AN ASSESSMENT OF THE ENVIRONMENTAL IMPACTS OF ORGANIC FARMING

A review for Defra-funded project OF0405

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CONTENTS

SUMMARY

There is currently considerable interest in organic farming as a method to deliver environmental goods. This review was therefore undertaken to assess the likely benefits to the wider environment from organic practices. On the basis of the published scientific evidence, we have concluded the following:

Biodiversity: Comparative reviews of the evidence base have been conducted for MAFF, English Nature, The European Commission and the Soil Association. The general conclusion is that, on average, there is a positive benefit to wildlife conservation on organic farms. In most studies, organic agriculture provides a conservation benefit, whereas there are few studies where a disbenefit is shown. While some of the practices that favour biodiversity are used on some conventional farms, it is only generally on organic farms where most of the relevant management is routinely and systematically carried out. Both organic and conventional farms will perform better when under agri-environmental schemes.

Soil quality: There are few UK studies on the relative benefits of organic or conventional systems for soil quality. However, such studies as have been done and those from other countries tend to show benefits for organic systems. Organic farmers pay particular attention to their soils, and it is a fundamental tenet of organic farming to operate a sound rotational system to "feed the soil" to maintain organic matter content and to keep it in good condition. However, organic matter additions are also made in conventional agriculture and, in some situations, the return may be similar or greater than in organic systems. Soil structure can benefit from regular returns of organic matter, and the evidence is that soil structure is at least as good under organic practices. Earthworm numbers tend to be greater in organic systems and studies into the microbial response of soils to organic management indicate there are benefits in many but not all situations and not always in all the attributes measured. The low concentration of soluble nutrients, the absence of most pesticides and reduced use of veterinary medicines such as antibiotics and ivermectins can be also expected to benefit soil organisms.

Nitrate in water: Many organic systems operate at a lower level of nitrogen intensity than conventional systems, with nitrogen inputs from fixation by legumes, or from importation of animal feed onto the farm. Variation in leaching losses from individual fields is large both in organic and conventional agriculture. Organic farming adopts many of the practices that should decrease losses: maximising periods of green cover, use of straw-based manure, lower stocking densities. The body of evidence suggests that leaching losses are generally less from organic systems – though this is not always guaranteed. It might also be argued that this differential would decline as conventional fertiliser practices improve under the increasing regulatory pressure. Losses after ploughing the fertility building leys is one area of organic farming where losses can be especially large.

Phosphorus in water: The main loss pathway for phosphorus is by movement of soil particles. Leaching is a smaller and more site-limited effect. There are some additional "incidental" losses following the application of fertilisers or manure. There is no direct evidence of differences in phosphorus losses between organic and conventional agriculture.

Pesticide pollution to water (and air): Pesticide use in organic farming is very restricted. A small number of pesticides are approved for organic use (principally copper, sulphur, natural pyrethroids, and derris), and they are only used as a last resort. The pyrethroids, copper and

derris are only permitted for use in protected cropping or for a restricted range of horticultural crops. With the exception of sulphur, on certain top fruit crops and pyrethroid sheep dip (which can be used in the same way on both organic and conventional farms), the use of the restricted range of pesticides is very limited by comparison with conventional agriculture. In particular, organic farmers do not use herbicides, some of which (such as isoproturon) have presented particular water pollution problems. Pesticide pollution from organic farming will be far less common than pesticide pollution from conventional agriculture. These differences are likely to hold whether assessed per area, or per unit of food produced.

Human Pathogens: Pathogenic organisms from livestock can contaminate surface waters used for drinking, bathing or irrigation. There is no reliable information on any differences in the incidence of zoonoses between organic and conventional farms that use manure, although there is on-going research. Studies have shown that composting manure and treating slurry, as encouraged under organic standards, decrease the survival of any pathogenic organisms but stacking or long-term storage can also be beneficial. The methods of handling manure between farming systems may not be sufficiently different to produce a consistent effect and, therefore, information on the incidence the organisms is needed before any conclusions can be drawn.

Ammonia: Ammonia is mainly lost from the surface of manure, either from animal buildings or hardstandings, which are soiled by manure, or during storage and handling. Manure produced in organic systems often has a lower concentration of nitrogen than does conventionally produced manure. Organic systems encourage the composting of manure, which leads to a relatively high loss of ammonia, although this will reduce the amount emitted when the compost is subsequently spread. Given the constraints on housing and stocking rate it is not possible to have organically certified intensive pig and poultry units, which are a major source of ammonia from conventional systems. Organic pigs and poultry are likely to have similar losses to conventional outdoor units at the same stocking densities. It seems likely that on balance there is little difference between organic and conventional systems in the amount of ammonia which is lost from the system per unit of yield, but it is likely that emissions are lower per unit area. Given that nitrogen is more valuable to organic systems than it is to conventional systems (which can purchase nitrogen fertiliser at about 30p per kg), there should be a greater incentive for organic farmers to control ammonia losses in the future.

Nitrous oxide: Nitrous oxide is emitted from manure and from soils. Emission tends to occur intermittently when there is a combination of the appropriate conditions. Within conventional agriculture, the main risks arise from manure and from the waterlogging of soils by heavy rainfall following fertiliser application. Within organic farming the risks are likely to come from manure and from waterlogging of soils where there is a legume crop. In the absence of direct measurement, it is not possible to assess whether there is any difference in risk from organic or conventional production.

Methane: About 75% of methane on farms is emitted directly from ruminant animals (chiefly cattle and sheep). There have been no direct comparisons of methane generation between organic and conventional production. Different types of fodder will generate different amounts of methane, with higher rates released from diets that are high in roughage relative to diets high in starch. This will tend to result in higher emissions from organic systems, as organic diets tend to be high in roughage and low in concentrates. Methane emission per unit of livestock product decreases as the intensity of animal production increases (two cows producing 5,000 litres of milk will generate more methane than one cow producing 10,000 litres). On average, production intensity is lower in organic than conventional systems, so methane generation from organic sheep and cattle farms is likely to be greater per unit of food produced. Because of the lower stocking densities, it maybe similar or less on an area basis.

Carbon dioxide: Net emissions of carbon dioxide from agriculture depend upon use of fossil fuel and the amount of carbon sequestration in soil organic matter. Emission from fossil fuel use will be lower on a per unit area and a per unit of yield basis, reflecting the greater energy efficiency of organic agriculture noted below. There is insufficient evidence on whether there is a significant difference in the amounts of carbon sequestered in soils.

Energy efficiency: The literature supports the statement that organic methods generally use less energy per unit area and per unit of output, both for individual crops and livestock types, and overall on a whole-farm basis. However, the setting of system boundaries, methods of calculating the energy values of inputs and methods of calculating energy use efficiencies vary substantially between studies. The intensity of production in the conventional comparison, particularly in relation to the level of use of mineral nitrogen fertiliser, also had a large impact on the relative performance of organic methods in comparative studies. This makes comparisons across studies difficult; there is a need for an agreed standard methodology. Information is lacking for non-ruminant livestock

Nutrient balance and use: Comparisons of nutrient budgets suggests that the balances can vary widely within a farming system. However, the general conclusion is that organic systems operate smaller nutrient surpluses. This is taken as an advantage, providing that nutrient reserves are not being depleted. Prohibition of various fertiliser additions is on the basis of encouraging self-sufficiency in a system and/or concern about damaging the soil ecosystem. However, evidence for the latter is largely anecdotal and there is a need to continually review the lists of allowed and disallowed products to ensure that choices are environmentally sound.

Controlled wastes: Waste is generally lower in organic farming since the system relies less on external inputs. Packaging materials for agrochemicals, veterinary medicine, animal feed, and fertilisers should all be lower on organic holdings. There is also little need for disposal of pesticide washings on organic systems.

The general conclusion from our review therefore concurs with other reports that organic farming can deliver positive environmental benefits. However, some of the benefits, particularly of lower levels of gaseous emissions, decrease or are lost if comparisons are made on the basis of unit production rather than area. It should also be noted that the differences depend on farming system, with fewer benefits likely to accrue from converting extensive upland production, compared with converting intensive lowland systems.

1. INTRODUCTION

1.1. Background

Defra published its *'Action Plan to Develop Organic Food and Farming in England'* in June 2002 (Anon., 2002a). The Action Plan aimed to ensure stable and strategic growth for the organic sector, and it set out steps that the Government and the food and farming industry would take to encourage a sustainable organic farming and food sector in England. Similar plans have been published for Wales and Scotland. Although organic farming is just one strand of The Government's Strategy for Sustainable Food and Farming, its uptake is being encouraged because it is considered to deliver benefits for the environment.

If environmental benefit is a key driver for Government support of organic farming, there needs to be a collation of the scientific evidence to confirm this. This same debate has been held across many countries as they reassess their agricultural policies (e.g. Stolze *et al*., 2000; Condron *et al*., 2000; Hansen *et al*., 2001; Stockdale *et al*., 2001). The aim of this report is to review the evidence in the UK context.

1.2. Objective

To provide a fully referenced and argued paper which collates the evidence used in developing "The Organic Farming and the Environment" paper published as Annex 3 of the Defra Organic Action Plan (Anon., 2002a).

1.3. Organic ethos

Organic agriculture is a systems approach to agricultural production that is working towards an environmentally, socially and economically sustainable production. The International Federation of Organic Agriculture Movements (IFOAM) defines organic agriculture as "… *a whole system approach based upon a set of processes resulting in a sustainable ecosystem, safe food, good nutrition, animal welfare and social justice. Organic production therefore is more than a system of production that includes or excludes certain inputs"* (Anon., 2002b).

Organic agriculture is based on a philosophy and a set of principles that are best encompassed by the IFOAM principles (Anon., 2002b). These are:

- To produce sufficient quantities of high quality food, fibre and other products.
- To work compatibly with natural cycles and living systems through the soil, plants and animals in the entire production system.
- To recognise the wider social and ecological impact of the organic production and processing system.
- To maintain and increase long-term fertility and biological activity of soils using locally adapted cultural, biological and mechanical methods as opposed to reliance on inputs.
- To maintain and encourage agricultural and natural biodiversity on the farm and surrounds through the use of sustainable production systems and the protection of plant and wildlife habitats.
- To maintain and conserve genetic diversity through attention to on-farm management of genetic resources.
- To promote the responsible use and conservation of water and all life therein.

- To use, as far as possible, renewable resources in production and processing systems and avoid pollution and waste.
- To foster local and regional production and distribution.
- To create a harmonious balance between crop production and animal husbandry.
- To provide living conditions that allows animals to express the basic aspects of their innate behaviour
- To utilise biodegradable, recyclable and recycled packaging materials.
- To provide everyone involved in organic farming and processing with a quality of life that satisfies their basic needs, within a safe, secure and healthy working environment.
- To support the establishment of an entire production, processing and distribution chain which is both socially just and ecologically responsible.
- To recognise the importance of, and protect and learn from, indigenous knowledge and traditional farming systems.

Within the EU, organic farming is a legally defined production system as set out under Council Regulation (EEC) 2092/91 and its amendments (Anon., 1991). Within the regulation, each member state is required to establish a competent authority to implement the regulation. Within the UK, until 2003, this authority has been the UK Register of Organic Food Standards (UKROFS) which provides baseline organic standards for the UK (Anon., 2001a) and approves and monitors the work of UK certification bodies. Within the UK, there were twelve certification bodies at the start of 2003 (Anon., 2003a), which set their own organic standards (based on, and with the UKROFS basic standard as a minimum) and register organic producers and processors. It is the production system that is being certified.

To become organic, a producer must become registered with one of the certification bodies and the land has to be converted. Conversion is typically a two-year process where the land is farmed under organic principles and standards, but any produce from the land is not certified as organic and so cannot be sold as organic although it may be identified as “in conversion”. It is the responsibility of the certification body to inspect the producer on a regular basis to ensure that the producer is complying with organic standards.

1.4. Organic farming systems in the UK

Organic food is a growth market in the UK. In 2001/02, organic food sales reached £920 million, which was an increase of 15% on the previous year (Anon., 2002c). However, the majority (65%) of organic food bought by UK consumers is imported (Anon., 2002c).

Defra statistics (Anon., 2002d) show that, in June 2002, the area of land farmed organically within the UK was 699,879 ha (approximately 3.8% of UK agricultural land). Of this land, 459,903 ha had completed the two-year conversion period and was fully organic, the remaining 239,976 ha was in the conversion period. Soil Association data differ slightly, but generally concur with 729,550 ha being farmed organically in April 2002 - of which 458,600 was fully converted (Anon., 2002c). The geographical distribution and the use of this organic land are not uniform.

The Soil Association data for regional distribution of organic producers shows a skewed pattern, with nearly 60% of producers being located in the Southwest, Scotland and Wales (Anon., 2002c). The Defra data show that 56.6% of UK organically farmed land (compared to 33.3% of total agricultural land) was in Scotland, 34.3% (compared to 51.8% of total agricultural land) in England, 8.1% (8.9% of total agricultural land) in Wales and 1% (compared to 6.0 of total agricultural land) in Northern Ireland (see Table 1.1).

Table 1.1. Organically farmed and total agricultural land in the UK in 2002 (Anon., 2002c; Anon., 2003b).

Region	**Organically managed land**			**UK agricultural land**	
	Area (hectares)	**% of Organic**	**% of agricultural land**	**Area (hectares)**	**% of agricultural land**
England	240,057	34.3	2.5	9,526,849	51.8
Scotland	396,142	56.6	6.5	1,632,504	33.3
Wales	56,621	8.1	3.5	6,120,730	8.9
N. Ireland	7,059	1.0	0.6	1,108,293	6.0
Total	699,879	100	3.8	18,388,376	100

Table 1.2. Organically farmed land area by enterprise in the UK in 2002.

Land Use	**Organically farmed land.** (Anon., 2002c)		**UK agricultural land** (Anon., 2003b)	
	hectares	**%**	**hectares**	**%**
Crops	32,734	7	4,605,000	25
Temporary pasture, ley and set-a-side	54,500	12	1,841,000	10
Permanent pasture including rough grazing	369,766	81	11,140,000	61
Other, i.e. woodlands, roads, yards	1,600	0.3	802,000	4
Total	458,600	100	18,388,000	100

The use of land in organic agriculture also shows an imbalance and is heavily skewed towards pasture (Table 1.2). In 2002, over 90% of organic land was under pasture and much of this would be permanent pasture in the uplands and highlands. When this is compared to the Agricultural Census data for 2002 (Anon., 2003b) it shows that organic agriculture has a very different land use structure than agriculture as a whole in the UK. Organic farms have less cropped area and increased pasture. This can be partly explained by the need for fertility building periods within the organic system. However, it may also be due to conversion of upland farms, particularly in Scotland; the economics of organic conversion and production being more favourable in livestock systems.

Watson *et al.* (2002a) stated that organic farming systems fall into similar categories as those for conventional farming:

- Mixed systems – built around fertility building (ley) and fertility exploiting (arable/horticultural crop) phases. Livestock are used to exploit the leys.
- Livestock systems – long-term pasture systems
- Stockless systems – tending to develop on farms converting to organic production where there is no expertise or infrastructure for livestock management. Systems employ fertility building and depleting phases. Managing stockless systems sustainably is challenging,

especially in terms of nutrient management, but this type of farm is relatively uncommon as an organic system.

- Horticultural systems – often intensive systems, providing challenges for nutrient management in particular.

1.5. Productivity levels and stocking rates

Crop yields are generally less in organic systems, although not in all cases and organic yields can be very variable. Table 1.3 shows average crops yields from organic and conventional systems, based on standard values.

Table 1.3. Yields of organic (Lampkin et al., 2002) and conventional crops (Nix & Hill, 2002).

Crop	Organic	Conventional	Crop	Organic	Conventional
Wheat (winter)	4.0	7.7 to 8.5	Potatoes[1]	25	42.5
Wheat (spring)	3.2	5.8	Cabbage	25 to 35	30
Barley (winter)	3.7	6.4	Carrots	36	45
Barley (spring)	3.2	5.8	Onions	20	35
Oats (winter)	4.0	6.8	Apples	10.4	13
Oats (spring)	3.5	5.5			

[1]main crop

Table 1.4. Yields of crops before and after conversion under Defra's Organic Farming Scheme (Anon., 2002e).

	Area (ha)[1]		%	Yield (t/ha)		%	Production (t)[1]		%
	before	after	change	before	after	change	before	after	change
All cereals	7,104	6,009	-15%	6.3	4.1	-35%	44,858	24,712	-45%
Winter cereals	6,219	4,882	-22%	6.5	4.3	-34%	40,610	20,993	-48%
Spring cereals	885	1,127	+27%	4.8	3.3	-31%	4,248	3,719	-12%
Potatoes	312	309	-1%	41	25	-39%	12,823	7,725	-40%
Fodder maize	667	293	-56%						
Field-scale veg.	608	734	+20%						
Market legumes	401	520	+30%						
Forage legumes	377	972	+157%						
Set-aside incl. leys and fallow)	938	1,842	+96%						
Temporary grass < 5 years	4,970	5,821	+17%						
Perm. pasture	8,858	9,360	+6%						
Farm woodland[2]	1,058	1,055	-0.3%						

[1] Totals over all survey farms.
[2] Including coppice.

The 'standard' values show large yield reductions in cereals (*c.* 41%, averaged across wheat, barley and oats) and potatoes (*c.* 43%), as well as reductions in other crop yields. However, these data can be supported by yield estimates taken before and after conversion under Defra's Organic Farming Scheme (Table 1.4: Anon., 2002e). The data suggest that conversion leads to significant reductions in the area given to supported commodities and that this, combined with smaller yields, results in a marked reduction in the production of these commodities.

There are also differences in the way that organic and conventional farms are stocked. Standards set maximum stocking levels based on the N output from each animal. This aside, stocking densities in organic systems are also generally limited by the productive capacity of the systems. Whereas intensive stocking rates in conventional systems are underpinned by N fertiliser (and feed) inputs, forage production to support livestock rates on organic holdings relies mainly on N fixation by legumes. Levels of forage productivity will be influenced by many factors but, generally, there are lower stocking rates (per ha) on organic farms than on non-organic farms. Padel (1997) reported that, in established organic dairy systems, stocking rates are on average at about 80-90% of the conventional system but, in beef and sheep systems, the variation is greater, ranging from similar levels of production as conventional to reductions of up to 60%.

However, the differences will be greater or lesser depending on a range of external influences and legal restrictions. Nitrate Vulnerable Zones (NVZs) are a good case in point. All farms within NVZs have restrictions on N loadings that, unless the farm exports manure, will restrict stocking rates. Organic farmers can also export manure to comply with rules but only to other organic farmers. Whereas the maximum N loading from manure on an organic farm is restricted to 170 kg/ha N (averaged over the cropped area) by the organic regulations, loadings for NVZs differ according to land use (Anon., 2001b):

- 250 kg/ha N, averaged over the area of grass
- 210 kg/ha N, averaged over the area of the farm not in grass

The 250 kg/ha N limit for grass is outside the Nitrates Directive recommended rates, but the UK is seeking a derogation based on a longer growing season for grass than arable/horticultural crops. The 210 kg/ha N limit for non-grass areas declines to 170 kg/ha N after four years, in line with the Nitrates Directive. Thus, considering that organic farms have to adhere to the lowest N loading of 170 kg/ha N, then it is likely that stocking rates will be less than on conventional farms. Also, currently only about 55% of England is designated as NVZs (though this may change in the future), so that conventional farms outside the NVZs have no restrictions on stocking rates apart from new intensive pig and poultry farms. These are covered by the EC IPPC (Integrated Pollution Prevention and Control) Directive, enacted in England through the Pollution Prevention and Control (England and Wales) Regulations 2000.

Table 1.5 shows examples of the range of stocking levels on various farm types. However, because of the different structures of these different farm types, comparisons are very difficult. In particular, pest and disease pressures might further decrease stocking densities on organic farms below the theoretical maximum based on N loadings. Note also that there is not always agreement between conventional stocking rates and those permitted within NVZs as calculated from manure loadings. This is because, without regulation, there was no limit on stocking rates: management considerations other than N loadings controlled the rate.

Table 1.5. Stocking rates (animals/ha) for livestock on Organic (Anon., 2001a), conventional (Nix & Hill, 2002) and NVZ farms (Anon., 2001b).

