Environmental Benefits of Livestock Manure Management Practices and Technology by Life Cycle Assessment

D.L. Sandars; E. Audsley; C. Cañete; T.R. Cumby; I.M. Scotford; A.G. Williams

Process Engineering Division, Silsoe Research Institute, Wrest Park, Silsoe, Bedford, MK45 4HS, UK; e-mail of corresponding author: daniel.sandars@bbsrc.ac.uk

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An environmental Life Cycle Assessment (LCA) procedure is constructed to compare the total emissions from different techniques for managing livestock wastes. Life Cycle Assessment is a method of holistically and systematically accounting for the environmental benefits and burdens of the production of goods and services including consequential burdens generated elsewhere. As waste emissions are very variable, the methodology is extended to include the uncertainty in the estimates in order to indicate the significance of differences between techniques. The object is to inform policy of whether options are better for the environment by quantifying potential emissions abatement, by highlighting priority environmental impacts and by revealing compromises for further investigation.

This paper reports comparative LCAs for several pig waste management options. For example, various slurry application techniques, including: splash plates, band spreaders and injection. If the splash-plate system is taken as a reference, the injector system causes only 64% of the environmental acidification and 71% of the eutrophication of surface waters. The benefits must be offset against the increase in nitrate leaching of 50%. In contrast, the band spreader system offers 28% of the benefits of injection.

The environmental impacts have also been expressed as a proportion of the UK national emissions. This gives each impact a weighted-value that enables direct comparisons of disparate impacts. Although band spreader systems showed an aggregated, or total, environmental impact reduction of almost 10%, the reduction is not significant when uncertainty is taken into account. Using an anaerobic digester shows few overall benefits due to the fugitive losses of methane. However, if these can be eliminated the global warming potential from waste management is reduced close to zero.

1. Introduction and background

In agriculture, environmental impacts are mainly due to the emission of chemicals, methane (CH$_4$), nitrous oxide (N$_2$O), ammonia (NH$_3$), and nitrate (NO$_3$) to air and water. Globally, there are between 300 and 450 Mt a$^{-1}$ of anthropogenic methane emissions, of which the UK emits between 3 and 4.5 Mt a$^{-1}$ with 0.846 Mt a$^{-1}$ due to farmed livestock alone (DETR, 1998; Sneath et al., 1997a). The figures for nitrous oxide emissions to air in the UK are 189.2 kt a$^{-1}$ nationally, of which 18.3 kt a$^{-1}$ is from farmed livestock (DETR, 1998; Sneath et al., 1997b). Ammonia emissions from UK farmed livestock are ca. 224 kt [NH$_3$-N] a$^{-1}$ (Misselbrook et al., 2000) and this is one of the UK’s main sources. Nitrate emissions to ground and surface water in the UK are also largely attributed to agriculture (DETR, 1998). Pretty et al. (2000) have made an assessment of the total external costs of UK agriculture and estimate that nitrate leaching alone costs the UK economy £16 m annually in treatment costs.

Concerns over emissions to the environment, of nitrate, ammonia and methane from pig production systems has led to a plethora of techniques and scientific studies of ways to reduce emissions from piggeries, from slurry and solid manure stores and from land spreading. For example Phillips et al. (1999) assess methods of abating ammonia emissions from livestock buildings and manure stores. However, savings in one area can lead to increased emissions in others. It is thus
important to be able to evaluate consistently a large number of alternative techniques in terms of aggregated environmental impacts of production for farmed livestock such as, beef and dairy cattle, pigs and poultry.

The Integrated Pollution Prevention and Control (IPPC) directive (The Council of the European Union, 1996) applies to a vast range of industrial enterprises, including new pig and poultry units over the threshold size, which is 2000 fattening places in the case of pigs. These enterprises are required to use Best Available Techniques (BAT) not entailing excessive cost to prevent and otherwise reduce emissions and the impact on the environment as a whole (Environment Agency, 2001). There is thus a need for policy makers to be able to make comparative assessments of the net environmental benefits of proposed techniques.

2. Theoretical considerations

A sound scientific basis for comparing alternative farm systems that create animal waste on the basis of overall environmental burdens needs to systematically analyse all the inputs and outputs involved in the complete process of waste production, storage, processing and utilisation. Some techniques, such as slurry injection, use considerable amounts of energy and may also increase emissions from other stages in the nutrient cycles, such as nitrate leaching, thus making it difficult to identify the best approaches. Considerable uncertainty is often associated with the performance of many techniques that are used under variable field conditions, due to lack of data, lack of knowledge, measurement difficulties, climatic variability, etc. It is quite common to see the environmental benefits of new technology reported as ‘up to’ a given level of reduction in emissions. The methodology used to compare different farm systems that create waste must apply to different methods of waste treatment and be easy to adjust in response to new information or factors concerning existing systems and methods.

