SCOPE 51 - Biogeochemistry of Small Catchments

11 Nitrogen Cycling

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11.1 INTRODUCTION

Nitrogen is an essential nutrient required by plants in substantial quantities. The nitrogen cycle is perhaps the most complicated among the plant nutrient cycles (Figure 11.1). Nitrogen exists in the form of inorganic ions, in more or less complex organic compounds as well as in gaseous forms. Considering this diversity of nitrogen compounds existing in the ecosystem, it is not surprising that a great diversity exists both within and between the nitrogen cycles of natural ecosystems (Gosz, 1981; Melillo, 1981). This diversity and complexity complicate the study of nitrogen cycling on ecosystem level (plots, forest stands) and even more in complex terrains such as a catchment.

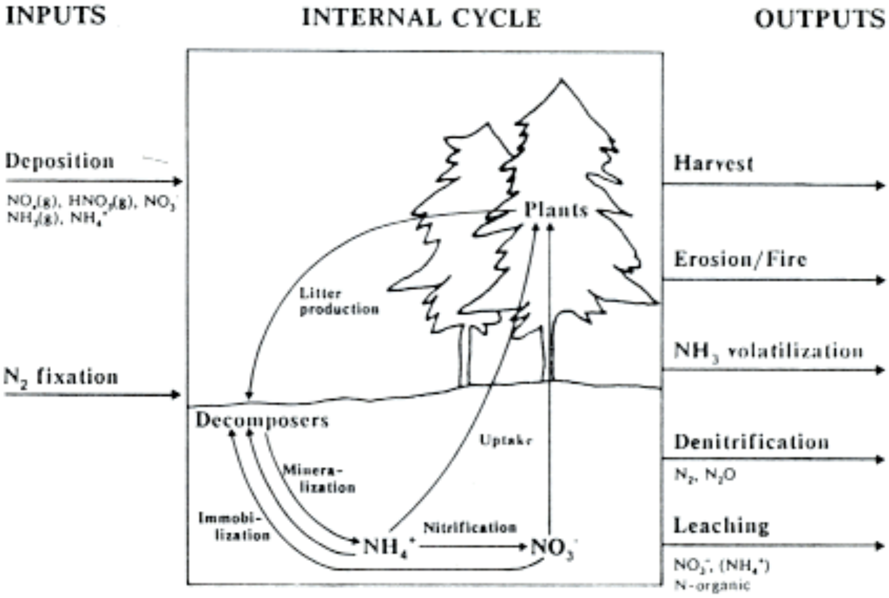

Figure 11.1 A simplified model of the N cycle as an internal cycle interacting with the surroundings by several processes (external cycle), The chemical forms of important inputs and outputs are indicated (from Gundersen, 1991; reproduced by permission of Elsevier Scientific Publishers BV).

Nitrogen is considered to be the growth-limiting factor in most terrestrial ecosystems, and natural ecosystems are characterized by a tight internal cycling of N. Leaching losses and gaseous losses are generally less than a few kg N ha^{-1} $year^{-1}$. High leaching losses may, however, occur after a disturbance of the system. In order to increase the yield of agricultural crops, addition of N fertilizer to agricultural ecosystems has increased dramatically during the last 50 years. This agricultural practice has markedly modified the N cycle. The high inputs are followed by large outputs by leaching, gaseous losses and crop removal. Due to this significant quantitative difference in N cycling in natural and agricultural ecosystems, this chapter is divided into two sections dealing with (i) changes in the bio-geochemical cycling of N in forested catchments as examples of natural or semi-natural ecosystems, and (ii) N cycling in agricultural or mixed catchments (agrogeochemical N cycling).

Because of the high leaching losses of N from agricultural ecosystems to groundwater and surface water, N is now recognized as an important pollutant. Nitrogen leaching, namely as nitrate from agricultural lands, affects drinking water quality and causes eutrophication of lakes and coastal areas. Nitrogen leaching is easily detected in the stream output from a catchment and may be related to major changes in the catchment such as disturbances, changes in management or fertilizer input. The small catchment concept is an important tool in monitoring changes in N cycling. This chapter will discuss the possible application of this concept in N research, rather than discussing processes of the N cycle which can be found elsewhere (Clark and Rosswall, 1981; Haynes, 1986; Sprent, 1987).

11.2 FORESTED CATCHMENTS

11.2.1 ENVIRONMENTAL PROBLEMS RELATED TO NITROGEN

The atmospheric N input to forests in Europe and North America has increased dramatically during the last decades due to the emission of NO_x from combustion processes and of NH_3 from agricultural activities (Pacyna, 1989). The N deposition to forest ecosystems generally exceeds 20 kg N ha^{-1} $year^{-1}$ in most of Europe and even reaches 100 kg N ha^{-1} $year^{-1}$ in some areas (Ivens *et al.,* 1990; Hauhs *et al.,* 1989). Forest ecosystems may accumulate considerable amounts of N in biomass and soil organic matter, but there is an increasing concern that forest ecosystems may be overloaded with N from atmospheric deposition. Indeed, increased leaching of nitrate has been observed in several areas of high N deposition (Nilsson and Grennfelt, 1988; Hauhs *et al.,* 1989).

On the other hand, nitrate leaching may also be a response of the N cycle to other factors such as management changes, forest decline and climatic change. Therefore the N problem must be addressed as a more complex interaction of causes, effects and environmental impacts. Management changes, forest decline and climatic change may alter N cycling by decreased plant uptake or enhanced mineralization and cause accumulation of inorganic N in the soil, nitrate leaching and/or increased de nitrification (partly as N_2O). Nitrogen in excess of plant demand may cause nutritional imbalances, leaching of nutrients and soil acidification, which independently or in combination with other factors contribute to forest decline (Nihlgård, 1985; Aber *et al.,* 1989; Gundersen, 1991). Further, nitrate leaching may affect groundwater and surface water quality in forested areas (Brown, 1988; Nilsson and Grennfelt, 1988) which are normally considered unpolluted with N, and even add to the eutrophication of coastal areas (Fleischer and Stibe, 1989). Increased N cycling and N availability in soils may lead to increased N_2O emission especially under acidic conditions (Schmidt *et al.,* 1988) and possibly also decrease methane consumption in soils (Steudler *et al.,* 1989). The increased atmospheric pool of these greenhouse gases contributes to global warming.

Some of the complex interactions and feedback of causes and effects in the N cycle may be addressed by studying small catchments at different environmental conditions or being manipulated. This part of the chapter will discuss the problems of defining changes in the N cycling of small forested catchments, and how to detect and interpret such changes.

11.2.2 NITROGEN SATURATION

Most temperate forest ecosystems are traditionally considered N-limited. Generally, fertilizer experiments have shown tree growth response to N additions, which has led to the concept that forest ecosystems are able to retain high N inputs from atmospheric deposition or from enhanced mineralization of the soil pool by increasing growth. This concept is, however, questioned by the observation of nitrate leaching from the root zone at several sites (Nilsson and Grennfelt, 1988; Hauhs *et al.,* 1989) which indicates that forest ecosystems have some kind of maximal capacity to immobilize N in soil and biomass. Other nutrients, water or light may become limiting for the primary production. This state of the ecosystem is called "N saturation".

The term N saturation is poorly defined in the literature and the validity of the concept is under discussion (e.g. Skeffington and Wilson, 1988). The discussion in the literature may be summarized by the following three attempts to define a N-saturated ecosystem: (i) an ecosystem where "availability of inorganic N is in excess of total combined plant and microbial nutritional demand" (Aber *et al.,* 1989); (ii) "an ecosystem where N losses approximate or exceed the inputs of N" (Ågren and Bosatta, 1988); and (iii) "an ecosystem where the primary production will not be further increased by an increase in the supply of N" (Nilsson, 1986).

These definitions can be understood as different stages of saturation related to different components in the ecosystem. Definitions (i) and (ii) are related to the state of the ecosystem, whereas (iii) is related to ecophysiological concepts (some kind of optimum curve for growth). A forest ecosystem leaching nitrate (or ammonium) is saturated in the sense of the first definition, but may still respond to N additions (e.g. in the spring) and still accumulate a considerable amount of N in the biomass. The second definition, which implies an accumulation in the system close to zero, may, from a theoretical point of view, be the most proper use of the term "saturation", and it is comparable with the concept of steady state in forest ecosystems. It is likely that the natural state of forest ecosystems in a long-term perspective is a "true saturation" where output equals input. But in that case, atmospheric N input should be constantly low (i.e. less than 2-5 kg N ha^{-1} $year^{-1}$, which is found as input in pristine areas). Moreover, in practice most forest ecosystems are harvested and would accumulate some N in the harvested biomass and for that reason never reach this kind of true saturation.

Although the term N saturation may be ambiguous, the concept is useful in the discussion of possible effects of chronic N additions to forest ecosystems and effects of forest decline, forest management and climate change on N cycling. For this purpose N saturation may be defined in accordance with definition (i) as a condition where the availability of mineral N exceeds the capacity of the ecosystem organisms to absorb N. By this definition N saturation implies a permanent change in the functioning of the N cycle from a virtually closed internal cycle to an open cycle, where excess N is leached from the system. Nitrogen saturation is easily detected as increased leaching of nitrate from the root zone or-in non-nitrifying or poorly nitrifying soils-increased accumulation of ammonium in the soil (at extreme atmospheric loads even ammonium leaching). Increases of nitrate leaching and/or ammonium accumulation should be considered in comparison with background levels from unaffected areas.

N saturation, by this definition, should be considered on a plot scale, since elevated nitrate concentrations under the root zone do not necessarily show up in the stream output of a catchment. This was for instance observed in the Strengbach catchment, Vosges massif, France where soil water concentrations below the root zone were 2.3 mg NO_3^--N l^{-1} but only 0.3 mg NO_3-N^{-1} appear in the draining stream (Probst *et al.,* 1990). Nitrate leached from the root zone of a N-saturated forest may be accumulated in bogs, lost by denitrification within the water- saturated parts of the catchment or transformed within the draining streams.

11.2.3 INTERACTIONS IN THE FOREST NITROGEN CYCLE

The majority of the available catchment studies were initiated to characterize S cycling and its effects on acidity in runoff (see Chapter 10). Studies focusing on nitrogen appeared later and were largely stimulated by the observations of N leaching in hydrochemical budgets (Chapter 8). But the interpretation of the budgets is much more complicated for N than for S. This can be illustrated by the biogeochemical cycling of S and N in a forest plantation where input and output of these elements are comparable, but the internal cycling and the soil pool of N is a factor 30 higher than for S (Table 11.1). It is clear that simple input-output relations cannot be expected for N. In addition, the N budget is complicated by the possibility of biological N fixation and the gaseous losses by denitrification. The comparison of nutrient cycles in Table 11.1 emphasizes the importance of combining catchment studies with plot studies to account for the internal processes and their possible changes.

