Chapter 2

ENVIRONMENTAL IMPACTS ASSOCIATED WITH PRODUCTION

With the widespread intensification of arable farming, environmental consequences have become apparent throughout the OECD area. Such environmental impacts include damage to, and removal of, soil thereby threatening agricultural sustainability, and water pollution. Modern arable systems also impact upon biodiversity within the system itself, and on associated non-cropped habitats such as grassland, field boundaries and watercourses, as well as on the aesthetic quality of the arable landscape.

There is a high degree of integration between the various environmental impacts of arable farming because crop production affects multiple environmental services through complex ecosystem linkages. For example, the conversion of grassland to an intensive form of arable crop production will reduce certain wildlife habitat and landscape values formerly provided by the grassland, increase erosion and release carbon emissions from tillage, increase the potential for nutrient and pesticide residue run-off and infiltration into surface and ground waters, and could increase surface or ground water withdrawals if supplemental irrigation is used. In this chapter, as far as possible, environmental impacts will be treated separately. Arable systems are also often highly integrated with livestock and forestry, and therefore references are made as appropriate. Generally, such multiple land use tends to be associated with higher biodiversity and landscape value compared with purely arable systems.

The following environmental impacts of arable farming systems are discussed in this chapter:

- soil-related impacts;
- water-related impacts;
- air quality; and
- biodiversity.

2.1. Soil-related impacts

Soils under arable crop cultivation may be susceptible to erosion; declining organic matter resulting mainly from frequent cultivation;

pollution by pesticides and, to a lesser extent, heavy metals (Stoate, *et al.,* 2001).[1] These processes are highly interrelated. Farming practices are important driving forces influencing soil properties.

2.1.1. Soil erosion

Soil erosion is widespread throughout OECD countries. Soil erosion can adversely affect crop productivity and damage the environment in a variety of ways. Impacts of soil erosion are felt both on-farm and off-site.[2] Moreover, there is a direct link between the magnitude of soil erosion and loss of soil biodiversity (OECD, 2003*c*).

There are two distinct, but related, facets of the on-site decline in productivity caused by soil erosion: short-term reduction in agronomic yield and long-term decline in soil productivity, resulting from a lessening in soil quality due to reduced rooting depth water-retaining capacity, soil organic matter and soil biodiversity. The two most important off-site impacts of erosion on the environment are, respectively, degradation of surface water by sediment and sediment deposition, and emission of greenhouse gases into the atmosphere (Heimlich, 1991).

The risk of soil erosion from wheat cultivation is normally low, with soybean, sunflower and maize cultivation systems generally being associated with higher levels of soil erosion.[3] For rice, soil erosion is constrained by the ground coverage offered by irrigation water during the early stages of growth and through the widespread use of terracing in upland rice cultivation. The system of terracing can prevent soil erosion and landslides. On the other hand, irrigated rice production systems may cause problems of soil salinisation and waterlogging, particularly in regions where irrigation water is often of poor quality and paddy fields are provided with inadequate drainage (van Tran, 1998). Expansion of upland rice farming systems may increase soil erosion and deforestation. The draining of coastal wetlands for rice cultivation can lead to the dehydration of soil, often causing sulphur to rise to the surface, with consequent acidification (Barbier and Mouret, 1998).

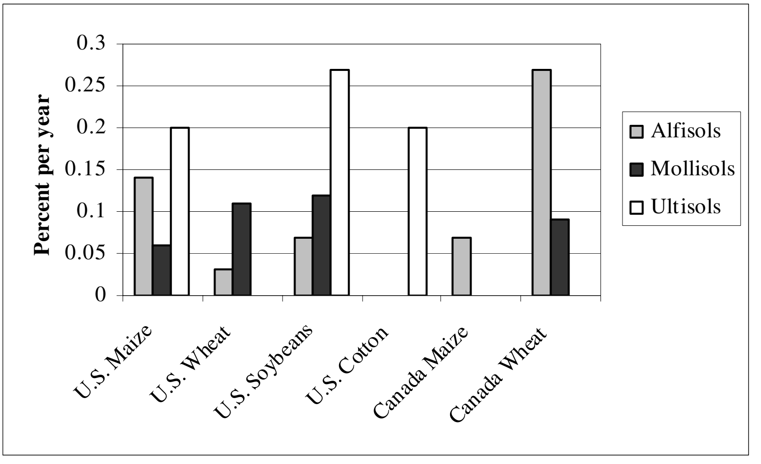
Soil erosion is caused by wind and water. The rate of erosion is influenced by a combination of physical factors such as climate, topography, soil texture, crop type and management factors such as cultivation practices, dates of seeding and harvest and post-harvest residue levels.

Higher rates of erosion can result from devoting larger areas to autumn cultivation, increasing field size, with the associated loss of hedges, and continuous arable cropping, all of which increase the exposure of soil to wind and water in space or time (Evans, 1996). Soil erosion is partly related to crop rotation. Available studies, mainly in the European context, seem to suggest that, generally, lack of crop cover tends to increase erosion rates on arable land (Boatman, *et al.,* 1999). Late-harvested spring-sown crops such as maize also increase the exposure of soils to erosion. Moreover, rainfall, slope and soil type can all have a major influence on erosion risk (Brouwer and Ervin, 2002).

The capacity of farmland to produce, and the link between agricultural production practices and soil erosion, has been studied extensively. Recent research shows that on-site productivity losses from erosion are relatively small. Erosion-induced productivity decline is estimated to generate a potential annual loss of 0.3% in the value of the global production of selected crops, ranging from 0.04% per year in **Europe**, to 0.61% per year in **Australia** (den Biggelaar, *et al.,* 2003). A USDA study found that average annual water-induced soil erosion rates vary widely by crop production area, soil, and region, but, in most cases, range between 12 and 17 t/ha/year (Eswaran and Reich, 2001). Estimates of annual production losses to erosion in the **United States** range from USD 40 million, to over USD 100 million (Crosson, 2004). Den Biggelaar, Wiebe and Breneman (2001), taking into account differences due to regional variations in soil and climate, but assuming unchanged farmer management practices, estimated the erosion-induced losses for wheat, maize, soybeans and cotton at only USD 56 million. The same study found that, although the erosion-induced yield loss varies widely by crop and region, there is, on average, a 0.3% per year loss in the value of global crop production, ranging from 0.04% in **Europe**, to 0.6% in **Australia**.[4] This average yield loss ranges from 0.03% for wheat on Alfisols (fertile soils that occur primarily in the Corn Belt) in the **United States**, to 0.3% for wheat on Alfisols in **Canada**, and for soybeans on Ultisols (fertile but acidic soils that occur primarily in the Southeast) in the **United States** (Figure 2.1).[5] The total loss in production for the **United States** was estimated at 229 000 tonnes for maize; 54 000 tonnes for wheat; 61 000 tonnes for soybeans; and 2 000 tonnes for cotton.[6]

While the estimated costs of erosion in terms of lost output are insignificant at the national level, there may be important regional and local impacts in terms of resource depreciation and off-site costs of crop production. For example, in the **United States**, Faeth (1993) found negative net economic value per hectare, after accounting for soil degradation and off-site costs, for Pennsylvania's best maize-soybean rotation over 5 years.

Figure 2.1. Annual yield losses due to erosion for selected crops in Canada and the United States, 1939-99



Source: USDA/ERS.

In **Australia**, rates of soil erosion associated with arable cropping are similar to those of native pastures. Of the land uses, sugarcane has the highest erosion rate (20.3 t/ha/year, as compared to 4.3 t/ha/year for oilseeds and 3.3 t/ha/year for cereals excluding rice) (Lu, *et al.,* 2003). The same study found that although acceleration of current erosion rates above natural rates is relatively evenly distributed across Australia, there is a great diversity across various land uses: cereals (excluding rice) 18 times the natural rate; oilseeds 33 times; sugarcane 33 times – while grazing lands have rates typically 2-5 times the natural rate. Soil erosion from cropland is an issue of concern regionally in **Canada**, particularly in the arable plains of the Canadian wheat belt.

Soil erosion is widespread in the **EU**, although levels of severity vary across countries, and between regions within countries (EEA, 2003*a*). Major causes are unsustainable agricultural practices, over-grazing, large-scale farming, construction activities, and poor water and irrigation management. Estimates of soil loss by erosion range from 3.6/t/ha to 40 t/ha/year (Boatman, *et al.,* 1999). The European Soil Bureau and the Pan-European Soil Erosion Risk Assessment programme show that the south European region is the most prone to soil erosion – with most erosion linked to the occurrence of high rainfall in short periods during storms. There is also evidence of significant erosion occurring in other parts of Europe (*e.g.* **Austria**, **Belgium**, the **Czech Republic**, **France** and the **United**

Kingdom). A study found that half of the arable fields surveyed in **England** and **Wales** showed signs of soil erosion at least every other year (Evans, 1996). Erosion by water is exacerbated by intense rainfall, steep slopes and sandy soil, late-harvested, spring-sown crops, such as maize and sugarbeet. It is lower where crops are drilled in early autumn and where minimum cultivation or direct seeding practices are used. Annual economic losses are estimated at around EUR 53 per hectare, while the costs of off-site effects on the surrounding civil public infrastructures reach EUR 32 per hectare (Torress, *et al.,* 2001).

In the **United Kingdom**, in grassland and arable regions, the timing of agricultural activities is as significant as considerations of cultivation practice, crop cover and soil type, in determining the scale and extent of soil erosion (McHugh, 2004). Approximately 70% of crops on arable soils are winter-grown, and therefore planted between August and December, when rainfall duration and intensity greatly increase the risk of erosion.

For arable crop farming in **Korea**, soil erosion by water is mainly due to the concentrated rainfall in the summer season. Annual soil loss is only 0.02 t/ha in paddy fields, as compared to 32 t/ha in uplands (on slopes greater than 15%) and 0.9 t/ha for forest. Annual total soil loss in paddy fields is 22 768 tonnes, while in upland and forest areas it is 26 and 488 million tonnes, respectively (Hur, *et al.,* 2004).

Water-induced soil erosion is an important by-product of cereal production in **Norway** (Oygarden and Gronlund, 2004). Erosion occurs mainly in autumn or winter as the result of rainfall, snow melt and partly frozen soil conditions. Since 1993, threshold values for soil loss were 2 t/ha/year. In **Switzerland**, average soil losses during the 1998-2001 period decreased by 6% compared to the 1987-89 period (Prasuhn and Weisskopf, 2004). Between 1998-2001 around 20% of the arable land was affected by soil erosion every year, with an average soil loss of 0.7 t/ha/year. Significant damage associated with erosion was estimated for winter wheat. Threshold values were 4 t/ha/year.

The *2001 National Resources Inventory* (NRI) showed that soil erosion on cropland in the **United States** declined from between 2.8 billion tonnes per year in 1982, to 1.6 billion tonnes per year in 2001 (NRCS, 2003). Sheet and rill erosion dropped from 8.9 t/ha/year, to 6.1 t/ha/year, and wind erosion dropped from 7.4 t/ha/year, to 4.7 t/ha/year. Water-caused erosion dropped by almost 41% during this period, while wind erosion dropped by 43%. Between 1982 and 2001, cropland acreage eroding at excessive rates dropped by 39%.[7] In 2001, 42 million ha of cropland were experiencing excessive erosion, down from 69 million ha in 1982. In 2001, about 72% of total cropland was eroding at, or below, the soil loss tolerance rate, up from

60% in 1982. Highly Erodible Land (HEL) cropland acreage declined from 50 million ha in 1982, to 41 million ha in 2001. The decline occurred in HEL acreage eroding at excessive rates, while HEL acreage eroding at acceptable soil loss tolerance rates increased slightly. Heavy concentration of both HEL cropland and high average erosion rates was located in cereal and oilseed producing areas such as the western plains, the western Corn Belt and in the Mississippi delta (Claassen, *et al.,* 2004*a*).

