This review has been necessary to bring together and interpret much of the diverse literature relating to the key areas of environmental impact associated with agriculture. The main environmental impacts of agriculture in the United Kingdom are associated with pesticides, nitrogen compounds, farm livestock wastes and soil erosion. Impacts on fauna and flora, water bodies and humans, as well as the economic costs (e.g. those involved in reducing levels of pollutants in water bodies) are considered within this paper. Agricultural activities, particularly those associated with emissions to the air and water, can have significant effects many kilometres from their place of origin. There may also be inter-related impacts. For example, pollution incidents which directly affect aquatic populations can indirectly influence recreational activities, and may have economic effects in the form of remedial water treatment and clean-up costs. It is important that management techniques and systems are developed to reduce detrimental environmental impacts associated with agriculture. The information outlined in this paper has been used to assist in the design of an integrated computer-based system to support environmental management in agriculture.

Keywords: environmental impact, agriculture, pesticides, nitrates, human impacts, livestock wastes, erosion, odour.

1. Introduction

According to Park (1988), up to the late 1970s and early 1980s U.K. agriculture had remained relatively free of environmental pressures and issues. In spite of environmental
damage in earlier decades, for example by DDT (Cremlyn, 1991), it was generally believed that farming was a benign function of the countryside. There were, therefore, few constraints on agricultural units, and planning constraints in particular were relatively unrestrictive (Park, 1988). However, with increasing environmental legislation and mounting popular concern for the environment, the importance of good environmental management has now been recognised. Although the agricultural industry has been slow to embrace these challenges, a voluntary move towards better environmental management in agriculture is emerging. Codes of Good Agricultural Practice for the Protection of Water, Air and Soil have been published by the Ministry of Agriculture, Fisheries and Food (Anon, 1991, 1992a, 1993a). There has also been a recent increase in research into the environmental impacts of agricultural activities and the development of focused environmental management techniques in agriculture (Allan, 1995; Freeland, 1995). This review has been necessary to bring together and interpret much of the diverse literature relating to the key areas of environmental impact associated with agriculture. Research in the 1970s and 1980s still provides a useful foundation for work in this field (Loehr, 1977; Canter, 1986), and, along with more recent literature, has been used to identify the areas of concern in the U.K. today. A scoping exercise (Wathern, 1988) was used to identify the principal impact areas associated with U.K. agriculture. This review is based on that exercise and focuses on farm pollution impacts, highlighting the following areas:

(i) pesticides;
(ii) nitrogen compounds;
(iii) farm livestock wastes; and,
(iv) soil erosion and associated nutrient losses.

The principle impact areas identified above can arise anywhere in the U.K., but their significance will vary, for example between uplands and lowlands. However, analysis of these variations would require further detailed study. Environmental impacts are not merely those that can be measured chemically or biochemically. There are also visual/aesthetic impacts such as the appearance of landscape, and socio-economic considerations, such as implications for rural communities. These are more difficult to quantify, and beyond the scope of this paper.

2. Pesticides

Research into pesticides (which include herbicides, insecticides and fungicides) has concentrated on three main areas: contamination of waters, impacts on fauna and flora and human impacts. The factors governing environmental impacts of pesticides are the type of chemical used, how much is applied (which partly depends on the accuracy of the machinery), where the spray lands and the weather conditions (Anon, 1993b).

2.1. Contamination of Waters

Water pollution, particularly by herbicides, has been recognised as a problem since the 1970s (Matthiessen et al., 1992). One area of concern is the extent to which pesticides reach rivers and lakes by leaching and runoff, since this may lead to impacts on aquatic life, and on humans if contamination extends to drinking waters. These concerns have been addressed by the EC Drinking Water Directive (Anon, 1980), which stipulates a maximum concentration of any pesticide in potable waters of 0.1 μg l⁻¹. Whilst the
maximum level is considered to be over-cautious for many pesticides, the large number of water samples that fail to comply have caused much public concern (Matthiessen et al., 1992). For example, atrazine, simazine, mecoprop, dimethoate and, more recently, isoproturon and chlortoluron have been found frequently in surface waters at concentrations of $0.5 \mu g l^{-1}$, with a maximum of $1.5 \mu g l^{-1}$ (Croll, 1991). Being highly soluble and persistent, atrazine (Anon, 1992b,c; Ivens, 1995; Steele, 1995; Williams et al., 1995) is the pesticide most frequently exceeding the $0.1 \mu g l^{-1}$ limit in water. For the years 1992 to 1994, the water companies spent almost £500 M on pesticide removal from water to meet the EC Drinking Water Directive. Severn Trent Water Authority, for example, has implemented a £100 M scheme for pesticide treatment plants on lowland surface water abstraction points. The running costs for these are estimated to be £8–10 M per year (Steele, 1995).