Table 11.1 External and internal fluxes and soil pool of N and S in a dense Norway spruce forest plantation in Stroedam, Denmark (Freiesleben, 1987; Gundersen, 1989)

	Sulphur	Nitrogen
Flux (kmol ha^{-1} $year^{-1}$)		
Atmospheric deposition (throughfall measurements)	0.8	1.4
Internal cycling (estimated as the litterfall flux)	0.2	7.1

Leaching loss (estimated from soil water concentrations and hydrological budgets)	0.9	1.2
Pool (kmol ha^{-1})		
Soil pool (forest floor and mineral soil to 25 cm depth)	10	300

Inputs of N by biological fixation are generally small in forest ecosystems compared to atmospheric deposition inputs, except for ecosystems with N fixing plant species (e.g. alder forests) (Boring *et al.,* 1988). It is difficult to estimate total N deposition because N compounds may be assimilated in the canopy by foliage, bark or epiphytic organisms (Eilers *et al.,* 1990; Gebauer *et al.,* 1991). N fluxes in throughfall provide a minimum estimate for the N input by wet and dry deposition. The measured throughfall fluxes in Europe are 10 to >100 kg N ha^{-1} year^{-1} (Ivens *et al.,* 1990) and in North America 2 to 40 kg N ha^{-1} year^{-1} (Aber *et al.,* 1989). The elevated N deposition is a continuous addition to the background flux of mineral N from net mineralization, which normally amounts to 30-50 kg N ha^{-1} year^{-1} in coniferous stands (Gosz, 1981) and to 50-150 kg N ha^{-1} year^{-1} in deciduous stands (Melillo, 1981). In the long term these additions may change the pattern of internal cycling and exceed the capacity of plants and soils to retain N.

The development of N saturation by increased N inputs or other environmental changes involves a complex interaction of the processes in the N cycle, and knowledge about these interactions is still very limited. Recently, at least two corresponding conceptual hypotheses for ecosystem response to chronic N additions have been published (Aber *et al.,* 1989; Gundersen, 1989, 1991). In the N-limited ecosystem, added N is effectively absorbed by plants (and microbial biomass). The canopy expands and primary production is increased. The internal cycling of N is accelerated by decreased C/N ratio of litter and increased litter production, decomposition, mineralization and tree uptake. As N availability is improved, the composition of the forest floor vegetation may gradually change towards more nitrophilic species, and the nitrification process may be stimulated. Nitrate may be formed at a high rate even at very low pH in the soil (Gundersen and Rasmussen, 1990). Nitrification is a crucial process for N losses by nitrate leaching and denitrification. Nitrate is relatively mobile in soils and is easily leached by percolating water, whereas ammonium is retained in the soil by cation exchange.

When the canopy has reached its maximal size, the N utilization efficiency will decrease. The primary production may at least periodically be limited by essential resources other than N, and the ecosystem approaches N saturation. At this stage the ecosystem may be destabilized by the interaction of a number of factors (Nihlgård, 1985; Roelofs *et al.,* 1988; Schulze, 1989; Gundersen, 1989, 1991): (i) increased potential for water stress by increased canopy size, increased shoot/root ratio and loss of mycorrhizal infection; (ii) root damage caused by climatic acidification pushes from nitrification may appear; (iii) absolute or relative nutrient deficiencies may develop and even aggravate from loss of mycorrhiza or root damage; (iv) high mineral N concentration in the soil may cause accumulation of N in foliage (e.g. as amino acids), which may affect frost hardiness and the intensity and frequency of insect and pathogenic pests.

After reaching the point of N saturation, N leaching will continue to increase. At this stage of "N excess", soil acidification from N transformations adds to the proton load from acid deposition (van Breemen *et al.,* 1984). Leaching of nitrate is accompanied by leaching of base cations and, in acid soils, by Al as counterion. Furthermore, in acid soils, acidification pushes from nitrification of excess N may totally determine the episodes with root-toxic soil conditions. The frequency, the duration and the effect of these acidification pushes are likely to increase with the degree of N saturation (Gundersen and

Rasmussen, 1990). Loss of base cations from the ecosystem may in the long term reduce site fertility and contribute to the onset of nutrient deficiencies.

The time and the amounts of N additions required to reach N saturation in a forest ecosystem have not been discussed. These will differ widely between ecosystems depending on preconditions, N load and influence from other sources of stress (e.g. "acid rain", ozone, climate). At the present state of knowledge it is very difficult to assess the time scale and the N storage capacity of a specific ecosystem. However, the observed leaching of considerable amounts of nitrate from a number of European forest ecosystems (Table 11.2) may indicate that N saturation developed within the last 40 to 50 years, where the atmospheric N deposition amounted up to 800-1200 kg N ha^{-1} (Gundersen, 1989).

11.2.4 CRITICAL PARAMETERS FOR NITROGEN SATURATION

Nitrogen saturation occurs as a disruption of the N cycle (overload, enhanced mineralization of the soil pool, decreased uptake). It is desirable to identify factors affecting the sensitivity and resilience of the ecosystems to such disruptions, and possibly to identify critical markers for N saturation. The importance of some factors (i.e. water surplus, drainage and nitrifying ability) is known from the studies of effects of forest harvesting (Vitousek *et al.,* 1979). Additional research is needed to re-evaluate the full suite of factors on a more general basis. Studies of small catchments could be useful in this context.

Nitrogen saturation, using the above definition, occurs when the soil flux density of mineral N (defined as throughfall deposition plus net mineralization) exceeds the capacity of N uptake by plants (Ingestad *et al.,* 1981). This capacity is limited by the supply of other nutrients or water. At fertile sites light may be a limiting factor. The development of high mineral N flux density in the soil or low uptake in plants (e.g. forest decline) is the precondition for N leaching and in that way N saturation. It is likely also that denitrification will increase at this stage, but compared to leaching this flux is small on a plot scale (Section 11.2.6).

As the ecosystem approaches N saturation, N leaching will occur in the dormant season, where N uptake is small. Gradually, this biological control of nitrate leaching may cease. This change in the seasonal pattern of N leaching was shown by Hauhs *et al*. (1989) who compared catchments exhibiting different levels of N leaching (Figure 11.2). It was even possible to indicate a proceeding elimination of the seasonal pattern at the sites Dicke Bramke and Lange Bramke in the Harz Mountains, Germany, over a seven-year period when N leaching was increased. Loss of biological control of N leaching may be a possible critical marker for N saturation.

Since N leaching occurs when the mineral N flux density exceeds plant uptake, it is obvious that a decrease in plant uptake induced by harvest or forest decline may as well cause N leaching. N leaching from harvested or windfelled stands is related to the abrupt disturbance of the system and not directly to the status of the N cycle. It seems evident, though, that the N losses by leaching (and denitrification) from harvested areas will increase with the degree of N saturation of the forest before harvest. The potential losses in the first years after a disturbance may approach the mineral N flux density of the soil. In a comparative study simulating forest harvest Vitousek *et al*. (1982) actually found a positive correlation between nitrate leaching and the amount of N in annual circulation.

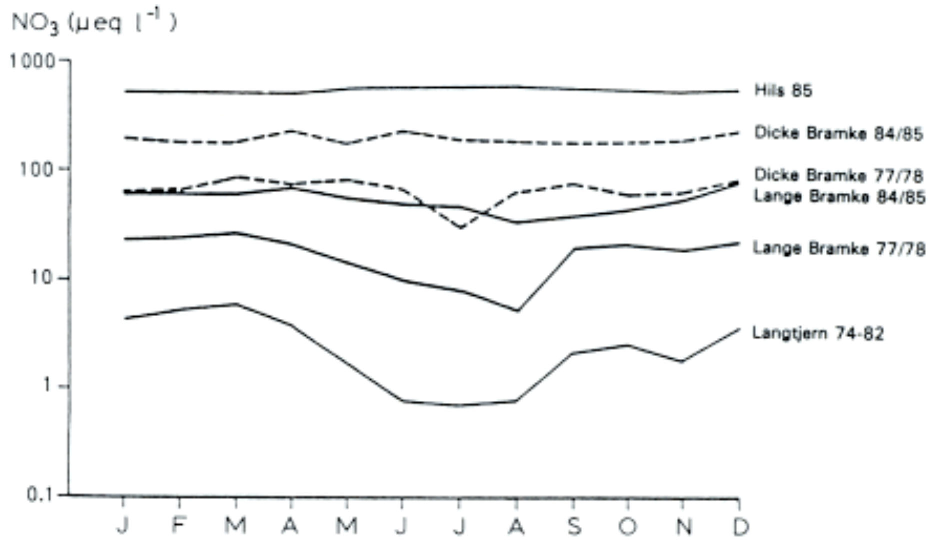

Figure 11.2 Seasonality of nitrate concentrations in streamwater output in forested catchments with mainly Norway spruce *(Picea abies)* at different levels of N leaching (Hauhs *et al.,* 1989; reproduced by permission of Michael Hauhs).

Table 11.2 Nitrogen cycle disruptions as indicated by various authors and the observed nitrate leaching or accumulation in the soil water
(see text). The studies shown are preferably from spruce forests

Disruption	Cause	Nitrate acc./leaching[a]	Method[b]	Site	Reference
Disturbance (removal of canopy cover)	Harvest clearcut	142 kg N ha^{-1} $year^{-1}$	r	Hubbard Brook, US	Likens *et al.,* 1970
		1-97 kg N ha^{-1} $year^{-1}$	1	several sites, US	Vitousek *et al.,* 1979
	Thinning	-	-	-	
	Natural causes				
	Windfelling		-	-	-
	Insect pests	0.5 kg N ha^{-1} $year^{-1}$	r 72-79	Coweeta, US	Swank *et al.,* 1981
		50 mg NO_3^- -N l^{-1}	1	Fredriksborg, DK (sitka)	Pedersen, 1993
	Fire	-	-	-	
Forestry	Liming	45 kg N ha^{-1} $year^{-1}$	1	Wülfersreuth,D	Hantschel *et al.,* 1990
manipulation		45 kg N ha^{-1} $year^{-1}$	1	Oberwarmensteinach,D	Hantschel *et al.,* 1990
		24mgNO_3^--N l^{-1}	1	Höglwald,D	Schierl and Kreutzer,1991

