Mitigation options for reducing nutrient emissions from agriculture

A study amongst European member states of Cost action 869

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Abstract

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The Water Framework Directive (WFD) requires improvement to the quality of surface water and groundwater. In the past many measures have been implemented to reduce the contribution of point sources, and as a result diffuse pollution from agricultural became more important. The main objective of COST Action 869 is to undertake a scientific evaluation of the suitability and cost-effectiveness of different options for reducing nutrient loss to surface and ground waters at the river basin scale, including their limitations in terms of applicability under different climatic, ecological and geographical conditions. In this report an overview is given of different categories of mitigation options and the individual measures has been described in terms of the mechanism, applicability, effectiveness, time frame, environmental side effects and cost in order to help policy makers, watershed managers and farmers to select the most relevant measures for their conditions.

Keywords: Measures, nitrogen, nutrients, phosphorus Water Framework Directive

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Alterra-Report 2141
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Contents

Preface

In 2006 a new COST Action was funded by the EC. This COST Action (869) focuses on the steps that need to be taken within the WFD in order to effectively reduce nitrogen (N) and phosphorus (P) losses from agricultural land to surface waters and groundwater. A total of 29 countries are involved in COST Action 869.

The COST Action 869 integrates different aspects of reducing nutrient losses: to localise critical source areas in catchments, where mitigation actions are most likely to be environmentally cost-effective (Working Group 1); to study the influence of nutrients on ecological processes in surface waters and the influence of ecology on the choice of mitigation options (Working Group 2); to evaluate for various mitigation options the cost-effectiveness, implementation aspects, and the influence of scale, climate and other physical factors (Working Group 3); and to evaluate ongoing mitigation projects in example areas across the EU (Working Group 4).

The funding of the EU concerns the travel reimbursement of the meetings organised within a COST Action. The daily work load of the scientists to collect information and writing the reports and the factsheets are not paid. From this point of view the authors want to thank all scientists who have contributed to collect information on the applicability of mitigation options under different circumstances, for the improvement of the factsheets and this document (see list below). And of course for all great discussions we had during our meetings in Devon (2007), Rome (2008) and Wageningen (2009).

This report is meant for national and regional governments, water managers, intermediaries and innovative farmers to help them to select proper measures to reduce the nutrient losses to our environment. The detail factsheets of mitigation options can be found on the website and will be improved based on new information from field experiments. We encourage all researchers to send information to improve the factsheets further in the future.

Together, with the information in this report, the factsheets, the help of the local expertise of agro- and environmental (research) institute and advisory services in your country, it should be possible to implement suitable and effective measures in the near future.

We hope that with this overview of mitigation options a step forwards is made to improve our water quality and that agriculture practices still may remain possible, because we also have to guarantee permanent food supply.

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Summary

The eutrophication of surface waters and the contamination of groundwater and surface waters due to elevated nutrient inputs have a serious impact on ecosystem health in many countries. The Water Framework Directive (WFD) requires improvement to the quality of surface water and groundwater. This may require drastic reduction in nutrient loss from agricultural land, with possible implications for the long term economic and environmental sustainability of agricultural systems. In addition, the situation of enclosed coastal waters (e.g. Baltic Sea, North Sea, and Mediterranean Sea) is of special concern for several countries in Europe. The eutrophic state of this brackish water has ended up in action plans that have to be implemented. The main objective of COST Action 869 is to undertake a scientific evaluation of the suitability and cost-effectiveness of different options for reducing nutrient loss to groundwater and surface water at the river basin scale, including their limitations in terms of applicability under different climatic, ecological and geographical conditions.

In this report the results are discussed of different potential measures to reduce nutrient losses from agricultural land, and of the impact of nutrient losses on surface water. The measures were based on an inventory among all participating countries of this COST Action 869. A total of more than 100 measures were suggested. Some of them could be combined and in the end 80 different measures were distinguished. For each measure a factsheet was written with a general description of the measure, and information on rationale/mechanism, applicability, effectiveness including uncertainty, time frame, environmental side-effects / pollution swapping, potential for targeting, cost in terms of investments and labour needs, and references.

The list of measures could be merged into eight categories: (a) nutrient management, (b) crop management, (c) livestock management, (d) soil management, (e) water management, (f) land use change, (g) landscape management and (h) surface water management. In this report each category is discussed from a general point of view, giving the main outline and possibilities of the underlying measures. In the Appendix all separate measures are described in a similar way. This information is a first attempt to collect information from all over Europe with respect to the potential of different mitigation options. Because COST action 869 provides only money for travel costs of meetings, work on gathering information on the measures and writing this report had to be done in the spare hours of many enthusiastic scientists. Therefore, still important steps have to be taken in order to collect missing information and to improve the reliability of the estimated effectiveness of mitigation measures under different circumstances within Europe. However, we hope that policy makers, watershed managers and farmers can use this information, in order to select relevant mitigation options applicable under their circumstances.

1 Introduction to COST Action 869

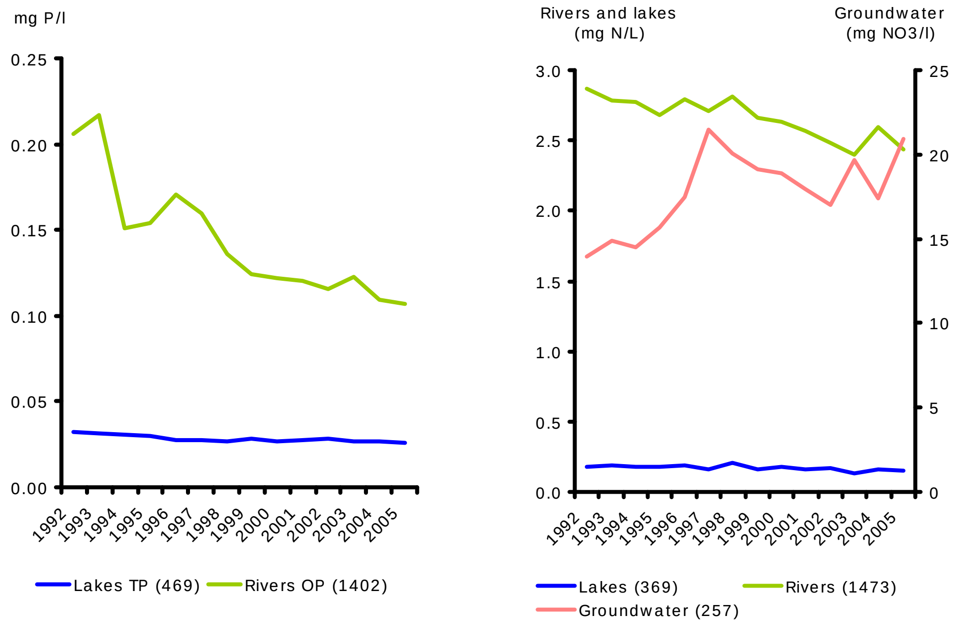
O.F. Schoumans and W.J. Chardon

The role of nutrients in the eutrophication of surface water and the contamination of groundwater and surface waters has long been recognised. Negative effects of eutrophication include: reduced biodiversity of aquatic ecosystems and surface water quality, algal blooms (sessile and planktonic; some are toxic) that restrict the use of surface waters for recreation, and excess nitrate concentrations having an impact on drinking water production. Toxic algal substances have caused fish kills and animal and human diseases in the past. Nitrogen (N) and phosphorus (P) are the elements that often determine the ecological status in most European inland waters.

To date, European-wide efforts have reduced the nutrient problem by a combination of accidental (e.g. diminishing industrial losses through industrial decline) and targeted options (e.g. increasing the number of households that are connected with sewerage systems in order to increase the denitrification of nitrogen and precipitating P in sewage water treatment plants, purification of the waste water from industries etc.). There are a number of EU Directives aimed at reducing the loads and impacts of organic matter and nutrients. These include: the Nitrates Directive (91/676/EEC) aimed at reducing nitrate pollution from agricultural land; Urban Waste Water Treatment Directive (91/271/EEC) aimed at reducing pollution from sewage treatment works and from certain industries; Integrated Pollution Prevention and Control Directive (96/61/EEC) aimed at controlling and preventing pollution of water from industry; and the Water Framework Directive (2000/60/EC) which requires the achievement of good ecological status or good ecological potential of rivers across the EU by 2015. Despite the reductions due to implementation of the EC Directives, water quality status remains poor in many rivers, lakes and estuaries. Since 1990, the nitrate concentration in major EU Rivers has been about constant and the phosphorus concentration has declined (Figure 1.1).

The decline in phosphorus concentrations in major EU-rivers is mainly due to improved wastewater treatment and a reduction in the amount of phosphorus in household detergents over this period (Figure 1.1). Natural ranges are considered to be approximately 0 to 10 µg P/l and are mainly still observed in Northern European countries, like Finland and Sweden (http://www.eea.europa.eu/data-and-maps/indicators/nitrogen-and-phosphorus-in-rivers).

Nitrate concentrations in Europe's groundwater increased in the first half of 1990s and have then remained relatively constant. The average nitrate concentration in European rivers has decreased approximately 10 % since 1998 from 2.8 to 2.5 mg N/l, reflecting the effect of measures to reduce agricultural inputs of nitrate. Nitrate levels in lakes are in general much lower than in rivers, but also in lakes there has been a 15 % reduction in the average nitrate concentration. Concentrations of nitrate below 0.3 mg N/l are considered to be natural or background levels for most European rivers though for some rivers levels of up to 1 mg N/l are reported (EEA, 2009). Concentrations of nitrate above 7.5 mg N/l are considered to be of relatively poor quality and exceed the guideline concentration for nitrate of 5.6 mg N/l as given in the Surface Water for Drinking Directive (75/440/EEC).



(Source: EEA, http://www.eea.europa.eu/data-and-maps/indicators/nutrients-in-freshwater/nutrients-in-freshwater-assessment-published-1)

Figure 1.1
Phosphorus and nitrogen in major EU Rivers and lakes

In large rivers the effects of nutrient management will improve the water quality at long term due to delays and dissolution of the nutrients losses from agricultural land. In small catchments with agricultural practices, nutrient management seems to be effective in reducing nitrate concentrations in surface water at the short term as shown for Flanders (Table 1.1).

Table 1.1
Evolution of the mean nitrate concentration in surface waters and of the percentatge of measurements exceeding at least once the limit of 50 mg NO3/l in agricultural areas of Flanders (VLM, 2009)

winter year (July-June)	1999 2000	2000 2001	2001 2002	2002 2003	2003 2004	2004 2005	2005 2006	2006 2007	2007 2008	2008 2009
number of measured locations	254	254	268	749	779	788	784	782	786	788
measurements per year	3075	3647	3217	6902	7624	7392	6432	6585	6945	6203
mean nitrate concentration (mg NO_3/l)	35.8	32.0	28.3	25.1	24.8	26.4	27.2	26.6	25.7	21.1
% exceedance	59	50	41	31	42	40	41	43	37	27

Within Europe, the Water Framework Directive (Directive 2000/60/EC) will force catchment management authorities to improve the ecological status of both surface waters and groundwater. Since both P and nitrogen (N) losses to surface and groundwater are largely driven by agriculture, there is an urgent need to determine the relationship between agriculture and chemical and ecological water quality, because the River Basin Management Plans (RBMP first 2009, second 2015, and third 2021) should only implement cost-effective mitigation options. In many European countries no mitigation options for agriculture have been implemented in the first RBMP (minutes of COST workshop WG3 May 2009). From this point of view there is still an urgent need for information on the effectiveness of measures in rural areas since it is expected that in 2015 additional measures have to be taken to improve the water quality.

In 2006 a COST (**CO**operation in **S**cience and **T**echnology) Action was funded by the EU. This COST action (869) focuses on the steps that need to be taken within the WFD in order to effectively reduce the N and P losses from agricultural land to surface waters and groundwater. The Action is undertaken in the context of balancing measures to reduce P losses from agricultural land with those necessary to reduce other nutrient losses such as N. Such measures often conflict, and need to be considered as part of an integrated programme of measures. The general objectives of COST Action 869 are to:

- Determine the techniques / tools that can be used to determine the main P and N sources within the agricultural system as a whole, that contribute to the nutrient losses to surface waters and groundwater, and also the main pathways.
- Determine the techniques / tools that can be used for evaluating the impact of a reduction of the nutrient input on the ecological status of surface waters.
- Evaluate different types of integrated mitigation options.
- Evaluate implementation strategies for different types of basins / catchments.

Mitigation options (tested options as well as potential new options) were formulated and discussed. This Action brought together the current expert knowledge base. The focus of COST 869 Action is on nutrient losses from agricultural land. The emphasis of this COST action is on improving the water quality of groundwater and surface water with River Basin Management Plans. Since in fresh water systems phosphorus is most of the time the limiting nutrient, the main focus is on phosphorus, but the positive or negative influences of mitigation options on the loss of fine sediment and nitrogen to either surface water or other environmental compartments is also discussed.

So, COST Action 869 integrates different aspects of reducing nutrient losses: to localise critical source areas in catchments, where mitigation actions are most likely to be environmentally, socially and economically cost-effective (WG1); to study the influence of nutrients on ecological processes in surface waters and the influence of ecology on the choice of mitigation options (WG2); to evaluate for various mitigation options the cost-effectiveness, implementation aspects, and the influence of scale, climate and other physical factors (WG3), and to evaluate ongoing mitigation projects in example areas (WG4).

References

EEA, http://www.eea.europa.eu/data-and-maps/indicators/nitrogen-and-phosphorus-in-rivers

EEC, 1975. COUNCIL DIRECTIVE 75/440/EEC of 16 June 1975 concerning the quality required of surface water intended for the abstraction of drinking water in the Member States

EEC, 1991a. COUNCIL DIRECTIVE 91/271/EEC of 21 May 1991 concerning Urban Waste Water Treatment.

EEC, 1991b. COUNCIL DIRECTIVE 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources.

EEC, 1996. COUNCIL DIRECTIVE 96/61/EC of 24 September 1996 concerning integrated pollution prevention and control.

EEC, 2000. COUNCIL DIRECTIVE 2000/60/EC of 23 October 2000 establishing a framework for Community action in the field of water policy.

VLM, 2010. Voortgangsrapport Mestbank 2009 betreffende het mestbeleid in Vlaanderen. Vlaamse Landmaatschappij. Brussel. 208 p.

2 Evaluation of measures

O.F. Schoumans, W.J. Chardon, M. Bechmann, C. Gascuel-Odoux, G. Hofman, B. Kronvang, M.I. Litaor, A. Loporto, P. Newell-Price and G. Rubæk

The main goals of Working Group 3 were to evaluate different types of mitigation options for reducing nutrient loss (P and N), to study their effectiveness and costs under different conditions, and the feasibility of their implementation. For this reason an inventory was made along COST 869-participating countries to summarise:
- mitigation options that reduce the impacts of different P and N sources,
- measures that reduce the contribution via specific pathways, and
- measures that reduce the impact of P and N load in surface waters.

At first an inventory was made of the mitigation options that are already part of legislation in the different participating countries. In addition to this inventory a meeting of WG3 of COST 869 (Wageningen, 18-19 May 2009) was organised to collect information on agricultural measures which are implemented in the first River Basin Management Plans (RBMP). It was concluded that in no country such additional measures were implemented in legislation. The main reason was the lack of information about the sources and pathways of nutrients that are lost from agricultural land and missing information about the effectiveness of specific measures. Contrary to N losses, it was not expected that agricultural measures would reduce the P losses at short term. From this point of view all partners are interested in the outcome of this pan-European study on the applicability and effectiveness of measures.