2.1. Life cycle assessment

Life Cycle Assessment methods systematically follow mass and energy balances from ‘cradle to grave’ to
ensure that improvements at one stage correspond to an overall improvement and do not simply move problems up or down the chain (SETAC, 1993). They are typically conducted using principles and guidelines laid out in ISO 14040-14043 (ISO 14040, 1997; ISO 14041, 1998; ISO 14042, 2000; ISO 14043, 2000). These break the analysis into sections: goal definition, system boundary, inventory and impact assessment. Life Cycle Assessment studies stress the importance of expressing impacts per unit of function and considering the whole production chain.

To illustrate the approach, this study was focussed on pig production systems, the largest of which will soon come under the Integrated Pollution Prevention and Control (IPPC) directive (The Council of the European Union, 1996). The following sections address the objective and scope of the LCA, the calculation of the Life Cycle Inventory (LCI) of emissions and finally the calculation of environmental impacts due to the emission.

2.2. Implementation issues

To cope with the heterogeneous nature of the effluent stream, the analysis of uncertainty and the need to systematically and rapidly evaluate numerous techniques we developed our own software to conduct LCAs of manure. The program, written in Visual Basic 6.0, uses Data Access Objects 3-5 to work on a relational database based on the SPINE Life Cycle Inventory data documentation format. The SPINE (Carlson et al., 1998) and SPOLD (Singhofen et al., 1996) LCA database formats aim to improve the productivity of the LCA methodology by providing a consistent means of storing and making electronically available reliable inventory data of commonly used goods and services.

3. Method

The method section is broken into subsections addressing:

(1) the scope of the work;
(2) how the manure stream was modelled;
(3) how the data were obtained for the Life Cycle Inventory;
(4) how the data that were aggregated and used for the Life Cycle Impact Assessment;
(5) how uncertainty was modelled; and
(6) scenarios used to illustrate the method.

3.1. Scope of the work and other related assumptions

Agricultural systems produce food products that have standardised commodity definitions. These commodity definitions are an excellent accepted starting point for choosing the measure of common function. Pork is traded on a dead weight basis, which can be related to liveweight using a killing out percentage of ca. 72–77% depending on slaughter weight (MLC, 1998). In this study, the functional unit is 1000 kg pork dead weight, which is used for the presentation of the results.

In a comparative LCA, the system boundaries only need to be set out just so far as to include those aspects that change. If a comparison is being made between two manure application systems then only the application technology and the fate of the nutrients in the soil need be considered. However, if the comparison is between two dietary-practices then all of the activities from the animal to the soil need to be included.

Emissions and resource depletion by the use of energy and material inputs are included in this study. Infrastructure emissions, e.g. those arising from the production of buildings and machinery, are not included. Audsley (1997) established that most items of agricultural infrastructure have no significant impact on the total LCA.

As only manure treatment and application to land are taken into account, there are no by-products from the production system that need to be considered. However, the production of bioenergy, increased crop yields and reduced requirement for nitrogen fertiliser are modelled as the displacement of the energy and emissions needed to produce them.

3.2. Modelling the manure stream

The manure stream is modelled by creating a vector of values $x$ representing the waste stream (see Table 1), that is the physical flow of livestock manure, through numerous process and transport steps. The principal functional flow is the total mass of the manure stream and the reference quantity is taken as one tonne. All inputs of energy and matter are defined with reference to this. In each of the process or transport steps it is necessary to quantify:

(1) inputs of energy or material $s$, such as straw;
(2) biochemical transformations, represented in matrix form $T$; and
(3) a vector of emission values, $e$ produced from the process, represented in matrix form, $B$.
Then for each step in the process, the transformations proportionally map inputs onto outputs (see Table 3):

\[
\begin{bmatrix}
\mathbf{x}' \\
\mathbf{e}'
\end{bmatrix} = \begin{bmatrix}
\mathbf{T} \\
\mathbf{B}
\end{bmatrix} \begin{bmatrix}
\mathbf{x} \\
\mathbf{s}
\end{bmatrix}
\]

where: \(\mathbf{x}\) and \(\mathbf{x}'\) are the manure stream vectors entering and leaving the process, respectively.

### 3.3. Derivation of data

The following subsections discuss the data and, where relevant, the methodology used to calculate the LCI. Data have chiefly been obtained by expert interpretation of the available literature, in-house data and models by a panel of manure experts from Silsoe Research Institute that met between April 1997 and March 2000. The panel comprised: T.R. Cumby, V.R. Phillips, I. Scotford and A.G. Williams. The summation of their expert knowledge has appeared in, amongst others: Pain et al. (1998a, 1998b, 1998c), Sneath et al. (1997a, 1997b) and Misselbrook et al. (2000).