The pesticide monitoring program started by Anglian Water in 1985 (Croll, 1991) has included rivers, reservoirs and groundwaters. With the exception of the triazines (atrazine and simazine), the pesticides detected are those used in the greatest tonnages in U.K. agriculture. The sources of triazine pesticides are difficult to establish, due to the problems in obtaining information about non-agricultural uses, such as maintenance of railways and roads (Anon, 1992c; Gomme et al., 1992). Pesticide levels were generally found to be much higher in surface waters than groundwaters. This is probably because sources included surface and drainflow runoff from farmland, as well as spray drift contamination, where soil contact and therefore degradation is reduced (Anon, 1993b). Point-source pollution of surface water by pesticides was estimated to form approximately 2% (50 point source incidents) of all farm incidents (2828) reported in 1984 (Anon, 1986); this figure has been rising to give a total of 70 point source pollution incidents involving pesticides in 1993, of which 16 involved sheep dip (Anon, 1994a).

Some pesticides are strongly adsorbed or degraded in soils. Laboratory experiments have shown that isoproturon is about five times more strongly adsorbed on soil than mecoprop (Nicholls et al., 1993). Isoproturon can therefore appear in drain water, especially when there is little contact between percolating water and the soil, as in clay soils with large desiccation cracks (Williams et al., 1995). Other factors that can limit leaching losses include degradation by soil microfauna. Degradation to less harmful compounds increases with rising soil temperatures and moisture contents (Nicholls et al., 1993). Hence, environmental persistence is compound specific, and soil type and weather dependent.

Research into the efficiency of herbicide applications at Brimstone Farm, Oxfordshire (Harris et al., 1992), showed that in gentle wind ($3–6 \text{ m s}^{-1}$) 87–93% of the spray was deposited on the target area, 2–3% on the soil outside the target area, 1–4% of this by drift up to 8 m downwind and the remainder, up to 10%, was lost by volatilization or further spray drift. Rainwater samples from the Brimstone site often contained pesticides such as atrazine, lindane and aldrin, which had not been applied to the plots since the beginning of the experiment in 1978 and must have come in as spray drift from external sources (Harris et al., 1992). Other pesticides found in the rainwater at Brimstone included simazine and isoproturon, which had both been used on the experiment. Half of the rainwater samples analysed contained amounts of at least one pesticide in excess of $0.1 \mu g l^{-1}$. For dry deposition the most abundant pesticides were those with high vapour pressures, e.g. lindane and aldrin. It was suggested that major atmospheric movements can bring compounds to the site from far afield (Harris et al., 1992).

Because of the nature of the compounds involved and the complexity of natural pathways of water, it is usually difficult to be sure of the sources of pesticide contaminants
and the ways by which they enter ground and surface waters. For example, Gomme et al. (1991, 1992) analysed surface and groundwaters from a Chalk catchment in the Anglian Water area for 20 different pesticides between 1987 and 1990. The catchment was dominated by arable agriculture, particularly cereal production. Concentrations of both atrazine and simazine in the groundwater were frequently less than other pesticides, and could have come from agriculture or other sources. Two uron herbicides, isoproturon and chlortoluron, were found in boreholes close to seasonal streams, but the flow mechanisms enabling these pesticides to reach the ground water were not established. The surface water study (Gomme et al., 1991) showed that concentrations generally increased with river flow rate. However, even at the lowest flow rates some pesticides were detected, including isoproturon, which had the highest agricultural use of the 20 pesticides identified. Results from the Boxworth Project (Anon, 1992d), which examined the environmental effects of pesticides used on cereals, showed considerable variation in the levels and persistence of pesticides in drainage water. For samples taken soon after application, herbicides were detected in amounts significantly above the EC drinking water limit, and only declined slowly. Substantial levels of simazine were detected, reaching a temporary peak of 35 μg l\(^{-1}\) 30 days after application. Residues of insecticides, which are applied at lower rates than herbicides, were almost absent.

2.2. IMPACTS ON HUMANS

Toxicity of different pesticides to humans probably varies greatly, but there is a general lack of epidemiological data on the impact of pesticides on human health (Anon, 1990a). This may be because research tends to be reactive, rather than proactive.

Pesticide uptake occurs through the skin, eyes, lungs or intestinal tract, and toxicity is influenced by the dose, type of chemical and its metabolites, the health of the individuals and their genetic susceptibility (Anon, 1990a). Adsorption through the lungs is effective for droplets smaller than 5 μm whereas larger droplets may be swallowed. Ingestion can also occur via contaminated food, eating utensils, hands or cigarettes. Following ingestion, the pesticide may be metabolised or stored in body fat. Synergistic effects can result in combinations of pesticides being more toxic than their individual components.

The World Health Organisation (Anon, 1990a) reported that unintentional occupational poisoning by pesticides from all continents approached several million cases and provided evidence that pesticides are responsible for severely affecting many aspects of human health. The extent of poisoning in the U.K. is indicated by the Health and Safety Executive, who reported 196 incidents in 1993/1994 relating to pesticides (Anon, 1994b). This compares with 226 incidents in 1992/1993 and 207 in 1991/1992. The 1993/1994 figures included 23 agricultural workers and 109 members of the public, with most incidents related to off-site drift. However, by no means all reported incidents were proved to be caused by pesticides.

