Chapter

21

Nitrogen as a threat to European soil quality

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Executive summary

Nature of the problem

- A large part of agricultural soils in Europe are exposed to high N inputs because of animal manure and chemical fertiliser use. Large parts of the European natural soils are exposed to high atmospheric N deposition.
- High N inputs threaten soil quality, which may negatively affect food and biomass production and biodiversity and enhance emissions of harmful N compounds from soils to water and the atmosphere.

Approaches

- An overview of the major soil functions and soil threats are presented, including a description of the objectives of the European Soil Strategy.
- The major N threats on soil quality for both agricultural and natural soils are related to changes in soil organic content and quality, soil acidification, and loss of soil diversity. These threats are described using literature.

Key findings/state of knowledge

- Generally, N has a positive effect on soil quality of agricultural soils, because it enhances soil fertility and conditions for crop growth. However, it generally has a negative effect on soil quality of natural soils, because it results in changes in plant diversity.
- Soil acts as a filter and buffer for N, and protects water and atmosphere against N pollution. However, the filter and buffer capacity of soils is frequently exceeded by excess of N in both agricultural and natural soils, which results in emission of N to the environment.
- Pyrite containing soils are found widespread in Europe. Nitrate removal from groundwater by pyrite oxidation increases concentrations of cations, heavy metals and sulphate. This causes problems when this water is used as drinking water.
- Combined application of N and C (e.g. in manures) has a positive effect on soil organic matter content in agricultural soils. There are indications that the use of only N fertilisers may result in a decline of soil organic matter under certain conditions.
- Application of fertilisers and manure and atmospheric deposition causes soil acidification. Soil acidification may lead to a decrease in crop and forest growth and leaching of components negatively affecting water quality, including heavy metals. Liming is widely used to reduce acidification of agricultural soils.
- Nitrogen affects populations of soil organisms, which may change N transformations in soil and emissions to the environment.
- The N inputs to agricultural and natural soils have decreased during the past ten years and the implementation of environmental policies may lead to a further decrease in the future. Consequently, the threat on the quality of European soils due to N will also decrease. Model simulations indicate that most of the European forest soils could recover from their acidified state within a few decades under the current N emission reduction plans.

Major uncertainties/challenges

- The effect of N on soil organic matter content and quality is uncertain. Some studies suggest that only use of N fertiliser may result in a decline of soil organic matter content.
- The effect of N on diversity of soil (micro) organisms and the effects of changes of soil biodiversity on soil fertility, crop production and N emissions towards the environment are not fully understood.

Recommendations

- Soil quality plays an important role in the production of food, feed and biomass, provides a habitat for biodiversity and controls emissions of pollutants to water and air. It is important to recognise the functions of soils in N cycling in both research and policy aiming at decreasing N emissions, and improving food production, and soil biodiversity.

The European Nitrogen Assessment, ed. Mark A. Sutton, Clare M. Howard, Jan Willem Erisman, Gilles Billen, Albert Bleeker, Peringe Grennfelt, Hans van Grinsven and Bruna Grizzetti. Published by Cambridge University Press.

21.1 Introduction

21.1.1 Soil quality and functions

Soil is the top layer of the earth and is composed of a mixture of mineral and organic compounds, water, air, and living organisms. The soil map (Figure 21.1) shows a large diversity in soil types in Europe. Soil types are classified on the basis of soil characteristics and properties (such as organic matter content, pH, clay content), and soil horizons (three-dimensional bodies containing one or more soil properties).

Soil quality can be defined as the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation (Karlen *et al.*, 1997; Schjønning *et al.*, 2004). Major functions of soils are (EC, 2006; Karlen *et al.*, 1997; Schjønning *et al.*, 2004):

- food and other biomass production;
- storage, filtering, buffering and transformations of natural and anthropogenic produced substances, including N;
- a biological habitat and gene reservoir;
- sink for C;
- source of raw materials;
- physical and cultural environment for human and human activities; and
- archival function for natural history.

21.1.2 EU soil policy

Recognising the extent of soil degradation and associated environmental and social risks in Europe, the European Commission proposed a Thematic Strategy for Soil Protection (EC, 2006). This strategy also contained a proposal for a Framework Directive. The overall objective of this European Soil Strategy is protection and sustainable use of soil, based on the following guiding principles:

- preventing further soil degradation and preserving its functions
- when soil is used and its functions are exploited, action has to be taken on soil use and management patterns, and

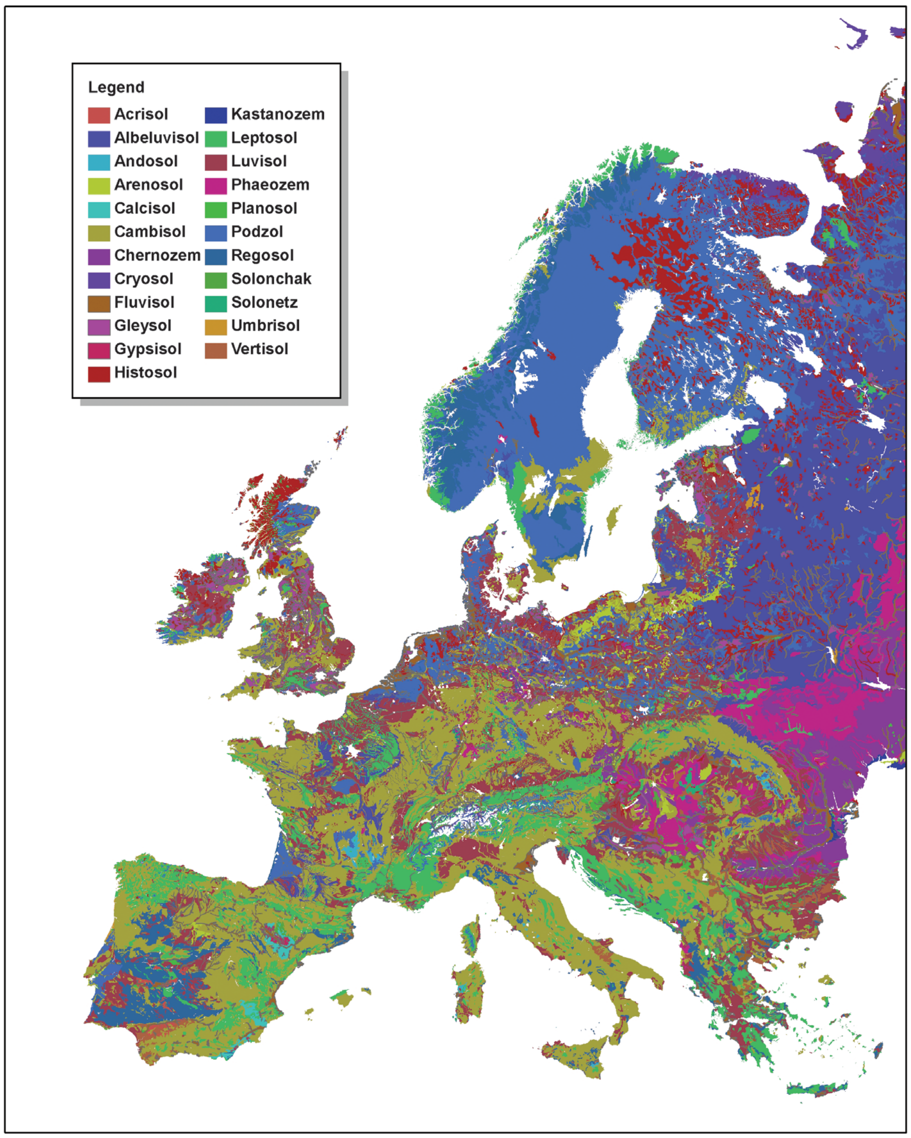

Figure 21.1 Soil Map of Europe (*Source*: European Soil Bureau).