2.1.2. Nutrients

Loss of nutrients and organic matter from the soil can represent a loss of fertility which ultimately can affect crop yields and also pollute water bodies. Losses of phosphates from the soil are largely due to soil erosion. Nitrates originating from organic and inorganic fertilisers are particularly prone to leaching and the degree of losses resulting from arable crop production depends on the type farming system operated as well as on specific site characteristics. The quantity of nitrate loss from a particular farming system is determined largely by the balance between nitrogen inputs in the form of fertilisers, and nitrogen outputs from the farm in terms of harvested crops. It also depends on whether the farming system protects the soil from leaching during winter, by avoiding spreading of nitrogen fertilisers (organic or inorganic) on the land in this period and ensuring vegetation cover. Leaching of nitrogen can result from applications of mineral fertilisers at very early stages of crop growth, so that little is taken up by plants, or from the excessive application of fertilisers. However, in some regions much of the nitrogen lost from soil is associated with mineralisation of soil organic matter, normally during the period following the harvest or the ploughing of pasture for planting arable crops (Bloem, *et al.,* 1994).

Hoffmann*, et al.* (2000) estimated long-term changes in nitrogen leaching from cereals, grass and bare fallow for three different soil types in nine **Swedish** agricultural regions, covering a range of climatic conditions. They found that leaching of nitrogen was approximately the same in the 1860s as it was in mid-1980s. For cereals, in particular, both N input and N-uptake efficiency have exhibited upward trends.

To gauge whether nutrients from arable crops pose an environmental risk, nitrogen balances for arable crops were calculated. A negative balance indicates that the amount of nitrogen removed from the soil through the harvested crop exceeds the amount of nutrient applied. Continued negative balances deplete nutrients in the soil, disrupt the soil ecosystem and can damage productivity (USDA, 2003*b*). Positive balances occur when farmers over-apply nitrogen.

Table 2.1 displays application rates of nitrogen (N) on arable crops, the share of N consumed by arable crops and the potential environmental risk from nitrogen loss in arable crop farming, as measured by the nitrogen balance (*e.g.* greater than 50KgN/ha). According to these results, countries where both the risk and the application rates of N are estimated to be highest include **Korea**, **Belgium** and **Denmark**. The **Netherlands** has the largest potential risk, but is ranked fourteenth in terms of application rates. **Poland**, **Canada**, **Australia** and **Turkey** are estimated to have both the lowest risk and application rates.

2.1.3. Waterlogging and salinisation

Waterlogging and soil salinisation have become important environmental concerns in some OECD regions. Waterlogging occurs as a result of a rise in the level of the water table, commonly caused by inefficient irrigation practices, such as inadequate drainage. The rise of the water table may also increase salinisation by drawing salt upwards from the lower soil horizons.

Most arable crops do not tolerate salt and are seriously affected when salts concentrate within the root zone. The main impact of increasing soil salinity is loss of production, yields and income. Other on-farm effects include the decline in the capital value of land, salinisation of water storage, loss of farm flora and fauna, and loss of shelter and shade. These effects are propagated at the regional level, where they could have a significant impact on biodiversity, water supplies and infrastructure. It is estimated that moderate-to-severe salinity on agricultural land can reduce the annual yields of most cereal and oilseed crops by about 50% (McRae, Smith and Gregorich, 2000).

In **Australia**, the incidence of soil salinisation is high on dry and irrigated land, predominantly in the Murray-Darling Basin and the south-western part of the country. In these areas, production of wheat is particularly affected. Around 30% of the grain farms in the west and 10% in the south of Australia are affected by significant dryland salinisation (AUDIT, 2001). It is estimated that in 2000 4.6 million ha of agricultural land in Australia were under a high risk of salinity hazard, and is projected that, unless effective solutions are implemented, the area could increase to 14 million ha by 2050. In the **United States**, some 5% of the cropland and pasture is affected by soil salinisation. Salinisation is also a problem in **Turkey** where it is associated with poor irrigation practices in some regions (OECD, 2001*a*).

Table 2.1. Potential environmental risk from nitrogen in arable crops[1]

Country	Application rates of N on arable crops (kg/ha)[2,3]	Share of N consumed by arable crops as % of the total amount of N consumed by total agriculture[2,3]	Nitrogen balance (kgN/ha) 1995-97
United Kingdom	156	48	87
Germany	140	63	61
Switzerland	135	57	61
Ireland	124	8	79
France	114	32	54
Korea	112	46	253
Belgium	111	37	181
Norway	104	33	73
Denmark	104	72	115
Italy	103	62	30
Austria	103	80	27
Czech Republic	101	87	54
Portugal	94	60	63
Netherlands	94	12	262
Greece	91	41	33
Sweden	90	61	34
Spain	89	59	44
New Zealand	86	6	6
Hungary	82	87	4
United States	80	88	32
Japan	78	35	135
Finland	74	53	64
Slovak Republic	71	86	45
Mexico	64	58	n.a.
Poland	59	55	29
Canada	56	86	14
Australia	37	72	7
Turkey	32	31	12

n.a. = not available.

Notes:

1. Environmental risk is indicated where nitrogen surplus is greater than 50 kgN/ha.

2. As time series data for N by crop are not available, the most recent data from IFA/IFDC/FAO (2002) were used. Nitrogen balance data are from OECD (2003*b*)

3. Caution should be exercised in interpreting these results due to a number of data and methodological problems. Data on fertiliser use by crop types should be taken to reflect the general magnitude rather than the exact measurement. Mixed-cropping, for example, makes it difficult to estimate the amount used for each crop. On the other hand, with double-cropping, although the fertiliser is applied to one crop, both crops benefit. Moreover, some countries (*e.g.* Australia) make estimates for a group of crops (*e.g.* cereals, oilseeds) rather than individual crops.

Sources: OECD Secretariat calculations, based on IFA/IFDC/FAO (2002), OECD (2003*b*); FAOSTAT.

2.2. Water-related impacts

Arable crop production can have environmental impacts on water through extraction for irrigation and pollution of watercourses with nutrients and pesticides.

2.2.1. Water use

Agriculture is a significant user of water resources in many OECD countries. Large volumes of water are used annually in some regions for the irrigation of arable crops. Non-irrigated crop systems generally use significantly fewer inputs of fertilisers and other agro-chemical inputs. Different crops are subject to irrigation at varying levels of intensity. Wheat requires little irrigation except in arid and semi-arid regions. Maize requires relatively high levels of water during the early stages of growth, and in some regions cultivation relies heavily on irrigation. Water also plays a prominent role in the cultivation of rice. Paddy rice consumes more water than any other crop, but much of it is recycled and put to other uses. Certain rice cultivation practices develop water storage capacity and help to control flooding during heavy rains.

In **Australia**, the agricultural sector is the most intensive user of water per unit of value created. Approximately 75% of Australia's water is used in irrigated agriculture (AUDIT, 2001). The intensity of water use varies within and across states, due to climate, soil crop type and method of application. Generally, rice is the most water-intensive crop sector, with application rates varying between 11.9 and 13.9 ml/ha, followed by grapes. The intensity of water use for cereals and oilseeds is, on average, 3 ml/ha, as compared to 7 ml/ha for all irrigated land uses.

In Europe, agriculture accounts for around 30% of total water use. The scale and importance of irrigation is significantly greater in southern areas of the **EU**, accounting for over 60% of water use in most countries. Within the EU, the main irrigated arable crops consist of maize and rice, particularly in **France**, **Greece**, **Italy** and **Spain** (IEEP, *et al.,* 2000). In **Portugal**, the application rate for maize varies between regions from 3.9 m^3/ha, to 6.6 m^3/ha (Plano Nacional da Água, 2002). On the other hand, in the **United Kingdom**, in the mid-1990s cereals accounted for only 12% of total area of irrigated crops and around 5% of the volume of water used for crops (potatoes, sugarbeet, cereals, other crops grown in the open) (DEFRA, 1997).

In **Mexico**, the total area planted for soybeans is irrigated, and for wheat and barley more than two-thirds of the area planted is irrigated. For maize, available evidence seems to suggest that the decline in maize production and yield observed since the mid-1990s was in the higher-yield irrigated sector. Between 1995 and 2000, production on irrigated land declined by 31% from its 1994 peak, whilst average rain-fed maize was 18% higher than the average rain-fed production of the previous six years. Likewise, the area cultivated by the irrigated sector, which applies more pesticides, has significantly declined, while the rain-fed maize sector, which uses significantly less pesticides, has expanded (Dyer-Leal and Yúnez-Naude, 2003).

In **Korea**, paddy fields take up about 77% of total water use in agriculture, 58% of which is used in irrigated paddy fields. Even though large areas of irrigated paddy fields have been converted for non-agricultural uses, the share of irrigated paddy field in total paddy field has increased steadily since 1970s (Hong-Sang, 2004).

Irrigated agriculture accounts for an important part of the **United States** cropland sector, contributing almost half the total value of crop sales on just 16% of total cropland harvested. Over time, the mix of irrigated crops has changed. From 1969 to 1982, irrigated area increased for almost all crops, with the biggest gains in the major export grains (maize, soybean and wheat). Since 1982, there has been a general trend towards crops with higher value per hectare irrigated. Acreage of irrigated soybean, maize, horticulture and mint has doubled, while declines occurred in irrigated areas of sorghum, wheat, oats, barley, dry beans, pasture and un-harvested cropland. In 2000, around 280 000 farms irrigated 22.4 million ha of crop and pastureland (USDA, 2003*b*). Irrigated acreages in 2000 were substantial for several crops, including maize for grain (4.1 million ha, or 18% of all irrigated crops), wheat (1.3 million ha, or 6% of all irrigated crops), barley (0.4 million ha, or 2% of all irrigated crops), rice (7.7 million ha or 6% of all irrigated crops) and soybeans (2.1 million ha, or 9% of all irrigated crops). All of the rice-growing area is irrigated.

2.2.2. Water pollution

For most OECD countries, nutrients, pesticide and soil sediments are the principal sources of water pollution associated with arable crop production. Inputs such as pesticides and nutrients can enter ground and surface waters, seriously affecting the quality of drinking water, and the cost of its treatment. Their presence in surface water can also have serious consequences for aquatic life. Greater impacts are associated with simplified, high-input arable systems. Nutrients, especially phosphates, cause eutrophication of water, which changes the ecological balance and can result in undesirable effects such as fish death and algal blooms. Problems are greatest where farming is intensive (Stoate, *et al.,* 2001).

Nutrient pollutants from arable crop production are comprised primarily of nitrogen and phosphates which reach water courses from the soil by leaching, surface run-off, sub-surface flow and soil erosion. Both nutrients can cause severe eutrophication of water. Arable farm systems are smaller sources of phosphate pollution than livestock systems. In the **United Kingdom**, for example, the UK Environmental Agency has estimated that agriculture is responsible for 43% of phosphates in surface water - 29% from livestock and 14% from fertiliser.