Enterprise	Organic	Conventional	NVZ (according to N loading, kg/ha) 250	210	170
Dairy	Max. 2	1.75 to 2.5	2.2 to 3.3	1.8 to 2.8	1.5 to 2.2
Beef (1-2 year old)	Max. 3.3	4 to10	5.3	4.5	3.6
Sheep	Max 13.3	8 to14	28	23	19
Pigs (fattening)	Max. 14	12 to25	13	11	9

A further difference between organic and conventional farming is that intensive pig and poultry units are not permitted in organic farming. This means that organic systems avoid the large-scale production units with a heavy reliance on imported feed and limited land on which to spread the manure (Table 1.6). For poultry, the UKROFS state that poultry must be reared in open-range conditions and cannot be kept in cages. Buildings for all poultry must meet the following minimum conditions:

- poultry houses must be structures with their own dedicated grazing, air space, ventilation, feed and water;
- at least one third shall be solid, that is, not of slatted or of grid construction, and covered with a litter material such as straw, wood shavings, sand or turf;
- in poultry houses for laying hens, a sufficiently large part of the floor area available to the hens must be available for the collection of bird droppings;
- they must have perches of a size and number commensurate with the size of the group and of the birds;
- they must have exit/entry pop-holes of a size adequate for the birds, and these popholes must have a combined length of at least 4 m per 100 m^2 area of the house available to the birds;
- each poultry house must not contain more than 4800 chickens or 3000 laying hens.

Also, all mammals (i.e. including pigs) must have access to pasturage or an open-air exercise area or an open-air run which may be partially covered and they must be able to use these areas whenever the physiological condition of the animal**,** the weather conditions and the state of the ground permit, unless there are EU or National requirements relating to specific animal health problems that prevent this. Herbivores must have access to pasturage whenever conditions allow.

Table 1.7, reported by Anon. (2002e), shows the effects on livestock production on a sample of farms following conversion to organic farming under Defra's Organic Farming Scheme. The data do not give any indication of stocking densities, but illustrate how a range of farms adjusted production in light of the organic standards. The greatest reduction was in pig production, presumably because of the need for more extensive systems. Milk yield per cow fell, but this was compensated for by more cows, so that milk quotas were fulfilled. Anon. (2002e) noted that decreases in milk yield are not as severe as decreases in arable production following organic conversion and this may be one reason why conversion is more popular with livestock than with arable farmers. Poultry (egg) production fell by 10% but, again, this does not inform about how production was restructured. Anon. (2002e) report that one producer reduced production by 80,000 birds during conversion, but this was compensated for by an increased number of farmers who started small to medium-scale egg production.

Table 1.6. Stocking rates for housed organic and conventional intensive pig and poultry housing units.

Enterprise	Max. house size		Max. stocking density	
	Conventional	Organic[a]	Conventional	Organic[a]
Poultry (eggs)	No limit[b]	3000	22 birds/m^2 [c e]	6 birds/m^2 [d]
Poultry (meat)	No limit[b]	4800	34 kg/m^2 (17 birds)[c]	21 kg/m^2 (10 birds)[d]
Pigs for fattening	No limit[g]	n/a[f]	1 - 6 pigs/m^2 depending on size	0.75-1.25 pigs/m^2 depending on size

[a]UKROFS – Individual Certification Bodies may have smaller limits.
[b]Typically, house size 40,000 to >100,000 birds.
[c]No outdoor access required.
[d]Outdoor access required.
[e]There may be several tiers of cages.
[f] Pigs must have access to an outdoor area under organic regulations (apart from the final fattening stage – maximum 20% of lifetime)
[g]Typically, house size >2000 fattening pigs

Table 1.7. Livestock production before and after conversion under Defra's Organic Farming Scheme (Anon., 2002e).

	Numbers		**%**	**Yield (l/cow/year)**		**%**	**Production (l/year)**		**%**
	before	**after**	**change**	**before**	**after**	**change**	**before**	**after**	**change**
Ewes	20,560	16,962	-18%						
Dairy cows	6,491	6,984	+7.5%	6,520	6,157	-5.6%	42 M	43 M	+1.6%
Beef cows/bulls	2,915	3,702	+27%						
Fattening pigs	6,800	4,177	-39%						
Breeding sows	945	458	-52%						
Poultry for egg production	264,908	238,620	-10%						

1.6. Nutrient Use and Balance

Organic farming systems attempt to be as self-sufficient as possible in terms of resource use. This potentially has two advantages when considering nutrient use: minimising import of fertilisers onto the farm and the associated costs of fertiliser production, as described earlier; and minimising nutrient surpluses on the farm. Both of these aspects, however, require further investigation. Firstly, a number of fertiliser materials are permissible for use, although often only in restricted situations. Nevertheless, the environmental implication of their use needs to be assessed. Secondly, there is a risk of depleting soil nutrient reserves and therefore, degrading a valuable resource.

Nutrient balance calculations are increasingly used as a tool for farm planning (Watson *et al.*, 2002b) and policy planning. They are useful as a guide to resource use and for judging the sustainability of a system. Regarding the latter, a large negative balance would suggest that the system was relying on utilising soil nutrient reserves that, in the long-term, would not be sustainable. A large surplus would be of concern because it could be taken as an indicator that losses to the environment could potentially be large. This would be of concern, particularly for N or P. However, it should be stressed that the link between large surpluses and greater environmental losses has not been fully explored. Indeed, Lord *et al.* (2002) found that the relationship between N balance and nitrate leaching was different for grassland and arable systems and was also strongly influenced by climate, level of inputs and management practices.

However, it is sometimes difficult to define ‘optimum’ nutrient status, because this will depend on the objectives of the system – species rich meadow in extensive livestock production vs. e.g. potato production, for example, will require quite different levels of soil fertility. However, the aim should be neither to run down fertility to detrimentally low levels, nor should it be to enrich the soil unnecessarily. The former degrades a valuable resource and the latter can cause pollution of N and P, as described above.

Nutrient balances are generally calculated for the ‘farm gate’ or ‘soil surface’. Table 1.8 provides a summary of what might be included in these calculations, though actual methodologies can differ between workers. Neither balance takes account of losses: the potential for these is assumed to be related to the size of the surplus, as described above.

Table 1.8. Calculation of nutrient balances – summary of methodology.

Type	Inputs	Outputs
Farm Gate	Nutrient content of purchased materials (feed, fodder, livestock, etc.), imported manure, biological N fixation, atmospheric deposition.	Nutrients in produce sold off the farm, manure exported from the farm
Soil Surface	N fixation, mineral fertilisers, manure applications, atmospheric deposition	Nutrients in harvested produce (plus crop residues if also removed)

There have been many comparisons of farm nutrient budgets, both for different organic farms and also comparing conventional and organic farming.

Watson *et al.* (2002b), summarising several datasets, demonstrated that NPK surpluses could vary widely between organic farms: +1 to +400 kg N/ha/year, -7 to +90 kg P/ha/year and –27 to +280 kg K/ha/year. This was also demonstrated by other case studies for UK organic farms (Berry *et al.*, 2002). In both cases, those with the largest surpluses generally imported more nutrients, either in manure or by having a large proportion of N fixing crops in the rotation (Watson *et al.*, 1994; Berry *et al.*, 2002). Given this wide variation between individual farms, care has to be taken when comparing budgets between conventional and organic farms, particularly as these studies are generally made across a few farms of each type. However, Table 1.9 compares nutrient budgets for England and Wales (Webb *et al.*, 2001) with means produced by Watson *et al.* (2002b). The nutrient balances were calculated on the same basis (soil surface), though the England and Wales budgets were calculated from national data and the organic farms were based on individual farm case studies (and are therefore limited in numbers).

Table 1.9. Comparison of soil surface balances (kg/ha) for England and Wales (Webb et al., 2001) and a collection of organic farms (Watson et al., 2002b), with standard error. Number of organic farms indicated as a superscript.

Nutrient	**Conventional '96**		**Conventional '97**		**Organic**	
	Arable	**Grass**	**Arable**	**Grass**	**Arable**	**Grass**
N	84	96	102	154	26 ± 24^{2}	82 ± 7^{67}
P	25	20	15	17	-6^{1}	3 ± 1^{56}
K	46	24	33	32	57^{1}	10 ± 2^{58}

Nevertheless, the general conclusion that can be drawn from the literature is that nutrient surpluses are smaller for organic than conventional farms, when comparing the same farm types. Further support is provided by Watson & Younie (1995) who compared pairs of conventionally and organically managed beef units and found greater N surpluses on the conventional, both when expressed on a unit area and unit stock basis.

This has important implications for the environmental effects of organic farming. Smaller nutrient surpluses will impact on N and P losses from these systems, and this is discussed in more detail later.

2. COMPARING SYSTEMS

Before assessing the impact of organic farming on the environment, two issues need to be addressed.

2.1. How to judge sustainability

Whereas most would sign up to 'sustainable agriculture', there would probably be many disagreements in the detail of what constitutes sustainable farming. Rigby *et al.* (2001) tried to develop an Indicator of Sustainable Agricultural Practice (ISAP) by scoring five aspects of farm production for cropping systems:

- Seed source
- Soil fertility
- Pest control
- Weed control
- Crop management

The challenge is allocating correctly weighted scores to each attribute. If a single threshold value is used to assess sustainability, it is possible for the system to score poorly in one or more sectors but still achieve 'sustainability' if scores in other sectors are high enough to compensate. This would not be truly sustainable. Also, because the indicator focused predominantly on horticultural systems, no assessment of livestock management is included.

Whereas the attempt to develop such an objective system is laudable, the complexity of defining what constitutes 'sustainable farming' means that our report has opted for a qualitative or semi-quantitative assessment of key indicators of environmental impact. Several reviews of the environmental impact of organic farming have recently been completed (Stolze *et al.*, 2000; Condron *et al.*, 2000; Hansen *et al.*, 2001; Stockdale *et al.*, 2001). All have generally used the same indicators.

Hansen *et al.* (2001) used the driving force – state – response (DSR) framework to structure the choice of indicators. Table 2.1 shows the same approach for our choice of indicators, focusing on State and Driving Force, but not Response (consumers, farmers and authorities).

Table 2.1. Choice of key indicators of environmental impact.

	Category	Indicators
State	Ecosystem	Biodiversity
	Soil Quality	Organic matter content; Biology; Structure; Erosion susceptibility.
	Water Quality	Nitrate leaching; Phosphorus loss; Pesticides; Human pathogens.
	Air Quality	Ammonia; Nitrous oxide; Methane; Carbon dioxide.
Driving forces	Input/output (Resource use)	Energy efficiency; Nutrient balance; Controlled wastes.

2.2. Comparing systems

Comparing conventional and organic agricultural systems is not straightforward:

- **Basis of comparison:** Arable and horticultural crop yields from organic systems tend to be less than in conventional systems. Organic yields have been reported to be, on average, 50-95% of the conventional yield, depending on species and position in the organic rotation (Watson *et al*., 2002a). Therefore, one issue is how to take account of the lower yield potential of organic systems when assessing environmental impact. For example, should environmental impact be measured per unit of land area, per unit of economic activity or per unit of produce?
- **Type of farms compared:** Most trials have compared lowland mixed crop and livestock organic farms with similarly structured conventional farms, as this review demonstrates. Therefore, this would not include comparisons of organic farming systems with the most intensive conventional farms, which is perhaps a comparison that should be made. There are also few comparisons between organic and conventional extensive farms (i.e. upland grass-based livestock systems).
- **Lack of clear definition of what is meant by 'conventional' agriculture.** Whereas organic agriculture is defined in EU and Sector Body standards, there is no similar definition for what is meant by conventional agriculture, and practices in both systems will change over time, especially in relation to market signals.

We have tried to address these difficulties in the synthesis of the existing information (see Discussion and Conclusions).

ASSESSMENT OF EFFECTS

3.1. Biodiversity

3.1.1. Introduction

Biodiversity can be divided into three components:

- diversity between and within ecosystems and habitats (habitat diversity)
- diversity of species (species diversity)
- genetic variation within individual species (genetic diversity)

Biodiversity is an insurance for the future. It provides the variability on which every species relies to help to adapt to change. For this reason alone, it is important to maintain, or improve biodiversity. However, the declining state of Europe's diversity is well documented, with 64 endemic plants extinct and 38% of bird species threatened across Europe (Anon., 2003c). In addition, some aspects of biodiversity provide some of the most visually attractive features of the landscape (flora, birds and arthropods). These aesthetic effects are important to the general public and are underpinned by key quality of life indicators such as number of farmland birds. There is a requirement under organic management to protect and enhance biological processes and wildlife habitats. Some Organic Certification Bodies have worked with English Nature towards the development and inclusion of specific conservation objectives within the organic production standards (Anon., 2002f).

Maintaining and enhancing biodiversity is considered central to developing a sustainable organic system. As well as protecting and enhancing biodiversity *per se*, increased biodiversity plays a functional role by improving nutrient cycling, pest control and disease control in the production system.

Biodiversity needs to be considered at all levels:

- Soil biomass, including bacteria and fungi (including mycorrhizae)
- Earthworms
- Arthropods
- Birds and other animals
- Flora

The Organisation for Economic Co-operation and Development (OECD) divides the indicator category of 'ecosystem' into four component parts: floral diversity, faunal diversity, habitat diversity and landscape (Anon., 1997a). The assessment of species diversity can also take place at three levels, diversity within a species (generic level), changes in the number of species and their populations (species level) and changes in habitats (ecosystems level). The OECD recommends that biodiversity be measured in terms of domestic and wild species, thereby assessing the widest possible genetic resource pool.

Whereas the soil biomass and earthworms are important for nutrient cycling and 'soil health', these are dealt with in more detail in Section 3.2.4. Here we focus mainly on other aspects of biodiversity.

3.1.2. The farming system.

Many aspects of organic farming will favour increased biodiversity:

- Organic standards require the sympathetic management of wildlife-rich infrastructure features, such as hedges, and ditches. These features also play a role for the organic farmer, providing reservoirs for the predators of crop pests as part of the integrated pest control strategies practiced on organic farms.
- A higher proportion of organic lowland farms are in mixed farming.
- Use of synthetic fertilisers, agrochemicals and veterinary medicines is prohibited or much restricted, which removes direct and indirect problems for wildlife.
- There is a greater variety of crop structure because of more spring cropping in more varied rotations.
- Organic farms use more undersowing, such as with stubble turnips with the land then used for autumn grazing. This can produce attractive over-winter habitat for seed eating birds and helps boost populations of some farmland invertebrates.
- Stocking densities are limited by productive capacity underpinned by the Organic Standards and so tend to be less in organic systems. The lower density can be an advantage when grazing sensitive habitats. A wider range of species of livestock are more often maintained on organic farms. This helps to control parasite burdens and has advantages in maintaining structurally diverse swards.

Stolze *et al*. (2000) undertook a thorough review of the effects of organic farming on the ecosystem and concluded that organic farming clearly performed better than conventional farming in respect of floral and faunal diversity, and that organic farming had greater potential to deliver wildlife conservation and landscape effects.

Several reviews have addressed the impact of organic farming on biodiversity of the whole system under UK conditions (Unwin *et al*., 1995; Younie & Baars, 1997; Gardner & Brown, 1998; Anon., 2000a). Other numerous studies have investigated the impact on biodiversity within components of the farming system, e.g. the farmed area that has been broken down into crop production and protection, livestock production and protection, post crop and rotational factors. These studies have included many short- to medium-term studies to evaluate farming systems impacts on biodiversity, for example (Feber *et al*., 1997, Fuller *et al*., 2000) but there are relatively few long-term comprehensive research projects on environmental benefits and impacts on the whole system.

3.1.3. Impact of system components on biodiversity

Gardner & Brown (1998) reviewed the effects of common agricultural practices from conventional, integrated and organic farming systems on biodiversity (Table 3.1). Organic farming performed best in all four aspects of agricultural management.

In the House of Commons Select Committee on Agriculture Report (Anon., 2001c), it was surmised that biodiversity studies might underestimate the benefits of organic farms for three main reasons:
There would have been a tendency to match organic farms, which have tended to be relatively small, with similar sized conventional farms. Consequently, the larger intensively managed farms, which usually support the lowest populations of wildlife, may not have been represented in the studies.

- In some studies, recently converted farms were selected due to the shortage of organic farms. As it is possible that wildlife populations build up over the years from the time a farm begins conversion, the results may not be representative of a fully established organic farm.

- Wildlife populations are likely to increase on organic farms when organic systems become more established as part of the landscape, as opposed to the current situation whereby most exist in isolation, surrounded by conventional farms.

Table 3.1. An assessment of the impacts of farming operations with farming systems on biodiversity of soil organisms, plants invertebrates, birds and mammals. The higher the score, the more beneficial the impact. From Gardner & Brown (1998).

Agricultural Practice	Conventional Arable	Conventional Mixed Lowland	LEAF	Organic
Cultivation	-1.5	-1.5	-1.5	1.5
Production	-2	-1	-1	+4
Protection	-6	-6	-6	-0.5
Post Cropping	+4.5	+9.5	+9.5	+11.5
OVERALL	**-5**	**+1**	**+1**	**+13.5**

Pesticide use

The use of synthetic pesticides in conventional farming has been one of the most significant impacts on wild flora and fauna (Unwin *et al.*, 1995). Organic regulations do not allow the use of synthetic pesticides. Only a small number of natural pesticides are permitted, and then only as a last course of action.

The potential effects of pesticide use include both direct and indirect effects:

- Herbicides can virtually eliminate broad-leaved weeds from the cropped area. Some of these weeds are desirable on aesthetic grounds and the seeds of some are important food sources for some farmland bird species.
- Accidental poisoning of non-target animals.
- Risk to beneficial insects after application of pesticides.
- Negative effects on soil organisms.

The use of chemicals for parasite control in conventional livestock production tends to be routine rather than by need as under organic standards. Some antihelminthic products have been shown to have adverse effects on dung dwelling invertebrates, resulting in a reduction in number and variety of dung insects that are important food sources for insectivorous birds and animals (Strong, 1992). Consequently, some Certifying Bodies do not permit the use of the most detrimental compounds.

Cropping diversity

The trend towards increasing specialisation of crops and near continuous cropping of cereals on many farms, aided by the development and use of pesticides and inorganic fertilisers, on conventional farms has led to the polarisation of regional cropping patterns: grass predominates in the north and west, cereals in the south and east (Unwin *et al.*, 1995).

In contrast, organic rotations are more diverse. On average, organic farms were growing 4.5 different crop types compared with 3.4 on integrated farms and organic farms are also likely to grow a greater number of perennial crops than their conventional counterparts (Stolze *et al*., 2000). This observation is also supported by Table 1.4, which shows cropping before and after conversion under Defra's Organic Farming Scheme (Anon., 2002e). These data show a move from winter to spring cereals, substantial decreases in the areas of rape, sugar beet and fodder maize, substantial increases in the areas of vegetables, legumes (market and forage, set-aside and temporary grassland).

This wider variety of crops on a farm provides greater structural diversity, habitat diversity and, therefore, should lead to a greater diversity of wild flora and fauna (Unwin *et al*., 1995). The alternative approaches to pest and disease management include the use of inter-cropping and under sowing. These can have beneficial effects on within-crop biodiversity (Altieri & Letourneau, 1982; Armstrong & McKinlay, 1996), although there is evidence that some species abundance can be reduced, probably due to species habitat preference (den Boer, 1977; Gardner, 1991; Armstrong & McKinlay, 1996; Gardner *et al*., 1997).

The importance of short-term grass leys is the contribution they make to diversifying the arable rotation. The fertilisation regime is the most significant difference between organic and conventional grassland, with organic grassland relying on biologically fixed N rather than N fertiliser. Increased sward diversity has been reported within organic grassland (Haggar & Padel, 1996; Younie & Armstrong, 1996). Organic short-term leys may have a greater species diversity than a comparative conventional ley. Lampkin *et al*. (2002) recommend at least four grass species and up to four legume species for a short-term ley, for a longer term grazing ley the list of recommended species rises to sixteen and in addition to grasses and legumes includes a variety of sown herbs e.g. chicory, plantains and yarrow. Cotswold Seeds Ltd, who specialise in the supply of forage seed mixtures to the organic sector, confirm that a wide range of varieties are generally used in seed mixes (Anon., 2003d).

Green manure crops or fertility building crops are important because they provide over-winter ground cover, offering a range of niches for botanical and invertebrate species. They also provide a different structure of cropping from cereal crops, which may prove beneficial to invertebrates (Armstrong & Younie, 1996). These cover crops may also take the form of weedy stubbles that are extremely important food sources for seed-eating birds (Anon., 2001d).

Furthermore, uncropped areas (sown grass strips or 'beetle banks', grass margins, uncropped wildlife and flower strips, hedges, ditch and bank habitats) are intrinsic in organic regimes where their management is central to the philosophy (Stockdale *et al*., 2001). This was demonstrated in a limited study of 15 farms funded by English Nature (Anon., 2003e). Whole-farm conservation plans were drawn up to establish where and how organic farming practices were contributing to improvements in biodiversity. This work was done in conjunction with development of the Soil Association's conservation standards.

Gardner & Brown (1998) concluded that the nature and extent of these habitats are the key to determining the overall biodiversity of the agricultural areas, because it is these non-cropped areas that are the reservoirs for faunal and floral diversity

Permanent pasture, together with natural and semi-natural grassland, accounts for 80% of organic registered land. Permanent pasture is particularly important for its potential to provide stable and less disturbed environments, providing a refuge for biodiversity. Organic management offers environmental protection in a number of ways: reduced nutrient inputs, less intensive grazing, avoidance of herbicides and later cutting dates for mown swards. Several organic/conventional studies of permanent grassland have shown that the organic swards contain a greater number of plant species (Frieben & Köpke, 1996).

