

**Recommendations for establishing Action Programmes
under Directive 91/676/EEC concerning the protection of waters
against pollution caused by nitrates from agricultural sources**
Contract number N° 07 0307/2010/580551/ETU/B1

Part C
**Analysis of the processes in soil that
influence nutrient leaching and runoff**

Final Report

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Consortium

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Administrative summary

Contract number N° 07 0307/2010/580551/ETU/B1

The contract “*Recommendations for establishing Action Programmes under Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources*” (ENV.B.1/ETU/2010/0063) was signed by the Commission of the European Community on 6 January 2011, and by the consortium on 17 December 2010. Total duration of the contract is 10 months, starting on the day of signing the contract (6 January 2011) and ending on 6 November 2011.

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ABSTRACT

Effect of environmental and climatic conditions on NH₃ emission, N leaching to ground and surface waters and P leaching to ground and surface waters as well as the effect of eutrophication in surface water; Recommendations for establishing Action Programmes under Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources (ND-Act). Alterra, Wageningen-UR, Wageningen. 94 pp.

This report provides a detailed analysis and scrutiny of the existing literature related to the relationships between soil type and characteristics and the risk for leaching/runoff to groundwater and surface waters and eutrophication. Special emphasis is given to the links between soil processes, related to the transformation and transport of nitrogen and phosphorus in soils, and the risk for leaching/runoff to groundwater and surface waters, for each of the identified pedo-climatic (sub)zones.

Keywords: Nutrient losses, Nitrates Directive, Eutrophication, N leaching, P leaching, ammonia volatilization, Pedo-Climatic zones

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Executive summary

This report gives a detailed analysis of the processes in the soil, both in the surface soil and the subsoil that influence nutrient leaching and runoff that could lead to pollution of waters and eutrophication processes. This report is a Part C of the study “*Recommendations for establishing Action Programmes under Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources*”. The aim of the study is ‘to build further on the ERM-2001 study, cover the whole EU-27, and take account of the most recent relevant scientific and technical data on nitrogen emissions, agricultural practices and environmental and climatic conditions’. The ERM-2001 study aimed at assessing action programmes established at that time (EU-15) and aimed at enunciating main principles which should guide the establishment of measures, taking into account agronomic concepts, pedo-climatic zones and local conditions. As such the current report has to provide useful inputs for a better implementation of the Nitrates Directive across the whole EU-27.

The Part C report comprises an overview of existing literature on the relation between soil properties, nutrient losses, eutrophication and pedo-climatic zones. The focus is on the nutrients nitrogen (N) and phosphorus (P), with an emphasis on N and addressing both the aqueous losses and the gaseous losses. The results are applicable to the farming systems within the EU-27. However, research results from outside the EU-27 were also used, when relevant and related to pedo-climatic (sub) zones that are comparable with those within the EU-27. The general approach of the analysis between soil characteristics and soil processes and risks of water pollution by nutrients was:

- a detailed analysis and scrutiny of the existing literature related to the relationships between soil type and characteristics and the risk for leaching/runoff to groundwater and surface waters and eutrophication (Chapters 3, 5 and 9);
- a conceptual cause – effect schematization of soil type characteristics and processes that affect the vulnerability to nitrogen and phosphorus leaching (Chapter 4) and
- an analysis of the links between climatic factors and soil properties and soil processes, related to the transformation and transport of nitrogen and phosphorus in soils, and the risk for leaching/runoff to groundwater and surface waters (Chapter 6 to 8).

C, N and P interaction in farming systems Chapter 3 provides a global overview of the relations between carbon (C), N and P in farming systems. Aspects addressed in this chapter are (i) a general overview of C, N and P in the soil plant water system, (ii) the interaction between C, N and P and mineralization immobilization turnover, (iii) the anthropogenic changes of N and P and (iv) the role of nutrient limitation.

Carbon is the dominant element of living organisms, about 50% of the dry weight. It plays an important role in the structure, biochemistry, and nutrition of all living cells.

Photosynthesis enables plants to transform solar energy and CO₂ into sugars, starches and other forms of organic matter, being the bases for the food chain in both natural ecosystems and agro-ecosystem. Nitrogen is predominantly found in the atmosphere as N₂, i.e. ~80% of N on earth. Sedimentary rocks contain roughly the remaining 20%. Just a very small amount, <1% is found in oceans and in living and dead organic matter. Phosphorus is almost absent in the atmosphere. P occurs in small quantities in the earth's lithosphere, biosphere and hydrosphere.

In managed agricultural systems, N inputs occur largely via synthetic N fertilizers, animal manures, biological N₂ fixation and atmospheric deposition. Nitrogen withdrawal from these systems through harvesting, but also through N loss processes as denitrification, leaching, volatilization and soil erosion are compensated mainly by applications of synthetic N fertilizers and animal manure. In general, only about half of these N inputs are taken up by the crops, while the remainder is lost to the environment.

The main impact of humans on the phosphorus cycle is (i) through the use of P fertilizers and through the import of animal feeds and food from elsewhere (ii) through the discharge of residues and wastes in the wider environment (rivers, landfills, cement, etc.). The two main factors controlling the availability of soil P to plant roots are the concentration of phosphate ions in the soil solution and the P-buffer capacity, i.e. the ability of the soil to replenish these ions when plant roots remove them.

There is a strong linkage between the cycles of N, P and C in agriculture ecosystems. N and P are the two most important nutrients limiting biological production and are the most extensively applied nutrients in managed terrestrial systems, mainly as inorganic fertilizers and animal manures. Global N and P fluxes are projected to increase substantially, as the demand for food will increase by ~50 to 100% during the next few decades, and fertilizer use will increase to help meet the growing food demand.

Runoff and groundwater recharge

For the quantification and characterisation N losses through volatilization, denitrification and leaching, P losses through leaching and runoff, and N and P transport and retention in groundwater, information is needed on soil properties and hydrologic characteristics of the groundwater system. Hence, Chapter 4 focuses on the relation between soil properties and hydrological characteristics on the one hand and surface runoff, subsurface runoff and groundwater recharge on the other hand. Important factors influencing surface runoff are: precipitation (amount, type intensity and duration), factors affecting the evapotranspiration, land use soil texture and slope. Important factors influencing subsurface runoff are: precipitation surplus, soil texture, effective porosity and groundwater level. In view of N and P processes in groundwater there is a need to distinct in two subsystems (i) the shallow groundwater with the (partly) unsaturated zone with rapid transport of solutes in surface runoff and flow through shallow groundwater to local water courses (subsurface runoff) and (ii) the deep groundwater saturated zone with slow transport towards larger streams

and rivers. An important parameter of the deep groundwater is the residence time. This parameter is important for the prognosis of the long-term behaviour of groundwater systems in response to N and P inputs. The groundwater residence times specify the time scale for measures to remediate polluted groundwater to lead to a substantial groundwater quality improvement.

N and P losses

The N and P surpluses of agricultural land are indicators for the potential N and P losses to the environment, either to the hydrosphere and/or to the atmosphere (in case of N). In Chapter 5 an overview is given of the principal properties and processes determining N and P losses from agricultural systems, and how these losses are affected by soil properties. The fate of the N and P surpluses is controlled by a combination of factors, including type and rate of N and P inputs, soil type and properties, weather conditions, crop type and the hydrology of the soils.

The N losses from the soil comprise (i) gaseous emissions (ammonia (NH_3), nitrous oxide (N_2O), nitrogen oxides (NO and NO_2), di-nitrogen (N_2)) and (ii) aqueous losses, through surface runoff, subsurface runoff and leaching. In addition, there may be losses of solid (particulate) N and P through erosion and runoff. Many factors regulate the NH_3 loss from soil-plant systems to the atmosphere, including manure type, fertilizer type, application technique, weather conditions, the presence of a crop and especially the management of soils and fertilizers. In soils, there is equilibrium between NH_3 and NH_4^+ , which is affected by pH, moisture, clay content and wind. Factors shifting this equilibrium towards NH_3 , such as high pH, removal of NH_3 by wind, low moisture content and low clay content, enhance NH_3 emissions. Nitrous oxide, being the most important N related greenhouse gas, is produced during nitrification and denitrification by nitrifying and denitrifying bacteria, respectively. During this process also NO and NO_2 emissions take place. The impact of environmental factors on N_2O and NO_x is very diverse. Some factors enhance the activity of bacteria, but decrease the production of N_2O during nitrification and denitrification (e.g., temperature, pH). These counterbalancing effects complicate the estimation of the net effect of environmental factors on N_2O emission.

Surface runoff (overland flow) is the lateral flow of dissolved compound over the soil surface to lower lying areas and streams. Erosion is the lateral flow of soil particles, including the N and P attached to these particles over the soil surface to lower lying areas and streams. In general, surface runoff of N is more important than the erosion of N, but depending on land use and fertilization practices. However, for P it is the other way around; erosion of soil P is a more important loss process than runoff of dissolved P. Indicators for the risk of N and P surface runoff are: heavy precipitation, soil with low infiltration rate, high amount of fertilizer, type of vegetation, soil tillage and steep slopes. Indicators for the risk of erosion of N and P are: rainfall, the geomorphology, soil texture, land cover, land management, thickness of the soil and the permeability of the soil.

The N surplus is subjected to leaching and denitrification. The most important factors controlling denitrification are (i) the presence of an energy source for the

denitrifying bacteria, mostly available organic carbon, (ii) anoxic conditions, and (iii) the nitrate content in the soil. If any of these conditions is not fulfilled, denitrification is unlikely. Denitrification is a microbial process, and therefore environmental factors affecting biological processes, such as the temperature and pH, may also affect denitrification.

In contrast, the P surplus is not a strong indicator for P losses to the environment, as P losses are determined also by the soil P status, which is strongly influenced by the retention of the P surplus over the past years. The retention of the P surplus by the soil is controlled by a combination of factors, including soil type and properties, slope, weather conditions, crop type and the hydrology of the soils. Important differences with N are that (i) there are no gaseous P losses, (ii) the retention of P in the terrestrial ecosystem is much stronger due to low solubility of P compounds, (iii) P losses due to erosion play a major role and (iv) the relative contribution of point sources to the total loss is much larger for P than for N. Point sources discharges directly into water bodies, for example, disposal through wastewater treatment plants, industries and households (not connected to a sewage treatment plant). By contrast, nonpoint sources affect a water body from diffuse sources such as runoff from agricultural areas. Phosphorus applied to soils is involved in a multitude of complex reactions that remove it from the solution and incorporate it into a large variety of much less soluble and stable compounds. Related to P leaching, the most relevant question is to find out how much of the P is adsorbed in the soil soon after application and how much is available for crop uptake and leaching. Sandy soils and soils with nearly neutral pH have relatively little sorption capacity, whereas acidic, clayey soils with high iron (Fe) and aluminium (Al) content have the highest retention potentials.

Factors influencing N and P losses

A detailed analysis of climatic and soil properties affecting NH_3 volatilisation, N leaching and P leaching in view of factors used for the identification of the pedo-climatic subzones defined in Part A is presented in the Chapters 6 to 8.

NH_3 volatilisation

Analysis of European data of NH_3 volatilisation after manure application show that variables significantly affecting NH_3 volatilization throughout Europe are soil water content, air temperature, wind speed, slurry properties (pH, dry matter content and content of total ammoniacal nitrogen), application method, time and rate, slurry incorporation and measuring technique. Experiments have shown that the most important climatic factors controlling NH_3 losses are wind speed together with slurry dry matter content, and for applied solid manure, rainfall.

N surface runoff and downward leaching

A detailed analysis of climatic and soil properties affecting the transformation, retention and leaching and runoff of nitrate to groundwater and surface water showed that the effect of temperature is ambiguous since it increases the nitrate availability through enhanced mineralisation and nitrification, but the nitrate availability is also decreased through enhanced denitrification and uptake. The intra

annual temporal distribution of precipitation as well as dry and wet climatic cycles greatly affected N leaching. Soil texture influences drainage which influences the quantity of solute leaching and it influences the oxygen content of the soil, which controls the (de)nitrification and mineralisation processes. On a well-drained, sandy soil, NO_3^- leaching is higher than on poorly drained wet clay soil. Soils high in organic matter can mineralize a substantial amount of nitrate, which is susceptible to leaching, especially when wet years follow very dry years. High organic matter contents (large amount of available organic C) also increases denitrification potential and thereby decreases nitrate leaching. Hence, the effects of soil organic matter on nitrate leaching are complex. Finally, drainage and lowering of the ground water level results in increased mineralisation, especially in peat soils, which in turn may result in an increased N leaching.

The results of the aforementioned criteria, the reviews presented in Part A and B of this report have been synthesized into formulae, which allow the derivation of the surface runoff risk potential and nitrate leaching risk potential, based on a combination of land, soil and climate factors (i.e. pedo-climatic information). The formulae also provide the underpinning for the recommendations for the measures (1-12) in Annex II and Annex III of the Nitrates Directive, as function of pedo-climatic zones. The factors that most strongly control the surface runoff risk potential of N are included in the formula, i.e. the slope, the precipitation, soil type, crop, and the depth to rock. The factors that most strongly control the nitrate downward leaching risk potential are included in the formula, i.e. land use, soil organic C, precipitation, soil type and rooting depth.

P surface runoff and downward leaching

The concentration of dissolved P in the soil is largely determined by the P sorption characteristics of the soil. The P sorption capacity of soils may be influenced through changes in drainage and ground water level; at high ground water levels, the sorption capacity is relatively low due to the reduction of iron (hydr)oxides. This is also a main reason why sediments release dissolved P to the overlying surface waters. The effects of climatic factors on P leaching are very similar to those on N leaching. All factors related to the runoff of N also apply to P. However, since the water solubility of P is much less compared to N, runoff of P is of less importance. The opposite, however, is true for P losses by erosion. It is estimated that for the European continent the amount of eroded P is ~80% of the amount of P fertilizer use.

Eutrophication of surface waters

In Chapter 9 the factors influencing eutrophication of surface waters are considered. Eutrophication is the enrichment of surface waters with N and P with the resulting adverse biological effects. Both N and P typically promote the growth of algae, which in some cases lead to so-called 'algae blooms'. Increased algae growth leads to reduced transparency of the water. When the algae die and subsequently decompose, the decomposing organisms deplete the water of available oxygen, causing the death of other organisms, such as fish. Furthermore, some algal blooms produce toxins and change the taste and odour of the water and thereby make the water unsuitable

for use as drinking water and recreation. Some species emit toxic gases while decomposing, which are lethal to humans and animals.

(EEA, 2005). Eutrophication is a natural process, but human activity greatly speeds up the process.

Eutrophication is a natural process, but human activity greatly speeds up the process. Eutrophication is caused by large (natural and anthropogenic) inputs of the N and P to the aquatic environment. This can either be caused by surface and subsurface runoff of N and P to surface waters or downward leaching. Although natural sources of N and P may occur locally, in large parts of Europe, agriculture is a dominating anthropogenic source of surface water pollution with N and P.

The scientific knowledge about the effects of climate and soil characteristics on eutrophication are scarce. Still, it is assumed that climate change will be associated with a change in rainfall patterns and a more intense precipitation will increase surface and groundwater nutrient discharge into water bodies. Moreover, an ecosystem shift may occur when the temperature changes, since many organisms react upon changes in temperature regarding their productivity. However, responses are complex and may vary depending on the specific context. So, nutrient loading and changes therein, may affect ecosystems differently in different climatic areas, but it is impossible to give a general description and predict the exact effects.

Concluding remarks

Global N and P fluxes are projected to increase substantially, as the demand for food will increase by ~50 to 100% during the next few decades, and fertilizer use will increase to help meet the growing food demand. In general, only about half of these N inputs are taken up by the crops, while the remainder is lost to the environment, whereas the P inputs are more in balance with the requested crop uptake.

The N losses from the soil comprise (i) gaseous emissions (ammonia (NH_3), nitrous oxide (N_2O), nitrogen oxides (NO and NO_2), di-nitrogen (N_2) and (ii) aqueous losses, through surface runoff, subsurface runoff and leaching. In addition, there may losses of solid (particulate) N through erosion and runoff.

The P losses to the environment are surface runoff and leaching of dissolved P and the erosion and surface runoff of solid (particulate) P. The leaching losses of P are determined by the retention of the P surplus by the soil.

Factors affecting NH_3 volatilisation after manure application are soil water content, air temperature, wind speed, slurry properties (pH, dry matter content and content of total ammoniacal nitrogen), application method, time and rate, slurry incorporation and measuring technique.

For N surface runoff is of more importance than the erosion of N, but depending on land use and fertilization practices. For P erosion of soil P is a more important loss process than runoff of dissolved P. Indicators for the risk of N and P surface runoff are: heavy precipitation, soil with low infiltration rate, high amount of fertilizer, type

of vegetation, soil tillage and steep slopes. Indicators for the risk of erosion of N and P are: rainfall, the geomorphology, soil texture, land cover, land management, thickness of the soil and the permeability of the soil.

The eventual downward leaching of N of the remaining N, i.e. after volatilization, runoff, plant uptake and (im)mobilisation, is strongly determined by denitrification. Most important factors controlling denitrification are (i) the presence of an energy source for the denitrifying bacteria, mostly available organic carbon, (ii) anoxic conditions, and (iii) the nitrate content in the soil. If any of these conditions is not fulfilled, denitrification is unlikely.

The eventual downward leaching of P is strongly determined by the retention of the P surplus by the soil, that is controlled by the sorption capacity. The sorption capacity of soils for P is determined by the characteristics of the soil, i.e., clay mineralogy, Al and Fe hydroxides, organic matter, pH, oxidation-reduction status, ground water level.

1 Introduction

The European Commission, Directorate-General Environment has contracted Alterra, Wageningen UR for the Service contract *Recommendations for establishing Action Programmes under Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources*.

In 2001, the Commission has carried out a study aimed at assessing Action Programmes established at that time in the EU-15 and aimed at enunciating main principles which should guide the establishment of measures, taking into account agronomic concepts, pedo-climatic zones and local conditions. The EU has in the meantime been enlarged, meaning that new states with different climatologic, environmental and agronomic characteristics have joined. Also, following discussions with Member States, the Commission identified the need for a reference document containing a strong and detailed scientific base for the establishment of action programmes. These needs will be addressed in this service contract.

1.1 Problem statement and Aim of the study

The general objective of the study is to build further on the ERM-2001 study (Rodhe et al., 2006) and covering the EU-27, while taking into account the most recent relevant scientific and technical data on nitrogen emissions, agricultural practices and environmental and climatic conditions. The ERM-2001 study aimed at assessing action programmes established at that time (EU-15) and aimed at enunciating main principles which should guide the establishment of measures, taking into account agronomic concepts, pedo-climatic zones and local conditions. The specific objectives of the study are: divided in three main task:

1. Review and further differentiate the pedo-climatic zones in Europe: definition of pedo-climatic (sub) zones.
2. A detailed analysis of the link between farming practices and the risks for leaching/runoff towards waters and eutrophication processes, with two subtasks:
 - 2.1. Farming practices in relation to water pollution risks
 - 2.2. Detailed analysis of the processes in the soil (surface and subsoil) that influence nutrient leaching and runoff that could lead to pollution of waters and eutrophication processes
3. Recommendations for all measures to be included in the action programmes, differentiated for each pedo-climatic (sub) zone.

The outcome of this study should provide useful inputs for a better implementation of the Nitrates Directive across the whole EU-27.

The study consists of four parts:

Part A: Review and further differentiation of the pedo-climatic zones in Europe: definition of pedo-climatic (sub) zones

Part B: Detailed analysis of the link between farming practices and the risks for leaching/run off towards waters and eutrophication processes

Part C: Detailed analysis of the processes in the soil (surface and subsoil) that influence nutrient leaching and run off that could lead to pollution of waters and eutrophication processes

Part D: Recommendations for the measures referred to in Annex II and Annex III of the Nitrates Directive, differentiated for each pedo-climatic (sub) zone.

This report is part C and presents the results of Subtask 2.2, which comprises an overview of existing literature on the relation between soil properties, nutrient losses, eutrophication and pedo-climatic zones. The focus is on the nutrients nitrogen (N) and phosphorus (P), with an emphasis on N and addressing the losses to groundwater, surface waters and atmosphere. The results are applicable to the farming systems within the EU-27. However, research results from outside the EU-27 was also used when relevant and related to pedo-climatic (sub) zones that are comparable with those within the EU-27.

1.2 Overview of this report

The general approach of this desk study is outlined in Chapter 2. Chapter 3 provides a global overview of the relations between carbon (C), N and P in farming systems. In Chapter 4 the relation between soil properties and hydrological characteristics and the runoff and groundwater recharge is described. Chapter 5 gives an overview of the properties and processes determining N and P losses. The principal part of this report is described in the Chapters 6 to 8 where the relations between pedo-climatic subzones and NH₃ volatilisation (Chapter 6), N leaching (Chapter 7), P leaching (Chapter 8) and Eutrophication of surface waters (Chapter 9) are considered.

2 Methodology

Our general approach of the analysis between soil characteristics and soil processes and risks of water pollution by nutrients was:

- A detailed analysis and scrutiny of the existing literature related to the relationships between soil type and characteristics and the risk for leaching/runoff to groundwater and surface waters and eutrophication. Soil is perceived here as the three-dimensional continuum of the soil surface and slope, the upper soil layer where plants absorb water and nutrients from the soils, as well as the subsoil including the groundwater aquifers. Soil characteristics include both the components of the soil (texture, soil organic matter content, clay mineralogy, iron and aluminium contents, nutrient contents), as well as the soil physical, chemical and biological characteristics. Special emphasis was given to the results of recent and current EU-funded studies.
- A conceptual cause – effect schematization of soil type characteristics and processes that affect the vulnerability to nitrogen and phosphorus leaching. The purpose of this schematization is to describe “the nitrate / phosphorus leaching / runoff vulnerability spaces” across the EU-27 in a coherent and qualitative ‘cause-effect’ manner, and to identify the shape of the relationships, and algorithms.
- A detailed analysis of the links between soil processes, related to the transformation and transport of nitrogen and phosphorus in soils, and the risk for leaching/runoff to groundwater and surface waters, for each of the identified pedo-climatic (sub)zones in Part A. Special attention will be given also to the manure type (slurry, solid manure, TAN content, C/N ratio, mineralization rate) and fertilizer type (urea, AN, UAN, etc.)
- Discussion of the conceptual scheme and the draft report with the project team and with the Commission. Based on these reviews, the final report was completed.

The analyses were largely based on literature and expert knowledge. The results of this particular part of the study were used as input for Part D.

3 Relations between C, N and P in farming systems

In this chapter a brief overview will be given on the linkages between C, N and P in farming systems. Linkages between C, N and P are manifold, but start with photosynthesis, with the production of biomass.

Aspects addressed in this chapter are:

- General overview of the functions of C, N and P in the soil-plant-water system.
- The interaction between C, N and P and the mineralization immobilization turnover (MIT).
- Anthropogenic changes of N and P inputs
- Nutrient limitation.

3.1 General overview of C, N and P in the soil plant water system

The most important biogeochemical element cycles include those of carbon (C), nitrogen (N), oxygen (O), hydrogen (H), phosphorus (P), and sulphur (S). These cycles are intimately linked also with the water cycle. The availability, circulation and interaction of C, N, P, S, O and H in nature are essential for life on earth and ecosystems. Humans on earth have modified the distributions and circulation of these elements tremendously (Bolin et al., 1981).

The main features of especially the biogeochemical cycle of C have been investigated and described thoroughly during the last decades, especially also because of the importance of the C cycle for climate change. The emphasis on the N cycle and to a lesser extent to the P cycle is growing, but the research is mainly related to agricultural production and eutrophication. In addition, N is also related to climate change (N_2O) and human health (NH_3 , NO_x and fine particles). Evidently the human impacts are drastically changing the speed, intensity, and balance of these cycles.

