Leaching Losses of Nitrogen from Agricultural Soils in the Baltic Sea Area

8

Arne Gustafson
Swedish University of Agricultural Sciences, Uppsala, Sweden

Background

Eutrophication by nitrogen of coastal zones and seas is a major and growing water problem for the Baltic Sea. Both point sources and diffuse sources contribute to the problems. This is fully clear from the results of pollution load compilation PLC5 of HELCOM- the Helsinki Commission (Figure 8.1)

In a wider European perspective analysis of source apportionment of nitrogen load in selected regions and catchments shows the importance of diffuse load to the water bodies. Agriculture is also a very significant contributor to this diffuse load (Figure 8.2).

In Sweden and other countries there is and has been an ongoing work to reduce the nitrogen contribution from large sewage treatment plants, by introducing tertiary treatment, a programme that seems to be successful. However in transition countries there is still substantial potential for reduction of these point sources (Table 8.1). This is especially important since point sources are emitted directly into stream waters in contrast to emissions from diffuse sources (as arable land) which are significantly subject to retention before entering the water bodies.

However the efforts to reduce the impact from arable land on the water bodies have so far, in spite of considerable efforts, shown only emerging evidence of **substantially** decreasing the nitrogen-loads in small streams. This is a tendency both in Sweden and in other countries around the Baltic Sea. Evaluation of this statement can be made

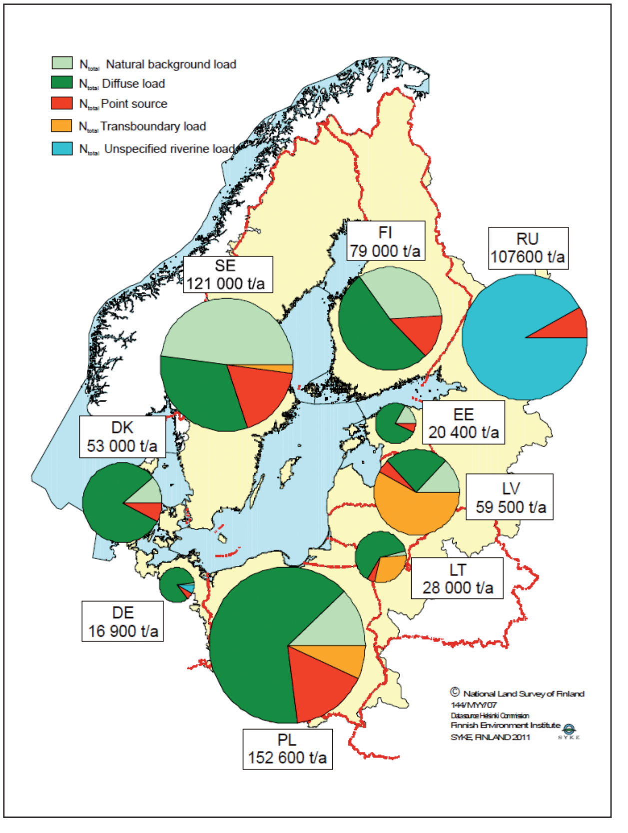


Figure 8.1. Nitrogen losses from diffuse sources into inland surface waters within the nine Contacting Parties' Baltic Sea catchment areas in 2006 based on the source-orientated approach (HELCOM, SYKE, Finland. PLC5, 2011).

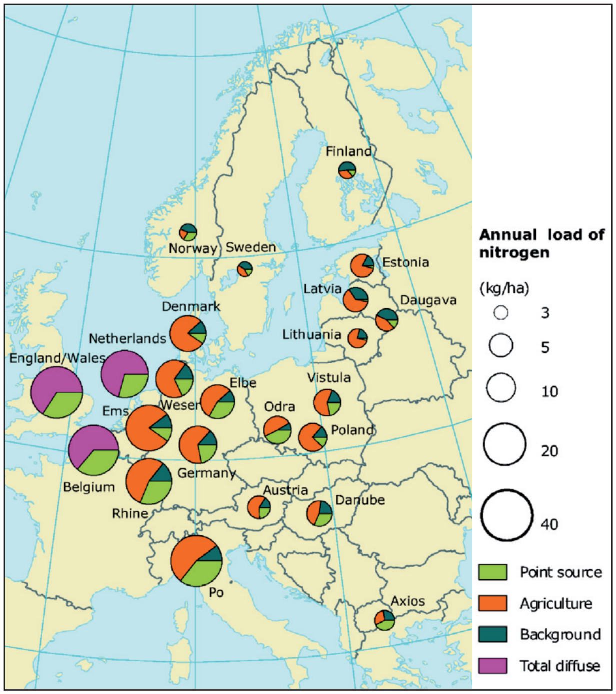


Figure 8.2. Source apportionment of nitrogen load in selected regions and catchments. The area of each pie chart indicates the total area-specific load. Mixed approaches. (European Environment Agency, EEA 2005)

Table 8.1. Levels of sewage treatment by country in 2004. Percentage of population connected to treatment plants of different levels.(from Humborg et al., 2007).

Country	Primary	Secondary	Tertiary
Belarus	0	50	0
Czech republic	0	61	0
Denmark	2	5.2	81
Estonia	2.2	34	34
Finland	0	0	80
Germany	0	9	85
Lithuania	33	6	18
Latvia	1.8	35	33
Norway	0	5.8	86
Poland	2.2	23	34
Russia	0	50	0
Sweden	0	5.8	86

FACT BOX 1

Overland flow:
The water flow takes place at the soil surface.

Preferential flow:
The water flow takes place in cracks and worm-holes.

Matrix flow:
The water percolates the whole soil profile.

through a network of small agricultural catchments in the Baltic Sea region. In Sweden a downward trend for nitrate nitrogen was found only in seven out of 24 agricultural catchments (Kyllmar et al., 2006) and downward trends in several minor rivers in Estonia and Denmark have also been reported (Iital et al., 2005; Kronvang et al., 2005).

Furthermore there are large differences in the leaching magnitude as shown by measurements made at the outlet of the catchments (Figure 8.3). There are lots of factors regulating the final leaching magnitude, such as soil types, farming practices, climatic conditions and denitrification rates in the plough layer and also deeper in the soil profile. Dominating water flow pathways are also important – i.e. *overland flow* or *subsurface flow*, the latter can be divided into *matrix flow* and *preferential flow*. We have also to take into account the interaction with the deeper groundwater system. (Gustafson, 1983; Vagstad et al., 2001). It is the interaction between agricultural practices and basic catchment characteristics, including the hydrological processes, that determines the final losses of nitrogen to the water bodies. It is necessary to stress that we need both a nutrient source and a transport mechanism to create a nutrient leaching situation.

Intensity of crop production increased after the Second World War. The breeding of high-yielding varieties of cereals and other crops, and chemical control of pests and diseases, required a higher input of mineral nitrogen, principally in the form of synthetic fertilisers. The amounts applied per hectare reached a plateau in the 1980s (Figure 8.4).

The largest outputs are normally in the form of crop off-take, but quantities of readily mineralisable nitrogen in the form of crop residues are also considerable in spite of greater percentage crop uptake in harvested products at actual fertilisation levels.

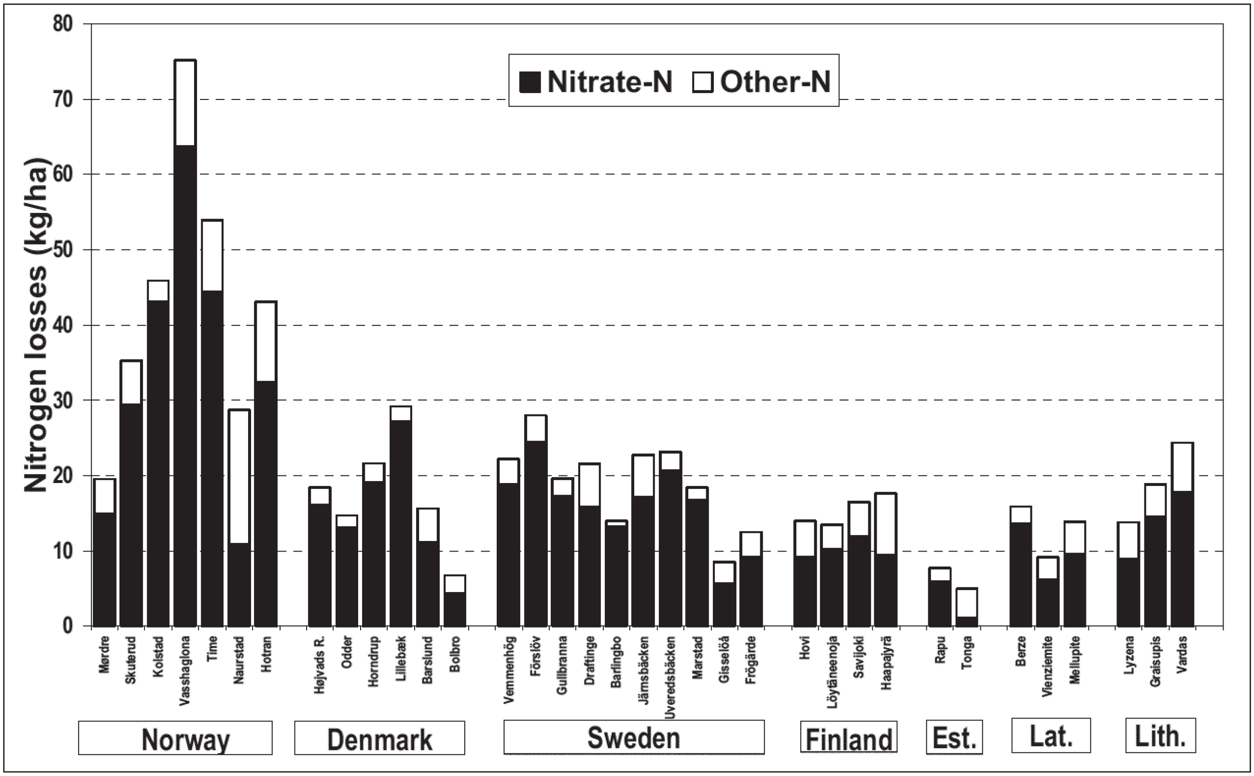


Figure 8.3. Mean annual losses of nitrogen from 35 agricultural catchments in the Nordic and Baltic countries (from Vagstad et al., 2001).