Conventional management of marginal and upland permanent pasture differs very little from organic management of similar situations in terms of use of inputs, e.g. fertiliser or pesticides. Even so, within the conventional system, the stocking rates have been maintained at artificially high levels due to the importation of feed, and this has lead to a decline in the biodiversity value of conventionally managed upland permanent pasture in comparison with the mixed stocking and lower stocking rates required under organic management (Hopkins & Hrabe, 2001).

Cultivation

Generally, there are some indications that inversion ploughing and deep tillage reduces the numbers of invertebrates (Mäder *et al*., 1996a; Fuller, 1997), particularly earthworms (Edwards & Lofty, 1982a; Scullion *et al*., 2002) and collembola and some oribatid mites (Wallwork, 1970). However, it may encourage small mammals (Brown, 1997). Both conventional and organic farming use inversion ploughing, though there is more scope for adopting minimal tillage regimes on some soil-types under conventional farming, where soil conditions are suitable and weed control can be achieved by herbicide use.

Currently there are no formal guidelines for mechanical weed control in organic systems. A recent review of inter-row hoeing by Welsh *et al.* (2002) has suggested that weeding operations should be conducted at an early stage in the growing season just as the weeds emerge and there is little benefit to weeding on more than two occasions. Mechanical weed control can have a negative impact on ground nesting birds (Jones *et al.*, 1996; Fuller, 1997), but this will depend on the timing and method of control (Welsh *et al.*, 2002). For example, weeding in winter-sown cereals should be completed before skylarks begin to nest, or at least in time to allow relaying. Inter-row hoeing may be less detrimental than spring-tine weeding or harrowing since less of the soil surface area is cultivated and fewer passes are required to achieve good levels of control. In addition, the use of wide crop row spacing sometimes seen in organic systems, which is required for inter-row hoeing in cereals, may in itself encourage ground nesting birds into the crop (Welsh *et al*., 2002).

3.1.4. Floral Diversity

Crop rotation exerts a considerable influence on biodiversity. The proportion of grassland to arable cropping, the variation in sowing dates for cereal crops and the inclusion of both autumn and spring sown cereals are all key components of the organic system that contribute to the richness in biodiversity.

Studies on wild flora demonstrate that greater species diversity occurs within the crop (Cosser *et al*., 1997), at the crop margins (Hopkins & Feber, 1997) and in the non-farmed areas (Frieben & Kopke, 1996) on organic farms. This increase within-crop can result in six times more species than on conventional farms (Rasmussen & Haas, 1984; Vereijken, 1985). These then become vital food sources for invertebrates, birds and small mammals.

In terms of endangered species (rare arable plants), a number of studies have found 50-80% of one or more endangered species on organic farms in comparison to 15-30% on conventional farms (Cobb *et al.*, 1998; Kay & Gregory, 1998; Kay & Gregory 1999). The occurrence of these rare arable plants can be attributed to a number of management factors including the restriction on the use of herbicides, and the avoidance of soluble fertilisers.

It is not only the farmed area that can be influenced by management strategies. Field margins and hedgerows on organic farms tend to have greater abundance and diversity than the equivalent areas on conventional farms (Critchley, 1994; Stopes *et al*., 1996; Hopkins, 1997; Hopkins & Feber, 1997). Frieben & Kopke (1996) characterised the benefits provided by uncropped habitats:

- Refuges for endangered plant species
- Areas of floral diversity

The greater floral diversity has an impact on the faunal diversity.

- Over-wintering sites for invertebrates and vertebrates
- Refuges for species after harvest
- Areas with network links to other habitats.

3.1.5. Faunal Diversity

There is anecdotal evidence that organic farming systems are more likely to use rare, native or traditional breeds, but there are few studies investigating the role of organic livestock in maintaining the genetic diversity of domesticated stock (Bremond, 2002). However, there are numerous studies on wild faunal diversity comparing different farming systems.

All the indicator groups studied, including Arachneae, Carabidae, Formicidae, Isopoda and Diplopoda, have been found to have generally higher or at least similar species numbers as on conventional systems. The DOC (bio-**d**ynamic, **o**rganic and **c**onventional) experiment reported by Pfiffner & Niggli (1996) showed higher diversity and abundance on organic plots (90% greater) than in the conventional plots.

A number of studies have documented either greater diversity of species or greater numbers of a specific species of beetles, parasitic flies and wasps, spiders and millipedes within organic farming systems (after Stolze *et al*., 2000). However, some reports have not found these differences to be so clear (Gardner & Brown, 1998). Feber (1998) has reported significantly more butterflies and more species of butterflies in organic fields and in the uncropped boundary on organic fields than on conventional sites.

Ongoing work is investigating the factors influencing biodiversity (plants, invertebrates and bird species) within organic and conventional systems of arable farming (Norton, 2002). In contrast to other studies, preliminary results show that there are significant differences in the number of species found in arable margins and within fields between organic and conventional farms, but there was only a marginal difference in species number in the non-cropped habitat.

Birds

The decline in farmland bird populations is well documented (e.g. Fig. 3.1) and is of concern to NGOs and Government alike. Reversing this decline is a priority and a key quality of life indicator.

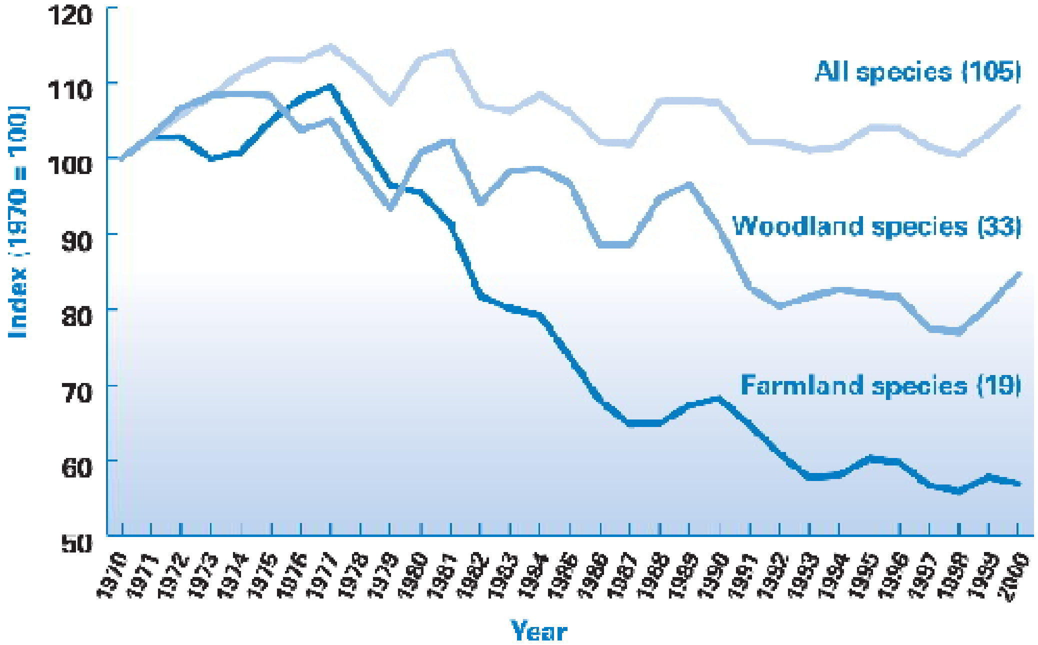

Figure 3.1. Trends in bird numbers.
Source: www.rspb.org.uk/science/survey/quality_of_life_indicators.asp

There have been a number of research and monitoring projects evaluating the impact organic farming systems have on bird populations. A number of British Trust for Ornithology (BTO) studies have shown higher densities of bird populations on organic farms. The Fuller *et al.* (1995) study on habitat selection and breeding success in Skylarks on organic and conventional farmland concluded that organic farming systems derived benefits from the 'whole system' rather than just from the non-cropped areas alone.

Subsequently, BTO has jointly undertaken a number of research projects to evaluate the effect of organic farming systems on breeding and wintering bird populations (Fuller, 1995). These constituted a comprehensive study that dealt with comparison of bird populations on organic and conventional farms, an intensive study on biology of skylarks, food resources, and habitat selection. The conclusion was that there were higher densities of birds on organic farms than on conventional comparisons, and that this was especially true during the winter. Fuller (1995) concluded that these differences could not be accounted for by non-cropped habitat or cropping patterns alone. Food resources were found to be more abundant on organic farms: this included both plant and invertebrate food sources. One study tried to eliminate habitat effects by pairing conventional and organic farms according to cropping, hedge density etc. Despite this, the study still found positive benefits of organic farming, though potentially not as large as real differences between the two farming systems, where improved habitat on organic farms will be a feature.

As well as birds, recent research into species richness on organic and conventional farms (Wickramasinghe *et al*., in press) has shown total bat activity was significantly higher on organic farms than on conventional farms. This study concluded that these differences were driven by a number of factors including taller hedgerows, better water quality and greater prey availability.

In conclusion, drawing upon evidence and data from a number of comparative reviews, organic farming systems have demonstrated that there is a positive benefit to wildlife conservation on organic farms.

3.1.6. Habitat Diversity

A habitat is defined as a place where organisms of a species are found. The OECD in 1997 agreed a measure of habitat diversity using the following indicators:

- Changes in selected large-scale areas (woodlands, wetlands and semi-natural or natural grassland).
- Fragmentation in agro-ecosystems and natural habitats.
- Length of the contact between different types of habitat feature.

There is, to date, little information available to analyse habitat diversity within farming systems. However, the requirement of organic standards to have both fertility building and cash cropping results in habitat diversity, so providing suitable conditions for some species that require different environments at different times of the year, e.g. lapwings nesting in grassland but requiring arable crops for food sources. There is also some (limited) evidence that organic farming systems positively enhance the habitat diversity. Baumgartner & Imhoff, (2002) reported upon a development programme that attempts to integrate ecologically sound and economically viable food production into a landscape that can accommodate a full range of native species and evolutionary processes.

3.1.7. Landscape

The impact of farming system is, by its very nature, subjective. There is very little information available relating the effect of organic farming in the UK. A previous study, (Anon., 1995) failed to locate any significant work in this area. Unwin *et al.* (1995) reported on a study that considered the visual impact of the farming system with landscape and farm context. The study comprised 48 farms providing a mix of conventional, short and long term organic farms; the sample included upland, mixed lowland farms and horticultural units. The sample size was too small to produce statistically significant results. However, with some reservations, the study concluded that overall, organic farmers did provide net benefits to the landscape, largely due to their general environmental awareness.

The pattern of conversion in the lowlands has resulted in organic farms representing ‘island communities’ rather than integrated landscape features. However, this may not be the case in some areas of upland and moorland where, as a reflection of the policy for conversion payments, this may have encouraged a greater number of farms to have converted large areas to organic management (Section 1.4; Anon., 2002c).

The assessment of landscape in terms of an area’s visual character is likely to become increasingly important as public money is used to support delivery of ‘social goods’. The assessment will need to produce an inventory of physical landscape features, be they natural, historical or cultural and the second indicator needs to establish a monetary value for a given landscape. How an organic farm can contribute to the landscape is an area of interest for further indicator development.

3.2. Soil Quality

3.2.1. Introduction

Soil 'quality', 'health' and 'fertility' are all terms used to describe the status of the soil, often interchangeably. Soil fertility could perhaps be considered to be a measure of the soil's ability to sustain satisfactory crop growth both in the short and longer-term. However, Stockdale *et al.* (2002) argue that soil quality is a wider concept than this because it encompasses attributes relating to protecting the soil as a resource. Soil fertility is determined by a set of interactions between the physical and chemical environments of the system and by biological activity. Organic matter is linked intrinsically to soil fertility, because it is important in maintaining good soil physical conditions (e.g. soil structure, aeration and water holding capacity), which contribute to soil fertility, and it is an important nutrient reserve. Stolze *et al.* (2000), in their review of the environmental effects of organic farming, concur with the view that soil organic matter, biological activity and soil structure are all important aspects of soil quality (chemical status not specifically mentioned), but also include susceptibility to soil erosion. We therefore review impacts on soil quality in terms of:

- Soil organic matter (SOM) content
- Soil structure
- Biological activity
- Soil erosion risk

3.2.2. Soil organic matter

Factors affecting soil organic matter content

Soil type (texture and drainage status), long-term cropping or other history (i.e. return of crop residues), topography and climate affect the SOM content of all soils. Under most circumstances, total SOM levels change slowly (Johnston *et al.*, 1989; Fig. 3.2). In terms of total amounts, soils under long-term grassland generally contains more than under long-term arable. Cultivation causes oxidation of SOM so that levels decline compared with undisturbed soils, but the rate of change will be determined by factors that influence the balance between residue return and rate of oxidation (Johnston, 1986). Here, not only is cultivation frequency important but soil texture also plays a role because light textured soils offer less protection for SOM and hence mineralise more rapidly. Figure 3.2 shows the slow change in soil organic matter and, also the increase when inputs of organic matter (in this case, as manure) exceed the rates of oxidation.

Organic farms maintain SOM levels by several methods (Hodges, 1991):

- mixed farming systems
- crop rotation (e.g. ley/arable)
- recycling manures
- green manures
- importing fertility (e.g. importing manures and composts)

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Increases in SOM arise when C inputs (crop residues, manures, etc) exceed the rates of oxidation. It should be noted that:

- Fertiliser increases SOM relative to unfertilised soils under similar cropping because it produces greater crop yields and residue returns (Johnston, 1986).
- Regular organic additions (manures, long-term grass) have the largest effects on SOM (Johnston, 1986; Khaleel *et al.*, 1981).

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- The effects of organic matter addition on soil organic matter content are more noticeable on light textured soils.

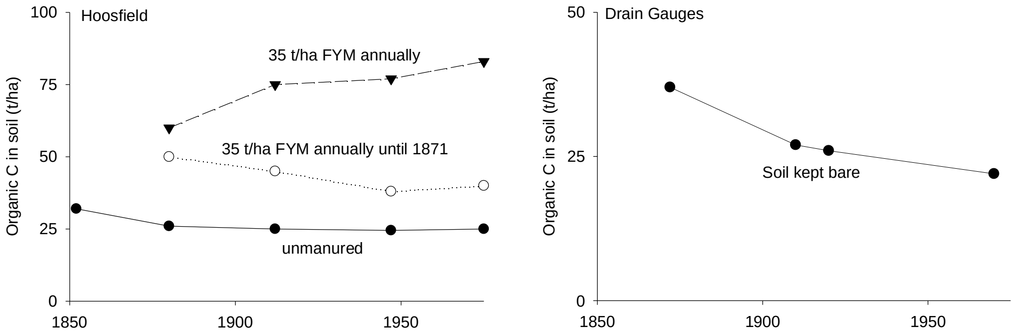

Figure 3.2. Effect of organic inputs on changes in soil organic carbon content with time, 0-23 cm. Redrawn from Jenkinson & Rayner (1977). Hoosfield was under continuous barley, Drain Guages was kept fallow (and undisturbed).

Soil organic matter content under organic farming

Raupp (1995a) reported results from a long-term plot experiment that demonstrated SOM differences between systems in the order conventional (0.79% C) < organic (0.92% C) < biodynamic (1.02% C) after *c.* 10 years of treatments. The differences were clearly linked to differences in organic matter input. Armstrong-Brown *et al.* (1995) also demonstrated this relationship with inputs. A paired comparison was made between organic and conventional farms in the UK. Organic horticultural and arable farms had more SOM than their conventional counterparts, which was related to greater manure inputs under organic. However, it was not possible to differentiate between organic and conventional pasture.

Many others have reported increased SOM under organic and/or biodynamic farming compared with conventional systems (Goldstein & Young, 1987; Garcia *et al.*, 1989; Clark *et al.*, 1998; Mäder *et al.*, 1993; Mäder *et al.*, 1995; Petersen *et al.*, 1997).

These results are not surprising: the organic systems generally had a greater return of organic matter (as manures), so that SOM levels would be larger than in their conventionally fertilised (and/or less frequently manured) conventional counterparts. However, the pasture soils described by Armstrong-Brown *et al.* (1995) did not differ in SOM, presumably because organic matter inputs were similar in both organic and conventionally managed pastures. The importance of actual organic matter inputs to the soil in influencing SOM contents is illustrated by the study of Amman (1989), where no differences between organic and conventional in SOM levels were noted. Stolze *et al.* (2000) interpreted this as relating to the lower stocking densities used in organic systems.

3.2.3. Soil structure

Defining 'good' soil structure

In agronomic terms, a 'good' soil structure is one that shows the following attributes:

- Optimal soil strength and aggregate stability, which offer resistance to structural degradation (capping/crusting, slaking and erosion, for example).
- Optimal bulk density, which aids root development and contributes to other soil physical parameters such as water and air movement within the soil.
- Optimal water holding capacity and rate of water infiltration.

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To a large extent, the inherent soil texture will influence soil structure. Soils consisting mostly of sand cannot form structural units and even under good management the soil's consistence remains loose. The structure in soils composed of clay can vary widely but these soils are more capable of forming fine granular structures because of their ability to shrink, swell and fracture during drying and wetting. Predominantly silty or fine sandy soils are the least stable. Soil structure should be described in terms of grade or degree of structure, shape and size of aggregates, and stability of the aggregates. SOM has been shown consistently to have a large influence on soil physical properties within textural groups (Haynes *et al*., 1991). Thus, SOM and management are the primary factors affecting soil structure, within these limits.

Effect of SOM on soil physical attributes

The stability and long residence time of the humus component of SOM in soil means that it plays an important role in structure. SOM strongly influences many soil properties including bulk density, water holding capacity, infiltration rate, hydraulic conductivity and aggregate stability.

Aggregate stability is a key property in relation to the development and maintenance of soil structure. Tisdall & Oades (1982) concluded that it is particularly the younger SOM (with a larger content of polysaccharides, roots and fungal hyphae) that is important for developing aggregate stability. Fungal hyphae (the biological agent) and extracellular polysaccharides (the chemical agent) link particles together to provide aggregate stability (Haynes & Naidu, 1998). Exudates are released by growing roots and rhizosphere microflora (Haynes *et al*., 1991). The simpler polysaccharides act strongly for 2-3 weeks, but decline over the following 4-6 months; cellulose achieves maximum effect after 6-9 months (but is not as effective as polysaccharides); ryegrass residues increase in effect up to 3 months, persist for 4-6 months and decline thereafter (Tisdall & Oades, 1982). Therefore, the most important SOM components exert their effect for at most a year, which matches the observations that aggregate stability is greatest under grass (continued production of these components) and decreases rapidly under arable cultivation (Loveland & Webb, 2003). This also explains why aggregate stability can change over the short-term (e.g. after ploughing a ley), although the total SOM is hardly affected (Haynes & Swift, 1990). Shepherd *et al*. (2002a) argue, therefore, that optimal aggregate stability requires the frequent turnover of young organic matter residues. Thus, a 'biologically active' soil is better predisposed to aggregate stability.

Other factors affecting soil structure

Whereas increasing SOM content can contribute to good soil structure, much still relies on good management decisions by the grower. This human influence should not be forgotten. Maintenance of good structure relies on timely cultivations, i.e. the correct type of cultivation, using appropriate equipment and when the soil is at the correct moisture content. Travelling and/or cultivating when too wet can destroy soil structure. The risk of 'poaching' (compaction) of the surface of grassland soils in wet conditions by livestock also needs to be minimised.

Soil structure under organic farming

Stockdale *et al*. (2001) reported evidence of increased aggregate stability under organic farming (Jordahl *et al*., 1993; Gerhardt, 1997; Siegrist *et al*., 1998). However, Stolze *et al*. (2000) reported that others have found no consistent differences in aggregate stability, and that measurable differences have not been found in other parameters. However, physical assessments are notoriously variable within the field. One approach to overcome this is to adopt a simple scoring system, based on a visual assessment of the soil. This seems to often better represent the soil structure than do detailed measurements of individual physical properties. Shepherd *et al*. (2002a) adopted this approach and concluded that soil structure was at least as good under organic as under conventional management. However, rotational position also strongly influenced soil structure, with structure being considerably better immediately after ploughing the leys, as would be expected from the preceding discussions

Reganold (1995) showed highly significant differences in soil structure score when 16 fields of biodynamic or conventional commercial farms were compared in a paired study in New Zealand. There were also highly significant differences in total topsoil C and a range of physical parameters (e.g. reduced bulk density and penetration resistance and increased topsoil depth under organic and/or biodynamic farming). Reganold (1988) undertook a similar paired study on a conventional and organic farm in the USA and again found improved physical properties under the organic system. Mytton *et al*. (1993) suggested that white clover (central to many organic rotations) was more effective than ryegrass in developing soil structure. Research is continuing. Mäder *et al*. (2002) found that soil aggregate stability was 10-60% higher on organic than on conventional plots in the long-term DOC trial in Basle, Switzerland. There were also positive correlations between aggregate stability and microbial biomass and between aggregate stability and earthworm biomass.