3.1.1 C cycle in the soil plant water system

About 50% of the dry weight of most living organisms is carbon. It plays an important role in the structure, biochemistry, and nutrition of all living cells. Photosynthesis enables plants to transform solar energy and CO_2 into sugars, starches and other forms of organic matter, being the bases for the food chain in both natural ecosystems and farming systems (Figure 1). In this cycle organic C is entering the soil through litterfall, root decay, debris or crop residues, where it becomes a part of the soil organic matter pool (SOM). Through decomposition and mineralization this will be partly respired back to the atmosphere via decomposer organisms and partly incorporated into stable soil organic matter. In addition also dissolved organic carbon (DOC) and inorganic carbon (HCO_3) will be produced which may leach to groundwater and surface waters.

Since the photosynthesis capacity is directly related to nutrient uptake, among which N and P are the most important, the C cycle can be considered as a principal driver of the N and P cycle.

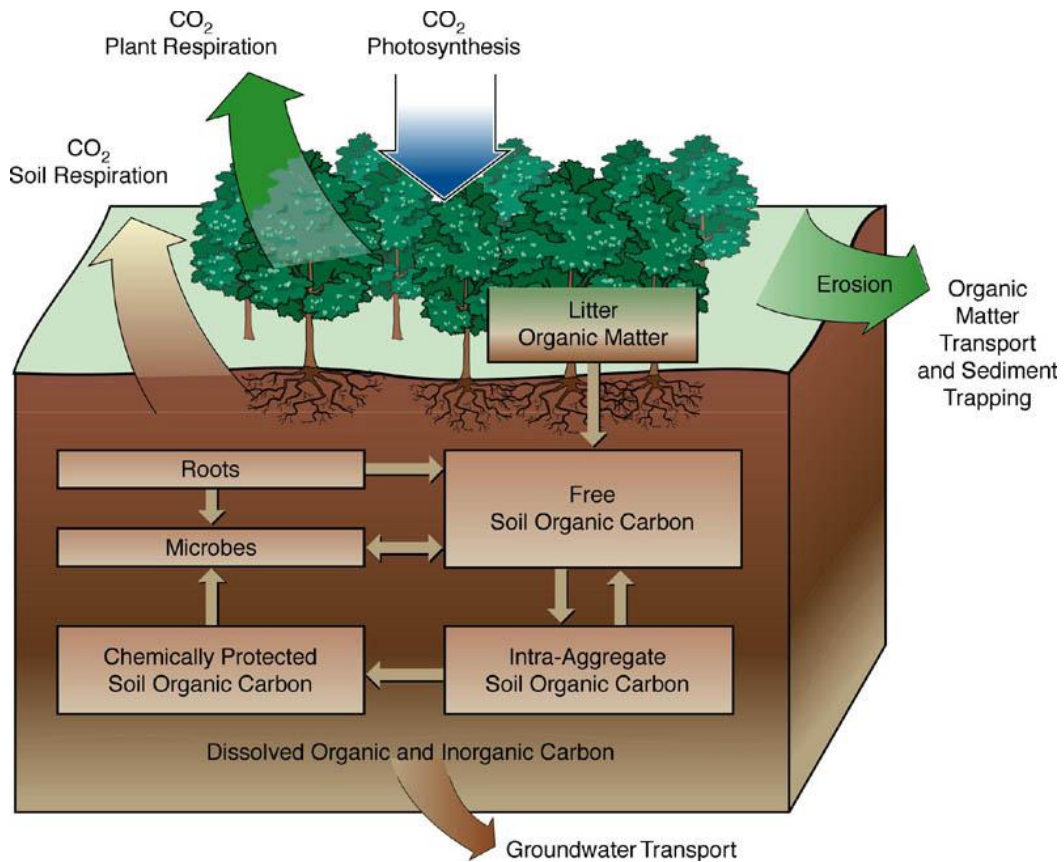


Figure 1 Soil-plant cycle C cycle

(Source: <http://www.climatechange.gov/Library/stratplan2003/final/ccspstratplan2003-chap7.htm>).

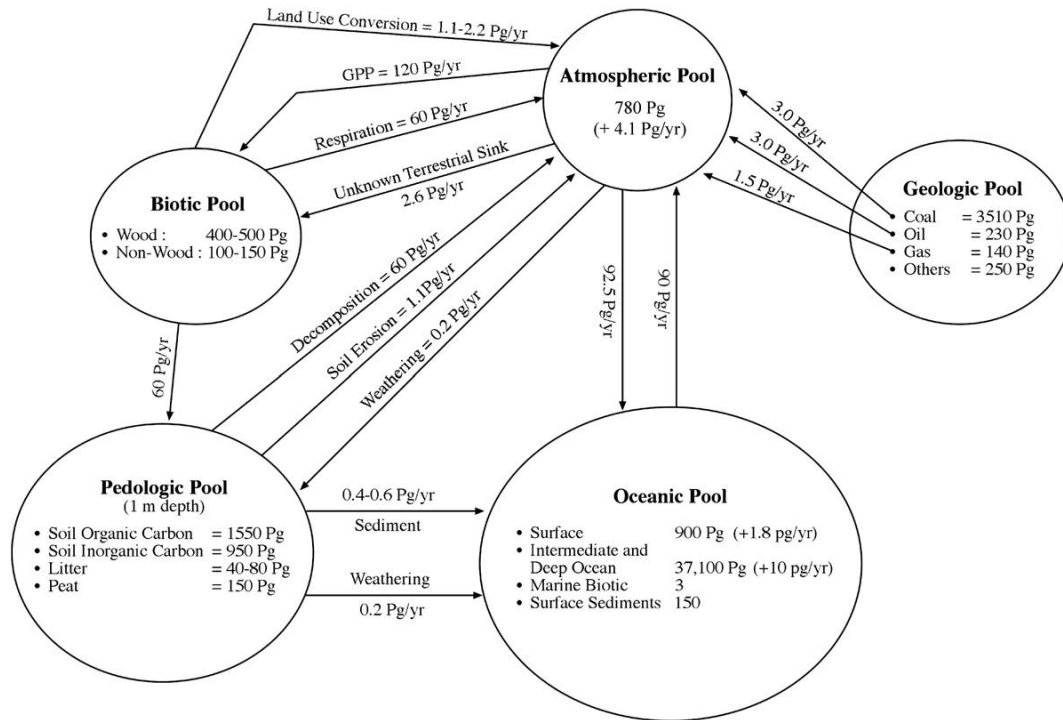


Figure 2 Estimates of the global pools and fluxes between them (Source: Lal, 2008).

Globally, living biomass holds ~ 550 Pg of carbon, most of which is wood. Soils hold ~ 1550 Pg, mostly in the form of organic carbon, with $\sim 35\%$ of the total soil pool as inorganic forms of carbon such as calcium carbonate Lal (2008), see Figure 1). Carbon storage in the biosphere is influenced by a number of processes on different time-scales: while net primary productivity follows a diurnal and seasonal cycle, carbon can be stored for several hundreds of years in trees and up to thousands of years in soils. Human-induced changes in those long-term carbon pools, e.g. through deforestation or afforestation or through temperature-related changes in soil respiration, may thus affect global climate change but also the mobilisation of nutrients such as N and P (see section 3.2).

3.1.2 N cycle in the soil plant water system

The vast majority, i.e. $\sim 80\%$ of N is found in the atmosphere as N_2 . Sedimentary rocks contain roughly the remaining 20%. Just a very small amount, $<1\%$ is found in oceans and in living and dead organic matter (Galloway et al., 2003).

All N in the biosphere (i.e. all living organisms), soils, sediments and waters originally stems from atmospheric N_2 . This N_2 has to be converted to reactive N (N_r), i.e. biologically active N, before it enters the biosphere.. The N cycle is closely related and linked with the C cycle because N is essential for the formation of proteins, an important constituent of living biomass. As for C, the atmosphere plays a major role in the N cycle, through gaseous compounds like N_2 , N_2O , NO and NH_3 (see Figure 3). Nitrogen is most often limiting plant growth in both natural and agricultural

systems and the rate of internal nitrogen cycling in soil is a crucial factor for plant production (see section 3.3). In this cycle organic N is transformed by mineralization, ammonification, and nitrification, resulting in inorganic N as either NH_4 or NO_3 , these forms are prone to losses through (1) denitrification, the reduction of nitrate (NO_3^-) to nitrite (NO_2^-), nitric oxide (NO), nitrous oxide (N_2O) and N_2 , (2) leaching of NO_3^- , NH_4^+ and dissolved organic nitrogen; (3) ammonia volatilization; and (4) soil erosion (Brady, 1990; Bouwman et al., 2009).

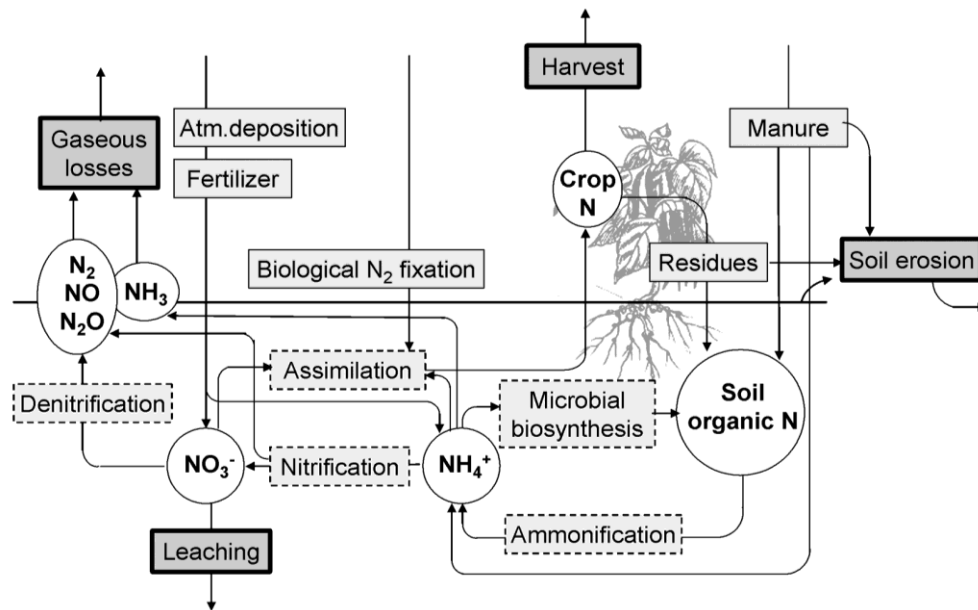


Figure 3 Nitrogen cycle in soil-plant systems. Circles indicate pools, boxes with dashed lines are processes, light-grey boxes with solid lines are inputs, and dark-grey boxes with bold lines represent outputs (Source: Bouwman et al., 2009).

In natural ecosystems these N losses are compensated through biological N_2 fixation (BNF) by leguminous plants living in symbiosis with *Rhizobium* and other N_2 fixing bacteria, or free-living bacteria, as well as through nitrogen deposition. A number of factors limit symbiotic N_2 fixation in ecosystems (Vitousek et al., 2002). Leguminous species are more widespread in the tropics than in temperate climates (Cleveland et al., 1999). The demand of leguminous species and free-living N_2 fixers for P and other elements like molybdenum exceeds that of non N_2 fixers; in many ecosystems with low availability of P, biological N_2 fixation may be constrained, leading to N limitation (Vitousek et al., 2002).

In farming systems N_2 fixing leguminous crops (pulses, soybeans) may play a role. However, most N inputs in farming systems occur via synthetic N fertilizers. Nitrogen withdrawal from these systems through harvesting, but also through N loss processes as denitrification, leaching, volatilization and soil erosion thus have to be compensated mainly by applications of synthetic N fertilizers (and/or animal manure).

In general, only about half of these N inputs are taken up by the crops (Smil, 1999) while the remainder is lost to the environment. Although in experimental fields nitrogen-use-efficiency (NUE), may be much higher. Here, we defined NUE as kg harvest product per kg N applied, irrespective of the type of N that is applied. There are, however, various definitions on NUE for crop production used see Dobermann (2004) for an overview. Under common agricultural practices it appear difficult to match precisely the N supply from fertilizer and from soil organic matter mineralization with the dynamics of crop N uptake demand (Dobermann & Cassman, 2005). Especially under increasing nitrogen inputs, the NUE tends to decrease.

3.1.3 P cycle in the biosphere

Contrary to N, P is almost absent in the atmosphere. P occurs in small quantities in the earth's lithosphere, biosphere and hydrosphere. In terms of mass, P ranks at the 11th place in the lithosphere, and 13th in seawater (Smil, 2000). The Earth's biomass contains small amounts of P. In tree polymers, that make up most of woody phytomass, P is absent in cellulose, hemicellulose and lignin. It is also absent in N-rich amino acids that make up proteins of living organisms (Smil, 2000). Despite its scarcity, P is essential for formation of carbohydrate polymers, proteins and nucleic acids (Schröder et al., 2011).

Unlike natural C and N cycles, which are driven by microorganisms and plants, and have an important atmospheric component, there is only a very small atmospheric reservoir of P (Smil, 2000). On a time scale of thousands of years, the natural P cycle appears to be a one-way flow, with an important role of living organisms. The weathering of parent material, which generally contains small amounts of P, may release up to 1 kg of P per ha per year. This P may be taken up by plants and take part in the biogeochemical P cycle. Ultimately, through erosion and runoff (Figure 4) it is transferred as soluble and particulate P to the ocean where it is eventually buried in sediments (Mackenzie et al., 2002). Given the low solubility of phosphates in soils, leaching of P generally occurs at low rates, apart from P-saturated soils in countries with intensive agriculture (Smil, 2000; Schröder et al., 2011). In soils, the cycling of organic P has rapid turnover times, and is driven by decomposition, mineralization and assimilation by autotrophic production.

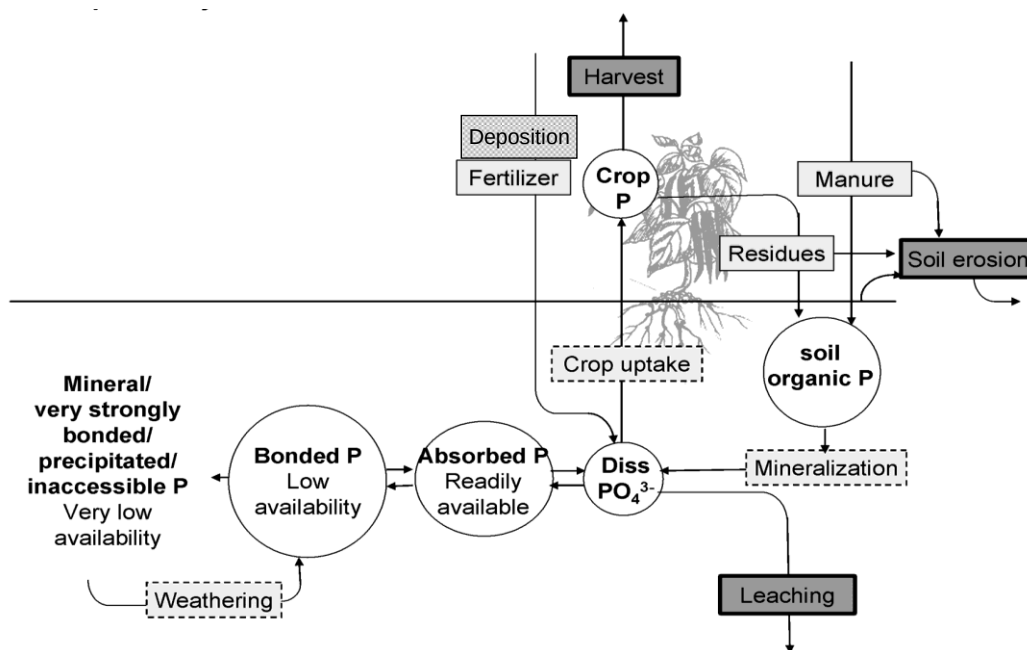


Figure 4 Phosphorus cycle in soil-plant systems. Circles indicate pools, boxes with dashed lines are processes, light-grey boxes with solid lines are inputs, and dark-grey boxes with bold lines represent outputs (Source: Bouwman et al., 2009).

The cycling of P is efficient in natural ecosystems. Contrary to N, there is little or no biotic mobilization of P. P that is lost from the soil-plant cycling is replaced by the slow process of rock weathering and a small amount from deposition (Figure 4). P in rocks is present in poorly soluble forms. Apatite, a calcium phosphate mineral, contains 95% of all P in the Earth's crust. In soils, soluble P released by weathering is usually rapidly immobilized into insoluble forms (Brady, 1990). Precipitation of phosphates with alumina occurs at low pH, and with calcium in calcareous soils. As a result, only a miniscule fraction of P in soils is directly available to plants as dissolved phosphate (PO_4). The major impact of humans on the phosphorus cycle is through the use of P fertilizers (see section 3.5).

The two main factors controlling the availability of P to plant roots are the concentration of phosphate ions in the soil solution and the P-buffer capacity, i.e., the ability of the soil to replenish these ions when plant roots remove them (Koopmans et al., 2004b; Syers et al., 2008). Root length and diameter determine the rate and extent of P uptake. In soils, inorganic P can become absorbed by diffusive penetration into soil components. This may result in a reversible transfer of P between plant-available and non-available forms. P is retained in soil components with a variety of binding types with varying degrees of reversibility. These pools can be related to the availability of P to plants. Soils rich in soluble iron or aluminium oxides, clay minerals like kaolinite, or with a high calcium activity, react with P to form insoluble compounds inaccessible to plant roots (Brady, 1990). This is often referred to as P fixation.

Since the readily available pool provides most of the plant-available P, it is necessary to maintain a certain critical amount of P in this pool in farming systems to obtain good crop yields (Syers et al., 2008). Fertilizer P recovery in crops is often only 10–20% within one growing season. Hence, only a small part of the P added to soil in fertilizer and manure is used by the plant in the year of application. The remainder (corrected for some possible losses) accumulates in the soil as “residual P”. This reserve can contribute to P in soil solution and be taken up by crops during the next years. Where the amount of readily available P is below the critical level, the rate of P release from residual P may not be sufficiently rapid to sustain optimal crop yields. While building up the soil P status to the critical value the P availability is sufficient. Alternatively, a higher P status due to a persistent surplus of P fertilisation leads to P leaching. P is present in the readily available pools and annual P inputs equal to the plant P uptake may be adequate to maintain good crop yields (Syers et al., 2008).

3.2 The interaction between C, N and P

There is a strong linkage between the cycles of N, P and C in farming systems. Both N and P are an essential element for plant growth, being a component of chlorophyll, amino acids, proteins and enzymes (see e.g. Sommer & Olesen, 1991)) Sufficient supply of N and P is required for plant metabolism, and addition of N and P will essentially increase the efficiency of photosynthesis to produce carbohydrates. An overview of the linkage between C, N and P in managing ecosystem is given in 5.

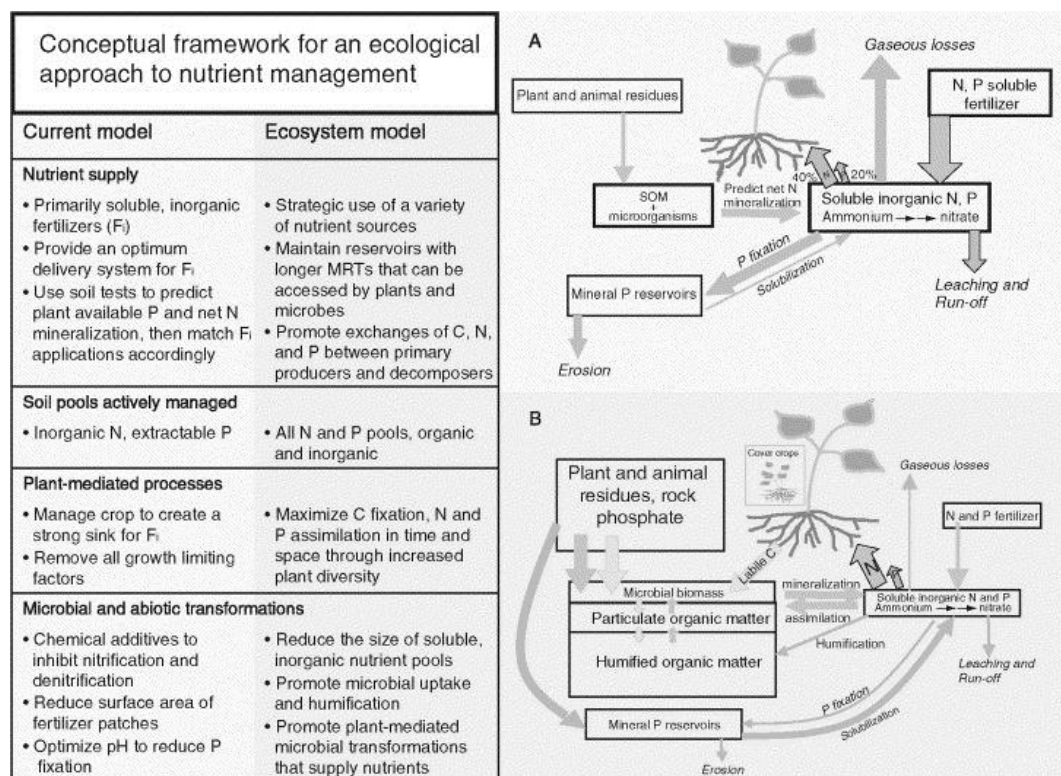


Figure 5 The (re)coupling of C, N, and P in managing ecosystem (Source Drinkwater and Snapp (1991)).

3.3 Nutrient limitation

Nitrogen (N) is one of the single most common nutrients that limits agricultural productivity. Phosphorus (P) is also important for plant growth and development, and is often called the second most limiting nutrient. Next, potassium (K) and other essential nutrients are needed and may limit productivity. In practice, growth and yields of plants are frequently limited by shortage of nutrients and/or water. As the two most important nutrients limiting biological production, N and P are the most extensively applied nutrients in farming systems, mainly as soluble inorganic fertilizers. Global N and P fluxes are projected to increase substantially as the demand for food and bio-energy increases and also countries with less intensive agriculture increase fertilizer use (Tilman, 1999; Galloway et al., 2004; Sutton et al., 2011b).

3.4 Natural sources of N and P

The natural inputs of nitrogen and phosphorus into surface waters are usually small compared to other diffuse and point sources. Regionally, however, the mineralization of soil organic matter from sediments and peat soils may lead to a substantial background loss of N. Examples of such soils may be found in delta's and low lands such as in the Po-delta and the Netherlands, but also certain areas in northern Scandinavia or the Alps. Sediments rich in N are shale, which may contain up to 1900 mg N per kg while limestone is almost depleted with N at concentrations between 4 and 200 mg N per kg (Salminen, 2005).

The natural input of phosphorus can be relatively significant compared to other sources. It depends on the geological conditions and may differ even over short distances. In areas dominated by marine sediments, there may be naturally high phosphorus concentrations in surface water and groundwater.

Volcanic sediments may contain considerable amounts of P; basalt for instance may contain up to 1100 mg P per kg. Under certain conditions specific form of P may easily dissolve in water. Apatite ($\text{Ca}_5(\text{PO}_4)_3\text{F}$) is remarkably soluble in acidic environments, whereas Al and Fe phosphates are highly insoluble. Sorption of phosphate to clays and Al-Fe hydroxides depends on the soil pH. P-adsorption in organic rich soils occurs in the presence of the metal ions Fe^{2+} , Al^{3+} and Ca^{2+} .

The aqueous geochemistry of P is complicated. Between pH 4 and 6, H_2PO_4^- is the dominant form, under neutral conditions HPO_4^{2-} dominates while PO_4^{3-} is most stable under alkaline conditions. Inorganic polyphosphate and organic P can be dissolved as well, but most P in water is particle bound. Generally, dissolved P concentrations in natural waters are low due to complexes formed with metal ions, which are highly insoluble (De Vos & Tarvainen, 2006).

Salminen (2005) shows a map with the P_2O_5 concentrations in sub soil in different areas in the EU. In areas with a relatively high concentration, mineralization may lead to a relatively high natural input of P. This may occur in areas with young marine sediments (e.g. Northern part of the Netherlands, South of France) but also in other regions (e.g. Central Spain, Southern Austria and Slovenia). Areas with very low P concentrations in the sub soil are found in Denmark, Northern Germany, Poland and Eastern Spain.