The LCA requires robust estimates of the average performance of a technique. Experimentation reveals the wide range of possible performances, due to the influence of an array of interacting environmental factors, from which an average could be calculated. However, experimental data are also by their nature limited to the constraints of experiments, such as site, duration, measurement error, etc., and are as such an abstraction or an isolated ‘snapshot’ from the real world. The expert panel was used to bridge this gap and provide an estimate of the average, and where relevant, a measure of their confidence in this value, by interpreting what is known.

### 3.3.1. Animals

There are various slaughter weight regimes for pigs that range from pork pigs in the range 50–90 kg liveweight to heavy hogs at 105 kg+ (Varley, 1988). The national average fattening systems purchase pigs at 18.2 kg and fatten to 88.2 kg liveweight with 3.8% mortality and a daily liveweight gain of 0.597 kg (MLC, 1998). The production cycle is thus ca. 118 days. Data and assumptions on manure composition (MAFF, 2000; Vanderholm, 1984) and enteric methane production (OECD, 1991) enabled the quantification of the animal’s outputs (see Table 1).

### Table 1

Values of state variables and inputs at various stages in the treatment system, per 1000 kg of pig dead-weight

<table>
<thead>
<tr>
<th>Stage</th>
<th>From animal</th>
<th>From housing</th>
<th>From anaerobic digester</th>
<th>From storage lagoon</th>
<th>For transfer of waste to field</th>
<th>For application of waste to land</th>
<th>Land</th>
</tr>
</thead>
<tbody>
<tr>
<td>State variables</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Waste total, kg</td>
<td>5450</td>
<td>5450</td>
<td>5450</td>
<td>7903</td>
<td>7903</td>
<td>7903</td>
<td></td>
</tr>
<tr>
<td>Waste total solids, kg</td>
<td>545</td>
<td>545</td>
<td>545</td>
<td>545</td>
<td>545</td>
<td>545</td>
<td></td>
</tr>
<tr>
<td>Waste total water, kg</td>
<td>4905</td>
<td>4905</td>
<td>4905</td>
<td>7358</td>
<td>7358</td>
<td>7358</td>
<td></td>
</tr>
<tr>
<td>Ammoniacal N, kg [N]</td>
<td>18.5</td>
<td>20.4</td>
<td>21.6</td>
<td>21.6</td>
<td>2.6</td>
<td>2.6</td>
<td></td>
</tr>
<tr>
<td>Organic N, kg [N]</td>
<td>43.3</td>
<td>10.8</td>
<td>8.6</td>
<td>6.3</td>
<td>6.3</td>
<td>6.3</td>
<td></td>
</tr>
<tr>
<td>Total organic carbon, kg [C]</td>
<td>174.4</td>
<td>153.7</td>
<td>61.3</td>
<td>60.6</td>
<td>60.6</td>
<td>60.6</td>
<td></td>
</tr>
<tr>
<td>Total copper, kg [Cu]</td>
<td>0.19</td>
<td>0.19</td>
<td>0.19</td>
<td>0.19</td>
<td>0.19</td>
<td>0.19</td>
<td></td>
</tr>
<tr>
<td>Total potassium, kg [K]</td>
<td>21.5</td>
<td>21.5</td>
<td>21.5</td>
<td>21.5</td>
<td>21.5</td>
<td>21.5</td>
<td></td>
</tr>
<tr>
<td>Total phosphor, kg [P]</td>
<td>11.3</td>
<td>11.3</td>
<td>11.3</td>
<td>11.3</td>
<td>11.3</td>
<td>11.3</td>
<td></td>
</tr>
<tr>
<td>Direct emissions</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>CH₄, kg [CH₄]</td>
<td>2.43</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Inputs</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rain water, kg</td>
<td></td>
<td></td>
<td></td>
<td>2453</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Diesel, MJ</td>
<td></td>
<td></td>
<td></td>
<td>14.3</td>
<td>79</td>
<td>28.5</td>
<td></td>
</tr>
<tr>
<td>Light fuel oil, kg</td>
<td></td>
<td></td>
<td></td>
<td>3.4</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Electricity, MJ</td>
<td></td>
<td></td>
<td></td>
<td>–478</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Triple superphosphate, kg [P]</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>–11.3</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Muriate of potash, kg [K]</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>–21.5</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ammonium nitrate, kg [N]</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>–4.8</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wheat grain (conv.), kg [dry matter]</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>–72.9</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Note:** Negative inputs represent displaced inputs (savings)
3.3.2. Housing systems

In the housing system the urinary nitrogen, urea, is rapidly transformed by urease into ammoniacal nitrogen with consequent gaseous losses as ammonia and nitrous oxide. Additionally, anaerobic microbial processes release methane and carbon dioxide (see Table 2).