	Fertilization[c]	45 kg N ha^{-1} year^{-1}	1	DK	Holstener-Jörgensen, 1990
		20 kg N ha^{-1} year^{-1}	r	Gårdsjön,S	Westling and Hultberg, 1990
		14 kg N ha^{-1} year^{-1}	r	Schluchsee,D	Feger *et al.,* 1990
	Ditching	-	-	-	Fleischer and Stibe, 1989
	Coniferous after	8 kg N ha^{-1} year^{-1}	r	Schluchsee,D	Feger *et al.,* 1990
	deciduous	5-20 mg NO_3^--N l^{-1}	1	-, D	Kreutzer, 1981
		7-14 mg NO_3^--N l^{-1}	1	Höglwald,D	Kreutzer *et al.,* 1986
Forest decline	Reduced	13-20 kg N ha^{-1} year^{-1}[d]	r 81-87	Robinette, B	Hornung *et al.,* 1990
	growth rate	6 to 24 kg N ha^{-1} year^{-1}[d]	r 77-86	Dicke Bramke, D	Hauhs *et al.,* 1989
		18 kg N ha^{-1} year^{-1}	1 86-88	Strengbach, F	Probst *et al.,* 1990
		15 kg N ha^{-1} year^{-1}	1 85-88	Strödam,DK	Gundersen, 1992
		12kg N ha^{-1} year^{-1}	r 78-85	X-14, CS	Paces,1985
		11 kg N ha^{-1} year^{-1}	1	Oberwarmensteinach, D	Horn *et al.,* 1989
		1 to 6 kg N ha^{-1} year^{-1}[d]	r 77-86	Lange Bramke, D	Hauhs, 1989; Hauhs *et al.,* 1989
Nitrogen	N_2-fixation	51 kg N ha^{-1} year^{-1}	1	Cedar	
saturation	in alder forest			River Watershed, US	van Miegroet and Cole, 1984
		22 mg NO_3^--N l^{-1}	1	-, D	Kreutzer, 1981
	High atmospheric	23-87 kg N ha^{-1} year^{-1}	1 81-84	Hackfort, NL (oak)	van Breemen *et al.,* 1987
	deposition[e]	31 kg N ha^{-1} year^{-1}	1	Speuld, NL (doug. fir)	van der Maas, 1990
		29 kg N ha^{-1} year^{-1}	1 83-87	Tongbersven, NL (pine)	Mulder *et al.,* 1990
		27 kg N ha^{-1} year^{-1}	1 83-87	Gerritsfles, NL (pine)	Mulder *et al.,* 1990
		23 kg N ha^{-1} year^{-1}	1	Kootwijk, NL (doug. fir)	van der Maas, 1990
		20 kg N ha^{-1} year^{-1}	1 84-86	Hils (dept. 79), D	Wiedey and Raben, 1989
		19 kg N ha^{-1} year^{-1}	1	Wingst, D	Nilsson and Grennfelt, 1988
		17 kg N ha^{-1} year^{-1}	1	Leuvenum, NL (dfir/pine)	Tietema,1992

16 kg N ha^{-1} $year^{-1}$	1 85-88	Tange, DK	Gundersen, 1992
14 kg N ha^{-1} $year^{-1}$	1,r	Beddgelert, UK	Stevens and Hornung, 1988
13 kg N ha^{-1} $year^{-1}$	1 73-85	Solling, D	Matzner, 1988
13 kg N ha^{-1} $year^{-1}$	1	Wülfersreutz, D	Horn *et at.,* 1989
10 kg N $ha^{-1}year^{-1}$	1	Söderåsen,S	*Wiklander et al*., 1991
25 mg NO_3-_N l^{-1}	1	Ysselsteyn, NL (pine)	van Dijk *et al.,* 1992
21 mg NO_3^--N l^{-1}	1	Höglwald, D	Schierl and Kreutzer, 1991
18 mg NO_3^--N l^{-1}	1	Bohemian forest, AU	Kazda,1990

[a]Concentrations are yearly mean.

[b]l =in lysimeters below the rooting zone; r= in catchment runoff. If the data record is more than a few years the monitoring period is indicated.

[c]100-150 kg N ha^{-1} applied.

[d]Increasing over the monitoring period.

[e]Nitrogen deposition with throughfall exceed 20 kg N ha^{-1} $year^{-1}$ at these sites.

The mineral N flux density in the soil (throughfall deposition plus net mineralization) may thus be an important parameter, which could predict the degree of saturation in a forest ecosystem (Gundersen, 1989). It seems reasonable that N saturation only occurs at high N flux density. On the other hand, when the N flux density is increased, the nitrifying capacity of the soil, the water surplus and texture of the site, the type of forest, etc., may also be important parameters for N leaching (Table 11.3). If we assume almost steady state in the soil, the N flux in throughfall plus litterfall provides a simple estimate of the annual N flux density. A compilation of existing data on throughfall, litterfall and N leaching in forest stands may indicate if N flux density is relevant as a critical parameter. This approach is an improvement compared with input-output balances, since the internal cycling of N is also taken into account.

Table 11.3 Possible factors influencing the risk of N saturation and nitrate leaching after disturbance

Factor	Increasing risk	Decreasing risk
Precipitation surplus	High	Low
Soil texture	Coarse-sandy	Fine
Soil depth	Shallow	Thick
Atmospheric N load	High	Low
Vegetation	Coniferous	Deciduous
Rooting depth	Shallow	Deep
Internal N flux density	High	Low
Nitrifying capacity	High	Low
Nutrient pool and/or deposition		

(base cations, P or micro nutrients)	Low	Big

There are some indications of a qualitative importance for N saturation of the other factors listed in Table 11.3. The physical factors (precipitation, soil texture and depth) regulate the flow of water as the transport media for nitrate and the contact time of the root system with inorganic N. It is clear that the atmospheric load is important. Similarly the forest type is important, as the deposition to coniferous systems generally is higher than to deciduous systems (Ivens *et al.,* 1990). Further, the rooting system of coniferous trees is shallower than for deciduous trees, which may affect the ability to retain N. Not all soil types are able to nitrify. Non- or poorly nitrifying soils seem to be characterized by high C/N ratio and very low content of other nutrients (e.g. P, K, Ca) (Kriebitzsch, 1978). The available pool of macro and micro nutrients may also limit the primary production and thereby the capacity to store and circulate N. Water or light may also become limiting. Regression analysis of these factors comparing available plot and catchment studies may reveal the relative importance of different factors.

11.2.5 ELEVATED NITROGEN LEACHING AND ITS CAUSES

Elevated nitrate leaching needs to be considered in comparison with background levels from unaffected areas. There may be difficulties defining such areas in Europe. According to data in Andersen (1986), Hauhs *et al*. (1989), Driscoll *et al*. (1989) and Nilsson and Grennfelt (1988) nitrate leaching in natural coniferous forests is less than 1 kg N ha^{-1} $year^{-1}$; somewhat higher in deciduous forests, but still less than 2-3 kg N ha^{-1} $year^{-1}$. Well-drained soils subjected to high nitrate deposition may, however, leach nitrate during the dormant season without any actual disruption of the N cycle. The highly mobile nitrate ion may simply follow the water flux. Furthermore, elevated nitrate deposition may cause elevated nitrate concentration in lakes and streams due to surface runoff, e.g. during the snowmelt period (Brown, 1988, Driscoll *et al.,* 1989, Schofield *et al.,* 1985).

In addition to the leaching of nitrate, some nitrogen is lost as dissolved organic nitrogen (DON), but this flux is often not measured. In natural undisturbed forests DON leaching may be the most important output flux both on plot and catchment scale. In three small catchments in Gårdsjön, Sweden DON accounts for 60% of a total flux of 1.5 kg N ha^{-1} $year^{-1}$ in runoff (Westling and Hultberg, 1990/91).

A disruption of the N cycle may be triggered by many causes and mechanisms (Table 11.2), and the extent and duration may vary over wide ranges. To make safe conclusions on real changes in the N cycle and their causes or disruption mechanisms, long-term monitoring (decades) is required. Short-term monitoring may lead to misleading conclusions. The first portion of the 25-year monitoring record from watershed 6 at the Hubbard Brook Experimental Forest (HBEF), USA (summarized in Likens et al., 1977) could be interpreted as evidence of N saturation from increased atmospheric deposition (Ågren and Bosatta, 1988), since the N output in streamwater increased from 1.5 kg N ha^{-1} $year^{-1}$ in the mid-1960s to 5-6 kg N ha^{-1} $year^{-1}$ in the mid.:1970s approaching the input flux. But looking at the full record, the output decreased again and was constantly low during the 1980s (Driscoll *et al.,* 1989). The cause(s) of the periodically elevated nitrate leaching has not been determined.

In Table 11.2 a list of disruption mechanisms and examples of observed rates of nitrate leaching are given. If the data record is more than a few years, the monitoring period is indicated.

The most extensively investigated N cycle disruption is the disturbance by forest harvest. In the classical work of Likens *et al*. (1970) at HBEF, USA, watershed W2 was clearcut and treated with herbicides to prevent regrowth of vegetation. Since plant uptake was totally cut off, the nitrate leaching of 142 kg N ha^{-1} $year^{-1}$ constitutes an estimation of the maximum nitrification rate in this soil. The acidity produced by this N transformation (10 kmol H^+ ha^{-1} $year^{-1}$) decreased pH in runoff from 5.1 to 4.3 and released base cations. In disturbed ecosystems where plant regrowth is not inhibited or delayed, the impact of nitrification is much smaller, and the duration of elevated nitrate leaching is only a few years (Vitousek *et al.,* 1979). Disturbances by natural causes (e.g. fire, windfelling, insect pests) or by thinning may exhibit the same effect on N leaching as forest harvest, depending on the scale of disturbance and the decrease in tree uptake. The effects of harvest, fire, etc., are further discussed in Chapter 17.

Further long-term changes in the forest ecosystem can result in a shift of the humus and N storage in the soil to a lower level. An example of direct man-made induced shift in soil N storage is the plantation of spruce following a deciduous forest on a nitrifying site with high soil organic matter storage. Kreutzer (1981) and Kreutzer *et al*. (1986) observed elevated nitrate levels (50-20 mg NO_3^--N l^{-1}) at such sites. Based on runoff data from the Schluchsee catchment, Black Forest, Germany, Feger *et al*. (1990) suggest that change of forest type even causes elevated nitrate leaching after more than one rotation period. Similar losses of soil-borne N by nitrification might occur on lower-quality sites after application of nutrients limiting mineralization by liming or fertilization, but the duration may be short due to immobilization of nutrients by trees and microflora.

Recent studies have shown nitrate leaching in areas of forest decline in spruce forests (Hauhs and Wright, 1986; Hauhs *et al.,* 1989). This may be due to the reduced growth rate and N uptake by the damaged trees. It seems reasonable to speculate that such effects will more easily appear on sites subjected to high N deposition. Lysimeter and runoff data from these sites show nitrate outputs in the range 6 to 24 kg N ha^{-1} $year^{-1}$ (Table 11.2). All of these sites receive more than 20 kg N ha^{-1} $year^{-1}$ in throughfall. From a catchment in the area of severe forest dieback in the Krusne Hory Mountains, Bohemia, Czechoslovakia, increased nitrate leaching during and after the dieback of forest was reported by Paces (1985). Data from a regional survey of small streams in the Krusné Hory and other mountains show that elevated nitrate leaching is a general phenomenon in regions of severe forest dieback (Vesely, 1990). At the Lange Bramke catchment, Germany, nitrate leaching in runoff was strikingly correlated with the appearance of decline symptoms (yellowing, Mg deficiency) on the trees. Therefore nitrate leaching was proposed as a predictor of forest dieback (Hauhs and Wright, 1986; Hauhs *et al.,* 1989). Forest decline caused by shortage of base cations and/or Al toxicity may aggravate from accelerated soil acidification due to nitrification. This may result in a destabilizing, positive feedback loop involving increased decline, decreased retention of N in the ecosystem and possibly mobilization of the soil N pool.