In our study a total of more than 100 measures were distinguished by all members of the COST action 869. Some of these measures were quite similar and a list was reduced to 80 measures. Examples of options that could reduce the impact of different sources are e.g. reducing nutrient input or increase nutrient output; reducing P use within animal husbandry, adding immobilising agents to manure or to the soil. The contribution via specific pathways can be reduced by e.g. changes in cropping; cultivation management reducing erosion, buffer or riparian zones or sedimentation ponds and artificial wetlands. Possibilities to reduce the negative consequences of a historic P load of surface water are for example stimulating the growth of submerged water plants, removing biomass from ditches and streams, and place it away from ditch borders, removing sediment, setting out fish and flushing eutrophic lake water with nutrient poor water.

Finally, the list of 80 measures was grouped into eight categories:
1. Nutrient management (25)
2. Crop management (1)
3. Livestock management (7)
4. Soil management (18)
5. Water management within agricultural land (11)
6. Land use change (1)
7. Landscape management (8)
8. Surface water management (9)

Furthermore, we decided to write a factsheet for each measure. A small group of scientists wrote the first drafts of the factsheets which were reviewed by other participants. The draft version was published on the web-site in order to get a broad reaction from other scientists involved in this Action. All factsheets were set up according to a standard format, with the following headings:

a. General description of the measure
b. Rationale/mechanism
c. Applicability
d. Effectiveness including uncertainty
e. Time frame
f. Environmental side-effects / pollution swapping
g. Potential for targeting
h. Cost in terms of investments and labour needs
i. References

The outcome of the exercise is presented in this report. In the chapters 5 to 12 the mitigation options of the eight categories are discussed in general terms. On the website http://www.cost869.alterra.nl/Fs/List_of_options.htm all 80 mitigation measures are described separately ('factsheets'). In many factsheets information on other evaluation studies were taken into account. Important sources of information are the UK (Cuttle et al., 2006; Newell-Price et al., in prep.; Denmark (Schou et al., 2007) and the SERA-17 group in the USA (SERA-17; http://www.sera17.ext.vt.edu/SERA_17_Publications.htm).

In chapter 3 a conceptual framework of nutrient losses is presented in order to understand which sources or pathways are important and to link the mitigation options to major sources and/or pathways. In chapter 4 the legislation of the European Water Directives is discussed to determine which legislation is already mandatory.

References

Cuttle, S., Macleod, C., Chadwick, D., Scholefield, D., Haygarth, P., Newell-Price, P., Harris, D., Shepherd, M., Chambers, B. and Humphrey, R. 2006. An Inventory of Methods to Control Diffuse Water Pollution from Agriculture (DWPA) USER MANUAL. Defra report, project ES0203, 115 pp. http://www.cost869.alterra.nl/UK_Manual.pdf

Schou, J.S., Kronvang, B., Birr-Pedersen, K., Jensen, P.L., Rubæk, G.H. Jørgensen, U. and Jacobsen, B. 2007. Virkemidler til realisering af målene i EUs Vandrammedirektiv. Udredning for udvalg nedsat af Finansministeriet og Miljøministeriet: Langsigtet indsats for bedre vandmiljø. Faglig rapport fra DMU, no. 625, 132 pp. (in Danish with English summary).

SERA-17. Organization to minimize phosphorus losses from agriculture. See (http://www.sera17.ext.vt.edu/).

Newell Price, J.P., Harris, D., Taylor, M., Williams, J.R., Anthony, S.G., Duethmann, D., Gooday, R.D., Lord, E.I., Chadwick, D.R., Misselbrook, T.H and Chambers, B.J., in prep. *An Inventory of Methods and their effects on Diffuse Water Pollution, Greenhouse Gas Emissions and Ammonia Emissions from Agriculture - USER GUIDE*. Defra project WQ0106, 181pp. Draft version.

3 Conceptual framework

O.F. Schoumans, W.J. Chardon, C. Gascuel-Odoux, M.I. Litaor and G. Hofman.

3.1 Introduction

In order to structure the different types of mitigation options to reduce nutrient losses, a conceptual framework for the losses to surface water was set up. The next step was to link the mitigation options to this framework to show where and how the measures influence specific sources, processes and/or pathways. The conceptual framework was discussed during meetings in Waidhofen/Ybbs (Austria, 21 May 2008) and Wageningen (The Netherlands, 18-19 May 2009). The final conclusions of these discussions are described in this chapter.

3.2 General approach

In order to identify and recommend mitigation options, it is necessary to have an overview of the implied systems and relation we are looking at: 1) which production system controls the sources, and 2) the impacts which determine the factor to be controlled (Figure 3.1).

Figure 3.1
General overview of the focus of the study

With respect to the impact our concern is about the aquatic ecology. There are many types of water, like oceans/seas, coastal/lagoon/bays, rivers, lakes, brooks, ditches. Within our study the focus is on the water quality of the open freshwater systems within the catchment or river basin, which indirectly influence also the water quality of the coastal waters and the seas. Within a catchment or river basin many production systems will contribute to the loads of the ecological system, e.g. industry, sewage works, urban areas, scattered dwellings, direct atmospheric deposition, agriculture, and nature. Our study is also restricted to the impact of nutrient losses (nitrogen and phosphorus) to the surface waters. In HARP guideline 6 for both nitrogen and phosphorus the main sources (production systems) that influence the water quality are visualized, resp. in Figure 3.2 and Figure 3.3.

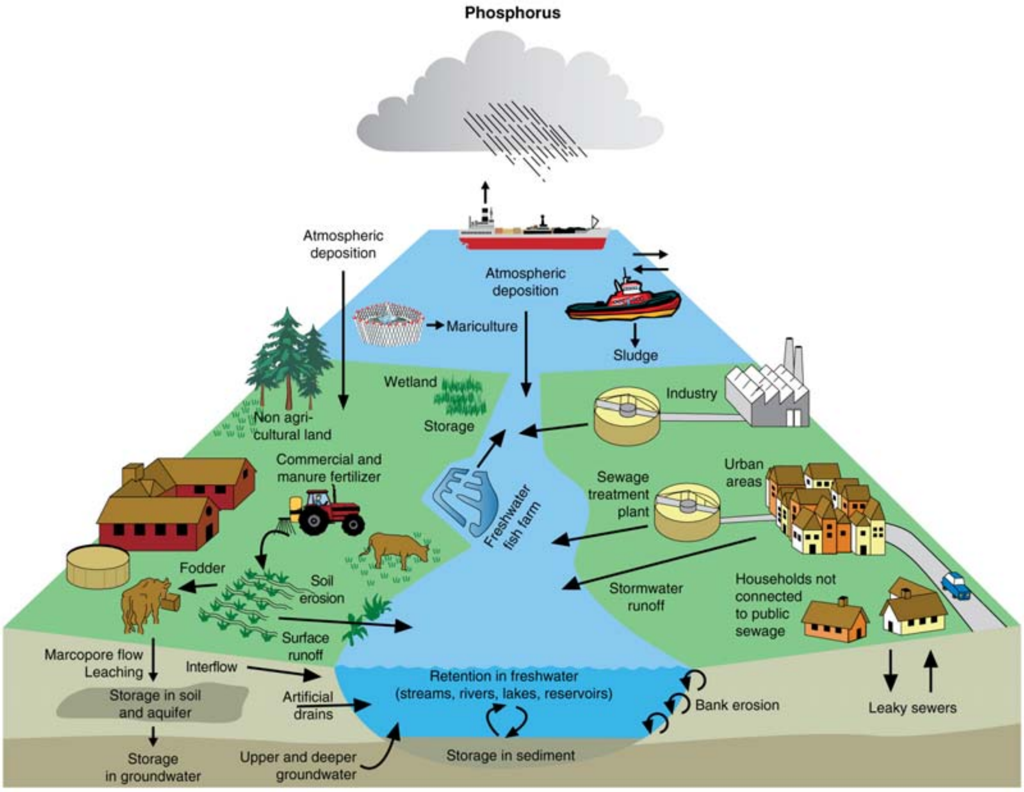


Figure 3.2
Phosphorus loads from different sources (after HARP guideline 6)

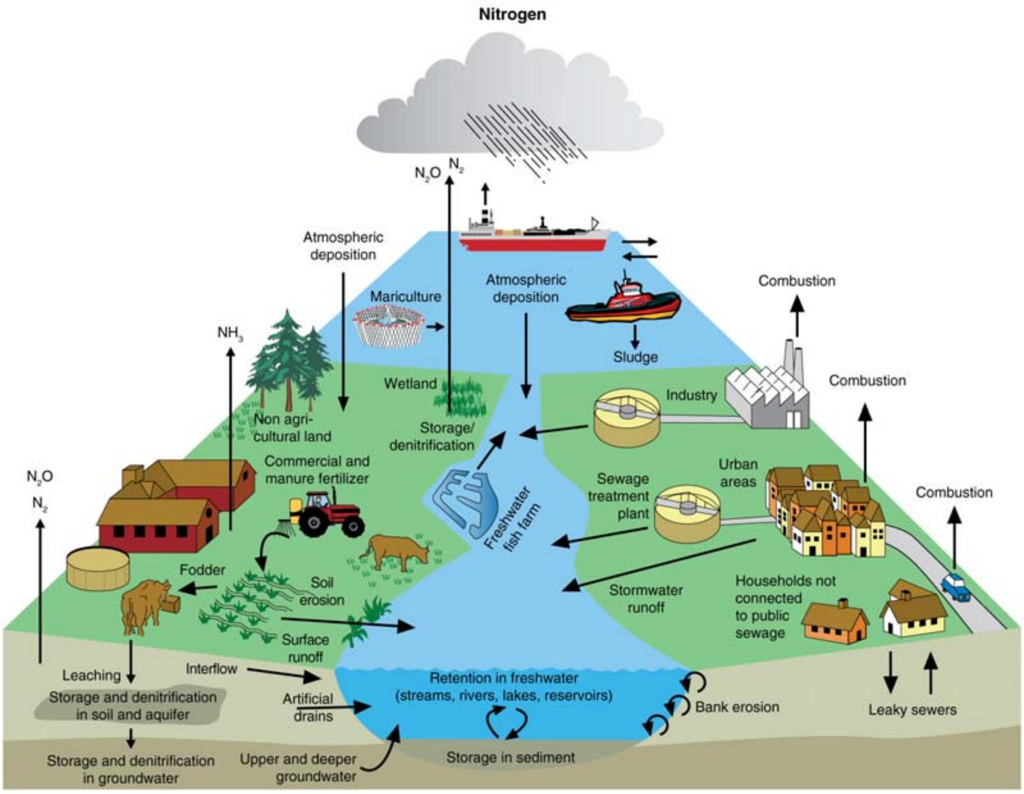


Figure 3.3
Nitrogen loads from different source (after HARP guideline 6)

Figure 3.4 shows for nitrogen and phosphorus the contribution of the most important production systems (point sources, agriculture and background concentration) to the nutrient loads of fresh surface waters. With respect to nitrogen, agriculture is the main source of the water pollution. Nitrogen runoff (which means in this figure total nitrogen loss) in areas with intensive agriculture is more than 5, and often more than 10 times higher, than in forested areas. With respect to phosphorus, the contribution of households and industries are still important, although in many countries the contribution of urban wastewater and industrial effluents, and the restriction of phosphate in detergents, has reduced more than 50 % since 1980. As a result, the relative contribution of agriculture to the nutrient loads to surface waters has increased. In the parts of Europe with intensive agriculture, the contribution from agriculture approaches 50 % of the total P loss.

In the Nordic countries the nitrogen and phosphorus loads are lowest, because of the low population density and because only a small part of the land is cultivated. In the five areas with highest amounts of nutrient loss per ha about 40-50 % of the area is cultivated and losses seem to be related with the intensity of agriculture practice; the higher the fertiliser applications, the higher the nutrient losses (up to 28 kg N per ha and 2.7 kg P per ha, both expressed per ha total land). In most European inland waters phosphorus is the most critical nutrient in relation to the eutrophication of the surface waters, and in coastal (salt) waters nitrogen is the most critical element.

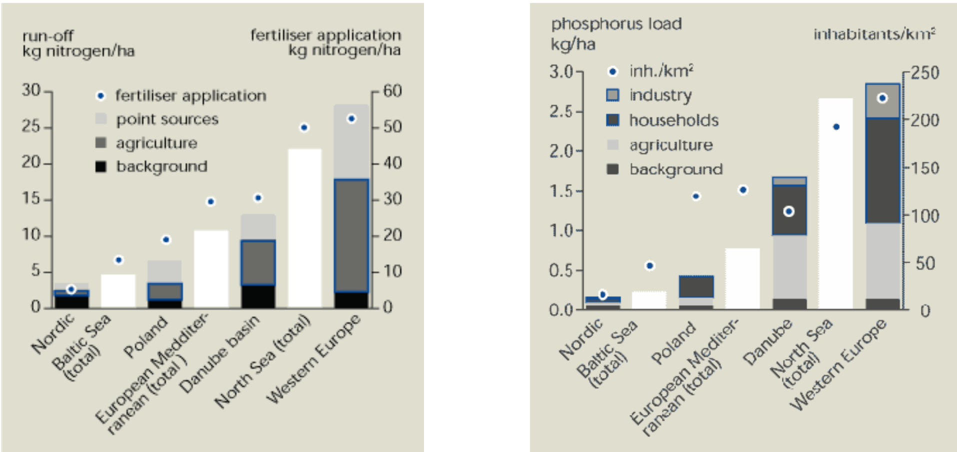


Figure 3.4

Nitrogen and phosphorus runoff (total loss), fertiliser application and population density in selected European areas between 1988 and 1996 (Source: EEA-ETC/IW). Note: All areas are greater than 300 000 km². Runoff (total loss) and fertiliser application per hectare of total land area

COST action 869 mainly gives attention to the nutrient losses from agriculture to surface waters, because it is difficult to reduce this kind of losses. The main reason is the complex combinations of sources and pathways that determine the nutrient losses in rural areas. Often the Driver-Pressure-State-Impact-Response (DPSIR) is used to analyse problems. With respect to livestock based agriculture, the following scheme (Figure 3.5) can be used (Rekolainen, 2006).

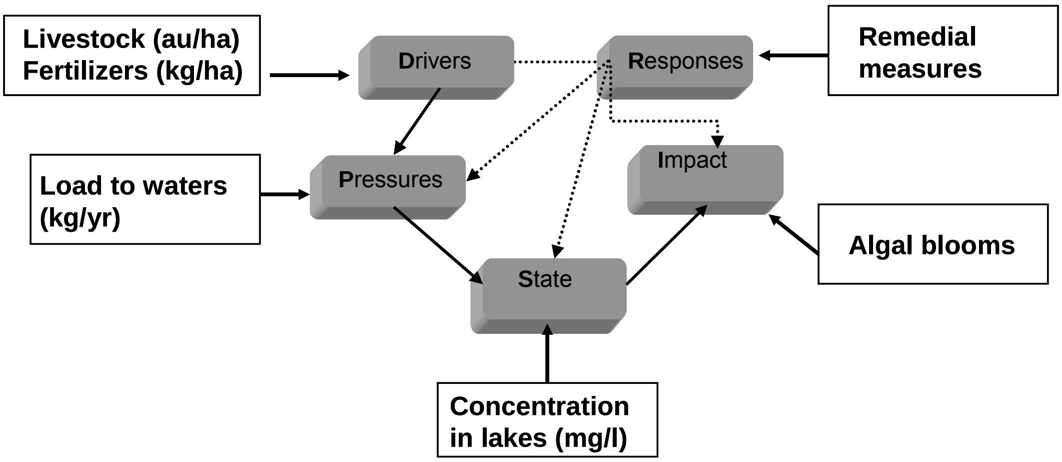


Figure 3.5
The DPSIR approach for a livestock based agricultural system. (after Rekolainen, 2006)

In fact, on a catchment scale, the relation between the mentioned Driver (livestock) and the Pressure (load to surface water; Figure 3.5) is more complex than what is presented in the DPSIR scheme, because all available sources of nutrients should be taken into account (e.g. also fertiliser use) and the way these sources can become mobile for transport and move with the water flow over the land surface and through the soil to surface waters. The source - transfer system by Heathwaite et al. (2003) provided a general scheme of the Driving forces of nutrient loss to surface water (loads/pressure). More recently, Withers and Haygarth (2007) and Haygarth et al. (2005) expanded this approach into a source-mobilization-delivery-impact continuum for phosphorus transfer from agricultural land to water (Figure 3.6), which is based on physical principles that describe the mechanisms of P transport. With respect to nitrogen this figure is also useful; however, then also deep or shallow aquifers flow to surface water should be taken into account.