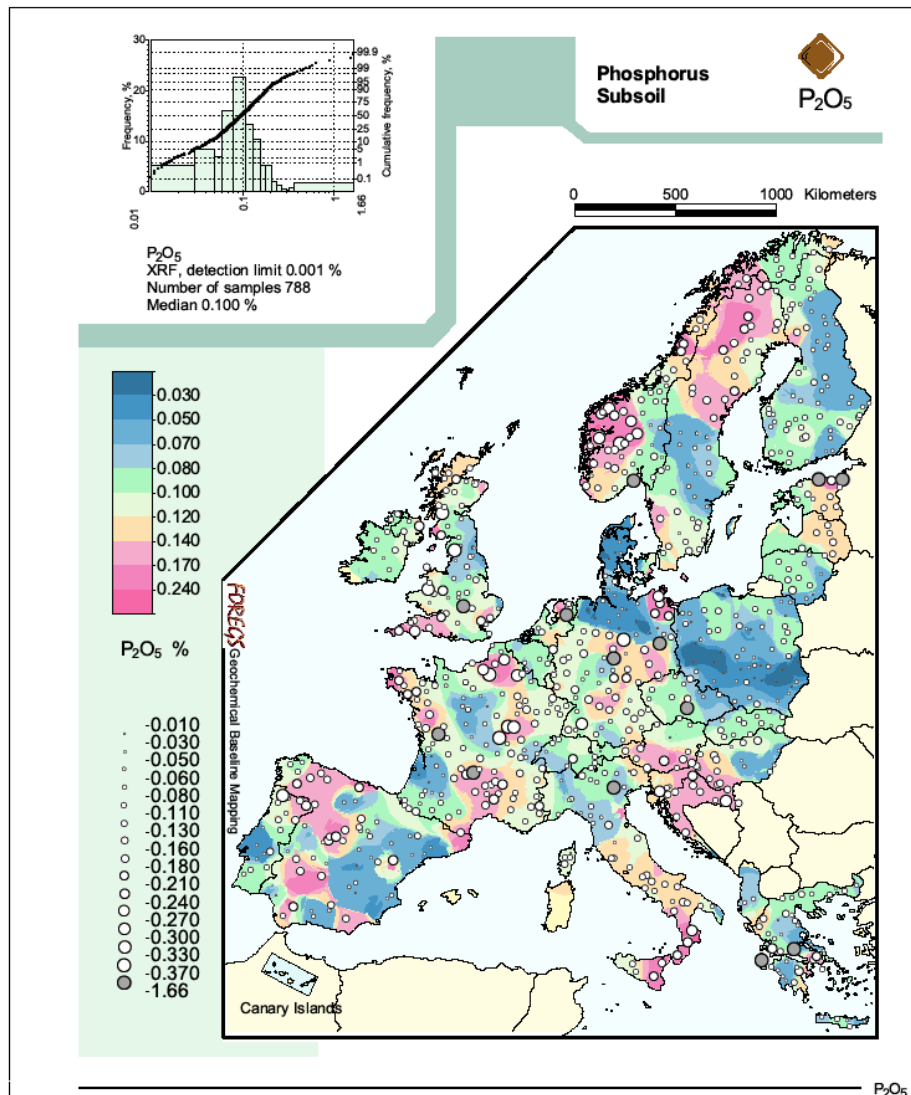


Figure 6 Map showing the natural sub-soil concentration (%) of phosphorus as P_2O_5 (source Salminen, 2005).

3.5 Anthropogenic alteration of N and P

Human behaviour has had a profound influence upon the biogeochemical cycles of C, N and P (Schlesinger, 1997; Vitousek et al., 1997a; Vitousek et al., 1997b).

During the past five decades, global population, food production, and energy consumption have increased with a factor of ~ 2.5 , ~ 3 and ~ 5 , respectively. In the early 1990s, the industrial production of N fertilizers was ~ 100 Tg N per yr, a factor of ~ 10 increase over 1960, about three times as much as the amount of biological fixed N in cultivated soils (Galloway et al., 2004).

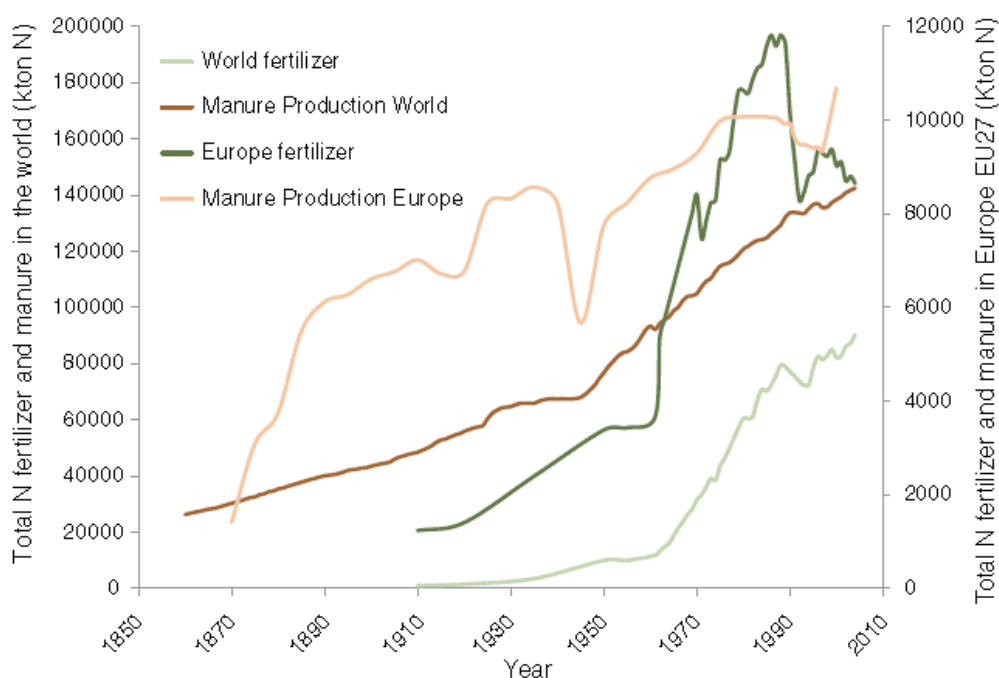


Figure 7 Global and European fertilizer and livestock manure nitrogen consumption (Source: Erisman et al. (2011)).

The global phosphorus (P) cycle has also been altered by human activity. Mining of phosphate rock and subsequent production and use as fertilizer, detergent, animal feed supplement and other technical uses has more than doubled P inputs to the environment over natural, background P from weathering (Mackenzie et al., 2002).

The changes in global nutrient cycles have had both positive and negative effects. The increased use of N and P fertilizers has allowed for producing the food necessary to support the rapidly growing human population (Galloway & Cowling, 2002). However, losses of N and P to the environment have also increased greatly, and have induced N and P pollution of groundwater, surface waters and coastal marine systems. This has resulted in numerous negative human health and environmental impacts such as loss of habitat and biodiversity, an increase in frequency and severity of harmful algal blooms, eutrophication, hypoxia and fish kills (Vollenweider, 1976; Diaz & Rosenberg, 1995; Howarth et al., 1996; Rabalais, 2002). Nitrogen losses comprise also emissions of NH_3 and N_2O to the atmosphere, with impacts on biodiversity and climate change. The relation between N and P losses and

eutrophication is further addressed in Chapter 9. The adverse impacts of atmospheric losses of N on biodiversity and climate change (Sutton et al., 2011b) are not further addressed in this report.

3.5.1 Agriculture

In large parts of Europe, agriculture is a dominating anthropogenic source of pollution by nitrogen and phosphorus (see section 3.5). The estimates of agricultural diffuse losses range from 0 to 30 kg/ha for N and from 0 to 1 kg/ha for P. The highest loss is found in agriculturally intensive regions in the north-western part of Europe, where the average (mineral) fertiliser consumption per country is commonly about 40–70 kg/ha of N and 8–13 kg/ha of P (FAO). The risk of eutrophication increases with the imbalance of the amounts of N and P applied and the amounts removed with the harvested crops.

Figure 8 and Figure 9 show the total N and P application in the EU, respectively, indicating the highest risk of losses in the agriculturally intensive regions.

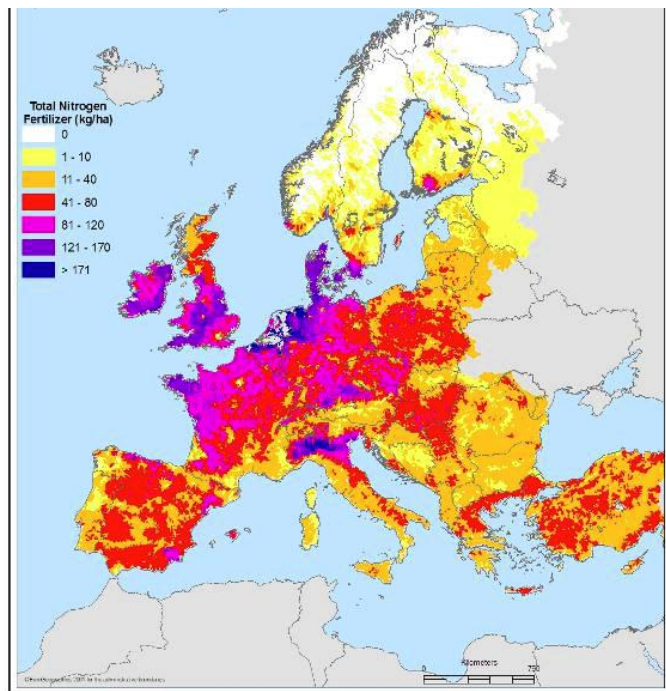


Figure 8 Total nitrogen application for the year 2005 in the EU (EC, 2010).

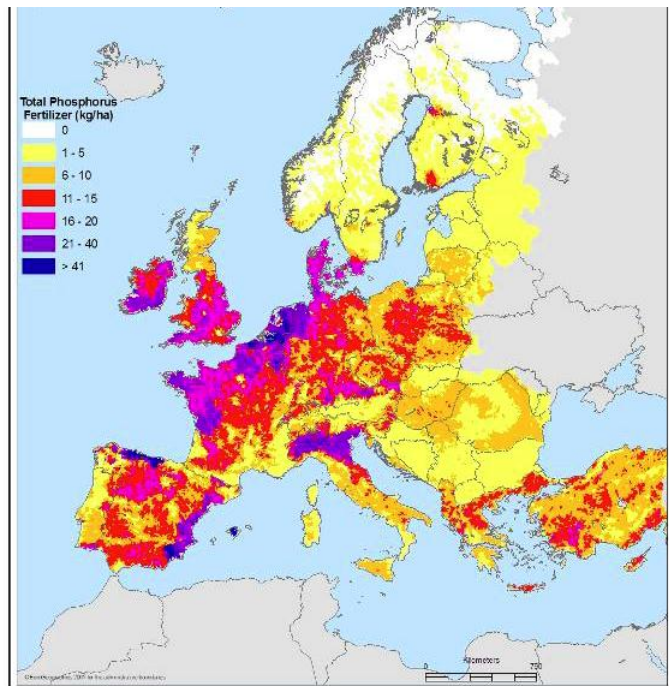


Figure 9 Total phosphorus application for the year 2005 in the EU (EC, 2010).

3.5.2 Atmospheric deposition

Both nitrogen and phosphorus are deposited in water and soil in different forms: nitrogen as ammonia, emitted from animal manure, and as NO_x coming from combustion of fossil fuels, i.e. power plants and transportation; phosphorus as dust, falling leaves and bird faeces.

The annual total N deposition ranges from 1 to 10 kg ha⁻¹ (Søgaard et al., 2002), being highest in the centre of western Europe around the Netherlands, where it may even be above 20 kg ha⁻¹ and lowest in northern Scandinavia, where it may even be below 1 kg ha⁻¹.

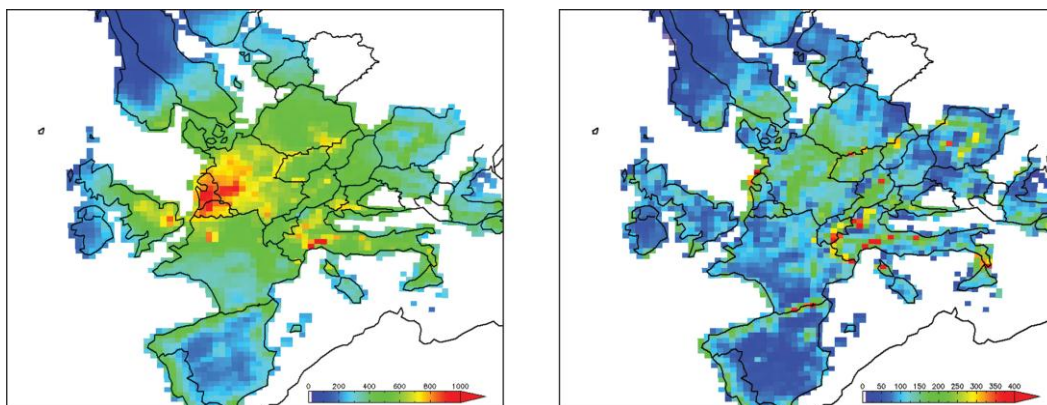


Figure 10 Calculated total N deposition (mg m^{-2} or 0.01 kg ha^{-1}) from an ensemble of seven models for 2001 (left), together with the standard deviation (right) (Source: Søgaard et al., 2002).

The deposition of phosphorus is generally small and difficult to estimate. Based on an extensive global literature research, Newman (1995) estimated an atmospheric input that ranges from $0.07\text{--}1.7 \text{ kg P ha}^{-1} \text{ yr}^{-1}$.

3.5.3 Rural population

About 80 % of the population is connected to waste water treatment in Northern and Southern European countries. The connection rate in Central European countries is even higher, at 90 %. On the basis of data reported in 2006–2007, about 65 % of total population is connected to wastewater treatment in the countries of Eastern Europe. Average connection in South-Eastern Europe (Turkey, Bulgaria and Romania) is about 40 % (Svensson, 1994). In sparsely populated countries with a relative high proportion of the population living in scattered dwellings, these are not connected to collecting systems and normally are served by individual waste water treatment (e.g. septic tanks or a percolation system), but the wastewater may also be discharged directly to surface water (Thorman et al., 2008). At large scale (OSPAR and Helcom data), the annual discharges from scattered dwellings are typically $0.1\text{--}0.5 \text{ kg/ha}$ for nitrogen and $0.01\text{--}0.1 \text{ kg/ha}$ for phosphorus, and occasionally higher.

3.5.4 Point sources

In Europe, nutrient discharges from municipal wastewater treatment plants are in general higher than for any other point source. Results from large inland and marine catchments show that municipal wastewater constitutes about 75% of the point source discharges of both nitrogen and phosphorus. Industrial sources constitute about 17% and other point sources are also relatively insignificant. Locally, in smaller catchments, all types of point sources may be significant in relation to pollution management (Grizzetti & Bouraoui, 2006).

4 Geomorphology, soil properties, and hydrological characteristics in relation to runoff and groundwater recharge

In order to describe N volatilization, leaching and transport of N and P, P retention and denitrification in groundwater, information is needed on soil properties and hydrologic characteristics of the groundwater system. Van Drecht et al. (2003) made a distinction in two subsystems (i) the shallow groundwater with the (partly) unsaturated zone with rapid transport of solutes in surface runoff and flow through shallow groundwater to local water courses (subsurface runoff) and (ii) the deep groundwater saturated zone with slow transport towards larger streams and rivers. Shallow groundwater flow is assumed to occur in the top layer of the (partly) unsaturated zone. Deep groundwater flow occurs in unconsolidated aquifers of ~50 m thickness. The shallow groundwater system is characterised by short residence times before water enters local surface water (small rivers) or deeper groundwater and is described in section 4.2. The deep groundwater system has often a much longer residence times before water enters large rivers and is described in section 4.3. A brief overview of the hydrologic cycle is presented in section 4.1.

4.1 The hydrological cycle

Evapotranspiration of water from land and water is the driver of the hydrological cycle (Figure 11). Evapotranspiration is water transpired from plants and evaporated from the soil. As moist air is raised, it cools and water vapour condenses to form clouds. Moisture is transported around the globe until it returns to the surface as precipitation. Some precipitation falls as snow or hail, sleet, and can accumulate as ice caps and glaciers. Most water falls as rain back into the oceans or onto land, where (i) some of the water may evaporate back into the atmosphere, (ii) directly enters surface waters through surface runoff or (iii) the water may penetrate the surface and become groundwater and ultimately seeps its way to oceans, rivers, and streams, or is released back into the atmosphere through transpiration.

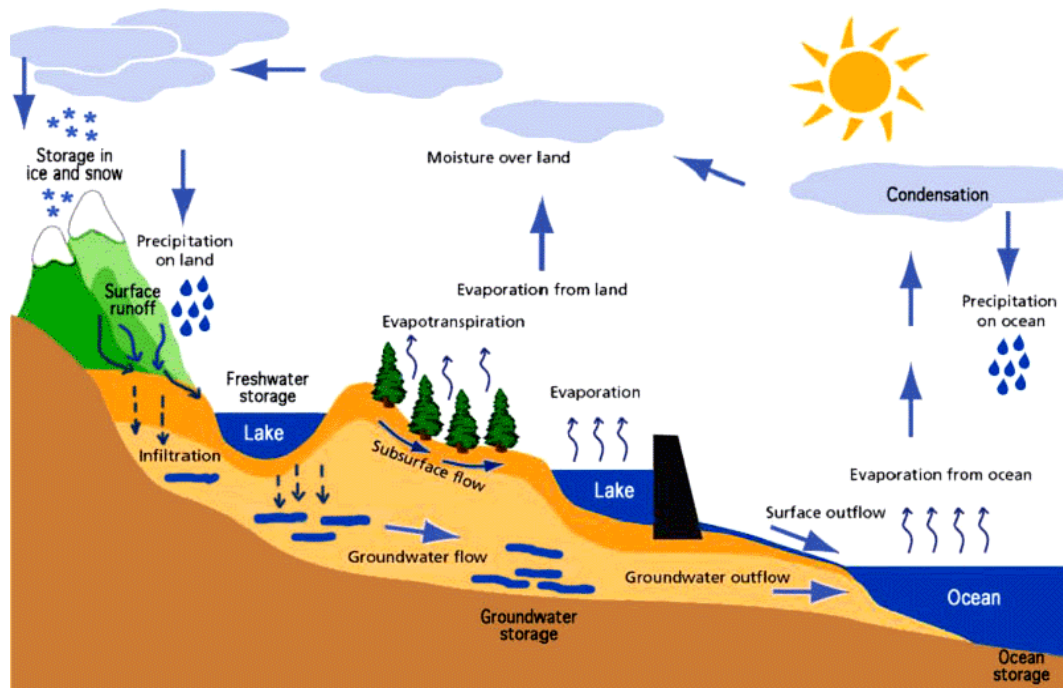


Figure 11 The hydrologic cycle (Source: <http://geofreeke.wordpress.com/the-hydrosphere>).

4.2 Shallow groundwater: geomorphology, soil properties and drainage to small rivers

The important role of the geomorphology is the delivery of water from hill slopes to stream channels. Important processes affecting the water budget are plant-atmosphere interactions, surface runoff, infiltration, flow in the unsaturated-saturated zone and subsurface runoff. Besides the recharge of a regional groundwater system the dynamics of the groundwater level itself also play a crucial role in, especially at high groundwater levels. These dynamics determining the vertical interaction and the drainage flow and the water quality processes in the shallow groundwater. Agro-ecological evaluations of the hydrologic regime of the top-system processes generally require a dynamic hydrological model (see Box 1).

Box 1. Modelling the hydrological processes at the soil surface

The modelling of the hydrological processes at the soil surface is often based on the so-called Richards' equation. An example of a Richards-type model is the soil and groundwater quantity modelling (SWAP) (Van Dam, 2000; Schoumans & Silgram, 2003). In this model (Figure 12) water discharge to groundwater and surface water is schematised by a pseudo-two-dimensional flow in a vertical soil column with unit surface. The ground level provides the upper boundary of the model and the lower boundary is at the hydrological basis of the system defined. The lateral boundary consists of one or more different drainage systems.

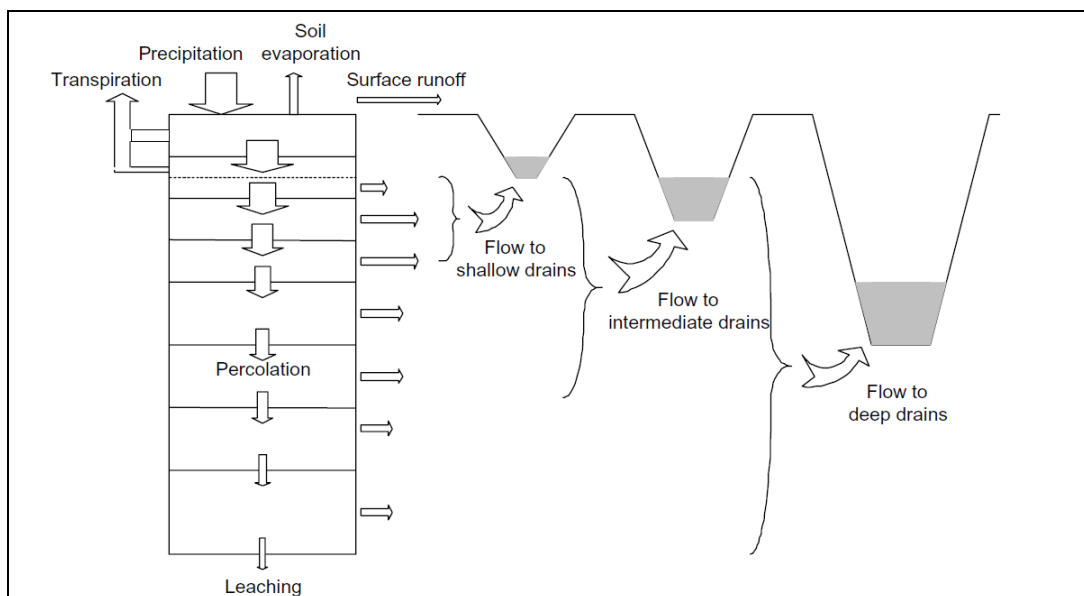


Figure 12 Scheme of water flows in a soil profile and the main terms of the water balance (from (Groenendijk & Boers, 1999))

In regions with high groundwater levels and water discharge towards surface water, residence times are strongly influenced by the size and depth of the drainage system. In non-point water quantity models, the extent of water flows to each of the drainage systems must be calculated by using drainage formulae applicable to the local flow (Groenendijk & Boers, 1999). In the non-point water quality models, regional spatially distributed patterns of soil type, land use and hydrology are schematised by a number of homogeneous subregions. The size of a subregion depends on the heterogeneity of these factors and on the ultimate goal of the model application. The boundary between local and regional flow can be defined as the depth below which no discharge to local surface water occurs. Above this depth, the greater part of the precipitation surplus flows to water courses and other drainage systems. This depth depends on the deepest streamline discharging water to the drainage systems. Once the regional and local flow has been segregated by the position of the boundary surface, the streamline pattern within the top system is schematised into vertical fluxes between soil layers and into lateral fluxes in the saturated zone. Information on water discharges and drainage distances is needed to simulate residence times of water and solute in the saturated zone (Groenendijk & Boers, 1999).

Using the model SWAP (Van Dam, 2000), Tiktak et al. (2006) calculated the mean average precipitation surplus for the EU-15 with mean average precipitation surplus of $\sim 350 \text{ mm yr}^{-1}$ (Table 2). The table shows that the differences between the regions are larger than the differences between the crops. It is obvious that climate zones with high precipitation fluxes generally also have a high precipitation surplus.