Animal-based systems have also been intensified over the same period, and large applications of nitrogen to arable land in the form of manure have become common. Losses of nitrogen from livestock-based agriculture have also increased with intensification, and contribute to a very significant part of the total losses from soils.

Gas losses are also important but little is known in detail of how to manage these losses under field conditions. It is also necessary to optimise the agricultural system in such a way that a decrease in losses in one way does not increase losses in another way. Thus a holistic knowledge of causes for nitrogen losses to water and air is of the utmost importance to be able to manage an environmentally friendly food production system.

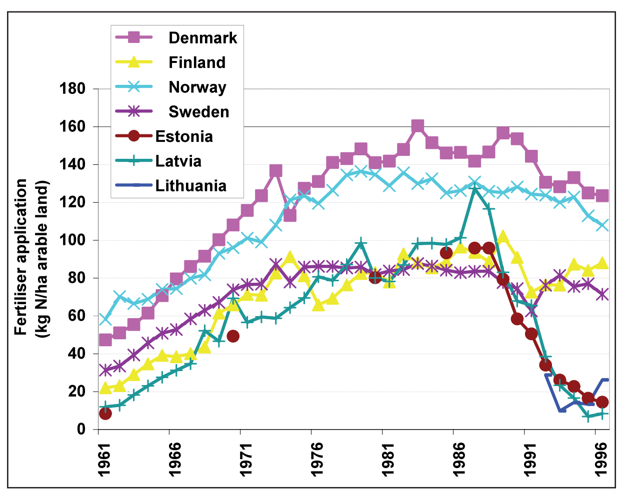


Figure 8.4. Changes in application of mineral nitrogen fertilisers (kg N ha^{-1} arable land) in the Nordic and Baltic countries from 1961 to 1996 (FAO, Statistics).

The Complexity of Nitrogen Losses to Water and Air – *Some Processes and Management Factors Involved*

Mineralisation/Immobilisation

As has been indicated earlier, fluxes of nitrogen through the soil by drainage water vary greatly. Only a part of available nitrogen is removed with the harvest and thus in spite of successful cropping, the losses might be high due to mineralisation of crop residues and easily decomposable organic material in the soil during the autumn period. Climatic conditions, pedological conditions, type of production and tillage management influence the mineralisation conditions. Soil disturbance through cultivation also increases the rate of mineralisation. The result is an increased amount of mineral nitrogen in the soil profile, which is vulnerable to leaching and /or denitrification.

Of major importance for the balance between mineralisation and immobilisation is the C:N ratio in the decomposing organic substances. Although there is a general trend relating net immobilisation to the C:N ratio, there is no precise critical value which marks the point at which reversal from immobilisation to mineralisation occurs (See page 129 for more information on immobilisation and C/N ratio). This is because other aspects of substrate quality have a major impact on the rate of decomposition. The rate of mineralisation of nitrogen from soil organic matter generally increases with increasing moisture content between permanent wilting point and field capacity. As the soil moisture content is raised above field capacity, however, mineralisation rates fall because of limited aeration.

It is not only moisture content that is important; temporal changes in content, i.e. cycles of drying and wetting, have a profound effect on the rate of mineralisation. There is evidence that rewetting of a dried soil results in a burst of microbial activity associated with an expansion in microbial populations. The substrate responsible for the stimulation is partly microbial cells killed during the drying phase, with a low C:N ratio, and partly soil organic matter newly exposed to microbial attack as a result of physical disruption of aggregates due to swelling and shrinking of the soil.

Freezing and thawing have comparable effects to those initiated by drying and rewetting. The freezing process kills a substantial part of the soil microbial biomass, which is then available for decomposition by the surviving population, once the temperature increases to allow the resumption of microbial activity. In conditions such as those of a Swedish winter, the effects of freezing and thawing may exceed those of drying and rewetting.

Rates of organic matter decomposition generally rise rapidly with increasing temperature, above the range normally found in soils in the field. This may result in large differences in the rate of nitrogen mineralisation between typically cool conditions in early spring, especially in the north of Sweden, and conditions in midsummer. This is of special interest because of the possible implications for organic farming systems.

In conclusion, mineralisation/immobilisation of nitrogen in soil is a complex process dependent on many factors. Much is known from laboratory experiments and much less from field experiments, especially for cold (autumn, winter, and early spring) conditions. The conditions during the cold period, however, play an important role in the leaching of nitrate to the water bodies. More should be known about the effect of catch crop management and the tillage regimes and more attention should be paid to this so that nitrogen leaching can be managed by proper control of the mobilisation/immobilisation processes.

Denitrification

Denitrification – the microbial reduction of nitrate to NO, N_2O and N_2 – is the major biological process by which the nitrogen cycle is completed and fixed nitrogen returned to the atmosphere. The environment in which the greatest quantities of nitrate, the essential substrate for denitrification, are likely to be found is agricultural land receiving substantial inputs of nitrogenous fertilisers or manure. Estimates of the quantities of nitrogen lost by denitrification from agricultural land differ widely; more than 50% of the applied nitrogen has been reported. There are concerns about N_2O since this gas is one of the more important contributors to the greenhouse effect and is also considered to be a partial cause of the depletion of the Earth's stratospheric ozone layer.

Recent research related to denitrification confirms greater losses in the presence of manure. Increased soil carbon content after long-term manure applications also promotes the process, as does straw incorporation. It appears to be a readily decomposable fraction of the organic matter that affects the capacity of soil to denitrify.

Denitrification rates are to some extent correlated with concentrations of nitrate in the soil. Where fertilisers containing nitrogen in the nitrate form are applied, much of the loss due to denitrification occurs in the period immediately following the application. This usually means that the maximum losses from cereal-growing land and grassland occur in spring, under Swedish conditions, with a tendency towards another peak in autumn from arable land, following the release of nitrate from the mineralisation of crop residues, and an increase in soil water content.

The effects of plants on denitrification are complex. On the one hand, they can promote it by providing carbon in the form of exudates and root cell material. On the other hand, water demand by the plants dries the soil and improves aeration; plant uptake of nitrate removes it from the danger of loss by denitrification.

Many studies have shown that denitrification activity in soils is correlated with water content. This dependence on water content is a direct consequence of the fact that the diffusion rate of oxygen through a water-filled pore is only one ten-thousandth of that through an air-filled pore. The potential for the development of anaerobic zones is thus to a greater degree dependent on water content than any other variable.

Agricultural land is a significant source of emissions of N_2O. Normally, but not always, increased fertiliser rates correspond to greater emissions. Several studies have shown that very high rates of N_2O emissions may occur when peat soils are drained and cultivated.

Soil pH is another factor affecting the ratio of N_2O to N_2 in the gaseous products of denitrification. Inhibition of N_2O reduction to N_2 occurs at all concentrations of nitrate at low pH, resulting in an increased proportion of the emissions occurring as N_2O. Studies have shown that the effect of acidity on N_2O is an immediate one, and thus not due to a change in the balance of microbial population.

The conclusion is that the possible risk for formation of N_2O lends great importance to the denitrification process and the manipulation strategies to avoid both major denitrification of a valuable N resource and the formation of N_2O. Not much is done under Nordic conditions concerning this issue and even less when it comes to interactions between mineralisation/denitrification and countermeasures against nitrogen leaching through different field management strategies. This must be an important field for research in the future.