The red alder forest is an exception to the general pattern of low nitrate leaching in natural forests. A high N_2 fixation rate can cause an increase of the soil N pool as well as of the nitrification rate. In a red alder forest in Washington, USA, up to 70 kg N ha^{-1} $year^{-1}$ was leached (van Miegroet and Cole, 1984).

Nitrogen saturation of forests caused by atmospheric deposition was first indicated in a study by van Breemen *et al*. (1982) at two locations in The Netherlands. The N input estimated from throughfall and stemflow exceeded 60 kg N ha^{-1} year in a deciduous as well as a pine stand. Van Breemen *et al*. (1987) summarize data from 1981 to 1984 from four plots at this deciduous forest site. Estimated annual nitrate leaching was in the range of 23-87 kg N ha^{-1} $year^{-1}$. Extreme soil acidification due to nitrification (4-14 kmol ha^{-1} $year^{-1}$) in the 10 cm surface soil was found at three of the four sites, but was partly alleviated at greater depth by removal of nitrate (van Breemen *et al.,* 1987). Similar examples of N saturation leaching 13-31 kg N ha^{-1} $year^{-1}$ are found in coniferous forests in The Netherlands (Mulder *et al.,* 1990; van der Maas, 1990; van Dijk *et al.,* 1992; Tietema, 1992), Germany (Matzner, 1988; Nilsson and Grennfelt, 1988; Schierl and Kreutzer, 1991; Wiedey and Raben, 1989) and Austria (Kazda, 1990). All these sites received 30 to 70 kg N ha^{-1} $year^{-1}$ in throughfall. At a spruce site in SoIling, Germany, data from the period 1973-85 showed an average nitrate leaching of 13 kg N ha^{-1} $year^{-1}$ receiving 30-35 kg N ha^{-1} $year^{-1}$ in throughfall. Although this elevated nitrate leaching was permanent over the monitoring period, considerable amounts of N were still accumulated in the ecosystem. A nearby beech forest showed no N cycle disruption (Matzner, 1988). Comparable nitrate leaching rates are also found in spruce forests at lower N deposition levels in throughfall (20 kg N ha^{-1} $year^{-1}$) in UK (Stevens and Hornung, 1988), South Sweden (Wiklander *et al.,* 1991), Denmark (Gundersen, 1992) and South Germany (Horn *et al.,* 1989). Leaching of ammonium is also found in extremely NH_3 polluted areas such as Peel in The Netherlands and Wingst in Germany (Nilsson and Grennfelt, 1988).

Forest decline symptoms and the appearance of nitrogen saturation coincide in many cases with deficiency of other elements such as Mg (Roelofs *et al.,* 1988; Schulze, 1989; Horn *et al.,* 1989; Feger *et al.,* 1990; Kazda, 1990; Probst *et al.,* 1990) and K (Roelofs *et al.,* 1988; Gundersen, 1992). This deficiency may limit the capacity of the vegetation and the soil to retain N inputs, causing nitrate leaching.

A compilation of data from input-output studies in Europe (Hauhs *et al.,* 1989) shows that outputs tend tobe either negligible (<1 kg N ha^{-1} $year^{-1}$) or high >5 kg N ha^{-1} $year^{-1}$ often approaching an input-Output balance. They hypothesized that this

> "may be caused by a self-accelerating effect when nitrate concentrations in soil solution increase in parallel with potentially toxic cations such as Al or Mn. If the nitrate uptake is temporarily lowered (e.g. in a dry period) the result may be damage to fine roots in the mineral soil. Such damage will further decrease uptake, and thus increase nitrate concentrations in the mineral soil. The damage will be manifested in needle loss and elevated nitrate export" (Hauhs *et al*., 1989).

The changes in rooting of declining forest have been observed in areas with elevated nitrate leaching (Murach, 1984).

From the case studies presented in Table 11.2 it seems evident that N deposition, forest decline and often forest management increase nitrate leaching on plot and small catchment scale. Recent studies have also documented increased N runoff from large catchments (river catchments). Nitrate concentration has increased during the last decade in a large number of lakes in South Norway and South Sweden and a similar trend was found in three rivers Bjerkereimsåna, Lygna and Ekso in southern Norway, which are not significantly affected by agriculture (Henriksen and Brakke, 1988). Also three forest rivers Lagan, Fylleån and Nissan in South Sweden draining more than 9000 km^2 showed a significant increase in N export during the period 1972 to 1987 (Fleischer and Stibe, 1989).

11.2.6 DENITRIFICATION AND NITROUS OXIDE EMISSIONS

The process of denitrification in forest soils has received more attention lately (Ineson *et al.,* 1991b), since this process might balance some of the N inputs, and thereby reduce the effects of excess N. But still field measurements of denitrification fluxes in forest soils are rare. Most observations indicate very low denitrification rates in soils of undisturbed forests (<1 kg N ha^{-1} $year^{-1}$), whereas denitrification losses after clearcut, where N availability, moisture and temperature are much more favourable for denitrification than in growing forests, amount to 3-6 kg N ha^{-1} $year^{-1}$ (Klemedtsson and Svensson, 1988; Robertson *et al.,* 1987). However, higher denitrification losses-5 to 7 kg N ha^{-1} $year^{-1}$-were identified by Brumme *et al*. (1989) and Ineson *et al*. (1991a) in some forest soils. Denitrification may thus alleviate effects of excess N in some ecosystems, but it is not yet clear which soil factors determine the appearance of high denitrification losses in these ecosystems (cf. Chapter 6). Although denitrification in many cases seems to be of minor importance in forest ecosystems on a plot scale, it may be an important process of the N cycle for an entire forest catchment, which includes bogs and stream sides with more favourable conditions for denitrification.

Increased denitrification losses from forested ecosystems due to elevated atmospheric N input and changes in management practice (e.g. clearcut harvest) are not desirable, since gaseous N losses from acid forest soils mainly occur as N_2O (Klemedtsson and Svensson, 1988) which may alter atmospheric chemistry and contribute to global warming (Wang *et al.,* 1976). Schmidt *et al.* (1988) estimate a N_2O emission of 0.7-1.5 Tg N $year^{-1}$ from temperate forest soils, which seems to be the largest individual source of atmospheric N_2O.

11.2.7 CONCLUSIONS AND RESEARCH RECOMMENDATIONS

Catchment studies are an important tool in monitoring changes of the forest N cycle. Observed changes in the catchments may relate to regional changes in N cycling and storage, and show up as changes in runoff from river catchments. On the other hand, it is very difficult to draw conclusions about the causes of such changes, due to the complexity and feedback between causes and effects in the N cycle. There is a need to combine whole catchment studies with several plot-scale studies representing the variability of N cycling within the catchment to be more conclusive about the cause-effect relationship. Long-term monitoring (decades) is needed to increase knowledge about the natural variability in cycling processes and nitrate output. There is a need of research focusing on internal cycling and processes of the total N cycle. A valuable approach are long-term manipulation studies (i.e. increasing and/or decreasing N input, increasing temperature, etc.) at both plot-scale and catchment-scale. Such experiments are under way on a number of sites in the USA (Aber *et al.,* 1989) and in Europe (the NITREX project, Wright *et al.,* 1992).

Specific research needs identified by Nilsson and Grennfelt (1988), Malanchuk and Nilsson (1989) and Hantschel and Beese (1991) include the following:

1. Accurate estimates of total N deposition. Critical data gaps exist for the dry deposition of N compounds (NO_x, HNO_3 vapour, NH_3 (including co-deposition with SO_x)), role of organic N deposition, and canopy uptake and turnover of N.
2. Better estimates for natural background of N emission, deposition and leaching (including dissolved organic N), and its natural variability.
3. Importance of humus type (mull/mor), soil fauna, microorganisms, forest floor vegetation and forest type for N cycling and storage.
4. N cycling and storage in relation to other macro and micro nutrients. Will additions of limiting elements (e.g. Mg, K, Ca, P, Mo) reverse N saturation?
5. Regulating factors for mineralization, nitrification and denitrification (especially N_2O emission) in terrestrial ecosystems.
6. Critical markers for changes in N cycling and the importance of forest decline for N leaching.

Research priorities should include:

1. Compilation of existing data on the internal N cycle and N storage in forest ecosystems in relation to management and input-output relations.

2. Investigation of the total N cycle on different sites (e.g. by use of the stable nitrogen isotope ^{15}N on plot and catchment scale), including studies of the interaction of several processes in the cycle, such as large-scale ecosystem studies manipulating inputs and N cycling, and simulating climate change.

3. Development and verification of predictive mathematical models for N dynamics in forests.

11.3 AGRICULTURAL CATCHMENTS

The input of increasing amounts of technogenic nitrogen into small agricultural catchments affects all components of the biogeochemical nitrogen cycle. This will transform the landscape into an agrogeochemical ecosystem with open nutrient cycles characterized by (i) high nitrogen export in crops, (ii) appearance of nitrate in groundwater and (iii) increased denitrification.

Small catchments with similar soil and climate conditions and similar anthropogenic loads are most suitable for investigation of geochemical cycles in agricultural land. The small catchment concept is now used in agricultural systems in Denmark and Sweden to evaluate the effects of different soil types, crops and management strategies on nitrate leaching (Anonymous, 1989). The significance of this approach, which includes quantification of all possible pools, sources and sinks of nitrogen, as well as the rate of circulation between them, is apparent; nutrient budgets are a synthesis of experimental and theoretical biological and geochemical approaches incorporating plot, ecosystem and landscape structure and function (Ayers and Branson, 1973; Messer, 1978; Bashkin and Bochkarev, 1979; Robertson, 1982; Bash kin, 1987a, b).

Nitrogen inputs to agricultural and mixed catchments include mineral and organic fertilizers, atmospheric deposition, irrigation water, lateral migration between landscape elements, symbiotic and non-symbiotic nitrogen fixation. Nitrogen export from agricultural landscapes results from removal of agricultural production (harvest), surface, subsurface and groundwater runoff, denitrification and volatilization of NH_3 (Bash kin, 1987b). Using also plot experimental data it is possible to assess the biogeochemical metabolism inside catchments and to explain the causes, direction and magnitude of many fluxes of nitrogen (Table 11.4).

Table 11.4 The suitability of catchment-scale and plot scale experimental data for biogeochemical studies of N cycling in small catchments

Processes	Approaches	
	Catchment	Plot
Input		
Fertilizers	+	-
N-fixation	-/+	+
Atmospheric deposition	-	+
Seeds	+	+

Internal cycling		
Organic matter mineralization	-	+
Nitrification	-	+
Crop uptake	-/+	+
Leaching	-	+
Output		
Harvest	+	-
Denitrification	-	+
NH_3 volatilization	-	+
Surface runoff	+	-/+
Subsurface runoff	+	-/+
Groundwater	+	-

11.3.1 CROP UPTAKE AND ACCUMULATION

A good example of plot data useful in catchment studies is provided by detailed investigation of nitrogen cycling in wheat *(Triticum aestivum L.*), which examined the effects of N surplus and deficit in the soil and atmosphere in relation to translocation within the plant (Harper *et al.,* 1987). Nitrogen concentrations were measured concurrently in soil, plant and atmosphere. Isotope and total N studies showed that after anthesis about half of grain N came from remobilization from leaves and stems and the other half directly from the soil. A progressively larger percentage of N came from mineralized organic matter as the season progressed. Nitrogen was lost as ammonia from the plant after fertilization and during the senescence.