Most concern from agricultural workers relates to organophosphate sheep dips (Burns, 1993; Mason, 1993a,b; Anon, 1994c; Davies, 1994). A survey by Friends of the Earth between 1988 and 1992 (Anon, 1993c) showed that over 450 farmers and workers who had handled wool were affected by exposure to organophosphate pesticides in sheep dip. Organophosphorus compounds produce a range of chronic symptoms (Savage, 1988), including those associated with acute poisoning, acute intermediate and delayed neurological effects, psychological and behavioural changes and respiratory effects (Anon, 1990a, 1994c; Burns, 1993; Mason, 1993a,b; Davies, 1994; Stephens et
al., 1995). In some cases, exposure resulted in hospitalisation and prolonged ill health. Recent research on breast cancer has identified atrazine, endosulfan and lindane, which are still in regular use in the U.K., as potential breast carcinogens (Anon, 1995a). It also identified the need for extreme caution when dealing with these chemicals directly, particularly in their concentrated forms, and emphasised the need for the precautionary approach adopted by the EC Drinking Water Directive (Anon, 1980).

2.3. FAAUNA AND FLORA

The effects of pesticide spray drift on fauna and flora have been reported by English Nature (Anon, 1993b) and for the Boxworth Experimental site (Anon, 1992d). It seems to cause measurable short-term impacts to wildlife, but the time scale and long-term effects are uncertain. For most pesticides, populations of woodmice at Boxworth decreased in the short term but there was no evidence of any long term effects. Molluscicide pellets, used for slug control, did kill most of the local populations of mice, but they recovered as migration into the field from untreated areas took place. The harmful effects were greatly reduced by drilling the molluscicide with the seed instead of surface broadcasting. Rabbit populations at Boxworth showed minor liver cell changes following field spraying with the organophosphorus insecticide demeton-S-methyl, but no evidence of long-term damage. Spray drift of demeton-S-methyl reduced catches of arthropods in an adjacent hedgerow. Rew et al. (1992) found that fertilisers and herbicides together can play a major role in the degeneration of field boundary vegetation and associated wildlife (Boatman, 1989). Other studies (Williams et al., 1995) have shown that pesticide flushes can occur at the headwaters of streams, where stream fauna could be affected. This is of particular concern because such waters are otherwise of high quality and may be fish nursery grounds.

The Advisory Committee on Pesticides (Anon, 1993d) provides statistics on pesticide incidents. In the U.K. in 1993, there were a total of 212 incidents where pesticide poisoning was identified. Deliberate abuse of pesticides to kill animals was found in 110 incidents. There were 23 further incidents of pesticide misuse, commonly resulting from careless practice. Incidents involving non-invertebrates arising from the approved use of pesticides increased from 6 in 1992 to 17 in 1993. There were 22 pesticide incidents involving honeybee deaths in 1993, with a total of 92 colonies affected by 13 different pesticides. In some cases, the cause of death was identified as pesticide used on nearby crops, although in others the poisoning was considered to be malicious. Misuse usually resulted from poor storage, chemicals not being used in the approved manner, or incorrect disposal. Chemicals in the misuse category tended to be rodenticides, molluscicides and seed treatments. Nearly half of the reported incidents involved metaldehyde slug pellets, uncleared spillages being the primary cause. Although dogs were the main species affected, a badger was also found poisoned with metaldehyde. Spillages may kill other wildlife, though their carcasses are less likely to be found (Anon, 1993d).

The Advisory Committee on Pesticides concluded that when consideration is taken of the large pesticide usage in the U.K., the small number of incidents shows that if pesticides are used in the approved manner there is a negligible risk to wildlife and other animals (Anon, 1993d). However, this could partly reflect the imperfect system for recording incidents.
3. Nitrate

Nitrate concentrations in U.K. waters increased during the 1970s and 1980s (Croll, 1990), the highest levels occurring in eastern England. The source of the additional nitrate in this region is often assumed to be arable agriculture because increased fertilizer inputs have led to increased leaching from the soils into both surface and ground waters, and because of low rainfall volumes to dilute the leachate (Croll, 1990). Whitmore et al. (1992) showed that ploughing up long established grassland, as occurred after World War II in eastern England, also increases nitrate losses.

The introduction of the EC Drinking Water Directive in 1985 led to concern that the maximum admissible concentration of 50 mg NO₃⁻ l⁻¹ would be exceeded in many groundwater abstractions. Croll (1990) estimated that nitrate concentrations in groundwaters would reach 150 to 200 mg l⁻¹ if agricultural losses remain at 1980 levels. During 1991 a total of 94 water supply zones with nitrate concentrations above the EC limit were identified (Anon, 1992b). In eastern England, nitrate concentrations in surface waters over four times the limit were measured in 1989/1990 (Harris and Rose, 1992).