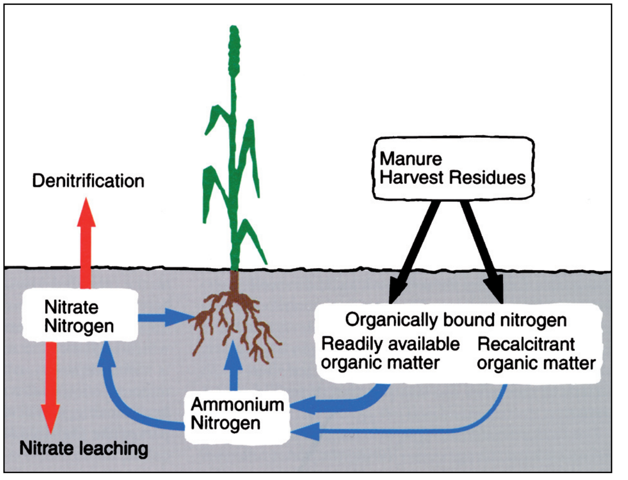


Figure 8.5. Soil nitrogen types, turnover, storage and losses (Claesson and Steineck, 1996).

Ammonia Volatilisation

Nitrogen can be lost from agricultural soils by the release of gaseous ammonia, NH_3, into the atmosphere. The predominant source of the ammonia in the farming systems is urea in the faeces and urine of livestock, either voided directly onto land by grazing animals or spread as slurry or farmyard manure. Ammoniac fertilisers also contribute to the release, when applied to calcareous soils. The ammonia lost to the atmosphere is a major contributor to acid deposition. Some of the NH_3 deposition is very local, within a few hundred metres of the source; at the other extreme, some is dispersed over large areas. Ammonia volatilisation contributes to acidification of land and in some limited areas even to nitrogen saturation in forest soils, as well as to eutrophication in lakes, rivers and the sea. It can also affect biological diversity negatively.

When urea is added to a soil, the urease enzyme rapidly hydrolyses it to ammonium and bicarbonate ions. The latter tends to raise the soil pH near the surface, and promote the loss of NH_3 by volatilisation. The amounts of ammonia lost are influenced by a number of factors, such as aerodynamic factors affecting the transfer of NH_3

from the soil surface to the atmosphere, the amount of urea applied, the rate of hydrolysis, the initial pH and the buffer capacity of the soil, the soil moisture level and the depth of application.

There exists rather good and detailed knowledge concerning individual processes regulating losses of NH_3 from arable soil but on the combined effect of all simultaneous ongoing processes there is still a considerable lack of knowledge. Research must therefore be directed towards a more complete understanding of the combined effect of all ongoing processes to reach the final goal of better utilisation of the nitrogen resource and thereby save the surrounding environment.

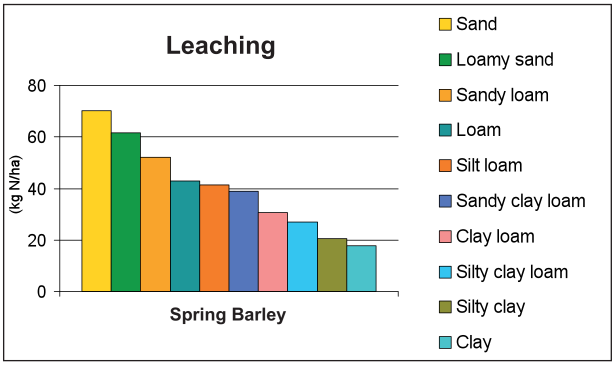


Figure 8.6. Leaching after spring barley in southern Sweden in relation to different soil textures. (Calculated values with the SOILNDB, Johnsson et al., 2002).

How Nitrate Moves in the Soil

Influence of Soil Texture

Of the various combined forms of nitrogen present in soils or added as fertiliser, only the nitrate ion is leached out in appreciable amounts by water passing through the soil profile. This is because there is no significant adsorption of nitrate onto soil surfaces, and there are no common insoluble nitrates. Thus nitrate in the soil solution is displaced downwards by rainfall or irrigation water and if sufficient water is added it can be carried beyond the root zone and eventually to the groundwater and/or to a tile drainage system if present.

The water content of the soil affects the rate of downward movement of nitrate during leaching. The depth of displacement by a given quantity of rainfall is generally greater for sandy soils than for clays, making sandy soils more vulnerable to leaching than clay soils (Figure 8.6). However, nitrate movement in the field is a complex process, and the effect of soil structure increases as clay content increases. Variations in pore size, in the spatial distribution of pores and their continuity all contribute to irregular movement of water down the soil profile. The effect of this is to spread out the front between the resident soil solution and the displacing rainfall, a phenomenon known as hydrodynamic dispersion. Superimposed on this effect is diffusive dispersion of nitrate in the soil solution, due to differences in concentration within the soil profile.

Recognition of the high hydrodynamic dispersion in structured soils has led to the concept of mobile and immobile water. The immobile water is retained in the aggregates, from which nitrate can only be transferred to the mobile water phase by diffusive transfer across the mobile-immobile water interface. This concept has been used with good effects in improving simulation of solute transport in structured soils. However, under intensive rainfall snow melt, water and solute may completely bypass the mobile pore system and move via large macro pores. The description of water movement under these conditions is being developed (Larsson and Jarvis, 1999), but detailed analysis of solute transport under these conditions is still not complete. One problem with improving the description of bypass transport is the highly transient nature of this type of transport. Time steps during simulation need to be the same order as rainfall events (i.e. hours rather than days), and data with such high time resolution are often lacking.

Sources of Leachable Nitrogen

Obviously, the size of the sources of nitrogen available for leaching will vary as regards both place and time. An example from a 9-year-old experiment on a clay-till in Skåne (southern Sweden) may serve as an example of the relative size of the sources in this part of the country (Table 8.2).

The amount of residual nitrogen and mineralisation during the winter in this example were of about the same magnitude. Atmospheric deposition was by far the smallest component. Almost half of the nitrogen available for leaching was in fact leached. Discharge was, on average, 237 mm. A larger discharge would have increased the leaching, while a smaller discharge would have decreased the leaching. Since the size of the discharge depends

Table 8.2. Sources of available nitrogen for leaching. Results from a 9-year investigation on clay till in southern Sweden. The dominant crops were spring wheat, barley and sugar beet. (from Gustafson, 1987)

Nitrogen source	Time or period of the year	N(kg/ha)
Nitrate in the soil, residual-N down to a depth of 1 m in the soil	1 Sept.	31
From mineralisation of litter and other organic material in the soil	1 Sept. – 31 March	34
Atmospheric deposition	1 Sept. – 31 March	6
Total available		71
Leached through drain pipes		31

largely on the amount of precipitation and its distribution, we are unable to influence the factor that regulates leaching apart from using irrigation. However, the amount of nitrogen available for leaching can be controlled to some extent. In the short-term, attempts can be made to reduce the amount of residual nitrogen by better dosing of the fertiliser in both amount and time. The amount of organic material available for mineralisation can also be influenced. This is particularly important in a long-term perspective. It is essential to attack both sources in order to achieve a sustainable reduction of leaching. Another possibility is to make use of catch crops during the winter so that the mineral nitrogen becomes incorporated into the plant material instead of being leached out; this is discussed in greater detail below.

The Role of Soil Organic Material

The availability of relatively easily mineralised organic nitrogen in the soil is, as has been shown, of major importance for the magnitude of the leaching. Soils given large amounts of organic material will, in the long-term perspective, have a larger capacity for net mineralisation. Agriculture with different lines of production and cropping systems will therefore, when "equilibrium" is finally reached after a fairly long period (decades), have clearly different contributions of net mineralisation from the soil. Both Swedish and foreign studies confirm this. It is mainly the semi-stable young humus pool in the soil that contributes to increased nitrogen mineralisation. This contributes to the nitrogen supply of the crops during the growing season but also to the formation of nitrate outside the growing season, which is less desirable from the leaching viewpoint.

Naturally, a good organic content has many positive effects on the soil, when regarded as an environment in which plants grow, but from the leaching viewpoint, the formation of organic material must not proceed too far. It is important to find an optimal situation. In a monoculture of grain crops where only fertilisers are applied there may, in the long run, be a reduction in the organic content, leading to undesirable effects on the soil structure, which may cause reduced crop growth and a decreased ability to utilise supplied and mineralised nitrogen. This should lead to increased leaching but if the monoculture is balanced with the ploughing-in of straw, the system can, nonetheless, survive for a long period and leaching losses may probably be kept at an acceptable level.

Mineralisation has been found to be greater on fields that are regularly treated with organic manure. Manure

Table 8.3. Mean annual losses and concentrations of nitrogen during a five-year period on sandy soils in southern Sweden when growing spring cereals with fertilization according to crops nutrient requirements and without manure for a long period of time (from Gustafson et al., 1990).