In the **United States**, nutrient pollution is the most important cause of water quality impairment in lakes and the third-largest cause of river pollution. Phosphate pollution from arable crops production may be important in regions with low absorption-capacity soils, such as sandy soils, and in areas where phosphorus-demanding crops (*e.g.* maize) are grown. For example, in the **United States**, some evidence shows that the Corn Belt has a high potential for nitrate contamination of both groundwater and surface water from commercially applied fertiliser, and for phosphorus contamination of surface water the same source (USDA, 2003*b*). Whether nitrogen actually contaminates surface or groundwater depends on the amounts of nitrogen applied to agricultural land, the leaching characteristics of the soil, precipitation, crop type, timing of cultivation and on farming practices. Early ploughing of rape residues can lead to nitrogen leaching. Nitrates are particularly prone to leaching during the autumn, when nitrate passes through the root zone faster than the crop is able to exploit it, and also following the ploughing of grassland, when organic nitrogen is mineralised (Young, 1986). Leaching is greater under cereals than under permanent grass (Croll and Hayes, 1988), but can also be high under rotational set-aside (Meissner, *et al.,* 1998). The likelihood of nitrate leaching is higher for spring-sown of cereals in northern Europe, unless cover crops, under-sowing or stubble regeneration are adopted. In contrast, nitrate leaching for autumn sowing is similar to winter cover crops (Boatman, *et al.,* 1999).

Pesticides reach water via surface run-off, through soil cracks and drains. Spray drift and acute pesticide pollution incidents can adversely affect aquatic organisms, as can the silt burden from eroded soil particles, which may also have phosphates and pesticides bonded onto their surfaces. Inappropriate cropping and cultivation techniques can exacerbate these problems.

Pesticides may enter water from point-source contamination or from diffuse sources, following application to crops. The risk of pesticide pollution depends on its solubility, mobility in soil and rate of degradation. As with nutrients, rates of pesticide use over much of southern Europe are lower and pesticide pollution of water is less of a problem than in northern

Europe, but it does occur where intensively managed, irrigated crops, such as maize are grown. Some evidence suggests that in the Po Valley in Northern **Italy** in early 1990s, use of the herbicides atrazine and molinate on irrigated maize and rice caused contamination of local drinking water and led to a ban on their use in vulnerable areas (Boatman, *et al.,* 1999).

Ground- and surface-water vulnerability to pesticides varies geographically, depending on soil characteristics, pesticide application rates, and the persistence and toxicity of the pesticides used. Areas with sandy, highly leachable soils and high application rates of toxic or persistent pesticides generally have high vulnerability ratings for pesticide leaching. Areas with heavy soils and high application rates of toxic or persistent pesticides generally have higher vulnerability ratings for pesticide run-off.

The relatively high levels of inorganic fertiliser used in the cultivation of rice may lead to the contamination and eutrophication of water. However, nitrogen leaching into surface water and groundwater from paddy fields is low compared to dryland crops and orchards, due to denitrification. Both lowland and upland systems make heavy use of pesticides. The draining of coastal wetlands for rice cultivation leads to the dehydration of soil, often causing sulphur to rise to the surface, with consequent acidification.

2.3. Air quality

Although arable crop production is not in itself a major source of air pollution, it can contribute to air pollution and climate change in a multitude of ways. Air quality concerns arising from arable crop farming include emissions into the air of greenhouse gases (GHGs), ammonia, wind-borne soil and other particulates (*e.g.* from burning crops). The focus of this report is on GHGs. The main arable crop activities which lead to airborne emissions include emissions of GHGs arising from the use of fertilisers, fossil fuel combustion (primarily through long-distance transport of arable inputs and products), wetland rice cultivation and the burning of crop residues. Burning crop residues in fields produces methane and nitrous oxide, while, of all arable crops, wetland rice cultivation is the principal source of methane (UNFCCC, 2003). On the other hand, production of biofuels from crops such as wheat and maize (for ethanol) and soybeans and rapeseed (for biodiesel) provide significant benefits for GHG reductions and air quality improvements (OECD, 2004*d*).

Notwithstanding considerable uncertainty and lack of data, it is generally accepted that agriculture is an important contributor to emissions of three GHGs: carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4). Carbon dioxide emissions from agriculture occur primarily in areas where land-use changes have taken place, or fuel use occurs; nitrous oxide, where there is crop cultivation using organic and inorganic fertilisers; while methane emissions are generally related to livestock and rice production. Most of the greenhouse gases result from intensive livestock rather than arable farming. As shown in Table 2.2, the contribution of arable crop production in agricultural GHGs is, on average, just over 10%, with considerable variation among countries (ranging from 1% in **Switzerland** to 49% in **Japan**).

Table 2.2. Contributions of agriculture and arable crop farming to GHG emissions, 2001

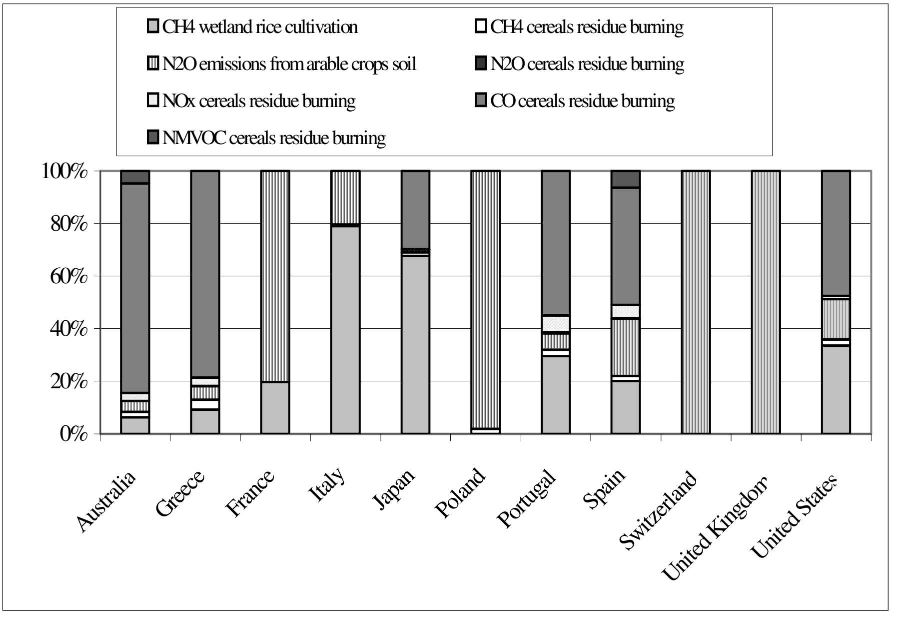
	Share of agriculture in total GHG emissions	Share of arable crops in agricultural GHG emissions	Share of agriculture in total CH_4 emissions	Share of wetland rice cultivation in agricultural CH_4 emissions	Share of agriculture in total N_2O emissions	Share of N_2O emissions from arable crops soil in agricultural N_2O emissions
	%	%	%	%	%	%
Australia	20	2	62	1	81	26
Greece	8	26	33	4	59	17
France	20	2	68	0	68	20
Italy	8	10	50	8	55	24
Japan	3	49	67	43	57	7
Poland	8	4	24	0	68	43
Portugal	14	9	55	3	73	10
Spain	12	5	58	1	66	24
Switzerland	10	1	66	0	72	24
UK	7	2	42	0	64	19
US	8	11	27	5	74	16

Source: OECD Secretariat calculations, based on UNFCCC (2003).

Farming practices associated with arable crops such as tillage methods, soil protection, crop timing and rotation, crop selection and land use can all play a role in the emissions of CO_2, N_2O, and CH_4 (OECD, 1998*b*). Agricultural soil is a major source of nitrous oxide emissions mainly originating from inorganic and organic fertilisers, while incorporation of crop residues, biological nitrogen fixation and cultivation of some soil also generate nitrous oxide emissions. Crop practices often affect the carbon content in the soils. Extreme differences can be found between wetlands and sandy soils. Wetlands can contain far more carbon than other types of soils. Changes in land use can affect the exchange of carbon between the soil carbon and atmospheric carbon dioxide.

Arable crop production is the most important source of nitrous oxide emissions from agricultural soil in **Switzerland**, the **United Kingdom**, **Poland** and **France**, while emissions of methane from rice cultivation are the most important source in **Japan** (Table 2.3 and Figure 2.2). Over 90% of methane produced by the cultivation of arable crops is caused by rice cultivation in all the countries listed, with the exception that of **Poland**. Emissions from burning arable crop residues in the field are important in **Australia**, **Greece**, **Japan**, **Portugal**, **Spain** and the **United States** (Annex Table 2.A1).

Figure 2.2. Gross emissions of GHGs from arable crop farming, 2001



Source: OECD Secretariat calculations, based on UNFCCC (2003).

Overall, recent estimates show that rice cultivation accounts for a much smaller share of methane emissions than was previously believed. Although in most of the countries listed, methane emissions from rice cultivation increased during 1990-2001, they represent only a small share of CH_4 emissions from agriculture, if **Japan** is excluded. In 2001, methane emissions from rice cultivation represented 43% of the methane emitted from all agricultural sources in **Japan**, although on average in OECD rice-producing countries, methane from rice represented less than 5% of agricultural methane emissions (Table 2.3).

Table 2.3. Methane emissions from agriculture, 1990-2001

(1000 tonnes)

	1990	1995	2001	Annual Growth Rate (%)
Australia				
Agriculture	3579.0	3413.0	3707.9	0.3
Rice	23.4	30.9	35.1	3.5
Share (%)	0.7	0.9	0.9	
France				
Agriculture	2185.0	2102.0	2087.5	-0.4
Rice	8.6	10.9	8.5	-0.1
Share (%)	0.4	0.5	0.4	
Italy				
Agriculture	913.8	901.2	871.1	-0.4
Rice	73.3	81.4	74.0	0.1
Share (%)	8.0	9.0	8.5	
Japan				
Agriculture	741.4	737.1	651.3	-1.1
Rice	336.9	342.9	281.3	-1.5
Share (%)	45.4	46.5	43.2	
Portugal				
Agriculture	302.1	278.4	279.8	-0.6
Rice	12.2	7.8	8.6	-2.8
Share (%)	4.0	2.8	3.1	
Spain				
Agriculture	912.4	957.6	1120.6	1.7
Rice	10.8	6.5	14.0	2.2
Share (%)	1.2	0.7	1.3	
United States				
Agriculture	7473.4	7972.4	7717.7	0.3
Rice	339.1	362.8	363.7	0.6
Share (%)	4.5	4.6	4.7	

Source: OECD Secretariat calculations, based on UNFCCC (2003).

The amount of methane released from the cultivation of paddy rice depends on a number of factors, including water management during the growing season, soil characteristics – such as soil temperature and type – application of inorganic and organic fertilisers, and also other cultivation practices (Yagi, 1997). Long periods of submersion promote the aerobic decomposition of organic material and reduce the amount of oxygen in the soil. As the oxygen is depleted, anaerobic decomposition by methanogenic bacteria begins. The resulting methane is partially released into the air through evaporation of water and transpiration of the rice plants. Other agricultural practices conducive to GHG reduction from rice cultivation include lowering the levels of organic fertiliser used, reducing the amount of crop residue left in the paddy fields and increasing the use of varieties of rice that emit low levels of methane.

2.4. Biodiversity

Arable crop farming can affect biodiversity and landscape in several ways. In particular, factors such as cropping systems, field size, use of agro-chemicals, drainage and irrigation can influence habitat and farmland species.