Interestingly, Alföldi *et al*. (1995a) reported earlier results and stated that, after fourteen years, crop production systems did not show any influence on the volume of total or large-sized pores, bulk density or aggregate stability. Others also have not found such differences between conventional and organic management. Raupp (1995b) investigated a long-term experiment (1958-1990) but found no clear differences in soil structure. Droogers & Bouma (1996) reported that soil structural differences were relatively small between biodynamic and conventional farms. Gardner & Clancy (1996) found general trends in apparently improved structure on organic farms but differences in parameters were rarely statistically significant.

3.2.4. Soil biological activity

Role and composition

The role of soil organisms is central to soil processes. The soil hosts complex interactions between vast numbers of organisms, with each functional group playing an important role: from the macrofauna (e.g. earthworms) responsible for initial incorporation and breakdown of fresh residues through to the bacteria with specific roles in mobilising nutrients.

Maintaining a diverse population of soil flora and fauna should theoretically offer advantages in terms of aiding soil processes. However, this is difficult to demonstrate conclusively. An additional advantage of maintaining soil biodiversity is the potential for protection against plant damaging organisms. Soil fauna are recognised as potential suppressants of root pathogens. Some species of fungivorous amoeba, nematodes, *Acari* (mites) and *collembola* (springtails) can selectively feed on phytopathogenic fungi.

Earthworms

Earthworms have many direct and indirect effects on soil quality, both in terms of their effects on soil physical properties (e.g. porosity) and nutrient cycling through their effects on microfloral and faunal populations (density, diversity, activity and community structure). These effects are complex, though many of the resultant effects are beneficial:

- reduction of plant parasitic nematodes and pathogenic fungi
- increased enzymatic activities
- increased nutrient release
- spread of biocontrol agents
- spread of mycorrhiza and *Rhizobium* species

Thus, although micro-organisms predominantly drive nutrient cycling, earthworms play a key role in soil organic matter turnover. Factors that reduce their abundance, be it natural environmental factors (e.g. soil drying) or management factors (e.g. cultivation, biocides), will therefore also affect organic matter turnover.

There is no straightforward relationship between soil management and earthworm populations because there tends to be an interaction between several factors. For example, whereas there have been some reports of fertilisers reducing worm populations, Edwards & Lofty (1982b) found larger populations with inorganic N than without: this was attributed to greater production of crop residues and roots, with the additional organic matter encouraging worms. Another example of the complexity of factors is that white clover has been found to inhibit worm activity (Lampkin, 1992) but, overall, organic rotations tend to favour earthworms because of the other beneficial effects of management: organic matter additions, leys, no biocides, etc. Mohamed Abdalla *et al.* (1995) studied the effects of pesticides on worms and found that the toxic effects could be ranked in the order of insecticides > herbicides > fungicides. Ramesh *et al.* (1997) linked low populations of earthworms to lack of adequate moisture in the soil surface, intensive pesticide use, frequent tillage, and absence of ground cover.

Siegrist *et al.* (1998) compared earthworm populations in a long-term field trial comparing organic and conventional land management; earthworm biomass and density, and population diversity were significantly greater on organic than conventional plots. Gerhardt (1997) compared organic and conventional farms, and found greater earthworm abundance and activity on the organic farms. Whalen *et al.* (1998) found earthworm numbers and biomass greater on organic manure treated plots than inorganically fertilised plots, though populations declined on both treatments during 5 years of continuous cereal production. The results of Scullion *et al.* (2002) were less conclusive, with fewer differences between organic and conventional. However, site selection was such that both organic and conventional rotations included grass leys at the same frequency.

Arable soils usually contain a smaller biomass of earthworms than pasture soils, unless the soil is given regular applications of FYM (Newman, 1988). It seems, therefore, that cultivation in some way reduces earthworm populations. Larger populations under direct drilled crops (Edwards, 1983) suggest that the physical act of ploughing reduces the population. Thus, because organic rotations tend to plough less frequently (because of the fertility building stages) this is likely to be an advantage for earthworm populations.

However, conversely, there is less scope for reduced cultivation systems in organic farming (as previously discussed), which would work against earthworm populations.

Soil microbial biomass

The soil microbial biomass (the living part of the soil organic matter excluding plant roots and fauna larger than amoeba) performs at least 4 critical functions in soil and the environment:

- a labile source of carbon (C), N, P, and sulphur (S);
- an immediate sink of C, N, P and S;
- nutrient transformation;
- pesticide degradation.

In addition, micro-organisms form symbiotic associations with roots, act as biological agents against plant pathogens, contribute towards soil aggregation and participate in soil formation.

The relative importance of various environmental variables in governing the composition of microbial communities could be ranked in the order: soil type > time > specific farming operation (e.g., cover crop incorporation or application of mineral fertiliser) > management system > spatial variation in the field. The fungal:microbial biomass ratio also changes with farming system. Fungi tend to dominate in self-regulating ecosystems that do not receive fertiliser inputs.

Negative effects of pesticides on various micro-organisms have been demonstrated (e.g. Selim *et al*., 1970; van Schreven *et al*., 1970; Banerjee & Dey, 1992; Taiwo & Oso, 1997; Martineztoledo *et al*., 1998; Yardim & Edwards, 1998; Welp & Brummer, 1999). However, it must also be said some others have failed to find significant effects (e.g. Martyniuk & Wagner, 1978; Hicks *et al*., 1989; Tu, 1992; Hart & Brookes, 1996; Biederbeck *et al*., 1997). Soil properties also influence effects by determining the degree of sorption and the speciation of toxicants in the liquid phase. Thus, soils can either buffer high loads of toxicants or can be very sensitive toward contamination. However, generally, it can be concluded that pesticides affect the population of micro-organisms. Also, fungicides tend to inhibit or kill soil fungi, including mycorrhizae (Johnston & Pfleger, 1992; Scullion *et al*., 1998), which are particularly important in organic systems (see later).

Nitrogen fertilisation, manure and tillage can all influence microbial activity. Often, N fertiliser increases activity because of a greater return of organic N and C in crop residues. However, most comparisons have been made with unfertilised or poorly fertilised crops, rather than with organic systems. There have been some suggestions that water-soluble fertilisers are harmful to the soil microbial biomass through their salt effects (by inducing osmotic stress etc.). For example, sulphate of potash (K_2SO_4) is considered significantly less toxic than muriate of potash (KCl), though the latter is the most common potassium fertiliser used in conventional agriculture and which supposedly can have serious detrimental effects on soil micro-organisms. However, the literature does not implicate salt effects and we suggest that evidence for this is generally scant and/or anecdotal.

The evidence for increased microbiological activity under organic farming is mixed. This was also the general conclusion reached by Stolze *et al.* (2000). However, they suggested that it might be as long as 10 years under organic conversion before any differences in microbiology might be observed, supported by the work of Peeters & van Bol (1993). Elmholt (1996) also demonstrated the age effect by showing that the abundance of the mainly soil-borne penicillia was significantly higher at the ‘oldest’ organically cultivated farm in the study than at the other localities. Biological activities of 21 agricultural soils in Rheinland-Pfalz, Germany managed organically for 2 to 56 years, were monitored by Schulte (1997). The general conclusion was that soil biological activity was greater in soil managed organically in the long-term compared with soils managed organically for shorter periods.

As with measuring soil physical properties, there are also methodological issues when testing for differences between farming systems. For example, Elmholt (1996) demonstrated the importance of crop type by measuring significantly higher microbial activity in the ley soils than in the wheat soils. This might be expected, due to differences (again) in organic matter inputs. Ritz *et al.* (1997) found that the effects upon microbial activity of sampling and adjusting the moisture status were as great as the addition of the manures. There are also the complications of spatial and temporal variability in populations.

Positive effects on microbiology have been reported by many workers:

Raupp (1995c) summarised the conclusions of several papers from experiments in Germany, Sweden, Denmark and Finland on the effects of organic and mineral fertilisation on soil microbiological processes. In general, microbial biomass, enzyme activities and soil respiration were increased by organic compared with mineral fertilisation. However, different types of organic fertilisers (e.g. fresh vs. composted manure) also influenced the parameters of biological activity to different degrees dependent upon type and quality of the applied manure and agronomic techniques (crop rotation, soil tillage).

Soil fertility and biological parameters were measured by Scow *et al.* (1994) on the Sustainable Agriculture Farming Systems (SAFS) Project, USA. By the end of the first four years, microbial biomass levels were consistently higher in organic and low input than conventional systems, while plant parasitic nematode numbers were also consistently lower. Nematode-trapping fungi, nematodes, and microbial biomass were quantified in conventionally and organically managed field plots in the SAFS project. Bacterivorous nematodes were more abundant and microbial biomass (substrate-induced respiration) was found to be greater in the organic than in the conventional plots (Jaffee *et al.*, 1998).

Sivapalan *et al.* (1993) monitored populations of soil micro-organisms during a conversion from a conventional to an organic system of vegetable growing system. They concluded that microbial populations were greater in the organic conversion area than in the conventional area. Soil in the organic conversion area supported approximately twice the number and a wider range of fungal species than soil in the conventionally cultivated area. Others have also reported greater active fungal populations under organic production (Cook *et al.*, 1995; Yeates *et al.*, 1997).

Although a single microbial indicator can not sufficiently characterise soil quality, the long-term DOC experiment in Switzerland looked at a range of physical, chemical and biological characteristics.

Soil microbial biomass increased in the following order: unfertilised< mineral< conventional (mixed mineral/organic)< organic< biodynamic. The organic and biodynamic treatments also showed a greater microbial activity and a greater potential than the conventional treatments to mineralise organic compounds (Mäder *et al.*, 2002).

Wander *et al.* (1994) investigated whether 10 years of organic or conventional management generated differences in biologically active soil organic matter, and found that the conventionally managed soil had the lowest biological activity (N supply and soil respiration rates).

However, others have reported no or negative effects of organic farming:

Shannon *et al.* (2002) reported recent work under UK conditions and concluded that differences in the size, activity and diversity of the soil microbial biomass were subtle, rather than dramatic. They found no consistent differences between organic and conventional farming, and they argued that the scientific literature was also contradictory with reports of negative, positive and neutral effects.

Yeates *et al.* (1997) studied paired conventionally and organically managed grasslands and concluded that, whilst microbial activity differed between management and sites, there were no consistent effects. The effects of organic management on soil fauna were investigated in grasslands on different soils (silt, loam, sand) where fields had been managed either with conventional fertiliser inputs or to the organic standards of the Soil Association (Cook *et al.*, 1995). Soil mesofauna and microfauna were counted and soil microbial activity was estimated. There were found to be no consistent changes associated with management in microbial activities measured as microbial C, respiration, and dehydrogenase activity.

Mycorrhiza

Mycorrhizal fungi can significantly increase the growth of some plant species, for instance *Allium* spp., particularly on soils low in available P (Lynch & Wood, 1988). Maize is another such crop. Colonisation by mycorrhizae is therefore important in organically managed soils. Lower concentrations of available P in organically managed soils also selects for more efficient mycorrhizae, resulting in better crop growth. However, the use of soluble P fertilisers in conventional agriculture can suppress mycorrhizae. Martensson & Carlgren (1994) measured reduced hyphal length with increasing P additions, for example. Fungicides, used as crop protection chemicals, can also adversely affect mycorrhizae (Scullion *et al.*, 1998). The evidence for increased mycorrhization under organic farming is quite strong, as reported by Stolze *et al.* (2000). The organic and biodynamic treatments of the long-term DOC trial also exhibited greater root colonisation by mycorrhizae (Mäder *et al.*, 2002).

3.2.5. Susceptibility to soil erosion

There are few studies that have directly compared erosion under organic and conventional farming (Unwin *et al.*, 1995). The most often cited study is that of Reganold (1988), who compared adjacent organic and conventional farms. The organically managed soil had significantly more SOM and a significantly lower modulus of rupture, more granular structure, less hard and more friable consistence and 16 cm more topsoil (due to erosion on the conventional farm over a period of 40 years). The difference in erosion rates was attributed to different crop rotation systems and different tillage practices.

Stolze *et al.* (2000) argue that organic farming employs as standard the main erosion control methods (grass, cover crops/undersowing and regular manure additions) as well as some practices that might encourage erosion (frequent tillage and wider rows for weed control, slower developing cover because of N shortage). They argue that the positive control measures outweigh the risk factors, although no evidence is provided.

Other management factors that, potentially, could decrease water and/or wind erosion include: reduced stocking rates (compared with conventional); the requirement to maintain grass cover under outdoor pigs; more cloddy seedbeds (fine seedbeds unnecessary because herbicides are not used). Again, currently there is no comparative data for organic and conventional systems.

Recent work in the uplands of England and Wales has demonstrated that a major factor in upland erosion is animal stocking density: where grazing histories of monitored sites was known, increases and decreases in erosion rates corresponded to times when grazing levels intensified and reduced, respectively (McHugh, 2003). Though no work has been specifically undertaken comparing organic and conventional upland systems, we can surmise that grazing pressure in the uplands will be less under organic, as stocking densities are generally less. For example, at the Pwllpeiran organic unit, stocking densities were set at 60% of the conventional rate (Frost *et al.*, 2002).

3.3. Nitrate Leaching

3.3.1. Introduction

The main loss of N in drainage is by leaching of nitrate: ammonium is less mobile. Leaching occurs when water drains through the soil, taking with it nitrate from the soil profile. Consequently, most nitrate leaching occurs during the autumn/winter drainage period, though nitrate can be lost at anytime if there is sufficient rain to fully wet the soil. Thus, the amount of nitrate lost depends on soil-type and rainfall, and is modified by management practices. In short, to minimise nitrate losses, management practices that minimise the amount of nitrate in the soil during the main drainage event must be adopted. Goulding (2000) has recently produced a thorough review of the main techniques.

Nitrate leaching can be split into 'direct' and 'indirect losses'. Direct loss results from adding nitrate (or materials that are quickly converted to nitrate) when drainage is occurring: late summer/early autumn applications of slurries, for example. Indirect loss occurs when nitrate has accumulated in the soil in the autumn as a result of crop/soil/management activities in the previous growing season. Examples are:

- A crop is supplied with too much nitrogen for its needs (e.g. from fertiliser and/or manure, or from ploughed out grass)
- Lack of synchrony between N supply and crop uptake, e.g. if ploughed grass residues are mineralised after the crop has matured.

Farming systems therefore need to manage nitrogen carefully, to avoid these circumstances wherever possible.

3.3.2. Factors affecting nitrate leaching from farming systems

Manures

Animal manures applied to agricultural soils can be significant contributors to nitrate leaching. The greatest risk is from late summer/early autumn applications of manures containing significant proportions of 'readily available N' (i.e. the fraction that can be nitrified quickly). ADAS studies on conventional farms in the UK have shown large losses from such applications of slurries and poultry manures. Losses were much smaller from applications of FYM (Smith & Chambers, 1993; Unwin & Smith, 1995). Large amounts of N can also be lost from the soil in surface run-off when heavy rain falls in the first few days after slurry application (Sherwood & Fanning, 1981). It is the 'readily available' nitrogen fraction that is most at risk from leaching: ammonium-N, uric acid-N (poultry manures) and nitrate-N (generally only trace amounts in most manure). This knowledge of manure management on conventional farms can tell us about the likely impacts of manure management on nitrate leaching on organic farms. For example, it can be hypothesised that organic farming usually offers an advantage: most manures are produced from straw-based systems, and have a relatively small readily available N content, thus presenting a small nitrate leaching risk. Some manures are also composted, which tends to reduce their ammonium N content still further. However, it should be noted that nitrate can accumulate during composting and it may be that well-composted manures have potential to leach substantial nitrate (either from an uncovered heap or after application to land in autumn). This was suggested from work by Shepherd & Smith (2000) on conventional manures: Shepherd *et al.* (2002b) found more nitrate in organically managed cattle FYM than is usually reported in standard values for non organically produced cattle FYM.

Under conventional agriculture, manures are used in combination with inorganic fertilisers. The aim is to apply some of the crop's requirements with manure and then 'top up' with fertiliser. This practice can lead to significant leaching if the combined manure plus fertiliser N supply is greater than the crop's requirements. Over-fertilisation results in a large soil nitrate residue that can be leached after harvest (Chaney, 1990). This over-supply is unlikely to occur under organic farming. First, the use of supplementary fertilisers is generally not permitted and this, combined with the fact that the manures within organic systems have a low readily available N content (Shepherd *et al.*, 2002b), should guarantee no over-fertilisation. One situation where over-supply can occur is after ploughing leys. This is discussed later.

Another route for N loss is that of direct run-off of N in leachate from manure stores (Stockdale *et al.*, 2001). Clearly, manures have to be managed in such a way as to minimise this risk by having facilities to collect the leachate. Covering the manure will not necessarily eradicate the risk, because much of the N is contained in the liquor that leaks from the FYM heap in the first few days (Shepherd *et al.*, 1999). The N content in leachate leaving the heap declines with time, because the readily available N becomes assimilated into the organic fraction of the manure heap. There is likely to be little difference in losses by this route between conventional and organically produced manures.

Fertility building phase

Nitrate leaching losses from cut grassland, where herbage is removed from the field, are generally small. Greater losses occur where pastures are grazed because of the large returns of N in excreta. Urine deposition from grazing animals, though limited to only a proportion of the pasture area, can provide the equivalent of up to 1000 kg N/ha in urine patches. Much of the nitrate leached from grazed grassland originates from these localised 'hot-spots', irrespective of whether N is supplied as fertiliser or by biological fixation.

Most studies of leaching from grassland have examined pastures receiving N fertiliser. There is a direct relationship between the level of N input and the quantity leached (Barraclough *et al.*, 1992) and research has tended to concentrate on heavily fertilised swards where the risk of leaching is greatest. Ryden *et al.* (1984) demonstrated that leaching losses from grazed grass/clover swards were much less than those from intensively fertilised grass monocultures. However, differences are less evident where conventional grass/clover swards are compared with grass receiving moderate fertiliser inputs. The productivity of grass/clover pastures is considered to be broadly equivalent to fertilised grass swards receiving 100-200 kg N/ha (Davies & Hopkins, 1996). At these levels of fertiliser input, leaching losses from grazed swards are typically in the range 1-12 kg N/ha (Barraclough *et al*., 1992) and are similar to those reported for grass/clover swards.

Tyson *et al*. (1996) reported annual leaching losses of 13 kg N/ha from grazed grass/clover pastures on a heavy clay soil in Devon and 50 kg/ha from equivalent grass swards receiving 200 kg fertiliser N/ha. Cuttle *et al.* (1998) compared leaching from unfertilised grass/clover swards and grass swards receiving 250 kg fertiliser N/ha. Herbage production and the numbers of sheep that could be supported by the sward appeared to be the main factor determining the amount of N leached from pastures. The 6-year study indicated that where pastures of similar productivity were compared, losses were similar whether N was supplied by fixation or as fertiliser. Hutchings & Kristensen (1995) modelled the factors influencing nitrate leaching from grassland and similarly concluded that differences in the quantities leached from clover- and fertiliser-based swards were likely to be small at the stocking rates commonly found on grass/clover pastures. In contrast, very large losses of about 200 kg N/ha occurred where pure stands of clover were grazed (Macduff *et al*., 1990). Eriksen *et al*. (1999) reported that leaching losses were greater from second year grass/clover leys than in first-year leys on an organic farm in Denmark, presumably as N accumulated in the system.

Arable phase

Leaching from arable land is increased where fertiliser rates exceed the crop’s requirement (MacDonald *et al*., 1989), as described above. In particular, losses are associated with the temporary nature of annual crops and, sometimes, the lack of synchrony between release of N from organic matter and crop uptake. If soils are left bare in autumn or crops are poorly developed, there will not be an effective rooting system to utilise the soil N that is mineralised after harvest and this will be at risk of leaching over the winter. Increasing the fertility of organically farmed soils by building up the content of SOM and incorporating organic residues and manures increases this risk.

The greater risk of leaching during the arable phase was demonstrated in a study on 17 Norwegian farms that were either organic or in the process of converting to organic production (Solberg, 1995). The potential for nitrate leaching (determined as nitrate-N in the 0-60 soil depth in October) increased in the order; leys (6 kg N/ha) < undersown grain = green fodder (14 kg/ha) < turnips/vegetables (17 kg/ha) < grain without undersown ley (30 kg/ha) < potatoes (33 kg/ha) < fallow (100 kg/ha). Similar measurements (0-75 cm depth) on 26 organic farms in Denmark showed the potential for nitrate leaching to increase in the order; grass/clover or lucerne fields (12 kg N/ha) < bare fields following cereals (48 kg/ha) < fields cultivated with cereals (57 kg/ha) (Kristensen *et al*., 1994). Eriksen *et al*. (1999) demonstrated marked differences in nitrate leaching at different stages of a dairy/crop rotation on an organic farm in Denmark. The lowest losses were from first-year grass/clover leys (20 kg N/ha) and increased to 28 kg/ha for the second-year ley. Greater quantities of nitrate were leached (43-61 kg/ha) during the three years of arable cropping after the ley was ploughed. The overall annual leaching loss from the farm was equivalent to 38 kg N/ha.