Table 1 Major climate zones of the Europe Union, based on mean annual rainfall and mean annual temperature. Classification according to FOCUS (Howarth & Marino, 2006) (Source: De Vos & Tarvainen, 2006).

Climatic zone	Representative location	Mean annual rainfall (mm)	Mean annual temperature (°C)	Surface areaa (km ²)
Cold	Jokioinen (Finland)	<600	<5	83,000
Temperate 1	Châteaudun (France)	<600	5–12.5	36,300
Temperate 2	Hamburg (Germany)	600–800	5–12.5	429,400
Temperate 3	Kremsmünster (Germany)	800–1000	5–12.5	269,000
Temperate 4	Okehampton (UK)	>1000	5–12.5	167,800
Warm 1	Sevilla (Spain)	<800	>12.5	329,800
Warm 2	Piacenza (Italy)	800–1000	>12.5	148,400
Warm 3	Porto (Portugal)	>1000	>12.5	32,600

Table 2 20 Years predicted average water fluxes (mm yr⁻¹) for winter wheat and maize. Water balances are presented for Europe as a whole and for the climate zones as described in Table 1 (source: De Vos & Tarvainen, 2006).

Climatic zone	P	I	E_i	E_t	E_s	Q	$E_{t,pot}$	$E_{s,pot}$
<i>Winter wheat</i>								
Europe	821	0	33	247	200	337	282	554
Temperate 1	650	0	39	211	208	191	248	542
Temperate 2	720	0	25	219	211	264	229	398
Temperate 3	900	0	31	258	239	371	264	426
Temperate 4	1119	0	47	238	228	605	253	410
Warm 1	644	0	36	274	121	210	418	1123
Warm 2	898	0	28	275	196	385	292	528
Warm 3	1069	0	61	330	179	494	354	571
<i>Maize</i>								
Europe	821	117	30	335	224	346	407	492
Temperate 1	650	59	28	275	227	178	409	451
Temperate 2	720	34	32	269	203	249	300	366
Temperate 3	900	10	33	301	233	342	325	407
Temperate 4	1119	13	52	278	219	583	320	365
Warm 1	644	434	10	525	243	298	725	944
Warm 2	898	180	25	384	223	430	457	460
Warm 3	1069	185	16	330	283	618	491	580

P is precipitation, I is irrigation, E_i is interception loss, E_t is transpiration, E_s is soil evaporation, and Q is seepage flux. The suffix *pot* refers to potential.

4.2.1 Role of geomorphology

4.2.1.1 Precipitation

Precipitation volumes collected at different sites within a relatively small area can vary significantly in response to local surface topography, vegetation cover, precipitation type and storm type, and extreme care should be taken in the extrapolation of volumes over larger areas.

The infiltration capacity of the soil depends on its texture and structure, as well as on the soil moisture content before the rainfall started. The initial capacity of a dry soil

may be high but, as the rain continues, it decreases until it reaches a steady state value, the final infiltration rate (see Figure 8).

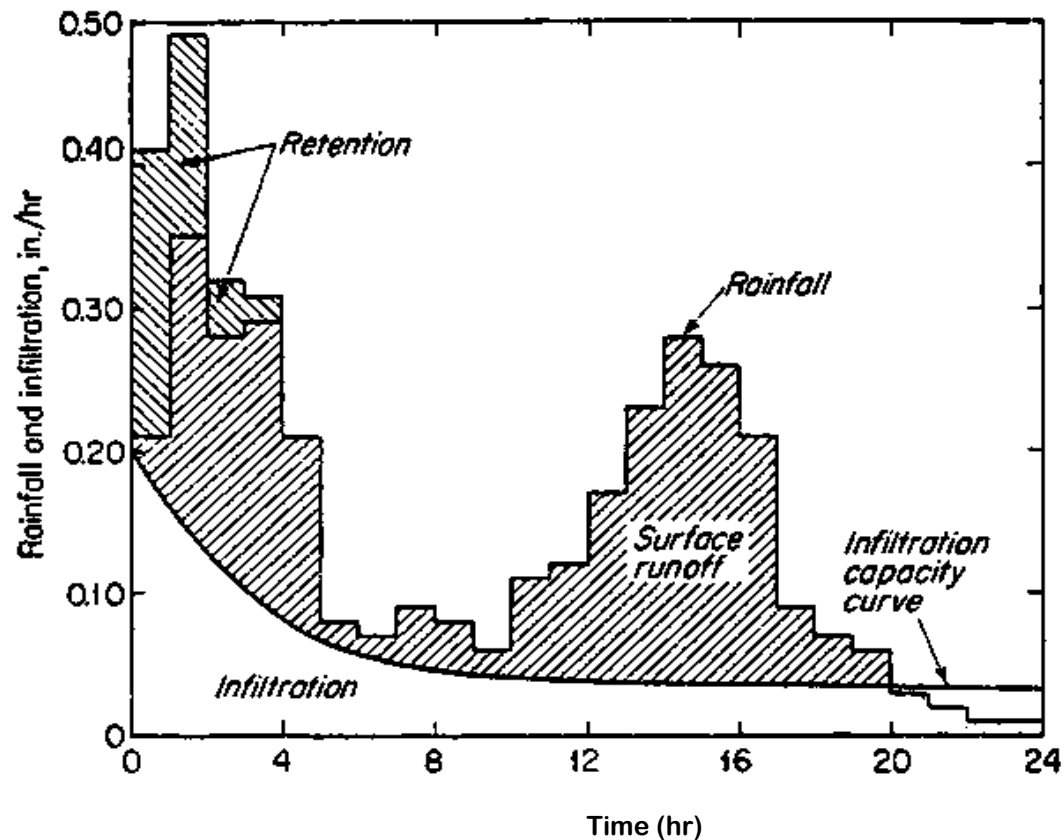


Figure 13 Schematic diagram illustrating relationship between rainfall, infiltration and runoff (Source: Linsley et al., 1958).

4.2.1.2 Surface runoff

The portion of water which does not infiltrate the soil but flows over the surface of the ground to a stream channel is called surface runoff or overland flow. As the rate of rainfall (intensity) exceeds the infiltration capacity of the soil and the retention capacity, runoff will be generated (see Figure 8 and Critchley & Siegert (1978)). Runoff continues as long as the rainfall intensity exceeds the actual infiltration capacity of the soil. In addition the vegetation has a significant effect on the infiltration capacity of the soil. A dense vegetation cover shields the soil from the raindrop impact and reduces the crusting effect. Furthermore, it affected the retention capacity through interception storage, i.e., above ground storage of water mostly in vegetation.

Factors affecting runoff can be distinguished in meteorological factors, biophysical factors and human activities.

Meteorological factors

The main meteorological factors that affect runoff are:

- Type of precipitation (rain, snow, sleet, etc.)
- Rainfall intensity
- Rainfall amount
- Rainfall duration
- Distribution of rainfall over the drainage basin
- Direction of storm movement
- Precipitation that occurred earlier and resulting soil moisture
- Other meteorological and climatic conditions that affect evapotranspiration, such as temperature, wind, relative humidity, and season

Biophysical factors

Runoff is affected by the following biophysical factors:

- Land use
- Vegetation
- Soil type (texture, porosity, cracks, etc.)
- Depth to bedrock
- Type of underlying bedrock (e.g. permeable limestone stone with karst characteristics)
- Drainage area
- Basin shape
- Elevation
- Topography, especially the slope of the land
- Drainage network patterns
- Ponds, lakes, reservoirs, sinks, etc. in the basin, which prevent or delay runoff from continuing downstream

Human activities

Human activities that may affect runoff are the removal of vegetation and soil, grading the land surface, and constructing drainage networks. These activities increase runoff volumes and shorten runoff time into streams from rainfall and snowmelt. Also, soil sealing in urban and infrastructural areas, and soil compaction by heavy machinery decrease the infiltration of water into the soil and thereby surface runoff. As a result, the peak discharge, volume, and frequency of floods increase in nearby streams. The root system as well as organic matter in the soil increase the soil porosity thus allowing more water to infiltrate. Vegetation also retards the surface flow particularly on gentle slopes, giving the water more time to infiltrate and to evaporate.

4.2.1.3 Drainage system

The geomorphology determines the drainage system that is formed by the pattern of streams, rivers, and lakes in a particular drainage basin. They are governed by the topography of the land, whether a particular region is dominated by hard or soft rocks, and the gradient of the land. Streams are often view as being part of drainage basins. A drainage basin is the topographic region from which a stream receives

runoff, throughflow, and groundwater flow. A catchment represents all of the stream tributaries that flow to some location along the stream channel. The number, size, and shape of the drainage basins found in an area vary and the larger the topographic map, the more information on the drainage basin is available.

Steep slopes yield more runoff than gentle slopes. In addition, the quantity of runoff decreases with increasing slope length. This is mainly due to lower flow velocities and subsequently a longer time of concentration (defined as the time needed for a drop of water to reach the outlet of a catchment from the most remote location in the catchment). This means that the water is exposed for a longer duration to infiltration and evaporation before it reaches the measuring point.

4.2.2 Role of soil properties

The infiltration capacity is among others dependent on the porosity of a soil which determines the water storage capacity and affects the resistance of water to flow into deeper layers. Porosity differs from one soil type to the other. The highest infiltration capacities are observed in loose, sandy soils while heavy clay or loamy soils have considerable smaller infiltration capacities (Figure 14).

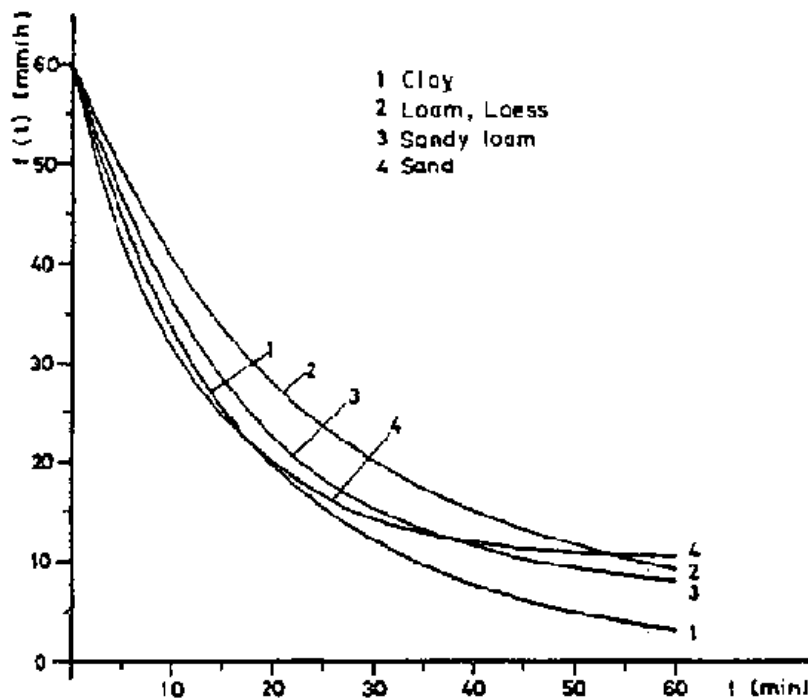


Figure 14 Infiltration capacity curves for different soil types

Furthermore, the infiltration capacity depends on the moisture content prevailing in a soil at the onset of a rainstorm.

4.3 Deep groundwater: Sub soils and drainage to large rivers

An important parameter of the deep groundwater is the residence time. This parameter is important for the prognosis of the long-term behaviour of groundwater systems in response to N and P inputs. The groundwater residence times specify the time scale for measures to remediate polluted groundwater to lead to a substantial groundwater quality improvement. The deep groundwater subsystem (aquifer) is recharged by water flowing from the shallow system. The residence time is determined by the groundwater velocity in the aquifer, that can be expressed by the (saturated) hydraulic conductivity, the permeability for water through the pore spaces or fractures. The hydraulic conductivity in loose and solid rocks follows a lognormal distribution (see Figure 1).

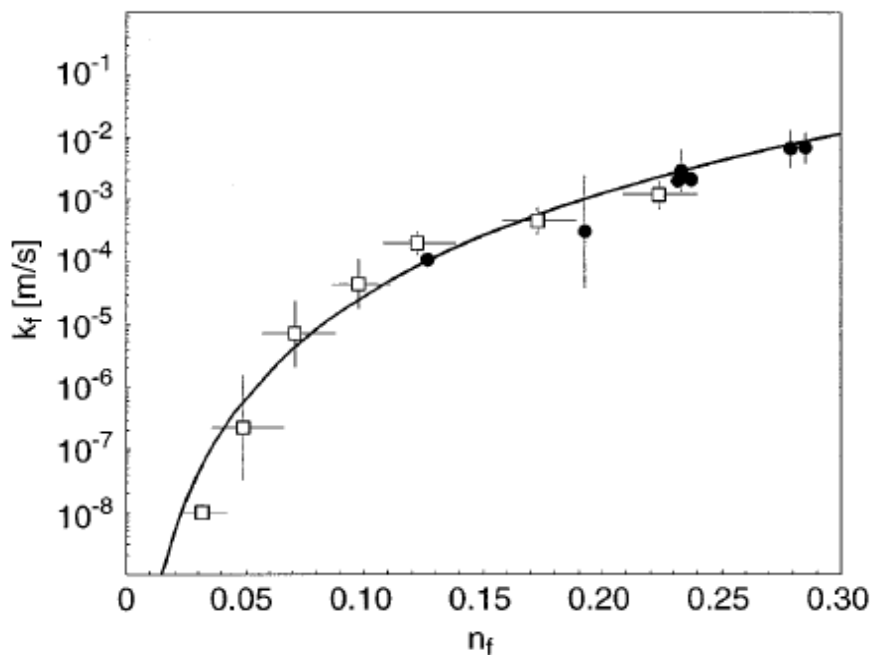


Figure 15 The hydraulic conductivity (k_f) as function of the effective porosity (n_f) for the upper limestone of the river Main catchment area (Source: Kunkel & Wendland, 1997).

Effective porosity can also be derived from other hydro-geological parameters using the lithological map (Dürr et al., 2005). Based on such a map Keuskamp et al (2012) derived a European wide map indicating where substantial deep groundwater flow occurs (Figure 16). In other formations groundwater flow is hampered due to small pore size (compressed loess deposits) or is restricted to fractures.

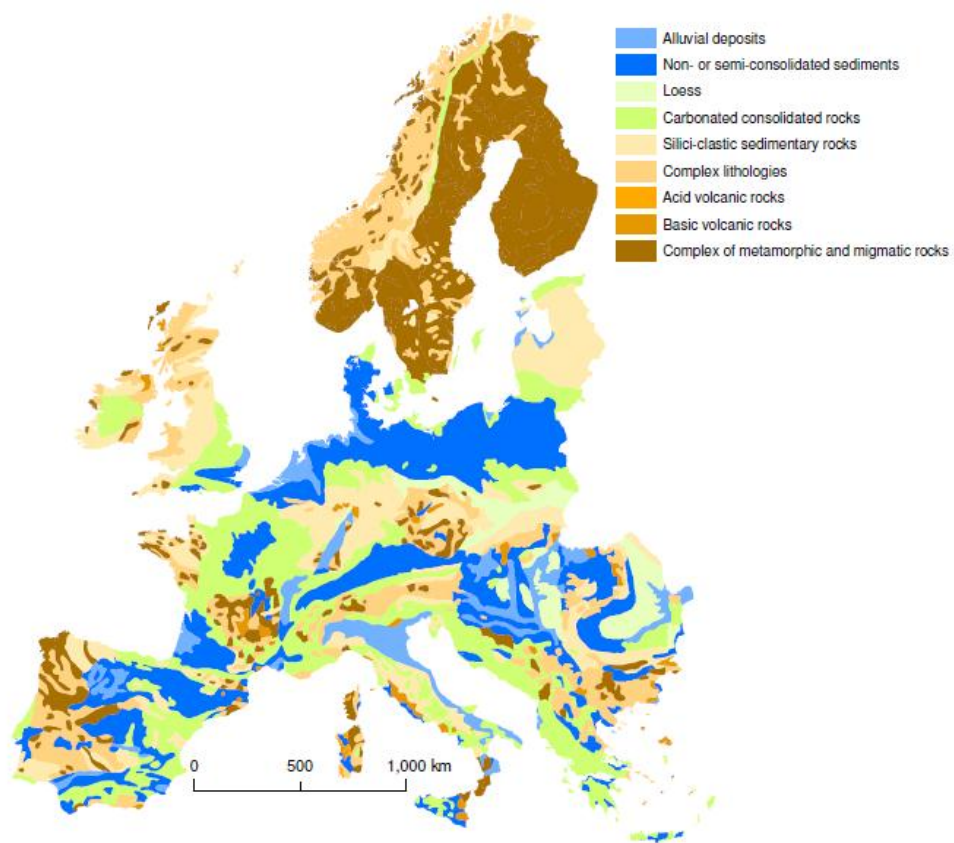


Figure 16 Lithological classes. Blue coloured areas are aquifer classes with a high conductivity and effective porosity (Dürr et al., 2005).

Keuskamp et al (2012) also derived a map with the infiltration fraction, representing the fraction of the precipitation surplus that contributes to groundwater recharge (Figure 17). The infiltration fraction was based on slope, texture and groundwater level.

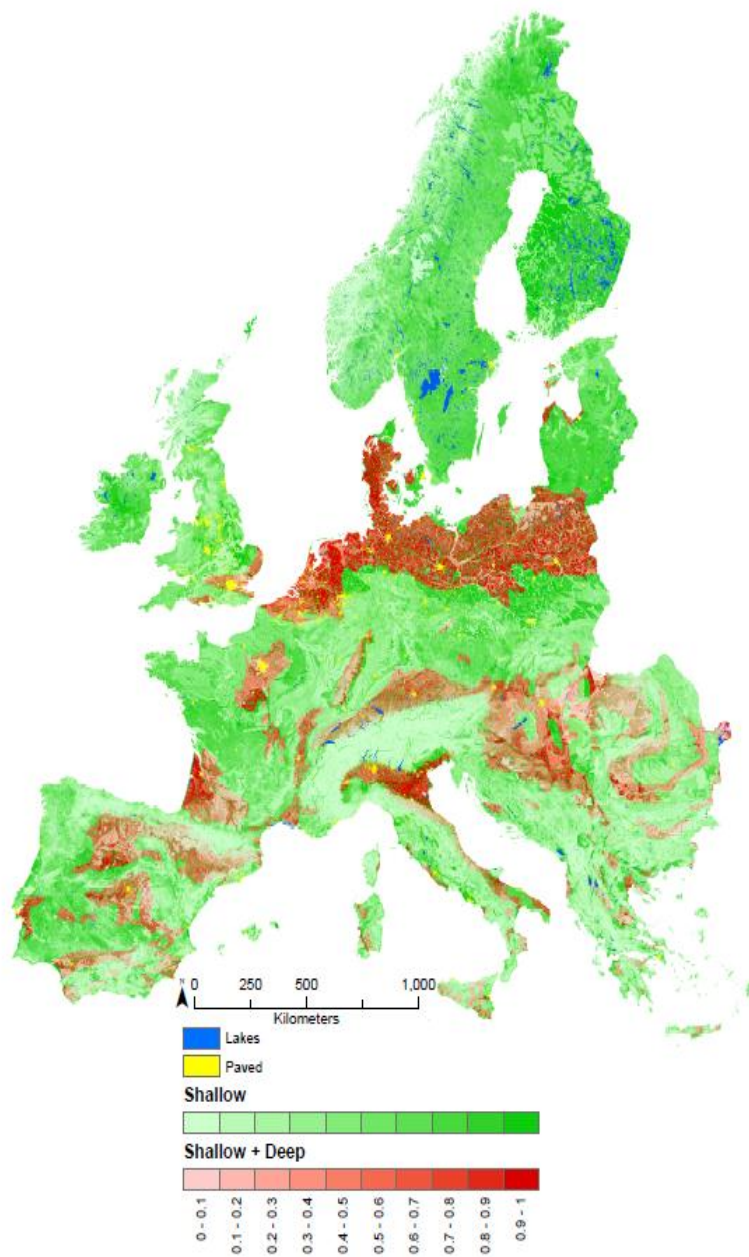


Figure 17 Infiltration fraction (f_{inf}) for areas having only shallow groundwater (green) and for areas with a shallow and a deep groundwater system (red). f_{inf} was based on slope, texture and groundwater level. Deep groundwater occurrence is based on lithology (from: Keuskamp et al., 2012).

4.4 Concluding remarks

For the quantification and characterisation of the N losses through volatilization, denitrification and leaching, P losses through leaching and runoff, and N and P transport and retention in groundwater, information is needed on soil properties and hydrologic characteristics of the groundwater system. Important factors influencing

surface runoff are: precipitation (amount, type intensity and duration), factors affecting the evapotranspiration, land use soil texture and slope. Important factors influencing subsurface runoff are: precipitation surplus, soil texture, effective porosity and groundwater level. In view of N and P processes in groundwater there is a need to distinct in two subsystems (i) the shallow groundwater with the (partly) unsaturated zone with rapid transport of solutes in surface runoff and flow through shallow groundwater to local water courses (subsurface runoff) and (ii) the deep groundwater saturated zone with slow transport towards larger streams and rivers. An important parameter of the deep groundwater is the residence time. This parameter is important for the prognosis of the long-term behaviour of groundwater systems in response to N and P inputs. The groundwater residence times specify the time scale for measures to remediate polluted groundwater to lead to a substantial groundwater quality improvement.

5 Properties and processes determining N and P losses

The N and P surpluses of the soil balance of agricultural soils are an indicator for the N and P losses to the environment to the hydrosphere and in case of N to the atmosphere. The soil surface balance includes all relevant N and P inputs and outputs from the soil. The fate of the N and P surplus is controlled by a combination of factors, including type and rate of N and P inputs, soil type and properties, weather conditions, crop type and the hydrology of the soils (see e.g. Velthof et al., 2011). A schematic representation is given of the relevant transport routes and processes determining N and P losses from farming systems.

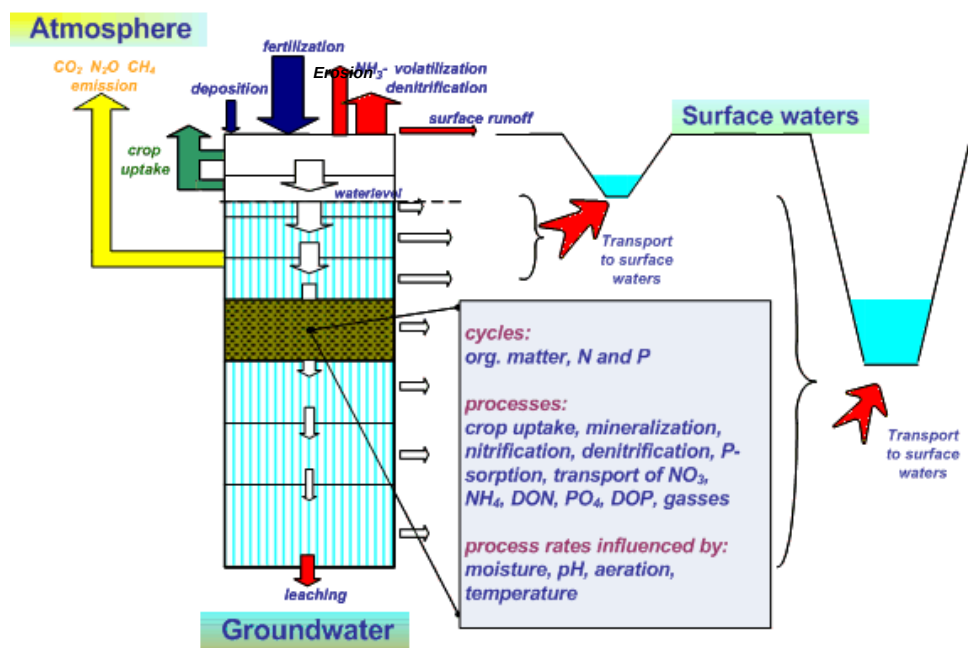


Figure 18 Schematic presentation of relevant transport routes and N and P related processes determining the N and P losses to the environment (Source: Schoumans & Silgram, 2003).