- when soil acts as a sink/receptor of the effects of human activities or environmental phenomena, action has to be taken at the source.
- restoring degraded soils to a level of functionality consistent at least with current and intended use, thus also considering the cost implications of the restoration of soil.

In the Strategy, human activities, like inadequate agricultural and forestry practices, tourism, urban and industrial activities and construction works are indicated as the main factors that have an impact on soil functions. Main degradation threats to soil quality are erosion, salinisation, compaction, loss of organic matter, landslides, contamination, and sealing (Tóth *et al*., 2008). Soil degradation has a direct impact on the quality of water and air and threatens food and feed safety as well.

Soil protection objectives are also included in the EU Water Framework Directive, because soils act as a filter and a buffer for water bodies and are a source of point and diffuse pollution sources of aquatic ecosystems.

21.1.3 Effect of soil degradation on N emissions

The focus of this chapter is on the threats of N on soil quality. Inversely, however, soil quality also strongly affects N cycling and soil degradation processes may enhance N emissions to the environment. In this paragraph a short overview is presented about the effects of soil compaction, erosion, salinisation, contamination, and organic matter decline on N emissions to water and air.

Soil compaction occurs by mechanical stress on the soil surface by agricultural and construction machinery and by overgrazing. The threat of compaction of European soils increases because of the increasing use of heavy machinery (Van den Akker *et al*., 2003). Compaction leads to a decreased soil porosity and a reduced water infiltration capacity (Eckelmann *et al*., 2006). Low soil porosity enhances denitrification and thereby the risk of increased N_2O emission (Ruser *et al*., 1998). The low water infiltration capacity of compacted soils also increases the risk of surface runoff of N to surface waters.

Erosion leads to displacement of soil particles by water or wind (Eckelmann *et al*., 2006). It causes irreversible soil loss over tens or hundreds of years. The Mediterranean region is particularly prone to erosion, because of long dry periods followed by heavy rainfall, especially in regions with steep slopes. Erosion results in loss of fertile soil and pollution of surface water with N and other soil compounds. Landslides, the movement of a mass of soil induced by physical processes such as excess rainfall, also cause loss of fertile soil and may pollute surface waters with N.

Salinisation is the accumulation of soluble salts in soil to the extent that soil fertility is severely reduced (Eckelmann *et al*., 2006). Salinity poses problems in Hungary, and many Mediterranean countries and is expected to increase owing to larger extremes in droughts and rainfall periods as a possible consequence of climate change. High salt concentrations inhibit biological N transformations in soil (Curtin *et al*., 1999), as well as N fixing capacity by legumes (Delgado *et al*., 1993). High salt contents also decrease plant growth, which may lead to a lower N use efficiency of applied N and higher N emissions in the form of gaseous losses or leaching towards the environment.

Contamination of soils with organic (micro) compounds and heavy metals may hamper crop growth and decrease N use efficiency. Moreover, biological N transformations in soils are affected by contaminants (Bååth, 1989).

Around 45% of soils in Europe have a low or very low organic matter content (0%–2% organic C) and 45% have a medium content (2%–6% organic C; EC, 2006). Low organic matter contents are mainly found in Southern Europe (Figure 21.2). Organic matter decline is in particular an issue in Southern Europe, but also in parts of France, the United Kingdom, Germany, the Netherlands and Sweden. Smith *et al.* (2005) suggested that C stocks in European cropland decline because of changes in land use or agricultural management such as tillage practices or manure use. The inputs of N may also play a role in changes in soil organic matter contents (section 21.3.2). A decrease of soil organic matter contents negatively affects the biological, chemical, and physical soil fertility, N transformations in the soil (mineralisation, and denitrification) and biodiversity. This may result in reduced crop growth and a decrease in the buffering and filtering capacities of soils for N, leading to N losses to water and atmosphere.

These examples show that soil degradation hampers growth of crops and biological N transformations in soil, which may enhance N emissions to water and air. Proper soil and water management, as proposed in the European Soil Strategy, are needed to decrease N losses induced by soil degradation.

21.2 Fates of excess nitrogen inputs to soils

Nitrogen that enters the soils is generally biologically or chemically transformed, e.g. via mineralisation, immobilisation in soil organic matter, nitrification, and denitrification (Butterbach-Bahl *et al*., 2011, Chapter 6 this volume). Soils protect the quality of air and water by storage, filtering, buffering and transformations of N. This soil function is threatened when the N inputs to the soils exceeds the N output by removal of the crop, tree, or vegetation. An excess input of N to the soil changes the rate of the different N transformation processes, affects soil organic matter content, decreases soil biodiversity and decreases the filtering and buffering capacity of the soil. Moreover, transformations of N often result in soil acidification, that may be reflected in a lower soil pH. Soil acidification may decrease crop growth, change N transformations in the soil, and decrease soil biodiversity.

The N surplus of the soil balance of agricultural soils is an indicator for N emissions to the environment (atmosphere and hydrosphere). The soil surface balance includes all relevant N inputs and outputs from the soil. The fate of the N surplus is controlled by a combination of factors, including type and rate of N input, soil type and properties, weather conditions, crop type, tree species, and the hydrology of the soils. Differences in fertiliser and manure use (rate and application method), soil and crop type and weather conditions cause differences in the fate of the N surplus between EU countries (Figure 21.3). The N surplus on the soil surface