Increased intensification and specialisation of arable cropping is characterised by significant economies of scale, which could trigger declines in diversity of habitats and in farmland species. In some regions, particularly in **Europe**, with farm amalgamation, many rotations have been simplified so that crops such as wheat or maize may be grown continuously without any breaks, which often requires higher applications of fertilisers and pesticides and increases the erosion risk. Moreover, increased monocultures and reduction in the number of mixed arable and livestock farms have led to loss of biodiversity and created a less diverse landscape (Baldock, Dwyer and Vinas, 2002; Boatman, *et al.,* 1999). Increased drainage and irrigation have also caused habitat degradation in many areas where irrigation of crops (*e.g*. maize) is usually associated with increased fertiliser and pesticide applications.

In contrast, the cultivation of rice can increase the local diversity of birds and the aquatic invertebrates on which they feed. Paddy fields can play a particularly valuable role in the conservation of wetland wildlife, including breeding, wintering and migratory birds, where rice is grown close to estuary habitats. The seasonal wetland habitat provided by flooded paddy fields also supports a number of ecosystems, including many species of birds and small mammals. Rice fields also host many natural enemies or predators, which provide a mechanism to control harmful insects and pests, and thereby reduce the need for pesticides. In some OECD countries such as in **Japan**, rice production is considered the single most important factor of "multifunctional" agriculture (Nakashima, 2001). On the other hand, the introduction of upland rice production can result in deforestation on marginal, steep hillsides, whilst lowland systems are often extended at the expense of coastal wetlands and mangrove swamps, with the consequent loss of habitats and destruction of ecosystems. Further, chemicals, agricultural run-off, sedimentation and other forms of pollution could accumulate in rice fields and cause environmental damage and loss of plant and animal species.

A number of biodiversity indicators has been established by OECD within the general framework of genetic, species and ecosystem diversity (OECD, 2001*a*). According to OECD work on Agri-environmental Indicators, the number of new crop varieties has increased between the mid-1980s and the mid-1990s. This work also suggests that the trend in the share of one of the top five dominant varieties in the total marketed production for certain arable crops (*i.e.* wheat, barley, maize and soybeans) increased for many OECD countries. Wetterich (2003) reports increasing diversity in **Germany** in terms of the number of registered varieties of maize and wheat over the 1992-2000 period. McRae and Weins (2003) found positive trends in **Canada** for wildlife habitat in cropland (land used for grains, oilseeds, fruits, nuts, vegetables, tames hay). Scott (2003) calculated changes in stock and condition of habitats for the **United Kingdom** over the 1990-98 period. It was found that for arable crops and horticultural farming there has been little change in the stock, but unfavourable trends have appeared in the condition of wildlife habitat. Some other studies have calculated changes in biodiversity using farmland bird indicators as a proxy. These studies, which are mainly for Europe, found declining trends, especially in the **United Kingdom**. Heath and Rayment (2003), for example, report that although the number of common birds has remained stable in the United Kingdom since 1970, the variety of farmland species has declined.

2.5. Management practice approaches to reduce environmental impacts of arable crop production

The improvement of arable crop yields described earlier stems to a great extent, from changes in agricultural practices and techniques. Few practices have remained unaltered by the increased intensification and modernisation of arable crop production. Tilling, sowing and harvesting have become increasingly mechanised, and application of chemicals has become more sophisticated. Contemporary agricultural practices – such as monoculture or the continuous production of row crops, fewer rotations with forages, shorter rotations, intensive tillage, inappropriate fallowing and crop residue management, and the cultivation of marginal lands – are often held responsible for many of the adverse environmental effects of arable crop farming discussed in the preceding section. This section will endeavour to discuss those farming practices which are deemed to be benign to the environment.

Various approaches have been developed over the past 15 years to minimise the environmental effects of agricultural production. Among the foremost of those concerning arable crop farming are Soil Management and Conservation Systems, Integrated Plant Nutrient Systems and Integrated Pest Management. These practices are interrelated and may be substitutes or complements, but they are treated separately here, as far as possible.

2.5.1. Soil management and conservation systems

Awareness of the need for protection of the soil resource is increasingly on the research agenda and also the wider political agenda. Different combinations of crops, rotations and tillage practices may have different impacts on soil and water quality. Decisions on crop selection, rotation and tillage can affect the risk of erosion, compaction, salinisation and nutrient loss (OECD, 1994). These choices are also likely to affect water quality. Concentrations of wildlife may also be affected, as different crops and tillage methods provide different levels of habitat. Large shifts in crops and tillage practices can also affect emissions.

Research on a wide range of agricultural husbandry systems and techniques has revealed direct beneficial implications in mitigating impacts on water quality. For example, the use of contour cultivation, or minimum tillage, silt traps, cover crops, the technical application of fertiliser, and riparian buffer zones can significantly reduce sediment and fertiliser run-off losses from arable cropping activities. A **United Kingdom** survey shows that nitrogen surpluses for winter wheat have dropped from 70 kgN/ha/year in the early 1980s, to around 25 kgN/ha/year in the late 1990s due to improvements in crop protection, plant breeding and agronomy (DEFRA, 2002).

2.5.1.1. Rotational cropping systems

Different land uses have different effects on natural resources. Generally speaking, annual cropping is the most disruptive type of land use and, depending on local soil conditions, it may reduce surface and groundwater quality. It also tends to provide less wildlife value. On the other hand, perennial forages, improved pasture, and native grassland or woodland are less disruptive.

Cropping systems which involve crop rotation could reduce the environmental risk posed by arable crops because they affect soil conservation, soil fertility and pest control. For example, row crops on erodible soils can be rotated with soil-conserving crops to reduce soil loss.[8]

Closely sown row grain crops such as wheat, barley and oats provide additional vegetative cover to reduce soil erosion and add organic matter. As such, these crops tend to be less erosive than rapeseed, which is in turn less erosive than wide-row crops, such as maize and sunflowers (AAFC, 1996). Wide-row crops are associated more with soil degradation, silt and nutrient infiltration to surface water, and the leaching of nutrients and pesticides to groundwater (USDA, 2003*b*). Rotations that include forages, green manure and winter cover crops tend to erode less and improve soil quality. Rotations that include tilled summer fallow may raise the risk of salinisation and erosion.

In the **United States**, rotational cropping of arable crops is predominant with soybeans and maize. Most rotational cropping of maize and soybeans alternates, while winter wheat rotates with a row crop and small grains, and fallow. About 60% of the acreage in maize and soybeans and 40% of winter wheat were rotated in 1999 (USDA, 2003*b*). Because maize production leaves more residue after harvesting than soybeans, a maize-soybeans rotation reduces soil erosion to a greater extent than continuous soybeans (although to a lesser extent than continuous maize). Over time, rotating maize with other crops, particularly soybeans, has increased.

Empirical studies in the **United States** found that crop rotations were associated with higher yields than those achieved with continuous cropping under similar conditions. For example, in 1996 returns to maize averaged 5% to 51% higher, depending on the region, when in rotation with soybeans rather than in continuous maize production (USDA, 2003*b*).

However, agricultural support policies could be an important impediment to the adoption of crop rotational cropping systems. For example, while farmers may be able to increase nitrogen to crops and decrease susceptibility to pests and diseases through crop rotations with leguminous crops, they may be able to earn greater profits through monocultures of crops. For example, in the **United States** maize grown in rotation with soybeans received deficiency payments was generally less profitable for farmers than continuous maize production in Iowa and Nebraska (Hrubovcak, *et al.,* 1999).

The amount of cover and residue left on the soil also affects soil quality and productivity and alters the effects of the soil on environmental quality. Cover crops are a management option to reduce nitrate leaching under cereal grain production. Soil organic matter in agricultural topsoils, derived from crop residues, organic manures, microbial biomass and soil microflora and fauna, plays a key role in maintaining soil quality, structural stability, and water-holding and buffering capacity. Crops that provide a high level of ground cover tend to have lower erosion rates compared to other crops. A cover crop of small grains, meadow, or hay planted in the autumn after harvest of a row crop provides vegetative cover to reduce soil loss, hold nutrients and add organic matter to the soil. Except for winter wheat, the cover crop is usually not harvested, but is sometimes grazed by livestock.

A study undertaken in **Sweden** on the effects of rye-grass cover crops on nitrate leaching in spring barley found that rye-grass cover reduced leaching by two-thirds in the first year and by more than 50% over a two-year period (Bergström and Jokela, 2001).

Soil residue cover provided by arable crops depends on tillage practices. For example, in **Canada**, the highest soil cover is provided under no-till and the lowest is produced under conventional tillage. Conservation tillage is associated with medium soil cover for maize, rapeseed and soybeans, and high for wheat, barley and oats (AAFC, 1996).

2.5.1.2. Tillage practices

Tillage systems are defined by the amount of crop residue remaining on the soil after the previous crop has been harvested. In the **United States**, conventional tillage leaves a maximum of 15% of the previous crop residue covering the soil, whereas conservation tillage maintains a maximum 30% of the previous crop residue covering the soil.

The adverse effects of conventional tillage practices (such as ploughing) on farm productivity and on the environment are being increasingly recognised (EEA, 2003*b*). The recurring disturbance of topsoil buries any soil cover and may destabilise the soil structure so that rainfall can cause soil dispersion, sealing and crusting of the surface. It often results in compacted soil which, in turn, negatively affects productivity.

In response to these problems, conservation tillage practices have been developed in a number of OECD countries. Conservation tillage reduces soil erosion and the risk of soil salinisation, and has the potential to improve surface-water quality (Derpsch, 2000; Pieri, *et al.,* 2002). It maintains and improves crop yields and resilience against drought and other hazards, while at the same time protecting and stimulating the biological functioning of the soil.

Studies in the **United States** found that pesticide use on maize, soybeans and wheat differs among tillage systems and it is difficult to distinguish the effects related to tillage systems from differences in pest populations between areas and from one year to the next, and from use of other pest control practices (USDA, 2003*b*). The study by Caswell, *et al.,* 2001, which is based on a detailed field-level survey across the US, found that tillage choice had no effect on yields for soybeans and maize.

Almost half of the area worldwide where conservation tillage practices have been applied is in the **United States,** although a considerable share of this is under monoculture. Adoption of conservation tillage has also increased over time. In the **United States**, for example, farmers employ conservation tillage practices on over 36% of planted area to maize and 56% of planted area to soybeans in 2000, compared with 30% in 1990 (Table 2.4).

Table 2.4. Adoption of alternative tillage practices in the United States, 1990-2000

	1990	1995	1997	2000
		(%)		
Corn				
Conservation tillage	**32.3**	**41.3**	**41.5**	**36.5**
No-till	8.7	18.1	17.5	17.9
Ridge-till	2.6	3.1	3.1	2.1
Mulch till	21.0	20.1	20.9	16.5
Non-conservation tillage	**67.7**	**58.8**	**58.5**	**63.5**
Reduced-till	24.4	22.6	24.2	23.2
Intensive-till	43.3	36.2	34.3	40.3
Soybeans				
Conservation tillage	**30.4**	**50.4**	**53.6**	**56.1**
No-till	9.6	30	30.5	32.8
Ridge-till	1.4	1	1	0.9
Mulch till	19.4	19.4	22.1	22.4
Non-conservation tillage	**69.6**	**49.6**	**46.4**	**43.9**
Reduced-till	24.2	20.8	20.2	18.8
Intensive-till	45.4	28.8	26.2	25.1
Small grains				
Conservation tillage	**24.4**	**31.2**	**32.2**	**30.4**
No-till	3.0	6.6	8.3	9.8
Ridge-till	0.0	0.0	0.1	0.1
Mulch till	21.4	24.6	23.8	20.5
Non-conservation tillage	**75.5**	**68.7**	**67.9**	**69.6**
Reduced-till	30.4	33.7	35	27.1
Intensive-till	45.1	35	32.9	42.5

Source: USDA (2003*b*).