Site	Discharge(mm)	Losses N (kg/ha)			Concentrations N (mg/l)		
		NH_4	NO_3	Tot. N	NH_4	NO_3	Tot. N
Fertiliser							
1	239	0.09	31	33	0.04	13	14
2	263	0.06	31	35	0.02	12	13
3	232	0.06	31	35	0.03	14	15
Fertiliser and manure							
4	291	0.10	41	44	0.03	14	15
5 *	290	0.31	62	67	0.11	22	23

*Large application rates of manure

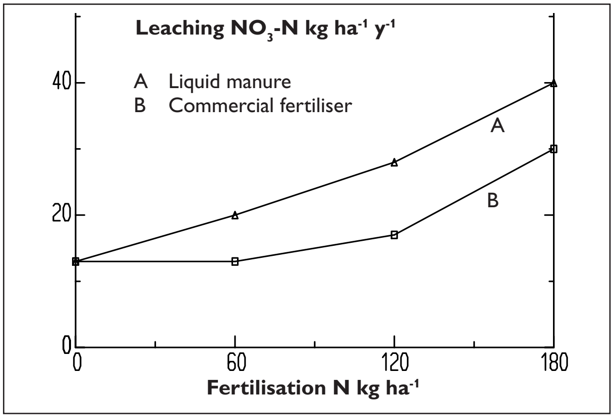


Figure 8.7. Mean annual leaching of nitrogen in tile drainage water following different fertilisation rates of commercial fertiliser and liquid pig slurry.

contains both mineral and organic nitrogen and the latter contributes to the enrichment of organic material in the soil. As a consequence, leaching under otherwise similar conditions will be greater on fields spread with manure or other organic fertilisers (Table 8.3).

Thus, it is important from a leaching point of view to take into account the organic part of manure when fertilising. In a five-year experiment, the impacts on leaching from commercial fertiliser and liquid manure were compared when adding equal amounts of inorganic nitrogen. The organic part in the liquid manure clearly contributed to elevated leaching magnitude at all fertilisation levels used (Figure 8.7). When using manure, a combination of manure and commercial fertilisers could be good, just to avoid to high losses from the organic part of the manure outside the growing season.

Climate-related Factors

Temperature and availability of water, oxygen and suitable nutrients control microbial processes. The longer the period between completed nitrogen uptake and the formation of frozen soil, the better the possibilities for enrichment of nitrogen in the soil. Both mineralisation and conversion to nitrate are favoured by good access to oxygen, heat and soil water. During summer, drought is often an inhibiting factor. Rainfall will then favour nitrogen mineralisation. In autumn, however, a shortage

Table 8.4. Mean nitrate losses by tile drainage water on a sandy soil in southern Sweden during (1991-94).

Fertilisation (N kg ha^{-1} y^{-1})	0	50	100	150
Losses (N kg ha^{-1} y^{-1})	34	36	45	66

of water is fairly unusual and then it is the availability of heat and oxygen that mainly restricts mineralisation. A warm autumn and early soil tillage, which increases the availability of oxygen in the soil at a time when its temperature is relatively high, will increase the autumn mineralisation and thereby the availability of nitrate and, consequently, possibly lead to increased leaching.

The colder conditions prevailing in the north of the Baltic area cause the formation of nitrate between the time of harvesting and the arrival of winter to decrease. Quite simply, there is insufficient time during autumn for particularly large quantities of nitrate to be formed, and as a result the leaching will be less the further to the north we proceed.

Another reason for the smaller leaching in the north is the different flow patterns of water as a result of frequently frozen ground. When the ground is frozen, a larger proportion of the water leaves as surface runoff and thus the soil is not leached of nitrate.

The increasing share of grassland in the north, where the soil has a crop cover during winter, together with late nitrogen uptake, also contributes to the leaching of nitrate in northern areas being relatively moderate. Consequently, there are considerable differences in leaching pattern and amount depending on the geographical location as can be demonstrated from the results from observation fields located from south to north in Sweden (Figure 8.8).

Overdoses of Fertilisers

Experiments illustrating the massive increases in leaching following excessive applications of fertiliser have been conducted in many countries. Results from a sandy soil in southern Sweden may illustrate this (Figure 8.9). Cereals were grown except in 1988 when the land was under set-aside and no fertilisers applied. In spite of this the leaching was high, illustrating the capacity of the soil to deliver mineralised nitrogen from the organic N-pool. Modern methods of predicting nitrogen requirement are available to ensure that excessive applications of fertiliser

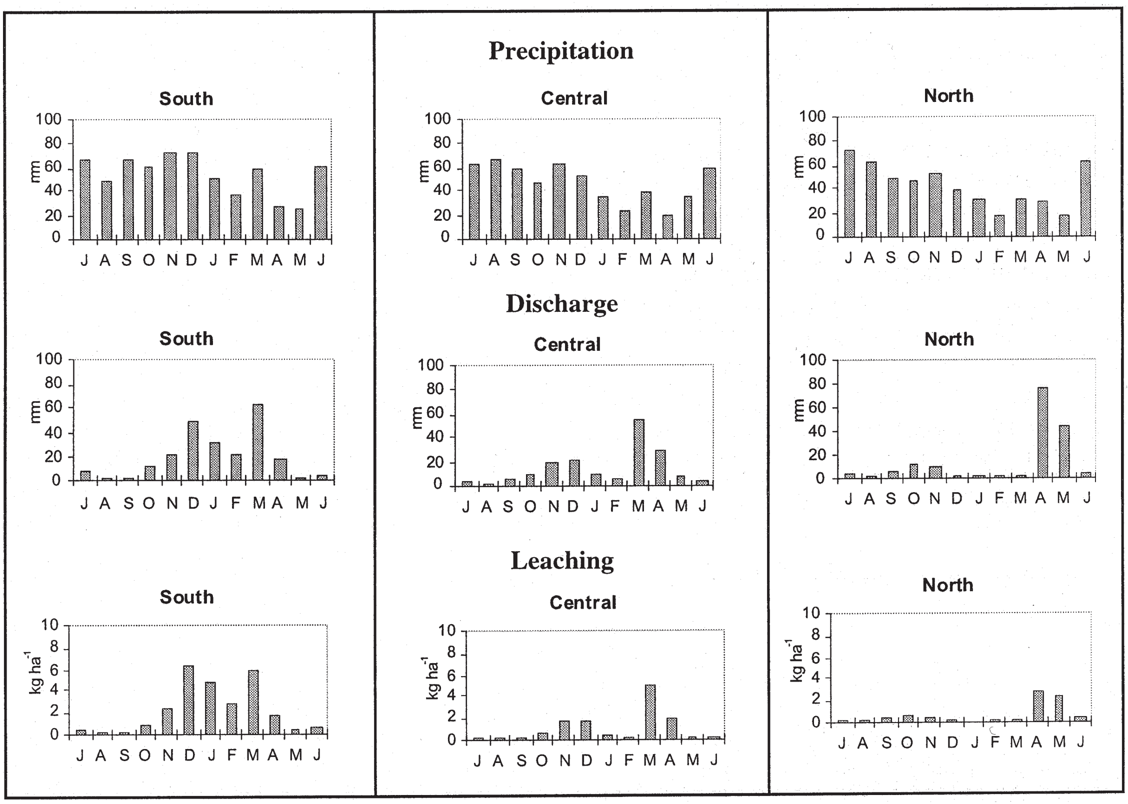


Figure 8.8. Precipitation, discharge and losses nitrogen, as mean values, from observation fields on clay and loamy soils in different parts of Sweden.

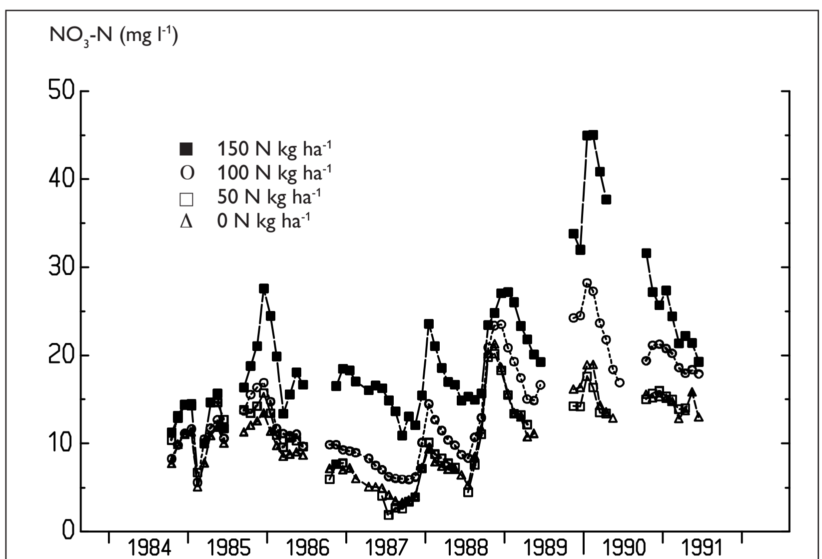


Figure 8.9. Effect of fertilisation levels on nitrogen concentrations in tile drainage effluent from a sandy soil. Recommended dose is 100 N kg ha^{-1} and 150 N kg ha^{-1} is an excessive dose. The crop rotation was : Barley (84), winter rye (85), oats (86), winter rye (87), fallow (88), winter rye (89), potatoes (90).