The components of the source-mobilization-delivery-impact continuum are a quite generic approach to identify surface areas at risk with the focus on transport (as discussed within WG 1 of the COST action). However, with respect to the sources, it is important to separate the *actual* manure and fertiliser applications (sources) from the nutrient status of the soil and shallow aquifers as a result of the nutrients stored in environment due to applications during the last century. The main reason is that the amount of nutrients that is accumulated in the soil and aquifers is much greater than the actual annual nutrient applications rates, and different types of mitigation options are necessary to reduce the impact of this source (polluted soil and aquifers) on water quality. Furthermore, with respect to manure applications and the purchase of fertilisers, at farm scale the farmer takes into account the nutrient status of the fields and the crops to be produced.

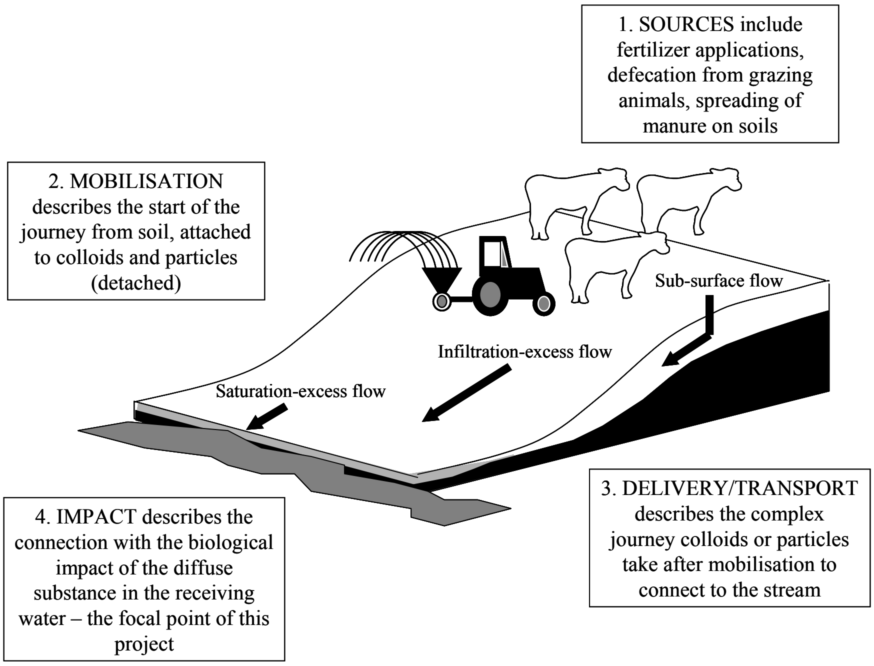


Figure 3.6

The components of the source-mobilization-delivery-impact continuum for phosphorus transfer from agricultural land to water (after Withers and Haygarth, 2007)

As a result a broader, but more general, starting point for the sources was developed (Figure 3.7). With this scheme we will put also more attention to the inter- and intra-relationships between a farming system and the accompanied other systems (cropping-soil system; landscape and hydrological system and ecological system). In the following section (3.3) the individual systems will be described in more detail.

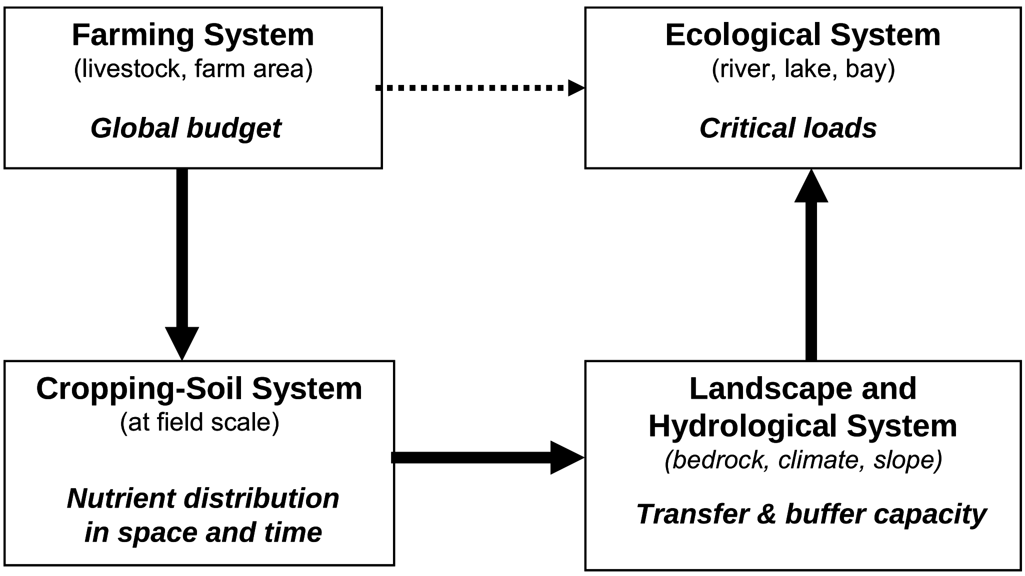


Figure 3.7

Schematic visualization of the elements of the conceptual framework

3.3 Systems

In the conceptual framework as depicted in Figure 3.7 we identify four systems:
- Farming system;
- Cropping-Soil system (field scale);
- Landscape-Hydrological system (catchment scale)
- Ecological system.

This will be discussed in more detail below.

3.3.1 Farming system

The enrichment of the agricultural land and the potential losses of nutrients to surface water are highly dependent on the farming system (farm area and the production type e.g. dairy, pig, poultry, and arable farms) that are present within the catchment.

At farm level import decisions are made which determine the overall nutrient balance of the farm, since the nutrient input (mainly fertilizers, animal feed and animal manure) is related to the assessment by the farmer of the need of nutrients to make a huge quantity and quality of products (arable crops, vegetables and meat). Major socio-economical decisions are made at this level (buying or selling farm land, crop rotation and number of animals, economic situation, transport of manure from the farm, ideology of the farmer etc), which lead to huge differences in annual nutrient surpluses between farms (e.g. dairy farms or arable farms). Nutrient balances at farm scale are very important information for policy makers (indicating potential risk areas), intermediaries (giving recommendations) and the agriculture itself (study groups), in order to develop a more sustainable agricultural practice from an agro-environmental point of view (Simon et al., 2000; Schröder et al., 2003). An example of such a budget procedure is shown in Figure 3.8.

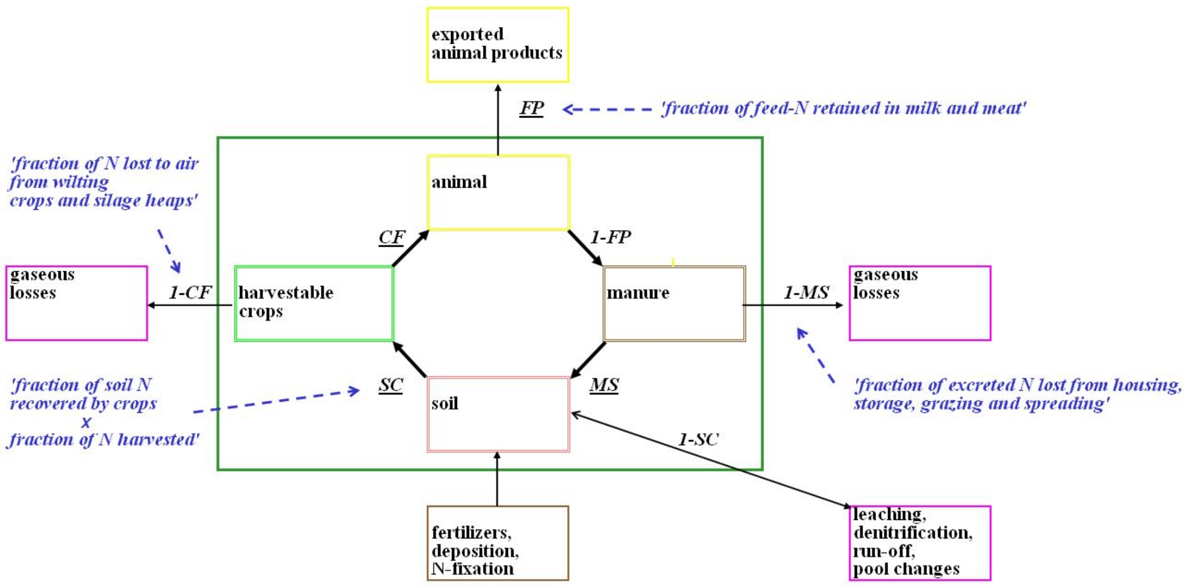


Figure 3.8
Flow diagram of mixed farming systems (after Schöder et al., 2003)

The recommendations and legislation (Directives and national legislation) have been implemented in many different ways by farmers, because a large variation of nutrient balances at farm scale is still being observed, especially on dairy farms. (Aarts et al., 2008). Because the behaviour of the farmers differ in terms of farm and field management it is important to look in more detail which options there are to minimize the nutrient surplus, not only at farm scale, but also at field scale, and which options there are to reduce the nutrient losses from farmer's individual fields, that actually contributes to the surface water quality.

3.3.2 Cropping-Soil system

The type of farming system and the farm management itself will determine the cropping sequence on the fields and the associated nutrient distribution over the fields during the year, and as a result the mobilisation of the nutrients. The amounts of animal manure and fertiliser to be applied to the field highly depend of the type of crop and the field conditions. Especially, the manure storage capacity at farm scale and the nutrient status of the fields are important factors that control the nutrient management of the fields. Fields receive N and P in many different ways, such as application of animal manure and fertilizers, dry and wet deposition, crop and root residues, organic matter (e.g. fresh material and inert organic material). Much of the amount of applied nutrients is taken up by the crop, and a part of that is harvested. During and after the application of nitrogen a part of the N may volatilize to the air (as NH_3, N_2 or NO_x). A nutrient surplus mainly accumulates in the soil and due to transformation processes nutrient losses can occur from the field to surface water (Figure 3.8).

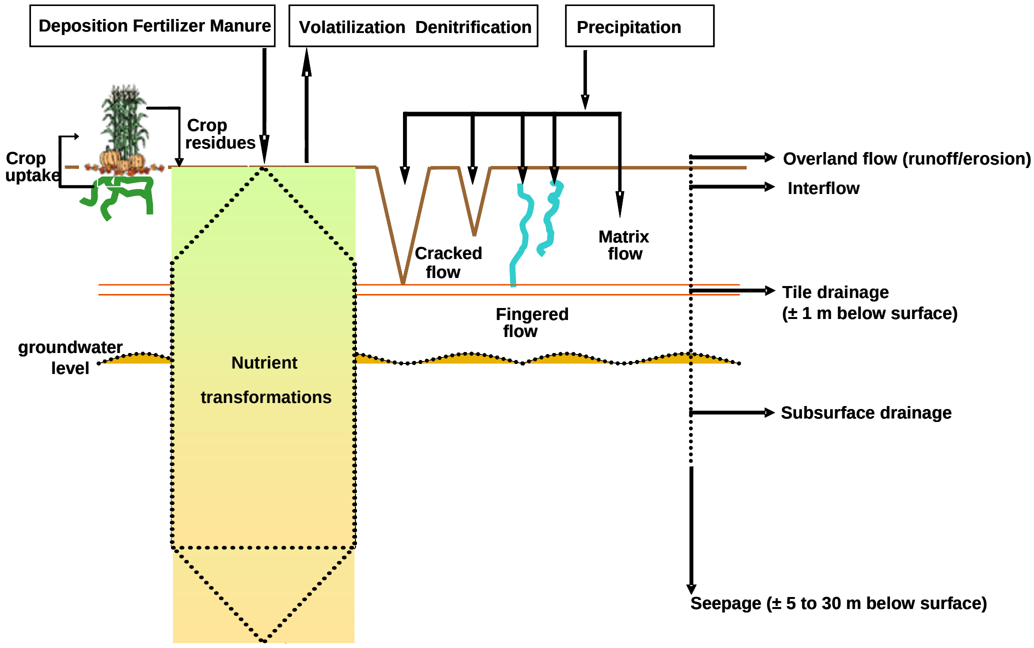


Figure 3.8
Schematic visualization of the nutrient losses at field scale (after Schoumans and Chardon, 2003) (Subsurface drainage may also occur above tile drains at high water table levels during high rainfall events)

A general schematization of the available sources and transformation/turnover and release processes that influence the amount of mobile nutrient forms that can be transported is shown in Figure 3.9. In the left part of the figure different sources of nutrient are mentioned. Due to turnover (biological processes) and chemical and physical transformations different nutrient forms exist within a field. A simple and rough division can be made between organic and inorganic forms. A part of all nutrients can be mobilized, mainly as a result of biological processes, solubilisation processes (e.g. dissolution, desorption) and detachment of material. Transport of nutrients attached to small-sized soil particles, can be very important, especially for phosphorus. These particles may be very fine or even colloidal sized (< 1 µm). For water quality a separation is made between soluble and solid forms. Soluble inorganic and organic forms are that part that passes a 0.45 μm filter, the part that remains on the filter is defined as solid mobile material (Haygarth and Sharpley, 2000).

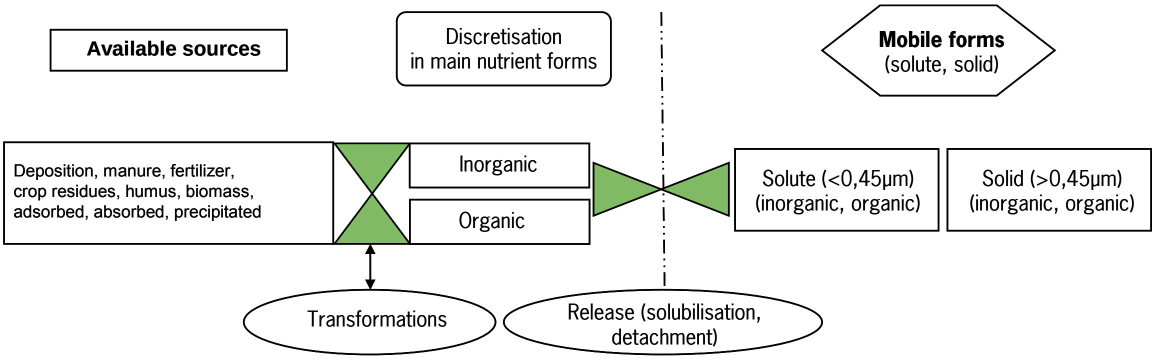


Figure 3.9

Schematic visualization of the available nutrient sources in the field (left part) and the release of mobile nutrient forms

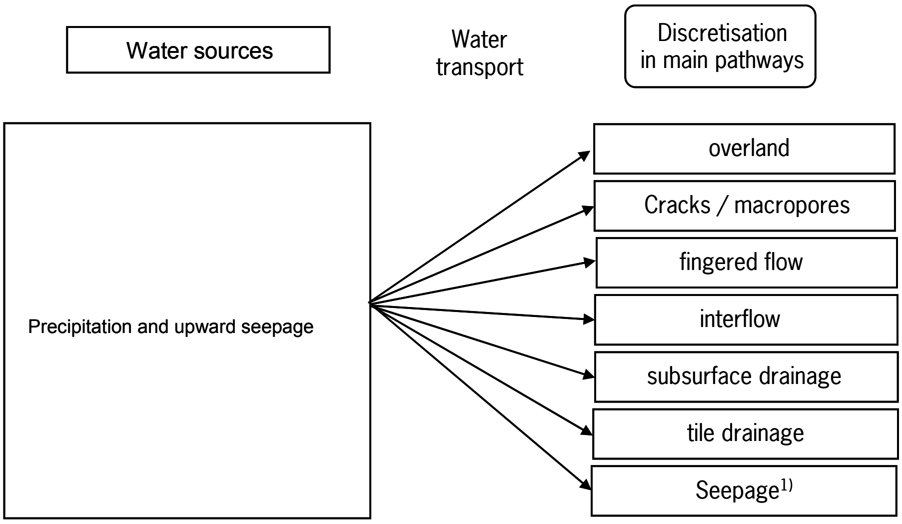
The amount of mobile nutrients is controlled by 1) amount and form of applied nutrients, 2) the uptake of nutrients by the crop during the year, 3) the current nutrient status (amounts and forms of the nutrients in the field), 4) weather conditions and available water amount and 5) the internal capacity of the soil to absorb nutrients into non-mobile forms. This capacity highly depends on the transformation processes, such as composting, adsorption/solubilisation and denitrification, which can differ in time, and will influence the release and the mobility of nutrients. The capacity of the soil to bind or release phosphorus is an important process regarding P mobilisation. With respect to nitrogen denitrification is a major important process.