In order to meet the nitrate standard for drinking water, the U.K. water industry is having to invest heavily in treatment plant and extensive distribution systems. The options available to the water industry for controlling the concentrations of nitrate in drinking water include blending with low nitrate water, source modification or replacement, extended storage to permit natural denitrification and removal of nitrate by treatment (Croll, 1990; Anon, 1992e). The total U.K. costs of achieving the 50 mg NO₃⁻ l⁻¹ standard have been estimated as £199 M over the next 20 years. Anglian Water, the most affected Water Authority, estimated their costs would be £70 M over the next 10 years (Croll, 1990).

Aquifers with nitrate levels above 50 mg l⁻¹ are generally located in eastern and southeastern England and in the West Midlands (Croll, 1990). High levels predominantly occur where there is no layer of clay between the soil and the water-bearing rock. The most severely affected aquifer is the Lincolnshire limestone where levels above 100 mg NO₃⁻ l⁻¹ have occurred and several borehole stations have consequently been taken out of public supply (Croll, 1990). Groundwater nitrate concentrations in Lincolnshire boreholes were relatively stable up to 1975, then showed a sharp increase until 1979 (Scott et al., 1992). Since then levels have been slowly declining. Analysis of agricultural practices for this area suggested that ploughing up grassland contributed more to the increases in leached nitrate for this period than did increased use of nitrogen fertilisers, though diminished rainfall was also identified as a factor contributing to increased nitrate concentrations (Scott et al., 1992).

Typically, nitrate concentrations in rivers are very seasonal, remaining low until soil field capacity is reached in early winter and flow begins from land drains. Most rivers in eastern England exceed 50 mg NO₃⁻ l⁻¹ during the winter, and although data since 1980 indicate that levels in some rivers have remained steady, analysis of long-term data indicated an upward trend (Croll, 1990). Since 1990 there has been no further increase in nitrate levels in U.K. rivers (Anon, 1996).

3.1. Human Impacts

High levels of nitrate in drinking water can cause methaemoglobinaemia (blue-baby syndrome) in bottle fed infants. The illness is actually caused by nitrite produced from nitrate in the gastro-intestine. The nitrite prevents the blood transporting oxygen,
resulting in a blue colouration of the skin (Bruning-Fan and Kaneene, 1993). There have been 14 reported cases in the U.K. since 1945 (Addiscott et al., 1991), the last being in 1972. The diets of some vegetarians may contain increased levels of nitrate because of nitrate in leafy vegetables (Dudley, 1990; Anon, 1994d). Laboratory research on animals (Wild, 1993) has suggested a link between nitrosamines, which can be produced from nitrate in the intestine, and gastric cancer. Although other studies have found no evidence that this causes cancer in humans (Addiscott et al., 1991; Wild, 1993), it is considered to be a risk factor (Clough, 1983; Bruning-Fan and Kaneene, 1993). Human impacts have been reduced by a number of measures taken to minimise exposure, such as taking highly contaminated wells out of use, water blending and treatment (Croll, 1990), and using nitrate-free bottled water for infants.

3.2. WATER CONTAMINATION

Research into nitrogen fertiliser use in agriculture has highlighted three main areas of environmental impact: water contamination, gaseous losses and effects on humans. The secondary economic costs, notably to the water industry, are also significant, although research is starting to show how good management practices can reduce losses (Anon, 1995b). The management practices are incorporated into regulations covering agriculture in Nitrate Sensitive Areas (NSAs) and management practice guidelines for Nitrate Vulnerable Zones (NVZs), where a combination of computer modelling and field sampling has shown that reductions in nitrate losses from agricultural land can be achieved (Lord, 1992; Lord et al., 1993). Nevertheless, relationships between leached nitrate and applied nitrogen, atmospheric inputs, and the production of nitrate from organic nitrogen in the soil under different land management systems are complex (Milford et al., 1993; Watson et al., 1993; Francis et al., 1995). Specific agricultural activities contribute substantially to losses of nitrate. These include the ploughing of permanent pasture, which releases large amounts of nitrate through the mineralisation of soil organic matter (Harris and Skinner, 1992; Scott et al., 1992; Watson et al., 1993; Francis et al., 1994), leaving land fallow over the winter (Addiscott et al., 1991), and application of animal manures or nitrogen fertilisers during the autumn when plant uptake is low and overwinter rainfall will increase leaching (Addiscott et al., 1991). Nitrate losses are also influenced by the type of cultivations used (Catt, 1993); for example in the Brimstone leaching experiment nitrate lost from direct drilled land was 24% less than from ploughed land. The average amount of nitrate lost from ploughed land in this experiment was 39 kg N ha$^{-1}$ yr$^{-1}$ with concentrations of nitrate in the drainflow from both ploughed and direct drilled land often exceeding the EC drinking water limit of 50 mg NO$_3$ l$^{-1}$. The first flows in autumn usually contained over 50 mg NO$_3$ l$^{-1}$, even when no fertiliser was applied to the seedbed. Concentrations subsequently decreased through the winter until spring fertiliser was applied, and then further rainfall events often resulted in elevated concentrations exceeding 50 mg NO$_3$ l$^{-1}$. From the first phase of the Brimstone experiment (1978–1988), Goss et al. (1993) concluded that nitrate losses were influenced by:

(i) soil mineral N residues present in autumn;
(ii) the amount of drainflow, which occurs mainly in winter;
(iii) the amount of fertiliser-N applied to the seedbed in the autumn;
(iv) the amount of rainfall following spring top-dressing;
(v) uptake of soil mineral-N by the autumn-sown crops which was generally small;
(vi) mineralisation of crop residues and other soil organic matter; and
(vii) the tillage regime.
The contribution of nitrate from plant residues incorporated into the soil is influenced by management practices and the carbon : nitrogen ratio of the residues. As much as 200 kg N ha⁻¹ can be released through mineralisation when permanent pasture is ploughed (Anon, 1989a). However, the addition of carbon from crop residues with a high C:N ratio, such as wheat straw, can result in temporary immobilisation of nitrogen into the soil microbial biomass (Shepherd, 1993). Ploughing land and a lack of green winter cover both promoted nitrate losses (as a result of mineralisation of soil organic matter) in excess of those under natural regeneration or clover (Sinclair et al., 1992; Milford et al., 1993; Watson et al., 1993; Francis et al., 1995). The presence of legumes and prolonged grass leys also increase organic N accumulation in the soil and thus decrease nitrate leaching up to the point of ploughing (Webb and Sylvester-Bradley, 1994). However, it is important to distinguish short and long term effects of management practices. In Phase II of the Brimstone leaching experiment (1988–1993) there were short-term (1–2 year) decreases in nitrate leaching resulting from straw incorporation, ungrazed grass leys, minimal cultivation (as opposed to ploughing) and growth of winter cover crops (as opposed to winter fallow). Ploughing up the grass leys or cover crops and land that had been minimally cultivated resulted in flushes of nitrate, and repeated straw incorporation led to increased losses, compared with burning, after about two years. The largest total losses over the five years were from land on which winter cover crops, followed by spring cereals, were grown in the first and third years (Catt and Howse, 1995). The reason for the changes from short-term decreases in nitrate losses to long-term increases was that each of these management practices temporarily increased the content of easily mineralisable soil organic N; this was initially retained in the soil but its subsequent mineralisation (e.g. by ploughing) led to increased leaching losses.

Livestock waste is also a significant source of nitrate. The U.K. produces an estimated 78 million tonnes of manure on farms requiring storage, handling and subsequent land application (Smith and Chambers, 1993). This poses a risk of nitrate leaching losses, particularly when applied in the autumn–winter period (Froment et al. 1992; Johnson, 1992). With manures, problems associated with uneven spreading and nutrient variability can increase the risk of nitrate leaching (Wadman and Neeteson, 1992). A “standard” dairy cow produces 47 kg of total nitrogen in dung and urine, during a six-month housed period (Anon, 1994e; Dampney, 1995), but few farmers make adequate allowances for this when they plan their annual fertiliser purchases. Liquid manure (slurry) nitrogen efficiency following land application is related to its dry matter content (Smith and Chambers, 1992); high dry matter (c. 10%) slurries are less efficient (approximately 20%) than low dry matter (c. 2% slurries) (approximately 60%).

The application of animal manures to soils in the autumn–winter period, followed by overwinter rainfall can result in substantial nitrate leaching losses. Organic manures are commonly applied to arable land in the autumn and winter months, prior to cultivations for the next crop. Consequently their “available” nitrogen is often poorly utilised and much can be lost by ammonia volatilisation, leaching of nitrate or denitrification. The last of these processes may lead to emissions of nitrous oxide (N₂O), which is a “greenhouse gas” and may increase global warming. In experiments on free draining light textured or shallow chalk soils, up to 40% of slurry N applied before December was lost by leaching (Froment et al., 1992). Substantial losses of nitrogen undoubtedly occur from a wide range of soils in the U.K., though free draining sandy soils are considered the most susceptible (Jones and Thomasson, 1990). Annual nitrate losses from arable areas on vulnerable soils are generally greatest under irrigated vegetables (including potatoes), which receive high rates of N and least under crops such as cereals and sugar beet which have deep root
systems, longer growing seasons or require less N fertiliser (Jones and Thomasson, 1990; Scott et al., 1992). Although low risk crops occupy most of the land area of the U.K., losses exceeding 50 kg N ha\(^{-1}\) yr\(^{-1}\) are likely to be common on the “better” soils (land classes 1 and 2, Davies et al., 1993) because of the more intensive management there. However, where heavily fertilised and irrigated vegetables are grown, losses may exceed 100 kg N ha\(^{-1}\) yr\(^{-1}\) (Jones and Thomasson, 1990).