In this chapter the most relevant processes and properties determining the N losses (section 5.1) and the P losses (section 5.2) will be addressed.

5.1 N losses

The amount of N added to soils as fertilizer and animal manure in many European regions exceeds the removal of N via harvested crop and grazing by animals. Some part of the N surplus can be immobilized in the soil, but the greater part of the N surplus is lost to the environment and these N losses lead to numerous problems related to human health and ecosystem vulnerability (Figure 19, Galloway et al. (2003), Sutton et al. (2011a)).

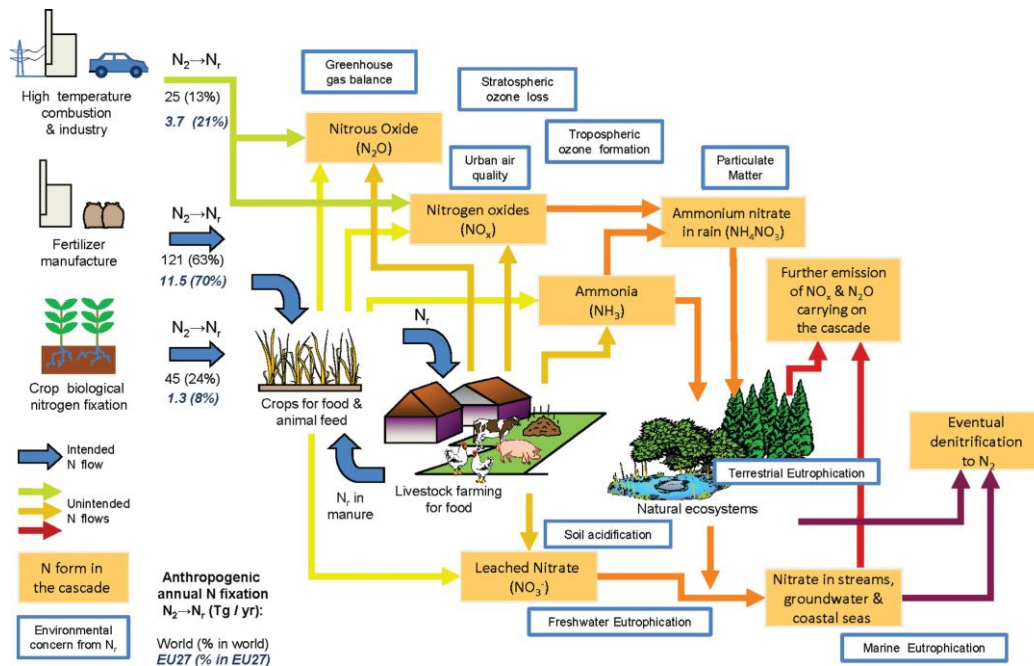


Figure 19 Overview of the N cascade, with the major anthropogenic sources of reactive nitrogen (Nr) from atmospheric di-nitrogen (N₂), the main losses of Nr (orange boxes) and the environmental concerns (boxes outlined with blue) (Source: Sutton et al., 2011a).

The N losses from farming systems comprise (See e.g. Jarvis (2001)):

- Gaseous emissions:
 - o Ammonia (NH₃) emissions
 - o Nitrous oxide (N₂O) emissions
 - o Nitric oxide (NO) and nitrogen dioxide (NO₂) emissions (the sum of both is referred to as NO_x emissions)
 - o N₂ emissions
- Surface runoff and Erosion
- Denitrification and Leaching

Significant N losses may occur via volatilization of NH₃, and emissions of N₂O, NO, and N₂ from nitrification and denitrification processes. Emissions of gaseous N compounds occur especially from feces and urine during livestock housing and manure storage, after deposition on pastures and paddocks by free ranging animals, and after application of manure and mineral N fertilizers to agricultural land (e.g. Oenema et al., 2007). Leaching and (surface)runoff of N to groundwater and surface water may occur from uncovered and unsealed manure storage systems and from agricultural fields on sloping lands and on flat terrain when rainfall exceeds evapotranspiration or during heavy rains (e.g. Van Beek et al., 2004), especially in cases with persistent N surpluses (Sutton et al., 2011b). The increase in N losses to the environment since 1900 in relation to the fertilizer and manure use in Europe is shown in Figure 20.

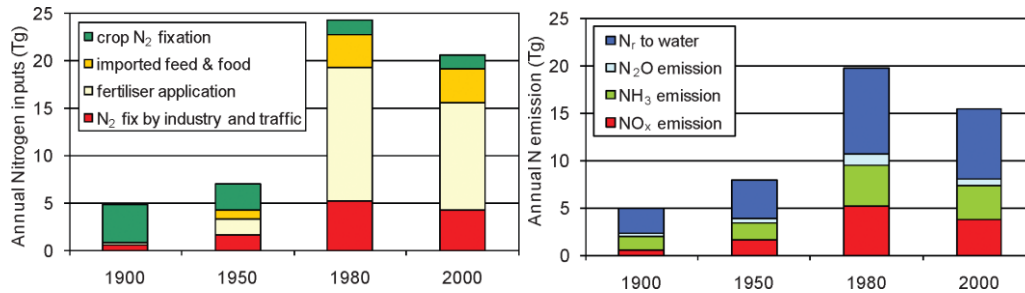


Figure 20 Temporal changes in annual N_r inputs (left) to EU-27 and the losses to the environment: surface runoff and leaching water, emission to air as ammonia (NH₃) mainly from agriculture, emission to air as nitrogen oxides (NO_x) mainly from industry and traffic (Source: Sutton et al., 2011a).

An overview of the estimated N losses for the EU-27 countries for the year 2000 is given in Figure 21. It is clear that the losses vary among the countries, which is due to differences in farming systems, N inputs, but also related to the differences in pedo-climatic factors and management.

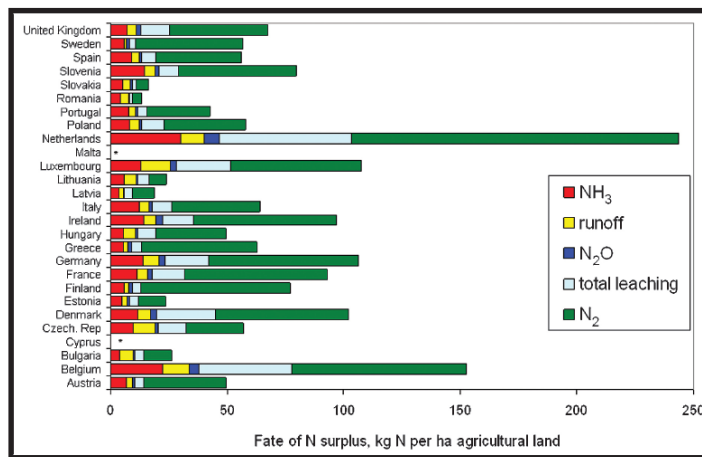


Figure 21 Fate of the N surplus of the soil balance in EU-27 in 2000 in kg N per ha agricultural land calculated using the model MITERRA-EUROPE (Source: Velthof et al., 2009).

5.1.1 Gaseous losses

5.1.1.1 NH₃ emissions

Losses of NH₃ emissions mainly originate from fertilizer use and animal excreta. Table 3 gives an overview of the global atmospheric NH₃ emissions in 1990, and Figure 22 shows the national emissions of NH₃ in 2007 for the EU Member States.

Table 3 Global Sources of Atmospheric NH_3 in 1990 (Source: Bouwman et al., 1997; Bouwman et al., 2002).

Source	$\text{NH}_3\text{-N}$ emissions (Mt yr^{-1})
Fossil fuel combustion, including aircraft	0.1
Industrial processes	0.2
Animal excreta	21.7
Fertilizer use	9.0
Croplands ¹⁾	3.6
Biomass burning, including biofuel combustion	5.9
Human excreta	2.6
Soils under natural vegetation	2.4
Oceans	8.2
Total	53.6

¹⁾ Due to emissions from crops and the decomposition of crop residues

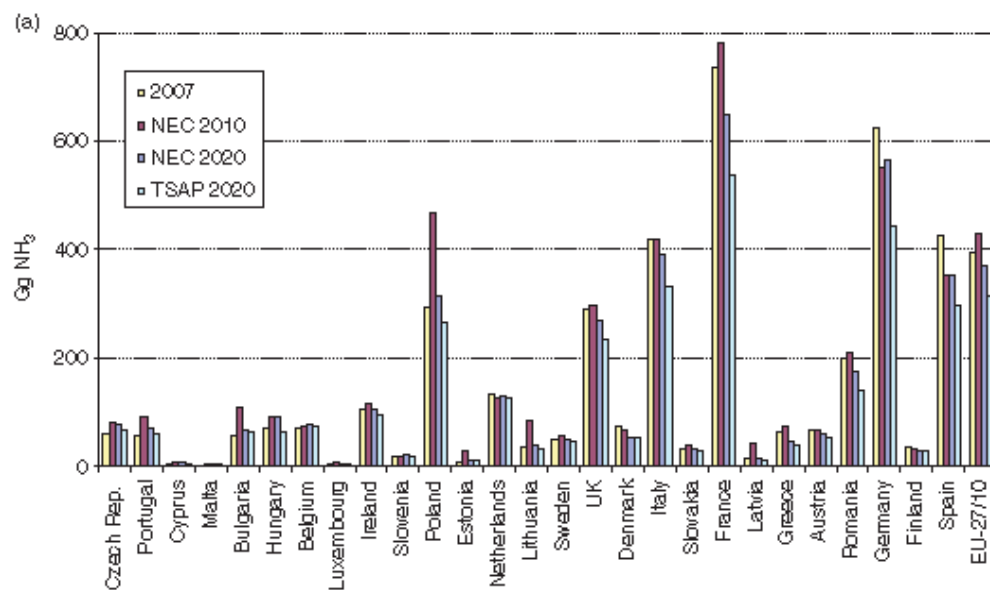


Figure 22 National emissions of NH_3 (Gg yr^{-1}) in 2007, the national emissions ceilings for year 2010 (NEC 2010) and the 'Current policy' scenario for 2020 for the EU Member States. The EU-27 emission totals are divided by 10 (Source: Moldanová et al. (2011)).

Many factors regulate the NH_3 loss from soil-plant systems to the atmosphere, depending on the crop, and management of soils, water, and fertilizers (cf. Bouwman et al., 1997). An overview of the environmental factors affecting NH_3 emissions is given in Table 4. Ammonia is produced by hydrolysis of urea (a compound of urine and also of urea fertilizers) by the enzyme urease. The rate of urea hydrolysis increases when the temperature increases. Once in the form of ammonia, it is subjected to the equilibrium between NH_3 and NH_4^+ , which is affected by pH, moisture, clay content, and wind. Factors shifting this equilibrium towards NH_3 , such as high pH, removal of NH_3 by wind, low moisture content and low clay content, enhance NH_3 emissions.

Table 4 The influence of environmental factors on NH_3 , and $\text{N}_2\text{O}/\text{NO}_x$ emissions. When an environmental factor is increased, this may lead to an increase (+) or decrease (-) in emissions (Source: Peierls et al., 1991).

Increase of environmental factor	NH_3	$\text{N}_2\text{O}/\text{NO}_x$
Soil moisture	0	+
Temperature	+	+
pH	+	-
C availability	0	+
NH_4^+ availability	+	+
NO_3^- availability	0	+
Wind	+	0
Clay content	-	+
+ = increase - = decrease 0 = no effect		

5.1.1.2 N_2O , NO and NO_2 emissions

Nitrous oxide, being the most important N related greenhouse gas, is produced during nitrification and denitrification. During this process also NO and NO_2 emissions take place. The expected impact of environmental factors on NO, NO_2 and N_2O emissions is also given in Table 4. Nitrous oxide, NO and NO_2 (the latter two are referred to as NO_x) are produced by nitrifying and denitrifying bacteria, but also by fungi and archaea (Vollenweider, 1968). Some factors enhance the activity of bacteria, but decrease the production of N_2O during nitrification and denitrification (e.g., temperature, pH). These counterbalancing effects complicate the estimation of the net effect of environmental factors on N_2O emissions. In the table, rough estimates are given, but these effects may differ between locations. Although the contribution of N_2O to the total N losses is rather small (see Figure 21), the nitrification and denitrification processes together with mineralization are crucial processes for the fate of N in the farming ecosystem (see section 3.1.2). The impact of environmental factors on NO_x and N_2O emissions as well as on nitrification and denitrification is further elaborated below.

Denitrification is thought to be generally a larger source of N_2O than nitrification, but under certain conditions (e.g. drained peat soils or urine patches) nitrification is probably also an important source of N_2O . The N_2O emissions from nitrification is controlled by the input of ammonium (via deposition, fertilizer, manures and mineralization of soil organic N), the oxygen content (which is strongly dependent on soil moisture content), pH and temperature. The N_2O emissions from denitrification is controlled by the NO_3^- input (directly via fertilizer or indirectly via nitrification of NH_4^+), content of degradable organic carbon, oxygen content (moisture content), pH and temperature. Degradable organic carbon is an energy source for the denitrifying micro-organisms, thus having an impact on denitrification and thus also on N_2O and NO_x emissions. This effect is apparent in view of differences between grassland and arable land (grassland contains much more degradable organic C than arable land), but also as regards to application of manure (organic C in pig manure is more degradable than that in cattle manure; chemical fertilizer does not contain C) and crop residues.

Environmental factors affect N₂O production by influencing the nitrification rate, the denitrification rate and also the N₂O fraction of N products produced (i.e., N₂, N₂O and NO_x) by (de)nitrification. These influences are described in literature and have been summarized in Table 5. The process rates of nitrification and denitrification and the N₂O fraction released are often inversely related. An increase in soil moisture causes lower nitrification rates, whereas denitrification rates increase (Weier et al., 1993). The fraction of N₂O produced depends on the dominant process. At saturation ‘complete’ denitrification occurs, implying no N₂O production in saturated soils, because all N is eventually converted to N₂. Wetlands are therefore only significant sources of N₂O when the soil falls dry during a part of the year, because cycles of drying and wetting of soils promote mineralization-nitrification-denitrification processes and thereby the emissions of N₂O. In wetlands that are saturated during the whole year, only some N₂O may be produced after atmospheric nitrate deposition, but this is mostly of minor importance. N₂O production from nitrification increases with temperature, whereas N₂O production from denitrification decreases with temperature. However, nitrification and denitrification are at an optimum at a different temperature range, causing N₂O production to increase with temperatures up to about 30°C.

Table 5 The influence of environmental factors on nitrification, denitrification and N₂O fractions (fN₂O). When an environmental factor is increased, this may have a positive (+) or negative (-) influence on nitrification, denitrification, and N₂O production by both processes. (Source: Peierls et al., 1991)

Increase of environmental factor	Nitrification	N ₂ O fraction nitrification	Denitrification	N ₂ O fraction denitrification
NH ₄ ⁺ availability	+	+	+	+
NO ₃ ⁻ availability	0	0	+	+
O ₂ -content (decrease of soil moisture)	+	-	-	+
Carbon availability	0 autotrophic + heterotrophic	0 autotrophic - heterotrophic	+	-
Temperature	+	+	+	-
pH	+ autotrophic - heterotrophic	+	+	-

+ = increase - = decrease 0 = no effect

Nitrate has a positive effect on denitrification. This would cause a more complete process and therefore conversion of N₂O to N₂. However, when nitrate concentrations in soils are very high the N₂O/N₂ ratio from denitrification will decrease, because nitrate is favoured as a substrate over N₂O by denitrifiers (Wrage et al., 2001).

5.1.2 Surface runoff and erosion

There are two main processes at the soil surface that are causing nutrient losses, surface runoff and erosion. Nitrogen losses through surface runoff are in general much larger than N losses via erosion, whereas for P losses it is the other way around. Yet, the factors controlling runoff and erosion are comparable to both N and P. Therefore, we discuss the factors controlling losses via surface runoff in this section and the factors controlling erosion in section 5.2.1 .

Surface runoff occurs when rainfall exceeds a soil's saturation level. When the soil is saturated and the depression storage filled and rain continues to fall, the rainfall will immediately produce surface runoff. If the amount of water falling on the ground is greater than the infiltration rate of the surface, runoff or overland flow will occur. The rate of runoff flow depends on the ratio of rain intensity to the infiltration rate. If the infiltration rate is relatively low, such as when a soil is crusted or compacted, and the rainfall intensity is high, then the runoff rate will also be high. Runoff specifically refers to the water leaving an area of drainage and flowing across the land surface to points of lower elevation. When rainfall intensity exceeds the soil's infiltration rate, a thin water layer forms that begins to move because of the influence of slope and gravity. Important factors controlling N losses due to surface runoff are the type, period and amount of N and P application, the slope of the soil, soil type and properties, weather conditions (precipitation, frost) and hydrology. Schwaiger et al. (2006) carried out a desk study to quantitatively assess the surface runoff of N from agricultural soils as part of the MITERRA-EUROPE project (see Velthof et al., 2007). This desk study identified the following parameters for a high risk of N surface runoff:

- Weather conditions - heavy precipitation, snow melt, storm.
- Soil conditions - soil with low infiltration rate.
- N Fertilizer input - high amount of fertilizer.
- Type of vegetation (length of growing season). By comparing different land use Korsath & Eltun (2000) found that runoff can reach from 18 to 35 kg N ha⁻¹ yr⁻¹. Forage system had lower N runoff than arable systems.
- Tillage. Soil tillage is the key management to avoid or perpetuate surface runoff and erosion. In relation to tillage effects, it is difficult to distinguish between N losses from surface runoff and erosion. To separate the effects from different tillage methods data on tillage is needed, but these are hardly available.
- Steep slopes.

Beside the weather conditions, the soil infiltration rate or capacity is of great importance controlling surface runoff. The soil infiltration depends on the soil texture (a sandy surface soil normally has a higher infiltration rate than a clayey surface soil), crust formation (slaking), compaction (which may lead to an impervious layer, close to the surface, which restricts the entry of water into the soil and tends to result in pounding on the surface), aggregation/structure, water content (the infiltration rate is generally higher when the soil is initially dry and decreases as the soil becomes wet), organic matter (organic matter increases the entry of water by protecting the soil aggregates from breaking down during the impact of raindrops) and pores (top soil air capacity). The surface runoff is also influenced by crop type, presence of terraces and tree lines, and soil management.

There is no literature about N surface runoff as affected by application of different types of N fertilizers. The estimation of the N surface runoff from different fertilizers depends on several parameters. In general it is assumed that 10-20 % of total N load in surface water is caused by surface runoff, the rest by leaching (Velthof et al., 2007).

Summarizing, the most important factors causing N losses due to surface runoff are i) the amount of precipitation, ii) the slope gradient and iii) the vegetation (see Chapter 4).

5.1.3 Denitrification and leaching

The N surplus subjected to leaching and denitrification, is the difference between total N input to soil (via manures, fertilizer, atmospheric deposition, net mineralisation and biological fixation, but corrected for possible losses via NH_3 and N_2O emissions, surface runoff and erosion) and the total N output via harvested crops. To derive the N leaching from the corrected N surplus (i.e., corrected for possible losses via NH_3 and N_2O emissions, surface runoff and erosion) is usually done by using a leaching or denitrification fraction (with denitrification fraction + leaching fraction = 1). The most important factors controlling denitrification are (i) the presence of an energy source for the denitrifying bacteria, mostly available organic carbon, (ii) anoxic conditions, and (iii) the nitrate content in the soil. If any of these conditions is not fulfilled, denitrification is unlikely.

Most important factors determining the leaching fraction are (see Velthof et al., 2007):

- Soil type, geomorphology, soil hydrology and groundwater level affect oxygen concentration, which in turn affects denitrification losses.
- Land use has a strong effect on available organic C contents in the soil and thereby on the denitrification capacity.
- The precipitation surplus in combination with soil texture is an indicator for the wetness of the soil, influencing the oxygen content.
- A decreasing rooting depth increases the risk of N leaching.

The leaching below the rooting zone is subjected to either leaching to surface waters and/or to groundwater (see Figure 18). Groundwater transport of nitrogen takes place over long distances and time-scales. Due to long residence times, the groundwater system acts as a temporary sink. In groundwater reactive nitrogen is converted through denitrification, i.e. the reduction of nitrate (NO_3^-) to N_2O , NO and non-reactive N_2 . Van Drecht et al (2003) illustrated the effect of groundwater age and the history of fertilizer N input for the shallow and deep groundwater layers on calculated nitrate concentrations for Western Europe (Figure 23). However, the importance of groundwater denitrification is highly uncertain. Seitzinger et al. (2006) argue that groundwater is an important site for denitrification due to long groundwater residence times. Keuskamp et al. (2012) developed a European model to estimate the denitrification and the associated gaseous losses from groundwater.

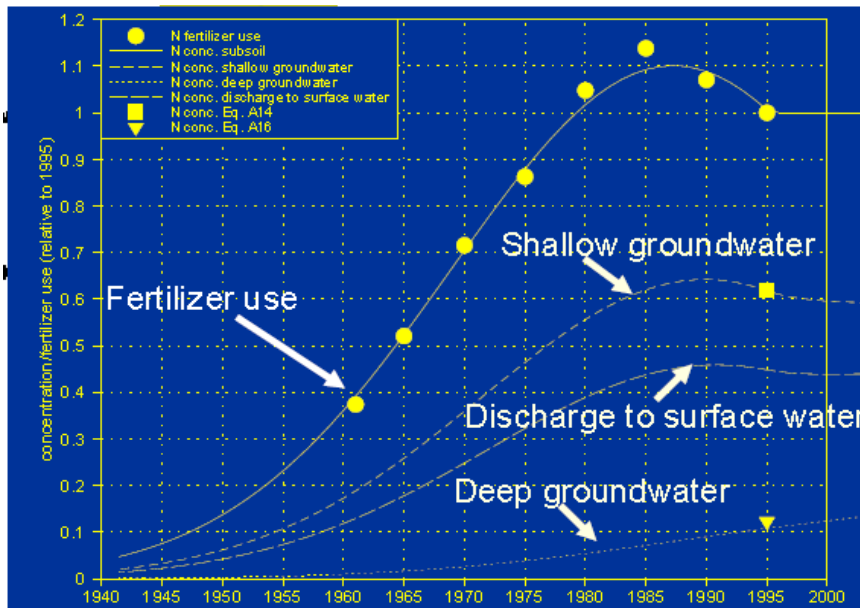


Figure 23 History of N fertilizer use for Western Europe expressed in relative terms compared to 1995, and the associated relative distribution of nitrate concentration for the outflow of groundwater in the shallow and deep groundwater layers. The mean relative outflow concentration represents the annual groundwater discharge to river systems from both the shallow and deep groundwater layers (Source: Van Drecht et al., 2003).

5.2 P losses

As for N, the P surplus is a possible, distal indicator for P losses to the environment. Factors controlling P losses from the soil comprise (see Figure 4 and Figure 18):

- Surface runoff and erosion.
- Sorption.
- Leaching.

The P soil balance includes all relevant P inputs and outputs from the soil. The fate of the P surplus is controlled by a combination of factors, including type and rate of P input, soil type and properties, geomorphology, weather conditions, crop type, and the hydrology of the soils. Important differences with N are that (i) there are no gaseous P losses, (ii) the retention of P in the terrestrial ecosystem is much stronger due to low solubility of P compounds, (iii) P losses due to erosion play a major role, (iv) the return of lost P to surface waters is at geological time-scale only; there is no other P sources than P containing sediments and rocks and (v) the relative contribution of point sources to the total loss is much larger for P than for N (see Figure 26). Point source discharges directly into water bodies, for example, disposal through wastewater treatment plants or households not connected with a sewage plant. By contrast, nonpoint sources affect a water body from diffuse sources such as runoff from agricultural areas (Schröder et al., 2011).