3.3.3 Landscape-Hydrological system

Precipitation (rainfall surplus) and upward seepage are the main sources that force water flow through and over the soil. With the water flow, mobile nutrient forms will be transported within the field and/or out of the field to surrounding fields or directly to surface waters. During this transport, the transported nutrients will interact with their environment, and often a part of the nutrients will be adsorbed or absorbed by the soil (mainly P), volatilized (N). The main pathways of water transport that can be considered are (see Figure 3.10):

- overland flow,
- interflow (lateral transport of water through the upper layers of the soil above the water table during or after high rainfall events),
- subsurface flow (lateral flow of groundwater under free drainage conditions),
- artificial drainage (lateral flow of groundwater via tile drains / pipe drains), and
- downward seepage to deeper groundwater bodies.

Furthermore, shortcuts from the surface to deep layers are possible, especially in cracked soils or hydrophobic soils (fingered flow) or via very fast macropore flows. The total amount of water transported via overland flow, lateral water movements or groundwater recharge is usually the most important factor for estimating the nutrient losses out of the field to the stream.



1) Recharge to deep groundwater

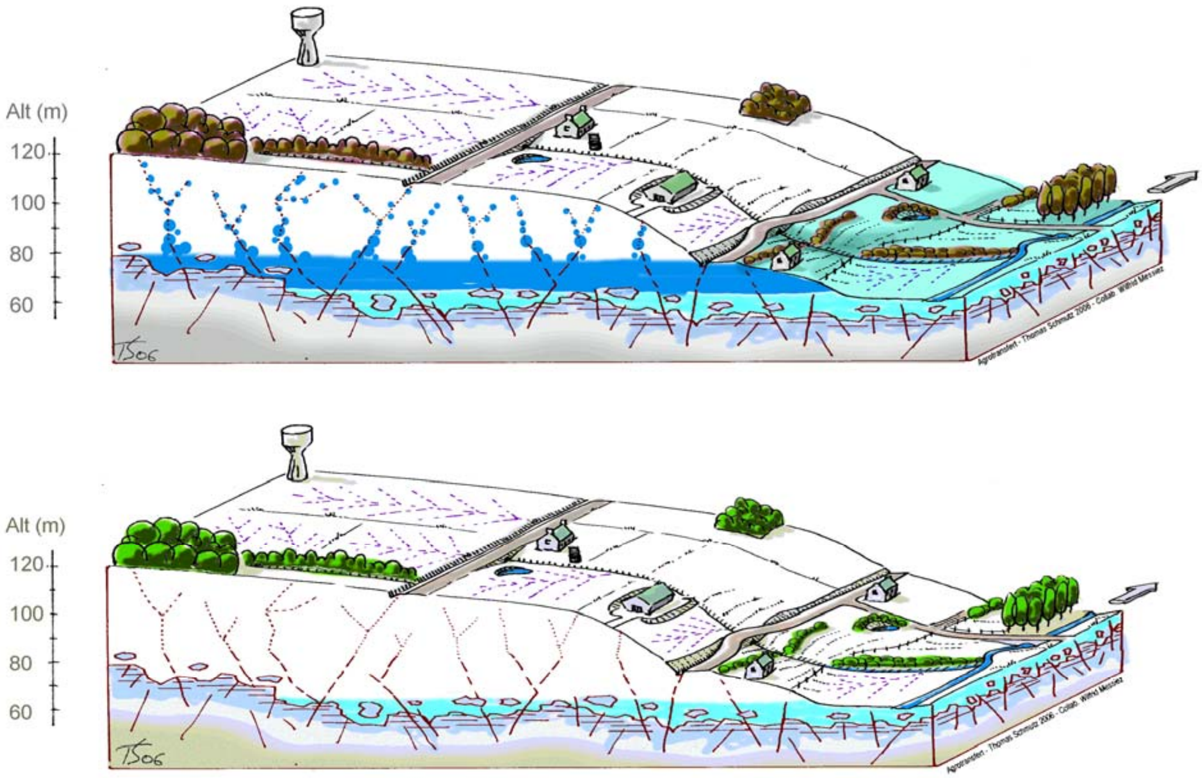


Figure 3.11

Schematic visualization of nutrient losses at catchment scale. Comparison of winter (upper part) and summer (lower part) conditions of a hill slope (or catchment). While the shallow groundwater depth is closer to the soil surface in the bottom domain during winter, preferential flow, saturated surface flow, drainage are higher, interaction between soil upper layer are more important, inducing denitrification in bottom domain and P export. (Adapted from Territ'eau tools: http://agro-transfert-bretagne.univ-rennes1.fr/Territ_eau/)

Due to overland water flow (surface runoff) soluble nutrients in runoff water and the nutrients bound to eroded material (e.g. particulate bound phosphorus) will be transported. Not all these nutrients will be delivered to the surface waters. This depends strongly on:

- the (actual) connectivity of the field with the receiving surface water (e.g. river, lake, ditch), since runoff water will (partly) infiltrate into the soil,
- the buffering capacity of the landscape (local depressions within the field or along the slope; buffers like hedges, roads and other landscape line elements),
- denitrification in wetlands and deeper groundwater connection between the field and surface waters and within stream processes (in the surface waters).

Transport through the soil (via subsurface runoff and tile drainage) and weathered layers to surface water, which can concern flows from the soil surface up to 30 m in depth, the main transport route in flat areas is more difficult to manage due to the inherent complexity of the groundwater system. With respect to phosphorus, a subdivision can be made in (a) fields that are directly located at the border of surface water (ditch, river, lake), (b) fields with artificial drains, a shortcut to the surface water, and (c) fields at a greater distance from surface water (more than 20-50 m; Schoumans et al., 2009). Agricultural P losses through the soil to surface water of fields at greater distance can often be neglected in flat areas. With respect to nitrates, a subdivision can be made in hill slope domain and saturated bottom lands where denitrification processes occur.

The controlling factors of nutrient losses via runoff, erosion and leaching are mainly defined at catchment level, where the relevant physiographic settings can be defined. Important factors are landscape conditions in combination with meteorological conditions, and the watershed drainage network, which determine the amount of runoff/water flow, pathways and discharges. Mitigation measures mainly intend to control/reduce nutrient status and the water flow of specific pathways which are enriched with nutrients by blocking these pathways or by increasing buffering capacity in the streams.

3.3.4 Ecological system

The impacts of farming on ecosystems can be evaluated at a lake, brook, river or estuary / sea / coastal scale. Controlling and managing the loads and the nutrient bioavailability in time of these systems is important. There are three major driving factors that should be determined:

1. The use, the key species, the diversity or specificity of the ecological system which reflect the ecosystem weaknesses;
2. The trophic status, in relation to N and P loads, and other parameters which can modify the ecological system;
3. The hydrodynamic nature of the ecological system including water flow and the stratification of water impact e.g. benthic processes, in addition to advection, because these factors influences the chemical reactivity of the system and therefore can accelerates of delays the eutrophication.

The analysis of an ecological system is important, because it determines what we have to control. It also governs how the ecological system as a whole works, the analysis of the contribution of the different sources to the fresh water system and an analysis of the internal processes and conditions that control the ecological status of the fresh water systems. Management design should consider the entire water course/catchment. Nitrogen may be transported from a P limited lake to N limited coastal waters. Therefore, we should be able to estimate the retention of N in P-limited systems and vice versa. The biogeochemical cycling of N and P are closely linked to each other, and thus the measures focusing on one of the nutrients can affect the other.

The relative concentrations of total N and total P have been used to estimate which of these nutrients is limiting the growth of algae in aquatic systems. The approach is simple and easy to use provided that there are data on N and P concentrations. Yet, interpretation of the results should be done with caution as the N:P ratio may not correctly indicate the limiting nutrient of the system. The approach has mainly been used for standing waters, i.e. lakes and coastal areas, where nutrients rather than physical conditions tend to limit algal growth.

Working Group 2 is carrying this impact analysis in more detail. Such an analysis is also the first step in order to determine which mitigation options will really improve the water quality from an ecological point of view. With respect to mitigation options that focus on the improvement of the ecological status of the water system, it can be relevant to determine the trophic status and which nutrient is the limiting factor for the ecological system, but possible other factors should also taken into account. Furthermore, it is important to mention that also the absolute concentration level also plays a role. If the concentration of N and P are very low neither P nor N may be limiting. Finally, a problem related to the use of N:P ratio in estimating the limiting nutrient is brought about by the biological unavailability of some forms of nutrients. For example, P bound to eroded soil particles, forming a major P fraction in areas with arable farming and surface runoff, is not entirely available to algae. The same applies to dissolved organic N. More details can be found in the factsheet on 'N:P ratios in estimating nutrient limitation in aquatic systems' (http://www.cost869.alterra.nl/FS/FS_NPratio.pdf) as produced by WG 2.

Although there are limitations, the ratio of nitrogen to phosphorus compounds in a water body is used as an indicator suggesting which of the two elements will be the limiting, and consequently which one has to be controlled. Table 3.1 gives general criteria for N/P ratios. As to freshwaters (lakes, rivers, reservoirs), it has traditionally been assumed that P is the nutrient present in lowest amount in relation to the requirements of phytoplankton. In marine systems, N has been identified as the growth limiting nutrient, especially in summer, whereas in estuaries P may be limiting in the fresh-water part and N in the marine part. Intermediate areas such as rivers, brooks and creeks are often phosphorus-limited during spring. Finally, in very nutrient-rich or turbid waters light rather than any of the nutrients may be in too short a supply. Physical conditions (morphology and hydrological regime) limit algal growth especially in rivers and creeks. The current view is that limitation of N can also be observed in freshwaters. Nitrogen limitation may be found when there is a high P level due to human perturbations or to P-rich soil type, or in low productive conditions with low N deposition.

Table 3.1
Nitrogen/Phosphorus ratios (expressed in weight) for various limiting conditions in freshwater and estuarine/coastal water

N-limiting (Ratio N/P)		**Intermediate (Ratio N/P)**	**P-limiting (Ratio N/P)**
Freshwater`	≤ 4.5	4.5 - 6	≥ 6
Estuarine/coastal water	≤5	5 - 10	≥ 10

(Source: EC, Eutrophication and Health, 2002)

3.4 Measures

As described in Chapter 2 the measures were combined in eight categories of mitigation options. The categories of the mitigation measures fit into the operational scheme of Figure 3.7 and are presented in Figure 3.12.

Regarding the farming system the focus is on category 1 - Nutrient management strategies and category 3 - Livestock management. Within nutrient management strategies attention is given to fertilisation recommendations in relation to crop requirements and farming balancing approaches. Livestock management discusses ways to reduce the manure surplus or to decrease the amount of nutrients in the manure by fodder management.

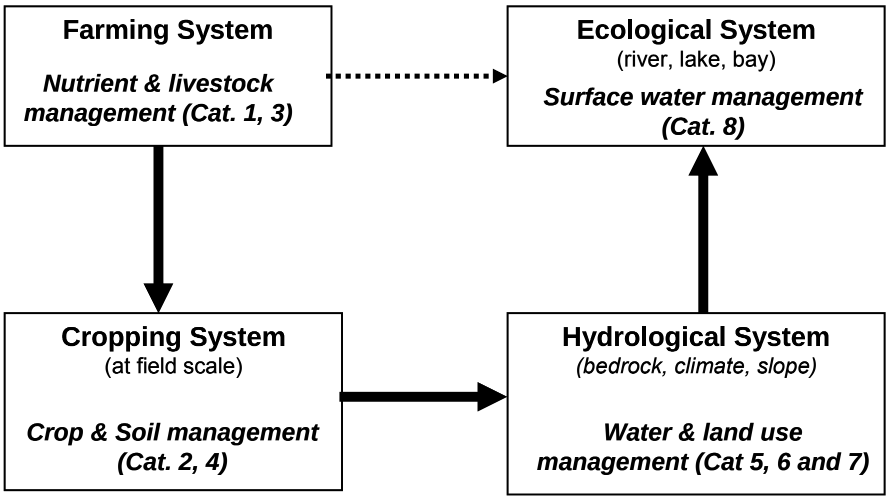


Figure 3.12
Schematic visualization of the conceptual framework including the categories of the mitigation measures (1-8; see text)

The cropping system handles aspects of crop and soil management, resp. category 2 and 4 of the group of mitigation options. Via crop management the nutrient losses to the environment can be reduced via e.g. application of catch crops, early sowing systems, apply crops that are less sensitive for surface runoff, phosphorus mining and improve rooting systems. Soil management of the fields can help to minimize the nutrient losses via e.g. changing soil tillage practices which increase the infiltration capacity and reduce surface runoff.

The third block of figure 3.12 relates to ways to reduce or block the transport of nutrients by manipulating the water flow. This can be done via proper water management on agricultural land (category 5), land use changes (category 6) and landscape management (category 7). Water management practices (category 5) focus on the main pathways of nutrient losses to surface water by (1) constructing ponding systems, grassed waterways, installing sediment boxes to reduce overland flow, (2) changing the drainage system, (3) controlling the surface water level and (4) irrigation practices. Land use changes (category 6) deals with reallocation of land uses or convert land to nature or less extensive agricultural systems. In addition, landscape management (7) is possible, such as delineate the functional hydrographical network, manage surface water boundaries and/or field boundaries.

Finally, also in the ecological system, i.e. surface water (Block 4 in Figure 3.12) a group of mitigations options (category 8) are possible. A distinction is made into (a) channel management, that is mainly focussed on increasing the retention time and sedimentation in surface water, (b) restoration of the surface water by re-establishing e.g. wetlands and lakes, (c) abatement of eutrophication by nutrient inactivation. In the next chapters the mitigation options of these eight categories are discussed in more detail.

References

Aarts, H.F.M., Daatselaar, C.H.G. and Holshof, G., 2008. Bemesting, meststofbenutting en opbrengst van productiegrasland en snijmaïs op melkveebedrijven. Rapport PRI 208. European Commission, Eutrophication and Health, 2002. Luxembourg: Office for Official Publications of the European Communities. 28 pp. ISBN 92-894-4413-4,

EC, 2002. Eutrophication and Health,

HARP, 2000. Development of HARP Guidelines. Harmonised Quantification and Reporting Procedures for Nutrients. SFT Report 1759/2000. TA-1759/2000. ISBN 82-7655-401-6.

Heathwaite, A.L., Fraser, A.I., Johnes, P.J., Hutchins, M., Lord, E. and Butterfield, D., 2003. The phosphorus indicators tool: A simple model of diffuse P loss from agricultural land to water. Soil Use Manage. 19:1-11.

Haygarth, P.M. and Sharpley, A.N. 2000. Terminology for phosphorus transfer. J. Environ. Qual. 29:10-15.