The trend towards adoption of conservation tillage, and the corresponding decline in intensive tillage, is attributable to many factors including the prospect of higher economic returns with conservation tillage and by government policies and programmes promoting tillage for its conservation benefits. Higher economic returns resulting from conservation tillage stem primarily from increased or stable crop yields and an overall reduction in input costs, with both heavily dependent on the characteristics of the resource base and appropriate management.

Farm size and cropping practices affect the likelihood of farmers' adopting soil conservation and tillage practices. According to the ERS/USDA study (Caswell, *et al.,* 2001) farm size and cropping practices, especially crop type and use of crop rotations proved to be important determinants in the adoption of till conservation practices. However, the most important determinant was the influence of policies concerning areas such as conservation compliance and technical assistance.

2.5.2. Nutrient Management

Any method of crop production – extensive or intensive, conventional or organic – removes plant nutrients from the soil. Nutrient uptake varies according to the type of soil and the intensity of production. An increase in biomass production results in a higher plant nutrient uptake. As mentioned in earlier sections, the major nutrients required by arable crops are nitrogen, phosphate and potash.

Enhanced nutrient management aims to optimise the uptake of plant nutrients by the crop and thereby increase productivity. It involves efficient use of nutrients from commercial fertilisers and animal wastes. Enhanced nutrient management practices include improving existing practices in regard to assessing nutrient needs and the timing of applications, placing fertiliser closer to the seed, using alternative products, changing crop and irrigation management, and using manure and organic wastes. Nutrient management practices may have a significant effect on nitrogen fertiliser use and crop yields.

OECD countries use a wide range of nutrient management practices to enhance fertiliser use efficiency and reduce nutrient losses into environment. These practices, *inter alia*, include: assessing nutrient need through regular soil and crop tissue testing before applying nutrients; timing nutrient application to tailor feeding to crop-growth needs; applying nutrients close to the root zone; selecting the nutrient product according to the soil's chemical stability; rotating nitrogen-using with nitrogen-fixing crops; using nitrogen inhibitors and other products to retard the release of nitrates from ammonium fertilisers until later in the growing season; and applying manure and organic waste based on nutrient management plans.

Soil nutrient tests are carried out in almost all OECD countries. In **Australia**, the focus has shifted from broad regional fertiliser guidelines to site-specific nutrient management.[9] In the **United States**, results from the 1996 USDA Agricultural Resources Management Study survey of maize farmers indicate that soil tests were the most extensively used (44% of maize acreage), whilst nutrient-testing techniques were used only modestly. Numerous studies have examined the factors determining the adoption of nutrient management systems. A survey of the literature suggests that these are both regional and practice-specific (Christensen, 2002). Adoption depends on the method of farming in the region (*e.g.* irrigated or not), the type of soil, and the presence of regulation. Moreover, some tests, such as manure testing, may more commonly adopted by livestock farmers.[10]

2.5.3. Integrated Pest Management

Arable crop production systems suffer losses caused by diseases, weeds, insects and other pests. The goal of integrated pest management is to avoid or reduce yield losses by pests, while minimising the negative impacts of pest control through the application of the most appropriate pest control methods. Under the system of integrated pest management, the presence and density of pests and their predators and the degree of pest damage are systemically monitored.[11] Pest management practices include biological controls, cultural controls (including crop rotation and strategic controls such as planting dates and location) and the use of pest-resistant plans.

Integrated pest management can reduce the need for pesticides, which can also have a beneficial effect on the quality of groundwater. Unfortunately, quantitative evaluations of the uptake of integrated pest management in terms of hectares covered and reduction in pesticide use is only available for a few projects, making generalisation difficult.

Integrated pest management has been introduced in many countries and for many different arable crops. According to FAO, worldwide integrated pest management applied to rice has shown significant improvements in production, in some cases simultaneously reducing costs (FAO, 2003). In the **United States**, farmers have used integrated pest management for more than 20 years (Hrubovcak, *et al.,* 1999), but many of the techniques under the umbrella of integrated pest management have been used for some considerable time, the large-scale adoption of integrated pest management elements is a relatively new phenomenon.

Farm structure, including human capital, is an important factor in the adoption of integrated pest management. Studies in the **United States** have found that human capital and farm size had a positive effect on the uptake of modern integrated pest management technologies (Caswell, *et al.,* 2001). On the other hand, human capital had a negative impact on the use of the more traditional pest management strategy of destroying crop residues and farm size had no influence on the use of traditional pest management strategies of crop rotation and crop residue destruction. Cropping practices, especially crop choice and use of irrigation significantly affected the use of all of the pest management practices that were analysed. Moreover, natural endowment was found to be important in explaining farmers' use of traditional pest management technologies, but not the use of integrated pest management. Large farms are more likely to adopt integrated pest management than smaller farms. The availability of operator and unpaid family labour was found to be associated positively with integrated pest management adoption.

2.5.4. Organic farming practices

Organic farming is a method of production comprising a range of land, crop and animal management systems. It is based on minimising the use of synthetic chemical inputs such as fertilisers, pesticides, additives and medicinal products and represents a deliberate attempt to make the best use of natural resources. Organic agriculture is circumscribed by a set of rules enforced by inspection and certification mechanisms. Organic farming generates less stress for the environment than conventional agriculture, in terms of lower pesticide residues and soil erosion, increased biodiversity and resilience to drought (OECD, 2003*a*; FAO/WHO, 1999). Organic farming systems also have the potential to lower nutrient run-off and reduce greenhouse gases. There is evidence to suggest that organic farming and no-till are more effective in reducing soil erosion than conventional farming practices and, therefore, in maintaining soil productivity (Loucks, 2003).

However, the overall long-term effects of organic methods of food production on the sustainability of agriculture require more investigation. Although the environmental costs of organic systems are generally lower than those of conventional farming, their unit production costs are higher. Compared with conventional farms, organic yields on a given area of land are often lower and more variable (OECD, 2003*a*; FAO, 2003). In such cases, a significant expansion of organic farming could mean more land under cultivation, which may have an alternative value in terms of its potential use, depending on its current and historical use. From the perspective of potential environmental impacts on the arable crop sector, an expansion in crop production will have immediate impacts on land use and land-use change. The extent of the change in land use depends on the type of crop and the method of crop production introduced.

However, yields might be improved if agricultural research were to place greater emphasis on organic farming. Any comprehensive assessment of the value of different farming systems needs to take account of the relative economic, social and environmental costs and benefits of these systems in terms of varying yields, soil and water depletion, pollution, landscape, wildlife habitats, and animal and human health.

Organic farming systems for arable crops include practices such as organic fertilisation, manipulation of crop rotations and strip cropping, biological pest management and composting. Soil fertility and crop nutrients are managed through tillage and cultivation practices, crop rotation, and cover crops, supplemented with manure and waste material from crops and permitted synthetic substances. Crop pests, weeds and diseases are controlled through physical, mechanical and biological control management methods. Crops produced by organic grain and oilseed farmers include traditional grains and oilseeds such as maize, soybeans, wheat, barley, oats and rice, as well as non-traditional grains, including millet, buckwheat, rye and spelt.

Organic agriculture is practised in almost every country in the world, and its share of agricultural land, farms and production has accelerated in recent years. This shift has been encouraged by changes in consumer demand. Moreover, in some OECD countries, particularly in Europe, government support has been instrumental in the development of organic farming. The share of farm area accounted for by organic agriculture varies considerably in OECD countries, from under 0.2% in **Japan**, **Korea** and **Mexico**, to over 10% in **Austria** (Table 2.5).

For arable crops, as depicted in Table 2.5, there is considerable variation between countries, ranging from less than 1% of area harvested under arable crops in the majority of countries, to 6% in **Austria**. **Austria** has the highest share of land under organic arable production, followed by **Finland** and **Italy** (4%). In absolute terms, the **United States** has both the largest organic area devoted to arable crops as well the largest number of organic farms, followed by **France**.

In the **EU**, major growth of the organic farming sector has taken place in the last decade, following the implementation in 1993 of EC Regulation 2092/91, defining organic crop production. The widespread application of policies to support conversion to, and maintenance of, organic farming has been ensured by *Regulation 2078/92* in the framework of the agri-environmental measures (see Chapter 4). Land area under organic arable crops production has more than tripled in the EU since the early 1990s (Foster and Lampkin, 2000).

In **Australia**, rice is one of the most important organic crops. In **Canada**, organic grain production is the fastest-growing organic sector. In **Korea**, the market for organic products is still very small. In 2001, locally grown organic produce, comprising rice, fruits and vegetables, accounted for only 0.2% of total agricultural production. In **Mexico**, soybeans are amongst the most important organic crops.

In the **United States**, organic farming has been one of the fastest-growing segments of US agriculture for nearly a decade (Dimitri and Greene, 2002). Certified organic cropland for maize, soybeans and other major crops more than doubled from 1992-97, and doubled again between 1997-2001. Even so, less than 1% of maize, soybeans and wheat were grown under certified organic farming systems in 2001.

Table 2.5. Arable crop area under organic farming, 2001

Country	Arable Crops Sector (Cereals and oilseeds)[1]				Total Agriculture			
	Number of organic farms	% of total arable crop farms	Organic hectares[2] (1000)	% of arable crops area	Number of organic farms	% of ALL farms	Organic hectares (1000)	% of total area
Australia					1380	1.4	10500	2.3
Austria[3]	7804	7.0	77	6.0	18292	9.3	276	11.3
Belgium					694	1.0	22	1.6
Canada					3236	0.6	431	0.6
Czech Republic			19	1.0	654	2.4	218	5.1
Denmark					3525	5.6	175	6.5
Finland			51	4.1	4983	6.4	148	6.6
France	4600		78	0.7	10364	1.6	420	1.4
Germany					14703	3.3	632	3.7
Greece	879	0.3	4	0.3	6680	0.8	31	0.6
Hungary					1040		105	1.8
Japan[2]							5	0.1
Korea[2]					1237		1	0.0
Iceland					27	0.8	5	0.6
Ireland					997	0.7	30	0.7
Italy			250	3.5	56440	2.4	1230	7.9
Luxembourg					48	1.6	2	1.7
Mexico					34862	0.1	143	0.1
Netherlands[2]	576		12		1507	1.6	38	1.9
New Zealand					983		63	0.4
Norway					2099	3.1	27	2.6
Poland			19	0.2	1787	0.1	45	0.3
Portugal			16	3.0	917	0.2	71	1.8
Slovak Republic					82		59	2.4
Spain					15607	1.3	485	1.7
Sweden					3589	4.0	194	6.3
Switzerland	1414	4.3	5	2.2	5441	7.9	94	8.7
Turkey					18385	0.1	57	0.1
United Kingdom			57	1.7	3981	1.7	680	4.0
United States			299	0.3	6949	0.2	950	0.3

Notes:

1. For the Czech Republic: arable land; Finland: includes dried pulses; France: includes protein plants; United Kingdom: includes other crops.
2. The data for Japan refer to 1999, for Korea to 1998 and for the Netherlands to 2002.
3. Data from IACS.

Sources: Foster and Lampkin (2000); Yussefi and Willer (2003); USDA/ERS; Delegations.

2.5.5. *Factors influencing adoption of environmentally benign farming practices*

The environmentally friendly practices and technologies described above are interrelated and complementary, seeking to meet the dual goals of increased productivity and reduced environmental impact. Yet, experience today suggests that, despite their higher rate of returns, wide-scale adoption has not yet occurred across OECD countries. There are several reasons for the continuing dominance of conventional farming practices.