The data on the nitrogen transformations has been adapted from data on monogastric nitrogen metabolism and excretion in McDonald et al. (1985) and from data on emissions from livestock housing facilities of ammonia (Misselbrook et al., 2000) and \( \text{N}_2\text{O} \) (Sneath et al., 1997b). Similarly, data on methane have been adapted from the national methane inventory (Sneath et al., 1997a) and the IPCC methodology (OECD, 1991).

3.3.3. Transport steps

The energy required for transporting slurry, either pumping along a pipeline or using a tractor and tanker combination, was calculated from established first principles (Moncaster, 1989; Crowther, 1936). Obviously, the energy required as electricity or diesel fuel, for a particular duty, depend on a variety of factors, e.g. slurry density, distance travelled, flow rate or velocity, etc. However, for this study the average energy requirement for pumping slurry along a pipeline from housing to storage was taken as 0.018 and 0.018MJt^-1 of slurry moved, depending on situation. This value concurs with figures provided by various pump manufacturers. Whereas the average energy required to move slurry with a tractor and tanker from storage to field, including the return trip, was taken as 10 MJt^-1 slurry moved this value concurring with those quoted by Dwyer (1985).

3.3.4. Slurry lagoons

Tables 1 and 2 show the effect on the manure of a 6 months period of lagoon storage over winter. The data have been derived by expert interpretation of the United
Kingdom inventories for methane (Sneath et al., 1997a), ammonia (Misselbrook et al., 2000) and nitrous oxide (Sneath et al., 1997b).

3.3.5. Anaerobic digester

Anaerobic digestion is a process that is applied to manure both on farm and in centralised plants. The process is interesting to study in an LCA because it has the potential benefits of renewable energy to offset any energy inputs and environmental impacts due to the modification of the manure.

Carbon and nitrogen are the most important elements undergoing change inside the anaerobic digester. It is assumed that there is an 85% reduction in biological oxygen demand (BOD) due to converting carbon to biogas, most of which is used to generate electrical energy with an emission of carbon dioxide. A fugitive loss of 13.5% of the biogas is not unreasonable for a small-scale farm based digester where energy supply and demand rarely balances. The digestion process converts some of the nitrogen from the relatively stable organic form to the volatile ammoniacal nitrogen form, thus increasing the potency of the nitrogen (see Table 2).

3.3.6. Application systems

The amount of ammonia lost during application of slurry or solid manure to land, depends on many factors including: percentage of dry matter in the slurry or solid manure, temperature, time of the year and method of application. Table 2 illustrates some general figures for ammonia loss for land spreading, which are taken as representative of the typical use in the UK of these techniques. The data assumes that run-off losses should be zero if best practice is used.

3.3.7. Nitrate losses from manure applied to land

Manure applied to land contains a considerable amount of nitrogen, perhaps up to 7 kg [N] t [fresh weight]⁻¹ (MAFF, 2000), even after discounting the ammonia lost shortly after it is applied. In the first year, the amount of this nitrogen available to the crop ranges from about 50% for pig slurry, 30% for cattle slurry to 10–49% for broiler litter (Nicholson et al., 1999). The actual amount varies due to the variable content of manures, the handling and processing that has been applied to the manure between production and spreading, and the rate of bacterial activity in the soil, which is a function of moisture and temperature. For this reason farmers tend, on average, to err on the side of caution when valuing the amount of nitrogen they expect to be contributed to the crop from the manure. Even if the amount is accurately estimated, after the crop is harvested in the autumn, organic matter turnover continues and indeed will increase, as the warm dry soil becomes warm and wet. Therefore, manure added to the crop in spring will contribute to an increase in nitrate in the soil in the winter and to more leaching. Manure nitrogen will also be lost by denitrification due to a series of microbial processes that convert ammoniacal nitrogen to nitrate as well as dinitrogen and nitrous oxide as reviewed in Conrad (1996) amongst others. Of course this will also increase the nitrate in the soil available to the following crop. In decreasing amounts, this effect will continue, slowly increasing nitrate in the soil, until at infinite time the manure nitrogen is fully accounted for. However, because of the variability of the plant available nitrogen available after the first year, few farmers take much account of the nitrogen from the manure after the year of application. Contrarily, a field that has previously received large amounts of manure and now no longer does will require less fertiliser due to the increased organic matter. Work at Rothamsted Experimental Station on soil organic matter turnover (Jenkinson & Rayner, 1977) and nitrogen recommendation programs (Smith & Bradbury, 1996) supports this view.