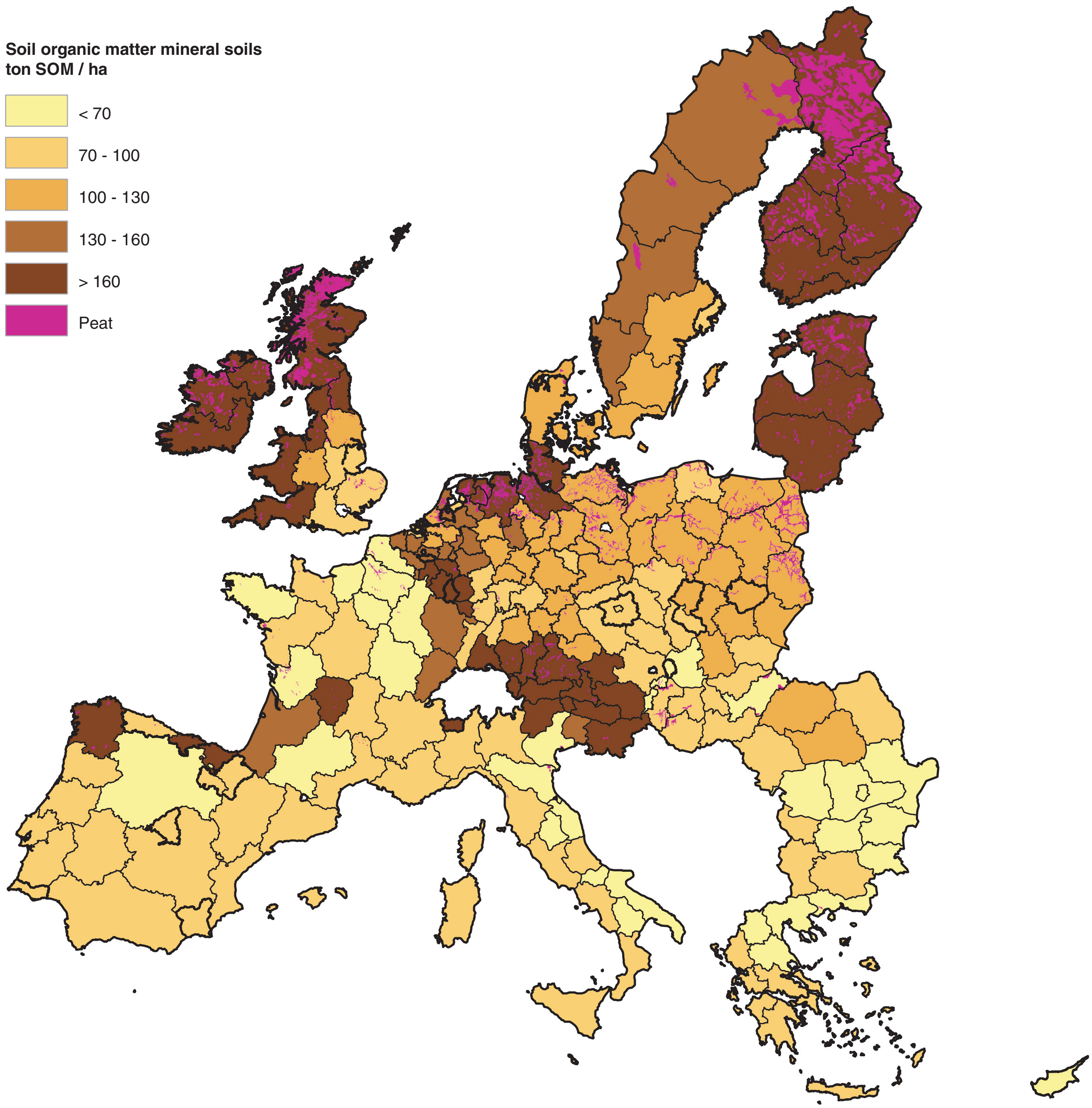

Figure 21.2 Organic matter stocks in mineral soils in Europe. The location of peat soil is indicated (Lesschen *et al.*, 2009).

balance ranges from less than 50 kg N/ha per year to more than 200 kg N/ha per year. Most of the N surplus on the soil surface balance (on average about 50%–60% of the N surplus) is lost as the harmless gas N_2, followed by NH_3 emission, NO_3 leaching, and emissions of N_2O, and NO_x. Part of N surplus may (temporally) accumulate in organic matter. More details on the fate of the N surplus are given in (De Vries *et al.*, 2011, Chapter 15 this volume).

In non-agricultural soils, the N input is limited to N deposition and N fixation. Nitrogen deposition to natural ecosystems in Europe is generally higher than 5 kg N/ha per year and often exceeds 40–50 kg N/ha per ha per year in areas near intensively used agricultural production regions. Leaching of N generally increases at N deposition rates higher than about 10 kg N/ha per year (Figure 21.4; Dise *et al.*, 1998; Gundersen *et al.*, 1998; Butterbach-Bahl *et al.*, 2011, Chapter 6 this volume). Elevated N deposition to forests also increases nitrification and denitrification and thereby emissions of gaseous N (N_2O, NO_x and N_2). A large number of controlling variables and complex interactions influence the net N_2O

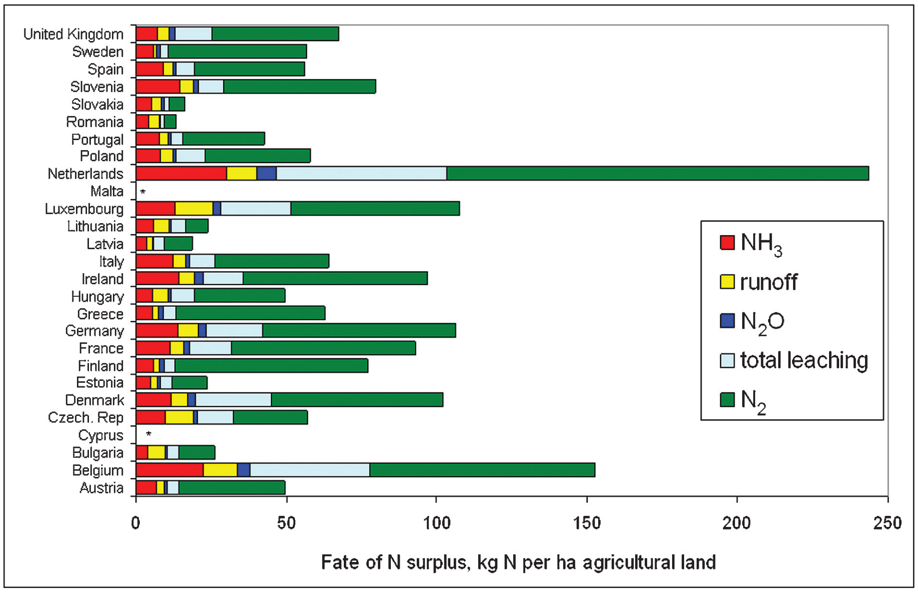

Figure 21.3 Fate of the N surplus of the soil balance in EU-27 in 2000 in kg N per ha agricultural land calculated using the model MITERRA-EUROPE (after Velthof *et al.*, 2009).

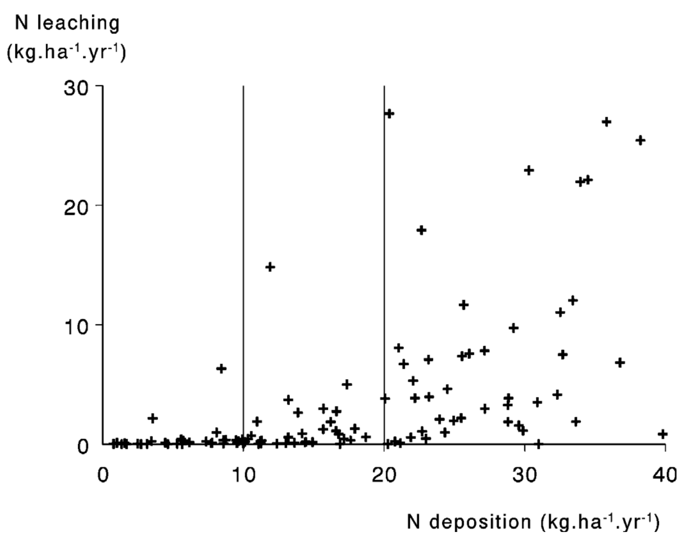

Figure 21.4 Leaching of total N against atmospheric deposition rate measured at 121 forest plots in Europe (after De Vries *et al.*, 2007b).

and NO_x emissions, such as N deposition rate, precipitation, temperature, pH, clay content and tree species composition are important variables (Kesik *et al.*, 2005; Bloemerts and de Vries, 2009; Butterbach-Bahl *et al.*, 2011, Chapter 6 this volume).