Each of the environmentally benign practices is "information- and management-intensive", because a farmer is required to have a thorough understanding of how the physical characteristics associated with farming, such as soil type, rainfall and temperature, interact with inputs such as pesticides, nutrients and soil, to affect crop production. Each practice uses inputs efficiently and may dramatically affect farm profits, the quality of the environment, and the pattern of natural resources (Hrubovcak, *et al.,* 1999). While decisions on the amount of conventional inputs to apply are made on a seasonal or annual basis, the adoption of new technologies entails extra costs for tools and equipment, and requires complex management skills. For example, production systems that include crop rotation are more complex, they require coherent management over the longer term. The adoption of information-intensive technologies requires a certain level of educational attainment on the part of the farmer. Evidence from the **United States** reveals that small grain farms are generally operated by older and less educated farmers than their counterparts on larger farms. Moreover, larger grain farms are more likely to use risk management strategies, conservation or no-till systems than operators of small farms. However, larger maize farms are likely to irrigate maize and to make heavier use of chemical inputs (Foreman, 2001).

The overall policy framework is also an important determinant of the type of environmentally benign practices adopted and their rate of uptake (OECD, 2001*b*). For example, commodity programmes that restrict base acreage to one or two crops could be an important impediment, as they encourage monoculture or the continuous planting of the same crop. In the **EU**, cuts in the compulsory set-aside rate brought about by the 1992 CAP reforms have encouraged some increase in the areas under cereal cultivation. In the **United States**, policy changes brought about by the *1996 FAIR Act*, including elimination of set-aside requirements, changes in prices and loan deficiency payments (*LDP*) led to some farmers transferring land previously used for maize production to the production of other crops, mainly soybeans or rotations with other crops (Lin, *et al.,* 2000). Farmers also adopted conservation tillage partly in response to incentives associated with conservation compliance provisions of the *1985 Food Security Act (FSA)* (see Chapter 5).

The fact that the nexus of environmental benefits-profitability exhibits spatial variation could be another factor hindering the adoption of environmentally benign technologies and farm practices. A given technology may be appropriate in one region, but inappropriate for another. Further, there could be environmental trade-off associated with the adoption of new technologies, as controlling one type of problem might exacerbate another (for example, it is possible that conservation tillage may reduce soil erosion, but increase herbicide use). The costs and benefits of conservation tillage vary according to farm and location. Studies in the **United States** comparing profitability of conservation and conventional tillage systems produced mixed results. Studies at the regional level for wheat found that higher yields resulted with conservation tillage than with conventional tillage in semi-arid areas (see Hrubovcak, *et al.,* 1999, for more discussion).

2.6. Transgenic crops[12] and the environment

The main purpose of this section is to summarise the current commercial status of transgenic crops and to identify some of the main environmental issues associated with them. It is not intended to provide an exhaustive overview of the *"GMO debates"*.[13]

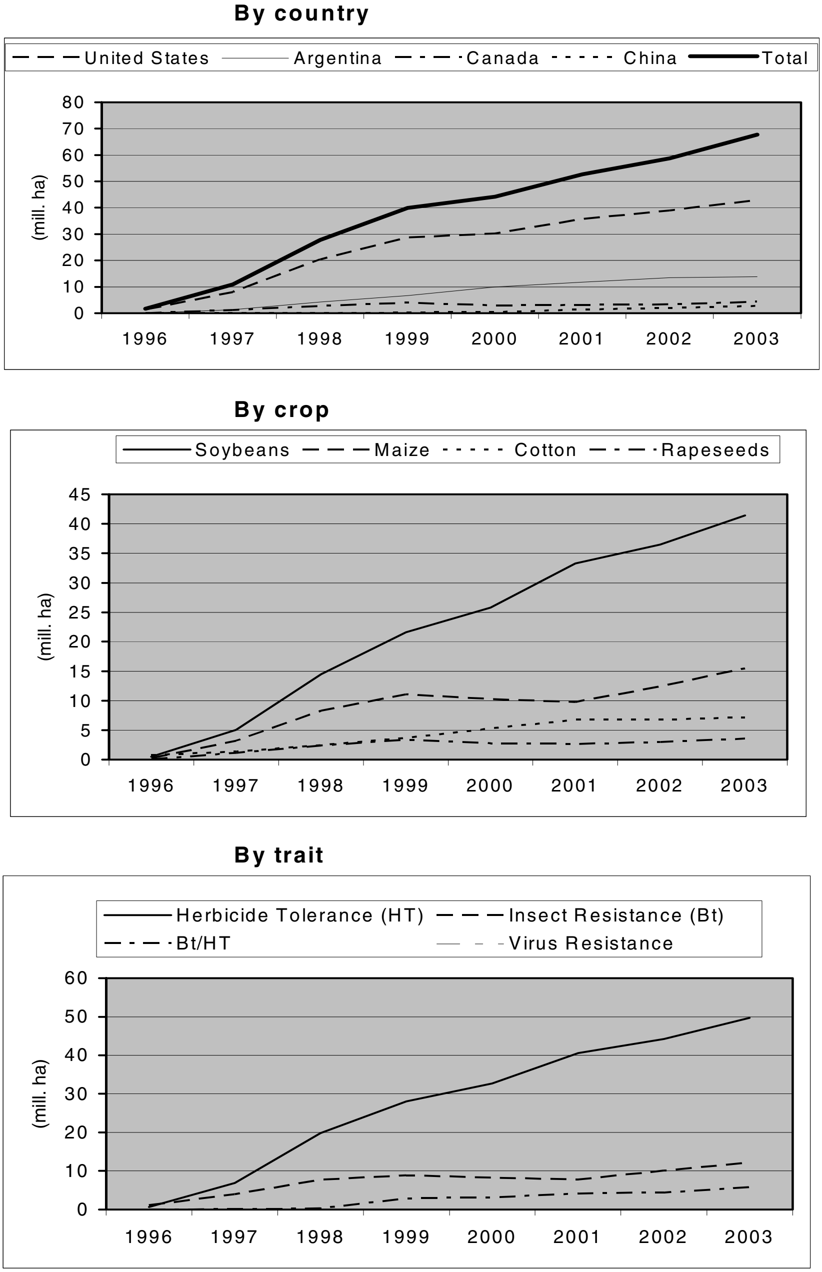
2.6.1. How widespread are transgenic crops?

The first transgenic crops became commercially available in the mid-1990s. Since then, their uptake has been rising. During the period from 1996 to 2003 there was a large increase in the area grown with transgenic crops worldwide, from 1.7 million ha in 1996 to 67.7 million ha in 2003 (Figure 2.3).

So far, adoption has been uneven across countries and commercialisation has involved only a few crops and traits. In 2003, two-thirds of the transgenic crop area worldwide was found in developed countries. Six countries, four crops (soybeans, cotton, maize and rapeseed) and two traits (insect resistance and herbicide tolerance) account for almost the totality of global transgenic crop area. The **United States** grew 63% of the global total, followed by **Argentina** (21%), **Canada** (6%), **Brazil** (4%), **China** (4%) and **South Africa** (1%).

In addition to the producing countries, many others have approved importation of transgenic crops for domestic consumption. In the **EU**, for example, 18 GMOs are approved for marketing, including amongst others GM maize, GM soy and rapeseed oil.

Figure 2.3. Global area of transgenic crops, 1996-2003



Source: James (2003).

Globally, most of this area is divided among four crops: soybeans (61%), maize (23%), cotton (11%) and rapeseed (5%). Of these crops, 55% of soybean acreage, 21% of cotton, 16% of rapeseeds and 16% of maize was transgenic in 2003. The uptake has been more rapid in the **United States,** growing from zero in 1996 to approximately 80% of soybean, 70% of cotton, and 38% of maize acreage being planted with transgenic varieties in 2003 (USDA, 2003*d*). Transgenic rapeseed is planted in two countries (**Canada** and the **United States**).

Currently, there are three main types of traits used in commercial cultivation: herbicide tolerance; insect resistance; and virus resistance. *Insect-resistant* transgenic crops are used as a way of controlling specific pests. Insect-resistant crops have been developed by integrating genes derived from various strains of a bacterium *Bacillus thuringiensis* (*Bt*), which produces toxins that kill certain insect pests, for example, the European maize borer and the Southwestern maize borer. Insect-resistance genes have been introduced in maize and cotton. For *herbicide-tolerant traits*, the insertion of a herbicide-tolerant gene into a plant enables farmers to spay wide-spectrum herbicides on their fields to control weeds without harming the crop. Herbicide tolerant crops include soybean, maize, rapeseed and cotton. *Virus resistance* genes have been introduced in tobacco, potatoes, papaya and squash. Transgenic crops have also been developed which involve two or more traits (*e.g. stacked events).* The most common stacked events at present are combinations of herbicide tolerance (HT) and insect resistance (*e.g.* Bt).

During the 1996-2003 period, herbicide tolerance has consistently been the dominant trait introduced, followed by insect resistance. Seventy-four percent of all transgenic crops in 2003 were herbicide tolerant, 18% insect resistant and a further 8% contained both these traits. HT soybean was the most dominant transgenic crop grown commercially (occupying 41.4 million ha or 61% of the global total), followed by Bt maize (13%) (James, 2003). OECD's Product Database (*http://www1.oecd.org/scripts/biotech/*) has information on most transgenic crops which have been approved for commercial use in OECD member countries.

Despite the focus of this discussion on the relatively small number of transgenic crops which have been commercialised so far, it is important to note that there is an impressive range of crops and traits in research and development, many of which have already been in field trials. Many of these are likely to be commercialised in the near future. It takes around a decade for a new transgenic crop variety to be developed from the field-trial stage to commercialisation. Arable crops in the pipeline include soybeans with improved animal nutritional qualities through increase protein and amino acid content; crops with modified oils, fats and starches to improve processing and digestibility, such as high stearate canola, low phytate or low phytic acid maize.

2.6.2. What are the environmental implications?

The environmental impact of transgenic crops may be either positive or negative. They may accelerate the damaging environmental effects of agriculture or contribute to more sustainable agricultural practices and the conservation of natural resources, including biodiversity depending on how and where they are used.

Releasing transgenic crops into the environment may entail risks such as gene transfer to wild relatives or conventional crops, weediness, trait effects on non-target species and other unintended effects. These risks are similar for transgenic and conventionally produced crops. Although scientists differ in their views on these risks, there appears to be an agreement on the need that environmental impacts should be assessed on a case-by-case basis and regularly monitored. Transgenic crops may also entail positive or negative indirect environmental effects through changes in agricultural practices such as pesticide and herbicide use and cropping patterns.

Main environmental benefits

The increasing cultivation of transgenic crops could contribute to more sustainable agriculture. Transgenic crops have been developed in order to increase the value or reducing the costs of producing crops. In addition to market effects, there could also be positive environmental impacts, depending on the crop and trait under consideration. These benefits include use of environmentally benign methods for managing weeds and insect pests due to smaller use of chemical inputs, thereby conserving biodiversity. Table 2.6 provides a snapshot of potential environmental benefits of transgenic crops, while Box 2.1 discusses the findings of selected empirical studies.