Haygarth, P.M., Condron, T.L.M., Heathwaite, A.L., Turner, B.L. and Harris, G.P. 2005 The phosphorus transfer continuum: Linking source to impact with an interdisciplinary and multi-scaled approach. Science of the Total Environment 344, 5-14.

Rekolainen, S., 2006. Presentation start up meeting COST 869. Brussels, d.d. 6-7 november 2006.

Schoumans, O.F. and Chardon, W.J. 2003. Risk assessment methodologies for predicting phosphorus losses. J. Plant Nutr. Soil Sci. 166: 403-408.

Schoumans, O.F., Groenendijk, P., van der Salm, C. and Pleijter, M. 2009. Methodiek voor het karakteriseren van fosfaatlekkende gronden: beschrijving van het instrumentarium PLEASE, Alterra, Wageningen, Alterra-rapport 1724. pp. 76.

Simon, J.C., Grignani, C., Jacquet, A., Le Corre, L. and Pagès, J. 2000. Typologie des bilans d'azote de divers types d'exploitation agricole: recherche d'indicateurs de fonctionnement. Agronomie 20, 175-195.

Schröder, J.J., Aarts, H.F.M., ten Berge, H.F.M., Van Keulen, H. and Neeteson, J.J. 2003. An evaluation of whole-farm nitrogen balances and related indices for efficient nitrogen use. European Journal of Agronomy 20: 33-44.

Withers, P.J.A. and Haygarth, P.M. 2007. Agriculture, phosphorus and eutrophication: A European perspective. Soil Use Manage: 23:1-4.

4 Environmental European legislation in relation to agriculture

O.F. Schoumans and O. Oenema

4.1 Introduction

Technological developments, changes in markets and the Common Agricultural Policy (CAP) of the European Union (EU; initially European Economic Community, or EEC) have greatly contributed to the modernization of agriculture in its Member States from the early 1960s onwards. The modernization of agriculture has led to increased productivity and food security, but has also changed rural landscapes. Less labour was needed in agriculture while more labour was needed in industry and services in urban areas. The countryside changed because of removal of landscape elements like hedgerows, stonewalls, ditches and restructuring of parcels and waterways. The intensification of agriculture also led to elimination of species-rich semi natural habitats (meadows, pastures), loss of biodiversity, and to deterioration of water, soil and air quality. More specifically, the main environmental problems caused by agricultural practices are:

Water:	Enrichment by nitrates, phosphates, pesticides, heavy metals and antibiotics.
Air:	Increased concentrations of ammonia, greenhouse gases (CO_2, CH_4 and N_2O), pesticides.
Soil:	Erosion, declining organic matter content, contamination, compaction, pollution by heavy metals.
Biodiversity:	Habitat disturbance and destruction, decline in farmland birds.
Landscape:	Changes in physical structure and land abandonment. Removal / change of features contributing to protection of water, soil, biodiversity and decreasing heterogeneity.

In response, the EU Council has implemented various environmental policy measures to improve our environment (Table 4.1). With respect to the legislative status, the EU makes a difference between a *Regulation* and a *Directive*. Accenttekesn en curiserf is dubbel accent

- *Regulations* have a general application, are binding in their entirety and are directly applicable in all Members States. They do not require a national act to be imposed. An example is the Regulation on agricultural production methods compatible with the requirements of the protection of the environment and the maintenance of the country side (Council Regulation EEC 2078/92)
- *Directives* are also binding upon each Member State to whom they are addressed and all objectives of the directive must be reached. However, the objectives of each directive can be obtained in different ways by each national or regional authority. It means that in this case the Member States have the choice of forms and methods to be applied to reach the goals, regarding local human, technical and physical settings.

In this chapter the most important Directives in relation to water, air, soil and nature (landscape and biodiversity) development will be discussed. Table 4.1 gives an overview of these relevant Directives.

Table 4.1
Important European Directives in relation to environmental aspects

Water		
	Waste Framework Directive	75/442/EEC amended by 91/156/EEC
	Bathing Water Directive	76/160/EEC amended by 2006/7/EC
	Dangerous substances	76/464/EEC = 2006/11/EC
	Groundwater Directive	80/68/EEC
	Waste Water Directive	91/271/EEC
	Nitrates Directive	91/676/EEC
	Water Framework Directive	2000/60/EC
	Marine Strategy Framework Directive	2008/56/EC
Air		
	IPPC Directive	96/61/EC
	Air Quality Framework Directive	96/62/EC
	Air Quality	1999/30/EC
	National Emissions Ceilings Directive	2001/81/EC
Soil & Landscape		
	Forthcoming Thematic Soil Strategy	COM(2006) 232
	Currently, Soil Strategy	Annex IV of R1782/2003
Nature		
	Conservation of Wild Birds	79/409/EEC
	Conservation of Natural Habitats	92/43/EEC

4.2 Directives

Water
With respect to the improvement of the water quality, many directives have been introduced. The Waste Water Directive (91/271/EEC) concerns urban waste-water treatment. Its objective is to protect the environment from the adverse effects of urban waste water discharges and discharges from certain industrial sectors. To ensure good bathing water quality, the EU has introduced the Bathing Water Directive (2006/7/EC) and has set limits for physical, chemical and microbiological parameters. National authorities must ensure that these limits are not exceeded. Today the limiting factor for bathing water quality is microbiological pollution, originating from waste water or from agricultural runoff. The Dangerous Substances Directive of 1976 (76/464/EEC amended by 2006/11/EC) deals with pollution caused by certain dangerous substances discharged into the aquatic environment. It had the ambitious objective of regulating potential aquatic pollution by thousands of chemicals already produced in Europe at that time. The Directive covered discharges to inland surface waters, territorial waters, inland coastal waters and groundwater. In 1980, the protection of groundwater was taken out of 76/464/EEC, and was regulated under the separate Groundwater Directive (80/68/EEC) on the protection of groundwater against pollution caused by certain dangerous substances. Recently this Directive became part of the Water Framework Directive (2000/60/EC article 22 and article 16).

The Nitrates Directive (91/676/EC) gives special attention to nutrient losses from agriculture. The main objective is to protect water quality across European Union by preventing nitrates from agricultural sources polluting groundwater and surface waters (eutrophication) and by promoting the use of good farming practices. In order to achieve this, Member States have to (1) Indentify polluted or threatened waters, (2) Designate nitrate leaching vulnerable zones (NVZs), (3) Establish Code(s) of good agricultural practice, to be implemented by farmers on a voluntary basis, (4) Establish Action Programmes, to be implemented by farmers within NVZs on a compulsory basis and (5) Monitor the progress of implementation, and report every 4 years on nitrate concentrations, eutrophication, assessment of the impact of Action Programmes and if necessary revision of NVZs and Action Programmes.

The purpose of the Water Framework Directive (2000/60/EC; WFD) is to establish a framework for the protection of surface waters (including rivers, lakes, transitional and coastal waters) and ground waters throughout the EU territory. In fact, it has become an important umbrella of many Water Directives. The main environmental objectives are to achieve and maintain a good status for all surface waters and groundwater by the target date of 2015, and to prevent deterioration and ensure the conservation of high water quality where it still exists. This is to be accomplished by implementing the measures necessary to:

- prevent deterioration of the status of waters,
- protect, enhance and restore all bodies of surface waters and groundwater,
- promote sustainable water use (through effective pricing of water services),
- progressively reduce discharges of priority substances and cease or phase out discharges of priority hazardous substances for surface waters,
- ensure progressive reduction of pollution of groundwater,
- mitigate the effects of floods and droughts,
- ensure sufficient supply of water and
- protect the marine environment.

In order to achieve these objectives of the WFD, the River Basin Management Plans (RBMP) have to be implemented. In 2009 the first RBMP of all Member States were published.

The approved latest Water Directive deals with marine waters. The main aim of the Marine Strategy Framework Directive (2008/56/EC) is to achieve a good environmental status of the EU's marine waters by 2020 and to achieve the full economic potential of oceans and seas in harmony with the marine environment. Since the water quality of estuaries and sea waters can be too trophic, there is a need to reduce the nutrient emissions to sea waters (OSPAR, HELCOM).

Air

In 1996 the Air Quality Framework Directive became active (96/62/EC), which addresses ambient air quality assessment and management by the Member States. In the same year also the Integrated Pollution and Prevention Directive (IPPC; 96/61/EC) was adopted by the European Council. This Directive aims to minimize pollution from point sources and NH_3 from agriculture. Large pig and poultry farms have to minimize NH_3 emissions by applying Best Available Techniques (BATs). The Air Quality Directive (1999/30/EC) relates to limit values for sulphur dioxide (SO_2), nitrogen dioxide (NO_2) and oxides of nitrogen (NO_X), particulate matter (PM_{10}, $PM_{2.5}$) and lead (Pb) in ambient air. The annual limit values for NO_2 and NO_x are resp. 40 and 30 mg l^{-1} (to be met in 2010 and 2001). The National Emissions Ceilings Directives (NEC; 2001/81/EC) aims to limit emissions of acidifying and eutrophying pollutants and ozone precursors. Upper limits are set for the total emissions of each Member State in 2010 regarding SO_2, NO_x, VOCs and NH_3. One of the main targets of the NEC is to reduce ammonia emissions from agriculture, because agriculture contributes for about 90% of EU total NH_3 emissions (http://www.eea.europa.eu/data-and-maps/indicators/eea-32-ammonia-nh3-emissions/eea-32-ammonia-nh3-emissions).

Soil

Until now there is no active soil quality protection directive in the EU-27, although various EU policies indirectly contribute to soil protection (like reducing chemical emissions, industrial pollution prevention, nature protection, emissions from agriculture). In 2006, the Commission adopted a Soil Thematic Strategy (COM(2006) 231) and a proposal for a Soil Framework Directive (COM(2006) 232) with the objective to protect soils across the EU. However, the Soil Framework Directive has not been accepted (yet). The strategy and the proposal have been sent now to the European Institutions for further steps in the decision-making process. Soil protection and preservation of the capacities of the soil is necessary, i.c. for biomass production, storing and filtering and transforming of substances, biodiversity-pool, source of raw materials, etc. Special attention is given to preventing erosion, organic matter decline, compaction, salinisation and landslides. In combination with Annex IV of the R1782/2003; Common Agricultural Policy; CAP) and its Good Agricultural and Environmental Condition framework (GAECs; EEC 2005) requirements regarding soil protection with respect to farming becomes clear (soil erosion, soil organic matter, soil structure, minimum level of maintenance and avoidance of deterioration of habitats).

Nature and Landscape
With respect to nature conservation, Member States have to meet the obligations of Directive 79/409/EEC dealing with bird species and those of Directive 92/43/EEC for the conservation of natural habitats. The main aim of the Habitats Directive is to promote the maintenance of biodiversity by requiring Member States to take measures to maintain or restore natural habitats and wild species at a favourable conservation status, introducing robust protection for those habitats and species of European importance. This Directive contribute to a coherent European ecological network of protected sites by designating Special Areas of Conservation (SACs) for habitats. The measures need also to be applied to Special Protection Areas (SPAs) of the Birds Directive. Together SACs and SPAs make up the Natura 2000-network.

4.3 Pollution swapping

This rough overview of important European Directives shows that agriculture has to comply with many of these Directives. Oenema and Velthof (2007) illustrated this for nitrogen (Figure 4.1). The application of manure and fertilizer has to comply with policy measures dealing with the emissions to air (NH_3, N_2O, CH_4, CO_2 and NO_x), leaching of NO_3 to groundwater and N- en P-losses to surface water. Furthermore, it becomes clear that it is difficult for farmers 'to do it always all right', because e.g. measures to reduce nitrate losses to surface water will often have an impact on the emissions to other compartments (groundwater and/or air) as a result of the nitrogen cycle. This is called the side effect of measures also known as pollution swapping (Stevens and Quinton, 2009). Another possibility of pollution swapping is the side effect on another substance. For example, increasing the water level to shallow water levels will reduce nitrate leaching (more denitrification) but may increase phosphorus losses. This is one of the reasons why these policies are not always effective and efficient and also why there is often a delay in the implementation and response of these measures (Oenema et al., 2011). Therefore, possible side effects of mitigation options are also discussed in this report.

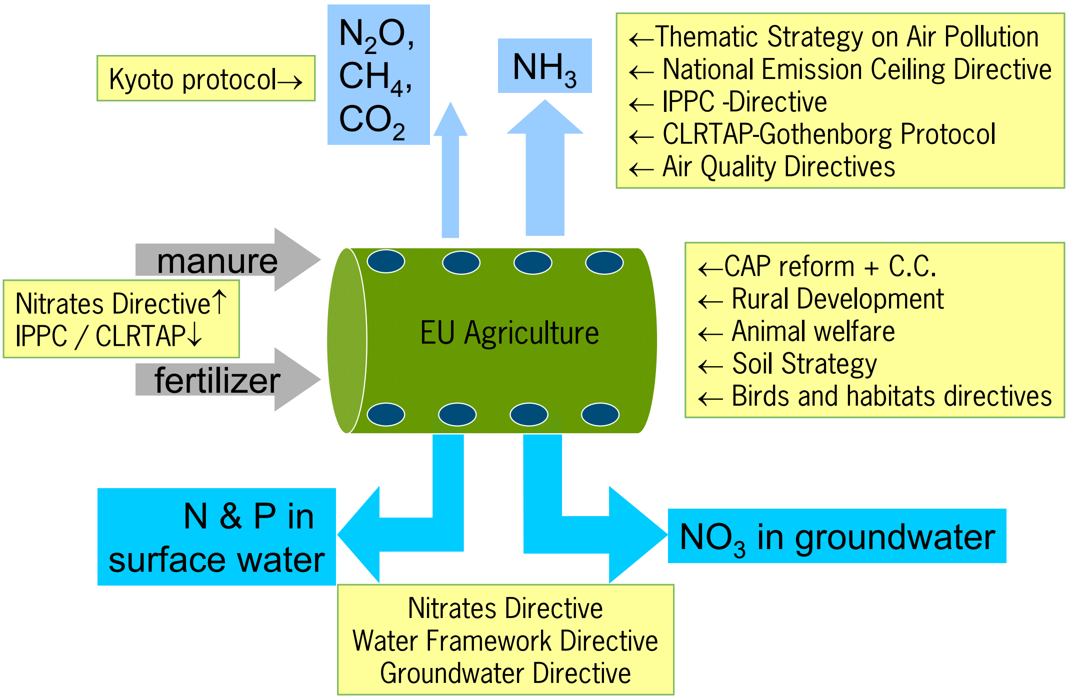


Figure 4.1
Overview of the EU policy instruments that directly or indirectly affect the use and loss of nitrogen in agriculture (after Oenema and Velthof, 2007)

References

COM 231, 2006. Communication from the commission to the council, the European parliament, the European economic and social committee and the committee of the regions. Thematic Strategy for Soil Protection.

COM 232, 2006. DIRECTIVE OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL establishing a framework for the protection of soil and amending Directive 2004/35/EC.

EEC 1975. COUNCIL DIRECTIVE 75/440/EEC of 16 June 1975 concerning the quality required of surface water intended for the abstraction of drinking water in the Member States.

EEC 1979. COUNCIL DIRECTIVE 79/409/EEC of 2 April 1979 on the conservation of wild birds.

EEC 1980. COUNCIL DIRECTIVE 80/68/EEC of 17 December 1979 on the protection of groundwater against pollution caused by certain dangerous substances.

EEC 1991a. COUNCIL DIRECTIVE 91/156/EEC of 18 March 1991 amending Directive 75/442/EEC on waste.

EEC 1991b. COUNCIL DIRECTIVE 91/271/EEC of 21 May 1991 concerning Urban Waste Water Treatment.

EEC 1991c. COUNCIL DIRECTIVE 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources.

EEC 1992a. COUNCIL DIRECTIVE 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora.