Fertilizer N uptake and mineralization rates were determined four times during the spring growing season in an agricultural plot (Parton *et al.,* 1988). Fertilizer N levels in the surface layer (0-7.5 cm) decreased rapidly due to plant uptake and immobilization. About 80% of the fertilizer N (73 kg N ha^{-1}) were utilized by plants, 61% of this amount was taken up within 28 days. During early growth, N uptake was 1.3 kg ha^{-1} day^{-1}, but during the elongation stage, fertilizer N was immobilized and uptake ceased. Input from rainfall was 3 kg N ha^{-1} $year^{-1}$ and output in runoff 0.3 kg N ha^{-1} $year^{-1}$. For the period studied in this plot system the net accumulation was 33 kg N ha^{-1} $year^{-1}$.

Table 11.5 Nitrogen budget of a small agricultural catchment (46 ha), Sabadmezo in Czechoslovakia (Bashkin *et al.,* 1986)

Process	Year (September to August)		
	1981/82 (kg N ha^{-1})	1982/83 (kg N ha^{-1})	1983/84 (kg N ha^{-1})
Initial N content	253 .5	244 .5	200 .0

N-fixation	-	-	110 .0
Plant uptake	-131 .9	-122 .7	-183 .2
Surface runoff	-2 .7	-14 .4	-14 .0
Subsurface runoff	-	-27 .2	-36 .5
Balance	118 .9	80 .2	76 .3

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Maximum accumulation of N seems to occur in water-saturated landscapes. A comparison of accumulation in grey forest and floodplain soils was made in a calcareous catchment in the Oka River valley (Bash kin, 1989). Due to heavy application of mineral fertilizers and irrigation on the eluvial grey forest soils, N was leached by lateral runoff into the floodplain soils below and accumulated. The accumulations in eluvial, transeluvial, and saturated areas were 85, 64 and 101 kg N ha^{-1} year^{-1}, respectively. Fertilizer application to eluvial soils (165-220 kg N ha^{-1} year^{-1}) was the main input and denitrification in the saturated area was the main output (71-128 kg N ha^{-1} year^{-1}). Similar results were obtained with heavily fertilized yellow-red soils used for intensive grapefruit cultivation and the grey humic soils of pastures on Isla de las Juventad, Cuba. In humid climate nitrogen leaches from eluvial and transeluvial landscapes and lateral runoff might cause accumulation in the saturated zone.

The N balance of the Sabadmezo catchment in the Chechejovka River valley was determined for three years (Table 11.5). Under conditions of low precipitation and low hydrological flow, N accumulation was substantial (119 kg N ha^{-1} year^{-1}). Accumulated N decreased (80 kg N ha^{-1} year^{-1}) with greater flow the second year, while in the third year, high flows leached 50 kg N ha^{-1} year^{-1}, but the system accumulated N due to a leguminous cover crop.

Nitrogen accumulation due to excessive application of mineral fertilizers has been demonstrated in many small agricultural catchments with varying soils and climates. The remainder of applied N is taken up by plants, leached in the groundwater or evolved into gas through denitrification. Such measurements are easily conducted on a plot scale, but are more difficult to extrapolate to the catchment scale. Cropping, cultivation and erosion have been shown to deplete N reserves in marginal farmlands of the southern plains in the USA. Some pastures were fertilized for 20-22 years (45 kg N ha^{-1} year^{-1}), but these differed from non-fertilized fields in N content only in the uppermost 5 cm of the soil, so N was not accumulating but being removed by cropping. The N accumulation rate appears to be considerably slower than the N depletion rate under past farming practices (Berg, 1989). A three-year nitrogen budget was estimated for a small (16 ha) hill pasture catchment in New Zealand (Cooke and Cooper, 1988). In this case 7 kg N ha^{-1} year^{-1} were exported in one year, 86% in reduced forms (TKN- Total Kjeldahl nitrogen) from saturated overland flow and the remainder as nitrate via soil water. TKN export could be predicted from peak flow during the event and peak flow for the seven days preceding the event. The stream system was a net sink for TKN except during large floods, which scoured organic-rich seepage areas

11.3.2 EXPORT OF NITROGEN

Denitrification losses from arable soils are influenced by drainage and cultivation as well as fertilizers and soil organic matter. Denitrification losses were compared on clay soils (drained and not drained, direct-drilled or conventionally ploughed) over four years (Colbourn, 1988). Cultivation limited denitrification through soil aeration. In drained land, direct-drilled soil lost 9 kg N ha^{-1} year^{-1}, while plowed soil lost only 3 kg N ha^{-1} year^{-1}. Drainage reduced denitrification losses by 50% in ploughed soils but had no effect on direct drilled treatments. Losses from denitrification amounted to 1-6% of fertilizer N applied.

A significant share of N has been shown to be lost as N_2O from coastal plain soils in USA through denitrification because most of them are acidic (Weier and Gilliam, 1986). Wetting and drying cycles did not appear to influence denitrification rates, but warm soil temperatures increased them; highest rates occurred during initial spring thaw. Microbial assays for nitrification and denitrification activity have indicated that the main nitrate sources are well-aerated soils and the main nitrate sinks are water-saturated soils (Cooke and Cooper, 1988). Spatial and temporal variability of denitrification in an acidic (pH 3.8) sandy loam soil were determined by the soil cover method (Christensen and Tiedje, 1988). They found that production of nitrous oxide by denitrification was very unevenly distributed Areas of consistently high activity were associated with pools of organic matter (dead *Escherichia coli* cells) decaying under anaerobic conditions. Spatial and temporal variations in denitrification activity may be used as an assay of organic matter decomposition in soils.

The spatial distribution of denitrification activity in sediment cores from a stream draining an agricultural catchment was studied in England using enzymatic assay and acetylene reduction (Cooke and White, 1987). It was estimated that denitrification reduced nitrate load in the River Dorn by 15% under summer base flow conditions. Sediment denitrification activity accounted for only 1% removal of added nitrate, the remainder being taken up by plants in a stream draining a pasture catchment (Cooke and Cooper, 1988). Retention of near-stream seepage areas is suggested as a measure for minimizing nitrate export. Mires and other water-saturated habitats near the draining stream can significantly decrease nitrate leaching due to denitrification activity (Kruk, 1990; Fleischer *et al.,* 1991).

The largest sources of atmospheric NH_3 are from fertilizer application and live-stock washes. Studies of long-term trends imply a 50% increase in NH_3 emissions in Europe between 1950 and 1980 (ApSimon *et al.,* 1987). A substantial amount of N is lost as volatile NH_3 from wheat plants after fertilizer application and during the senescence period; about 16 kg N ha^{-1} of the applied fertilizer was lost as volatilized NH_3 (Parton *et al.,* 1988).

11.3.3 LEACHING AND RUNOFF

Excessive accumulation of N in small agricultural catchments leads to enhancement of surface and subsurface runoff as well as leaching into groundwater. Nitrogen mobility in catchment plots amended with ^{15}N-labelled fertilizer was followed (Bash kin, 1987b); Nitrate was the most prevalent form of migrant, its movement was closely related to hydrological flow. Application of N fertilizers enhanced the nitrogen mineralization capacity of the soil, releasing nitrate from soil organic matter as well as fertilizer.

Surface runoff in agricultural catchments is greater than in forested areas, with N being exported mainly in soluble form (97%), mostly nitrate. Concentration of N in surface runoff is variable and discontinuous though more export occurs during high surface flows (Dorioz *et al.,* 1987). It is beneficial to use both plots and a catchment approach to determine surface runoff, drained agricultural systems can unify both methods (Gerds, 1987). Long-term nitrate leaching to surface waters is often studied using selected systematically drained agricultural fields under different cropping systems and climatic conditions; crop type was the most important factor in nitrate leaching (Gustafson, 1987). Similar results have been reported elsewhere (Jaakola, 1984; Kjellerup and Kofoed, 1983; Uhlen, 1978). Grass and clover leys of several years standing and a two-year-old grass ley showed mean nitrate losses 4-6 times smaller compared to losses from cereal production under similar discharge conditions. Leaching losses were reduced by almost the same rate when cereal fields were re-seeded, suggesting the use of re-seed as a catch crop when growing crops such as spring cereals. When leys were ploughed in late autumn, losses were comparable to standing ley conditions. Early ploughing (July) followed by repeated cultivations increased the losses by at least three-fold in comparison with the situation after cereals. If early ploughing (August) was combined with sowing of a winter crop then leaching level was less than after cereals. Lysimeter estimates of nitrate leaching losses in agricultural catchments in Sweden show that catch crops (such as rape), straw incorporation and application of manure very late in the season can substantially reduce nitrate leaching (Bertilsson, 1988).

Table 11.6 Nitrate leaching in agricultural catchments in relation to soils, crops and fertilizer load

Soil	Permeability	Fertilizer (kg N ha^{-1})	Crop	Method	Leachate (kg N ha^{-1})	References
Loess/loam	Mean	80	Rape	L	150-170[a]	Guster *et al*<.,1987
Meadow gley	Poor	0	Meadow/grasses	L	0.8-56[a]	Mrkvicka and Velich, 1988
		100			0.8-61[a]	
		200			0.8-91[a]	
		300			0.8-197[a]	
		400			1.3-297[a]	
Brown/humic/ clayey/loam	Poor	0	Grasses	L	2.2	Ulehlova 1987
		100			3.3	
		200			6.8	
Sand	High	0	Spring	L	67	Bertilsson,1988
		60	cereals		77	
		120			81	
		240			88	
Loam	Poor	0			36	
		100			43	
		200			45	
	slurrey				48	
	FYM				51	
Clay	Poor	100	Corn/soybean	D	18.1	Gilliam,1987
Loam		0	Forest		1.8	
Clay	Poor	100	Corn/soybean		6.6	
Loam		0	Forest		0.9	

Organic			Pasture:cereal	D		Gustafson, 1987
		41	67:33		13	
		88	23:69		10	
		108	18:73		7	
		118	8:61		29	
Sand	high	120	Cereals/winter	D	100-130[a]	Strebel *et al.*, 1984
		120	Winter intercrops		60- 70[a]	
		250			15-30[a]	
		250			60-90[a]	
		0	Forest		10-40[a]	
Loess	Mean	150	Cereals		30-60[a]	
Clay	Poor	96	Bermuda grass	W	2.1-3.6[a]	Sharpley *et al.*, 1985
Loam		45-90	Cotton/oats/sorghum		5-19.5[a]	
Clay/loam	Mean	174	Cereal/potato/corn	W	22	Bash kin, 1987a
		80			15	
Loam	Poor	35	Cereals		8	
Sand	High	40	Cereal/potato		7	

L, lysimeters; D, drainage; W, well. [a]($mg\ l^{-1}$).