Factors, such as, application rate, timing of application, soil type and over winter rainfall, which affect nitrate leaching (Beckwith et al., 1998) at a site–specific level, have, in effect, been averaged out. The work here represents the behaviour of the nitrogen cycle under typical UK conditions. Policy makers need to understand the impact of a technology at a national or possibly a regional level.

Appendix A shows the derivation of the nitrogen loss equation for typical pig slurries

\[
L_n = \frac{w(1 - N^*r)(1 - f)(a + b)}{(1 -\mu N_2 r - (1 - w)(1 - N^*r))}
\]  

where: \(L_n\) is total leaching in kg ha⁻¹, \(w\) is proportion of nitrate leached each year, \(N^*\) is the nitrogen content of the harvested crop, \(r\) is the response of the crop harvested yield to nitrogen at the economic optimum on which the farmer bases his application rate, \(f\) is the proportion of the N in waste assumed by the farmer to be available to the crop and thus not applied as manufactured nitrogen in the first year. Thereafter, the farmer makes no adjustment for this waste application, \(a\) is the ammonium-N content converted to nitrate-N, \(b\) is the organic matter nitrogen content, \(\mu\) is the ratio of the crop residue incorporated as organic matter to the harvested yield and \(N_2\) is nitrogen content of crop residues.
Additional nitrogen can similarly be calculated where:

\[ Y = \frac{r}{w(1 - N^s)} L_n \]  

where: \( Y \) is the increase in yield in kg ha\(^{-1} \).

Table 3 shows the effect of changing some of the parameters on the amount of N lost to the environment, the increase in yield and the reduction in nitrogen fertiliser. The application of manure contains 100 kg of total N per hectare. The analysis shows that for most situations the cumulative loss is around 50% of the N applied. Where soils are susceptible to leaching, nitrate is lost rapidly; in the opposite case, the nitrate is lost steadily over many years. In the long term and thus in the steady state (where the same amount of manure is applied every year), there is very little difference between soil types.

In this work it has been assumed that the crop in question is winter wheat, which gives the terms the following values: \( r = 3.5 \) kg [grain dry matter] kg [N fertiliser]\(^{-1} \); \( N_1 = 0.02 \) kg [N] kg [grain dry matter]\(^{-1} \); \( N_2 = 0.001 \) kg [N] kg [grain dry matter]\(^{-1} \); \( \mu = 1 \) kg [straw dry matter] kg [grain dry matter]\(^{-1} \); and \( w = 0.5 \) kg [N leached over winter] kg [soil mineral N]\(^{-1} \).

3.3.8. Goods and services supplied to the production system

Existing product life cycle inventories (Audsley, 1997; Pira International, 1995) have been used to estimate the emissions of a collection of goods and services that are either inputs into the system being modelled or are displaced by the system. Table 4 lists a representative few of the many goods and services currently available in the database. For simplicity the data is expressed as aggregated impacts, which are explained in Section 3.4, but is held within the database as an inventory of individual flows of matter and energy.

3.4. Impact assessment

The Life Cycle Inventory is converted into environmental impacts using the weightings from Audsley (1997). The following environmental impacts are considered: global warming at 20, 100 and 500 yr horizons,
acidification, eutrophication, nitrate leaching and photochemical oxide formation (smog).

These impact categories are due to a variety of different emissions as discussed in O'Neill (1993) amongst others. Global warming is caused by the emission of greenhouse gases, such as carbon dioxide, methane, nitrous oxide, etc. Acidification of aquatic and terrestrial habitats is due to the gaseous release of the oxides of nitrogen and sulphur. Eutrophication or nutrification of aquatic and terrestrial ecosystems is caused by the emission of nitrogen and phosphorus compounds. Smog or photochemical oxide formation is due to the emission of non-methane volatile organic compounds (VOCs), such as pesticides, etc.

A set of environmental valuation weights facilitates a comparison between different impacts. There are many methods of determining such environmental valuations. The method that is used here is based on the size of the current UK environmental emissions inventory. Table 4 shows the values that have been used.

3.5. Modelling of uncertainty

The approach adopted within the model is to use Monte-Carlo simulation to propagate uncertainty based on the coefficients of variation on several key variables. Gaussian distributions and no significant correlation between emission factors are assumed. Estimates of a mean tend to a Gaussian distribution according to the Central Limit Theorem.

In estimating uncertainty it is important to distinguish between the variance from the mean of all possible outcomes, which could be quite wide, and the variance of the mean, which will be much narrower. It is the uncertainty with which the mean performance of a technique is known that is used here. For example, slurry injection might, across all UK field conditions, emit a range of 0–85%, of manure ammoniacal nitrogen as ammonia, but the national mean maybe 10%, as shown in Table 2, which could vary by ±3% say. This study used an expert panel to estimate both the mean performance of a technique under all UK field conditions, methods of use, etc. and their confidence in that mean (Section 3.3).

