, damage characterisation is currently undeveloped.
- Communication of multiple environmental impacts that are scientifically complex on an environmental label would be very difficult and require aggregation and considerable simplification. Aggregation techniques themselves are highly complex and whilst they have been used in academic applications they are insufficiently tested and validated and have not been widely accepted by the scientific community to enable them to be used with confidence on an environmental label. Composite impacts are not transparent and there is the risk of losing data resolution.
- There is a lack of appropriate data available. In many instances the data necessary to measure and/or calculate mid-point impacts and/or damage cannot be measured easily, reliably or cost effectively. It is also subject to large uncertainties.
- There are issues of spatial and temporal resolutions to be resolved. Data sets are not harmonised in terms of their spatial or temporal characteristics. Environmental issues significant in one location may not be so important in a different location.

Table 10: Summary of techniques and methods used for assessing environmental impact in the context of an environmental label

<table>
<thead>
<tr>
<th></th>
<th>Standard Techniques Available?</th>
<th>Sound measurement techniques available?</th>
<th>Data available or measureable?</th>
<th>Aggregation required? Possible?</th>
<th>Value to an eco-label</th>
</tr>
</thead>
<tbody>
<tr>
<td>Climate Change</td>
<td>Yes – PAS 2050</td>
<td>Yes</td>
<td>Data available for common processes, gaps exist for agricultural processes.</td>
<td>Needed to combine various GHG’s – std technique exists and is robust.</td>
<td>Valuable</td>
</tr>
<tr>
<td>Air Quality</td>
<td>No</td>
<td>No single tool. Separate measures available for some issues e.g. ozone, VOC’s &amp; photo-oxidant</td>
<td>Severely limited</td>
<td>Need to combine various pollutants/effec ts – techniques do not appear sound</td>
<td>Limited to certain impacts – needs further scientific development and debate</td>
</tr>
<tr>
<td></td>
<td>Standard Techniques Available?</td>
<td>Sound measurement techniques available?</td>
<td>Data available or measureable?</td>
<td>Aggregation required? Possible?</td>
<td>Value to an eco-label</td>
</tr>
<tr>
<td>--------------------------</td>
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</tr>
<tr>
<td><strong>Soil Quality</strong></td>
<td>No</td>
<td>No single tool. Indicators and indices main techniques used.</td>
<td>Some data sets available but their value in this context is limited.</td>
<td>Need to combine various pollutants/effects – techniques require further development.</td>
<td>Limited to certain impacts – needs further scientific development and debate</td>
</tr>
<tr>
<td><strong>Water Quality</strong></td>
<td>No</td>
<td>Some but all have problems in the context of labelling</td>
<td>Good data sets available</td>
<td>Need to combine various pollutants/effects – techniques require further development.</td>
<td>Limited to certain impacts – needs further scientific development and debate</td>
</tr>
<tr>
<td><strong>Biodiversity</strong></td>
<td>No</td>
<td>No single tool, highly complex. Multiple problems. Limited to the use of indicators and indices.</td>
<td>Limited, expensive to collect, influenced by multiple, interacting parameters.</td>
<td>Need to combine various pollutants/effects – techniques complex and do not appear sound</td>
<td>Needs significant development.</td>
</tr>
<tr>
<td><strong>Stratospheric Ozone Depletion</strong></td>
<td>No</td>
<td>Yes - Ozone Depletion Potential</td>
<td>Data available for common processes, some gaps exist for agricultural processes.</td>
<td>Need to combine various pollutants – technique exists and is robust.</td>
<td>Reasonable potential</td>
</tr>
<tr>
<td><strong>Resource Depletion</strong></td>
<td>No</td>
<td>Many techniques proposed – all have problems.</td>
<td>Severely limited and uncertain.</td>
<td>Aggregation required but techniques have been severely criticised.</td>
<td>Needs significant development.</td>
</tr>
<tr>
<td><strong>Waste and Recycling</strong></td>
<td>No</td>
<td>Many simple but effective methods in use by industry.</td>
<td>Measureable.</td>
<td>Need to combine various wastes – technique exists and is robust.</td>
<td>Reasonable potential</td>
</tr>
<tr>
<td><strong>Landscape &amp; Heritage</strong></td>
<td>No</td>
<td>Some have been proposed in literature but not sound in labelling context</td>
<td>Not measureable. Available data severely limited and its value questionable</td>
<td>Need to combine various effects – techniques do not appear sound</td>
<td>Needs significant development.</td>
</tr>
<tr>
<td><strong>Noise, Dusts &amp; Odours</strong></td>
<td>No</td>
<td>Few scientifically accepted techniques available. Very subjective.</td>
<td>Measureable but its value limited in this context. Not routinely collected on farm. Could be expensive.</td>
<td>Three issues need to be considered separately. Aggregation not required.</td>
<td>Limited potential and value.</td>
</tr>
</tbody>
</table>
Table 11: Summary of techniques and methods used for assessing environmental impact in the context of an environmental label

<table>
<thead>
<tr>
<th>Pros</th>
<th>Cons</th>
<th>Value to an eco-label</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Distance to Target Method</strong></td>
<td>• Criticisms concentrate on the difficulties associated with setting targets.</td>
<td>Could be useful providing suitable targets can be identified and agreed.</td>
</tr>
<tr>
<td>• Widely used &amp; accepted.</td>
<td>• Targets can be seen as politically biased, arbitrary or subjective.</td>
<td></td>
</tr>
<tr>
<td>• Simple to use.</td>
<td>• Does not consider environmental damage – based on mid-point impacts.</td>
<td></td>
</tr>
<tr>
<td>• Transparent.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>• Easy to understand.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>• Useful for measuring progress.</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Damage Function Approach</strong></td>
<td>• Science not well developed.</td>
<td>May be useful where damage can be soundly measured.</td>
</tr>
<tr>
<td>• Based on actual environmental damage.</td>
<td>• Few damage functions scientifically identified.</td>
<td></td>
</tr>
<tr>
<td>• Consumers may relate to damage better than arbitrary targets.</td>
<td>• Relationship between cause and effect is highly complex.</td>
<td></td>
</tr>
<tr>
<td><strong>Environmental Valuation</strong></td>
<td>• Not easily understood.</td>
<td></td>
</tr>
<tr>
<td>• Cost/ societal value has always been considered to be a sound driver for change.</td>
<td>• Not transparent to the lay-person.</td>
<td></td>
</tr>
<tr>
<td><strong>Panel and Scoring Methods</strong></td>
<td>• Highly subjective.</td>
<td>Limited due to practical problems and its subjectivity.</td>
</tr>
<tr>
<td>• Expensive.</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

In conclusion, in the context of environmental labelling our opinion is that a considerable amount of scientific development and debate towards achieving standardised techniques for measuring and assessing environmental impacts is required before a true omni-label for food could be a reality, particularly if it was to be used for a statutory application. However, that is not to say that it is not possible to begin with some simple measures for a voluntary application particularly if the scheme and its associated label were designed to be flexible and expandable.
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