Nitrate leaching from the root zone is controlled by an interacting suite of factors, including: climate (precipitation, irrigation, evapotranspiration), soil (topography, texture, nutrient levels, porosity), land-use (crop, cultivation and harvest practices) and N-fertilization (type, application procedures and times) (Duynisveld *et al.,* 1988). Nitrate leaching from agricultural land in the Rhine Valley of Germany was estimated to be 46-593 mg NO_3^--N l^{-1} using a soil water simulation model (Simon *et al.,* 1988). Nitrate concentration and loading in relation to fertilizer application to sandy soils of the coastal plains in the USA was evaluated; concentrations of 9 mg l^{-1} under 87 kg N ha^{-1} applied and 24 mg l^{-1} under 336 kg N ha^{-1} applied were reported (Habbard *et al.,* 1986). Fertilizer application rates seem to be the most important single factor in nitrate leaching from agricultural catchments (Table 11.6).

11.3.4 MIXED CATCHMENTS

Mixed catchments contain both agricultural and forested land and occur most frequently in small river basins and lowlands. Biogeochemical flux is altered by both components but N flux is dominated by fertilizer application. In the Gorodnyanka River Basin, the highest N, P and K accumulation occurs in catchments containing the highest proportions of agricultural land receiving the heaviest rates of mineral fertilizer (Bashkin, 1987a). Excess N can reach 50% of input values (100-200 kg N ha^{-1} $year^{-1}$) in many catchments (Kudeyarov and Bash kin, 1984) and may be leached into the groundwater. Groundwater pollution under mixed farming and woodland areas occurred deeply in coarse sandy soils under farmland and the borders of forests (Krajenbrink, 1987). Lateral transport of excess N can occur through surface runoff between landscape elements. Loadings of inorganic N on farmlands can directly influence N export from wetlands associated with them in the catchment (Richardson, 1987).

11.4 SUMMARY

The natural biogeochemical cycle of N in terrestrial ecosystems is conservative. Disruption of the natural biological controls over N cycling opens the cycle and causes significant leaching losses. In natural or semi-natural ecosystems like forests a disruption of the N cycle is often related to the breakdown of the ecosystem by felling or decline. Recently it has become evident that also excess N deposition loadings may disrupt the N cycle and cause increased N losses. This situation is often referred to as "nitrogen saturation". Further readings on changes of N cycling in forest may be found in Skeffington (1988), Nilsson and Grennfelt (1988), Malanchuk and Nilsson (1989), Brandon and Hüttl (1990), Hantschel and Beese (1991) and Grennfelt and Thörnelöf (1992).

Nitrogen saturation is a well-known phenomenon in agricultural systems where excess fertilizer N loadings in combination with crop removal, soil tillage, etc., result in open systems with high input-output fluxes. This cycling situation can be described as "agrogeochemical". More details on N cycling of arable land may be found in Haynes (1986), Bashkin (1989) and Andrén *et al*. (1990).

Excess N deposition and/or fertilizer application causes nitrate leaching by direct leaching (lack of plant uptake or microbial immobilization) and stimulation of mineralization of soil organic matter. These factors contribute to depletion of long-term soil fertility, increasing soil acidity and, eventually, acidification and eutrophication of surface waters. Excess N availability may increase denitrification and production of nitrous oxide, which affects ozone levels and global warming. Nitrate leaching is an easily measurable indicator of disruption in the terrestrial N cycle, and in many cases it may be the only one. Changes in nitrate leaching, i.e. increased leaching or changes in seasonal pattern, are early warnings of an opening of the N cycle, but it does not give information about the cause. We still need to define criteria for an absolute change in nitrate leaching compared to the natural background situation, and to set criteria for acceptable changes in N cycling patterns of arable land. Long-term monitoring and research programmes are required to reach such criteria.

The monitoring of small catchments appears to be an important tool in this context, although the output in the draining stream is the integrated result of a variety of cycling patterns in different habitats within the catchment. In the study of agricultural systems the small catchment concept should serve well because the output fluxes are high (50-100 kg N ha^{-1} $year^{-1}$). Nitrate leaching may relatively easily be related to fertilizer amount, cropping system, management practice and climate parameters. In contrast, the interpretation of N leaching from forested catchments is much more complex, since the output fluxes are small compared to the possible natural variation. A combination of plot-scale and catchment-scale studies is essential in order to be more conclusive about the cause-effect relation. Further, this will improve the understanding of the interaction of different landscape elements or habitats. Understanding of the interaction of areas of N sources and sinks in agricultural systems and managed forests may be useful for preservation and creation of landscape elements which function as barriers to N loss.

To improve the understanding and management of N cycling in both natural and agricultural systems, predictive mathematical models are required on both plot and catchment scale. Good mathematical models for natural N cycling may be the only approach to reach estimates of critical deposition loads for N to specific ecosystems such as forests. In agriculture predictive models and expert systems can be used as an instrument in management to minimize N loss (for example, by calculating fertilizer amount, and by coordinating application time with plant uptake needs).

11.5 REFERENCES

Aber, J.D., Nadelhoffer, K.J., Steudler, P. and Melillo, J.M. (1989) Nitrogen saturation in northern forest ecosystems. *BioScience* **39**: 378-386.

Ågren, G.I. and Bosatta, E. (1988) Nitrogen saturation of terrestrial ecosystems. *Environmental Pollution* **54**: 185-198.

Andersen, B. (1986) Impact of nitrogen deposition. In Nilsson, J. (Ed.): *Critical Loads of Nitrogen and Sulphur,* Nordisk Ministerråd, pp. 159-197.

Andrén, O., Lindberg, T., Paustian, K. and Rosswall, T. (Eds) (1990) *Ecology of Arable Land--Organisms, Carbon and Nitrogen Cycling. Ecol. Bull. (Stockholm)* **40**.

Anonymous (1989) *Water Environment Plan: Monitoring Program*. Danish Environmental Protection Agency, Project-report 115, in Danish.

ApSimon, H.M., Kruse, M. and Bell, J.N.B. (1987) Ammonia emissions and their role in acid deposition. *Atmospheric Environment* **21**: 1939-1946.

Ayers, R.S. and Branson, R.L. (1973) *Nitrates in the Upper Santa Ana River Basin in Relation to Groundwater Pollution*. Riverside, California, 59 pp.

Bashkin, V.N. (1987a) Landscape-agrogeochemical mass-balance of nitrogen in agricultural regions. In Moldan, B. and Paces, T. (Eds): *GEOMON International Workshop on Geochemistry and Monitoring in Representative Basins*. Geological Survey, Prague, pp. 224-226.

Bashkin, V.N. (1987b) *Agrochemistry of Nitrogen*. Puschino, 276 pp.

Bashkin, V.N. (1989) Biogeochemistry of nitrogen in agrolandscape. *Soviet Soil Science* **21**: 12-20.

Bashkin, V.N. and Bochkarev, A.N. (1979) Study of regional nitrogen balance. In *Cycle and Balance of Nutrients in Agriculture*. Puschino, pp. 170-176.

Bashkin, V.N., Mocik, A., Skorepova, I., Kudeyarov, V.N. and Nikitishen, V.I. (1986) Application of nitrogen (Fertilizers and state of the environment. In *Nitrogen Balance and Transformation of Nitrogen Fertilizers in Soils*. Puschino, pp. 104-154.

Berg, W.A. (1989) Soil nitrogen accumulation in fertilized pasture of the Southern Plains. *J . Range Management* **41**: 22-25.

Bertilsson, G. (1988) Lysimeter studies of nitrogen leaching and nitrogen balance as affected by agricultural practices. *Acta Agric. Scand.* **38**: 3-11.

Boring, L.R., Swank, W.T., Waide, J.B. and Henderson, G.S. (1988) Sources, fate and impacts of nitrogen inputs to terrestrial ecosystems: review and synthesis. *Biogeochemistry* **6**: 119-159.

Brandon, O. and Hüttl, R.F. (Eds) -(1990) *Nitrogen Saturation in Forest Ecosystems*. Proceedings of a workshop held in Aberdeen, UK, 21-23 Sept. 1988. Kluwer Academic Publishers, Dordrecht, The Netherlands, 135 pp.

Breemen, N. van, Burrough, P.A., Velthorst, E.J., van Dobben, H.F., de Witt, T., Ridder, T.B. and Reijnders, H.F.R. (1982) Soil acidification from atmospheric ammonium sulphate in forest canopy throughfall. *Nature* **299**: 548-550.

Breemen, N. van, Driscoll, C.T. and Mulder, J. (1984) Acidic deposition and internal proton sources in acidification of soils and waters. *Nature* **307**: 599-604.

Breemen, N. van, Mulder, J. and van Grinsven, J.J.M. (1987) Impact of acid atmospheric deposition on woodland soils in The Netherlands: II. Nitrogen transformations. *Soil Sci. Soc. Am. J.* **51**: 1634-1640.

Brown, D.J.A. (1988) Effect of atmospheric N deposition on surface water chemistry and the implications for fisheries. *Environmental Pollution* **54**: 275-284.

Brumme, R., Beese, F. and Loftfield, N. (1989) Gaseous nitrogen losses from soils-effects of liming and fertilization. *Berichte des Forschungszentrums Waldökosysteme, Reihe B,* **15**: 83-89.

Christensen, S. and Tiedje, J.M. (1988) Denitrification in the field analysis of spatial and temporal variability. In Jenkinson, D.S. and Smith, K.A. (Eds): *Nitrogen Efficiency in Agricultural Soils*. Elsevier Applied Sci., pp. 295-301.

Clark, F.E. and Rosswall, T. (Eds) (1981) Terrestrial nitrogen cycles. *Ecol. Bull. (Stockholm)* **33**.

Colbourn, P. (1988) The influence of drainage and cultivation on denitrification losses from an arable clay soils. In Jenkinson, D.S. and Smith, K.A. (Eds): *Nitrogen Efficiency in Agricultural Soils*. Elsevier Applied Sci., pp. 283-294.

Cooke, J.G. and Cooper, A.B. (1988) Sources and sinks of nutrient in New Zealand hill pasture catchment. III. Nitrogen. *Hydrol. Proc.* **2**: 135-149.

Cooke, J.G. and White, R.E. (1987) Spatial distribution of denitrification activity in a stream draining an agricultural catchment. *Freshwater Biol.* **18**: 509-519.

Dijk, H.F.G. van, Boxman, A.W. and Roelofs, J.G.M. (1992) Ysselsteyn, The Netherlands. In Dise, N. and Wright, R.F. (Eds): *The NITREX Project (Nitrogen Saturation Experiments)*. Ecosystem Research Report No.2. Commission of the European Communities. Directorate-General for Science, Research and Development, Brussels, 101 pp.