Some nitrogen inputs to the land arise from sources other than direct application of fertilisers and manures. These include deposition of nitrate and ammonium in rain and of NH\(_3\) and NO\(_x\) from the air, which are influenced by proximity to intensive livestock units, road traffic and industrial units. Total deposition from the atmosphere can vary from 10 to 50 kg N ha\(^{-1}\) per year (Anon, 1994f). 50 kg N ha\(^{-1}\) is equivalent to a quarter of the average nitrogen applied annually to tilled land. Approximately a third of this nitrogen is leached because it is deposited when the crop cannot use it (Goulding and Poulton, 1992).

A regional model for nitrate leaching losses to surface waters in Northern Ireland (Jordan et al., 1994) predicted that agriculture accounted for 70% of the annual total nitrate load to a major water course, the remaining 30% coming equally from rainfall and domestic sewage. The value of the nitrogen leached was estimated to be around £8.8 million per year for the Province as a whole or £6.50 annually for every hectare of land.

Some crop rotations and specific management activities can increase nitrogen losses (Shepherd et al., 1992). Examples include incorporating pea or rape haulm soon after harvest, rather than delaying until drilling, and late autumn cultivations to establish wheat after sugar beet or potatoes. However, where cover crops can successfully be established in the early autumn period (August–September), they have the potential to take up nitrogen and decrease nitrate leaching (Christian et al., 1992), at least in the winter when they are grown.

Eutrophication causes accelerated growth of algae, reducing water quality. The process is influenced by nutrient levels, particularly nitrogen and phosphorus (Anon, 1992b). However, phosphorus rather than nitrogen is considered to be the limiting nutrient in most freshwater systems. Consequently when runoff rich in phosphorus reaches water bodies, where nitrate levels are already high, algal blooms can occur (Thanh and Biswas, 1990). Agriculture has been estimated to contribute c. 20% of phosphorus inputs to surface waters (Anon, 1992b), with sewage being the main input. However, the recent removal of phosphorus from sewage outfalls has decreased these inputs. Research is now being directed towards minimising phosphorus losses from arable land and grassland.

Toxic blue-green algal blooms in 1989 were thought to be responsible for the deaths of dogs and sheep. This resulted in a National Rivers Authority (NRA) survey which found that, out of 915 U.K. stillwaters (lakes and reservoirs) surveyed, 169 (18.5%) had sufficiently high densities of blue-green algae to warrant alerts to owners and Environmental Health Officers, and 28% of these were in the Anglian region (Figure 1). Other problems associated with excessive algal growth in lakes and reservoirs include de-oxygenation and reduced light levels resulting in the long term disruption of ecosystems (Anon, 1992b).

3.3. BIODIVERSITY

The application of nitrogenous fertilisers significantly reduces species diversity in grassland (Thurston et al., 1976) as it stimulates the growth of competitive grasses at the expense of broadleaved herbs. Theaker and Rew (1992) found a direct relationship
between the number of species in field boundary vegetation and the amount of nitrogen fertiliser applied. Additionally, when a combination of agricultural inputs (fertilisers, pesticides, lime and manures) are used, the impacts on species diversity are particularly strong (Boatman, 1989; Rew et al., 1992; Williams et al., 1995).

4. Farm livestock wastes

4.1. Ammonia

Nitrogen is lost to the air by two main routes: first, by volatilisation of ammonia where manures are spread on the soil surface without being incorporated, or urea is applied to grassland or tillage crops without incorporation into the soil; and second, denitrification by soil bacteria converting nitrate to gaseous nitrous oxides or nitrogen, a process which occurs under anaerobic, waterlogged conditions. There is also some ammonia loss from crops and losses of nitric and nitrous oxide from the soil following mineralisation of organic matter (Anon, 1989a; Jarvis and Pain, 1990).

Livestock farming accounts for about 80% of the U.K.’s ammonia emissions (Istas et al., 1988; Anon, 1990b; 1994f). Jarvis and Pain (1990) estimated losses from U.K. agricultural systems to be 186 kt NH$_3$ per annum, but later estimates are greater (Anon, 1994f). These losses are responsible for more deposited nitrogen in the U.K. than from vehicles and power station sources (Anon, 1994f). The extent of ammonia volatilisation depends principally on how animal excreta are managed in housing,
Table 1. Estimates of annual losses of NH$_3$-N from U.K. agricultural systems

| Source                          | kt NH$_3$-N | %
|---------------------------------|-------------|---
| Fertiliser*                     |             |   
| : tillage                       | 4.3         |   
| : grassland                     | 4.8         |   
| Grazed grassland†               |             | 5  
| : cattle                        | 27.5        |   
| : sheep                         | 14.9        |   
| Livestock wastes:               |             | 23 
| spreading on land ‡             |             |   
| : cattle                        | 40.4        |   
| : pigs                          | 10.1        |   
| : poultry                       | 10.1        |   
| buildings‡                      |             | 32 
| : cattle                        | 25.3        |   
| : pigs                          | 23.7        |   
| : poultry                       | 6.0         |   
| Waste storage§                  |             | 29 
|                                 | 19.3        | 10 
| Total                           | 186.4       |   