EEC 1992b. Council Regulation (EEC) No 2078/92 of 30 June 1992 on agricultural production methods compatible with the requirements of the protection of the environment and the maintenance of the countryside. Official journal of the European Communities, L215, 30/07/1992, 85-90.

EEC 1996a. COUNCIL DIRECTIVE 96/61/EC of 24 September 1996 concerning integrated pollution prevention and control.

EEC 1996b. COUNCIL DIRECTIVE 96/62/EC of 27 September 1996 on ambient air quality assessment and management.

EEC 1999. COUNCIL DIRECTIVE 1999/30/EC of 22 April 1999 relating to limit values for sulphur dioxide, nitrogen dioxide and oxides of nitrogen, particulate matter and lead in ambient air.

EEC 2000. DIRECTIVE 2000/60/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 23 October 2000 establishing a framework for Community action in the field of water policy.

EEC 2001. DIRECTIVE 2001/81/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 23 October 2001 on national emission ceilings for certain atmospheric pollutants.

EEC 2005. ANNEX IV of R1782/2003 List referred to in Article 17 of the Protocol: supplementary adaptations to acts adopted by the institutions. Official Journal of the European Union.

EEC 2006a. DIRECTIVE 2006/7/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 15 February 2006 concerning the management of bathing water quality and repealing Directive 76/160/EEC.

EEC 2006b. DIRECTIVE 2006/11/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 15 February 2006 on pollution caused by certain dangerous substances discharged into the aquatic environment of the Community 76/464/EEC = 2006/11/EC.

EEC 2008. DIRECTIVE 2008/56/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive).

Oenema, O., and Velthof, G.L., 2007. Analysis of international and European policy instruments: pollution swapping. Report 1663.2, Alterra, Wageningen.

Oenema, O., Bleeker, A., Braathen, N.A., et al., 2011. Nitrogen in current European policies. In: *The European Nitrogen Assessment*. ed. M.A. Sutton, C.M. Howard, J.W. Erisman, et al., Cambridge University Press. (in press).

Stevens, C.J., and Quinton, J.N., 2009. Diffuse pollution swapping in arable agricultural systems. Critical Reviews in Environ. Sci. Techn. 39: 478-520.

5 Nutrient management

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This chapter deals with mitigation options related to nutrient management strategies. In Europe nutrient management at farm scale is highly regulated by legislation and influenced by recommendations by intermediaries. Therefore, these aspects are discussed first (Section 5.1 and 5.2). In Section 5.3 and 5.4 are environmental recommendation methodologies and management strategies. In Section 5.5 measures in relation to inorganic fertilizer management are discussed and in Section 5.6 measures related to manure management. Finally, a manure surplus management strategy is discussed (5.7). Some of the measures are already compulsory in some countries or will be part of a Code of Good Agricultural Practices (compulsory or voluntary), other are potential measures that can be implemented in the nearby future.

5.1 Nitrate Directive and Nitrate Action Programmes

The intensification of agriculture, as a consequence of an increased demand for food and feed, has resulted in higher inputs of production means like fertilizers, pesticides, etc. As the efficiency of nutrient inputs is never 100%, a non scientific based fertilization can lead to excessive losses of nutrients to the environment. Furthermore, in some EU countries or EU regions, there is a huge production of animal manures. Since transportation to regions with fewer cattle breeding and processing the surplus of manure are both expensive, the easiest and cheapest way is to apply the manure on local agricultural land. Therefore, farmers will use as much as possible manure on their own fields. Without national or international regulations, emissions of nitrogen and phosphorus from agricultural activities to groundwater, surface water and the atmosphere would have been a significant problem in the European Union. Since the '80's of the last century, international EU and national legislative bodies became involved in environmental legislation (De Clercq and Sinabell, 2001) (Chapter 4).

A number of EU directives relate to agricultural practice and water quality standards. The most important are the Nitrate Directive (91/676/EC) and the Water Framework Directive (2000/60/EC). Today, the Nitrates Directive is the most important Directive that regulates the amount of manure applications. In the so called nitrate vulnerable zones (NVZ)' Member State have to establish a Nitrate Action Programme (NAP) or several NAPs (e.g. for different NVZs). There is also the possibility to set up a NAP for the whole country without defining NVZs. In the NAP the maximum amount of manure application is regulated (max. 170 kg N from manure per ha). Furthermore, the total amount of nitrogen has been regulated based on the balance between crop requirement and nitrogen supply by manure and fertilizer, taking also into account the release of nitrogen from the soil. Based on objective criteria like a long growing season, crops with high N uptake, high precipitation and/or soils with a high denitrification capacity, it is possible to obtain derogation of the 170 kg N per ha. Especially, for pasture land higher values have been granted in some countries. Finally, the Nitrate Directive promotes the use of good farming practices. The water quality of groundwater and surface water has to be reported every fourth year. If the water quality did not ameliorate the Action Programmes have to be adapted.

The implementation of the Nitrate Directive has led to different nutrient management legislation within EU countries as a result of local differences (http://ec.europa.eu/environment/water/water-nitrates/index_en.html). These are due to variation in natural conditions, like climatic conditions or geographical variations, variation in soils, structure and size of the farms, intensity of farming, etc. (Schröder et al., 2004). Therefore, environmental problems linked with nutrient use can differ between regions and it is quite evident that policy makers respond differently to environmental problems in each country.

5.2 Fertilizer recommendation methodology

Fertilizer recommendations are very important for a high quality crop production. In the past these recommendations were only focussed on agronomical and economical aspects. Nowadays more and more attention is given to environmental aspects (agro-environmental approach), which will be discussed in the next Section (5.3). With respect to fertilizer recommendations some characteristic aspects have to be mentioned in general, which are:

- soil fertility rating
- soil sampling
- soil analyses

Soil fertility rating
Crop productivity depends on several factors, one of them is the availability of nutrients in the soil, which is called (chemical) soil fertility. Many techniques can be used to evaluate the status of soil fertility, one of them is soil analysis. Soil analysis techniques are chemical extraction methods which measure a part of the total nutrient supply from the soil, i.e. the nutrient-supplying power of a soil. These methods must be calibrated previously against nutrient rate experiments in the field. The advantage of soil analyses compared to nutrient-deficiency symptoms of plants or plant tissue analysis is that they determine the needs of the soil before the crop is planted. In most countries there is a well developed soil testing programme and this is the common way to evaluate soil fertility. There is no doubt that using a fertilizer recommendation system, based on soil test data (Sharpley et al., 2005) the input of nutrients will be more in agreement with the real needs of the crop.

Table 5.1 shows the relation between nutrient content of the soil, deficiency status and fertilizer recommendations.

Table 5.1
Relation between nutrient content of soil, deficiency status and fertilizer recommendations

No	Nutrient content	Deficiency status	Recommended fertilization
1	Very low	Acute deficiency	Much higher fertilization than crop uptake
2	Low	Latent deficiency	Higher fertilization than crop uptake
3	Sufficient	Optimal content	Maintenance (normal) fertilization
4	High	Luxury content	Lower fertilization than crop uptake
5	Very high	Luxury content to toxicity	No fertilization

This scheme holds for most of the nutrients, except nitrogen. The P recommendation for a soil with a very low P content will be to apply more than the output by the crop while at a high P level in the soil, the P recommendation will be to apply less than the output.

For nitrogen, the recommendation based on soil testing is much more complicated because, beside the N in soil organic matter, there is no real accumulation of nitrogen in the soil. Using a system based on measurements of N_{min}, the N recommendation should be based on the total N-need of the crop (roots included) + the accepted residual nitrogen at harvest time, minus the N_{min} before sowing or planting, minus the estimated N released from soil organic matter (SOM) and applied organic material as given in Table 5.2. In this balance, the most uncertain factor is the N release from soil organic matter and, to a lesser extent, the effective nitrogen from applied organic material.

Table 5.2
Calculation of the theoretical N fertilization (Hofman and Van Cleemput, 2004)

N need of the crop +	=	N_{min} before planting +
Residual N_{min} in the soil profile at harvest*		N mineralisation + N fertilisation

* The residual N_{min} in the soil profile at harvest is the amount of mineral N which remains in the rooting zone at optimum N fertilization and at the time of maximum N uptake

Two factors in soil testing are important for the relevance of the data and/or the interpretation of the data, i.e. sampling procedure and the extraction procedure.

Sampling procedure
To obtain reliable results, soil samples must be representative for the field or area under study. Selecting a sampling strategy involves a decision on how many samples to collect and what sampling design to use. Ideally, samples should be located evenly over the field or area. A completely regular sampling network of systematic sampling is preferable (Figure 5.1a). However, it can be biased if it coincides in frequency with regularly spaced drains or with a banded placement of fertilizers. For this reason, statisticians sometimes prefer a kind of random sampling for computing unbiased means and variances (Figure 5.1b) A drawback of this procedure is that complete randomization can lead to an uneven distribution of sampling points (Figure 5.1b) unless many points can be measured which is usually a problem because of costs drawbacks. A good compromise between regular and random sampling is to locate individual points at random within regularly lay out strata or blocks. This is called a stratified random sampling design (Figure 5.1c) where sampling is done according to soil types, different histories of crop management, etc.

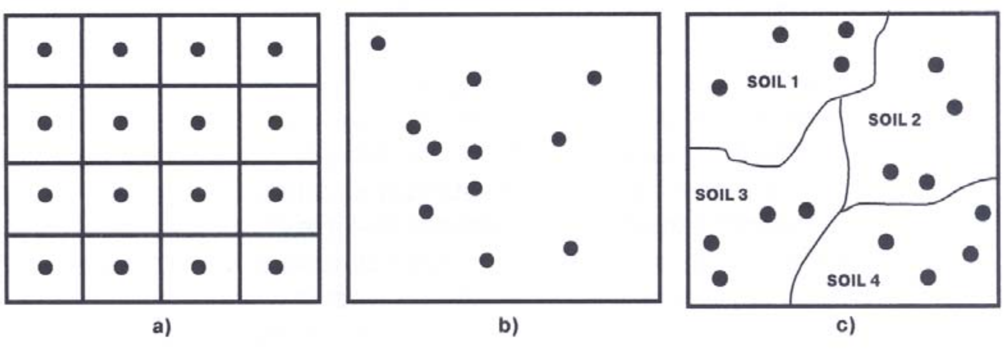

Figure 5.1
Examples of sampling design (a: regular sampling and b: simple random sampling design and c: stratified random sampling)

A working unit for a sampling area is about 2 ha, which means that if a field is much larger than 2 ha, several samples should be taken. Per maximum unit of 2 ha, as a rule of thumb about at least 15 augerings should be done and the 15 subsamples should be put together, mixed and a bulk sample should be taken.

Soil analysis procedure / extraction procedure
Chemical analysis of soils is based on the principle that chemical solutions can rapidly, reproducibly and inexpensively assess the nutrient supplying capacity of a soil and other soil properties that effect plant growth, i.e. pH, SOM, etc. Chemical extraction is almost always conducted on dried, ground and sieved soil samples (except measurements of mineral N in soil).
There are numerous extraction solutions which all give an idea about the nutrients in the soil available for plant growth. To make a choice of the extraction solution, the following criteria have to be taken into account:

- the extraction solution must have a sufficient power to bring the nutrient under study in solution;
- it must contain an ion which can replace the sorbed ions on the adsorption complex;
- it can make complexes so that the released ions remain in solution.

Furthermore, it must be known if we want to measure the nutrient intensity (I) or the nutrient capacity (Q). The nutrient intensity represents an amount of a certain nutrient which is in soil solution. This amount is often determined with a very weak extractant. The nutrient capacity refers to the amount of a certain nutrient which can be desorbed from the soil complex by the extraction solution. The ratio Q/I is a measure of the buffer capacity of the soil for this specific nutrient.

Because NO_3-N is not sorbed on the soil complex, due to a negative charge, an extraction with water, KCl or even a more aggressive extraction solution will give more or less the same result. In this case, we measure the nutrient intensity. The behaviour of phosphorus in soils differs completely from nitrate. In the soil solution, only a very small part of the total P is present under the form of $H_2PO_4^-$ in acid soils or as HPO_4^{2-} in more alkaline soils. Most of the phosphorus is present in the soil in different ways:

- sorbed at the surface of clay minerals or organic matter together with cations like Ca^{2+}, Fe^{3+}, Al^{3+} making bridges between the clay or organic matter and the phosphate;
- sorbed at the surface of Fe- and Al-hydroxides in more acid soils;
- sorption with Ca^{2+} or $CaCO_3$ and possible precipitation of calcium phosphates.

After choosing a certain extraction solution, some of the fractions mentioned above will go into solution and are measured afterwards. It means that in this case both the intensity and the capacity are measured. Table 5.3 gives an overview of used soil P tests based on a questionnaire within COST action 832.

Many laboratories have their own extraction procedures, which makes it difficult to compare results. Therefore, it is necessary always to mention the methodology used, because extraction solutions can strongly differ in their ability to extract a nutrient. Neyroud and Lischer (2003) give an excellent overview of the methodologies used to extract phosphorus as soil P test. In order to compare systems, they organized a soil exchange program: 16 P methods were compared on 135 soils from 12 countries. The amount of extracted P decreased in the order $P_{total} > P_{oxal.} > P_{AL} > P_{Me3} > P_{Bray} > P_{AAEDTA}$, P_{DL}, $P_{CAL} > P_{Olsen} > P_{paper\ strip}$, P_{AAAc}, $P_{Morgan} > P_{H2O}$, P_{CO2}, P_{CaCl2}.

Table 5.3

Overview of soil P tests methods used within Europe

P test	Countries	Method (soil:solution ratio)	Reference
AL	BE, HU, NL, NO, RO, SE	1:20 (w/v) with 0.1 M ammonium lactate + 0.2 M acetic acid, pH 3.75, 2 h shaking, N & SE 1.5 h shaking, BE 4 h shaking	Egnér et al., 1960
AmAc	CH	ammonium acetate	Anonymous, 2004
An Ex	UK	An ion exchange resin	Somasiri and Edwards, 1992
Bray-1	EL, IT	1:7 (w/v) 0.03 M ammonium fluorid + 0.0125 M HCl, pH 3.5, 1 min shaking	Bray & Kurtz, 1945
Bray-2	ES	1:8 (w/v) 0.03 ammonium fluorid, 0.1 M HCl, pH 1, 40 minutes shaking	Bray & Kurtz, 1945
$CaCl_2$	NL, PL	1:10 (w/v) with 0.01 M calcium chloride, 2 h shaking	Houba et al, 1990
CAL	AT, DE	1:20 (w/v) 0.05 M calcium lactate + 0.05 M calcium acetate + 0.3 M acetic acid, pH 4.1, 2 h shaking	Schüller, 1969
DL	AT, DL, PL	1:50 (w/v), 0.02 m calcium lactate + 0.02 M hydrochloric acid, pH 3.7, 1 h shaking	Egnér and Riehm, 1955
DYER	FR	1:5 (w/v) 2% Citric acid, pH 2.0, shaking time 4/15 h.	Dyer, 1894
EDTA-Ac	BE	1:5 (w/v), 0.5 N ammonium acetate + 0.002 M EDTA, pH 4.65, 0.5 h shaking	Cottenie et al., 1979
EDTA-Ac	CH	1:10 (w/v), 0.5 M ammonium acetate + 0.5 M acetic acid, 0.02 M EDTA, pH 4.65, 1 h. shaking	Anonymous, 1996
Joret-Hebert	FR	0.2 M ammonium oxalate, pH 7.2, 2 h. shaking time	Joret and Hebert, 1955.
MoCa	RO	0.3% ammoniumheptamolybdate, 0.01 M calcium chlorid, pH 4.3, 1 h. shaking	Borlan et al., 1982
Morgan	FI, IE	6.5:30 (v/v), sodium acetate, pH 4.8., 0.5 h shaking	Vuorinen & Mäkitie, 1955 (FI) Aura, 1978 (FI) Morgan, 1941 (IE)
Olsen	DK, EL, IT, EL, UK	1 : 20 (w/v), 0.5 M sodium acetate, pH 8.5, 1 h shaking	Olsen et al, 1954
Water	AT	1:20 (w/v) extraction with water	Önorm, 2005
Water	NL	C, 22 h°1:60 (v/v) with water at 20h incubation, 1 h shaking	Sissingh, 1971
Water	NL	1:2 (v/v) water, 20 minutes shaking	Sonneveld, 1990
Water	CH	1:10 (w/v) with CO_2-enriched water, pH 3.9	Anonymous, 1996.