3.6. Scenarios

Three slurry application techniques, which reduce ammonia emissions, are compared. The ammonia volatilisation coefficients of the techniques have been given 10% coefficients of variation. In addition the methane conversion factors of housing, storage and anaerobic digestion have been given a coefficient of variation of 20% (see Table 2). Table 5 lists the definitions of the scenarios that have been used.

4. Results

The results are presented in progressively increasing detail from the identification of better environmental options to investigating why results are as they are and what that implies about future developments.

4.1. Top level output — indication of best environmental options

Figure 1 shows the results plotted with error bars set at a 95% confidence interval. The move from splash plate application to slurry injection technology results in a significant environmental gain. The band spreader systems are better on average than the splash plate systems, however not significantly. The advantages are maximised if the farmer makes full use of the available nitrogen in the manure, as might be the case with in-line nutrient sensing on manure application systems, the use of standard tables (MAFF, 2000) or computerised decision support systems, such as MANNER (Chambers et al., 1999).

However, the use of anaerobic digestion only has a marginal negative impact (Fig. 1). The advantage of constraining fugitive biogas emission appears to be of
small benefit relative to the reduction in aggregated environmental impact.

4.2. Weighted environmental impact scores

The Life Cycle Assessment methodology can estimate the likely change in UK national inventory if any given set of techniques is substituted for another. Another way of expressing this is how much of the industry needs to change practice to bring about a given, say 0-1%, reduction in an environmental impact. Fig. 2 shows the proportion of the UK pig industry that needs to change from the reference scenario for each improved system. Where columns are missing then no net reduction is possible and similarly where columns exceed 100% of the industry then the 0-1% reduction cannot be achieved. The results suggest that modest 11–20% of pig numbers changing to slurry injection technology from splash plate systems would bring about this level of reduction in acidification and eutrophication, but no other advantage. Acidification and eutrophication, nutrient-related impacts, are a larger proportion of the national inventory than global warming potential (GWP) and smog formation and are strongly influenced by the techniques that have been considered.

4.3. Environmental impacts

Figure 4 shows a multi-axis profile of the environmental impacts for different slurry application techniques. The figure plots the change, relative to the splash plate system, of four major environmental concerns and
also the nitrate-leaching component of eutrophication, which can be a specific issue. If the splash-plate system is taken as a reference, the injector system causes only 64% of the acidification and 71% of the eutrophication. In contrast, the band spreader system offers 28% of the benefits of slurry injection. Ammoniacal nitrogen is conserved in the slurry in the case of the band spreader and slurry injection systems, which leads to additional nitrate leaching and a marginal benefit to the productivity of following crop, assumed to be winter wheat. This reduces the global warming potential and photochemical smog formation due to the use of manufactured ammonium nitrate fertiliser on this crop and displacing winter wheat elsewhere with its requirements for chemical inputs.

The nitrate leaching increase shown in Fig. 3 shows the importance of the farmer making a full allowance for the increased availability of the nitrogen in the manure. Fig. 4 shows three scenarios where the assumed availability of the manure nitrogen increases from 10 to 50%. The reduced nitrate leaching, however, is associated with an increase in photochemical smog formation due to the fact that as more nitrogen is utilised there is less surplus to increase crop productivity. Fertiliser nitrogen optima are often lower than the maximum crop response so surplus manure nitrogen leads to a crop productivity boost, which displaces the need to grow that quantity elsewhere. The results imply that there is more smog production and global warming from 1 kg [N] as winter wheat production and nitrate leaching than 1 kg [N] manufactured as ammonium nitrate. Examination of the data shows this is in fact the case by a factor of 1.05 for global warming and 1.27 for photochemical smog production. The use of pesticides in winter wheat generates the emission of 28.9 g [Volatile Organic Compounds (VOCs)] kg [winter wheat grain dry matter]⁻¹ produced compared to 81.2 g [VOCs] kg [nitrogen as ammonium nitrate]⁻¹. An extra yield of 3.5 kg of winter wheat is produced for every kilogram of additional fertiliser nitrogen at the economic optimum.

Figure 5 shows results of some scenarios using the anaerobic digester. The splash plate system with 13-5% fugitive methane losses shows that these losses generate the same global warming potential as is saved by the generated energy. Thus these fugitive methane losses need to be constrained to realise the benefits of renewable energy. However, smog production is substantially reduced due to the displacement of fossil-based energy.