An overview of the P balance in Europe is given in Figure 24, where the P balance (P_{bal}) is defined as:

$$P_{bal} = P_{man} + P_{fer} + P_{graz} + P_{dep} - P_{up}$$

Where:

- P_{man} the input by animal manure application
- P_{fer} the input by synthetic fertilizer
- P_{graz} the input by manure deposited by grazing
- P_{dep} the input by atmospheric deposition, weathering and in some cases through irrigation by surface water.
- P_{up} crop removal

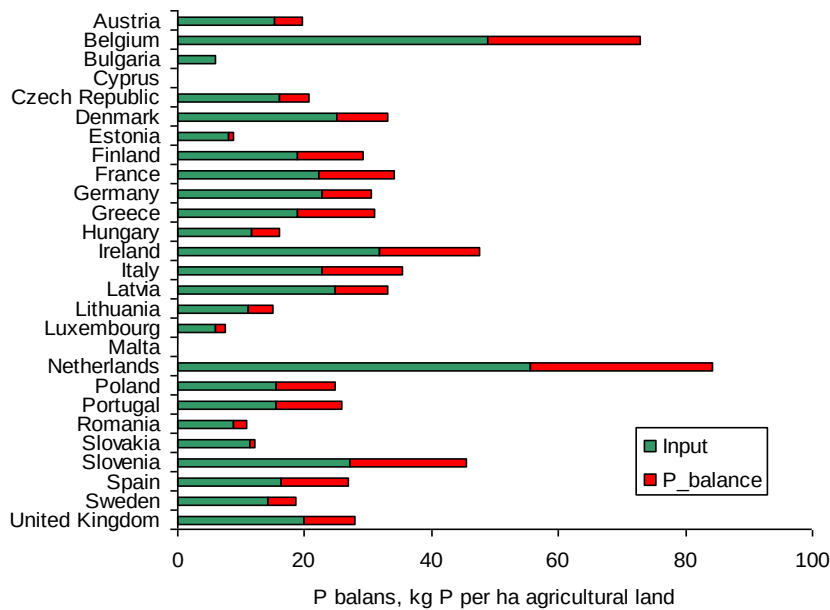


Figure 24 The P soil balance in EU-27 in 2000 in kg P per ha agricultural land calculated using the model MITERRA-EUROPE (Source: Velthof et al., 2009).

Contrary to N inputs, the P inputs are more in balance with the requested crop uptake. The mean distribution for 2005-2008 of the P inputs over the various input terms is given in Figure 25.

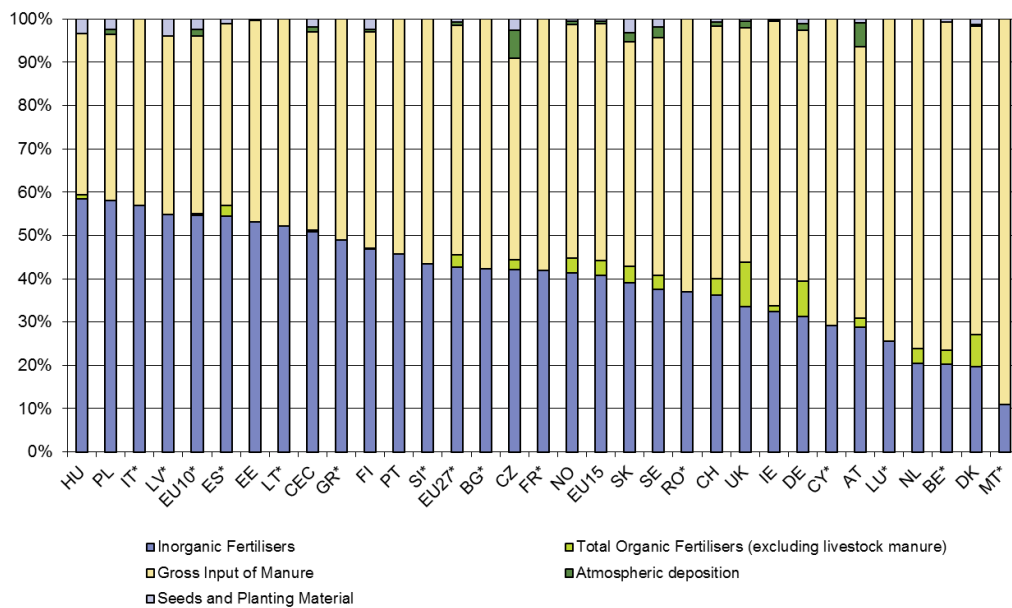


Figure 25 The share (%) in total Phosphorus inputs per MS, average 2005-2008 (Source: Eurostat).

Based on a model study Grizzetti & Bouraoui (2006), estimated a much higher contribution of diffuse sources to the total load of rivers for N (generally between 50-100%) compared to P (generally less than 25%) (Figure 26). However, a comparison with observation data indicates that diffuse emissions of P may be underestimated. Other literature data for European catchments show a contribution of ~60% from point sources for P and ~30% for N (Grizzetti & Bouraoui, 2006; Grizzetti et al., 2008).

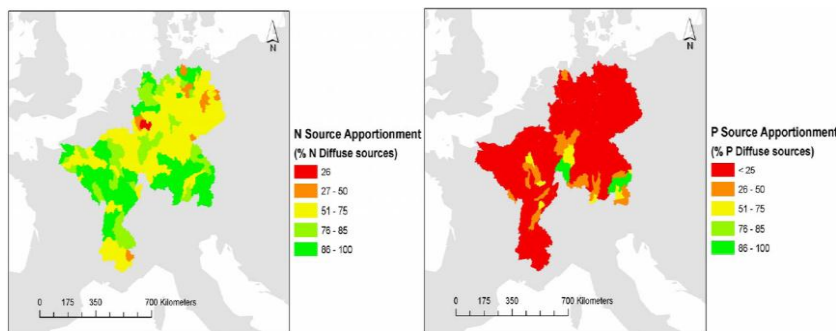


Figure 26 Calculated N (left) and P (right) source apportionment (average of 1995- 2002). It is assumed that the upstream load is proportionally distributed between point and diffuse (Source: Grizzetti & Bouraoui, 2006).

5.2.1 Surface runoff and erosion

Erosion leads to displacement of soil particles by water or wind. It causes irreversible soil losses. The Mediterranean region is particularly prone to erosion, because of long dry periods followed by heavy rainfall, especially in regions with steep slopes

(Lesschen et al., 2009). Erosion results in loss of fertile soil and pollution of surface water with N and P and other 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 and P.

All factors related to the surface runoff of N as described in paragraph 5.1.2, also apply to P. Since the water solubility of P is much less compared to N, runoff of P is of less importance. The opposite, however, is true for erosion. Quinton et al. (2010) estimated that for the European continent the amount of eroded P is ~80% of the amount of P fertilizer use (Figure 27). For the Odense catchment in Denmark, Kronvang et al. (1968) estimated a net P input from stream bank erosion from 17 to 29% of the annual total P export and 21 to 62% of the annual export of P from diffuse sources.

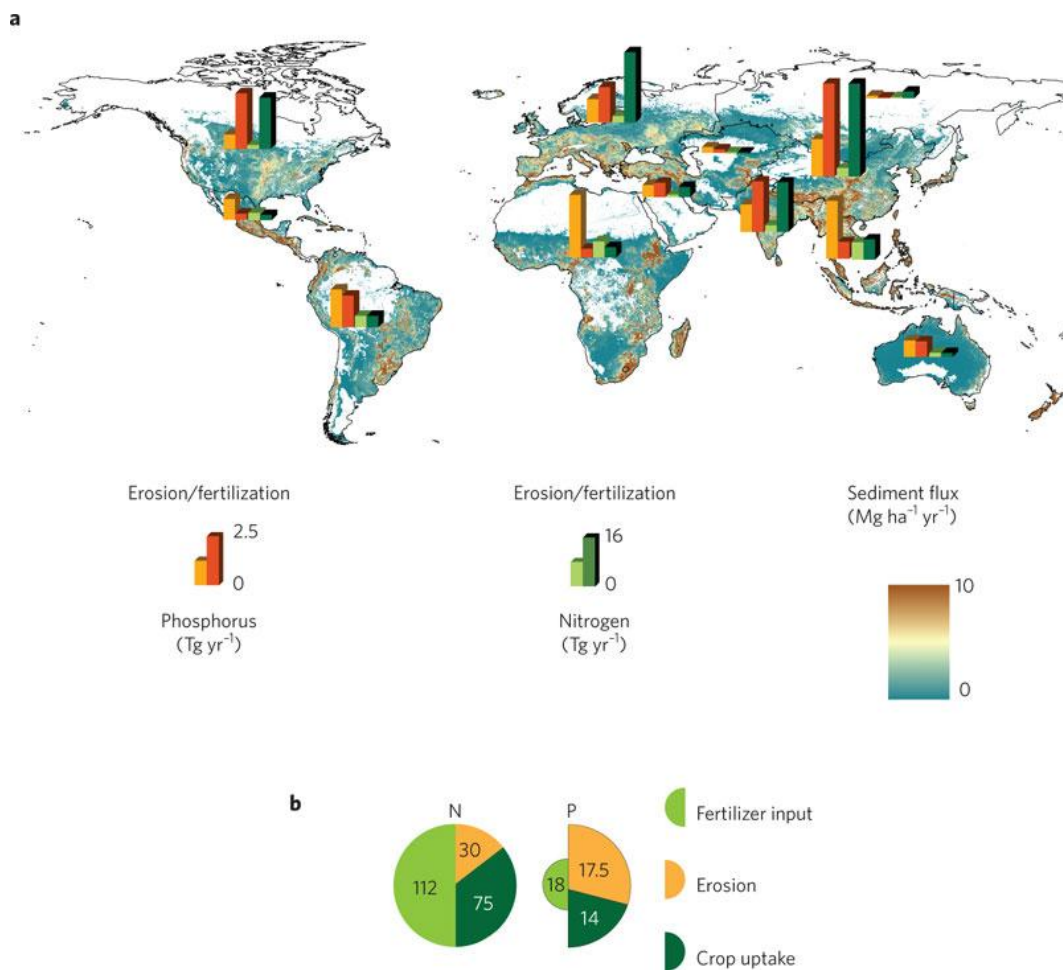


Figure 27 The global distribution of sediment fluxes of nitrogen and phosphorus by water and tillage erosion compared with fertilizer use (Source: Quinton et al., 2010). a.: Shaded areas show the global distribution of sediment fluxes. Bars show the continental fluxes of nitrogen and phosphorus by water and tillage erosion compared with fertilizer use. b: Global fluxes of nitrogen and phosphorus (Tg yr⁻¹) due to fertilizer input, erosion and crop uptake.

Compared to arable land, grasslands and forests have negligible soil erosion rates. Consequently, 75–80%, and often more than 90%, of all soil erosion comes from arable land (Smil, 2000).

Factors involved in erosion are indicative of any water-borne transport process along slopes. The main factors relevant for erosion are rainfall, the geomorphology, soil texture, land cover, land management (see Figure 28), thickness of the soil and the permeability of the soil (see Part B). Since erosion and the elemental cycles The impact of land management on soil erosion on carbon is illustrated in Figure 28., which is linked with P (and N) through the presence of P (and N) inorganic matter (Quinton et al., 2010).

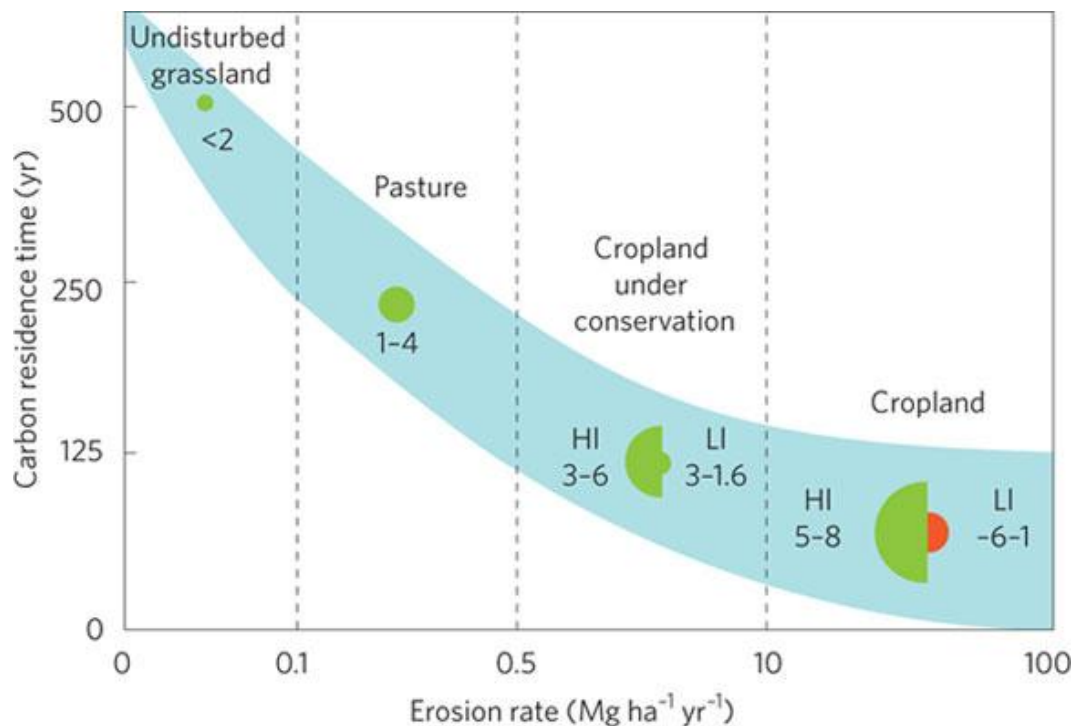


Figure 28 Interaction between soil erosion, land use/soil management and carbon cycling at sites of erosion. Size of the circles represents the maximum size of the carbon sink (positive, green) or source (negative, red) (Quinton et al., 2010).

5.2.2 Sorption and leaching

The continuing addition of P through manure and fertilizer application, may result in considerable annual surpluses of the nutrient in a variety of farming systems (Schoumans & Groenendijk, 2000), resulting ultimately in downward leaching of P. Phosphorus applied to soils is involved in a multitude of complex reactions that remove it from the solution and incorporate it into a large variety of much less soluble, or insoluble, labile and stabile compounds.

Related to P leaching the most relevant question is to find out how much of the P is retained in the soil soon after application and how much is available for crop uptake and leaching. Sandy soils and soils with nearly neutral pH have relatively little sorption, whereas acidic, clayey soils with high iron (Fe) and aluminium (Al) content have the highest fixation. The sorption/retention of inorganic P in soil is characterized by a fast reversible process and a slow, almost irreversible process. The fast reaction is generally attributed to P adsorption on surface sites. The slow reaction is mostly viewed as diffusion or nucleation-controlled precipitation reaction. For example, diffusion of P into particles of microcrystalline Al and Fe oxides or slow nucleation and growth of a Ca-P-precipitate at the surface of calcium carbonate (Smith et al., 1999; Huijsmans et al., 2001; Misselbrook et al., 2005). Furthermore, dissolution of a superphosphate granule reduces the pH of soil water in its immediate surroundings and releases Al, Fe, Ca, K, and Mg from soil particles; they react with fertilizer P and produce relatively insoluble, and hence immobile, compounds (see e.g. Smil, 2000; Koopmans et al., 2004a; Koopmans et al., 2007).

5.2.3 P saturated soils

The degree of phosphate saturation of a soil may be defined in a general way as the fraction of soil surfaces (binding sites) coverage with phosphate and indicates the potential risk of leaking of soil P towards groundwater and surface water. There are different indices that have been proposed to calculate the degree of soil P saturation (Jeppesen et al., 2010). In the beginning, the index was defined as the ratio of the soil ammonium-oxalate-extractable P content (P_{ox}) to its P sorption capacity (see section 8.3.1). The P sorption capacity was estimated with either sorption parameters derived from laboratory experiments or with the oxalate-extractable Al and Fe contents ($Al_{ox} + Fe_{ox}$) (Schoumans et al., 1987; Van der Zee & van Riemsdijk, 1988).

5.3 Concluding remarks

In contrast to N, the P surplus is not a strong indicator for P losses to the environment, as P losses are determined also by the soil P status, which is strongly influenced by the retention of the P surplus over the past years. The retention of the P surplus by the soil is controlled by a combination of factors, including soil type and properties, slope, weather conditions, crop type and the hydrology of the soils. Important differences with N are that (i) there are no gaseous P losses, (ii) the retention of P in the terrestrial ecosystem is much stronger due to low solubility of P compounds and (iii) the relative contribution of point sources to the total loss is much larger for P than for N. Point sources discharges directly into water bodies, for example, disposal through wastewater treatment plants, industries and households (not connected to a sewage treatment plant). By contrast, nonpoint sources affect a water body from diffuse sources such as runoff from agricultural areas. Phosphorus applied to soils is involved in a multitude of complex reactions that remove it from the solution and incorporate it into a large variety of much less soluble, and stabile compounds. Related to P leaching the most relevant question is to find out how much of the P is adsorbed in the soil soon after application and how much is available for crop uptake and leaching. Sandy soils and soils with nearly neutral pH

have relatively little sorption capacity, whereas acidic, clayey soils with high iron (Fe) and aluminium (Al) content have the highest retention potentials.

6 Factors influencing NH₃ volatilisation

6.1 Introduction

Nitrogen may be lost in the form of ammonia (NH₃) from fertilisers containing nitrogen in the form of ammonium (NH₄-N). Many factors influence the emission rate, like the properties of the fertiliser, climate, soil conditions, spreading technology and rate (Svensson, 1994; Søgaard et al., 2002; Misselbrook et al., 2005). Ammonia volatilisation is depending on the area of the emitting solution source exposed to the atmosphere. Hence, reducing the source area exposed to atmosphere is therefore a very efficient method to reduce ammonia emissions. This can be done by covering the manure storage or using injection technique in order to place the slurry and mineral fertilizer into soil at spreading. There is also an interaction between soil and manure and/or mineral fertilizer properties, as for instance the combination of liquid forms of fertiliser and a porous soil may lead to a small time of exposure and thereby low ammonia emissions (Svensson, 1994). Rain after spreading may be another natural condition that reduces ammonia emissions by speeding up the infiltration of manure NH₄-N into soil (Webb et al., 2005).

6.2 Effect of climatic factors

6.2.1 Effect of temperature

Temperature has a large impact on ammonia volatilisation (Sommer & Olesen, 1991; Svensson, 1994) with an exponential increase with increased temperature below 15°C, and linearly from 15 to 25 °C.

6.2.2 Effect of precipitation

For applied solid manure, Misselbrook et al. (2005) identified rainfall after spreading as the parameter with most influences ammonia emissions. The same effect could be achieved with irrigation immediately after spreading. Rodhe et al. (1996) irrigated with 15 mm water immediately after spreading of solid manure on grassland, and thereby the ammonia emissions were reduced by up to 30%, while the reduction after the application of slurry on grassland were even higher, by 67% for pig and 80% for cattle slurry, with 10 mm water irrigated, (Klarenbeek, 1990). Diluting the slurry with water 1:3 had about the same effect in reducing ammonia emissions.

6.2.3 Effect of wind speed

Wind speed is one of the parameters significantly affecting NH₃ volatilization as shown in the statistical analysis of NH₃ data from Denmark, Italy, the Netherlands, Norway, Sweden, Switzerland and UK (Søgaard et al., 2002) Misselbrook et al. (2002). Sommer et al. (1991) found that the ammonia loss rate increased when wind

speeds increased up to 2.5 m/s. At higher wind speeds from 2.5 to 4 m/s there was no consistent increase.

6.3 Effect of pedological factors

6.3.1 Effect of soil texture, soil organic carbon and soil CEC

Soil texture and soil organic carbon are factors that can regulate NH_3 volatilization. Bouwman et al. (2002) assessed the influence of soil texture (coarse, medium, fine) and soil organic carbon, SOC ($\text{SOC} \leq 1\%$, $1 < \text{SOC} \leq 2.5$, $2.5 < \text{SOC} \leq 5$ and $\text{SOC} > 5$) using a meta analyses of literature data, and found that the effects of soil texture and organic carbon content were not clear. However, soil texture and soil organic carbon are the main determinants of soil CEC. Therefore the influence of soil organic carbon and soil texture on NH_3 volatilization can be assumed to be included in the factor CEC. The soil CEC influences the ammoniacal N concentration through the reaction of positively charged NH_4^+ with the negatively charged cation exchange sites. Hence, soils with low CEC are more prone to high NH_3 volatilization losses than are soils with high CEC. Bouwman et al. (2002) found also that the median NH_3 volatilization was about 40% lower for soils with a $\text{CEC} > 32 \text{ mol}_c \text{ kg}^{-1}$ than for soils with $\text{CEC} < 32 \text{ mol}_c \text{ kg}^{-1}$.

6.3.2 Effect of pH

The pH affects the equilibrium between NH_4^+ and NH_3 , as the relative concentration of NH_3 in the soil increases from 0.1 to 50% as pH increases from 6 to approx. 9 (Freney et al., 1981). Both the volatilization process itself and the nitrification process can reduce NH_3 volatilization by decreasing NH_4^+ availability and by producing acidity (He et al., 1999):

Volatilization: $\text{NH}_4^+ \leftrightarrow \text{NH}_3 + \text{H}^+$

Nitrification: $\text{NH}_4^+ + 2\text{O}_2 \rightarrow 2\text{H}^+ + \text{H}_2\text{O} + \text{NO}_3^-$

Bouwman et al. (2002) carried out a meta-analysis of results in literature, and found that increased soil pH had a consistent relationship with increased NH_3 volatilization, where the balanced medians of NH_3 volatilization from soils with $\text{pH} > 8.5$ exceeded those soils with $\text{pH} 5.5 - 7.3$ by 61% and those soils with $\text{pH} < 5.5$ by 80%. The balanced medians of NH_3 volatilization from soils with $\text{pH} 7.3 - 8.5$ exceeded those soils with $\text{pH} 5.5 - 7.3$ by 39% and those soils with $\text{pH} < 5.5$ by 55%.

6.3.3 Effect of soil drainage

NH_3 volatilization increases if the infiltration is reduced because of increased soil water content (EC, 2009). In a laboratory study, it was shown that the NH_3 volatilization from slurry applied to dry soil ($0.01 \text{ g H}_2\text{O g}^{-1}$ of soil) was 70% of the volatilization from slurry applied to soil with more than $0.8 \text{ g H}_2\text{O g}^{-1}$ soil (Sommer & Jacobsen, 1999). Model results confirms the findings of a number of previous studies showing that the NH_3 volatilization is relatively low when slurry is applied on

dry soil (Søgaard et al., 2002; HELCOM, 2009), even if the air or soil surface temperature is high (HELCOM, 2009).

6.4 Interaction soil, climate, fertiliser and crop

Reducing the manure exposure to the atmosphere is a very efficient method to reduce ammonia emissions. This could be achieved by many ways like making the soil more porous by cultivating, or increase the infiltration ability of the slurry into the soil by dilution. When spreading during the spring tillage, NH_3 losses after application of urine were limited to less than 5% of applied N due to the good infiltration ability of the urine into the seed bed (Rodhe et al., 2004). A very efficient way to reduce NH_3 is to incorporate or inject the slurry into the soil (Huijsmans et al., 2001; Rodhe et al., 2006). Lowering the pH of the slurry can be another way to reduce NH_3 emissions from storage and spreading of slurry. With a system decreasing slurry pH to 5.5 by adding 4-6 kg concentrated sulphuric acid per ton of manure, the NH_3 emissions were reduced by 70% (Pedersen, 2004).

It is assumed that NH_3 volatilization rates are generally lower in grasslands than in cultivated and managed areas, but Bouwman et al. (2002) found almost no difference between grasslands and croplands concerning NH_3 volatilization, using a meta-analyses of the results in literature.