Catch crops are effective at reducing nitrate leaching from what would otherwise be bare soil (Stockdale *et al*., 1995; Rayns & Lennartsson, 1995; Reents *et al*., 1997; Aronsson & Torstensson, 1998). A lysimeter study in Denmark demonstrated that ryegrass undersown as a cover crop halved nitrate leaching from spring barley with average annual reductions of 20-35 kg N/ha (Thomsen & Christensen, 1999). On sandy soils in the UK, the average leaching loss of 47 kg N/ha from bare soils following cereals was reduced to 22 kg/ha by sowing an over-winter catch crop (Shepherd, 1999). The catch crops were only effective where they had become well established before the start of drainage in autumn.

Cultivation of grass/clover

The flush of N mineralisation following cultivation of leys is another feature of organic systems that may increase the risk of nitrate leaching (Stopes & Philipps, 1992; Scheller & Vogtmann, 1995). This is often highlighted as an argument against organic farming.

Studies on organic farms have shown 38 kg N/ha leached where a grazed grass/clover ley was cultivated for winter wheat in September, compared with 10 kg/ha where cultivated in February for a spring crop (Philipps *et al*., 1995). Elsewhere, ploughing a 4-year ley in October resulted in 70 kg N/ha leached over the following winter (Watson *et al*., 1993). In New Zealand, cultivation of a 3-year ryegrass/white clover ley in either early or late autumn resulted in winter leaching of 78 and 40 kg N/ha, respectively, whereas delaying cultivation until late winter reduced this loss to 5 kg/ha (Francis *et al*., 1992). Considerable losses can also occur where green manures are cultivated. For example, over 100 kg N/ha was leached following ploughing a 1-year red clover crop in September, this was equivalent to about one third of the N in the above ground crop (Stopes *et al*., 1995). Again, leaching was substantially reduced where cultivation was delayed until spring. Unfortunately, the necessity of autumn cultivations to control weeds on organic farms may conflict with recommendations to minimise soil disturbance at this time of the year.

However, although the cultivation of grassland can result in large leaching losses, the overall impact is reduced because only a proportion of the ley area on a farm will be ploughed at any one time. Similarly, the overall impact on the N budget of individual fields will be reduced because these large losses will only occur in one or two years during the rotation. This argument is developed further, below.

3.3.3. Comparing farming systems

The risk of loss and the processes influencing leaching vary for different phases of the cropping rotation. The greatest risk follows the cultivation of the ley phase when large quantities of N are mineralised. Although large losses at a particular stage of the rotation will influence the immediate, short-term availability of N, the long-term effect on the N status of the soil can only be assessed over the full rotation.

Taking all these factors into account, overall leaching losses from organic farms are generally less than from conventional farms (Edwards *et al*., 1990; Younie & Watson, 1992; Eltun, 1995). However, the study by Kristensen *et al*. (1994) found average nitrate content in soils in autumn from organic farms (31 kg N/ha) to be similar to those in soils from conventional farms that also applied manure (29 kg/ha). Both were greater than for conventional farms that did not use manure (22 kg/ha) and it was concluded that nitrate contents were related to the use of manures rather than mineral fertilisers.

Condron *et al*. (2000) argue that there are few measurements to study losses through a rotation. One approach is modelling. They used the NLE model to calculate and compare leaching losses from conventional and organic dairy farms in New Zealand. Simulations showed annual losses to be 19-46 kg/ha and 9-12 kg/ha for conventional and organic, respectively. The differences were attributable to (a) lower stocking rates and (b) lower N inputs on the organic farms. Hansen *et al.* (2001) similarly adopted a modelling and N balance approach to simulate losses from arable, pig and mixed arable/dairy farms on sandy soils in Denmark. Calculated N losses were generally less from organic than from conventional due to lower N inputs and more winter cover.

The most relevant UK work has recently been reported by Stopes *et al.* (2002). They compared measurements of leaching from organic (legume based) and similar conventional rotations. Leaching losses were similar between organic and conventionally fertilised leys receiving less than 200 kg/ha fertiliser N (both before and after ploughing). Losses were greater for leys receiving more than 200 kg/ha fertiliser N, however. Losses were also greater from arable crops in the conventional systems than the organic. It was concluded that leaching from organic systems can be slightly less than conventional equivalents. Furthermore, it should be noted that comparisons were made between similar systems, rather than including the more intensive conventional farms.

Goulding (2000) constructed a 'typical' ley/arable rotation and compared literature values for each phase. The conclusion was that losses were generally smaller throughout the organic rotation, compared with conventional, except when ploughing the fertility building ley. Overall, losses were slightly smaller from the organic rotation. Stolze *et al.* (2000) undertook a comprehensive review of the nitrate leaching risks and concluded that losses were less or, at worst, the same from organic systems. It was also argued that the difference is decreasing as conventional farmers improve their N management. Hansen *et al.* (2001), also after reviewing the literature, argued that losses could be less overall from organic, but this is not guaranteed for each and every individual farm.

3.4. Phosphorus Loss

3.4.1. Introduction

Although the quantities of P lost from farmland are usually small in agricultural terms, losses of a few kg P/ha are sufficient to be of environmental concern. Transport processes of P to water are complex, and not necessarily simply related to the amount of P in the soil-crop system. Edwards & Withers (1998) concluded that the loss of P from agricultural land is controlled by factors that are independent of the annual P surplus.

Phosphorus losses from agriculture have been reviewed by Sharpley & Menzel (1987), Sims *et al.* (1998) and Haygarth & Jarvis (1999). In most soils there is little actual leaching of dissolved P because adsorption maintains low concentrations in the soil solution. Leaching is most likely on:

- Deep sandy soils or high organic matter soils, which have little capacity to adsorb P.
- Soils with high P concentrations resulting from long-term over-fertilisation and/or excessive applications of animal manures where the accumulation of P exceeds the soil's sorption capacity.

In the majority of soils, losses of P are most likely to occur in surface run-off or in subsurface drainage through the transport of P associated with colloidal clay or organic matter. These losses are less related to excessive P inputs and more related to soil and water management factors. For example, Sharpley & Menzel (1987) report quantities of P lost in subsurface drainage from fields in the United States, Canada and New Zealand ranging from <0.01 to

0.44 kg P/ha/year. Greater quantities were lost in surface run-off, ranging from 0.01 to 4.3 kg P/ha/year as soluble P and 0.02-18.2 kg/ha as particulate P. Similar losses have been reported for arable land in England (Catt *et al*., 1998). Annual losses through field drains and in catchment runoff were 0.37-2.6 kg total P/ha whereas up to 32 kg total P/ha was lost in surface run-off in a wet year due to erosion and transport of fine sediment.

Where storm events occur shortly after the application of soluble fertiliser, there may be a direct loss of P due to transport of fertiliser in surface runoff (Haygarth *et al*., 1998). Heathwaite *et al*. (1998) reported that greater quantities of P were lost in surface runoff from grassland receiving inorganic fertiliser than from farmyard manure or slurry treatments. Losses of fertiliser are less likely to occur in organic agriculture where fertilisers are applied less frequently and only relatively insoluble materials are used.

3.4.2. Effects of organic farming

There is little direct information about P leaching and runoff from organic agriculture. As budgets for organic farms rarely show a significant surplus of P (e.g. Table 1.9), losses are assumed to be small. However, this may not be a reliable indicator (Edwards & Withers, 1998). Losses may be determined more by differences between the dominant loss pathways in livestock and arable farming systems and by differing contributions of P arising from soil erosion and from the cumulative development of P surpluses.

Cropping

Conversion to organic agriculture will generally involve a change in cropping patterns and the proportions of arable land and grass, and this may affect the quantities and forms of P loss. Cultivation of leys and the introduction of arable crops may be expected to increase the risk of erosion of soil particles and sorbed P in runoff compared with grassland farms. Conversely, introduction of grass leys and catch crops into previously all-arable farms may lessen this risk. Erosion is less common on established grassland, which will limit particulate losses, although livestock can increase erosion through poaching and damage to stream banks.

Use of no-till systems, winter cover crops, grassing of valley floors and creation of riparian buffer zones have been proposed as means of reducing P loss (Withers & Sharpley, 1995). These measures will be most effective in controlling losses of particulate P, which represent the greatest risk in organic agriculture.

Manure management

It is important to distinguish between short-term losses occurring shortly after application of slurry or manure and losses resulting from an accumulation of P from heavy applications of manure over an extended period. Results of separate studies of short-term and cumulative effects were described by Smith *et al*. (1998). Direct losses following slurry applications were investigated on a silty soil that was prone to capping. High rates of application of cattle slurry (80 m^3/ha) resulted in high P losses, mainly in surface runoff. This was attributed to sealing of the ground surface as a result of the high application rate and high solids content (8%) of the slurry. The largest loss (1.8 kg/ha) was equivalent to only 3% of the applied slurry P which, although small in agronomic terms, could be significant in terms of water pollution. Losses were much smaller with diluted slurry or lower application rates. Little P was lost from applications of solid manure. At a second site, on a structured clay soil with underdrainage, only pig slurry produced significant losses of P. Little P was lost from applications of cattle manure or poultry litter. The characteristics of the soils at both sites would be expected to present a high risk of P loss by providing pathways for rapid water flow and limited opportunity for P sorption.

The cumulative effects of repeated manure applications were examined at 7 sites on freely draining soils with long-term histories of manure applications. High losses of P occurred from some sites receiving poultry manure and cattle slurry as well as from one site with pig slurry, but losses were only significant where soil levels of P had built to high levels as a result of years of over-application. Concentrations in drainage water were closely related to contents of extractable-P in the soil. There is limited information about the loss of P from composts.

Other factors

Phosphorus may also be leached from vegetation and crop residues (e.g. Bromfield & Jones, 1972; Mays *et al*., 1980; Miller *et al*., 1994). However, in the absence of surface runoff, much of the P leached from crops and residues would be expected to be adsorbed by the soil (Sharpley *et al*., 1981; Qualls *et al*., 1991).

Earthworms affect P leaching in a number of ways. Surface casts contain a high proportion of finer soil particles and can represent an important source of particulate P in surface runoff (Syers & Springett, 1984). Casts also contain more loosely-bound P than the bulk soil. Although these effects will tend to increase P losses in surface runoff, such runoff events are less likely to occur in soils with worms because of the increased infiltration that results from their burrowing. The increased infiltration is unlikely to increase leaching if the drainage occurs as macropore flow through worm channels and thus by-passes the soil matrix.

Comparison of organic and conventional farming

We were unable to find any comparative studies of P losses under organic and conventional systems and, in summary, there is insufficient evidence to make an assessment of the effects of organic farming on P loss.

Hansen *et al*. (2001) reported that the Bichel Committee in Denmark argued that there would be smaller surpluses of P under organic farming, and that this would decrease P leaching. However, we would argue that leaching, certainly under UK conditions, is a minor loss pathway. There have been conflicting reports of soil P levels increasing and decreasing under organic management (Stockdale *et al*., 2001), so that it is not possible to state categorically that P loss from erosion will be reduced under organic management.

Hansen *et al*. (2001) also pointed to two situations where organic farming might increase P loss: outdoor pigs, and fields receiving large organic matter inputs (ploughed leys, manures, cover crops), which might be expected to raise the mobility of P in the soil. However, we would argue that these arguments are tenuous. Outdoor pigs are increasingly common under conventional farming systems and therefore carry the same risk (perhaps an even greater risk, depending on stocking rate). Furthermore, organic pigs are moved frequently and should have grass cover maintained, which will reduce erosion and P loss risk. Also, many conventional systems receive manures and other additions of organic matter.

3.5. Pesticide leaching

3.5.1. Introduction

The term 'pesticides' covers a wide range of chemicals. In 2003, there are about 350 individual active ingredients, and a further 180 that contain these actives as part of a mixture, that are approved for use in the UK in conventional agriculture, horticulture and forestry. This includes acaricides, algicides, fungicides, herbicides, insecticides, lumbricides, molluscicides, nematicides, rodenticides and plant growth regulators. Only a very small number of these products are approved by the Organic Standards for use on organically farmed land.

Movement of pesticides from soil to water will depend on many of the same factors as for nutrients: soil-type and drainage flow path, rainfall after application. A further factor is the mobility of the pesticide itself. Highly mobile, water-soluble pesticides will clearly move more quickly than those that are adsorbed to soil particles. Rate of degradation is another factor in assessing the environmental impact of the pesticide. Application rate may also be important, with the amount of active substance (a.s.) ranging from a few grams to several kilograms depending on the product/formulation in use.

Recent work on the heavy clay soils at Brimstone Farm (Oxon.) showed that the most important factor influencing the leaching of moderately mobile compounds – in this case isoproturon (IPU) – was the time interval between application and the start of winter drainflow. The greatest concentration of pesticide in drainage water was recorded when applications were made under wet autumn conditions, or later in the winter when drains were flowing. Results from this study suggested that losses via drain-flow decreased by half for about every 10 days with no drainage (Jones *et al*., 2000). The decrease in concentrations and losses of IPU in drain-flow with increasing time from application to the first drain-flow was significantly greater than would be expected from degradation alone. These findings have led to recommendations that IPU is applied at the earliest opportunity in the autumn period, even if this means applying to dry soil. This study also demonstrated that reduction in the application rate of IPU (from label rate of 2.5 kg/ha) resulted in at least an equivalent reduction in concentrations (but not % losses) in drainage water. However, losses of a more mobile, moderately persistent herbicide were less dominated by the time from application to drainage event, with similar concentrations recorded under all weather patterns experienced during the study.

As well as diffuse losses, as described above. Pesticides can contaminate water from small point sources. Contamination events derive from spillages or discharges of product, tank mix, waste or washings directly to surfaces or drainage systems that can enter surface water or via soakaways to groundwater.

In general, it is apparent that losses of pesticides to receiving waters must be very small if the waters are to fall within the EU Drinking Water Directive limit of 0.1 μg/l, even when dilution factors are applied. The following chemicals were found above 0.1 μg/l on more than 1% of groundwater samples, when sampled in 2001:

- Atrazine (3.7%)
- Bentazone (2.6%)
- Simazine (1.4%)
- Diuron (3.1%)
- Mecoprop (1.5%)

- Isoproturon (1.6%)
- Pentachlorophenol (1.4%)

All are herbicides except pentachlorophenol, which is used for wood treatment and as a general biocide. None of the herbicides would be permitted within organic standards.

3.5.2. Effects of organic farming

The consensus of many reviews on this subject is the same: because synthetic pesticides are not approved for use in organic agriculture, the risk of contamination of air and water by these materials is avoided (Stolze *et al*., 2000; Condron *et al*., 2000; Hansen *et al*., 2001; Stockdale *et al*., 2001). Though entirely logical, there are no studies to support this, because the risk of pesticide effects on water quality in organic systems have been largely unstudied (Stockdale *et al*., 2001). However, the circumstantial evidence is further strengthened by the fact that no herbicides are allowed in organic farming, whereas most of the water contamination relates to herbicides.

Some chemicals are permitted for use in organic farming: copper, sulphur, natural pyrethroids (restricted use) and derris (restricted use), but they tend to be used as a last resort and their approved uses are for minority or protected crops. Sulphur is not a harmful chemical (in fact, it is a valuable plant nutrient). The others are not mobile in the soil and pyrethroids and derris are not very persistent in soil. Therefore, the risk of pollution from these materials is small. This is also supported by the fact that no examples of water contamination have been reported (Stolze *et al*., 2000), although it could be argued that they have not been looked for (Unwin *et al*., 1995). As with all materials, there is always a risk of pollution from spillage (Unwin *et al*., 1995).

Organic livestock production prohibits (a) the routine use of antibiotics, (b) all organophosphates and (c) some ivermectins. This will also contribute to a smaller pollution load from organic agriculture.

There is some debate about disposal of sheep dip and relative risks of pyrethroids versus organophosphates. Organic farmers only use the former and they are potentially more damaging to aquatic habitats. However, all disposal systems have to be licensed, care is needed in use and, also, some conventional farmers also use them because of health concerns for operators from organophosphates.

3.6. Pathogens

3.6.1. Introduction

The number of reported cases of food-borne illness has risen significantly in the UK over recent years, with a six-fold increase in the collective number of gastro-enteritis and food poisoning cases between 1982 and 1998. The main causative agents are bacteria, particularly *Salmonella*, *Campylobacter* and verocytotoxic *Escherichia coli* (VTECs) and viruses, in particular small round spherical viruses (SRSV). In addition, significant levels of human illness are caused by the parasitic protozoa *Cryptosporidium* and *Giardia*, and it is likely that in many cases transmission to man is via food or water contaminated with these pathogens. The application of organic manures to agricultural land is one route by which pathogens may be introduced into the human food chain during the primary food production stage.

3.6.2. Pathogen transfer in farming systems

There are relatively few data currently available on the relative risks of pathogen transfer from organic and conventional farming systems. Three phases of management need to be considered:

- Manure production, collection and transfer – currently few data on differential pathogenic burdens between management systems.
- Manure storage and treatment – pathogen levels can decline during storage of manure (Himathongkham *et al*., 1999; Kudva *et al*., 1998), particularly if solid manure is actively composted to increase the temperature of the heap: stacked manure may not reach the requisite high temperatures (Nicholson *et al*., 2002). Spreading manure directly from store to the land will increase the risk of pathogen transfer. Thus, it might be concluded that organic farming provides a lesser risk because manures are generally composted or stacked. However, there are no data yet to substantiate this, but research is on-going. Anaerobic and aerobic slurry treatment systems can reduce pathogen numbers in slurry (Bendixen, 1999).
- Landspreading – simple management procedures will minimise the risk of transfer to crops (crops eaten raw are the greatest risk) and water. These procedures are likely to be followed in both organic and conventional farming systems, so it is not possible to identify differences in risk without further work.

The Food Standards Agency (Anon., 2003f) have made the following statement about risk of microbial contamination of food:

> *There is no firm evidence at present to support the assertion that organic produce is more or less microbiologically safe than conventionally farmed produce. However, the Agency recognises that there is a potential risk to food safety from the use of organic wastes in agriculture, both conventional and organic, and, in conjunction with Defra, is carrying out a structured programme of research and risk assessment into the use of all organic wastes on agricultural land.*

3.7. Ammonia Emissions

3.7.1. Introduction

Reducing ammonia (NH_3) emissions is a policy requirement in the UK (Anon., 2002g). Ammonia causes acidification and eutrophication when redeposited to soils and waters (Roelofs *et al.*, 1991), and can damage sensitive habitats. Directive 2001/81/EC on National Emission Ceilings for Certain Atmospheric pollutants aims to limit emissions of acidifying and eutrophying pollutants and ozone precursors to improve protect against the risks from acidification, soil eutrophication and ground level ozone. This aim is consistent with the long-term objectives of not exceeding critical levels and loads and protecting people from the health risks of air pollution by establishing national emission ceilings. By 2010, Member States must limit their annual national emissions of sulphur dioxide, nitrogen oxides, volatile organic compounds (VOCs) and ammonia.

Agriculture, particularly livestock production, accounts for about 80% of NH_3 emissions in the UK (Anon., 2002g). Ammonia is produced when urea in urine and dung comes into contact with the enzyme urease. This enzyme is very common and can be found in both manure and soil. Therefore, animal housing, manure stores and the spreading of manures to land are major sources of NH_3. There has been a large amount of research into NH_3 emissions from conventional animal production systems but only a few studies specifically on organic farms (Stockdale *et al.*, 2001). Much of the research conducted using conventional systems may be applied to organic farms. However, differences in dietary N intake and N excretion, housing system and period, manure storage and spreading and livestock density, will affect the amount volatilised (Stolze *et al.*, 2000). There is therefore potential for NH_3 losses to be different from organic systems, which operate at a lower level of intensity than most conventional systems.