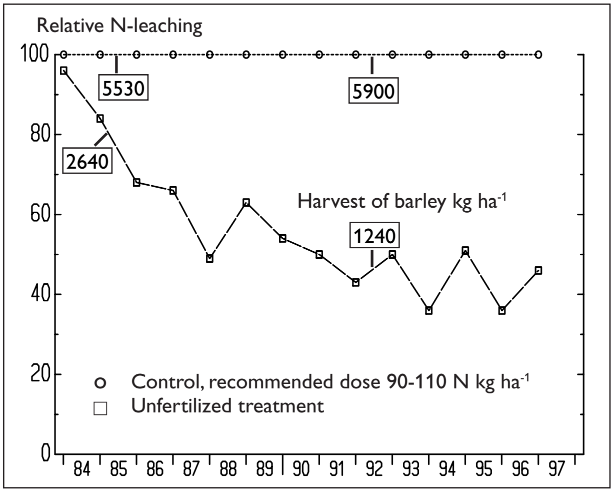


Figure 8.10. Relative N-leaching and harvest of barley (kg per hectare) in a treatment with recommended dose of fertiliser (yield=100) and a treatment without any N-fertiliser.

Figure 8.11. A well established ryegrass catch crop in the stubble of the main crop. Photo: A. Gustafson.

FACT BOX 2

Precision Farming

is an agricultural concept relying on the existence of in-field variability. It is about doing the right thing, in the right place, in the right way, at the right time. It requires the use of new technologies, such as global positioning (GPS), sensors, satellites, aerial images, and information management tools (GIS) to assess and understand variations. Collected information may be used to more precisely evaluate optimum sowing density, estimate fertilisers and other input needs, and to more accurately predict crop yields. It seeks to avoid applying inflexible practices to a crop, regardless of local soil/climate conditions, and may help to better assess local situations of disease or lodging

are not made. The importance of finding the right fertilisation level on every field each year must be stressed. Today precision farming techniques are also available so that farmers can automatically allocate the right dose within each field.

The results also clearly demonstrate that a reduction of the recommended dose by half, or avoiding the use of fertilisers, does not reduce the losses very much, at least in the short term (Table 8.4).

However, if fertilisers are not used, the yield will drop drastically (by half) in the second year and even more in the long run, since the nitrogen delivery capacity of the soil will decrease with time. A field trial in the Laholm Bay area in Sweden illustrates this. In the second year the yield of barley dropped by half in a zero-fertilised treatment compared with the control with a recommended dose of 90-10 N kg ha^{-1}. After 8 years the barley yield was only 20% of the control and the leaching 40% of the control (Figure 8.10).

The results demonstrate that the farmer cannot reduce the nitrogen level too much since yields will decrease. This is also meaningless from the leaching point of view. However the leaching magnitude from an environmental point of view might still be too high, even when using recommended fertilisation levels. In such cases the use of catch crops, and in some cases increased use of winter crops, can constitute possibilities for further decreasing the leaching magnitude.

Catch Crops

Many times not even optimal amounts of fertilisers or manure give an acceptable concentration in the tile drainage water. The nitrogen mineralised outside the cropping season must be utilised. Introduction of catch crops and increased use of winter crops can in such cases further reduce the leaching. A catch crop is grown over the winter or late in the autumn for no other purpose than to take up nitrate. The catch crops themselves have to be killed off by cold temperatures or ploughed in late in the autumn or the following spring. A typical undersown catch crop such as ryegrass is normally well established after the harvest of the main crop and ready to pick up available nitrate (Figure 8.11).

In an eight-year Swedish experiment, acceptable leaching losses were obtained, both after fertiliser applied

Table 8.5. Mean annual leaching for an eight year period on a sandy soil in southern Sweden

Time of application for liquid manure	Winter state	NO_3-N kg ha^{-1}	NO_3-N mg l^{-1}
Commercial fertiliser 90 N (kg ha^{-1}) in the spring			
	ploughed	44	16
	ryegrass catch crop	15	5
Pig slurry 90-110 Tot.N (kg ha^{-1}) and 45-55 N (kg ha^{-1}) commercial fertiliser in the spring			
Autumn	ryegrass catch crop	27	10
Spring	ploughed	49	15
Spring	ryegrass catch crop	19	8
Pig slurry 180-220 Tot.N (kg ha^{-1}) and 45-55 N (kg ha^{-1}) commercial fertiliser in the spring*			
Autumn	ryegrass catch crop	46	18
Spring	ploughed	63	23
Spring	ryegrass catch crop	40	16

*Overdose

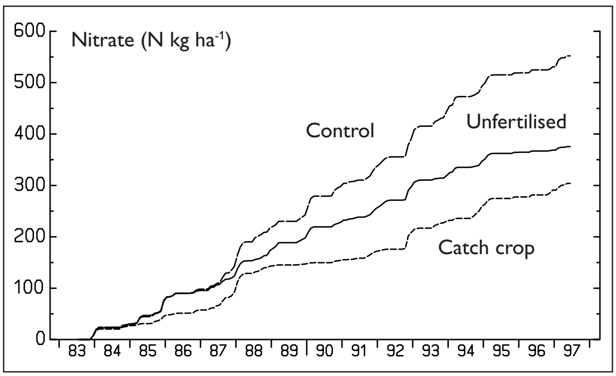


Figure 8.12. Cumulative monthly leaching of nitrate in a cropping system with mainly spring-sown grain crops. Two treatments received normal fertiliser doses (commercial fertilisers and manure, spring applications). One of these had a catch crop during the winter season, either winter rye (1984/89) or ryegrass (1989/96), ploughed in before sowing in the spring. The unfertilised treatment had no catch crop. In the first year (1983/84) all treatments were similar, with normal fertilisation rates and no catch crop.

FACT BOX 3

Organic Farming

is a form of agriculture that relies on crop rotation, green manure, compost, biological pest control, and mechanical cultivation to maintain soil productivity and control pests, excluding or strictly limiting the use of synthetic fertilisers and synthetic pesticides, plant growth regulators, livestock feed additives, and genetically modified organisms. Since 1990 the market for organic products has grown at a rapid pace, averaging 20-25% per year to reach $33 billion in 2005. This demand has driven a similar increase in organically managed farmland. Approximately 306,000 square kilometres (30.6 million hectares) worldwide are now farmed organically, representing approximately 2% of total world farmland. (IFOAM 2007:10) In Sweden, the increase in organic farming is to a large extent driven by political initiatives. The Swedish government has established the goal that by the year 2005, 20% of agricultural land in Sweden should be organically farmed. In 2009 this figure was 19%. For more information on organic agriculture see Part F, Chapter 37.

in spring and autumn or spring application of pig slurry in combination with fertiliser at normal doses, when growing ryegrass as catch crop (Table 8.5). The grass was sown in the main crop in the spring and remained during the winter before being tilled during the spring operations. However when an overdose of liquid manure was used in combination with fertiliser the leaching became too high, in spite of the ryegrass and time of application.

A long-term experiment (14 years) of continuous catch crop treatment (mainly rye grass) showed the sustainability of the catch crop system to decrease the leaching losses in a cereal-potato crop rotation (potatoes every fifth year) on a light soil in southern Sweden (Figure 8.12).

N-leaching in Organic Farming

In addition to what has been mentioned earlier as efficient countermeasures to reduce N-leaching, whole farming concepts have also been introduced as organic farming (see fact box).

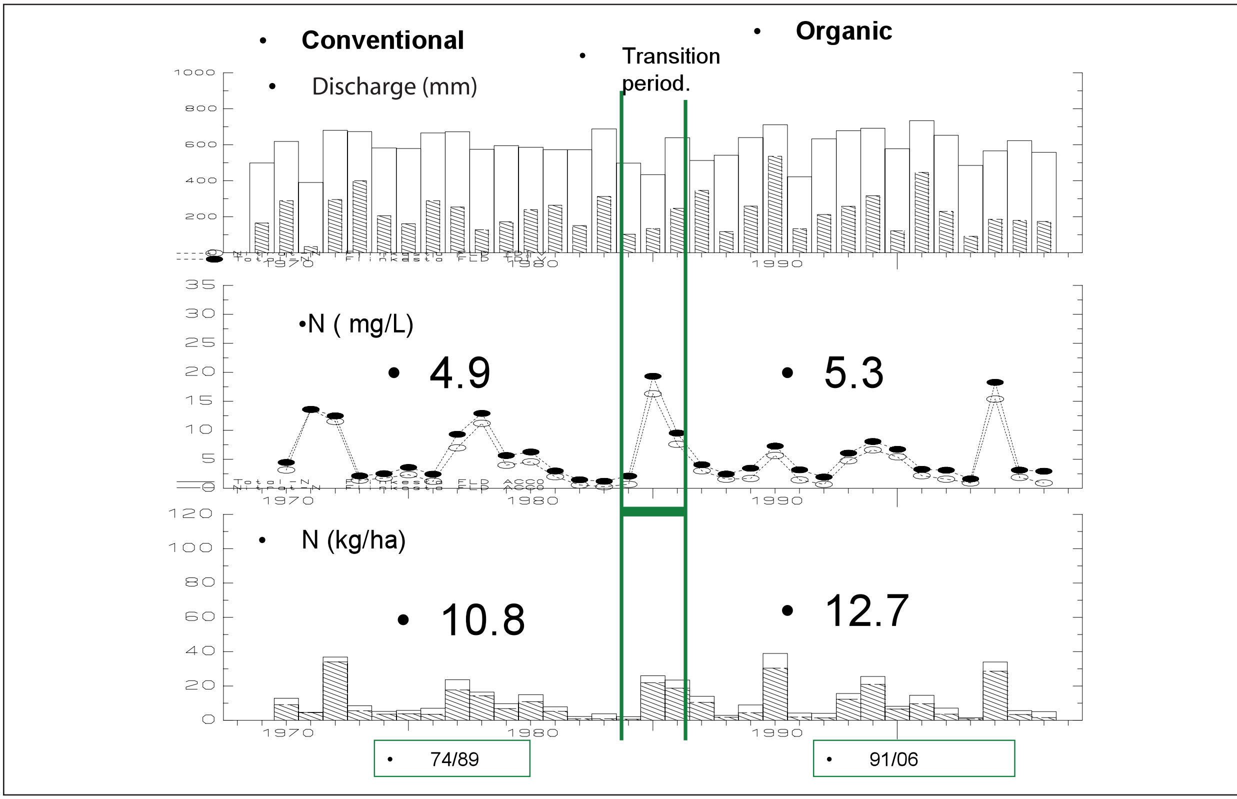


Figure 8.13. Leaching of nitrogen from an observation field in central Sweden before and after conversion from conventional to organic farming. Mean annual values as well as long-term mean values (15 years) for both the conventional and organic farming period. A transition period of three years is not included in the long-term values. (after Johansson and Gustafson, 2008).