Denitrification is the main mechanism of removal of nitrate in deep groundwater and subsoil. The organic C content of the subsoil is low, which limits denitrification. However, the subsoil may also contain reduced inorganic compounds, including pyrite (FeS_2), siderite ($FeCO_3$), and other ferrous containing compounds. These compounds can be used by denitrifying bacteria as energy source. Pyrites often contain trace elements, including the toxic elements nickel, arsenic, cobalt, copper, lead, manganese and zinc (Huerta-Diaz and Morse, 1992; Larsen and Postma, 1997). These trace elements are released when pyrite is oxidised during denitrification. Moreover, pyrite oxidation results in an increase of the sulphate concentration and a pH decrease of groundwater (Appelo and Postma, 1999). Thus, nitrate removal from groundwater by pyrite oxidation increases concentrations of cations, heavy metals, and sulphate. These high concentrations may cause problems when this water will be used as drinking water. Nitrate leached from agricultural and natural soils may thus negatively affect soil quality in pyrite containing soils. Pyrite containing soils are found widespread in Europe, and have been reported in Denmark (Jørgensen *et al.*, 2009), England (Moncaster *et al.*, 2000), France (Molénat *et al.*, 2002), Germany (Eulenstein *et al.*, 2008), the Netherlands (Hartog *et al.*, 2005), and Spain (Otero *et al.*, 2009).

21.3 Threats of nitrogen on soil quality

N has several effects on the soil quality of natural ecosystems (Table 21.1). Impacts on the inorganic N concentration are presented in the previous paragraph. In this paragraph the effects of N on soil acidification (Section 21.3.1), soil organic matter (Section 21.3.2), and soil biodiversity (Section 21.3.3) are presented.

21.3.1 Effects on soil acidification

Transformations of N are an important source and sink of hydrogen ions or protons (Tabel 21.1). Acidification considerably reduces the fertility of the soil, affects microbial transformations in the soil, and may cause depression of crop growth, and yields (Marschner, 1995; Bolan *et al.*, 2003). It may lead to (i) less availability (or deficiencies) of nutrients such as phosphorus, calcium, magnesium and molybdenum, (ii) a release of toxic compounds, including aluminium, and manganese, and (iii) hampering of the activity of soil micro-organisms involved in N transformations, such as mineralisation of organic N and biological N fixation. Low soil pH promotes the production of N_2O during nitrification and denitrification (Granli and Bøckman, 1994). Soil acidification results in leaching of cations. In the Netherlands, the observed increase of hardness (i.e.

Table 21.1 Effects of N on soil parameters of natural soils, their mechanisms, and the ecosystem response

Soil parameter	**Mechanism**	**Ecosystem response**	**Literature**
C/N ratio	Narrows at sites with high N availability, due to the incorporation of surplus N in soil organic matter.	Plant species richness ↕ Decomposition of SOM ↕ Microbial biomass ↑	(Von Oheim *et al*., 2008) (Friedel *et al*., 2008) (Dumortier *et al*., 2002) (Berg, 2000)
Inorganic nitrogen concentration	Nitrogen deposition is close to or exceeds ecosystem N demand. Input of inorganic N increases soil solution concentrations.	Plant productivity ↑ Leaf/needle N content ↑ Litter decomposability ↑ Plant species richness ↓ Vascular plants in wetlands ↑ Microbial N immobilisation ↓ Nitrogen leaching ↑ Soil N_2O/NO emissions ↑	(De Vries *et al*., 2006b) (Corré *et al*., 2007) (Kreutzer *et al*., 2009) (Gundersen *et al*., 2006) (Stevens *et al*., 2006)
Acidification and soil buffering capacity	Nitrification of deposited NH_3/NH_4^+ leads to H^+ formation. In the course of the acidification process base cations are leached.	Nutrient availab. (Ca/Mg) ↓ Al/Mn toxcity if soil pH<5.5 ↑ Biodiversity ↓ Microbial activity ↓ Root growth ↓ Nitrogen leaching ↑ DOC leaching ↓ Soil N_2O/NO emissions ↑ Wetland CH_4-emissions	(Matzner and Murach, 1995) (Raubuch and Beese, 2005) (Bowman *et al*., 2008) (Gauci *et al*., 2005) (Evans *et al*., 2008)
Soil C stocks and SOC stratification	Surplus N decreases fine root biomass and, thus, reduces belowground litter production, but increases aboveground plant production and litter fall.	Total soil C stocks ↑ Forest floor C stocks ↑ Mineral soil C stocks ↕	(Högberg, 2007) (De Vries *et al*., 2006b) (Hyvönen *et al*., 2007, 2008)
Soil aggregation	N can increase litterfall and improve litter quality and, thus, positively affect soil fauna and the formation of organo-mineral soil aggregates by e.g. earthworm activities	Soil aeration ↑ Water infiltration ↑	(Lavelle *et al*., 2007)

the contents of Ca and Mg) in groundwater used for drinking water has been associated with the acidification of agricultural soils (Velthof *et al*., 1999). Hardness is considered an aesthetic water quality factor and is not known to pose a health risk to users.

Agricultural soils

Ammonium based fertilisers acidify soils, because of a combination of nitrification, ammonium uptake by plants, and/or ammonia volatilisation (Table 21.2). Nitrate based fertilisers increase pH of soils, because of a combination of nitrate uptake by plants, and/or denitrification. The most used mineral N fertilisers in Europe (i.e. >90% of total N fertiliser consumption) are calcium ammonium nitrate, ammonium nitrate, NPK fertilisers, urea, and urean (Fertilizers Europe, 2010). All these fertilisers have an acidifying effect (Harmsen *et al*., 1990), indicating that the use of mineral N fertiliser causes acidification in a large part of the European agricultural soils. Besides the type of N fertiliser, soil acidification is dependent on crop type, soil type, weather conditions (leaching), and other N sources (Bolan *et al*., 2003). Other major inputs of N are manure, N excreted during grazing, biological N fixation, and atmospheric deposition. Manures contain high ammonium concentrations, which has an acidifying effect either by uptake or nitrification. Organic N results in alkalinisation if it is mineralised (consumption of H^+), but this effect is more than compensated by nitrification (production of $2H^+$) and the combined process leads to acidification unless NO_3 is consumed (consumption of H^+) (Table 21.2). The total effect of manure on acidification depends on rates of inputs and uptake of cations and anions, specifically of NH_4 and NO_3, by the crop and the rate of denitrification affecting nitrate leaching. Grazing causes a heterogeneous pattern, with acidification in urine patches (nitrification) and alkalinisation in dung pats (mineralisation).