Secondly, anaerobic digestion metabolises forms of stable organic nitrogen into forms more readily lost. However, it should be remembered from an earlier subsection that energy related impacts are not as important in agriculture as those related to nutrients using the weighting system used here.
5. Discussion

The data and procedures developed here permit the following things:

(a) evaluation of the effect of a change in anything from a single technique or process to a whole system on the net environmental performance of the system as a whole;

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**Fig. 3.** Comparison of slurry application techniques at 30% replacement of fertiliser nitrogen: —, reference system, splash plate application; ......, band spreader; ––, injection

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**Fig. 4.** Comparison of splash plate application technique at different replacement percentages of fertiliser nitrogen: —, 30% (reference system); ......, 10%; ––, 50%
(b) ranking a range of proposed techniques in terms of net environmental burdens, taking into account uncertainty in the performance data of each technique;
(c) investigating total net environmental impact and identify ‘hotspot’ areas, such as manure nitrogen utilisation and thus identify areas for further research and improvement; and
(d) assessment of the impact of changes in agricultural techniques on the national inventory.

However, the LCA does not attempt to consider the cost or practicability of the clean technology options considered here.

5.1. Evaluation of results

ISO 14043 gives insight into methods of interpreting LCA results and hence adding confidence to them. The two following points can be noted.

(1) This LCA study has adopted the important environmental emissions and hence impacts for livestock manures (Turner, 1999). However, it should be noted that the emissions of pathogens and odours have not been considered.
(2) The activities and processes used to model each system in this study are those identified as important in the compilation of relevant emission inventories from agriculture, namely the animal, housing, manure storage and manure application to land (Sneath et al., 1997a, 1997b; Misselbrook et al., 2000). As a comparative assessment, processes that are identical in all scenarios are omitted.

The remaining issue is data quality and this has been evaluated in part by the use of scenarios that include a range of possible parameter values and the explicit use of uncertainty by way of Monte-Carlo simulation. The measures of uncertainty that have been used are based on expert opinion and as such are uncertain themselves. Uncertainty is an important component in selecting techniques if farmers are legally required to use them to ensure that significant environmental benefits are achieved to justify the almost inevitable increase in costs.

One of the assumptions of the LCA is the allocation of only one product to be displaced by changes in the flows of manufactured inputs or outputs, such as the supra-optimal yield response, due to unaccounted manure nitrogen, displacing the need to grow ‘conventionally’ farmed winter wheat elsewhere. In reality manure can be applied to a range of crops farmed both intensively and extensively. Similarly energy from anaerobic digestion can be used to replace more than electricity; it can be used as heat to displace...
oil or gas. However, this assumption should not affect the result profoundly because the displacement products chosen are the most widespread and one would expect equivalent substitutes to behave in a similar way. One exception is the application of slurry to grassland. Additional grass yields do not involve the same pesticide inputs, hence VOC emissions, hence smog formation. Thus there may be fewer side effects from applying surplus nitrogen to grassland. However the fact that the energy related impacts, smog and global warming, have a low weighting dilutes any effects considerably.

The use of weighted-environmental impact categories introduces a value judgement when evaluating one impact category against another. In this case, a distance to target approach using the UK national inventory has been adopted. Another set of criteria could be adopted that elevates the importance of greenhouse gases and consequently global warming against eutrophication and acidification. In which case anaerobic digestion and the energy intensity of the slurry injection system may be much more important.

6. Conclusions

Systematic data and procedures for analysing livestock manure treatment systems have been constructed and used to compare three application and one treatment system. The following conclusions can be drawn:

(1) Slurry injection is significantly better for reducing eutrophication and acidification impacts than splash plate. The following secondary points can be made:
(a) if ca. 20% of the pig industry changed from splash plate systems to this system, the UK national emissions inventory of these impacts would be reduced by 0.1% and
(b) it is difficult to have a useful effect on global warming, using the UK national emissions inventory to weight the individual impacts.

(2) The band spreader system reduces the environmental impacts but not significantly, considering the uncertainty in emission estimates.

(3) Anaerobic digestion is of very marginal benefit because of fugitive losses and because of the valuation based on the UK national emissions inventory. A different valuation and no losses would increase the benefit.

(4) Technology to enable farmers to better utilise the N in the manure such as in-line nutrient analysis or other forms of manure analysis and decision support have an important role to complement ammonia abatement techniques that conserve nitrogen content of the manure.

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Appendix A Derivation of nitrogen loss equations

Consider a waste applied to land containing ammonium (a) and organic matter nitrogen (b). The ammonium is either converted quickly to nitrate available to the crop or lost as ammonia. In analysing the environmental impact of applying waste to land, thus accounting for 100% of the manure nitrogen, four factors need to be taken into account:

1. the immediate release of ammonia to air;
2. the supply of nitrogen to the first crop, reducing the use of manufactured nitrogen and increasing crop yield;
3. the increase in nitrogen losses after the first crop, taken to infinite time; and
4. the increase in nitrogen available to crops after the first crop and consequent increase in crop yield, again taken to infinite time.