Productivity gains encompass higher returns on all factors of production or lower input requirements per unit of production. This could lead to higher crop yields (due to the presence of fewer insects or pests), lower pesticide and fertiliser applications, less demanding production techniques, higher product quality, better storage and easier processing. These gains should be assessed in comparison with conventionally produced crops, produced under the same production system. Ultimately, higher productivity may result in lower producer and consumer prices. Moreover, the reduction in production cost has the potential to raise rural incomes in developing countries in a similar way to the Green Revolution in large parts of Asia during the 1960s to 1980s (FAO, 2003).

Table 2.6. Potential environmental benefits of transgenic crops

Characteristics	Rationale	Examples
Productivity enhancements	Higher output per unit of land	High-yielding rice and maize
Herbicide tolerance	More efficient herbicide use and/or safer herbicide use	Glyphosate-tolerant soybeans, canola, maize
Disease/insect tolerance	Reduction in pesticide use and/or more efficient pest control	*Bt* cotton, maize, potatoes; virus resistant papaya, tobacco, melon
Tolerance to biological stresses	Improved resistance to droughts, easier production in marginal areas, easier nitrogen fixation	Research on drought-tolerant maize

Source: Nelson and de Pinto (1999; 2001).

Changes in pesticide use associated with the production of transgenic crops have been considered as an important possible impact (Royal Society, 1998; Ervin, *et al.,* 2000; Fernandez-Cornejo and McBride, 2002; Wolfenbarger and Phifer, 2000). Transgenic crops could lead to a reduction in the use of environmentally harmful chemicals to control weeds and pests because certain pesticides are no longer used, the frequency of treatments is reduced, or the area treated is reduced.

Due to higher yields, transgenic crops might lower pressures on land resources and diminishing the need for clearing the land or for land conversion, thereby leaving more area available for habitat protection and preservation. In the future, transgenic crops might become available that are resistant to drought (thereby saving water). Salinity-resistance of the soil could contribute towards the continuation of agriculture in regions affected by this phenomenon, which is primarily linked to irrigation.

Box 2.1. What does the empirical evidence show?

Several studies have attempted to assess non market benefits and impacts associated with transgenic crops (e.g. an annotated bibliography can be found at: www.isb.vt.edu). However, they are non conclusive, partly because of the novelty of such crops, because some of these crops have been grown for a short period and there are different approaches as to what should be the benchmark of comparison. Overall, available empirical evidence tends to suggest that yields are somewhat higher with transgenic crops than with their conventional counterparts, although there is significant variation by crop, location and year.

The National Center for Food and Agricultural Policy, which estimated the impacts of nine transgenic crops in the EU, found that collectively the nine transgenic crops have the potential to increase yields by 8.5 million tonnes per year, increase grower net income by USD 1.6 billion per year and reduce pesticide use by 0.014 million tonnes per year. Transgenic tomato would offer the greatest yield and grower income increase, while herbicide tolerant maize would have the largest reduction in pesticide use. The largest increase in yields is estimated for transgenic sugarbeet, whereas for glyphosate tolerant maize, wheat and rice yields would be unchanged (Gianessi, Sankula and Reigner, 2003). Traxler (2003) found that yields of glyphosate tolerant soybeans are not significantly different from yields of conventional soybeans in either the United States or Argentina. A study by USDA (1999a) reports that while glyphosate tolerant soybeans appear to have low yields, in some US Midwest regions, farmers planting Bt maize had yields 26% higher than conventional, non modified crops. Brookes (2003) found that Bt insect resistant maize in Spain on yields varies depending, inter alia, on location, climatic factors, timing of planting and on whether insecticides are used or not, with a country average yield benefit 6.3%. In Australia, the yield advantage GM rapeseed offers over non GM varieties is estimated to be 12.7% (Foster, 2003), while in Canada it is estimated at 10% (Serecon, *et al.*, 2001).

The evidence also suggests that changes in pesticide use rates have been variable (van den Bergh and Holley, 2001). For example, USDA studies found that, in the aggregate, as more farmers adopted transgenic crops, insecticidal treatments have been reduced on maize, whereas, the use of glyphosate, such as Roundup ®, on maize and soybeans has increased (USDA, 1999a and 1999b). However, the use of other, more toxic, chemical decreased. The situation varies by production method and by region.

Studies published so far on the effects of transgenic plants on agricultural biodiversity indicate that there is lack of consensus of the consequences of gene flow and conclude that more data and new models are needed to analyse the possible long-term unexpected effects of transgenes (Ervin and Welsh, 2005). The Farm-Scale Evaluation study initiated by the United Kingdom government compared biodiversity in fields of glyphosate-tolerant sugarbeet, maize and rapeseed with that in comparable plots of equivalent non-transgenic varieties in adjoining fields (DEFRA, 2003). The findings showed that there were differences in the abundance of wildlife between genetically modified herbicide tolerant crop fields and conventional crop fields. However, the study stressed that the differences found arose not because the crops have been genetically modified, but because the GM herbicide tolerant crops gave farmers new options for weed control. The differences depended on which and how herbicides were used.

There may also be other types of beneficial environmental impacts. Transgenic crops could contribute to savings in energy and air emissions or reductions in soil erosion due to less frequent operations in the field. Herbicide-resistant crops may lead to environmental benefits by letting farmers use herbicides that do need not to be incorporated with the soil, thereby encouraging a shift to no-till and conservation tillage practices.[14] In contrast to crops requiring conventional chemical applications, herbicide-resistant crops may thus reduce wind and water sediment damages by allowing for reductions in plowing. These techniques also facilitate the use of winter cover crops, thereby limiting nutrients leaching (*e.g.* nitrates). Certain transgenic crops in the pipeline could also increase removal of toxic heavy metals from the soil, either by incorporating them in the cells or transforming them to less toxic substances (Engel, *et al.,* 2002; Wolfenbarger and Phifer, 2000).

Main environmental concerns

In certain areas, where transgenic crops are released widely into the environment, the main potential environmental risks include impacts stemming from gene flow to wild relatives. The development of resistance to pests and viruses is equally possible, as in the case of conventional crops showing similar resistance, especially in the case of monogenic resistance.

An important environmental concern is the possibility that genes may be transferred by pollen or seed to populations of the same crop species or wild relatives in the surrounding area, if the gene(s) is considered to present a hazard. This is an especially important issue when considering the impact of a transgenic crop in its centre of origin and diversity, which can be considered as the geographic region where the crop has its largest diversity and where a close relationship exists with its wild relatives.

Many of these issues were explored at an OECD Conference, *LMOs and the Environment*, which was held in the United States in 2001. A special session at the Conference considered the preliminary evidence of gene flow from transgenic maize to local varieties in Mexico, as well as issues related to the conservation of maize diversity given the possibility of gene flow from transgenic maize.

Another potential environmental concern is whether the use of transgenic crops will have adverse impacts on non-target organisms or cause ecosystem damage. The Bt toxin, for example, may have adverse effects on non-target organisms like butterflies or beneficial insect populations that help control pests.

There are also issues associated with the potential impacts of transgenic crops on organic agriculture due to the inadvertent presence of transgenic crops or material in organic land. Organic farmers are not allowed to have transgenic content in seed or plants. For example, the **EU** Regulation for organic farming (*EC No. 2092/91*) forbids the use of LMOs. In July 2003, the European Commission published guidelines for the development of strategies and best practices to ensure the co-existence of LM crops with conventional and organic farming. They are intended to help EU member states to develop workable measures for co-existence in conformity with EU legislation. The guidelines set out the general principles and the technical and procedural aspects to be taken into account. Approaches to co-existence should be developed in a transparent way, based on scientific evidence and in co-operation with all concerned. Measures should be specific to different types of crop and regional and local aspects should be fully taken into account.

In June 2004, a law on co-existence was adopted by the Danish Parliament, which lays down rules on the cultivation of LMOs. The key elements of the law, *inter alia*, is capacity building with LM farmers, information sharing between LMO- and non-LMO farmers, crop specific measures such as distances and cropping intervals, to minimise the adventitious presence of LMOs in other crops and setting up a compensation scheme. The law will be evaluated regularly, with the first evaluation planned two years after its implementation.

2.6.3. Environmental impact assessments

All OECD countries (and many others besides) have a system of regulatory oversight in place for assessing the environmental safety of transgenic crops. In the majority of countries, these systems have been in place for a number of years; in fact, for well over a decade in many cases. As indicated above, a number of countries have approved the production and commercial use of such crops for human consumption or feed and have accumulated experience in risk/safety assessment of the large-scale use of transgenic crops in the environment. A far greater number of countries (the majority of OECD countries) have approved field trials of transgenic crop plants, which also involve a risk/safety assessment. Most countries continue to make changes and improvements to their regulatory systems in light of this experience.

In parallel with this, many OECD countries have continued to sponsor large research programmes designed to address risk and safety assessment questions related to the release of transgenic organisms to the environment. The results of this research have been used to inform and improve the practice of risk/safety assessment. Similarly, a large number of countries have undertaken national studies on the implications of agro-biotechnology. In general, OECD countries have shown a practical commitment to a proactive and scientifically-based approach to the risk/safety assessment of environmental applications of genetically engineered organisms.

National approaches to biosafety have been enhanced by successful multilateral activities aimed at developing a common approach to both the principles and practice of risk/safety assessment. Much of this common understanding was developed through work at the OECD where biosafety projects, addressing, *inter alia*, transgenic crops, have been in place since around 1985.

An authoritative description of the internationally accepted principles and practice of risk/safety assessment, as it relates to transgenic organisms, is given in a report by OECD's Working Group for Harmonisation in Biotechnology, which was prepared for the G8 Okinawa Summit in 2000 at the request of the G8 Heads of State and Government.

This report shows how environmental risk/safety assessment takes into account the biological properties of the host organism, the gene(s) introduced and their source, how the gene(s) is (are) expressed in the transgenic crop and the nature of the gene product. The characteristics of the organism are taken into account, as well as its likely performance and impact in the environment where it is to be released. For example, exposure and toxicity data are used to examine potential ecological effects to resident wildlife and biodiversity (for example, plants with pesticidal genes may impact non-target species of insects). In addition, information on the eventual use of the product is necessary to ensure a complete assessment. The kinds of information risk/safety assessors use have been developed, in part, from experience with traditional organisms. The general issues assessed for transgenic plants were developed by OECD and include the following: gene transfer, weediness, trait or non-target effects, genetic or phenotypic variability, and the use of vectors and genes from pathogens. The report of OECD's Working Group to the G8 describes the issues addressed by risk/safety assessors in greater detail.

It is important to note another significant multilateral effort, the Cartagena Protocol on Biosafety, which is a key international instrument dealing with "living modified organisms" (LMOs) in transboundary movements. The objective of this Protocol is to contribute to ensuring an adequate level of protection in the field of the safe transfer, handling and use of LMOs resulting from modern biotechnology that may have adverse effects on the conservation and sustainable use of biological diversity. It has established an advance informed agreement (AIA) procedure to ensure that countries are provided with the information necessary to make informed decisions before agreeing to the import of such organisms into their territory. The Protocol has also established a Biosafety Clearing-House

(BCH) to facilitate the exchange of information on, *inter alia*, LMOs used for Foods Feeds or Processing. The BCH also assists countries in the implementation of the Protocol.

2.6.4. Current and future trends

Despite the large degree of similarity among OECD countries in terms of risk/safety assessment, there remain major differences among countries on the topic of the safety of genetically engineered crops/foods. Most of these differences appear to be focused around "risk management" issues. In other words, the measures which are taken once an application has been the subject of a risk/safety assessment and has been approved for release to the environment. These measures include, amongst other things, the monitoring and detection of transgenic material following release, labeling of products, and measures designed to avoid the development of pest resistance to insect-tolerant crops.