3.7.2. Factors affecting ammonia losses from farming systems

Diet

Most of the nitrogen fed to cattle is excreted. Clearly, the quality of the diet will influence the amount of N excreted and its distribution between dung and urine (with most NH_3 lost from urine, Sommer & Hutchings, 1997). Organically reared cattle tend to be fed more forage (containing a higher proportion of legumes) and less concentrates than those in conventional systems, although it is uncertain what effect this has on N excretion. N losses are likely to be different if the N content of dung and urine from an organically reared animal is different to that from a conventional system (Stolze *et al.*, 2000), as NH_3 emissions are dependent on the N content (particularly NH_4-N) of the manure (Shepherd *et al.*, 1999). In a survey of 43 cattle FYM and 14 cattle slurries from organic farms in the UK, Shepherd *et al.* (2002b) measured manure nutrient (N, P, K) concentrations that were c. 20-40% less than published values for 'conventionally' produced manures (Anon., 2000b). The ammonium-N content of the organically produced manures was also less than those from conventional farms (0.26 kg/t and 0.74 kg/m^3 NH_4-N in organic cattle FYM and slurry compared to 0.77 kg/t and 1.4 kg/m^3 in conventional cattle FYM and slurry as reported in RB209 (Anon, 2000)). However, these manures were largely sampled from stores and therefore reflect the outcome of all the management processes associated with the production of manures (housing and storage as well as diet). So although NH_3 emissions following land-spreading of organically produced manures may be lower than those from conventional manures (see below), the overall NH_3 loss from the organic system may not necessarily be lower, due to significant NH_3 emissions during housing and storage.

Housing system and period

Ammonia loss during animal housing is inevitable. However, factors such as the surface area, bedding material and ventilation will all affect the amount lost (Shepherd *et al.*, 1999). Emissions from housed animals are considered to be greater than from those grazed, as urine is quickly absorbed into soils. Therefore, as housing periods tend to be shorter in organic systems (maximum grazing is recommended), the potential for ammonia loss is likely to be less, although this has not been tested (Stolze *et al.*, 2000). Straw-based systems will also tend to have lower emission rates than systems based on slurry, due to the absorption of urine by straw (Pain *et al.*, 1998). Straw-based systems are more common than slurry systems in organic agriculture.

Manure storage

Ammonia emission from manure stores depends on the surface area of manure in contact with the air and level of disturbance. Organic systems encourage the composting of solid manure. This involves active turning of the manure to produce a more stable, uniform product, free of weeds and toxins and easier to spread (Lampkin, 1992). However, there is a large amount of evidence to suggest that ammonia losses are greater from composted manures compared to those which are just stockpiled (Kirchmann, 1985; Shepherd *et al.*, 1999; Gibbs, *et al.*, 2000), with losses ranging from 5-70% of the total manure N. Increasing the straw allowance (and therefore C:N ratio) can decrease losses, although this may not be cost-effective (Shepherd *et al.*, 1999). Losses can be less from slurry stores, particularly if they are left undisturbed or are covered (Shepherd *et al.*, 1999). Organic regulations encourage slurry aeration (for similar reasons to composting; Burton, 1997), but if this is done incorrectly, NH_3 emissions can increase due to removal of the surface crust and increased transport of NH_4 from subsurface layers to the surface (Stevens & Cornforth, 1974).

Spreading

Spreading systems will largely be the same regardless of whether the farm is organic or conventional. Generally, the amount lost will depend on the NH_4-N content of the manure (with greater losses from slurries compared to solid manures) and speed of incorporation (Chambers *et al.*, 1999). As mentioned above, the NH_4-N content of organically produced manures tends to be less than that of conventional manures, so losses during spreading are likely to be lower from an organic system, although this has not been tested. Also, the ammonium N component of composted manures is smaller then from fresh FYM, so losses after application will also be smaller.

Grazing

N in urine excreted during grazing can be a significant source of NH_3, whereas loss from dung pats tends to be insignificant (Ryden, 1996; Sommer & Hutchings, 1997). Potentially, there will be very little difference in the amount of NH_3 emitted from the dung and urine of an organically produced animal compared to a conventional one. However, livestock densities tend be lower on organic farms, as previously discussed in Section 1.5. This is likely to decrease the potential for NH_3 losses compared to conventional systems (Stockdale *et al.*, 2001).

Other factors

Ammonia may be lost from some organic systems (particularly stockless systems) during the cutting and mulching of fertility building crops (Whitehead *et al.*, 1988; Janzen & McGinn, 1991). However, losses are likely to be minimal. For example, Shepherd *et al.* (1999), based on a relationship developed by Schmidt *et al.* (1999), estimated a loss of just 0.4 kg/ha N as NH_3 during the cutting and mulching of a 2 year grass clover ley grown on a theoretical stockless organic farm.

3.7.3. Comparing systems

Ammonia losses occur right through the animal production system (Fig. 3.3), so that ammonia saved, for example, during housing might be susceptible to loss during manure storage and/or spreading (unless it is immobilised into non-ammoniacal forms). This whole needs to be considered when comparing ammonia losses from different production systems. Two factors working in favour of reduced emissions from organic farms are these: no intensive pig and poultry units, and lower stocking rates (Section 1.5).

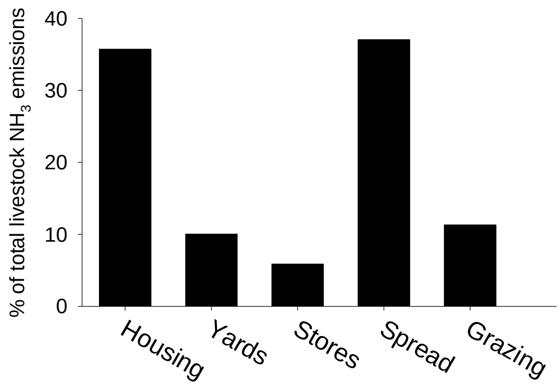

Figure 3.3. Proportion of ammonia losses from different phases of the manure production cycle (J. Webb, pers. comm.)

The UK ammonia inventory provides detailed emission factors for NH_3 losses from the major UK agricultural systems (Misslebrook *et al.*, 2002), with separate factors for housing, storage, land spreading and grazing. However, the emission factors do not differentiate organic and conventional systems. Stolze *et al.* (2000) concluded that organic systems are likely to emit less NH_3 compared to conventional systems due to lower N inputs, greater reliance on straw-based systems, shorter housing periods and lower stocking levels. However, this may not be the case, particularly if manures are actively composted. Very few studies have compared NH_3 emissions from organic and conventional farms. Where this has been done, case studies or theoretical farms have been constructed and losses calculated using the emission factors and 'rules' derived for conventional systems. Stolze *et al.* (2000) report a case study from Sweden conducted by Lundström (1997) where NH_3-N emissions were slightly higher from conventional farms (4.8 g N/kg milk) compared to organic farms (4.6 g N/kg milk). Shepherd *et al.* (1999) calculated a nutrient budget for a model organic dairy farm (70 LU) with an N surplus of 99 kg/LU (124 kg/ha). Ammonia emissions were estimated to account for 15% (15 kg/LU) of the surplus. This was compared to the nutrient budget calculated by Jarvis (1993) for a conventional dairy farm (165 LU) where the N surplus was 124 kg/LU (270 kg/ha). Here, total NH_3 emissions were greater (at 21 kg/LU) than in the model organic farm constructed by Shepherd *et al. (*1999), but they represented a similar proportion of the total N surplus (17%). It is generally assumed that organic pigs and poultry will have similar NH_3 emissions to conventional outdoor units (as reported in the NH_3-inventory: Misslebrook *et al.*, 2002). With all organic farming systems, in the absence of direct measurements, it seems reasonable to assume that the amount of NH_3 lost per unit of yield is unlikely to differ to that from conventional systems, but that losses per unit area are likely to be less, due to lower livestock densities.

3.8. Nitrous Oxide

3.8.1. Introduction

Nitrous oxide (N_2O) contributes to global warming and to the depletion of ozone in the stratosphere (Bouwman, 1996; Crutzen, 1981). It can be produced both aerobically during the nitrification of ammonium ions and anaerobically during the denitrification of nitrate ions, which are present in both soils and manures (Hutchinson & Davidson, 1993). Agriculture is therefore a major source of N_2O, estimated to contribute *c.* 47% of the total UK N_2O

emissions (Brown *et al*., 2002). Of this, the largest sources are soils fertilised with inorganic fertilisers, and manure stores (Chadwick *et al*., 1999; Brown *et al*., 2002). In the absence of inorganic fertiliser applications (as in organic systems) the main sources of N_2O will therefore be from the production, storage and application of livestock manures. N_2O can also be produced during the decomposition of fertility building crops in soils.

Most research on gaseous N emissions from livestock farming has concentrated on ammonia (NH_3). N_2O emissions are less commonly reported and are generally considered to be much smaller than NH_3 emissions: 320 kt ammonia vs. 140 kt nitrous oxide in 2000, of which approximately 65% of the N_2O derived from agriculture (Anon., 2003g). Although this means they are less significant in terms of nutrient loss, the environmental impact of N_2O emissions from livestock farming is still important. As with NH_3, in the absence of direct measurements on organic farms, much of the research using conventional systems can be applied. However, differences in dietary N intake, housing system and period, manure storage and livestock density may affect the amount emitted.

3.8.2. Factors affecting nitrous oxide losses from farming systems

Housing

There have been a few studies which suggest N_2O losses during animal housing will be greater from straw based systems (which are very common in organic farming) compared to slurries, because the presence of straw supplies a carbon source which encourages both nitrification and denitrification (Sneath *et al*., 1997a). However, evidence for this is inconsistent (Shepherd *et al*., 1999) and the UK N_2O inventory makes no differentiation between housing emissions from solid and slurry based systems (Chadwick *et al*., 1999).

Storage

Denitrification relies on a source of nitrate and carbon as well as shortage of oxygen. Stockpiled manures and slurry stores, which tend to be anaerobic, therefore provide ideal conditions for denitrification, although shortage of nitrate can limit the amount produced (Sibbesen & Lind, 1993). Composting (as encouraged in organic systems) increases the level of aeration in solid manures and may reduce N_2O losses. However, nitrate levels also increase in composted manures, which may enhance N_2O losses, although this has not been measured directly. N_2O losses from slurry are considered to be negligible due to the minimal amounts of nitrate present and the tendency for complete denitrification to N_2 gas rather than N_2O (Shepherd *et al*., 1999).

Spreading and grazing

N_2O losses following manure spreading should not differ between organic and conventional farms, unless there are substantial differences in the manure N content. The loss of N_2O from grazed grassland can result from the nitrification and denitrification of N within dung and urine patches and also from an increase in the number of anaerobic sites in the soil due to compaction by the treading of grazing animals (Oenema *et al*., 1997). As livestock densities tend to be lower on organic farms (see NH_3 section), the potential for N_2O losses via this route are likely to be lower also.

Incorporation of fertility building crops

Decomposition of crop residues following the incorporation of fertility building crops may significantly contribute to the total N_2O emission from cultivated soils. For example, Flessa *et* *al.* (2002) measured high annual N_2O emissions (7.4-12.9 kg N_2O/ha/yr) following the incorporation of legume residues on an organic farm in Germany. These were attributed to an increase in available N and enhanced microbial respiration giving rise to anaerobic microsites within the soil.

3.8.3. Comparing farming systems

Very few studies have compared N_2O emissions from organic and conventional farms. Stolze *et al.* (2000) reported a case study from Sweden conducted by Lundström (1997) where NO_x emissions, expressed per kg milk produced, were higher on 6 organic farms compared to 6 conventional farms. However, in the absence of direct quantitative data, Stolze *et al.* (2000) concluded that no definite differences between organic and conventional farms could be identified. Since the Stolze review, Ball *et al.* (2002) reported results from a study in Scotland where N_2O emissions were measured from both the arable and ley phases of an organic system. Peaks of emission were lower than those often observed in conventional systems, although substantial N_2O emissions were observed from the arable component (2.9 kg N_2O/ha/yr) that exceeded those from a separate study on conventional farmland (0.7 kg N_2O/ha/yr; Dobbie *et al.*, 1999). This was attributed to the use of FYM on the organic farm. However, in both studies, gaseous losses were more related to rainfall during the growing season than to cropping.

Fertiliser application can stimulate N_2O emissions, so that fertilised grassland is often responsible for the highest N_2O emissions (Skiba *et al.*, 1996). Thus, in the nutrient budget constructed by Shepherd *et al.* (1999), N_2O emissions accounted for less than 2% of the total N surplus of an organic dairy farm (1.5 kg/LU). This was compared to a loss of 25 kg/LU N_2O (c. 20% of the N surplus) from the conventional dairy farm budget calculated by Jarvis (1993). Ball *et al.* (2002) observed considerably lower N_2O losses from organically managed permanent grassland (2.9 kg/ha/yr) or ley (3.0 kg/ha/yr) compared to those measured by Dobbie *et al.* (1999) on conventionally managed mown grassland (9.0 kg/ha/yr). Both studies were conducted in Scotland over a similar period. A recent study by Flessa *et al.* (2002) also measured smaller N_2O emissions (c. 30% less) from an organic farm compared to an adjacent conventional system, when expressed on an area basis, but because yields were lower on the organic farm, there was little difference in emissions per unit yield.

N_2O emissions from field soils can be very sporadic, with emission peaks usually linked to rainfall events. This makes measurement very difficult. Emissions have been related to the total N input in the form of fertilisers, manures and crop residues (Flessa *et al.*, 2002). Consequently, it has been largely assumed that, because organic farms operate at a much lower intensity, with lower N inputs and less available mineral N in both manures (Shepherd *et al.*, 1999) and soils, N_2O losses will also be lower (Stolze *et al.*, 2000). However, until recently, there have been no quantitative comparisons between organic and conventional systems. Recent studies suggest that losses will be lower from organic grassland systems. However, these could be offset by higher losses from organic arable production due to the incorporation of leguminous fertility building crops. For example Ball *et al.* (2002) conclude that despite reduced losses during the ley phase of an organic rotation, the conversion to organically managed ley/arable systems may have little overall beneficial effect on N_2O emissions because of the enhanced losses associated with FYM additions during the arable phase. In the absence of any further studies it therefore seems reasonable to assume that (as with NH_3), the amount of N_2O lost per unit of yield is unlikely to differ to that from conventional systems, but losses per unit area may differ, depending on the cropping system and input of organic manures.

3.9. Methane

3.9.1. Introduction

Methane (CH_4) is responsible for contributing approximately 2.5% of total greenhouse gas emissions (Schonwiese, 1995). Approximately 40% of the UK's methane emissions in 2000 came from agriculture (Anon., 2003g). Agricultural processes in the UK result in net emissions of methane from the digestive processes in animals and from animal wastes. Total methane emissions from agriculture were around 1 Mt in 2000 (Anon., 2003g). The main contributor, *c*. 90%, was enteric fermentation in livestock (mainly cattle, *c*.75%, and sheep, *c*.15%). Livestock wastes were mainly responsible for the other emissions.

3.9.2. Comparing farming systems

The amount of CH_4 that is released depends on the type, age, and weight of the animal, the quality and quantity of the feed, and the energy expenditure of the animal (Anon., 1997b).

There is little direct data available to compare CH_4 actual emissions from different farming systems. Flessa *et al.* (2002) compared a conventional and an organic farm in southern Germany and calculated that CH_4 emissions were about 25% higher from the conventional farm (per 500 kg livestock unit, LU). The two farms reared beef cattle and the calculation was based on a constant methane emission factor per LU. The main factor accounting for the difference between farms was the larger methane emission from manure production because a slurry system was employed on the conventional farm. Methane emissions were smaller on an area basis, but the difference was less when based on a unit production basis.

To assess the emissions from farming systems, several factors need to be considered. Consequently, the interaction of these factors will affect the overall assessment of methane emissions from the farm. The result of any assessment will also depend on whether the assessment is based on an area or unit production basis:

- Animal numbers and type – *c*. 90% of CH_4 emissions comes from the animal's digestive process, as described above, with most from ruminant livestock production (Anon., 2003g).
- Diet – feeding systems that rely less heavily on concentrate rations and more on forage based feeding systems tend to produce more methane (Kulling *et al*., 2002). This is because methane emissions are closely related to the amount of rumen fermented organic matter (OM) or the amount of digestible OM since more than 50% of digestion occurs in the rumen (Moss *et al*., 2000). Methane production should be less when high concentrate diets are fed (Fahey & Berger, 1988). Van Soest (1982) indicated that a high grain diet and/or the addition of soluble carbohydrates gave a shift in fermentation pattern in the rumen, which gives rise to a more hostile environment for the methanogenic bacteria and passage rates are increased, ruminal pH is lowered and certain populations of protozoa, ruminal ciliates and methanogenic bacteria may be eliminated or inhibited.
- Manure management system – Methane is produced from the decomposition of manure under anaerobic conditions. When manure is stored or treated as a liquid in a lagoon, pond or tank it tends to decompose anaerobically and produce a significant quantity of methane. When manure is handled as a solid or when it is deposited on pastures, it tends to decompose aerobically and little or no methane is produced (Anon., 1997b). Hence the system of manure management used affects emission rates. Since slurry based systems

are less common under organic agriculture, methane emissions from the manure component of the system are likely to be less.
- Levels of productivity – greater emissions per unit of production will result from lower productivity animals (Moss, 1994), though evidence is only available for the dairy sector (albeit, the most important sector in terms of methane emissions). Because each animal will have a basal level of methane production, it may be more efficient, in terms of methane emissions per unit of production, to have fewer higher yielding animals on the farm. Reductions of total emissions would only result if livestock numbers were reduced correspondingly (Moss *et al*., 2000). This argument is pursued in more detail, below.

The IPCC methodology (Anon., 1997b) for calculating greenhouse gas emissions represents the best available information on factors affecting emissions from agricultural activities. The methane emission arising from enteric fermentation from a single ruminant depends on:
- average daily feed intake (megajoules (MJ) per day and kg per day of dry matter).
- methane conversion rate (i.e. the percentage of feed energy converted to methane).

Typically, the methane conversion rate is taken as 4-6% (±0.5%). The lower figure represents cattle fed on high quality diets confirming the conclusion, above, that feeding systems that rely less heavily on concentrate rations and more on forage based feeding systems tend to produce more methane.

Therefore, it can be seen that anything that increases daily feed intake potentially increases methane emissions. This might include: animal weight; average weight gain per day; feeding situation; milk production per day; average amount of work performed per day; percentage of cows that give birth in a year; feed digestibility. Table 3.2 shows how the calculated emission factors have increased over the last decade. Figure 3.4 shows that, although many factors that impact on energy intake will impact on methane emissions, there is a fairly good linear relationship with milk production because milk production is a major factor affecting energy intake by dairy cows.

Table 3.2. Methane emission factor for dairy cattle used in the construction of UK methane emission inventories produced by Anon. (2003g), based on the IPCC (Anon., 1997b) methodology.

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999
Average Weight of cow (kg)	550	556	561	567	572	578	584	590	596	602
Average Rate of Milk Production (litre/d)	14.3	14.2	14.5	14.7	14.7	15	15.1	15.9	16.1	16.4
Average Fat Content (%)	4.01	4.04	4.06	4.07	4.05	4.05	4.08	4.07	4.07	4.03
Enteric Emission Factor (kg CH_4/head/y)	104	104	106	107	107	109	110	113	114	115

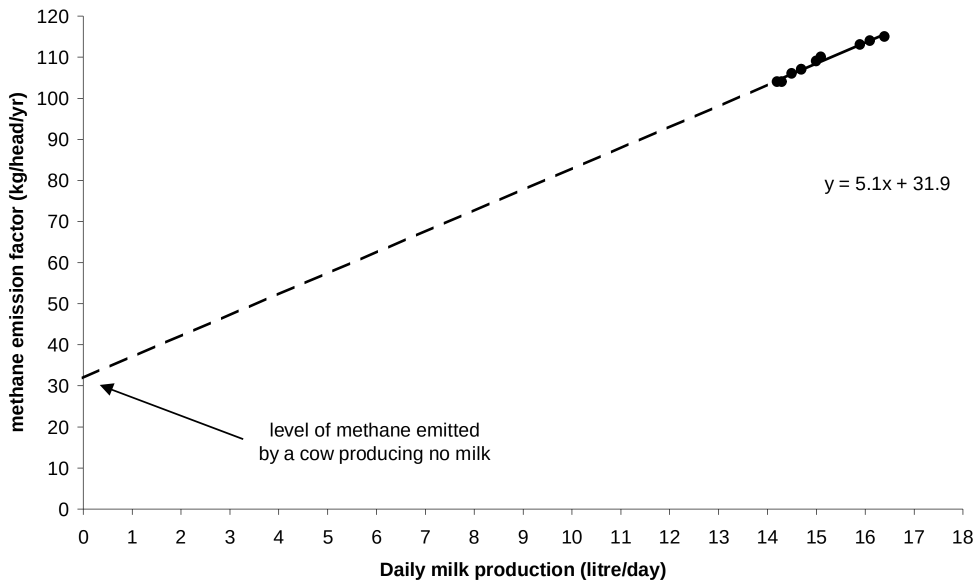

Figure 3.4. Relationship between daily milk production and estimated milk production (based on data from Anon., 2003g), and extrapolated to zero milk production (dashed line) to indicate the level of methane emitted by a cow producing no milk.

Thus, at least within the bounds of the relationship shown in Fig 3.4, decreasing the milk production of an animal by 12.5% (i.e. from 16 to 14 litres milk/day), for example, will increase methane emissions by 4%, assuming that the total target milk production is the same. This is because a greater number of lower-producing animals are required to meet the overall milk target, and Fig. 3.4 shows that, even at nil milk production, a dairy cow will emit approximately 32 kg methane/day.