Table 21.2 Generation (acidification) and consumption (alkalinisation) of protons (H^+) in N transformation processes

Process	Reaction[a]	H^+, mol/mol N
Biological N-fixation	$4ROH + 2N_2 + 3CH_2O \rightarrow 4RNH_2 + 3CO_2 + H_2O$	0
Mineralisation of organic N	$RNH_2 + H_2O + H^+ \rightarrow ROH + NH_4^+$	−1
Urea hydrolysis	$(NH_2)_2CO + 3H_2O \rightarrow 2NH_4 + + 2OH^- + CO_2$	−1
Nitrification	$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O$	+2
Ammonium assimilation	$ROH + NH_4^+ \rightarrow RNH_2 + H_2O + H^+$	+1
Nitrate assimilation	$ROH + NO_3^- + H^+ + 2CH_2O \rightarrow RNH_2 + 2CO_2 + 2H_2O$	−1
Ammonia volatilisation	$NH_4^+ \rightarrow NH_3 + H^+$	+1
Denitrification	$5CH_2O + 4NO_3^- + 4H^+ \rightarrow 2N_2 + 5CO_2 + 7H_2O$	−1

[a] R = in the reaction mean organic C compounds. (Based on Van Breemen *et al*., 1983; De Vries and Breeuwsma, 1986; Bolan *et al*., 2003).

Grazing in intensively managed grassland with high N inputs results in soil acidification (Oenema, 1990). Biological N fixation, in which N_2 is fixed in organic N, does not affect soil acidification (Table 21.2). However, actively N_2-fixing leguminous crops acidify the soil, because of an excess uptake of cations over anions (Haynes, 1983). Atmospheric N deposition causes acidification, because most of the N is present as NH_4.

The content of heavy metals cadmium, zinc, and copper are high in some agricultural soil, because of long-term inputs of heavy metals via fertilisers, manures, lime and organic products. The mobility of these heavy metals generally increases when soil pH decreases (Adriano, 2001). Liming of agricultural soils reduces the mobility of heavy metals and decreases risk of crop uptake and leaching to ground and surface waters (Bolan *et al*., 2003). However, if agricultural soils with elevated heavy metal contents are abandoned and not limed, there is a risk that these soils acidify and heavy metals are released to the environment (Boekhold, 1992).

Soils of natural ecosystems and forests

Influences of elevated atmospheric N (and S) deposition on natural terrestrial ecosystems have received particular attention since the 1980s, specifically with respect to forests. The acidifying effects of N and S include (Table 21.2) (i) loss of base cations from the soil causing deficiency of these nutrients for forest trees (notably Mg), (ii) release of soluble toxic Al affecting fine root growth and inhibiting the uptake of base cations (Cronan *et al*., 1989; Marschner, 1995) and (iii) a decrease in pH that may affect mineralisation processes and hence nutrient availability (De Vries *et al*., 1995, 2000; Erisman and de Vries, 2000).

In acid soils, atmospheric deposition of S and N compounds leads to elevated Al concentrations in the soil solution. Figure 21.5 shows that more than 80% of the variation in Al concentration in subsoils of European forested plots with pH below 4.5 could be explained by a variation in SO_4 and NO_3 concentrations. In soils with a pH above 5.0, the release of Al is generally negligible, since base cations release by weathering and cation exchange buffers the incoming net acidity in those soils.

21.3.2 Effects on soil organic matter

Soil organic matter is important for soil fertility, and affects both crop production (food and other biomass) and N and C transformations. Soil organic matter is also a sink for C, and important from a perspective of greenhouse gas emissions.

Soil organic matter content and composition (quality) affects many physical, chemical, and biological properties of soils, including soil structure, water holding capacity, aeration, compaction, risk of erosion, biodiversity, (micro)biological N transformations, and the cation exchange capacity (CEC). The C/N ratio and the degradability of the soil organic matter are major factors affecting the quality of organic matter. Soils with low C/N ratio and high degradability have a high N mineralization capacity and are often considered as fertile from an agricultural point of view. However, a high N mineralization capacity is considered negative for natural soils as a high N release may lead to a decrease in plant biodiversity.

One of the major controllers of soil organic matter content is land use and land management (Guo and Gifford, 2002) rather than N input. Land use and N availability are, however, closely linked and need to be jointly considered when discussing N effects on the quality of soils. In pre-industrial times, it was common practice across Europe that natural ecosystems were exploited for nutrients in order to maintain the soil fertility of the arable land, e.g. by extracting litter from forests, forest grazing, wood harvest or by converting grassland/heathland swards to arable land (Glatzel, 1991; Sieferle, 2001). These management practices have resulted in a large-scale depletion of natural soils in nutrients and soil organic matter and degradation of soil properties (Glatzel, 1991). The situation changed with the introduction of N-fixing legumes at the end of the nineteenth century, and with the increasing availability of synthetic fertilisers at the beginning to the mid of the twentieth century (Chorley, 1981). Owing to the large scale increased accessibility and availability of N for agricultural production, the pressure on natural ecosystems to serve as a reserve for nutrients ceased. However, the impacts of former land management on nutrient availability in soils may be traced for almost 2000 years (Dupouey *et al*., 2002). The demand for increased

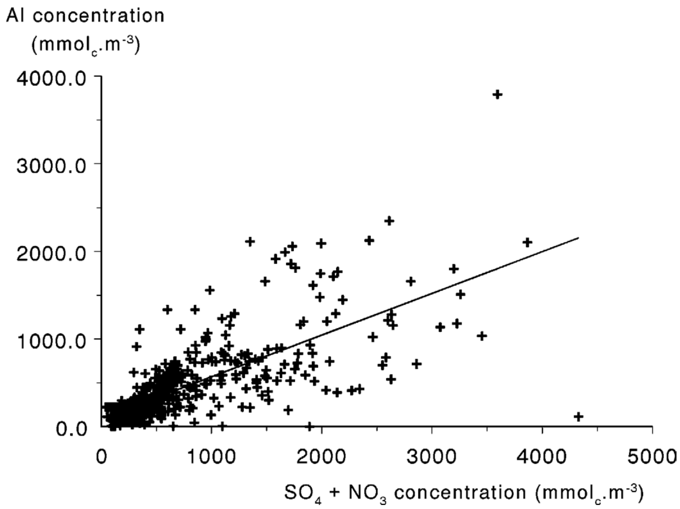

Figure 21.5 Concentration of total Al against total SO_4+NO_3 in the subsoil of monitoring plots with a pH < 4.5 (after De Vries *et al.*, 2007b). The solid line is a regression line being equal to: $y = -95 + 0.74x$ ($R^2 = 0.86$).

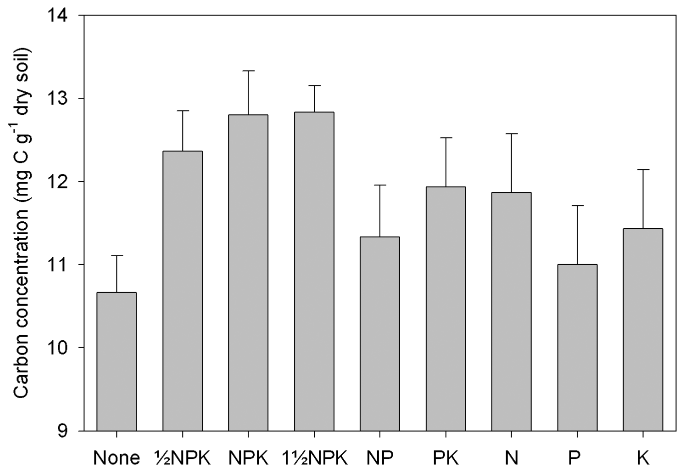

Figure 21.6 Carbon concentration in the topsoil (0–20 cm) for different mineral fertiliser treatments in the Askov long-term experiment in 2004 after 110 years of treatment differences on a sandy loam (Christensen *et al.*, 2006). A crop rotation with winter cereals, root crops, spring cereals and grass-clover has been subjected to a range of different fertiliser treatment over the period 1894 to 2004. Various levels (0, ½, 1, 1½, 2) of N, P and K applied as mineral fertiliser were tested in the experiment. Note that the positive effect of K on soil C content was presumably because K is an important nutrient for the grass-clover.

agricultural production was also the major driver to drain many huge wetland areas in central and northern Europe. The change to oxidative conditions has promoted mineralisation of C and N, previously stored for thousands of years.