Organic farmers use manures to dress crops with nitrogen or they use methods of supplying nitrogen in crop rotation. Growing a legume crop such as peas or beans brings nitrogen into the soil because bacteria living in association with these crops fix atmospheric nitrogen. Clover has the same benefit, so a field may be put down to grass with clover in it for a year or two at a time. Not all organic farmers have animals to supply manure and those that do not, rely heavily on crop rotations.

Plants must take up mineral nitrogen whether they are grown conventionally or organically. The ready availability of nitrogen from chemical fertilisers encourages fast growth and, if other conditions are favourable, large yields. Organic farmers usually produce less yield of what is perceived to be a higher quality and for which people may be prepared to pay a higher price. Arable organic farms may lose less nitrate by leaching than conventional ones but this is probably only when they are less productive. A long-term study from an observation field in central Sweden can confirm the small differences in leaching before and after transition from conventional to organic production on a dairy farm (Figure 8.13). Therefore, claims about water quality benefits associated with the use of animal or green manures should be viewed with great caution. This is especially critical for N due to the often poor synchronicity between release of inorganic nitrogen from animal or green manures and N uptake by the crop (Torstensson et al., 2006). Yields of cereal crops in organic systems can also be considerably lower than in conventional systems, which means that leaching losses per harvested crop unit can be significantly higher in the organic systems. (Aronsson et al., 2007)

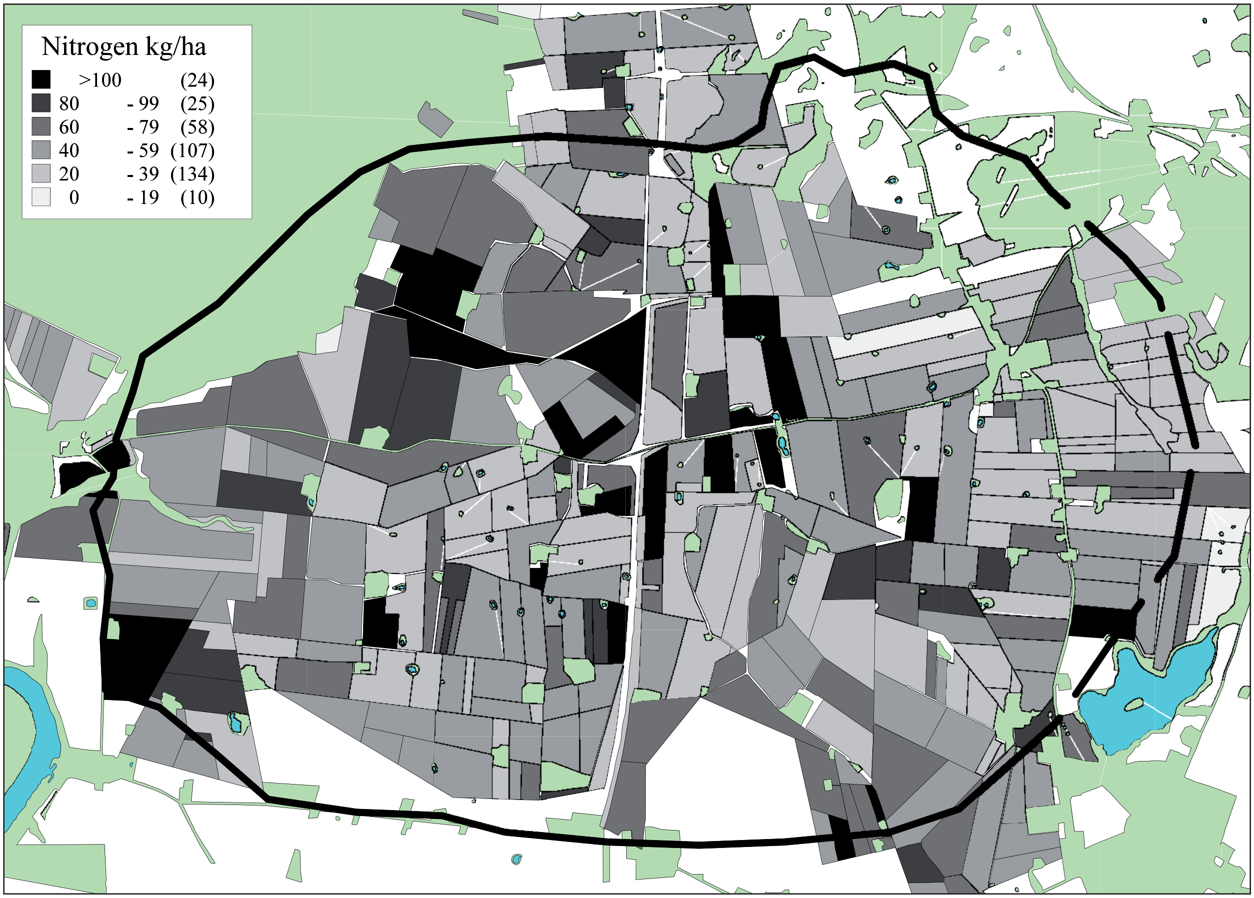


Figure 8.14. Spatial distribution of estimated nitrogen leaching (kg ha^{-1} yr^{-1}) from 331 fields in south-west Sweden as a function of soil type, fertilisation rate, crop type and soil tillage (within brackets = number of fields) (from Gustafson et al., 2000).

Perspectives of Counter-measures on the Farm and Local Watershed Level

Watershed Perspective

To be effective, the watershed perspective requires development and utilisation of more effective tools in water quality management work. Such tools include creation of a comprehensive watershed database concerning governing factors for nutrient losses on a suitable GIS media, indexing procedure to locate critical pollution areas within the watershed, and interaction between the GIS media and predictive mathematical modelling of nutrient losses to prescribe cost-effective and sustainable best management practices for pollution reductions. Without knowing the critical areas of concern, money and efforts may be spent wastefully or in the wrong order and non point source pollution may be hard to reduce. The advisory service in a region should have access to a GIS tool to be able to convince the farmer about necessary measures. An example of an analysis of leaching losses using a comprehensive database and a GIS tool for spatial distribution on a watershed level is demonstrated in Figure 8.14.

Ecotechnological Measures

Even if the farm is managed according to best management practices there still will be a need to do things close to or in water courses to achieve good water quality. One of these measures does not decrease leaching or emissions from soil, but increases removal of nutrients during runoff, i.e. restoration of ponds and wetlands.

The upper limit for N-removal is set by the hydrological conditions (Fleischer et al., 1994). Sedimentation of organic material must be favoured in order to obtain adequate conditions for denitrification at the sediment-water interface. In the long run, channel flow should be avoided by appropriate management.

Creation/restoration of wetlands has now become a part of the Swedish agro-environmental programme. One problem is, however, that ponds should be located at strategic sites in the watershed, rather than at sites identified by farmers. An inventory of optimal sites for a pond must therefore be made for each watershed subject to pond/wetland restoration. This can be included in the GIS tool and presented to the farmer.

Advisory Service and Co-operation Among Farmers

With an effective GIS tool, a programme for effective measures to be included in an environmental plan for good and sustainable farming and water quality can be set up for any watershed. The advisor and the farmer must cooperate in a positive way and the farmers can also work together to achieve the goals of the plan.

For water quality purposes the plan must as a minimum include proposals of measures to:

- Avoid overdoses of fertilisers.
- Improve manure management.
- Increase cultivation of winter crops, especially catch crops.
- Reduce soil tillage in autumn.
- Reduce erosion losses by leaving uncultivated strips of land alongside watercourses.
- Restoration or construction of ponds/wetlands in the watercourses to trap nutrients.

EU Directives, International Agreements and National Legislation and Regulations to Minimise Agricultural Leaching

Nitrate Directive 91/676/EEC

The aim of the Nitrate Directive (EU, 1991) is to reduce and prevent water pollution caused by nitrates from agricultural sources. The Directive obliges EU member states to monitor the nitrate concentration and trophic status of bodies of water. Member states must identify the bodies of water with a eutrophic level above 50 mg/l or those that might reach this eutrophic level if no action is taken.