The size of the potential effects of milk productivity on methane emissions can be considered in more detail by using statistics of annual milk production from organic and conventional dairy herds. However, annual average production data are variable across years and cattle type. For example, Defra statistics (Anon., 2003h) show that the average milk yield per cow (averaged across the National herd) were 5977, 6347 and 6531 litres for 2000, 2001 and 2002, respectively. Equivalent data for solely organic herds are not available. Lampkin *et al.*, (2002) suggest that annual yields from an organic dairy cow might be 4500 litres (Guernsey) to 6000 litres (Friesian/Holstein). Roderick *et al.* (2002) reported average milk yields of 5874 litres/cow (range 5127-7031 litres/cow) from a series of organic herds. Promar International (Anon., 2003i) report the rolling 12 month average yield (to March 2003) as 7400 litres for conventional and 6100 litres for organic (though data for organic are limited).

However, as an example, Table 1.7 provides example milk productivity before and after conversion under Defra's Organic Farming Scheme (Anon., 2002e). This shows a reduction in milk yield per cow from 6520 to 6157 litres/year. Overall production was similar (approximately 43 million litres), so cow numbers increased by 7.5% after conversion to sustain this level. Table 3.3 shows the effects that this change would have on methane emissions, based on the relationship from Fig. 3.4. Figure 3.5 extends this calculation to explore the relationship between decreasing milk yield per cow and increased methane emissions.

Table 3.3. Calculated methane emissions from organic and conventional dairy cows, based on an overall target milk production of 43 million litres/year, as reported by Anon. (2002e). Organic milk yields based on data from Anon. (2002e), conventional milk yield data are from a range of sources as indicated in the Table.

Production system	Milk yield (l/cow)		No. cows	Methane emissions	
and data source	Annual	Daily		kg/cow	Total (t)
Organic	6,157	16.9	6,984	118	824,307
Conventional 1	6,520	17.9	6,595	123	811,865
Change over organic		+6%	-6%		-2%
Conventional 2	6,917	19.0	6,217	129	799,752
Change over organic		+12%	-11%		-3%
Conventional 3	8,398	23.0	5,120	149	764,670
Change over conventional 3		+36%	-27%		-8%

Data sources: 1 – Assessment of Organic farming Scheme (Anon., 2002e); 2 – Kingshay Dairy Manager Costings, average 2001/02 (Anon., 2003j); 3 – Kingshay Costings, top 10% of herds (Anon., 2003j).

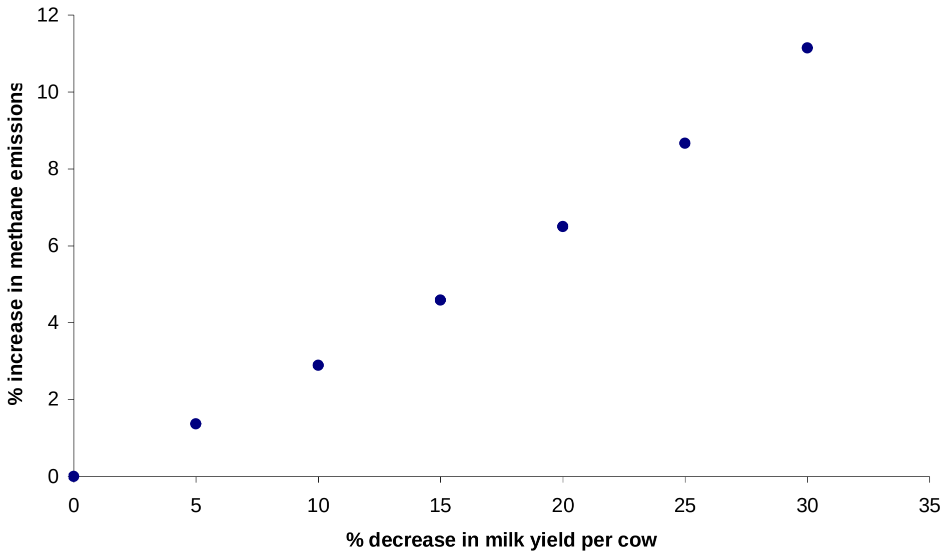

Figure 3.5. Relationship between reduction in milk yield per cow and increased methane emissions. Based on the assumption that the basal milk production is 6520 litres/year per cow and target production is 43 million litres/year (data from Anon., 2002e).

Thus, methane emissions expressed on a unit milk output basis would increase under organic production. However, emissions would decrease when expressed on an area basis because of the lower stocking densities. Assuming typical stocking densities of 1.8 and 2.2 cows/ha for organic and conventional cattle, respectively (Table 1.5), methane production per ha in the organic system would be 78% of the conventional system.

In short, ruminant methane emissions will be smaller in more efficient dairy systems. This does not only apply to dairy systems. Using the same IPCC methodology, Hyslop (2003) concluded that that the greatest potential to reduce gaseous pollutants from beef cattle systems could be achieved by the use of intensive, concentrate based finishing systems.

Most of the data relate to cattle production systems. Pigs are not ruminants and emit little methane. Sheep are generally kept extensively on organic and conventional systems, though stocking rates tend to be less on the organic units (e.g. Frost *et al*., 2002), and so there is probably some difference between systems in terms of methane emissions.

3.10. Carbon dioxide

3.10.1. Introduction

Agriculture is both a source and a sink for carbon dioxide. The source of CO_2 in agricultural systems derives from direct effects such as the burning of fossil fuels and the indirect consumption of energy resulting from processes, e.g. the production and transportation of fertilisers. The sink for CO_2 is essentially organic matter, which can act as a temporary store for atmospheric carbon.

3.10.2. Comparison of farming systems as CO_2 sources

Organic farming principles seek to 'reduce the use of non-renewable resources'. To date there have been only a limited number of studies that investigate the impact of organic farming on CO_2 emissions. Stolze *et al*. (2000) report that the available data generally deal with gross emissions on a commodity basis expressed as output per ha. There are no data available on CO_2 net balances in agriculture. A number of studies have sought to compare CO_2 emissions from organic and conventional farming systems (Table 3.4).

Table 3.4. Studies comparing carbon dioxide emissions from organic and conventional farming (from Stolze et al., 2000), based on average farm characteristics of crop management and rotations in Germany.

	CO_2 emissions t/ha		Organic as % of conventional
	Conventional	Organic	
Haas & Kopke (1994)	1.25	0.50	40
Anon. (1996)	1.75	0.60	34
Rogasik *et al*. (1996)	0.73	0.38	52

Flessa *et al*. (2002), in a study that integrated the evaluation of greenhouse gas emissions from two farming systems (conventional and organic) in southern Germany, reported that CO_2 emissions from both systems were the lowest contributory greenhouse gas (*c*. 15%). By combining the greenhouse gases to CO_2 equivalents for the farming systems the conventional system produced 4.2 Mg/ha CO_2 equivalents, compared with 3.0 Mg/ha CO_2 equivalents from the organic system. Flessa *et al*. (2002) concluded that converting from a conventional farming system to organic production methods led to a reduction in greenhouse gas emissions per ha, but yield related emissions were not reduced. Other studies have similarly concluded that gross emissions from organic farming systems result in smaller CO_2 emissions, based on an area basis. Table 3.4 shows that CO_2 emissions from organic are much smaller on an area basis than from conventional systems. The differential is sufficiently large that, even allowing for smaller yields in an organic system (Section 1.5), CO_2 emissions will be less from organic than from conventional, even when making the comparison on a unit yield basis.

Clearly, carbon dioxide emissions are also linked to energy use, and these aspects are discussed in Section 3.11.

3.10.3. Carbon sequestration

Theoretically, there is some scope for sequestering atmospheric carbon dioxide into soil organic matter. It is argued that, while only a temporary measure, it would 'buy time' to put other, more effective measures in place to decrease atmospheric CO_2 levels. Thus, any measures that could potentially increase soil organic matter content (organic manure applications, cover crops, minimising periods of fallow, etc.) will also sequester carbon. The effects of organic farming on soil organic matter were thoroughly reviewed above (Section 3.2). The conclusion was that there was potential for small increases, though much depended on the carbon balances for individual systems. Consequently, effects on additional carbon sequestration will also be small and likely to be insignificant on a global scale. Reducing soil disturbance by moving to minimal or no-till systems has the potential to decrease oxidation of soil organic matter. However, these systems are perhaps less appropriate for organic systems because of their need of herbicides for weed control.

3.11. Energy

3.11.1. Introduction

The dominant energy input in farming systems is solar energy, which drives photosynthesis. However, this has not been considered in studies as it is seen as a limitless energy source and is orders of magnitude greater than the other inputs, so a consideration of it would swamp other effects and make interpretation of the supply energy from fossil fuel difficult (Hulsbergen & Kalk, 2001; Refsgaard *et al*., 1998). Most published studies have split energy use on farms into direct energy as electricity, fuel oils etc, and indirect energy used in the manufacture and transport of fertilisers, pesticides, animal feeds and in the manufacture, transport and maintenance of machinery. A few studies have calculated the energy input in the form of human labour, but most have excluded it due to the difficulties in doing this (Refsgaard *et al*., 1998). Energy use is generally presented as the energy consumption per unit area, or unit of output, and by the efficiency of energy use calculated as the ratio of energy input to energy output. Several studies have been made on energy use in organic farming systems and there are some consistent messages from these. However, differences in methodology, for example in the method of calculating the energy inputs in machinery manufacture and maintenance, and in the setting the boundaries of the systems being studied, make it difficult to make meaningful comparisons of absolute or even relative values between studies (Refsgaard *et al*., 1998). Also, there is no single clearly defined system of conventional farming, it comes in a range of forms from very high input intensive systems to near-organic systems. This must be taken account of when drawing conclusions from studies comparing organic and conventional systems.

3.11.2. Energy data

Direct energy input

Inputs of direct energy per unit area in the long-term DOC trial in Switzerland were similar across conventional, low input and organic systems (Alföldi *et al*., 1995b). This is not surprising as basic operations such as ploughing, cultivation, sowing and harvesting are likely to be broadly similar irrespective of system. Reduced fuel costs in organic systems due to the absence of most pesticide applications, and lower harvesting energy inputs due to lower yields, can be balanced by increased fuel use for mechanical weed control. A notable exception is the energy in fuel for flame weeders in horticulture which can be greater than the herbicide energy used in a conventional system (Cormack, 2000). This would also apply where gas powered burners are used to kill potato haulm.

Indirect energy input

Organic systems generally have substantially lower indirect energy inputs than conventional systems. Machinery energy inputs per unit area are generally similar; the major difference is in the greater energy use in conventional systems to produce and transport fertiliser, particularly nitrogen, and pesticides (Berardi, 1977; Alföldi *et al*., 1995b; Stolze *et al*., 2000; Cormack, 2000). Nitrogen fertiliser is the dominant energy input. However, the production process became more efficient through the last century and the energy cost has fallen progressively (Hulsbergen & Kalk, 2001) so care must be taken in interpretation of older studies. Hulsbergen & Kalk (2001), reviewing the literature, concluded that energy cost fell from 190 to 574 MJ/kg NH_3 at the start of the 20th Century, to 63 MJ/kg NH_3 in the 1940s. The decline since then has been slower; to around 38 MJ/kg NH_3 by 1990. Hulsbergen & Kalk (2001) believe that the theoretical minimum efficiency should be 23 MJ/kg NH_3. The energy cost of fertiliser is the dominant energy cost in conventional systems, as much as the sum of all other indirect energy costs (Alföldi & Niggli, 1994; Cormack, 2000). For conventional winter wheat, Cormack (2000) calculated total indirect energy inputs from sowing to drying the harvested grain as 22,519 MJ/ha, of which fertiliser accounted for 11, 512 MJ/ha. The absence of this energy cost in organic systems is the single most important factor affecting total system energy inputs and efficiencies.

Energy efficiency

In most organic systems, the yield of crop and animal products is less than in conventional systems (Stockdale *et al*., 2001; Alföldi & Niggli, 1994). However, the size of the difference will depend on many factors including the farm type, soil type, climate, and the intensity of production (i.e. the level of mineral nitrogen application) in the conventional comparison. Stockdale *et al*. (2001) concluded that, in Europe, yield of arable crops was from 20 to 40% lower in organic systems and yield of horticultural crops could be as low as 50% of conventional. Grass and forage production was between 0 and 30% lower. As a result, when calculating the energy input in terms of unit physical output, the advantage to organic systems was generally reduced, but in most cases that advantage was retained.

A range of studies, mainly in Germany, reviewed by Stolze *et al*. (2000) showed between 21 to 43% less energy input per tonne of wheat grain grown in organic systems. These studies were concerned with the individual wheat crops and appear not to have included energy inputs to catch crops or fertility building crops in the rotation. Pimentel *et al*. (1983) recorded 29 to 70 % greater energy efficiency in wheat and corn crops and Halberg *et al*. (1994) recorded consistently greater energy efficiency in spring cereals and grass/clover. The data for potatoes are less clear. Stolze *et al*. 2000 review three studies, one of which showed 19% less energy input per unit of yield, but two others showed a 7% and 29% greater input than conventional. They considered that this was due both to a higher direct and indirect energy input for increased mechanical weed control, and to relatively modest mineral nitrogen applications to the conventional crops. Pimentel *et al*. (1983) also reported lower energy efficiency in organic potatoes and ascribed it to reduced yield due to insect and disease attacks that could not be controlled in the organic system, highlighting the difficulties of comparison studies. Conversely, a modelled system using typical crop yields and inputs showed a lower energy input per unit output for potatoes in the UK (Cormack, 2000). In drawing conclusions from comparison studies, levels of inputs into the systems, and saleable yields, must be representative of the place and time.

Fewer studies have been made on livestock. Stolze *et al.* (2000) quoted two studies that reported lower energy input on organic compared with conventional dairy farms, both per farm and per unit weight of milk produced. In the same paper, they quoted a Swedish study that calculated a lower energy input on organic dairy and beef farms compared with conventional equivalents. Refsgaard *et al.* (1998) also reported lower energy input per unit weight of milk produced. Cormack (2000) showed greater energy efficiency from organic in modelled dairy and upland livestock systems. Both Refsgaard *et al.* (1998) and Cormack (2000) commented on the sensitivity of the calculation to the proportion of purchased feeds, which have a greater energy cost, to the balance.

3.11.3. Whole-farm studies

Studies of individual enterprises are useful but, in practice, the mix of crop and livestock enterprises will differ between organic and conventional farms so the analysis should take account of overall farm energy balance. Also, account should be taken of activities that are not crop-specific. This includes the handling and application of manures, rotational applications of fertilisers (e.g. rock phosphate), winter catch crops and the use of fallows for weed control. However, apart from a few studies based on long-term rotation experiments (Alföldi *et al.*, 1995; Hulsbergen & Kalk, 2001) these energy inputs have not been considered. The growing in stockless systems of fertility-building legumes which incur an energy input but have no directly harvested output should also to be considered. Using actual physical input and output data from a long-term field-scale experiment, and energy values from the literature, Cormack (2000) showed that the overall energy efficiency of a stockless arable rotation was less than a conventional equivalent because of the inclusion of non-harvested fertility building crops. However, all other modelled crop and livestock farm systems showed greater energy efficiency from organic methods. Nguyen & Haynes (1995) compared three pairs of mixed sheep and arable farms in New Zealand. They found little difference in overall energy efficiency but they noted that the conventional farms relied more on legumes for nitrogen supply, used less nitrogen fertiliser, and so were more energy efficient than European equivalents. Smolik *et al.* (1995) compared conventional, minimum tillage and alternative (equivalent to organic) systems of growing soya, wheat and barley over seven years in South Dakota USA. Overall, the alternative system had the greatest energy efficiency. The minimum tillage system had the lowest efficiency; reduced direct energy input as tractor fuel was more than balanced by increased fertiliser and herbicide energy input.

3.12. Controlled Wastes

3.12.1. Introduction

National legislation for the disposal and recovery of waste stems from the EC Waste Framework Directive, principally in the form of Part II of the Environmental Protection Act 1990. Wastes from agricultural premises have traditionally been excluded from these controls, but this exclusion is inconsistent with the Waste Framework Directive. It is therefore likely that, pending a Government review, disposal of agricultural wastes will undergo tighter regulation.

The total quantity of non-natural waste is estimated as 500,000 tonnes per year of which approximately 225,000 tonnes are pesticide washings and spent sheep dips (Anon., 2003g). This compares with 245,000 tonnes of animal by-products and some 95 million tonnes of slurry and farmyard manure (though slurry/manure is not considered as a 'waste'). The main non-natural wastes include packaging, non-packaging plastics (e.g. silage and horticultural films); agrochemicals; animal health products (e.g. used syringes); waste from machinery (e.g. oil, tyres and batteries) and building waste (e.g. asbestos sheeting). Farmers use a variety of recovery and disposal methods, depending on circumstances. They include reuse on farm, take back by suppliers, inclusion with household waste, stockpiling, and the most common are burial or burning (especially packaging and plastic films). Environment Agency research indicates that some two thirds of all wastes are buried or burnt on farm (Anon., 2003g).

3.12.2. Comparison of farming systems

Organic farming principles and practice place a strong emphasis on recycling. This particularly applies to animal and crop residues, which maintain soil fertility on an organic farm. As organic farming systems rely less on external inputs, waste materials within organic farming systems are potentially less than for a conventional counterpart. There are less packaging materials from agrochemicals, fertilisers and pesticides that require disposal. There may be some disposal of, for example, horticultural plastic and silage wrap. There is also little requirement for disposal of pesticide tank washings. Based on these observations, it is concluded that organic farming systems produce less controlled waste than conventionally managed farms.

3.13. Nutrient Use and Balance

3.13.2. Nutrient balances

These were discussed in detail in Section 1.6. It was concluded that nutrient surpluses are smaller for organic than conventional farms, when comparing the same farm types.

3.13.3. Fertiliser use

Table 3.5 summarises the list of fertilisers permitted under the UKROFS standards, though this list is continually reviewed and adapted by individual Certifying Bodies. The emphasis is on recycling and use of non-synthetic sources, hence the heavy reliance of fertility building upon the use of clover and manures/composts.

There are continual debates about the sustainability of some fertiliser materials and, for, example, the relative merits of using mined rock from a long distance away compared with a nearby 'less natural' product. Renner & Jones (2002) provide a useful summary of most fertiliser materials and their relative scores in terms of many factors, including sustainability. Also, it should be noted that many of the materials in Table 3.5 have a restricted use and may only be used as a last resort.

It is necessary to continually critically review the list of products and assess whether these are truly sustainable. This applies also to new materials that might arise.

Table 3.5. Summary list of fertilisers/soil amendments for UK organic farms (Anon., 2001a). Individual Certifying Bodies may have further restrictions.

• FYM – fresh/stacked or composted • Slurry – aerated or on-aerated • Green waste composts	All permitted. Restricted use if the material is from a non-organic source.
• Sewage sludge	Prohibited.
• Peat	Use limited to horticulture (market gardening, floriculture, arboriculture, nursery).
• Mushroom culture wastes	The initial composition of the substrate must be limited to products of the present list.
• Guano	Need recognised by the inspection authority or inspection body.
• Products or by-products of animal origin (meals, etc.)	Need recognised by the inspection authority or inspection body.
• Seaweeds and seaweeds products	Need recognised by the inspection authority or inspection body.
• Sawdust and wood chips	Wood not chemically treated after felling
• Composted bark	Wood not chemically treated after felling
• Wood ash	From wood not chemically treated after felling
• Soft ground rock phosphate	Permitted. Cadmium content less than or equal to 90 mg/kg of P_20_5
• Basic slag	Need recognised by the inspection authority or inspection body.
• Crude potassium salt (for instance: kainit, sylvinite, etc.)	Need recognised by the inspection authority or inspection body.
• Potassium sulphate possibly containing magnesium salt	Need recognised by the inspection authority or inspection body.
• Stillage and stillage extract	Ammonium stillage excluded
• Calcium carbonate of natural origin (for instance: chalk, marl, ground limestone)	Permitted
• Magnesium and calcium carbonate of natural origin (e.g. magnesian chalk, ground magnesium limestone, etc.)	Need recognised by the inspection authority or inspection body
• Magnesium sulphate (e.g. kieserite)	Permitted
• Calcium sulphate (gypsum)	Only of natural origin. Permitted.
• Industrial lime from sugar production	Need recognised by the inspection authority or inspection body.
• Trace elements	Need recognised by the inspection authority or inspection body.
• Sodium Chloride	Only mined salt. Need recognised by the inspection authority or inspection body.
• Stone meal	Permitted

4. DISCUSSION AND CONCLUSIONS

4.1. Introduction

There have now been several, independently published, comprehensive assessments of the effects of organic farming on the wider environment. Together, these syntheses of the literature, often encompassing different literature, enable a robust assessment of the effects of organic farming on many environmental parameters. Most of these reviews have been undertaken to inform their respective national debates about the value of organic farming to the wider environment: Stolze *et al*., 2000 (EU); Condron *et al*., 2000 (New Zealand); Hansen *et al*., 2001 (Denmark); Stockdale *et al*., 2001. They have all also generally chosen the same indicators of environmental benefit.