Agricultural soils

Application of N generally increases crop production and thereby also the returns of organic matter as crop residue to the soil (Paustian *et al.*, 1997). A review by Glendining and Powlson (1995) showed that long-term use of only inorganic N fertilisers increased soil organic C content in the majority of long-term studies, but the increases tended to be small (see Figure 21.6). Results of long-term studies in Russia of Shevtsova *et al.* (2003) showed that high application rates of inorganic N fertilisers lead to a decrease in soil organic C content compared with unfertilised soils. This decrease was related to an increase in C mineralisation due to changes in soil quality by the long term use of the acidifying fertilisers. Results of an experiment of Olesen *et al.* (2000) showed that application of only mineral N fertiliser caused higher shoot biomass and lower root biomass in cereals, compared to application of animal manure. This was probably because of the higher availability of N in the soil with mineral N fertiliser, which resulted in less development of roots in comparison to the lower N availability in the manure treatment. This effect may be an explanation for the observed small effects of mineral N fertiliser on soil organic matter content in comparison to manure N. Khan *et al.* (2007) and Mulvaney *et al.* (2009) analysed results of a long-term experiment in the USA, and concluded that 40 to 50 years of inorganic N fertilisation caused a net decline in soil C content, despite massive residue C incorporation. They indicated that mineral fertiliser N promotes decomposition of crop residues and soil organic matter. However, Powlson *et al.* (2010) disagreed with this conclusion and suggested that the results of the long-term experiment were misinterpreted.

The combination of mineral fertiliser and animal manure is most effective in maintaining soil organic matter contents at a stable level (Glendining and Powlson, 1995). A study of Sleutel *et al.* (2003) in Belgium showed a decrease in soil organic C contents in agricultural soils during the period 1989–1999. This was attributed to a decrease in manure application, because of environmental legislation. Moreover, farmyard manure production in Belgium has been replaced by slurry based systems. Regular application of farmyard manure has a beneficial effect on soil organic C (see Hofman and van Ruymbeke, 1980).

The studies indicate that the effects of N fertilisation on soil organic C contents are diverse and related to the initial soil organic C content, soil type, climatological conditions, soil management, development of shoot and root biomass, N source and land use. Factors, such as soil tillage, changes in climatological conditions, and changes in land use probably have a larger effect on the soil organic matter content than the use of N.

Soils of natural ecosystems and forests

Natural soils have low N contents. Increased N deposition rates have significantly changed C and N cycling and soil C/N ratios during the last decades (Vitousek *et al.*, 1997; Gundersen *et al.*, 2006; Corré *et al.*, 2007). A high input of N leads to eutrophication or N saturation of the natural systems, mainly indicated by low C/N ratios of organic matter, elevated N leaching and sometimes elevated NH_4/base cation ratios (Table 21.1). Eutrophication also leads to reduced plant species diversity (Dise *et al.*, 2011, Chapter 20 this volume) and may cause damage to forests due to: (i) water shortage, since a high N input favours growth of canopy biomass (De Visser *et al.*, 1994), (ii) nutrient imbalances, since the increase in canopy biomass also causes an increased demand for base cation nutrients (Ca, Mg, K) (Boxman and Roelofs, 1988), and (iii) an increased sensitivity to factors such as frost (Bruck, 1985) and attacks by fungi. Several studies showed that soil respiration may decrease under conditions of increased N availability (Olsson *et al.*, 2005), which possibly can be attributed to the formation of recalcitrant organic material with narrow C/N ratios (Table 21.1).

At an ecosystem scale, a large positive effect of N input to the C sequestration capacity of boreal and temperate forests in living biomass has been demonstrated (Högberg, 2007; Magnani *et al.*, 2007; De Vries *et al.*, 2008). On the basis of forest inventory data and measured inputs of N, De Vries *et al.* (2006b) estimated that at an EU scale the average impact of an additional N input on the net C sequestration in both tree wood and soil was approximately 50 kg C/kg N deposited. A literature review of (i) empirical relations between spatial patterns of C uptake and influencing environmental factors including nitrogen deposition, (ii) ^{15}N field experiments, (iii) long-term low dose N fertiliser experiments and (iv) results from ecosystem models indicate a total C sequestration range of 5–75 kg C/kg N deposition for forest and heathlands, with a most common range of 20–40 kg C/kg N (De Vries *et al.*, 2009). It should be noted that the N addition effect on C sequestration will saturate with time or even decline due to detrimental effects of surplus N availability in forest soils on forest health.

Effects of N deposition to peatlands systems may differ from those to upland systems such as forests and heathlands. Moderate to low N deposition rates have been shown to promote the growth of sphagnum, even though nutrient imbalances, especially P deficiencies may set close limits for accelerated growth (Berendse *et al.*, 2001; Phuyal *et al.*, 2008). Furthermore, at higher N deposition plant community changes may occur, with vascular plants outcompeting mosses. Consequences of N deposition for peatland C storage are variable. Increased net ecosystem productivity following N deposition and stimulated growth of sphagnum and vascular plants may be offset by increased decomposition due to improved litter quality (Gunnarsson *et al.*, 2008; Trinder *et al.*, 2009).

21.3.3 Effects on soil biodiversity

Nitrogen affects the biodiversity of soil organisms. Belowground especially fungi, saprotrophic decomposers as well as mycorrhizal fungi, and N fixing bacteria are reduced by fertilisation and high N availability (Streeter, 1988; Johansson *et al.*, 2004; De Vries *et al.*, 2006a).

Bacteria and fungi are the primary decomposers of dead organic matter such as plant residues and manure, and they release mineral nutrients by mineralisation. Soil microorganisms are consumed by microbivores such as protozoa, nematodes and mites. Microbivores, in turn, are eaten by bigger predatory soil fauna. All these links in the soil food web contribute to mineralisation and nutrient cycling. Changes in bacteria and fungi will also affect the soil fauna via the bacterial and fungal channels in the soil food web.

While plants and soil microorganisms directly react to the availability of mineral N, soil fauna, such as protozoa, nematodes, enchytraeids, collembolas, insect larva or earthworms, mostly react indirectly to N through effects on plant growth and microbial dynamics (Bardgett, 2005). Since both plant litter and microorganisms are at the base of soil detritivore food webs, this is likely to lead to bottom-up effects on the whole belowground food web, on plants and on the aboveground food web (Wardle *et al.*, 2004).

While, trophic effects are likely to be very influential (De Ruiter *et al.*, 1994), non-trophic activities of soil fauna are also involved in soil response to inputs of mineral N (Lavelle and Spain, 2001). For example, the biomass of earthworms is likely to increase if N increases plant biomass production and plant litter production. This would finally change soil aggregation, water infiltration, and organic matter dynamics (Lavelle *et al.*, 2007). Because of the complexity of the interactions involved in soil fauna response to increasing N inputs, it is more difficult to make general predictions on this response than on the response of soil microorganisms or plants.