Thorman et al. (2008) found a clear relationship between crop height for band-spread slurry applications to cereal crops and grassland and NH_3 volatilization rates. Crop height provided a major explanation of the variation in NH_3 volatilization rates, where the developed algorithm predicted that for slurry applications to cereal crops, the reduction efficiency would increase by slightly less than 1% for every 1 cm increase in crop height. For slurry application to grassland, the reduction efficiency was predicted to increase by approximately 5% for every 1 cm increase in sward height. Also a number of other factors contributed; e.g., precision of the application rate, precision of the measurement of slurry TAN content and differences in the degree of fouling of crop leaves. For a given crop height a grass canopy appears to be more effective at reducing NH_3 emissions than the cereal canopy for a given crop height (Thorman et al., 2008). This is most likely due to the denser canopy structure of grass, which has a leaf cover of approximately twice that of cereals for crop height between 5 and 10 cm. In practice, slurry is most often applied immediately after the grass is cut. The reasons are that the farmer want to minimize the risk of contamination of the herbage with manure and that the crop require mineral N. Shallow injection is an effective strategy to reduce NH_3 volatilization and the risk of contamination of herbage (Schils et al., 2007).

Sommer et al. (1997) compared NH_3 volatilization from pig slurry applied to winter wheat with trail hoses on the soil below the canopy and by a splash plate technique spreading the slurry on both plants and soil. Accumulated NH_3 volatilization from pig slurry applied with trail hoses was from 4 to 26% of applied TAN, and when applied with splash plates, from 11 to 26%. Volatilization loss of NH_3 from trail hose applied slurry was significantly lower than from splash plates. The reduction in NH_3

volatilization was greatest when the slurry was applied to a tall and dense winter wheat crop and soil at low water content. There was no difference in NH_3 volatilization between trail hose and splash plates when slurry was applied to a 10 cm high crop with a leaf area index of 0.3. The NH_3 volatilized from trail hose applied slurry was absorbed by the wheat plants in rates from 0 to $0.74 \text{ g NH}_3\text{-N m}^{-2}$ leaf surface during a period of 7 days after slurry application. Canopy NH_3 absorption was responsible for up to 25% of the reduction on NH_3 loss when using trail hose application.

6.5 Concluding remarks

Analysis of European data of NH_3 emissions after manure application show that variables significantly affecting NH_3 volatilization throughout Europe are soil water content, air temperature, wind speed, slurry properties such as type, dry matter content and content of total ammoniacal nitrogen of slurry, application method and rate, slurry incorporation and measuring technique (Søgaard et al., 2002). Experiments by Misselbrook et al. (2002) confirms the importance of the climatic factors where the most important parameter for applied slurry was wind speed together with slurry dry matter content, and for applied solid manure rainfall data.

7 Factors influencing N runoff and downward leaching

7.1 Introduction

Loading of N into surface water and groundwater is a function of transport volume (amount of water) and nitrate concentration in the transported water. The amount of drainage water leaving the landscape is largely a function of climate and soil properties. This section describes the effect of the climatic factors such as temperature and precipitation and the pedological factors such as texture, slope, soil organic matter, pH and soil drainage on N leaching and runoff.

7.2 Effect of climatic factors

7.2.1 Effect of temperature

Temperature affects the biological processes determining the N availability in the soil solution and thereby subjected to leaching. Table 6 gives an overview of the effect of temperature on the N availability through the processes mineralisation, nitrification, denitrification, crop uptake and crop residue mineralisation.

Table 6 The influence of temperature on the N availability through the processes mineralisation, nitrification, denitrification, uptake and crop residue mineralisation. When temperature is increased, this may have a positive (+) or negative (-) on NH_4^+ availability, NO_3^- availability and the resulting N leaching (Source: Peierls et al., 1991).

Process	Effect on process	NH_4^+ availability	NO_3^- availability	N leaching
Mineralisation	Higher mineralisation rate $\rightarrow$ NH_4^+ production	+	0	+
Nitrification	Transformation of NH_4^+ to NO_3^-	-	+	+
Denitrification	Transformation of NO_3^- to N_2 , N_2O and NO	0	-	-
Uptake	Higher N uptake due to higher growth rate	-	-	-
Crop residue mineralisation	?	?	?	?

+ = increase - = decrease 0 = no effect ? = unknown

Nitrate availability generally increases the micro biological growth rate and can either cause an increasing or a decreasing effect on N leaching. Increasing temperature results in an increase in mineralisation and nitrification, which in turn yields higher NH_4^+ and NO_3^- availability, respectively. Temperature has no effect on denitrification. However, increasing denitrification leads to consumption of NO_3^- , which yields to a

decrease in N leaching. Tiedje (1988a) showed that at a mean annual temperature of 15°C N loss via denitrification is twice that at 5°C. Temperature increase may also yield higher growth rates, which enhances the N uptake and reduces the N leaching. In addition to N uptake rates, the N input through litter production or crop residues may increase. However, what the net effect of increasing crop residue mineralisation on N leaching will be, is difficult to estimate.

7.2.2 Effect of precipitation

Increases in precipitation will generally leads to an increase in N leaching. Based on the monitoring of temporal changes in nitrate N loads for a catchment of the Minnesota River basin, Randall & Mulla (2007) showed the relationship between nitrate loadings and growing season precipitation amounts (Figure 29). Generally, there is a good relationship between precipitation and nitrate N loads in the river.

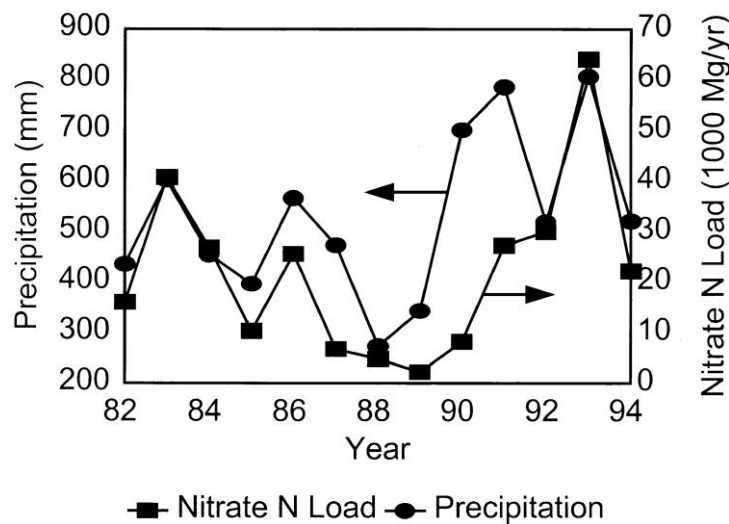


Figure 29 Temporal changes in nitrate N loading from the Greater Blue Earth River catchment, MN, USA in relationship to annual precipitation (source: Randall & Mulla (2007)).

Furthermore, drainage is also influenced by the temporal distribution of precipitation within a particular year. For instance, high rainfall in the spring, when evapotranspiration (ET) losses are low and soil moisture in the profile near field capacity, will have a much greater effect on drainage volume than the same rainfall during the middle of the summer, when daily ET losses are high and soil moisture is far below of field capacity.

Randall & Mulla (2007) (while referring to Goolsby et al. 1997) noted that the concentration and flux of nitrate tend to be highest in the spring or autumn, when stream flow is highest. This direct relationship between nitrate concentration and flow may result from leaching of nitrate from the soil during periods of high rainfall. Increased flows and elevated concentrations in agricultural tile drains were also speculated as contributing to this relationship.

N leaching is also greatly affected by dry and wet climatic cycles. Furthermore, nitrate concentrations is strongly influenced by variation in precipitation, both the between and the within year variation, esp. in shallow groundwater. When monitoring groundwater quality for quantification of the effectiveness of the action program, the measured nitrate concentration is biased by this variation. Therefore, in the Netherlands Fraters et al. (1998) adjusted the measured nitrate concentration for dilution associated with differences in actual precipitation and average precipitation.

7.3 Effect of pedological factors

7.3.1 Effect of soil texture

Soil texture affects surface runoff. The risk of surface runoff increases when clay content of the soil increases and is higher in shallow soils than in deep soils, because the infiltration of water is hampered in shallow soils.

Texture influences drainage, which in turn influences the quantity of solute leaching and the oxygen contents, which controls the (de)nitrification and mineralisation processes. Soil texture strongly affects the diffusion of oxygen into the soil and thereby the oxygen partial pressure in the soil. In general, denitrification losses will increase in the order: sandy soil < loamy soils < clay soils < peat soil. Based on a compilation of literature data and expert judgements, De Vries et al. (2005) derived denitrification fractions for several soil type- crop combinations in the Netherlands (Table 7).

Table 7 Ranges in nitrification and denitrification fractions for soil type – crop combinations derived for the Netherlands (source De Vries et al. (2005)).

Soil type	Wetness class	Nitrification fraction; fr_{n_2}		Denitrification fraction; fr_{d_2}			
		Grassland	Maize/arable	Grassland	Maize/arable	Groundwater	Ditches
Sand	Dry ¹	0.98–1.00	0.98–1.00	0.30–0.60	0.20–0.50	0.10–0.50	0.00–0.80
	Moist ²	0.95–1.00	0.95–1.00	0.40–0.80	0.30–0.70	0.20–0.70	0.10–0.90
	Wet ³	0.90–0.95	0.90–0.95	0.60–0.95	0.50–0.95	0.50–0.90	0.50–0.95
Loess	Dry	0.98–1.00	0.98–1.00	0.50–0.70	0.30–0.60	0.10–0.85	0.10–0.85
	Moist	0.95–1.00	0.95–1.00	0.60–0.85	0.40–0.80	0.30–0.90	0.30–0.90
	Wet	0.90–0.95	0.90–0.95	0.70–0.95	0.60–0.95	0.50–0.95	0.50–0.95
Clay	Dry	0.95–1.00	0.95–1.00	0.60–0.90	0.50–0.90	0.30–0.90	0.30–0.90
	Moist	0.90–1.00	0.90–1.00	0.70–0.95	0.60–0.90	0.50–0.95	0.50–0.95
	Wet	0.85–0.95	0.85–0.95	0.80–0.98	0.80–0.98	0.80–0.95	0.80–0.95
Peat	Dry	0.90–1.00	0.90–1.00	0.80–0.95	0.60–0.90	0.90–1.00	0.80–0.95
	Moist	0.85–0.95	0.90–1.00	0.80–0.95	0.70–0.95	0.90–1.00	0.80–0.95
	Wet	0.80–0.90	0.80–0.95	0.90–0.98	0.90–0.98	0.90–1.00	0.90–0.98

¹Dry is related to a MHW level of >80 cm; ²Moist is related to a MHW level between 40 and 80 cm; ³Wet is related to a MHW level below 40 cm.

Further, the risk of N leaching increases when the rooting depth decreases. Studies clearly indicate that deeply rooting crops, can remove NO_3^- from the soil profile up to more than 1 m (Rustad et al., 2006). Soils with a very shallow rooting depth are more susceptible for leaching than soil with a deep rooting depth. This is of concerns in case of thin soils, due to nearby bedrock, but also in horticulture, given the shallow root depth of most vegetable crops in combination with the intensive use of fertilisers.

7.3.2 Effect of soil organic matter

Soils high in organic matter can mineralize a substantial amount of NO_3^- , which is susceptible to loss in subsurface tile drainage, especially when wet years follow very dry years. Based on a long term study at the same catchment in the USA as mentioned before, Randall & Mulla (2007) showed that high concentrations of nitrate can easily be lost to tile drainage from high organic matter soils even if no N or very small amounts of N are applied, especially in wet years following dry years when crop production is limited. Hatfield (1996) found for another catchment (in Iowa, USA) that NO_3^- -N concentrations ranged from 15 to 20 mg N l⁻¹ throughout most of the year and stated that this loss is due primarily to the high organic matter content of the soils and their ability to mineralize N. Under these conditions, elevated levels of NO_3^- will be lost to drainage water regardless of soil or nutrient management practices. (see also Chapter 9 section 3.4).

High (available) organic C contents in the soil increases the denitrification capacity (Bijay-Singh et al., 1988; Munch & Velthof, 2006). Denitrification decreases and leaching increases when the organic C content of the soil decreases, because denitrification capacity decreases when total C content of the soil decreases (Burford & Bremner, 1975; Bijay-Singh et al., 1988). Available organic C contents is strongly affected by land use. Since the organic matter content in grassland is generally (for all soil types and climate zones) higher than in arable land for all soil types, grasslands have a higher denitrification, thus a lower risk for N leaching than arable land.

7.3.3 Effect of pH

Nitrogen mobilisation and immobilisation largely depends on the biological activity of mineralizing, nitrifying and denitrifying bacteria. These activities are affected by pH. An overview of the effect of pH on the N availability through the processes mineralisation, nitrification, denitrification and crop uptake is presented in Table 8).

Table 8 The influence of pH on the N availability through the processes mineralisation, nitrification, and denitrification and crop uptake. When pH is increased, this may have a positive (+) or negative (-) on NH_4^+ availability, NO_3^- availability and the resulting N leaching.

Process	Effect of pH increase on process	NH_4^+ availability	NO_3^- availability	N leaching
Mineralisation	Higher mineralisation rate --> NH_4^+ production	+	0	+
Nitrification	+ autotrophic - heterotrophic	- +	+ -	+ -
Denitrification	Transformation of NO_3^- to N_2 , N_2O and NO	0	+	-
Crop uptake	Higher N uptake due to higher growth rate	-	-	-
+ = increase - = decrease 0 = no effect ? = unknown				

7.3.4 Effect of soil drainage and groundwater level

Drainage leading to a lowering of the ground water level may result in increased mineralisation, especially in peat soils, which in turn may result in an increase in N leaching.

From water table management experiments in Ontario, Canada, nitrate concentrations in drainage water from sub-surface irrigation, i.e., water delivered from below soil surface, were 74% and 80% lower than those from free drainage in 1997 and 1998, respectively. Water table management can effectively reduce NO_3^- pollution of water (Heathwaite et al., 1998). The effect of climate on subsurface runoff in a Minnesota, USA study on clay loam with maize showed that drainage was least in 1989 when growing season precipitation was 35% below normal and greatest in 1991 when growing season precipitation was 51% above normal (Gäbler et al., 2007).

7.4 Interaction between factors

The precipitation surplus in combination with soil texture is an indicator for the wetness of the soil. It is likely that leaching on sandy and loamy soils is highest when precipitation surplus is highest. However, a high precipitation surplus creates anaerobic soil conditions in peat and clay soils and therefore higher denitrification losses and lower leaching losses.

7.4.1 Concluding remarks

The analysis of climatic and soil properties affecting the transformation, retention and leaching and runoff of nitrate to groundwater and surface water showed that the effect of temperature is ambiguous since it may increase the nitrate availability

through enhanced mineralisation and nitrification, but it may also decrease the nitrate availability through enhanced denitrification and uptake. The intra annual temporal distribution of precipitation as well as dry and wet climatic cycles greatly affected N leaching. Soil texture influences drainage which influence the quantity of solute leaching. It also influences the oxygen contents, which controls the (de)nitrification and mineralisation processes. On a well-drained, sandy soil, NO_3^- leaching is higher than on poorly drained wet clay soil. Soils high in organic matter can mineralize a substantial amount of nitrate, which is susceptible to loss in subsurface tile drainage, especially when wet years follow very dry years. High contents of soil organic matter (available organic C) increases denitrification and decrease nitrate leaching. Soil pH may affect N leaching through its effect on mineralisation, nitrification, denitrification and uptake. This can be either positive or negative. Finally, drainage and lowering of the ground water table results in increased mineralisation, especially in peat soils, which in turn, may result in an increase in N leaching.

7.5 Selected factors to determine the surface runoff risk potential and the downward leaching risk potential of N

The results of the aforementioned criteria, the reviews presented in Part A and B of this report and the model simulation experiments have been synthesized into formulae, which allow the derivation of the surface runoff risk potential and nitrate leaching risk potential, based on a combination of land, soil and climate factors (i.e. pedo-climatic information). The formulae also provide the underpinning for the recommendations for the measures (1-12) in Annex II and Annex III of the Nitrates Directive, as function of pedo-climatic zones (see Part D). The formulae are discussed below.

As discussed in Part A, B and this report, a distinction has to be made between (i) surface runoff of N to surface waters, which leads directly to eutrophication, and (ii) downward leaching of nitrate-N, which directly leads to groundwater pollution by nitrates, but upon seepage of this groundwater may lead also to eutrophication of surface waters.

Re (i) Surface runoff is used here as a collective term for overland flow, run-off and shallow lateral flow. Eutrophication is understood as the enrichment of surface waters by phosphates and nitrates. These nutrients typically promote the growth of algae. As the algae die and decompose, high levels of organic matter and the decomposing organisms deplete the water of available oxygen, causing the death of other organisms, such as fish. Some algae are produce as well toxic gases, such as H_2S , during decomposition. Eutrophication is a natural process, but human activity greatly speeds up the process. Also, nitrate-rich surface waters may not be used as source for drinking water.

Re (II) Downward leaching of N from agricultural sources leads to groundwater pollution and the nitrate-rich groundwater aquifers may not be used as source for drinking water. Moreover, the nitrate-rich groundwater may seep into surface waters in lower lying areas after some time and thereby contribute to

the enrichment of these waters with nitrates. Hence, downward N leaching may lead to firstly pollution of groundwater and secondly to eutrophication of surface waters. The source-pathway-receptor linkage can be highly complex, also because the traveling pathway of the groundwater may be very lengthy, and the traveling time of the groundwater before it seeps into surface waters may take decades.

Hence, two general formulae are needed for the estimation of the risk of surface runoff and downward leaching (see Box 2 and Box 3 for details):

$$\text{Risk}_{\text{surface runoff}} = \text{LF}_{\text{surface runoff, max}} \times f_{\text{lu}} \times f_{\text{p}} \times f_{\text{rc}} \times f_{\text{s}} \quad [\text{Eq. 1}]$$

$$\text{Risk}_{\text{downward leaching}} = \text{LF}_{\text{soil type, max}} \times f_{\text{lu}} \times f_{\text{p}} \times f_{\text{r}} \times f_{\text{t}} \times f_{\text{c}} \quad [\text{Eq. 2}]$$

Risk (of nutrient leaching and runoff) is perceived here as a combination of (i) incidence of occurrence, i.e. frequency of leaching, and (ii) the amount of nutrients, i.e. the mass of nutrients in for example kg per ha of agricultural land where leaching occurs. Risks are termed high when both the incidence of occurrence and the amounts of lost nutrients are high. Risks are termed low when both the incidence of occurrence and the amounts of lost nutrients are low. Risks are termed intermediate when the incidence of occurrence is relatively high but the amounts of lost nutrients are low, or when the incidence of occurrence is low but the amounts of lost nutrients are high.

Box 2. Deriving formulae for the risk of surface runoff

Surface runoff occurs when rainfall exceeds a maximum infiltration level of the soil. Factors controlling surface-runoff are the slope, weather conditions (precipitation, frost), the type, period and amount of N application and properties of the soil. Commonly, the leaching fraction for surface runoff is expressed as percentage of the N applied as fertilizer and manure. Reported surface runoff losses ranged from 0 to 35 kg N ha⁻¹ yr⁻¹ and from 0 to 3 kg P ha⁻¹ yr⁻¹ (Wu et al., 1997; Heathwaite et al., 1998; Korsath & Eltun, 2000). Studies of Heathwaite et al. (1998), Scholefield and Stone (1995) and Smith et al. (2001) showed that runoff from sloping soils were generally around than 10% of the N and P applied as fertilizer or manure. The factors that most strongly control surface runoff are included in formulae, i.e. the slope, the precipitation, soil type, crop type and the depth to rock (Sharpley, 1985; Eltun & Fugleberg, 1996a, b; Wu et al., 1997; Heathwaite et al., 1998; Velthof et al., 2009).

Here, the risk potential of surface runoff is estimated a maximal surface runoff, and a set of reduction factors:

$$\text{Risk}_{\text{surface runoff}} = \text{LF}_{\text{surface runoff, max}} \times f_{\text{lu}} \times f_{\text{p}} \times f_{\text{rc}} \times f_{\text{s}}$$

where

$\text{LF}_{\text{surface runoff}}$ is the surface runoff fraction in % of the N and P applied via fertilizer and manure,

$\text{LF}_{\text{surface runoff, max}}$ is the maximum surface runoff fraction for different slope classes,

f_{lu} is the reduction factor for land use or crop,

f_{p} is the reduction factor for precipitation,

f_{s} is the reduction factor for soil type, and

f_{rc} is the reduction factor for depth to rock.

The surface runoff fractions and reduction factors are presented in Table 9. The reduction factors are based on literature information. The maximum surface runoff fractions range from 10% for flat areas to 50% for steep slopes. It is assumed that the smallest surface runoff occurs in grasslands, because there is no tillage and because the grassland soil is covered the whole year by a crop. The precipitation surplus (i.e. the precipitation minus the evapotranspiration) is chosen as an indicator of rainfall effect on surface runoff. Soil type affects surface runoff, i.e. risk on surface runoff increases when clay content of the soil increases. The risk on surface runoff is higher in shallow than in deep soils, because the infiltration of water is more rapidly hampered in shallow soils overlying solid rock. The depth to rock was therefore included in the formulae to calculate surface runoff.

The potential surface runoff can be calculated as: $\text{Risk}_{\text{surface runoff}} \times \text{the applied amounts of fertilizer and manure N and P}$.

Table 9. Surface runoff fractions and reduction factors for surface runoff (Source: MITERRA-Europe; Velthof et al., 2009).

Parameter	Description	Value
$\text{LF}_{\text{surface runoff, max}}$	maximum surface runoff fraction for slope classes, % of the N applied via fertilizer and manure (including grazing)	0% for a slope of < 2% 10% for a slope of 2-8% 20% for a slope of 8-15% 35% for a slope of 15-25% 50% for a slope of > 25%
f_{lu}	reduction factor for land use	0.25 for grassland and 1.00 for other land use
f_{p}	reduction factor for precipitation surplus	1 for precipitation surplus > 300 mm 0.75 for 100-300 mm 0.50 for 50-100 mm 0.25 for < 50 mm
f_{s}	reduction factor for soil type	1 for a clay content of > 60 % 0.9 for a clay content of 35 - 60% 0.75 for a clay content of 18 - 35% 0.25 for a clay content of < 18% 0.25 for peat soils
f_{rc}	reduction factor for depth to rock	1 for a soil depth of $\leq$ 25 cm depth 0.8 for a soil depth of > 25 cm depth

Table 10. *Leaching fractions and reduction factors for downward leaching (Source: MITERRA-Europe; Velthof et al., 2009).*

Parameter	Description	Value
LFsoil type, max	maximum leaching fraction, in % of the N surplus corrected for NH ₃ emission and surface runoff	100% for sandy soils 75% for loamy soils 50% for clay soils 25% for peat soils
flu	reduction factor for land use	0.36 for grassland 1.00 for other land use
fc	reduction factor for soil organic content	1 for total C content of < 1% 0.90 for total C content of 1-2% 0.75 for total C content of 2-5% 0.50 for total C content > 5%
fp	reduction factor for precipitation surplus	for sand a loam: 1 for precipitation surplus of > 300 mm 0.75 for a surplus of 100 - 300mm 0.50 for a surplus of 50 - 100mm 0.25 for a surplus of < 50 mm for peat and clay: 0.5 for a precipitation surplus of > 300 mm 1.0 for a surplus of 100 - 300mm 0.75 for a surplus of 50 - 100mm 0.25 for a surplus of < 50 mm
ft	reduction factor for temperature	1 for a temperature of < 5 °C 0.75 for a temperature of 5-15 °C 0.50 for a temperature of > 15 °C
fr	reduction factor for rooting depth	1 for a rooting depth of < 40 cm 0.75 for a rooting depth of > 60 cm.

Box 3. Deriving formulae for the risk of downward leaching

Downward leaching occurs when rainfall exceeds evapotranspiration and the surplus rains infiltrates into the soil. Factors controlling downward N leaching are (i) rainfall surplus, (ii) N surplus, (iii) nitrate removal through denitrification, and (iv) nitrate formation through net N mineralization and nitrification. The leaching can be estimated at various soil depths, both in the unsaturated zone and saturated zone (i.e. the groundwater aquifer). Common depths below the root zone are 1, 5 m and 10 m below the soil surface. Common depths within the aquifers include the upper meter of the aquifer and at 5, 10 and 50 m within the aquifers.