Note that in fact there may be three sources of loss of nitrogen from the system: nitrate leaching, nitrous oxide and nitrogen gas (benign denitrification). Thus if $w_1$, $w_2$, $w_3$ are the three losses as the annual proportion of the nitrate left after the crop, then $w = w_1 + w_2 + w_3$. $w$ is the total annual proportion of nitrogen lost.

The nitrogen available to the crop after the waste is applied $A_1$:

$$A_1 = a + c + p(b + m) - f(a + b)$$

where: $c$ is the nitrate-N normally applied to the crop by the farmer in the absence of waste kg [N] ha$^{-1}$; $p$ is the organic matter-N converted to nitrate in a year (proportion); $m$ is the organic matter-N available annually before the waste was applied in kg [N] ha$^{-1}$; and $f$ is the N in waste assumed by the farmer to be available to the crop (proportion). Comparing this to $(c+pm)$ without the waste

$$\Delta y_1 = r \Delta A_1 = 1 - (a + pb - f(a + b))$$

where: $r$ is the response of the crop’s harvested yield to nitrogen at the economic optimum on which the farmer bases his application rate in kg [grain] kg [N]$^{-1}$.

The increase in residual nitrogen after the crop $R_1$ is

$$\Delta R_1 = \Delta A_1 - (N_1 + N_2(1 - \mu))\Delta y_1$$
where; \( N_1 \) is the nitrogen content of the harvested crop (kg [N] kg [dry matter] \(^{-1}\)); and \( N_2 \) is the nitrogen content of crop residue in kg [N] kg [dry matter] \(^{-1}\) and \( \mu \) is the residue incorporated as organic matter (proportion of harvested yield). To simplify let

\[
N^* = N_1 + N_2(1 - \mu)
\]

Nitrogen losses are \( wR \) and the nitrogen after losses is \( R(1-w) \).

The organic matter in the soil after the crop \((O_1)\) is

\[
O_1 = (b + m)(1 - p) + \mu N_2 (y + \Delta y_1)
\]

Note that this is increased due to the increased yield due to the extra nitrogen. Therefore this can be simplified to

\[
\Delta y_2 = r(1 - w)\Delta R_1 + rp\Delta O_1
\]

and in general

\[
\Delta R_n = (1 - w)\Delta R_{n-1} + p\Delta O_{n-1} - N^*\Delta y_n
\]

\[
\Delta O_n = (1 - p)\Delta O_{n-1} + \mu N_2 \Delta y_n
\]

which can be simplified to

\[
R_{n+1} - \alpha R_n + \beta R_{n-1} = 0
\]

where

\[
\alpha = (1 - w)N^* r + (1 - p + \mu N_2 p)
\]

\[
\beta = (1 - w)(1 - N^* r)(1 - p)
\]

The solution of this difference equation is

\[
R_{n+1} = k_1 M_1^n + k_2 M_2^n
\]

where

\[
M_1 = \frac{(x + \sqrt{x^2 - 4\beta})}{2} \quad M_2 = \frac{(x - \sqrt{x^2 - 4\beta})}{2}
\]

\[
k_1 = \frac{(\epsilon_1 - \epsilon_2 M_2)}{M_1(M_1 - M_2)} \quad k_2 = \frac{(\epsilon_1 - \epsilon_2 M_1)}{M_2(M_2 - M_1)}
\]

\[
\epsilon_1 = (\lambda(1 - w)(1 - N^* r) + pc(1 - p) + \mu N_2 r p); \epsilon_2 = \gamma(1 - N^* r)
\]

\[
\gamma = b + pc - f(b + c)
\]

Thus total nitrogen loss \( L \) due to the application of manure taken to infinite time is

\[
L = \frac{w_1 M_1}{(1 - M_1)} + \frac{w_2 M_2}{(1 - M_2)}
\]

This can be simplified to

\[
L = \frac{w(1 - N^* r)(1 - f)(a + b)}{(1 - \mu N_2 r - (1 - w)(1 - N^* r))}
\]

The increase in yield \( Y \) in kg [grain] ha\(^{-1}\) due to the additional nitrogen can similarly be calculated

\[
Y = \frac{r}{w(1 - N^* r)} L
\]

It is of course known that manure has other potential benefits to the soil other than simply its nitrogen content. The organic matter in the manure improves the water holding characteristics of the soil, and thus there could be an additional benefit due to reduced stress, dependent on the soil type. This can be directly included in the above analysis but as we are assuming a constant rate of release of nitrogen over time from the manure, the residual organic matter content is proportional to the nitrogen released. Therefore this effect can be incorporated simply, by increasing the marginal response of the crop yield to nitrogen.