Results of soil P tests are compared with tabulated critical limits (depending on the extraction procedure used) to evaluate soil fertility and to come up with a fertilizer recommendation (as in general form mentioned in Table 5.1). The P recommendations are usually based on soil-P values and there are three broad fertilizer strategies:

- No fertilizer required for optimum production for a number of years when the test is high
- Maintenance P required when the value is moderate / sufficient
- Build-up of P necessary when the value is low.

The defined deficiency classes (upper and lower boundaries) and fertilization recommendation differ between countries (Tunney et al., 1997; Sibbesen and Sharpley, 1997; Dawson and Johnson, 2006). Table 5.4 gives, as an example, an overview for the P ammonium lactate (PAL) extraction method that is used in some European countries. All values are expressed in mg P_2O_5 per kg.

Table 5.4
Defined deficiency classes for the PAL soil P test as used in some European countries (all expressed in mg P kg-1)

Country		soil type	soil P fertility classification					Reference
			very low	low	moderate	high	very high	
Belgium	grassland		<80	80 - 184	185 - 254	255 - 600	>600	Boon et al., 2009; Reubens et al., 2010
Belgium	arable land		< 50	50 - 114	115 - 184	185 - 500	> 500	Boon et al., 2009; Reubens et al., 2010
Hungary		Group I	< 14	14 - 26	27 - 43	44 - 78	≥ 79	
		Group II	< 18	18 - 35	36 - 52	53 - 86	≥ 87	
		Group III	< 22	22 - 43	44 - 65	66 - 104	≥ 105	Németh, 2003;
		Group IV	< 27	27 - 52	53 - 74	75 - 117	≥ 118	Csathó et al., 2007
		Group V	< 31	31 - 65	66 - 87	88 - 130	≥ 131	
		Group VI	< 81	81 - 180	181 - 250	251 - 400	≥ 401	
NL	grassland	löss soils	< 57	57 - 82	83 - 117	118 - 174	> 174	Anon., 2008
	grassland	river clay soils	< 61	61 - 99	100 - 134	135 - 200	> 200	Anon., 2008
	grassland	orther soils	< 70	70 - 117	118 - 156	157 - 218	> 218	Anon., 2008
Norway				< 45	45 - 74	75 - 144	≥145	Krogstad et al., 2008
Sweden			< 15	15 - 44	45 - 84	85 - 144	> 145	Kirchmann and Andersson, 2001

Losses can be minimized by maintaining moderate soil P levels, but can not always be prevented, because a build-up of phosphate in the ploughed layer is necessary to meet with the crop requirements. However, minimizing P applications on soil with high P levels is important because the P loss often increases strongly above moderate/sufficient soil P level due to the non-linear relationship between the soil P content and the soil P concentration (Schoumans and Groenendijk, 2000). Furthermore, reducing chemical P fertilizer at a high soil P status will not have a negative effect on crop yields, and will save some money if chemical P fertilizers are used.

It must be clear that an integration of nutrient supply via mineral fertilizers and via manure is necessary to reduce nutrient losses. Robust recommendation systems provide a good estimate of the amount of available nutrients supplied by manure application. Therefore, it is necessary that there is an analysis or at least a reliable estimate of the nutrient content and nutrient release of the applied animal manure or other organic by-products. The amount of phosphorus applied with organic manure can be subtracted from the calculated needed P fertilization. This is based on the assumption that the efficiency coefficient of P in manure is 100%. The efficiency coefficient for nitrogen in animal manure depends on a lot of factors like the C/N ratio, the way of application (direct incorporation or not), the pH of the soil, climatic conditions, etc. When using good agricultural practices the efficiency of nitrogen in animal manure can be around 70 %. There is no doubt that a correct interpretation of the available nutrients in animal manure and other organic amendments will mitigate nutrient losses to the environment and will be an economic benefit for the farmer since less mineral fertilizer has to be applied. The drawback is that analyses of all applied organic products are hardly feasible. Furthermore, there is still quite some uncertainty about the nitrogen efficiency coefficients.

5.3 Environmental recommendation methodologies

In order to reduce nutrient losses to groundwater and surface water the nutrient recommendations have already been overruled by legislation, e.g. by means of manure application standard in the Nitrate Directive for Nitrate Vulnerable Zones (max. 170 kg N/ha of animal manure). Furthermore, Action Programmes have to be adapted, and become more restrictive regarding N and P application, in case the water quality of groundwater or surface water does not improve. Although in many countries the focus is on nitrogen and nitrate losses to groundwater, sometimes also additional nutrient management strategies are applied at fields with a high P status (Ireland, The Netherlands) or phosphorus saturated areas (Flanders). To locate such P risk areas several approaches have been developed, like the phosphate saturation degree (Breeuwsma et al., 1995) and modifications of this approach (Hartikainen 1982; Nair and Graetz, 2002)). The phosphate saturation degree is only related to the soil (potential P risk) and not to aspects related to the transport of phosphorus. These aspects are incorporated into P index approaches which are mainly oriented on overland P losses via surface runoff and erosion: manure and fertiliser application rates, slope, precipitation surplus/rainfall event etc. (Lemunyon and Gilbert, 1993; Bechmann et al., 2007). Regarding P losses via leaching a few approaches are available, the PLEASE approach has been developed to assess such losses at field scale (Schoumans et al., 2009).

5.4 Agro-environmental nutrient management strategies

Balance

With respect to phosphorus it is important to set up a phosphorus balance at farm and field scale, taking into account the amount of P that is available in the soil, because in many western European countries the amount of available P in the soil is much higher than the annual amount of P applied. At high soil P status the P balance can be negative and at low P status the balance can be positive to increase the soil P status to a sufficient level (Tunney et al., 1997). In some countries such approaches have been introduced into legislation (e.g. via the Nitrate Action Programmes like in Ireland and The Netherlands). In fact, also with respect to nitrogen a balance approach is important and has been described in the Nitrate Directive (91/676/EC). So, the first step of mitigation options deals with the reduction of annual manure and fertiliser application rates by taking into account (Factsheets: Dana, 2010; Delgado, 2010; Garnier, 2010a and b; Garnier and Harris, 2010; Hofman, 2010a, b and c; Newell Price, 2010a and b; Krogstad and Bechmann, 2010b; Taylor, 2010b and c):

- soil nutrient level and crop response;
- risk of nutrient losses (such as nitrate vulnerable zones, phosphorus saturated or vulnerable zones);
- integrate fertilizer and manure nutrient supply.

Since such approaches are difficult to set up, Denmark used an approach where for nitrogen 80% of the economic optimum is used as application standard as a first approach to introduce a more environmental based approach. In other regions like Flanders, a reduction of fertilizer use can be done on a voluntary basis in certain regions (agreement with the government) which will be compensated financially for yield losses. One of the most important agreements are the so-called management agreements to protect surface and groundwater in zones where surface water is used for drinking water purposes and in nitrate vulnerable zones. The maximum amount of nitrogen that can be applied is at least 30% less than without an agreement and in autumn there must be a measurement of the residual mineral N in soil. This is compensated by an amount of 537 euro/ha for grassland and 302 euro/ha for arable land. The effect of reduced nitrogen fertilization on the residual NO_3-N in autumn is given in Table 5.5. From this table it can be concluded that the residual NO_3-N in autumn has been reduced by about 50 % with some variation between preceding crops. These results are somewhat biased because the agreement is already implemented since 2001, and all the fields were once a residual N higher than 90 kg NO_3-N/ha has been found were excluded. Nevertheless, it can be noticed that with some compensation to farmers, the residual mineral N in autumn can significantly decrease.

Special attention should be given to peat soils because these soils are very sensitive for P leaching. Especially in the Nordic countries of the EU, large areas of low decomposed sphagnum peat is cultivated and these soils can contribute significantly to the P loading of surface and groundwater (Renou-Wilson and Farrel, 2007; Van Beek et al., 2007).

Table 5.5
Mean residual NO_3-N (kg/ha) as a function of previous crop in autumn and amount of fields sampled in 2009 without and with further restriction of nitrogen application (VLM, 2010)

Crop	**Without further N-restrictions**		**With further N-restrictions**	
	Sampled fields	Mean NO_3-N residue	Sampled fields	Mean NO_3-N residue
Grass	3003	83	6135	44
Maize	1253	91	4521	57
Beets	48	54	1521	31
Cereals	107	89	4943	41
Potatoes	164	156	938	90
Vegetables	124	179	812	47
Fruit trees	26	100	866	24
Ornamental plants	10	154	68	91
Other crops	13	140	214	40
Total	4748	90	20018	47

Placement
Increasing economical pressure on agriculture and the protection of the environment are the driving forces towards a more efficient use of production means like fertilizers and pesticides (Nychas, 1989). Nitrogen losses can be minimized by a better adaptation of the fertilization to the needs of the crop (see balance) but also by a better placement of the fertilizers by row or band application (Factsheet: Delgado, 2010). Especially for crops with large distances between the rows and/or with a restricted root distribution and for crops which are earthed up soon after fertilization, positive results of row or band application of fertilizers can be expected, due to a better nutrient availability with lower residual mineral N at harvest and lower leaching losses (Hofman et al., 1992). Other benefits are a delay of nitrification of NH_4-N (due to a high salt concentration nearby the fertilisers) which reduces possible losses by leaching because NH_4-N is adsorbed at clay and humus components. Another advantage, although it has only indirectly to do with mitigation options for reducing nutrient emissions to surface waters, is a reduction of NH_3 losses. By row or band application the fertilizers are incorporated to a depth of a few cm which reduces possible NH_3 volatilization to about nil.

Also for P, the applied P is used more efficiently when it is placed nearby the roots instead of broad field spreading. Depending on the crop the amount of recommended P can be reduced remarkably when P is applied in bands compared to broad field spreading (Van Dijk, 2003).

Time of spreading
Besides the placement of nutrients also the time of spreading is important for reducing nutrient losses. Within the Nitrate Action Programme the Member States are obliged to restrict the period of spreading, but the closed period for manure and fertilizer application still differs between the Member States (http://ec.europa.eu/environment/water/water-nitrates/index_en.html). Table 5.6 gives some examples of closed periods in some of the European countries.

Table 5.6
Closed periods of some of the EC-member states

Country	Remarks[*)]	harvest	1-sep	15-sep	1-okt	15-okt	1-nov	15-nov	1-dec	15-dec	1-jan	15-jan	1-feb	15-feb	28-feb
Be (Flanders)	other		■	■	■	■	■	■	■	■	■	■	■	■	
	clay soils					■	■	■	■	■	■	■	■	■	
	farmyard manure & compost							■	■	■	■				
Netherlands	grassland sand and loss		■	■	■	■	■	■	■	■	■	■			
	grassland clay and peat soils			■	■	■	■	■	■	■	■	■			
Denmark	other	■	■	■	■	■	■	■	■	■	■	■			
	grassland and winterrape				■	■	■	■	■	■	■	■			
Norway							■	■	■	■	■	■	■	■	
Ireland	based on regional rainfall (South end 12/1)			■	■	■	■	■	■	■	■				
North Ireland				■	■	■	■	■	■	■	■	■			
Italy	NVZ: mineral fertilizers; organic fertilizer., farmyard manure (FYM)						■	■	■	■	■	■	■		
	NVZ: poultry manure						■	■	■	■	■	■	■	■	■
	NVZ: liquid manure on grass, winter cereals, horticult., fruit trees						■	■	■	■	■	■	■		
	NVZ: liquid manure other crops						■	■	■	■	■	■	■	■	■

[*)] closed also in other periods for frozen and snow covered soil

In fact, also in the non-closed periods high nutrient losses can occur, especially after high rain events or long rainy periods. From this point of view a good agricultural practice guideline could be to avoid manure and fertilizer application before predicted high or long rainfall events, and a demand for injection of manure or ploughing directly after application (Factsheets: Haygarth 2010c; Garnier, 2010b; Garnier and Harris, 2010 and Taylor, 2010c).

High risk areas
As mentioned before, the applied amount of nutrients (manure and fertilizers) should be reduced at high risk areas. This can be done on a whole field, but also on parts of fields, e.g. feeding places that have been trapped, nearby surface water (especially with a high slope), parts with compacted areas caused by wheel tracks etc. It is difficult to describe the exact circumstances, but both farmers and water managers can draw attention to such high risk areas (Factsheets: Haygarth 2010c; Garnier, 2010a; Newell Price, 2010a and Taylor, 2010 b).

5.5 Inorganic fertilizer management

One of the approaches to mitigate N and P losses is a reduction of fertilizer application rates. Limiting the amount of N fertilizer applied to crops will reduce the quantity of residual nitrate in the soil after harvest (Johnson et al., 2002) to a certain extent, that will depend on the mineralization capacity of the soil. On the short term, limiting P fertilizer rates will have no large effect on P losses. On the other hand, reducing P fertilizer rates will reduce both particulate and soluble P losses by erosion through a reduction in soil P reserves at the soil surface.

In addition to the application rate, attention should also be paid to the N, P and K ratio of fertilisers, in relation to the type and amount of manure that is applied to the fields. E.g. fine tuning on N can result in a high P and/or K surplus for some types of fertiliser. Furthermore, the chemical form of the nutrients in the fertilizer is important. Not only if N is applied as NO_3 or NH_4, but also the dissolution rates of the nutrients can be important, e.g. fast and slowly available N and P. There are many options to deliver fertilizer nutrients in such a way that they are in line with the need of nutrients of the different crops during the year (Factsheets: Dana, 2010 and Newell Price, 2010e). In fact, gradual nutrient fertilization is an example of such an approach that is more and more frequently used in practice by farmers.

5.6 Manure management

Similar to inorganic fertilizer management, also the manure application rate can be reduced in relation to plant requirements. However, with respect to manure application it should be taken into account that also the soil fertility has to be kept at a sufficient level (in relation to organic matter content, micronutrients, good quality status for the fauna etc). Therefore, it is important to notice that other options are available to reduce the nutrient losses via manure management. On farms where currently limited storage capacity is available, expanding of facilities for collection and storage of slurry and dirty water is needed in order to allow spreading at times when there is a low risk of runoff and when there is an actively growing crop to utilise the nutrients supplied in the manure (Factsheets: Newell Price, 2010c; Newell Price and Morvan, 2010 and Taylor, 2010d).

Minimizing the volume of dirty water produced can also help to control the volume of liquid manure to be stored and spread. This can be reduced via reducing (1) unnecessary dirty yard areas, (2) excessive use of water in washing down yards, buildings, etc. (3) unnecessary mixing with clean water from uncovered clean yard areas and from roofs, etc., and by (4) roofing over yard areas and (5) covering dirty water and slurry stores. The method reduces the volume of liquid to be stored and handled but has no effect on the total amounts of N or P, but helps to reduce the storage capacity (Factsheets: Newell Price and Morvan, 2010 and Taylor, 2010d).

Composting solid manure (Factsheet: Haygarth, 2010a) will lead to a more friable, stable, and spreadable product with a reduced volume. During this process, the manure is sanitised and the readily available N content is reduced, thereby lowering nitrate losses when the compost is spread. No effect on phosphorus is expected.

Since the risk of pollution by slurry during runoff events is higher than from solid manure, it would be better to change from a system where the manure from housed animals is collected as liquid slurry to a system where animals are kept on a bed of straw to produce a solid manure. Also, solid manure is more easily stored than slurries, which leads to a smaller risk of pollution. This measure will reduce both N and P loss (Factsheet Taylor, 2010a).