While it is difficult to predict the effect of N inputs on particular taxa of soil fauna, N inputs could have a clearer effect on the biodiversity of soil fauna. Since N decreases plant biodiversity, it may be suggested that this leads to a lower soil biodiversity if soil taxa are eating preferentially the organic matter coming from particular plant species. However, belowground and aboveground biodiversities do not seem to be linked in such a straightforward way (Hooper *et al.*, 2000). Some studies have shown that soil fauna is more sensitive to the quantity than to the composition (quality) of organic matter. Thus, while N inputs often decreases plant diversity, it also tends to change the quality of organic matter, which may increase soil fauna diversity (Cole *et al.*, 2005; Van der Wal *et al.*, 2009).

Agricultural soils

Most of the information about effects of fertilisers on soil biodiversity comes from studies where organic farming systems were compared with conventional intensive farming systems. One has to be aware that in organic farming systems, there are usually more differences with conventional farming systems than fertilisation alone, such as the avoidance of pesticides.

In a comparison of 23 farms in Estonia, animal manure increased soil microbial biomass, activity and N mineralisation, and chemical fertilisers resulted in negative effects compared to organic fertilisers (Truu *et al.*, 2008). In long-term experiments (25 years), Ge *et al.* (2008) found a negative effect of mineral N (300 kg N per ha per yr) on microbial diversity (genetic diversity, number of genotypes or species of bacteria). Also Jangid *et al.* (2008) found higher bacterial diversity with organic manure (poultry litter) and lower diversity with mineral fertilisers.

In grassland soils, fungal biomass increased with reduced N fertilisation (De Vries *et al.*, 2007a; Van Groenigen *et al.*, 2007). Negative effects of inorganic N on fungi were mostly attributed to changes in vegetation and organic matter characteristics (Bardgett *et al.*, 1999; Rousk and Bååth, 2007). When fungi are affected also bacteria can be affected because of competitive interactions between these two groups.

In agricultural soils, it is difficult to disentangle the effects on soil fauna of the various agricultural practices. Indeed herbicides, pesticides, tillage, organic fertilisation and mineral fertilisation impact soil fauna (Jordan *et al*., 2004). Large species such as earthworms are often affected by tillage that directly increases their mortality (Edwards and Lofty, 1982; Chan, 2001). The depletion in organic matter of many agricultural soils (Lal, 2004) is also a very influential factor for the whole soil detritivore food web (Lavelle and Spain, 2001), which may in turn hide effects of N. However, some studies have detected effects of mineral N fertilisation on some groups of soil fauna in crop or pasture soils. For examples, nematodes (Okada and Harada, 2007) or protozoa (Forge *et al*., 2005) have been shown to be impacted by mineral fertilisation, probably through a trophic impact on microorganisms and plant growth. Earthworms have also been shown to have higher densities in some N fertilised plots (Jordan *et al*., 2004).

Soils of natural ecosystems and forests

Soils of natural ecosystems and forests usually are more dominated by fungi and fungivorous soil fauna. Probably there are no fundamental differences between the effect of N input on microbial processes in agricultural soils and other ecosystems. Usually natural soils are not fertilised, are affected by atmospheric deposition of N, and are more acid. Also in natural soils long-term high N inputs have been shown to cause changes in the structure of the microbial community (Nemergut *et al*., 2008). Some of the microbial community shifts could provide explanation for changes in soil organic matter structure. Steep declines in basidiomycete fungi were related to higher lignin concentrations in N-amended alpine tundra plots. Multiple factors can alter soil organic matter pools following increases in N availability, including shifts in both plant productivity and species composition. The effects of increased N inputs on decomposition of organic matter, C storage and CO_2 emissions are not clear yet (Craine *et al*., 2007).

In natural ecosystems many trophic effects of N enrichment on soil fauna have been documented. For example, in a forest soil, a reduction in the diversity of nematodes and an increase in their total abundance were attributed to an increase in the relative abundance of bacterivorous and fungivorous nematodes due to the positive impact of fertilisation on microbial biomasses (Forge and Simard, 2001). Non-trophic effects of N enrichment on soil fauna have also been documented. N inputs can lead to soil acidification which negatively impacts many taxons. Xu *et al*. (2009) found that N deposition both decreases soil pH, the diversity of collembolas and the density of the most abundant collembolan species in a Swiss forest.

21.4 Management and future perspectives

21.4.1 N inputs to soils

The surplus of N in agricultural soils can be decreased by decreasing the total N input and/or increasing the outputs of N in harvested products. Nitrogen surpluses have been declining since the eighties in member states of the European Union (Figure 21.7). In Central Europe, the economic situation in the early nineties has caused a drop in the use of N fertiliser. In the intensively managed agricultural systems in the member states in North and West Europe, the decrease in the N surplus is mainly due to reform of the agricultural policy and environmental legislation, which has enhanced a more efficient N use. The decrease in the surplus indicates that N emission to the environment has decreased. However, many regions in the EU-27 still have a significant surplus on the soil N balance (see also Figure 21.3). Measures to decrease N inputs include balanced N fertilisation, in which the N inputs to the crop are tuned to the crop demand, low protein animal feeding, and/or decrease of the number of livestock. An increase in the N output can be obtained by proper management of soils, water and nutrients and control of pests. The Nitrates Directive, which is implemented in EU-27 requires measures in nitrate vulnerable zones increases, such as balanced N fertilisation is imposed. The Gothenburg Protocol of UNECE's Convention on Long-range Transboundary Air Pollution is under revision, and may also include balanced N fertilisation as one of the measures. It is expected that because of implementation of environmental policies, the N inputs to agricultural soils with a high N surplus will further decrease in the near future.

Following a series of control measures during the last two decades, emissions of NH_3 and NO_X and subsequent N deposition have been reduced in Europe. The concept of critical N load has been developed for natural systems to set targets for reduction of N emission (Hettelingh *et al*., 2001). The area where critical N loads are exceeded clearly decreased between 1980 and 2005. Nevertheless, high exceedances for critical N loads remain widespread especially in north-western European areas. Further reductions in N emissions to the atmosphere are predicted (Moldanová *et al*., 2011, Chapter 18 this volume).

21.4.2 Soil acidification

The expected decrease in N inputs to soils in the future will also decrease soil acidification. Model simulations of Reinds *et al*. (2009) showed that most of the European forest soils could recover from their acidified state within a few decades under the current emission reduction plans.

Liming is used to decrease soil acidification and optimise plant growth in carbonate free soils. The major sources of lime used in agriculture are lime stone ($CaCO_3$) and dolomite ($CaMgCO_3$), but also other compounds may reduce soil acidification (burned lime, rock phosphate, sugar beet pulp). Liming is widely used in agriculture and there are no indications that acidification of agricultural soils hampers crop production in Europe. The amount of limestone used in agriculture in several western and northern European countries has decreased strongly (Figure 21.8). This coincides with the decrease in N fertiliser consumption since 1990. These results suggest that acidification of agricultural soils in Europe is decreasing since the early nineties.