Under the Directive, member states must designate vulnerable zones which include polluted waters. They must carry out measures to reduce nitrate pollution in these zones and also monitor water quality. Member states also need to draw up codes of good agricultural practices that can be taken up by farmers on a voluntary basis. Several member states did not fully comply with the Directive's requirements in time (mid-1990s).

Member states must submit implementation reports every four years. Based on these reports the Commission publishes a summary of the information received. If the reports show that the objectives have not been achieved, remedial action must be taken by member states.

The implementation of the Nitrate Directive is essential to achieving good water status. The Water Framework Directive has incorporated several aspects of the Nitrate Directive in its provisions. For example, the nitrate vulnerable zones became protected areas under the Water Framework Directive and the measures under the Nitrate Directive became the measures of the River Basin Management Plan.

EU Water Framework Directive 2000/60/EC

The EC Water Framework Directive, which came into force on 22 December 2000, establishes a new, integrated approach to the protection, improvement and sustainable use of Europe's rivers, lakes, estuaries, coastal waters and groundwater.

The Directive introduces two key changes to the way the water environment must be managed across the European Community. The first relates to the types of environmental objectives that must be delivered. Previous European water legislation set objectives to protect particular uses of the water environment from the effects of pollution and to protect the water environment itself from especially dangerous chemical substances. These types of objectives are taken forward in the Directive's provisions for Protected Areas and Priority Substances respectively.

However, the Directive also introduces new, broader ecological objectives, designed to protect and, where necessary, restore the structure and function of aquatic ecosystems themselves, and thereby safeguard the sustainable use of water resources. Future success in managing Europe's water environment will be judged principally by the achievement of these ecological goals.

The second key change is the introduction of a river basin management planning system. This will be the key mechanism for ensuring the integrated management of: groundwater; rivers; canals; lakes; reservoirs; estuaries and other brackish waters; coastal waters; and the water needs of terrestrial ecosystems that depend on groundwater, such as wetlands.

The planning system will provide the decision-making framework within which costs and benefits can be properly taken into account when setting environmental objectives and proportionate and cost-effective combinations of measures to achieve the objectives can be designed and implemented. It will also provide new opportunities for anyone to become actively involved in shaping the management of river basin districts – neighbouring river catchments, together with their associated stretches of coastal waters. The key dates for delivery of the requirements of the directives as listed in Table 8.6.

Table 8.6. The Water Framework Directive (WFD) past and future key dates for delivery of the requirements of the Directive.

Year	Requirement
Dec 2000	Directive comes into force
By Dec 2003	Transpose requirements to Member State Law; Identify River Basin Districts (RBD) and competent authorities
By Dec 2004	**Undertake RBD characterisation**: Pressures and impacts upon water status; Economic analysis of water use; Identify heavily modified and artificial waters; Monitoring programmes operational; Register of protected areas
By 2006	Monitoring programmes operational; Publish, for consultation, a work programme for River Basin Management Plan (RBMP) production;
By 2007	Publish, for consultation, interim overview of significant water management issues inriver basin district (RBD)
By 2008	Publish full draft RBMP for consultation
By 2009	Publish final first RBMP; Designate heavily modified water bodies; Environmental objectives; Programme of measures; Monitoring networks
By 2010	Introduce pricing policies
By 2012	Programme of measures operational
By 2013	**Review, for the first RBMP**: Characterisation assessments; Economic analysis; Publish, for consultation, interim overview of significant water management issues for second RBMP
By 2015	Achieve environmental objectives of first RBMP; Publish second RBMP
By 2021	Achieve environmental objectives of second RBMP; Publish third RBMP
By 2027	Achieve environmental objectives of third RBMP; Fourth RBMP

Baltic Sea Action Plan – BSAP

The HELCOM Baltic Sea Action Plan is an ambitious programme to restore the good ecological status of the Baltic marine environment by 2021. The final version of the Baltic Sea Action Plan was complete in the beginning of November 2007. It was adopted at the HELCOM Ministerial meeting which was held on 15 November 2007 in Krakow, Poland.

The Baltic Sea Action Plan addresses all the major environmental problems affecting the Baltic marine environment. However of the many environmental challenges, the most serious and difficult to tackle with conventional approaches is the continuing eutrophication of the Baltic Sea. Clear indicators of this situation include problems with algal blooms, dead sea-beds, and depletion of fish stocks. Such problems call for immediate wide-scale action to put an end to the further destruction of the Baltic Sea environment. Failure to react now would undermine both the prospects for the future recovery of the sea and its capability to react to the projected stress by the climate change. Furthermore, inaction will affect vital resources for the future economic prosperity of the whole region and would cost tenfold more than the cost of action.

Concerning inputs of nutrients which are responsible for eutrophication, HELCOM has already achieved a 40% reduction in nitrogen and phosphorus discharges (from sources in the catchment area) and likewise a 40% decrease as regards emissions of nitrogen to the air. But in order to achieve "clear water", which is one of the main objectives of the HELCOM Baltic Sea Action Plan, phosphorus and nitrogen inputs to the Baltic Sea must be further cut by about 42% and 18%, respectively.

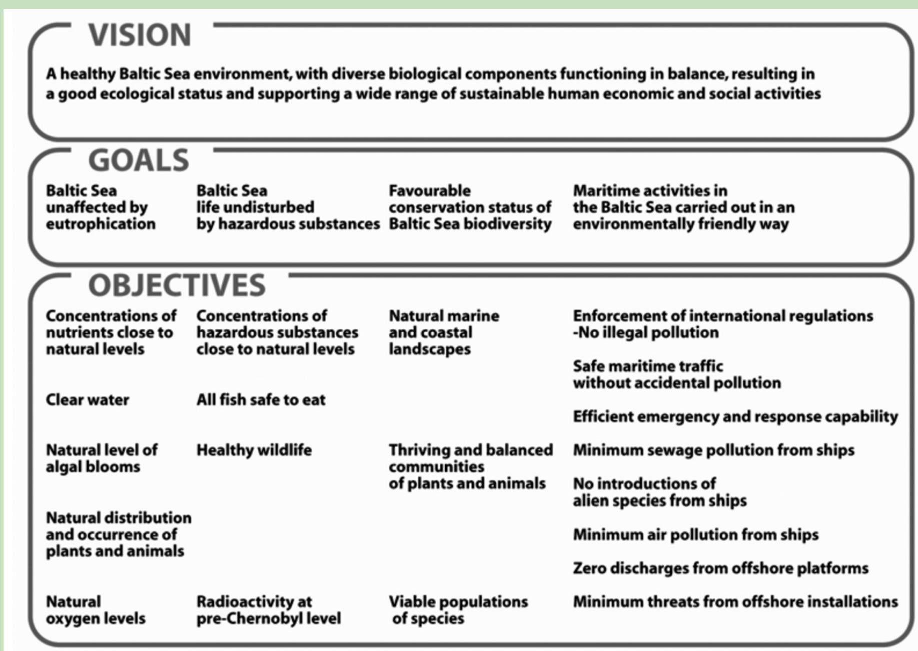
However, further progress cannot be achieved using only the old administrative measures of equal reductions in pollution loads. A completely different approach and new tailor-made actions are required to reach the goal of good ecological status. Moreover, the remaining challenges are more difficult than earlier obstacles. Reductions in nutrient inputs have so far mainly been achieved through improvements at major point sources,

Table 8.7. Country-wise provisional nutrient reduction burden in 2007.

Country	Phosphorus (tonnes)	Nitrogen (tonnes)
Denmark	16	17,210
Estonia	220	900
Finland	150	1,200
Germany	240	5,620
Latvia	300	2,560
Lithuania	880	11,750
Poland	8,760	62,400
Russia	2,500	6,970
Sweden	290	20,780
Transboundary Common pool	1,660	3,780

FACT BOX 4

The HELCOM system of vision, strategic goals and ecological objectives



National legislations and code of good agriculture practice
A comprehensive review of these issues in the European context has been published earlier (De Clercq et al., 2001) and is highly recommended to those interested in this matter.

such as sewage treatment plants and industrial wastewater outlets. Achieving further reductions will be a tougher task, requiring actions to address diffuse sources of nutrients such as run-off from agricultural lands.

The innovative approach is that the BSAP is based on a clear set of 'ecological objectives' defined to reflect a jointly agreed vision of 'a healthy marine environment, with diverse biological components functioning in balance, resulting in a good ecological status and supporting a wide range of sustainable human activities'. Example objectives include clear water, an end to excessive algal blooms, and viable populations of species. Targets for 'good ecological status' are based on the best available scientific knowledge.

HELCOM's plan is a cornerstone for further action in the Baltic Sea region, emphasising that the plan is instrumental to the successful implementation of the proposed EU Marine Strategy Directive in the region. The proposed EU Marine Strategy Directive foresees such an action plan for each eco-region, including the Baltic. HELCOM is in a unique position to deliver this already, given its embracing of all the countries in the Baltic Sea catchment area. HELCOM is also in a unique position to ensure that the special characteristics of the Baltic Sea are fully accounted for in European policies.