Commonly, the leaching fraction for downward leaching is expressed as percentage of the N surplus. Reported leaching percentages range from 5 to 100% of the N surplus, depending on soil type and soil depth and soil drainage class (which affect the rate of denitrification). Reported leaching rates range from 0 to >100 kg N ha⁻¹ yr⁻¹, and mainly depends on soil type and N surplus. The downward leaching of P is usually negligible, because of the strong sorption of P to the soil matrix. Therefore, downward leaching of P is not discussed further.

Denitrification is the microbial reduction of NO₃ to gaseous NO, N₂O and N₂, and is an important factor controlling NO₃ leaching. Factors controlling denitrification include the presence of i) an energy source for the denitrifying bacteria, mostly metabolizable organic carbon, ii) anoxic conditions, and iii) presence of nitrate (Tiedje, 1988b). If any of these conditions is not fulfilled, denitrification is unlikely. There is often no quantitative information available about the presence of metabolizable organic C, anoxic conditions and NO₃ in soil, and, as a consequence, proxy indicators have to be used. Land use and soil organic C are proxies for metabolizable C, precipitation, soil type and rooting depth are proxies for the presence of anoxic conditions, and the N surplus is a proxy for the NO₃ content in the soil.

The risk of downward leaching is estimated from a maximal leaching fraction, and a set of reduction factors:

$$\text{Risk}_{\text{downward leaching}} = \text{LF}_{\text{soil type, max}} \times \text{flu} \times \text{fp} \times \text{fr} \times \text{ft} \times \text{fc}$$

where

LF_{soil type, max} is the maximum leaching fraction in % of N surplus per soil type, assuming that soil type is the major factor controlling the ratio between leaching and denitrification

f_{lu} is the reduction factor for land use or crop

f_p is the reduction factor for precipitation surplus

f_r is the reduction factor for rooting depth,

f_t is the reduction factor for soil temperature, and

f_c is the reduction factor for soil organic carbon.

The maximum leaching fractions (Table 10) are derived from empirical leaching data (Schröder et al., 2007). The maximum leaching fractions were corrected for effects of land use (grassland versus arable land). Land use has a strong effect on available organic C contents in the soil and thereby on the denitrification capacity (Bijay-Singh et al., 1988; Munch & Velthof, 2006). The reduction factor for land use (f_{lu}) is set at and 0.36 for grassland and 1.00 for other land use. It was assumed that denitrification decreases (and leaching fraction increases) when the organic C content of the soil decreases (Burford & Bremner, 1975; Bijay-Singh et al., 1988). The precipitation surplus in combination with soil texture is an indicator for the wetness of the soil. The reduction factor for the precipitation surplus (f_p) is dependent on soil type. It is assumed that leaching on sandy and loamy soils is highest when precipitation surplus is highest. However, a high precipitation surplus creates anaerobic soil conditions in peat and clay soils and therefore higher denitrification losses and lower leaching losses. Therefore, a smaller leaching fraction is assumed for peat and clay at a high rainfall surplus (Table 8). Denitrification is assumed to be two times higher at a mean annual temperature of 15 °C than at 5 °C. Further, the risk of N leaching increases when the rooting depth decreases. Soils with a very shallow rooting depth are more susceptible for leaching than soil with a deep rooting depth, which is expressed in the reduction factor f_r.

The potential downward leaching can be calculated as the product of the risk of downward leaching and the N surplus: $\text{Risk}_{\text{downward leaching}} \times \text{Nsurplus}$

Nsurplus is the difference between total N input, corrected for surface runoff and NH_3 emissions, and total N output via harvested crop.

8 Factors influencing P leaching and runoff

8.1 Introduction

The risk of P transfer to water bodies in catchments with significant slopes is adequately predicted based on soil vulnerability to erosion-P losses, in other systems the risk related to dissolved P losses, particularly in excessively P-rich, flat soils. The fate of P leaching and runoff is also be significantly variable in soils of different slope positions of a landscape (see Figure 30). Another important factor is the occurrence of preferential flow or by-pass flow. As for N, the loading of P into surface water and groundwater is a function of transport volume (amount of water) and P concentration in the transported water. The amount of drainage water leaving the landscape is largely a function of climate and soil properties. Contrary to N, the concentration of P in the soil is largely determined by P sorption. Since the effects of climatic factors on P leaching are very similar to those on N leaching, we concentrate this chapter to the effect of pedological factors. The principal pedological factors determining P leaching and runoff are the factors that influencing the retention of P in the soil. The risk of a soil to leach phosphorus is related to the combination of the capacity to store phosphorus and the amount of phosphorus that accumulates in soil profiles.

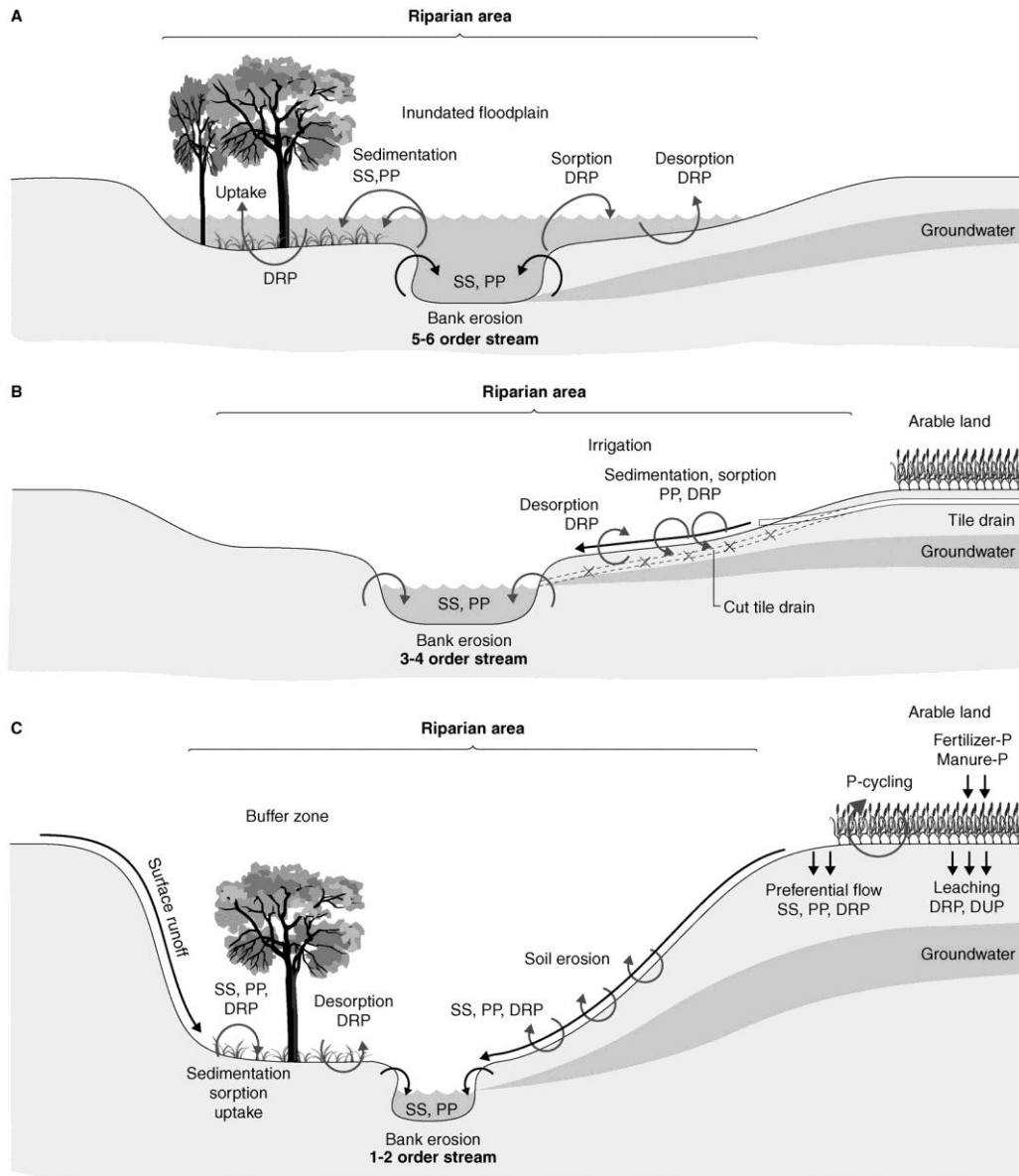


Figure 30 Schematic diagram of important P pathways and processes taking place along the border of agricultural fields and riparian areas. (A) Downstream river under natural conditions the river often inundates the floodplain. (B) Middle part where tile drainage pipes often penetrate the riparian areas with water and substances from upland agricultural fields, thus short-cutting the biogeochemical processes in riparian areas. (C) Most upstream agricultural fields bordering the streams are often steep, resulting in both soil erosion and surface runoff and tillage erosion transport soil and P toward low-lying areas (Source: Burford & Bremner, 1975).

8.2 Effect of climate factors

Phosphorus loading from land to streams is expected to increase in northern temperate coastal regions due to higher winter rainfall and to decline in temperature (Eltun & Fugleberg, 1996a). Based on model results Jeppesen et al. (1996a) suggest a 3.3 to 16.5% increase within the next 100 yr in the P loading of Danish streams depending on soil type and region.

8.3 Effect of pedological factors

8.3.1 P sorption and P saturated soils

Phosphorus sorption is strongly controlled by the amount and nature of some soil constituents such as clay minerals, organic substances, iron (Fe) and (Al) oxides and carbonates. Different parameters related to soil P sorption capacity or to measurements of labile P pools have been proposed to assess the effect of soil P accumulation on the P desorption and on the consequent risk of P loss in surface and subsurface pathways. In general, the P sorption capacity of soils for P is limited, and, as a result, P can be transported to ground water through leaching, especially in flat areas with a high ground water level (Sims et al., 1998; Schoumans & Groenendijk, 2000; Misselbrook et al., 2005).

From research in the Netherlands, it appeared that the P concentration in the upper ground water is likely to exceed the national surface water limit of 0.15 mg total P l⁻¹. An estimated an area of about 400,000 ha in the Netherlands (Breeuwsma et al., 1997), can be considered as P saturated soils. These soils have the potential to contribute to P enrichment of surface waters through subsurface runoff.

Several European countries and several states in the US use a “phosphorus index” (PI), which is a semi-quantitative tool to assess the risk of P loss from fields to surface waters. The PI is generally based on simple mass balance of the amount of P in the soil (based on soil tests), P applications as manure and fertilizer, plant residues and the losses due to transport factors such as erosion, surface runoff and leaching (Buczko & Kuchenbuch, 2007).

8.3.2 The role of aluminium and iron hydroxides

The total sorption capacity of inorganic P is largely related to the amount of amorphous Al and Fe (hydr)oxides, which are the main reactive solid phases (Beek et al., 1977a, b; Van Riemsdijk & de Haan, 1981). The overall reaction of inorganic P with Al- and Fe-(hydr)oxides is the result of a fast reversible adsorption reaction at surface sites (e.g. <1 d) and a slow one, that is, diffusion through the solid phase or through micropores of Al and Fe (hydr)oxides possibly followed by precipitation and/or adsorption inside the aggregates (Van Riemsdijk & de Haan, 1981). The total pool of sorbed P (F) in noncalcareous sandy soils has been interpreted to be the sum of reversibly adsorbed P (Q) and quasi-irreversibly bound P (S), that is, $F = Q + S$ (Schoumans & Groenendijk, 2000).

The P sorption capacity of soils for P is may be influenced through changes in drainage and determined by the ground water level; at higher ground water levels, the sorption capacity is relatively low due to the reduction of Fe (oxy)hydroxides. This is also a main reason why sediments release dissolved P to the overlying surface waters. Consequently, oxidation-reduction status (Redox, Eh) as well as the pH are important factors in poorly drained soils.

8.4 Concluding remarks

The concentration of dissolved and plant-available P in the soil is largely determined by P sorption. The sorption capacity of soils for P is determined by the characteristics of the soil, i.e., clay mineralogy, Al and Fe hydroxides, organic matter, pH, oxidation-reduction status (Redox, Eh), ground water level. At high ground water level, the sorption capacity is low, because of the reduction of Fe(III) (hydr)oxides. The effects of climatic factors on P leaching are very similar to those on N leaching. All factors related to the runoff of N also apply to P. However, since the water solubility of P is much less compared to N, leaching of P is of less importance. The opposite, however, is true for P losses by erosion. It is estimated that for the European continent the amount of eroded P is ~80% of the amount of P fertilizer use.

9 Factors influencing eutrophication of surface waters

9.1 Introduction

Eutrophication is the excessive enrichment of surface waters with nutrients and the resulting adverse biological effects. These nutrients typically promote the growth of algae and reduce the transparency of the water. As the algae die, decomposing organisms deplete the water of available oxygen, causing the death of other organisms, such as fish. Further, some algal blooms may produce toxins, which influences the taste and odour of the water, and thereby make the water unsuitable for use as drinking water and recreation (EEA, 2005). Some algae species produce toxic gases while decomposing, which are lethal to humans and animals. Eutrophication is a natural process, but human activity greatly speeds up the process.

Eutrophication is caused by (natural and anthropogenic) inputs of nutrients, mainly nitrogen (N) and phosphorus (P), to the aquatic environment. This can either be caused by surface and subsurface runoff of N and P to surface waters or downward leaching (see Chapter 5).

An important pathway of nutrients from agriculture to surface waters is via surface runoff, including overland flow, runoff and shallow lateral flow.

Although downward leaching of N from agricultural sources, directly leads to groundwater pollution by nitrates, indirectly it may also lead to eutrophication. The nitrate-rich groundwater may seep into surface waters in lower lying areas after some time and thereby contribute to the enrichment of these waters with nitrates. Hence, downward N leaching may lead to firstly pollution of groundwater and secondly eutrophication of surface waters. The source-pathway-receptor linkage can be highly complex, also because the traveling pathway of the groundwater may be very lengthy, and the traveling time of the groundwater before it seeps into surface waters may take decades.

9.2 Limiting nutrient

In most fresh waters P is the limiting nutrient for eutrophication, whereas most limnologists consider N to be the limiting nutrient in coastal and marine waters (Howarth & Marino, 2006). By the middle to the end of the 1970s a strong consensus had developed that P caused eutrophication in lakes strongly based upon experiments by Schindler (Schindler et al., 1978) and Vollenweider (1975; 1976). The primary reason for P limitation of freshwater streams, lakes, reservoirs, and rivers is that these systems are generally supported by large watershed areas, which capture, accumulate, and mobilize relatively large amounts of biologically available N relative to P (Peierls et al., 1991; Wetzel, 2001). Moreover, a number of microorganisms have the capability to fixate atmospheric N₂, thereby overcoming any N-limitation.

In order to predict the effects of N and P inputs on receiving waters, water body nutrient concentrations dependency on the external N and P inputs must be assessed. Therefore quantitative conceptual models that relate external nutrient inputs to the resulting water column concentrations of nutrients in the water body have been developed, which started with the mass-balance approach by Vollenweider (1968). In the 30 years since the publication of Vollenweider's report, however, a rigorous quantitative framework has developed to predict the responses of freshwater lakes and reservoirs to eutrophication. Essential to this framework are the calculations of mass budgets for both N and P and the parallel calculation of a hydraulic budget for the water body (Smith et al., 1999).

9.3 Effect of climatic conditions

There are many reports about climate change and eutrophication. However, the number of papers referring to the effect of climate on eutrophication is scarce. Papers dealing with climate change effects can be used to indicate the effect of climate on eutrophication. For instance, it is assumed that climate change will be associated with a change in rainfall patterns and a more intense precipitation, which in turn will increase surface and groundwater nutrient discharge into water bodies. Moreover, an ecosystem shift may occur when the temperature changes, since many organisms react upon changes in temperature regarding their productivity (Jeppesen et al., 2010).

Ecosystem structures in warm climates differ from those in more temperate regions. For that reason, it seems likely that they also respond differently to changes in nutrient supply, however, responses are complex and may vary depending on the specific context (Meerhoff et al., 2007).

So, nutrient loading and changes therein, may affect ecosystems differently in different climatic areas, but it is very complex to give a general description and predict the exact effects.

9.4 Methods to assess eutrophication

Many different methods to establish the level of eutrophication exists, however in most of them N, P or a combination of these two elements play an essential role. In this paragraph a number of these methods will be described briefly, without labelling them qualitatively. Currently, there is no consensus among ecologists with regarding the use of one specific method, although ecologists within the EU are trying to harmonize towards a generally accepted methodology. But still, many Member States are using their own system to assess eutrophication. Even if they are often based on the same parameters, the interpretation of the results may differ per Member State. A large part of the information below is from the Guidance Document on Eutrophication Assessment (EC, 2009).

9.4.1 Fresh waters

Lakes

The assessment of the degree of eutrophication in lakes is primarily determined through the application of nutrient (phosphorus and nitrogen) concentration criteria supplemented with the use of criteria for indicators of direct effects of eutrophication, e.g. chlorophyll a and Secchi depth. In addition, the occurrence of algal blooms and the occurrence of certain algal species may be used as an indicator.

Many countries are using the OECD classification scheme based on total P, annual chlorophyll-a concentrations and the clarity of the water. Some countries are using only part of the OECD scheme.

Rivers

Many countries within the EU use the French Seq-eaux system for rivers

However, a number of Member States are using different schemes. The Trophic status in Irish rivers for instance is assessed using both biological responses and supporting physico-chemical elements especially those indicative of organic enrichment such as phosphate, nitrate, ammonia, dissolved oxygen and BOD. The EPA Quality Rating System (macroinvertebrates) and diatom index have been intercalibrated with other European countries in the formal Water Framework Directive Intercalibration process.

In conclusion, many Member States are using a system which are based upon common principles, using P, chlorophyll-a and water clarity, however which differ regarding the final interpretation.

9.4.2 Estuarial, coastal and marine waters

As for fresh waters, also different systems exist within the EU to establish eutrophication in estuarial, coastal and marine waters. Partly these differences are explained by regional aspects. The water quality in the North Sea, for instance, is affected largely by other seas (the Atlantic Ocean), while such effect are far smaller in e.g. the Baltic Sea and the Black Sea.

The major approaches to establish eutrophication in the marine environment are:

The Helcom approach

The HELCOM assessments are based on an integration of the results from core set indicators on nutrient (nitrogen and phosphorus) concentrations, chlorophyll a concentrations, water transparency and zoobenthos communities using the HELCOM Eutrophication Assessment Tool (HEAT, Pyhälä et al., 2009). Chlorophyll-a, Secchi depth and oxygen concentration as important parameters. Figure 31 is an example of the results showing the transparency situation in the Baltic Sea during 2001-2006.

The HELCOM approach has also been used for the Black Sea area under the Istanbul convention

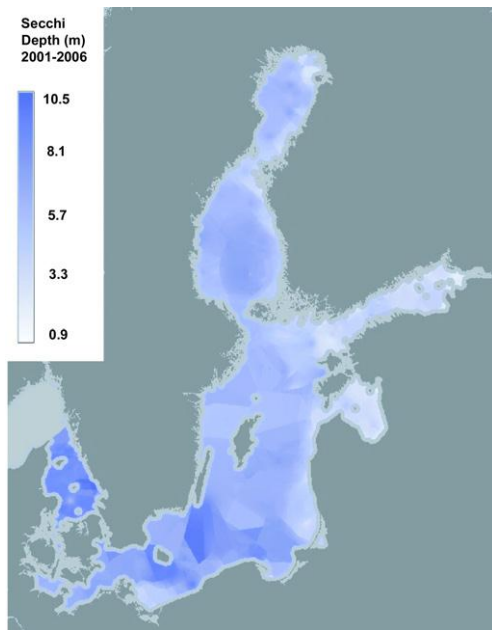


Figure 31 Map showing the water transparency status in 2001-2006 in the Baltic Sea area (HELCOM, 2009).

The OSPAR approach

The OSPAR approach is more or less comparable to the HELCOM approach. OSPAR is using a Comprehensive Procedure, i.e. a tool through with the integrated suite of Ecological Quality Objectives (EcoQO) for eutrophication for the North Sea is implemented, based upon five EcoQO's and a number of harmonized assessment parameters and associated assessment levels

The MEDPOL approach

EEA has developed an approach to establish eutrophication in the Mediterranean which is depicted in Table 11.

Table 11 EEA eutrophication assessment scheme for MEDPOL (UNEP, 2007).

Headline Indicator	EEA Indicator theme	General description	Indicator parameters	Comments
	Nutrients	Concentrations of substances	winter conc. NO ₂ + NO ₃ , PO ₄ , N/P ratio, Chlorophyll a Bottom oxygen TRIX (Trophic Index) = (Log(Chla*aD%O*DIN*TP)+ 1.5)/1.2	Mean summer tot-P, tot-N, inorg-N forms, PO ₄ Spring peaks, seasonal Mean, annual minimum duration of low oxygen values TRIX index allows for spatial information using SeaWifs images

Chla = chlorophyll-a concentration in mg/L; aD%O = as absolute deviation from saturation in %,
DIN = Dissolved inorganic N in µg/L; TP = Total phosphorus in µg/L

TRIX is distinguishing 4 different classes of water quality. TRIX could possibly also be useful in other marine areas (Vascetta et al., 2004)

Other approaches

Next to the approaches described above, also other assessment schemes are used by member states. Ireland for instance, uses the Trophic Status Assessment Scheme (TSAS, Anonymous, 2008), in which the assessment of ecological status is calculated on a basis of multiple criteria.

9.5 Eutrophication and pedo-climatic zones

Is there a relation between eutrophication and pedo-climatic zone stratification? The answer is probably positive with respect to the vulnerability risk of eutrophication. A high risk of eutrophication exists probably in zones with a high risk of nutrient leaching and runoff. Thus, areas with a high precipitation, heavy soils and relatively steep slopes in combination with shallow soils are most likely subject to an increased risk of eutrophication.

Areas with soils rich in nutrients by nature (see above) will have a higher risk of eutrophication than areas with soils low in nutrients. Also, areas with high precipitation rates will probably show a higher weathering rate leading to a higher runoff risk.

However, in the areas mentioned above, the risk of eutrophication has a natural background. One may wonder if agricultural measures to reduce the risk of eutrophication will sort any effect. Theoretically the combinations as shown in Table 12 exist. The overall vulnerability risk is depending on combination of natural and agricultural risks. The effect of measures in agriculture to reduce the risk of

eutrophication, probably will have an effect in only one of the four possible combinations (Table 12).

Table 12 Vulnerability risk of eutrophication under 'natural' and 'agricultural' conditions and the reducing effect of measures in agricultural management

Natural risk	Agricultural risk	Overall risk	Effect of reducing measures in agriculture
Low	Low	Low	n.a.
Low	High	High	Large
High	Low	High	Small
High	High	Very high	Large ¹⁾

¹⁾ In situations with very high natural inputs, such as the fen meadow area in the Netherlands, the effect of reducing measures is low.

9.6 Concluding remarks

Eutrophication is an important ecological phenomenon caused by an excess of nutrients from a natural or anthropogenic origin. Of these, anthropogenic sources are far more important than natural sources and agriculture is the predominant anthropogenic source.

Many different eutrophication assessments methods exist for both fresh and marine waters. However, most of them are based upon common principles, including chemical and biological parameters. The outcome of different systems may, however, differ due the final interpretation of the results.

There is probably a relation between eutrophication and pedo-climatic zone stratification, but this relation is probably stronger for the natural risk of eutrophication than for the risk caused by agriculture, although also then precipitation and soil properties will be important factors.

The risk of eutrophication caused by agriculture may be reduced by taking measures regarding farm management and fertilization. In areas where the risk of 'natural' eutrophication is high, measures to reduce the agricultural risk probably have only a small effect. This of course not necessarily means that such measures should not be taken.

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