Our review has drawn on these reviews, as well as using other scientific evidence. Much was also drawn from recent MAFF/Defra funded reviews on various aspects of organic farming, most notably biodiversity (Gardner & Brown, 1998), manures (Shepherd *et al*., 1999), soils (Anon., 2002h) and organic farming generally (Unwin *et al*., 1995).

4.2. Biodiversity

Biodiversity of soil borne organisms has already been considered under soil quality. Here, we have considered effects on flora and fauna at a range of scales. The general conclusion is that organic farming benefits biodiversity. Some of the potential causes for the biodiversity benefits of organic farming include:

- Organic standards require the sympathetic management of wildlife-rich infrastructure features, such as hedges, and ditches. These features also play a role for the organic farmer, providing reservoirs for the predators of crop pests as part of the integrated pest control strategies practised on organic farms.
- A higher proportion of organic lowland farms is in mixed farming.
- Use of synthetic fertilisers, agrochemicals and veterinary medicines is prohibited or much restricted, which removes direct and indirect problems for wildlife.
- Greater variety of crop structure because of more spring cropping in more varied rotations.
- Organic farms often use undersowing**,** such as with stubble turnips with the land then used for autumn grazing. This can produce attractive over-winter habitat for seed eating birds and helps boost populations of some farmland invertebrates.
- Existing unimproved grassland is protected under organic standards (although legislation on Environmental Impact Assessment gives protection to uncultivated land generally).
- Stocking densities are limited by productive capacity underpinned by the Organic Standards and so tend to be lower in organic systems. The lower density can be an advantage when grazing sensitive habitats. Different species of livestock are more often maintained on organic farms. This helps to control parasite burdens and has advantages in maintaining structurally diverse swards.

Conclusions – Biodiversity: Comparative reviews of the evidence base have been conducted for MAFF, English Nature, The European Commission and the Soil Association. The general conclusion is that on average there is a positive benefit to wildlife conservation on organic farms. In most studies organic agriculture provides a conservation benefit, whereas there are few studies where a disbenefit is shown. While some of these practices are used on some conventional farms it is only generally on organic farms where most of the relevant management is routinely and systematically carried out. Although, the evidence from several studies shows that birds do better on organic farms overall, there are some detrimental actions in organic farming, such as mechanical weeding or mulching operations taking place between April and July. If these practices were to intensify in the future they could reduce the overall benefits for ground-nesting birds. Both organic and conventional farms will perform better when under agri-environmental schemes.

4.3. Soil Quality

Soil organic matter benefits many aspects of soil quality. This has long been recognised by both organic and conventional farmers. Within soil textural constraints, soil organic matter levels will increase with greater organic matter inputs to the soil. There is evidence that soil organic matter contents will increase under organic farming. However, many conventional systems also encourage a build-up of organic matter through regular manure applications and returns of large amounts of crop residues etc. Due to the production systems, there may also be fewer differences in organic matter levels between conventional and organic grassland. Stockdale *et al.* (2001) stated that changes in organic matter drive/underpin many of the other changes in soil biological and physical properties. Our review has clearly demonstrated this.

For soil structure, we conclude that there is a large body of evidence to show that organic farms exhibit at least as good and generally better soil physical conditions than conventionally managed soils. Although SOM is often implicated in differences in soil physical properties the soil structure would be the result of all practices (SOM, rotational and tillage practices).

The evidence also tends to support the hypothesis that earthworm populations are more active in organic farming systems than those conventional systems with a great reliance on inorganic fertilisers and pesticides. Small populations of earthworms have been linked to lack of adequate moisture in the soil surface, intensive pesticide use, frequent tillage, and absence of ground cover. Organic management practices try to minimise these effects and are therefore more likely to encourage active earthworm populations.

Generally, organic farming practices have also been reported to have a positive effect on soil microbial numbers, processes and activities. Much of the cited literature has made direct comparisons between organic and/or biodynamic and conventionally managed soils. The evidence generally supports the view of greater microbial population size, diversity and activity, and benefits to other soil organisms too. However, little is currently known about the influence of changes in biomass size/activity/diversity on soil processes and rates of processes. Nor is it possible to conclude that all organic farming practices have beneficial effects and conventional practices negative effects. Pasture is the main element of agricultural systems where least difference would be likely to be seen in soil quality between organic and conventional systems, since both will accumulate organic matter. The majority of literature showing no benefit to microbial activity from organic systems is found in studies of pasture. In the few arable comparisons where lack of differences or greater activity in conventional systems were found, this might be related to greater residue returns in the conventionally fertilised systems. If so, this provides a pointer to the key factor that differentiates between conventional and organic systems as being return of organic matter.

Conclusion – Soil quality: There are few UK studies on the relative benefits of organic or conventional systems for soil quality. However, such studies as have been done and those from other countries tend to show benefits for organic systems. Organic farmers pay particular attention to their soils, and it is a fundamental tenet of organic farming to operate a sound rotational system to 'feed the soil' to maintain organic matter content and to keep it in good condition. However, organic matter additions are also made in conventional agriculture and, in some situations, the return may be similar or greater than in organic systems. Soil structure can benefit from regular returns of organic matter, and the evidence is that soil structure is at least as good and generally better under organic practices. Earthworm numbers tend to be greater in organic systems and studies into the microbial response of soils to organic management indicate there are benefits in many but not all situations and not always in all the attributes measured. The absence of soluble nutrients, most pesticides and reduced use of veterinary medicines such as antibiotics and ivermectins can also be expected to benefit soil organisms.

4.4. Water Quality

4.4.1. Nitrate

Nitrogen is difficult to manage and control in any farming system given its mobility in soils as nitrate and the huge amount of potentially oxidisable organic nitrogen in soils. Losses depend on many factors, not all of which are under the control of the farmer. Weather plays an important role. Practices that minimise risk of loss must be adopted, and it must be recognised that it is impossible to avoid some loss. Since nitrogen is often the limiting nutrient in organic systems and is expensive to replace, it seems sensible that growers aim to avoid losing as much as possible to the wider environment.

Organic farming aims to adopt many of the practices that should minimise loss – maximising green cover (leys, cover crops), use of straw-based manures or compost applications, lower stocking rates. Therefore, it might be expected that nitrate losses would be less than from conventional systems. The evidence, on balance, supports this. However, it must be said that there are few comprehensive studies making the comparison. Under UK conditions, the recent study of Stopes *et al*. (2002) perhaps provides the best evidence. However, even this study tended to compare organic and conventional farms at the same levels of intensity, i.e. low intensity conventional systems. It is known that nitrate losses are even greater from the more common highly intensive conventional farms and so it could be argued that the differential would be larger.

Much emphasis is always placed on the ley ploughing phase. Indeed, nitrate losses can be large after autumn ploughing and further research needs to examine other options. However, because we are discussing a farming system, nitrate losses from the whole rotation need to be considered, not just this one aspect of the system. Because organic systems operate at a lower level of N input, losses are generally less – but this is not always guaranteed.

Conclusion – Nitrate in water: Variation in leaching losses from individual fields is large both in organic and conventional agriculture. Many organic systems operate at a lower level of nitrogen intensity than conventional systems, with nitrogen inputs from fixation by legumes, or from importation of animal feed onto the farm. Organic farming adopts many of the practices that should decrease losses: maximising periods of green cover, use of straw-based manures, lower stocking densities. The body of evidence suggests that leaching losses are generally less from organic systems – though this is not always guaranteed. Losses after ploughing the fertility building leys are one area where losses can be especially large. It might also be argued that this differential will decline as conventional fertiliser practices improve under the increasing regulatory pressure.

4.4.2. *Phosphorus*

The transport processes for P transfer from soil to water are complex. Surface run-off, soil erosion and sub-surface flow are the most common routes. Under UK conditions, downward leaching of P is not a primary route unless the soil P status has been elevated to extreme levels. Because of the complexity of the transport mechanisms, P loss is not necessarily related to P surplus. Factors that encourage infiltration of water and avoid surface run-off and erosion will probably decrease P losses. However, there is no work that has directly compared losses from organic and conventional farming. Information to date is therefore inconclusive.

Conclusion – Phosphorus in water: The main loss pathway for phosphorus is by movement of soil particles. Leaching is a smaller and more site-limited effect. There are some additional "incidental" losses following the application of fertilisers or manures. There is no direct evidence of differences in phosphorus losses between organic and conventional agriculture.

4.4.3. *Pesticides*

An assessment of pesticide pollution risk from organic farming is straightforward because only a few are permitted for use under restricted conditions.

Conclusion – Pesticide pollution to water (and air): Pesticide use in organic farming is very restricted. A small number of pesticides are approved for organic use (principally copper, sulphur, natural pyrethroids, and derris). They have restrictions on their use, and can only be used as a last resort. The pyrethroids, copper and derris are only permitted for use in protected cropping or for a restricted range of horticultural crops. With the exception of sulphur, on certain top fruit crops and pyrethroid sheep dip (which can be used in the same way on both organic and conventional farms), the use of the restricted range of pesticides is very limited by comparison with conventional agriculture. In particular, organic farmers do not use herbicides, some of which (such as isoproturon) have presented particular water pollution problems. Pesticide pollution from organic farming will be far less common than pesticide pollution from conventional agriculture. These differences are likely to hold whether assessed per area, or per unit of food produced.

4.4.4. *Human Pathogens*

The application of organic manures is a potential mechanism for transferring pathogens into the food chain, either by directly contaminating crops or by contaminating water. This is currently an area of intensive research, mainly because data have been lacking to date. There have been no comparisons of the effects of organic and conventional farming. Manure storage methods can influence pathogen survival. Composting will increase kill, but current research projects are not sufficiently advanced to draw firm conclusions.

Conclusion – Human Pathogens: Pathogenic organisms from livestock can contaminate surface waters used for drinking, bathing or irrigation. There is no reliable information on any differences in the incidence of zoonoses between organic and conventional farms although there is on-going research. Studies have shown that composting manures and treating slurries as encouraged under organic standards decrease the survival of any pathogenic organisms but stacking or long-term storage can also be beneficial. The methods of handling manures between farming systems may not be sufficiently different to produce a consistent effect and therefore information on the incidence the organisms is needed before any conclusions can be drawn.

4.5. Air quality

4.5.1. Ammonia

The main source of ammonia from organic farming is manures. An additional source from conventional agriculture may be losses from urea fertiliser, if this is used. However, manure is the major source from agriculture. Many factors affect ammonia loss – diet (amount of N excreted), housing, storage and landspreading. Because ammonia losses occur right through the animal production system, ammonia saved, for example, during housing might be susceptible to loss during manure storage and/or spreading (unless it is immobilised into non-ammoniacal forms). This whole needs to be considered when comparing ammonia losses from different production systems.

Conclusions – Ammonia: Ammonia is mainly lost from the surface of manures, either from animal buildings or hardstandings, which are soiled by manures, or during storage and handling. Manures produced in organic systems often have a lower concentration of nitrogen than do conventionally produced manures. Organic systems encourage the composting of manures, which leads to a relatively high loss of ammonia, although this will reduce the amount emitted when the compost is subsequently spread. Given the constraints on housing and stocking rate it is not possible to have intensive pig and poultry organic units, which are a major source of ammonia from conventional systems. Organic pigs and poultry will have similar losses to conventional outdoor units at the same stocking densities. It seems likely that on balance there is little difference between organic and conventional systems in the amount of ammonia which is lost from the system per unit of yield, but it is likely that emissions are lower per unit area. Given that nitrogen is more valuable to organic systems than it is to conventional systems (which can purchase nitrogen fertiliser at about 30p per kilogram), there should be a greater incentive for organic farmers to control ammonia losses in the future.

4.5.2. Nitrous oxide

There are major methodological problems in measuring nitrous oxide emissions from soils, mainly because of the size of the emissions and their intermittent nature. Consequently, there has been no comparative study of emissions from organic and conventional systems. One of the sources – fertiliser N – will not occur from organic systems, so the main organic practices that influence loss will be manure management and soil management.

Conclusions – Nitrous oxide: Nitrous oxide is emitted from manures and from soils. Emission tends to occur intermittently when there is a combination of the appropriate conditions. Within conventional agriculture, the main risks arise from manures and from the waterlogging of soils by heavy rainfall following fertiliser application. Within organic farming the risks are likely to come from manures and from waterlogging of soils where there is a legume crop. In the absence of direct measurement, it is not possible to assess whether there is any difference in risk from organic or conventional production.

4.5.3. Methane

Nearly all methane emissions from agriculture are related to ruminant livestock production Comparative data for organic and conventional production systems are limited. We therefore have to draw conclusions on methane emissions from the three main factors that affect emissions: livestock numbers, diet and productivity.

Conclusions – Methane: About 75% of methane on farms is emitted directly from ruminant animals (chiefly cattle and sheep). There have been no direct comparisons of methane generation between organic and conventional production. Different types of fodder will generate different amounts of methane, with higher rates released from diets that are high in roughage relative to diets high in starch. This will tend to result in higher emissions from organic systems, as organic diets tend to be high in roughage and low in concentrates. Methane emission per unit of livestock product decreases as the intensity of animal production increases (two cows producing 5,000l of milk will generate more methane than one cow producing 10,000l of milk). On average, production intensity is lower in organic than conventional systems, so methane generation from organic farms is likely to be greater per unit of food produced. Because of the lower stocking densities, it maybe similar on an area basis.

4.5.4. Carbon Dioxide

Although agriculture can be both a sink for and source of CO_2, most of the literature has focused on CO_2 emissions. The likelihood of organic farming increasing carbon sequestration in soils is small, even though organic farming practices encourage an increase in organic matter (manure applications, minimising bare soil, cover crops, etc.). This is because the size of the organic matter increase is small, and is not consistent across farms, depending on the carbon balance of individual farms with widely differing practices.

For CO_2 emissions, the number of comparative studies is few. The limited evidence is in favour of decreased emissions of CO_2 when comparing organic with conventional systems on an area basis, but the evidence is less convincing when comparing on a unit production basis. Much also depends where the boundary of the study is drawn.

Conclusions – Carbon dioxide: Net emissions of carbon dioxide from agriculture depend upon use of fossil fuel and the amount of carbon sequestration in soil organic matter. Emission from fossil fuel use will be lower on a per area and a per yield basis, reflecting the greater energy efficiency of organic agriculture noted below. There is insufficient evidence on whether there is a significant difference in the amounts of carbon sequestered in soils.

4.6. Resource Use

4.6.1. Energy efficiency

The review of the current literature showed that organic lowland livestock systems tend to have lower energy use than conventional lowland livestock systems. For extensive upland livestock systems, the energy uses are more similar although, on average, organic production uses somewhat less. Some of the differences in energy ratio were large. Organic arable production used 35% and organic dairy 74% less energy than conventional per unit of product.

Conclusions – Energy efficiency: The literature supports the statement that organic methods generally use less energy per unit area and per unit of output, both for individual crops and livestock types, and overall on a whole-farm basis. However, the setting of system boundaries, methods of calculating the energy values of inputs and methods of calculating energy use efficiencies vary substantially between studies. The intensity of production in the conventional comparison, particularly in relation to the level of use of mineral nitrogen fertiliser, also had a large impact on the relative performance of organic methods in comparative studies. This makes comparisons across studies difficult; there is a need for an agreed standard methodology. Information is lacking for non-ruminant livestock

4.6.2. Nutrient use and balance

Calculation of farm gate and soil surface balances is becoming an increasingly popular tool for judging the sustainability of a farming system. There are no hard and fast guidelines for the optimum size of any surplus to judge sustainability, but they provide an indication of whether a system will deplete soil reserves in the long-term (and therefore be deemed unsustainable). A large surplus may also indicate the potential for large losses, though the relationship between surplus and loss to the wider environment is not straightforward, nor proven.

Conclusions – Nutrient balance and use: Comparisons of nutrient budgets suggests that the balances can vary widely within a farming system. However, the general conclusion is that organic systems operate smaller nutrient surpluses. This is taken as an advantage, providing that nutrient reserves are not being depleted. Prohibition of various fertiliser additions is on the basis of encouraging self-sufficiency in a system, but there is a need to continually review the lists of allowed and disallowed products to ensure that choices are environmentally sound.

4.6.3. Controlled wastes

Organic farming focuses on recycling and on minimising external inputs. Thus the likelihood of needing to deal with controlled wastes when practising organic principles is small.

Conclusions – Controlled wastes: Waste is generally lower in organic farming since the system relies less on external inputs. Packaging materials for agrochemicals, veterinary medicine, animal feed, and fertilisers should all be lower on organic holdings. There is also little need for disposal of pesticide washings on organic systems.

4.7. Overall Conclusions

The general conclusion from our review concurs with that from other reviews, i.e. organic farming can deliver positive environmental benefits. However, this statement needs to be covered by several caveats:

- Organic farming does not automatically deliver all of these benefits. Clearly, where regulations control the management activities (e.g. no herbicide applications), environmental benefits are delivered. However, for other aspects, benefit will depend very much on the individual farmer, as does the impact of conventional farming. Here, soil quality improvement is a good example. Organic matter build-up can occur on a conventional farm if the farmer has access to animal manures and they are applied regularly (in accordance with codes of practice). The benefit here may be greater than on an organic stockless farm with limited or no access to manures. It is therefore important to bear in mind that there is a continuum of farming systems even within ‘organic’ and ‘conventional’ classifications.

- The outcome of any comparison depends on the type of farms being compared. We have already stated that ‘organic’ is legally defined, whereas ‘conventional’ is not. The tendency with some of the reported research is also to compare organic systems with conventional systems at similar levels of production. However, it is the higher intensity of some conventional systems that can lead to most problems. It would be more appropriate to compare organic with ‘typical’ intensive systems if this is what a switch to organic would replace. This is most likely to be the case in lowland agriculture. There are likely to be fewer differences between conventional and organic extensive upland livestock production systems.
- For some impacts (e.g. gaseous emissions), the potential benefit depends on the basis of comparison, i.e. on area or unit of production. This is important, and not easy to interpret.

We have summarised our assessments in Table 4.1, assuming lowland agriculture and comparing organic with moderately intensive conventional systems.

Finally, there are some other considerations that need to be borne in mind for this exercise:

- The effects of scale of converted areas are unknown. Larger areas of contiguous organically farmed land could result in greater or, possibly, lesser environmental benefits than the conversion of individual farms.
- The implications at the macro-scale if a large proportion of agricultural land was converted to organic are uncertain. Organic systems tend to produce lower yields than conventional systems, and have a higher proportion of land occupied by animals, whereas many conventional livestock systems have a greater reliance on feed produced off-farm. This could lead to differences in food imports and in the balance of land-use within the country. It is not clear what the implication of these macro changes would be for the environment.

Table 4.1. Summary of the environmental impact of organic farming (compared with conventional farming)

	Indicator	Assessment of impact		Comments
		Per unit area	Per unit yield	
Ecosystem	Biodiversity	☺	☺	Organic principles encourage a wide variety of habitats.
Soil Quality	Organic matter content	☺/😐	☺/😐	Potential benefits from organic farming, depends on organic matter inputs on individual organic and conventional farms.
	Biology	☺/😐	☺/😐	Literature tends to support a benefit, <u>but not always</u>.
	Structure	☺/😐	☺/😐	Literature tends to support a benefit, <u>but not always</u>.
	Erosion susceptibility	☺/😐	☺/😐	Few direct measurements, but organic practices should decrease risk.
Water Quality	Nitrate leaching	☺	☺/😐	Potentially large losses from ploughed leys, but smaller losses, on average, from other points in the rotation.
	Phosphorus loss	😐	😐	Insufficient information.
	Pesticides	☺	☺	Few pesticides used in organic production.
	Human pathogens	😐	😐	Insufficient information – work ongoing.
Air Quality	Ammonia	☺	😐	No direct studies. Assessed from what is known about processes.
	Nitrous oxide	😐	😐	Insufficient information.
	Methane	☺	☹	Most data relate to dairy systems. Lower emissions on an area basis due to lower livestock densities.
	Carbon dioxide	☺	☺	Main energy input relates to fertiliser manufacture
Resource use	Energy efficiency	☺	☺	Depends where boundaries are drawn when comparing systems, but main energy input into conventional is fertiliser production.
	Nutrient balance	☺	☺/😐	Smaller surpluses: OK if not over-depleting soil fertility.
	Controlled wastes	☺	☺	Emphasis on recycling. Less packaging and no agrochemical waste.

Key:

☺ Organic is better than conventional

😐 No difference between organic and conventional

☹ Organic is worse than conventional

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