Dorioz, J.M., Pilleboue, E. and Ferhi, A. (1987) Phosphorus and nitrogen transfer in a small elementary agricultural watershed. In Moldan, B. and Paces, T. (Eds): *International Workshop on Geochemistry and Monitoring in Representative Basins*. Geological Survey, Prague, p.39.

Driscoll, C.T., Schaefer, D.A., Molot, L.A. and Dillon, P.J. (1989) Summary of North American data. In Malanchuk, J.L. and Nilsson, J. (Eds): *The Role of Nitrogen in the Acidification of Soils and Surface Waters*. Miljørapport 1989: 10 (NORD 1989:92), Nordic Council of Ministers, Copenhagen.

Duynisveld, W.H.M., Strebel, O. and Bottcher, J. (1988) Are nitrate leaching from arable land and nitrate pollution of groundwater avoidable? *Ecol. Bull. (Stockholm)* **39**: 116-125.

Eilers, G., Brumme, R. and Matzner, E. (1990) Investigations to the above ground uptake of ^{15}N from wet deposition by Norway spruce *(Picea Abies* Karst.). In Proceedings from a symposium on the 'Use of stable isotopes in plant nutrition, soil fertility and environmental studies', Vienna, 1-5 Oct. 1990, pp. 150-152.

Feger, K.H., Brahmer, G. and Zöttle, H.W. (1990) An integrated watershed/plot-scale study of element cycling in spruce ecosystems of the Black forest. *Wat. Air Soil Poll.* **54**: 545-560.

Fleischer, S. and Stibe, L. (1989) Agriculture kills marine fish in the 1980s. Who is responsible for fish kills in the year 2000? *Ambio* **18**: 347-350.

Fleischer, S., Stibe, L. and Leonardson, L. (1991) Restoration of wetlands as a means of reducing nitrogen transport to coastal waters. *Ambio* **20**: 271-272.

Freiesleben, N.E. von (1987) *Effekt af atmosfrisk deponerede svovlforbindelser på ionbalancer i en dansk na!leskov*. Ph.D. thesis, Lab. of Environmental Sciences and Ecology, Technical University of Denmark, 126 pp.

Gebauer, O., Katz, C. and Schulze, E.-D. (1991) Uptake of gaseous and liquid nitrogen depositions and influence on the nutritional status of Norway spruce. In Hantschel, R. and Beese, F. (Eds): *Effects of Forest Management on the Nitrogen Cycle with Respect to Changing Environmental Conditions*. Report from a European workshop held at GSF München 9-13 May, 1990. OSF-Bericht 43/91, pp. 83-92.

Gerds, W. (1987) The drainage-plot method. A possibility of controlling nutrient leaching in agroecosystems. In Moldan, B. and Paces T. (Eds): *GEOMON International Workshop on Geochemistry and Monitoring in Representative Basins*. Geological Survey, Prague, pp. 42-44.

Gilliam, J.W. (1987) Drainage water quality and the environment. In *Drainage Design and Management,* Proc. Fifth National Drainage Symposium. 14-15 December, 1987. St Joseph, MI, USA, Am. Soc. Agr. Eng., pp. 19-28.

Gosz, J.R. (1981) Nitrogen cycling in coniferous ecosystems. In Clark, F.E. and Rosswall, T. (Eds): *Terrestrial Nitrogen Cycles, Ecol. Bull. {Stockholm)* **33**, pp. 402-426.

Grennfelt, P. and Thornelof, E. (Eds) (1992): *Critical loads for Nitrogen*. NORD 1992: 41, Nordic Council of Ministers, Copenhagen, 428pp.

Gundersen, P. (1989) Air pollution with nitrogen compounds: effects in coniferous forest. Ph.D. thesis, Lab. of Environmental Sciences and Ecology, Technical University of Denmark (in Danish), 292 pp.

Gundersen, P. (1991) Nitrogen deposition and the forest nitrogen cycle: role of denitrification. *For. Ecol. Manage.* **44**: 15-28.

Gundersen, P. (1992) Nitrogen circulation in three spruce forests in Denmark. In Teller, A., Mathy, P. and Jeffers, J.N.R. (Eds): *Response of Forest Ecosystems to Environmental Changes*. Proc. of the First European Symposium on Terrestrial Ecosystems, Florence 20-24 May 1991. CEC, Brussels, pp. 724-725.

Gundersen, P. and Rasmussen, L. (1990) Nitrification in forest soils: effects from nitrogen deposition on soil acidification and aluminum release. *Reviews of Environmental Contamination and Toxicology* **113**: 1-45.

Gustafson, A. (1987) Nitrate leaching from arable land in Sweden under four cropping systems. *Swed. J. Agric. Res.* **17**: 169-177.

Guster, R., Heyen, J., Amberberg, A. and Brüne, H. (1987) Zur Stickstoff- und Mineralstoffauswashung aus Lössboden. *Landwirtsch. Forschung* **40**: 312-325.

Habbard, R.K., Oascho, G.J., Hook, J.E. and Knissel, W.G. (1986) Nitrate movement into shallow groundwater through a Coastal Plain sand. *Trans. ASAE* **29**: 1564-1571.

Hantschel, R. and Beese, F. (Eds) (1991) *Effects of forest management on the nitrogen cycle with respect to changing environmental conditions. Report from a European workshop held at GSF Munchen,* 9-13 *May,* 1990. OSF Bericht 43/91,186 pp.

Hantschel, R., Kaupenjohann, M., Horn, R. and Zech, W. (1990) Water, nutrient and pollutant budgets in damaged Norway spruce stands in NE-Bavaria (FRO) and their changes after different fertilization treatments. *Wat. Air Soil Poll.* **49**: 273-297.

Harper, L.A., Sharpe, P.R., Langdale, G.W. and Giddens, J.E. (1987) Nitrogen cycling in a wheat crop: soil, plant and aerial nitrogen transport. *Agron. J.* **79**: 965-973.

Hauhs, M. (1989) Lange Bramke: An ecosystem study of a forested watershed. In Adriano, D.C. and Hauhs, M. (Eds): *Acid Precipitation, Vol.* I. *Case Studies,* Springer-Verlag, New York, pp. 275-305.

Hauhs, M. and Wright, R.F. (1986) Regional pattern of acid deposition and forest decline along a cross section through Europe. *Wat. Air Soil Poll.* **31**: 463-474.

Hauhs, M., Rost-Siebert, K., Raben, G., Paces, T. and Vigerust, B. (1989) Summary of European data. In Malanchuk, J.L. and Nilsson, J. (Eds): *The Role of Nitrogen in the Acidification of Soils and Surface Waters*. Miljörapport 1989: 10 (NORD 1989: 92), Nordic Council of Ministers, Copenhagen.

Haynes, R.J. (1986) *Mineral Nitrogen in the Plant-soil System*. Academic Press, Orlando, Florida, 483 pp.

Henriksen, A. and Brakke, D.F. (1988) Increasing contribution of nitrogen to the acidity of surface waters in Norway. *Wat. Air Soil Poll.* **42**: 183-201.

Holstener-Jörgensen, H. (1990) Nitrogen saturation-sense or nonsense? In Brandon, O. and Hüttl, R.F. (Eds): *Nitrogen Saturation in Forest Ecosystems*. Proceedings of a workshop held in Aberdeen, UK, 21-23 Sept. 1988. Kluwer Academic Publishers, Dordrecht, The Netherlands, pp. 115-120.

Horn, R., Schulze, E.-D. and Hantschel, R. (1989) Nutrient balance and element cycling in healthy and declining Norway spruce stands. In Schulze, E.-D., Lange, O.L. and Oren R. (Eds): *Forest Decline and Air Pollution*. Ecological Studies, Vol. 77. Springer- Verlag, Berlin pp. 445-455.

Hornung, M., Rodá, F. and Langan, S.J. (Eds) (1990) *A Review of Small Catchment Studies in Western Europe Producing Hydrochemical Budgets*. Air Pollution Research Report 28, Commission of the European Communities, 186 pp.

Ineson, P., Dutch, J. and Killham, K.S. (1991a) Denitrification in a Sitka spruce plantation and the effect of clear-felling. *For. Ecol. Manage.* **44**: 77-92.

Ineson, P., Kjöller, A. and Struwe, S. (Eds) (1991b) *Denitrification inforests*. Report from a workshop held in Copenhagen, 6-7 Nov., 1989. EEC publication, Brussels, 58 pp.

Ingestad, T., Aronsson, A. and Ågren, G.I (1981) Nutrient flux density model of mineral nutrition in conifer ecosystems. *Stud. For: Suecica* **160**: 61-71.

Ivens, W.P.M.F., Lövblad, G., Westling, O. and Kauppi, P. (1990) *Throughfall monitoring as a means of monitoring deposition to forest ecosystems*. NORD Miljørapport 1990: 16, Nordic Council of Ministers, Copenhagen.

Jaakola, A. (1984) Leaching losses of nitrogen from a clay soil under grass and cereal crops in Finland. *Plant and Soi1***76**: 59-66.

Kazda, M. (1990) Indications of unbalanced nitrogen nutrition of Norway spruce stands. *Plant and Soil* **128**: 97-101.

Kjellerup, V. and Kofoed, A. (1983) Nitrogen fertilization in relation to leaching of plant nutrients from soil. Lysimeter experiments with ^{15}N. *Tidskr. Planteavl.* **87**: 1-22.

Klemedtsson, L. and Svensson, B.H. (1988) Effects of acid deposition on denitrification and N_2O emission from forest soils. In Nilsson, J. and Grennfelt, P. (Eds): *Critical Loads for Sulphur and Nitrogen*. Report from a workshop at Skokloster, Sweden, 19- 24 March 1988, NORD Miljørapport 1988: 15, Nordic Council of Ministers, Copenhagen, pp. 343-362.

Krajenbrink, G.J.M. (1987) Groundwater pollution under farming-land and woodland in a water catchment area. Verslagen en Medeling Commissie voor Hydrologisch Onderzock TNO **38**: 601-606.

Kreutzer, K. (1981) Die Stoffbefrachtung des Sickerwassers in Waldbeständen. *Mitt. Dtsch. Bodenkdl. ges.* **32**: 272-286.

Kreutzer, K., Deschu, E. and Hösl, G. (1986) Vergleichende Untersuchungen über den Einfluss von Fichte (*Picea abies* L.) Karst.) und Buche *(Fagus sylvatica* L.) auf die Sickerwasserqualität. *Forstw. Cbl.* **105**: 364-371.

Kriebitzsch, W.U. (1978) Stickstoffnachlieferung in sauren Waldböden Nordwest- deutschlands. *Scripta Geobotanica* **14**: 1-66, Goltze, Göttingen, FRG.

Kruk, M. (1990) Effects of mires on nitrogen and sulfate outflow from agricultural catchments. Inst. of Ecology PAS, Lomianki, Poland. Presentation at the SCOPE workshop 'Biogeochemistry of Small Catchments' held in Most, Czechoslovakia, 5-11 Nov. 1990.