*Assumes (i) 15% and 5% losses from urea applied to grassland and tillage, respectively; (ii) no significant losses from other N fertilisers; †Assumes a median rate of loss of 12.5 and 1.0 g NH$_3$-N animal$^{-1}$ day$^{-1}$ and grazing periods of 185 and 365 days, respectively. ‡Based on average figures from recent experimental data. §Assumes a 5% and 15% loss of excreted N for cattle and pig slurry, respectively. Adapted from Jarvis and Pain (1990).

during storage and following land application. As indicated in Table 1, losses of ammonia from animal sources occur from buildings (29%), storage of manure (10%), grazing (23%) and following spreading (32%). A further 9 kt (5%) of ammonia is also estimated to come from fertilisers applied to grazing and tilled land, especially calcareous soils (Istas et al., 1988; Jarvis and Pain, 1990; Johnson, 1992; Sommer et al., 1993).

Emitted ammonia is changed in the atmosphere to ammonium, which is present as acidic oxides of sulphur and nitrogen. About 50% is estimated to be deposited in the surrounding area, with the remainder retained in the air for up to approximately 13 hours, during which time it may be transported considerable distances (Istas et al., 1988). Deposition can be in either wet or dry forms. When it enters the soil, ammonium N is nitrified to nitrate, producing hydrogen ions which acidify the soil (Istas et al., 1988).

Ammonium absorbed by plants can have toxic effects under conditions of low soil pH, drought or low temperatures. Additionally, ammonium deposition has specific secondary effects, particularly on low nutrient status ecosystems. These include (Istas et al., 1988):

(i) an imbalance of the general nutrient status;
(ii) disrupting carbohydrate metabolism;
(iii) causing phosphorus deficiency by restricting mycorrhizal development; and,
(iv) increasing the formation of organic nitrogen compounds in the plant, making it more sensitive to insect pests and fungal diseases.
Environmental Impact of Agriculture in the U.K.

The impacts of ammonium deposition are likely to be most severe in biological communities where the nutrient status is low and pH is critical, with the rarity and diversity of species in such habitats giving rise to concern. Examples include heaths and bogs in the U.K. whose critical loads are only 5–10 kg N ha\(^{-1}\) yr\(^{-1}\) (Anon, 1994f; Pearce, 1995).

4.2. BIOCHEMICAL OXYGEN DEMAND

Agricultural wastes can have particularly high biochemical oxygen demands (BOD). Waste milk has a BOD value 350 times greater than raw domestic sewage (140 000 mg l\(^{-1}\)); silage effluent is 200 times greater (30 000–80 000 mg l\(^{-1}\)) and cattle slurry 50 times greater (10 000–20 000 mg l\(^{-1}\)) than domestic sewage (Anon, 1989b, 1991). Where pollutants with high BOD enter a river the dissolved oxygen concentration decreases immediately, recovering only when the effluent is broken down and re-aeration occurs. This may have fatal consequences for biological communities. The main impacts of agricultural pollution incidents are deaths of fish and macro invertebrates. For example, the NRA reported the death of 15 000 fish following the accidental discharge of 90 000 litres of farm slurry into the River Wheelock in 1993 (Anon, 1994a).

Most reported agricultural pollution incidents are from point sources caused by release of animal slurries, silage effluent, and yard washings (Seager et al., 1992). However, these do not take into account the effects of continuous or chronic pollution. The Water Authorities Association ascribed most agricultural pollution incidents to poor management and failures to comply with good agricultural practice (Anon, 1986).

Water pollution from farm wastes was a significant problem in 1993 with 2883 reported incidents in England and Wales, amounting to 11% of all water pollution incidents (Anon, 1994a). Similar numbers occurred in the previous four years. Analysis of water pollution incidents for 1993 showed that the greatest proportion came from dairy cattle wastes (35%) and beef cattle wastes (32%), followed by pig wastes (7%), with poultry and arable sources (e.g. crop protection chemicals and crop spillages) each comprising 2%. Other incidents came from fish farms, stables and horticulture (2%) and the sources of 19%, although thought to be from agriculture, were not identified (Anon, 1992f, 1994a).

Pollution incidents commonly have inter-related or linked secondary effects which tend to increase their environmental and economic impacts. For example, pollution of a water course by farm wastes might primarily result in the death of fish, but predatory animals higher up the food chain such as kingfishers may also be affected. Additional costs can result from clean up, the movement of fish to safety or re-stocking, water purification treatment and subsequent monitoring (Rau and Wooten, 1980). Increasingly farmers are being asked to pay the extensive clean-up costs (Purnell, 1995).