Dissolution of lime in soils leads to dissolution of carbonates and release of CO_2. Thus, liming results in emission of

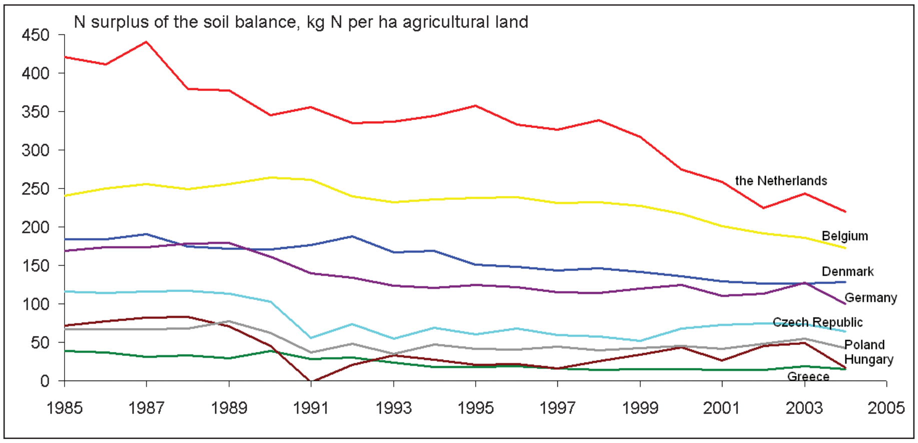

Figure 21.7 Nitrogen surplus of the soil balance of agricultural soils for selected countries (source data: OECD, 2010).

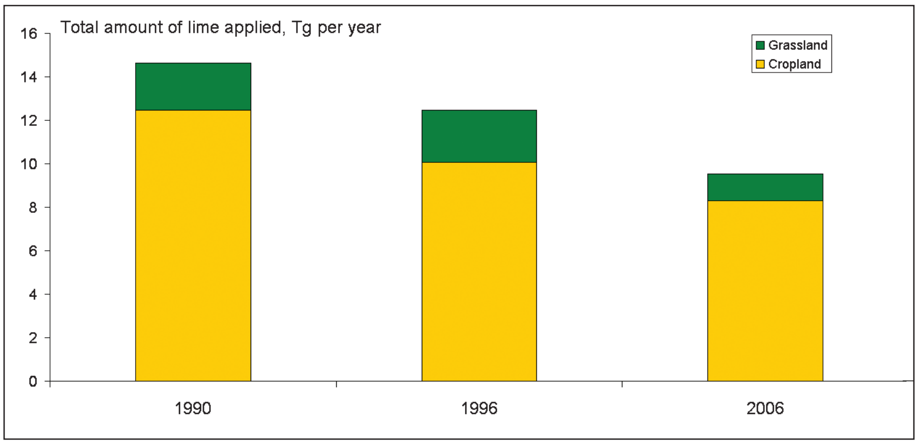

Figure 21.8 Total amount of lime (dolomite and limestone) used for grassland and cropland in 1990, 1996, and 2006 in Austria, Germany, Denmark, Finland, France, United Kingdom, Ireland, the Netherlands and Sweden. These European countries reported their lime use to UNFCCC. Data derived from the 2008 report of Greenhouse Gas emissions to UNFCCC (UNFCCC, 2010).

CO_2. The IPCC has recognised this source of CO_2 in the latest update of methodology of calculating greenhouse gas emissions (IPCC, 2006). Countries are obliged to report the CO_2 emission from limestone and dolomite use in agriculture in the annual inventories to UNFCCC (category 5G). The use of lime in agriculture is only a minor source of CO_2 and, for example, much smaller than the N_2O emission from agriculture.

Liming of forest soils has been widely discussed as a method of neutralising the effect of acidification. Beier and Rasmussen (1994) conclude that it is possible to reverse the acidification processes in the soil by liming, and that it is possible to increase growth and improve the nutritional balance in the trees by fertilisation and irrigation. However, the complexity of the ecosystem and the factors controlling vitality and sustainability of the ecosystem are still not fully understood. In Sweden, Andersson and Persson (1988) recommend a liming dose of 2–5 ton/ha when improvement of soil and root environment is required. Higher doses may be needed to avoid leaching of aluminium from catchments. However, liming may have negative effects on the development of tree fine roots, particularly in areas with a high N deposition (Persson and Ahlström, 1990). Results from liming experiments in Dutch forests for the period 2000–2005 (Wolf *et al.*, 2006) show that liming leads to an increase in plant species, especially nitrophilic species, which is considered as a non-desirable side effect. Furthermore, liming increases the decomposition of organic matter, leading to thin humus layers and a decrease in soil biota species. Wolf *et al.* (2006) thus considered permanent forest liming as an undesirable management option, but it can be beneficial as a once-only event in nutrient rich deciduous forests.

21.4.3 Soil organic matter

In agricultural soils, increasing or maintaining soil organic matter content can be obtained by strategies in which the input of organic matter to the soil is higher than the output by harvested crop and by decomposition in the soil. Sources of organic matter are crop residues, manures, and organic products like compost. There are large differences in decomposition of residues of arable crops, with highest decomposition in residues of vegetables and low decomposition in cereal straw (Velthof *et al.*, 2002). Changing crop types and including winter crops in the rotation are options to increase input of organic matter to the soil. Grasslands have a high biomass production and roots and stubbles are a large source of organic matter. Including grassland in a crop rotation will enhance soil organic matter contents in comparison to continuous cropland (Van Eekeren *et al.*, 2008). The organic matter in farmyard manure is less degradable than that in animal slurries and the use of farmyard manure instead of slurry has a beneficial effect on soil organic matter content of soils (Leinweber and Reuter, 1992). No tillage or reduced tillage decreases decomposition of soil organic matter. All these strategies to increase or maintain soil organic matter content of agricultural soils have also been suggested as strategies to increase C sequestration in soil (Smith *et al*., 2008).

Thinning and removal of the top soil layer are options to avoid eutrophication of natural soils. For coniferous forests the combination of sod cutting and felling and removal of trees, can lead to an improvement of soil quality. Thinning reduces litter fall and thereby the N input to soils. In heathlands, sod cutting is an efficient measure to halt invasion of grasses and increase plant species diversity (Diemont and Oude Voshaar, 1994). The complete removal of the organic top layer, including the vegetation, ensures the removal of accumulated N. A less rigorous measure is choppering. Although considerably more nutrients were removed by sod-cutting than by choppering, nutrient output by choppering was still sufficient to compensate for about 60 years of net N input (Niemeyer *et al*., 2007). These types of measures may lead to changes in soil biodiversity.

21.4.4 Soil biodiversity

The main option to reduce or prevent unwanted effects of N on soil biodiversity is the reduction of N inputs. Further, there is broad agreement on general principles to promote and maintain soil biodiversity, natural soil fertility and ecosystem services, i.e. the input of organic matter in the soil should be sufficient to meet the C and energy requirements of the soil biota, and the nutrient requirements of the crop (Swift *et al*., 2004; Barrios, 2007; Brussaard *et al*., 2007; Kibblewhite *et al*., 2008). It also helps when the soil remains covered by crops, which continuously feed the soil organisms via root exudates and residues. Intensive soil tillage and the use of pesticides should be kept to a minimum.

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