Kudeyarov, V.N. and Bash kin, V.N. (1984) Study of landscape-agrogeochemical balance of nutrients in agricultural regions. III. Nitrogen. *Wat. Air Soil Poll.* **23**: 141-153.

Likens, G.E., Bormann, F.H., Johnson, N.M., Fisher, D.W. and Pierce, R.S. (1970) Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed-ecosystem. *Ecol. Monogr.* **40**: 23-47.

Likens, G.E., Bormann, F.H., Pierce, R.S., Eaton, J.S. and Johnson, N.M. (1977) *Biogeochemistry of a Forested Ecosystem*. Springer-Verlag, New York, 146 pp.

Maas, M.P. van der (1990) *Hydrochemistry of Two Douglas Fir Ecosystems in the Veluwe, The Netherlands*. Dutch Priority Program on Acidification, Report no.1 02.1.01 , RIVM, Bilthoven.

Malanchuk, J.L. and Nilsson, J. (Eds) (1989) *The Role of Nitrogen in the Acidification of Soils and Surface Waters*. NORD 1989:92 Miljörapport 1989: 10, Nordic Council of Ministers, Copenhagen.

Matzner, E. (1988) Der Stoffumsatz zweier Waldökosysteme im Soiling. *Berichte des Forschungszentrums Waldökosysteme/Waldsterben. Reihe A* **40**: 1-217.

Melillo, J.M. (1981) Nitrogen cycling in deciduous forests. In Clark, F.E. and Rosswall, T. (Eds): *Terrestrial Nitrogen Cycles. Ecol. Bull. (Stockholm)* **33**, pp. 427-442.

Messer, J. (1978) Nitrogen mass balance in Florida ecosystems. Dissertation, U. Florida, Gainesville. 415 p.

Miegroet, H. van and Cole, D.W. (1984) Acidification sources in red alder and douglas fir soils- importance of nitrification. *J. Environ. Qual.* **13**: 586-590.

Mrkvicka, J. and Velich, J. (1988) Nitrogen leaching in permanent grassland applied graduated rates of nitrogen. *Rostl. Vyr.* **34**: 179-188.

Mulder, J., Breemen, N. van, Rasmussen, L. and Driscoll, C.T. (1990) Aluminum chemistry of acidic sandy soils with various inputs of acidic deposition in The Netherlands and in Denmark. In Lewis, T.E. (Ed.): *Environmental Chemistry and Toxicology of Aluminum,* Vol. 10. American Chemical Society, pp. 171-194.

Murach, D. (1984) Die Reaktion der Feinwurzeln von Fichten *(Picea abies* Karst.) auf zunehmende Bodenversauerung. *Göttinger Bodenkundliche Berichte* **77**: 1-127.

Nihlgård, B. (1985) The ammonium hypothesis-an additional explanation to the forest dieback in Europe. *Ambio,* **14**: 2-8.

Nilsson, J. (Ed.) (1986) *Critical Loads for Nitrogen and Sulphur*. NORD Miljörapport 11, Nordic Council of Ministers, Copenhagen, 232 pp.

Nilsson, J. and Grennfelt, P. (Eds) (1988) *Critical Loads for Sulphur and Nitrogen*. Report from a workshop held at Skokloster, Sweden, 19-24 March 1988. NORD Miljørapport 1988: 15, Nordic Council of Ministers, Copenhagen, pp. 225-268.

Paces, T. (1985) Sources of acidification in Central Europe estimated from elemental budgets in small basins. *Nature* **315**: 31-36.

Pacyna, J.M. (1989) Atmospheric emissions of nitrogen compounds. In Malanchuk, J.L. and Nilsson, J. (Eds): *The Role of Nitrogen in the Acidification of Soils and Surface Waters*. Miljörapport 1989:10 (NORD 1989:92), Nordic Council of Ministers, Copenhagen.

Pedersen, L.B. (1993) Nutrient cycling in Sitka spruce, Norway spruce and beech stands in Denmark. Research series No.1-1993, Danish Forest and Landscape Res. Institute, Lyngby, 252pp (in Danish, Engl. abstract).

Parton, W.J., Morgan, J.A., Altenhofen, J.M. and Harper, L.A. (1988) Ammonia volatilization from spring wheat plants. *Agron. J.* **80**: 419-425.

Probst, A., Dambrine, E., Viville, D. and Fritz, B. (1990) Influence of acid atmospheric inputs on surface water chemistry and mineral fluxes in a declining spruce stand within a small granitic catchment (Vosges Massif, France). *J. Hydrol.* **116**: 101-124.

Richardson, C.J. (1987) Freshwater wetlands: transformers, filters or sinks. In *FOREM,* pp. 3-9.

Robertson, G.P. (1982) Regional nitrogen budgets: approaches and problems. *Plant and Soil* **67**: 73-80.

Robertson, G.P., Vitousek, P.M., Matson, P.A. and Tiedje, J.M. (1987) Denitrification in a clearcut lobolly pine *(Pinus Taeda* L.) plantation in the southeastern US. *Plant and Soi1* **97:** 119-129.

Roelofs, J.G.M., Boxman, A.W., van Dijk, H.F.G. and Houdijk, A.L.F.M. (1988) Nutrient fluxes in canopies and roots of coniferous trees as affected by N-enriched air-pollution. In Bervaes, J., Mathy, P. and Evers, P. (Eds): *Relation Between Above and Below Ground Influence of Air Pollutants on Forest Trees*. CEC Air Pollution Report No.16, Proceedings of an international symposium, Wageningen, NL, 15-17 Dec. 1987, pp. 205-221.

Schierl, R. and Kreutzer, K. (1991) Effects of liming in a Norway spruce stand. *Fertilizer Research* **27**: 39-47.

Schmidt, J., Seiler, W. and Conrad, R. (1988) Emission of nitrous oxide from temperate forest soils into the atmosphere. *J. Atmos. Chem.* **6**: 95-115.

Schofield, C.L., Galloway, J.N. and Hendry, G.R. (1985) Surface water chemistry in the ILWAS basins. *Wat. Air Soil Poll.* **26**: 403-423.

Schulze, E.D. (1989) Air pollution and forest decline in spruce *(Picea abies)* forest. *Science* **244**: 776-783.

Sharpley, A.N., Smith, S.J., Menzel, R.G. and Westerman, R.L. (1985) *The chemical composition of rainfall in the Southern Plains and its impact on soil and water quality*. Techn. Bull. T-162, Oklahoma State University. 46 pp.

Simon, W., Huwe, B. and van der Ploeg, R.R. (1988) Die Abschatzung von Nitratanstragen ans landwirtschaftlichen Mulzflachen mit Hilfe von Nmin-daten. f. z. *Pflanzenernaehr. Bodenk.* **151**: 289-294.

Skeffington, R.A. (Ed.) (1988) Excess nitrogen deposition. *Environmental Pollution* **54**: 159-296.

Skeffington, R.A. and Wilson, E.J. (1988) Excess nitrogen deposition: issues for consideration. *Environmental Pollution* **54**: 159-184.

Sprent, J.I. (1987) *The Ecology of the Nitrogen Cycle*. Cambridge University Press, 151 pp. Steudler, P.A., Bowden, R.D., Melillo, J.M. and Aber, J.D. (1989) Influence of nitrogen fertilization on methane uptake in temperate forest soils. *Nature* **341**: 314-316.

Stevens, P.A. and Hornung, M. (1988) Nitrate leaching from a felled Sitka spruce plantation in Beddgelert Forest, North Wales. *Soil Use Management* **4**: 3-9.

Strebel, O.J., Böttcher, J. and Duynisveld, W.H.M. (1984) Einfluss von Strandortbedigungen und Bodennutzung auf Nitratauswaschung und Nitratkonzentration des Grundwassers. *Landwirtschaft. Forschung, Kongressband, Karlsruhe:* 33-34.

Swank, W.T., Waide, J.B., Crossley, D.A. Jr and Todd, R.L. (1981) Insect defoliation enhances nitrate export from forest ecosystems. *Oecologia (Berl.)* **51**: 297-299.

Tietema, A. (1992) Nitrogen cycling and soil acidification in forest ecosystems in The Netherlands. Thesis, University of Amsterdam. Amsterdam, The Netherlands, 140 pp.

Uhlen, G. (1978) Nutrient leaching and surface runoff in field Iysimeters on a cultivated soil. III. Seasonal variations in chemical composition of drainage water. *Meld. Norg. Landbr: Hogsk.* **57**(29): 1-22.

Ulehlova, B. (1987) Chemical composition of rain and Iysimetric waters from grasslands with different rates of fertilization. In Moldan, B. and Paces, T. (Eds): *GEOMON International Workshop on*

Geochemistry and Monitoring in Representative Basins. Geological Survey, Prague, pp. 105-107.

Vesely, J. (1990) Water chemistry in small streams in the mountain regions of Bohemia. Presentation at the SCOPE workshop 'Biogeochemistry of Small Catchments' held in Most, Czechoslovakia, 5-11 Nov. 1990.

Vitousek, P.M., Gosz, J.R., Grier, C.C., Melillo, J.M, Reiners, W.A and Todd, R.L. (1979) Nitrate losses from disturbed ecosystems. *Science* **204**: 469-474.

Vitousek, P.M., Gosz, J.R., Grier, C.C., Melillo, J.M. and Reiners, W.A. (1982) A comparative analysis of potential nitrification and nitrate mobility in forest ecosystems. *Ecol. Monogr:* **52**: 155-177.

Wang, W.C., Yung, Y.L., Lacis, A.A., Mo, T. and Hansen, J.E. (1976) Greenhouse effects due to man-made perturbations of trace gases. *Science* **194**: 685-690.

Weier, K.D. and Gilliam, J.W. (1986) Effect of acidity on denitrification and nitrous oxide evolution from Atlantic Coastal Plain soils. *Soil Sci. Soc. Am. J.* **50**: 1202-1205.

Westling, O. and Hultberg, H. (1990/91) Liming and fertilization of acid forest soil: Short-term effects on runoff from small catchments. *Wat. Air Soil Poll.* **54**: 391-407.

Wiedey, G.A. and Raben, G.H. (1989) Comparison of wind side and lee exposition: Site, deposition and soil conditions at Hils low mountain range. *Berichte des Forschungszentrums Waldökosysteme, Reihe B,* **15**: 90-94.

Wiklander, G., Nordlander, G. and Anderson, R. (1991) Leaching of nitrogen from a forested catchment at Söderåsen in southern Sweden. *Wat. Air Soil Poll.* **55**: 263-282.

Wright, R.F., Breemen, N. van, Emmett, B., Roelofs, J.G.M., Tietema, A., Verhoef, H.A., Hauhs, M., Rasmussen, L., Hultberg, H., Persson, H. and Stuanes, A.O. (1992) NITREX-nitrogen saturation experiments. In Teller, A., Mathy, P. and Jeffers, J.N.R. (Eds): *Response of Forest Ecosystems to Environmental Changes*. Proceedings of First European Symposium on Terrestrial Ecosystems, Florence 20-24 May 1991, CEC, Brussels, pp. 335-341

Back to Table of Contents

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