In order to reach the goal towards a Baltic Sea unaffected by eutrophication the BSAP includes an agreement on the principle of identifying maximum allowable inputs of nutrients in order to reach good environmental status of the Baltic Sea and further an agreement that there is a need to reduce the nutrient inputs and that the needed reductions shall be fairly shared by all Baltic Sea countries.

To identify maximum allowable input and the reductions needed, the Baltic Nest decision support system, including the MARE NEST model, was used (Johansson et al., 2007; Baltic Nest Institute; Mare model). This is believed to be the best scientific information available, and thus stressing the provisional character of the data. The conclusion is that the maximum nutrient input to the Baltic Sea that can be allowed and still reach good environmental status with regard to eutrophication is about 21,000 tonnes of phosphorus and 600,000 tonnes of nitrogen. Furthermore, based on national data or information from 1997-2003 in each sub-region of the Baltic Sea, the maximum allowable nutrient inputs to reach good environmental status and the corresponding nutrient reductions that are needed in each sub-region were calculated. In addition, country-wise provisional nutrient reduction burdens for each country were decided (Table 8.7).

Actions should be taken not later than 2016 to reduce the nutrient load from waterborne and airborne inputs aiming at reaching good ecological and environmental status by 2021.

According to the adaptive management principles, all figures relating to targets and maximum allowable nutrient inputs should be periodically reviewed and revised using a harmonised approach and updated information to be made available by the Contracting Baltic Sea countries. This should start in 2008, taking into account the results of the Fifth Pollution Load Compilation (PLC-5) and national river basin management plans.

In order to reach the above country-wise provisional reduction targets the countries must develop and to submit for HELCOM's assessment national programmes by 2010 with a view to evaluate the effectiveness of the programmes at a HELCOM Ministerial Meeting in 2013 and whether additional measures are needed.

The countries must also identify and, where appropriate include the required and appropriate measures into national programmes/River Basin Management Plans of the EU Water Framework Directive (Directive 2000/60/EC) for HELCOM Contracting States that are also EU member states.

Chapter 8

Aronsson, H., Torstensson, G. and Bergström, L. 2007. Leaching and crop uptake of N, P and K from a clay soil with organic and conventional cropping systems. In: *Soil Use and Management* vol. 23 71-81

Baltic Nest Institute (http://nest.su.se/)

Claesson, S. and Steineck, S. 1996. *Plant nutrient management and the environment*. SLU.

De Clercq, P., Gertsis, A.C., Jarvis, S.C. and Neeteson, J.J. 2001. *Nutrient management legislation in European countries*. Wageningen: Wageningen Pers.

EEA, 2005. *EEA web site*, http://www.eea.europa.eu

EU, 1991. *Nitrate directive*. http://ec.europa.eu/environment/water/water-nitrates/index_en.html

EU, 2000. *Water framework directive* http://ec.europa.eu/environment/water/water-framework/index_en.html

Fleischer, S., Gustafson, A., Joelsson, A., Pansar, J. and Stibe, L. 1994. Nitrogen removal in created ponds. In: *Ambio*, 23, 1994 pp. 349-357.

Gustafson, A. 1983. *Leaching of nitrate from arable land into groundwater in Sweden*. Environmental Geology, Vol. 5, No 2, pp. 65-71

Gustafson, A. 1987. *Simulation of Nitrate Leaching from Arable Land in Southern Sweden*. Acta Agriculturae Scandinavica, 38, pp. 13-23.

Gustafson, A., Kyllmar, K. and Ulén, B. 1999. Leakage of nitrogen and phosphorous. In: Lundin, L-C. (ed.) *Water use and management*. Uppsala: The Baltic University Programme. pp. 111-123.

Gustafson, A., Fleischer, S. and Joelsson, A. 2000. A catchment oriented and cost-effective policy for water protection. In: *Ecological Engineering*, 14, pp. 419-427.

HELCOM, 2007. *Baltic sea action plan* (adopted on 15 November 2007 in Krakow, Poland). http://www.helcom.fi/BSAP/ActionPlan/en_GB/ActionPlan/

Humborg, C., Mörth, C.-M., Sundbom, M. and Wulff, F. 2007. Riverine transport of biogenic elements to the Baltic Sea – past and possible future perspectives. In: *Hydrol. Earth Syst. Sci. Discuss*., 4, pp. 1095-1131.

Johansson, G. and Gustafson, A. 2008. *Observation fields on arable land. Discharge and nutrient losses for the agro-hydrological year 2006/07 and a long term review.* Technical report no. 121. Uppsala: Swedish University of Agricultural Sciences. (In Swedish) http://www-mv.slu.se/Vv/jrk/obs/obs_result.htm

Johnsson, H., Larsson, M., Mårtensson, K. and Hoffmann, M. 2002. SOILNDB: a decision support tool for assessing nitrogen leaching losses from arable land. In: *Environmental Modelling & Software,* 17, pp. 505-517.

Johansson, S., Wulff, F. and Bonsdorff, E. 2007. The MARE Research program 1999-2006: Reflections on Program Management. In: *Ambio*, 36, No. 2-3. pp. 119-122.

Iital, A. Stålnacke, P., Deelstra, J., Loigu, E. and Pihlak, M. 2005. Effects of large-scale changes in emissions on nutrient concentrations in Estonian rivers in the Lake Peipsi drainage basin. In: *J. Hydrol.,* 304, pp. 261-273.

Kirchmann, H. et al. 2007. Comparison of Long-Term Organic and Conventional Crop-Livestock Systems on a Previously Nutrient-Depleted Soil in Sweden. In: *Agronomy Journal* 99:960-972. doi:10.2134/agronj2006.0061.

Kronvang, B., Jeppesen, E., Conley, D.J., Sondergaard, M., Larsen, S.E., Ovesen, N.B. and Carstensen, J. 2005. Nutrient pressures and ecological responses to nutrient loading reductions in Danish streams, lakes and coastal waters. In: *J. Hydrol*., 304, 274-288.

Kyllmar, K., Carlsson C., Gustafson A., Ulen B. and Johnsson H. 2006. Nutrient discharge from small agricultural catchments in Sweden – Characterisation and trends. In: *Agriculture Ecosystems & Environment* vol 115, pp. 15-26

Kynkäänniemi, P. and Kyllmar, K. 2008. *Nutrient losses in small agricultural catchments 2006/07*. Ekohydrologi nr 101. Uppsala: Swedish University of Agricultural Sciences. (In Swedish) http://www-mv.slu.se/Vv/publ/Ekohydrologi_101.pdf

Larsson, M.H. and Jarvis, N.J., 1999. Evaluation of a dual-poporosity model to predict field-scale solute transport in a macroporous soil. In: *Journal of Hydrology* 215, pp. 153-171.

MARE NEST model, http://nest.su.se/nest/

Torstensson, G., Aronsson, H. and Bergström, L. 2006. Nutrient use efficiency and leaching of N, P and K of organic and conventional cropping systems in Sweden. In: *Agronomy Journal*, Vol 98, p 603-615.

Vagstad, N., Stålnacke, P., Andersen, H.E., Deelstra, J., Gustafson, A., Iital, A., Jansons, V., Kyllmar, K., Loigu, E., Rekolainen, S., Tumas, R. and Vuorenmaa, J. 2001. *Nutrient losses from agriculture in the Nordic and Baltic countries. Results of measurements in small agricultural catchments and national agro environmental statistics.* TemaNord 2001:591. 74 pp

Willer, H. and Yussefi, M. (eds.) 2007. *The world of organic agriculture – Statistics and emerging trends 2007*. International Federation of Organic Agriculture Movements (IFOAM), DE-Bonn and Research Institute of Organic Agriculture, FiBL, CH-Frick.

Chapter 9

Antikainen, R., Lemola, R., Nousianen, J.I., Sokka, L., Esala, M., Huhtanen, P. and Rekolainen, S. 2005. Stocks and flows of nitrogen and phosphorous in the Finnish food production and consumtion system. In: *Agriculture, Ecosystems & Environment*, Vol 107, Iss. 2-3 Pp 287-305.

Astover, A., Roostalu, H., Lauringson, E., Lemetti, I., Selge, A., Talgre, L., Vasilier, N., Mõtte, M., Tõrra, T. and Penu, P. 2006. Changes in agricultural land use and in plant nutrient balances of arable soils in Estonia. In: *Archives of Agronomy and Soil Sciences* 52(2): pp. 223-231.

Baker, D.B. and Richards, R.P. 2002. Phosphorus budgets and riverine phosphorus export in northwestern Ohio watersheds. In: *Journal of Environmental. Quality.* 31:1, pp. 96–108.

Beegle, D. and Lanyon, L. 2006. *Phosphorus balance*. SERA 17. Minimizing phosphorus losses from agriculture. (http://www.sera17.ext.vt.edu/SERA_17_Publications.htm)

Brandt, M., Ejhed, H. and Rapp, L., 2006. *Nutrient loads to the Swedish marine environment in 2006*. Swedish Environmental Protection Agency.