Annex 2.A. Selected Data

Table 2.A1. Gross emissions of GHGs from field burning of agricultural residues, 1990 and 2001

(1 000 tonnes)

		1990					2001				
		CH_4	N_2O	NOx	CO	NMVOC	CH_4	N_2O	NOx	CO	NMVOC
Australia											
	Agriculture	8.8	0.3				12.6	0.4	20.8	492.1	28.7
	Cereals	7.1	0.2				11.4	0.3	16.3	444.5	25.9
	Wheat	4.0	0.1				6.5	0.1			
	Barley	0.9	0.0				1.5	0.0			
	Maize	0.1	0.0				0.3	0.0			
	Oats	0.4	0.0				0.3	0.0			
	Rice	1.2	0.0				2.2	0.1			
Greece											
	Agriculture						2.7	0.1	2.3	56.7	0.0
	Cereals	0.0	0.0	0.0	0.0	0.0	2.5	0.1	2.1	52.6	0.0
	Wheat						1.6	0.0			
	Barley						0.2	0.0			
	Maize						0.6	0.0			
	Oats						0.1	0.0			
	Rice										
Italy											
	Agriculture	0.6	0.0				0.5	0.0			
	Cereals						0.5	0.0			
	Wheat						0.3	0.0			
	Barley						0.0	0.0			
	Maize						0.0	0.0			
	Oats						0.0	0.0			
	Rice						0.0	0.0			
Japan											
	Agriculture	8.0	0.4	0.0	149.1	0.0	6.4	0.5	0.0	123.4	0.0
	Cereals	6.8	0.4	0.0	149.1	0.0	5.6	0.4	0.0	123.4	0.0
	Wheat										
	Barley										
	Maize	1.6	0.0				1.2	0.0			
	Oats										
	Rice	5.0	0.3				4.1	0.3			
Poland											
	Agriculture	1.5					1.3	0.1	0.0	0.0	0.0
	Cereals						0.4	0.0			
	Wheat						0.2	0.0			
	Barley						0.1	0.0			
	Maize						0.0	0.0			
	Oats						0.0	0.0			
	Rice										

Table 2.A1. (*continued*). **Gross emissions of GHGs from field burning of agricultural residues, 1990 and 2001**

(1 000 tonnes)

	1990					2001				
	CH_4	N_2O	NOx	CO	NMVOC	CH_4	N_2O	NOx	CO	NMVOC
Portugal										
Agriculture	0.9	0.1	2.1	177.8	0.0	0.8	0.1	1.9	16.1	0.0
Cereals	0.1	0.0	0.2	1.5	0.0	0.1	0.0	0.1	1.1	0.0
Wheat										
Barley										
Maize										
Oats										
Rice	0.1	0.0				0.1	0.0			
Spain										
Agriculture	2.9	1.0	35.5	61.1	8.6	2.9	1.0	36.1	60.8	8.5
Cereals	1.1	0.1				1.5	0.1	3.7	31.4	4.4
Wheat	0.3	0.0				0.7	0.0			
Barley	0.6	0.0				0.2	0.0			
Maize	0.1	0.0				0.1	0.0			
Oats	0.0	0.0				0.0	0.0			
Rice	0.0	0.0				0.1	0.0			
United Kingdom										
Agriculture	12.7	0.3	9.1	266.0	35.0	0.0	0.0	0.0	0.0	0.0
Cereals	12.7	0.3				0.0	0.0	0.0	0.0	0.0
Wheat	11.6	0.2				0.0	0.0	0.0	0.0	0.0
Barley	0.9	0.0				0.0	0.0	0.0	0.0	0.0
Maize						0.0	0.0	0.0	0.0	0.0
Oats	0.1	0.0				0.0	0.0	0.0	0.0	0.0
Rice						0.0	0.0	0.0	0.0	0.0
United States										
Agriculture	32.6	1.2	28.1	684.8	0.0	36.3	1.5	34.9	762.0	0.0
Cereals	24.6	0.6	13.7	516.0	0.0	24.5	0.6	13.4	514.4	0.0
Wheat	6.5	0.2				4.7	0.1			
Barley	0.8	0.0				0.5	0.0			
Maize	13.4	0.3				16.1	0.3			
Oats										
Rice	3.9	0.1				3.3	0.1			

Source: Greenhouse Gas Inventory Database, 2003.

Notes

1. Heavy metal contamination of soil can arise from the use of sewage sludge, fertilisers and copper-based fungicides. However, copper is not used in most arable farming systems (Boatman, *et al.*, 1999).

2. It has been argued that the on-farm economic costs of soil erosion, including the costs of lost soil biodiversity, are less than the off-farm costs of damage caused by sediment (Crosson, 2004). Furthermore, when markets do not function well and property rights are not well established, soil erosion and associated productivity losses are larger than would otherwise be the case (Claasen, *et al.*, 2004*a*).

3. Tobey (1991) looked at soil erosion and agrochemical use of the ten primary crops grown in the United States. In terms of soil erosion, soybean production was found to be associated with some of the highest levels of soil loss, at 17.5 metric tons per hectare, being exceeded only by tobacco.

4. The estimates of potential production losses should be treated with care as the true value of production losses depends on how farmers change management practices to address erosion.

5. The loss in agronomic productivity due to water-induced soil erosion in North America is estimated at 235 x 103 Mg/y for maize, 60 x 103 Mg/y for soybean, 75 x 103 Mg/y or wheat and 2 x 103 Mg/y for cotton. Globally, the value of annual production losses is estimated at USD 15 million in Africa, USD 98 million in Asia, USD 15 million in **Australia**, USD 15 million in **Europe**, USD 206 million in **North America** and USD 90 million in Central and South America. These losses represent an annual loss of 0.3% of the value of the global production of selected crops.

6. In a more recent study, den Biggelaar, *et al*. (2003) found that absolute yield loss caused by erosion ranged between 0.5 and 1.4 kg/ha/Mg of soil erosion for grain and leguminous crops, and between 0.7 and 127.0 kg/ha/Mg for root crops. In **North America**, crop yields declined at the rate of 0.4%/Mg of soil erosion.

7. Cropland includes areas used for the production of adapted crops for harvest. Two subcategories of cropland are recognised: cultivated and non-cultivated. Cultivated cropland comprises land in row crops or close-grown crops and also other cultivated cropland, for example, hayland or pastureland that is in a rotation with row or close-grown crops. Non-cultivated cropland includes permanent hayland and horticultural cropland (NRCS, 2003).

8. See Orlick, Bauer and Jeffrey (1995) for a literature review of crop rotation and tillage literature.

9. In **Australia**, the Ricecheck Programme was developed in 1986 to improve the system of rice management (AUDIT, 2001). It covers seven areas of crop management or component factors: environment (land suitability and safe pesticide use); productivity (field layout, sowing time, crop establishment, crop protection, crop nutrition, panicle initiation date and water management); and grain quality (harvest grain quality). Key checks are provided for each target, allowing for easy self-assessment.

10. Precision farming, defined as a systems approach to optimise crop yields through systematic gathering and handling of information about the crop and the field, has the potential to contribute to nutrient management by tailoring input use and application more closely to ideal plant growth and management needs. Results from the 1996 USDA Agricultural Resource Management Study found precision agriculture adopters more likely to operate larger farms, have more maize acreage and higher yields, and have higher educational attainment than non-adopter farmers.

11. For a detailed explanation see: *www.nri.org/ipmeurope/homepage.htm.*

12. Different countries have different preferences for terms which describe products of modern biotechnology. This document uses the term “transgenic crops” or “transgenic organisms”. For the purposes of this text, the term transgenic organisms is equivalent to the terms “genetically modified organisms” (GMOs), “genetically engineered organisms ” or “living modified organisms (LMOs)”.

13. There is a large and still increasing body of literature concerning the potential economic, social and environmental effects of transgenic crops (*e.g.* Ervin and Welsh, 2005; Ervin, *et al.,* 2000; Nelson and de Pinto, 1999 and 2001, Wolfenbarger and Phifer, 2000; NRC, 2003; Alvarez-Buylla, 2004; van den Bergh and Holley, 2001).

14. The two most common herbicides are Roundup Ready, with the effective chemical glyphosate and BASTA, with the effective chemical glufosinate (Wolfenbarger and Phifer, 2000).

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Table of Contents

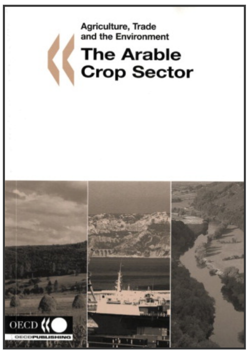
ACRONYMS AND ABBREVIATIONS

AP 2002	Federal Agricultural Law 2002 *(Politique agricole 2002),* Switzerland
ARP	Acreage Reduction Program, United States
AAFC	Agriculture and Agri-Food Canada
AAPS	Arable Area Payments Scheme, EU
ABARE	Australian Bureau of Agricultural and Resource Economics
AUDIT	National Land and Water Resources Audit, Australia
CAP	Common Agricultural Policy, European Union
CRP	Conservation Reserve Program, United States
CSP	Conservation Security Program, United States
CCP	Counter-cyclical Payments, United States
DEFRA	Department of Environment, Food and Rural Affairs, United Kingdom
DP	Direct Payments, United States
EFTA	European Free Trade Association (Iceland, Liechtenstein, Norway, Switzerland)
ERS	Economic Research Service of the USDA
ENS	Environmental News Service
ESAS	Environmentally Sensitive Areas Scheme, United Kingdom
EQIP	Environmental Quality Incentives Program, United States
EC	European Commission
EEA	European Environment Agency
EU	European Union
FSRI	Farm Security and Rural Investment Act, United States
FAIR	Federal Agricultural Improvement and Reform Act, United States
FAO	Food and Agriculture Organization of the United Nations
IEEP	Institute for European Environmental Policy, London
LEI	Agriculture Economics Research Institute *(Landbouw Economisch Instituut),* the Netherlands
LDP	Loan Deficiency Payments, United States
MLAP	Market Loss Assistance Payments, United States
NRI	National Resources Inventory, United States
NAFTA	North American Free Trade Agreement
PFCP	Production Flexibility Contract Payments, United States
PROCAMPO	Direct support for the countryside *(Programa de Apoyos Directos al Campo),* Mexico
RFISP	Rice Farming Income Stabilisation Programme, Japan
REPS	Rural Environment Protection Scheme, Ireland

SAPARD	Special Accession Programme for Agriculture and Rural Development (European Union, Czech Republic, Hungary, Poland, Slovak Republic)
UNFCCC	United Nations Framework Convention on Climate Change
USDA	United States Department of Agriculture
USITC	United States International Trade Commission
URAA	Uruguay Round Agreement on Agriculture
WRP	Wetland Reserve Program, United States
WHO	World Health Organization
WTO	World Trade Organization

TECHNICAL TERMS

AEI	Agri-Environmental Indicators
CRAM	Canadian Regional Agricultural Model
ESA	Environmentally Sensitive Areas
ESU	European Standard Unit, EU
GMO	Genetically Modified Organisms
GTAP	Global Trade Analysis Project
GFP	Good Farming Practices
GHG	Greenhouse Gas
HEL	Highly Erodible Land
LFA	Less Favoured Areas, EU
LMO	Living Modified Organisms
NPC	Nominal Protection Coefficient
PSE	Producer Support Estimate
SFP	Single Farm Payment, EU
TRQ	Tariff Rate Quotas
USMP	US Regional Agricultural Programming Model
UAA	Utilised Agricultural Area



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