4.3. ODOUR

The odour nuisance from farms originates primarily from animal production units. Odours are dispersed by extraction fans on farm buildings, and also come from manure heaps, waste spreading on the land, waste storage tanks (particularly during emptying) and silage pits. Their impacts depend on the proximity to local populations (Loehr, 1977; Rau and Wooten, 1980). Influencing factors include topography, prevailing wind direction and wind speed (Anon, 1992a).

In the year 1989/1990, there were 3700 complaints about odours from farms (Anon,
1992b). This was about a quarter of the total number of complaints received by Environmental Health Officers. Around 40% were considered “justifiable” and were attributed mainly to slurry storage and spreading (Figure 2) (Anon, 1992b). These incidents were particularly associated with dairy farms, the odour problems occurring on exposure to air after storage of slurry in anaerobic conditions (Loehr, 1977). Guidance on management techniques developed to reduce odour problems are given in the Code of Good Agricultural Practice for the Protection of Air (Anon, 1992a).

5. Soil erosion

Soil erosion in the U.K. is caused by the action of water and wind on arable lowlands, and also of frost and animals in the uplands. Soil may to be lost entirely from a field
or redeposited within it depending on topographic features. The coarse fraction of the soil (sand and stones) is transported short distances whereas the finer silt, clay and organic matter can be moved well away from the site. This causes the remaining soil to become progressively coarser-grained (Davies et al., 1993) and less fertile. Much of the phosphorus and pesticide losses from farm land to surface water are bound to eroded soil particles (Anon, 1993e; Powlson, 1993; Wild, 1993).

The Woburn Erosion Reference Experiment (Catt et al., 1994), set up in 1988 to assess the long term effects of erosion on soil fertility, crop production and nutrient losses, compared plots cultivated up and down a slope with those cultivated across the slope. Runoff, soil and nutrient losses were greatly increased where crops were drilled up and down the slope. Standard plough-based cultivations were also compared with minimal cultivations, the former commonly resulting in greater soil losses. Yields were often much less from the land cultivated up and down the slope, probably because more water is lost than where plant rows are parallel to the contours. As most U.K. farmers do not practice minimal tillage and usually cultivate land up and down slopes, these results suggest that pollution of surface waters, and other problems resulting from soil erosion, are unlikely to decrease in the future.

On a farm the main impacts of soil erosion can be reduced soil water storage capacity, loss of nutrients and deterioration of soil structure, as a result of the fine soil particles being lost (Wild, 1993). However, these impacts are usually only seen over very long periods. Off-farm impacts are probably the more significant immediate effects of soil erosion. Boardman (1990) identified various economic burdens on the South Downs, including flooding of property by soil laden water and the costs to councils of clearing roads and drains of eroded material. Secondary impacts can include decreases in water body capacity through sedimentation, deterioration of aquatic habitats and an increase in water treatment costs (Canter, 1986; Thanh and Biswas, 1990; Patrick et al., 1991; Anon, 1993e; Wild, 1993). Tertiary impacts may affect recreational activities such as fishing (Clark et al., 1985; Patrick et al., 1991) and human health (Croll, 1990; Anon, 1992b) and can have economic implications in the cost of water treatment or of dredging and cleaning watercourses (Rau and Wooten, 1980; Croll, 1990; Anon, 1992g).

7. Conclusions

The major environmental impacts associated with agriculture in the U.K. result from pesticides, nitrogen compounds, pollution by farm livestock wastes and soil erosion. Other areas requiring further investigation include the effects of non-animal wastes, socio-economic impacts and aesthetic effects, for example on landscape. Although we have treated the major problem areas in isolation, they are frequently inter-related, often involving several rounds of effects, some of which are “unseen”, such as the deterioration of aquatic habitats and the costs associated with water purification. The causes of pesticide loss into the environment need to be addressed and measures taken to help reduce this particular area of environmental impact. Environmental information needs to be widely available and publicised so that pesticide choice can be based on potential environmental effects. Research is currently focused on the development of management strategies to reduce losses of nitrogen and other nutrients from agriculture. The results of this research need to be interpreted on a farming system scale and converted into integrated management approaches. A first step in this has been the
publication of the Codes of Good Agricultural Practice (Anon, 1991, 1992a, 1993a), but these will need continual revision in the future as field research progresses.

Increasing environmental pressures on the agricultural industry mean that new environmental techniques need to be adopted by farmers. This overview has identified and evaluated the key impact areas associated with U.K. agriculture, updating earlier work in the field (Loehr, 1977; Canter, 1986). It has examined the extent of current knowledge and provides a foundation for research into the development of management systems for mitigating the detrimental environmental impacts of agriculture. Management approaches such as environmental auditing, developed for other industries (Anon, 1989c), are now being examined to assess their potential benefits for use in agriculture. The increased use of computers in agriculture provides opportunities for developing environmental management and decision support systems to assist farmers address environmental demands without loss of profitability. A system prototype developed at the University of Hertfordshire (Lewis et al., 1995) is currently under evaluation by practitioners from the farming industry.

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