Finally, rapid incorporation of manure into the soil after field application (Factsheet Haygarth, 2010b) is an important action to reduce P losses via surface run-off. Furthermore, ammonia volatilization will be reduced, but this can have a negative side effect on nitrogen losses because more nitrogen is added to the soil.

5.7 Manure surplus management

In Section 5.2 the nutrient balancing approach was discussed, that often leads to less manure being used. As a consequence, in many regions the manure surplus will increase in terms of volume and amount of nutrients, and consequently more manure storage capacity is needed. There are different ways to reduce this side effect. In the first place, often the nutrient content of fodder can be reduced, which will be discussed in more detail in Chapter 7. In case the amount of nutrients in fodder is reduced, also the amount in excreta will be reduced and more manure (with fewer nutrients) can be applied to the soil. A final surplus has to be transported to neighboring farms or less intensive husbandry areas. This measure is environmentally efficient if the distance is under a threshold. This one has been evaluated to 80 km in a study case, indicating that this distance has not to be a long distance (Lopez-Ridaura, 2009). Sometimes it can be worthwhile to separate a manure surplus in a liquid fraction (with a high N and a low P content) and a more solid fraction (high P and low N). The liquid fraction can be applied on the land, and less manure (only the solid fraction) has to be transported. Especially on dairy farms such an approach can be successful (Schröder et al., 2009). Instead of transporting the manure to another area, the manure can also be collected and processed, with energy recovery. Although the ash will be rich in P, there seem to be possibilities to reuse the P (Schoumans et al., in prep.). Since more than 80% of all P that is mined as phosphate rock is used by agriculture (as feed and fertilizers), this can be an important step to close the P cycle in a sustainable way.

In conclusion

1. Nutrient management strategies, like agro-environmental recommendations, are useful tools for setting up a more sustainable agricultural management practice.
2. With respect to the selection of, and placement of, fertilizers (NPK), the surplus of all components has to be minimised in relation to nutrient uptake of the crop and the composition of the applied manure composition.
3. Storage capacity of manure should be (more than) sufficient to avoid manure application under high risk conditions (high risk areas and high risk time).
4. Rapid incorporation of manure into the soil reduces N losses via ammonia volatilization and P losses via surface runoff considerably.
5. Since the phosphorus supply in the mines is running out as a result of the high agriculture P use, the phosphorus surplus of manure have to be, and can be, reused.

List of factsheets

Dana, D., 2010. Method for determining the economically optimal rate of phosphorus fertilizer application. [FS]
Delgado, A., 2010. Fertilizer placement near crops. [FS]
Garnier, M., 2010a. Do not apply fertilizer to high-risk areas. [FS]
Garnier, M., 2010b. Do not spread farmyard manure to fields at high-risk times. [FS]
Garnier and D. Harris, 2010. Avoid spreading fertilizer to fields at high-risk times. [FS]
Haygarth, P.M., 2010a. Compost solid manure. [FS]
Haygarth, P.M., 2010b. Incorporate manure into the soil.[FS]
Haygarth, P.M., 2010c. Adopt batch storage of slurry. [FS]
Hofman, G., 2010a. Use a fertilizer recommendation system with soil testing. [FS]
Hofman, G., 2010b. Reduce N-application. [FS]
Hofman, G., 2010c. Reduce P-application based on soil P status. [FS]
Newell Price, J.P., 2010a. Do not apply P fertilizers to high P index soils. [FS]
Newell Price, J.P., 2010b. Incinerate poultry manure. [FS]
Newell Price, J.P., 2010c. Minimize the volume of dirty water produced. [FS]
Newell Price, J.P., 2010d. Price Integrate fertilizer and manure nutrient supply. [FS]
Newell Price, J.P., 2010e. Reduce fertilizer application rates. [FS]
Newell Price, J.P., & T. Morvan, 2010. Adopt batch storage of solid manure. [FS]
Krogstad, T. & M. Bechmann, 2010a. P Index - a tool to evaluate risk of P runoff. [FS]
Krogstad, T. & M. Bechmann, 2010b. Reduced P application in peat soil. [FS]
Taylor, M.J., 2010a. Change from slurry to a solid manure handling system. [FS]
Taylor, M.J., 2010b. Do not apply manure to high-risk areas. [FS]
Taylor, M.J., 2010c. Do not spread slurry or poultry manure to fields at high-risk times. [FS]
Taylor, M.J., 2010d. Increase the capacity of farm manure (slurry) stores. [FS]
Turtola, E., 2010. Reducing P content of common NPK fertilizers. [FS]

References

Anonymous, 1996. Méthodes de référence de Stations Federale de Recherches Agronomiques, vol 1. Analyse de Terre pour Conseille de Fumure. Edition des Stations Fédérales, Zürich-Reckenholz. Switzerland.

Anonymous, 2004. Méthodes de référence des stations fédérales de recherches agronomiques, dernière mise à jour 2005. Agroscope Reckenholz-Tänikon (ART), 8006 Zurich-Reckenholz.

Anonymous, 2008. Adviesbasis bemesting grasland en voedergewassen. Commissie Bemesting Grasland en Voedergewassen. Animal Science Group. Lelystad.

Aura, E. 1978. Determination of available soil phosphorus by chemical methods. J. Sci. Agric. Soc. Finland, 50: 305-316.

Baert, L., Depuydt, S., De Smet, J., Hofman, G., Scheldeman, K., Vanderdeelen, J., Van Meirvenne, M., Lookman, R., Merckx, R., Schoeters, I., Vlassak, K., De Gryse, S., Hartmann, R., Seuntjes, P., Verplancke, H. and Verschoore, P. 1997. Fosfaatverzadiging van zandige bodems in Vlaanderen. Vlaamse Landmaatschappij, Brussel, 1997, 143 p.

Bechmann, M.E., Stalnacke, P. and Kvaerno, S.H. 2007. Testing the Norwegian phosphorus index at the field and subcatchment scale. Agric. Ecosyst. Environ. 120: 117-128.

Boon, W., Ver Elst, P., Deckers, S., Vogels, N., Bries, J. and Vandendriessche, H. 2009. Wegwijs in de bodemvruchtbaarheid van de Belgische akkerbouw- en weilandpercelen (2004-2007). Bodemkundige Dienst van België, Heverlee, België. 149 p.

Borlan, Z., Hera, Cr. et al. 1982. Opredelenie podvijnîh fosfatov, ekstraghiruiemîh molibdatom ammonia i hloristîm kaliem-MoCa în: Agrohimiceskie metodî issledovania fosfatnogo rejima pociv. Sbornik metodov.Acad. SH Nauk G.D.R.Instit. Pitania Rast. Jena:45-48.

Bray, R.H., and Kurtz, L.T. 1945. Determination of Total, Organic, and Available Forms of Phosphorus in Soils. Soil Sci. 1945, 59, 39–45.

Breeuwsma, A., Reijerink, J.G.A. and Schoumans, O.F. 1995. Impact of manure on accumulation and leaching of phosphate in areas of intensive livestock farming, In: K. Steele (ed.), Animal waste and the land-water interface. Boca Raton (USA), Lewis, 1995, pp. 239-249.

Cottenie, A. 1979. Workshop on the standardisation of analytical methods for manure, soils, plant and water. C.E.E. Agricultural series. EUR 6369 EN.

Csathó, P., Árendás, T., Fodor, N. and Németh, T. 2007. A legelterjedtebb hazai trágyázási szaktanácsadási rendszerek tesztelése szabadföldi kísérletekben. Agrokémia és Talajtan, 56:1. 173-190.

Dawson, C.J. and Johnston, A.E. 2006. Agricultural Phosphorus in Relation to Its Effect on Water Quality. Proceedings of the International Fertiliser Society. No. 590, ISBN 978-0-85310-227-4. York, UK.

De Clercq, P. and Sinabell, F. 2001. EU legislation and multinational environmental legislation with respect to nutrient management. In: De Clercq P., Gertsis A.C., Hofman G., Jarvis S.C., Neeteson J.J. & Sinabell F. (eds.). Nutrient management legislation in European countries. Department of Soil Management and Soil Care, Faculty of Agricultural and Applied Biological Sciences, Ghent University, 347pp.

Dyer, B. 1894. Determination des matieres minerals assimilabes par les plants. Ann. Agron 291-298.

EEC 1991. COUNCIL DIRECTIVE 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources. Official Journal of the European Communities, L 375, 31/12/1991,1-8.

EEC 2000. DIRECTIVE 2000/60/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 23 October 2000 establishing a framework for Community action in the field of water policy. Official Journal of the European Communities, L327, 22/12/2000, 1-43.

Egnér, H. and Riehm, H. 1955. Die Doppellaktatmethode. In: R. Thon, R. Hermann & E. Knikemann (eds.). Die Untersuchung von Boden Verbandes Deutscher Landwirtschaftlicher Untersuchungs und Forschungsanstalten. Methodenbucht I. Nuemann Verlag, radenbeul and Berlin. Based on: Egnér H., 1932. Meddlande report No. 425. Från Centralanstalten for Forsoksvasendet poa jordbrukssomrao det, Avdelnignen Forlantbrukschemie, Stockholm, Nr. 51.

Egnér, H., Riehm, H., and Domingo, W.R. 1960. Untersuchungen über die chemische Bodenanalyse als Grundlage für die Beurteilung des Nähstoff-zustandes der Böden. II. Chemische Extraktionsmethoden zur Phosphor- und Kaliumbestimmung. Kung. Lantbr. Hdgsk. Ann. 26: 199-215.

Hartikainen, H. 1982. Relationship between phosphorus intensity and capacity parameters in Finnish mineral soils II. Sorption-desorption isotherms and their relation to soil characteristics. J. Sci. Agric. Soc. Finland 54: 251-262.

Hofman, G., Van Meirvenne, M. and Demyttenaere, P. 1992. Stikstofbemesting. Toediening van N-meststoffen in de rij: potenti:ele voordelen. Landbouwtijdschrift, 45: 341-353.

Hofman, G. and Van Cleemput, O. 2004. Soil and plant nitrogen. International Fertilizer Industry Association, Paris, 48 pp.

Houba, V. J. G. , Novozamsky, I. , Lexmond, Th. M. and van der Lee, J. J. 1990. Applicability of 0.01 M CaCl2 as a single extraction solution for the assessment of the nutrient status of soils and other diagnostic purposes, Communications in Soil Science and Plant Analysis 21: 2281-2290.

Johnson, P.A., Shepherd, M.A., Hatley, D.J. and Smith, P.N. 2002. Nitrate leaching from a shallow limestone soil growing a five course combinable crop rotation: the effects of crop husbandry and nitrogen fertilizer rate on losses from the second complete rotation. Soil Use Management, 18: 68-76.

Joret, G. and Hebert, J. 1955. Contribution à la détermination du besoin des sols en acide phosphorique. Ann. Agron. 2,; 233-299.

Kirchmann, H., and Andersson, R. 2001. The Swedish system for quality assessment of agricultural soils. Environmental Monitoring and Assessment 72: 129–139.

Krogstad, T., Øgaard, A.F. and Kristoffersen, A. Ø. 2008. New P recommendations for grass and cereals in Norwegian agriculture. In: G. Rubaek. Phosphorous management in Nordic-Baltic agriculture - reconciling productivity and environmental protection. Nordiska jordbruksforskares förening (NJF). NJF report 4.

Lemunyon, J.L. and Gilbert, R.G. 1993. The concept and need for a phosphorus assessment tool. J. Prod. Agric. 6: 483-486.

Lopez-Ridaura, S., Van der Werf, H., Paillat, J.M., and Le Bris, B. 2009. Environmental evaluation of transfer and treatment of excess pig slurry by life cycle assessment. J. Environ.Manage. 90: 1296-1304.

Morgan, M.F. 1941. Chemical Soil Diagnosis by the Universal Soil Testing System. Connecticut Agricultural Experimental Station Bulletin 450, Connecticut.

Nair, V.D., and Graetz, D.A. 2002. Phosphorus saturation in Spodosols impacted by manure. J. Environ. Qual. 31:1279-1285.

Németh, T. 2003. Importance of agricultural chemistry in multifunctional crop production. Acta Agronomica Hungarica 51 (1): 101-107.

Neyroud, J.-A. and Lischer, P. 2003. Do different methods used to estimate soil phosphorus availability across Europe give comparable results? Journal of Plant Nutrition and Soil Science, 166: 422–431.

Nychas, A.E. 1989. Environmental policy with regard to fertilizers in arable farming/field production of vegetables in the E.C. In: Congres Meststoffen, Milieu en Akkerbouw, Misset BV, Doetichem, The Netherlands, pp. 7-20.

Olsen, S.R., Cole, C.V., Watanabe, F.S. and Dean, L.A. 1954. Estimation of Available Phosphorus in Soils by Extracting with Sodium Bicarbonate; USDA Circ. 939 U.S. Government Printing Office: Washington, DC, 1954.

Önorm, L. 2005. Chemische Bodenuntersuchungen: Bestimmung wasserlöslicher Stoffe. Reference L 1092.

Renou-Wilson, F. and Farrell, E.P. 2007. Phosphorus in surface runoff and soil water following fertilization of afforested cutaway peatlands. Boreal Environ. Res., 12, 693-709.

Reubens, B., D'Haene, K., D'Hose, T. and Ruysschaert, G. 2010. Bodemkwaliteit en landbouw: een literatuurstudie. Activiteit 1 van het Interregproject BodemBreed. Instituut voor Landbouw- en Visserijonderzoek (ILVO), Merelbeke-Lemberge, België. 203 p.

Schoumans, O.F. and Groenendijk, P. 2000. Modeling soil phosphorus levels and phosphorus leaching from agricultural land in the Netherlands. J. Environ. Qual. 29:111–116.

Schoumans, O.F., Groenendijk, P., van der Salm, C. and Pleijter, M. 2009. Methodiek voor het karakteriseren van fosfaatlekkende gronden: beschrijving van het instrumentarium PLEASE, Alterra, Wageningen, Alterrarapport 1724. pp. 76.

Schoumans, O.F., Rulkens, W., Ehlert, P.A.I. and Oenema, O. in prep. Phosphorus Recovery from Animal Manure. Technical opportunities and agro-economical perspectives, Alterra, Wageningen.

Schröder, J.J., Scholefield, D., Cabral, F. and Hofman, G. 2004. The effects of nutrient losses from agriculture on ground and surface water quality: the position of science in developing indicators for regulation. Environ. Sci. Policy, 7, 15-23.

Schröder, J., De Buisonjé, F., Kasper, G., Verdoes, N. and Verloop, K. 2009. Mestscheiding: relaties tussen techniek, kosten, milieu en landbouwkundige waarde. Rapport 287. Plant Research International, Wageningen.

Schüller, H. 1969. Die CAL-Methode, eine neue Methode zur Bestimmung des pflanzenverfügbaren Phosphates in Böden. With a summary: The CAL-Method, a new method to determine the availability phosphate in soils. Zeitschrift für Pflanzenernährung und Bodenkunde 123: 48-63.

Sharpley, A.N., Weld, J. and Kleinman, P.J.A. 2005. Soil testing, SERA-17, Description of BMPs. http://www.sera17.ext.vt.edu/Documents/BMP_soil_testing.pdf.

Sibbesen, E. and Sharpley, A.N. 1997. Setting and justifying Upper critical limits for phosphorus in soils. In: H. Tunney, O.T. Carton, P.C. Brookes and A.E. Johnston (ed). Phosphorus loss from soil to water. CAB International, Wallingford UK., p. 151-176.

Sissingh, H.A. 1971. Analytical technique of the method used for the assessment of the phosphate status of arable soils in the Netherlands. Plant Soil 34: 483-486.

Somasiri, L.L.W. and Edwards, A.C. 1992. An ion exchange resin method for nutrient extraction of agricultural advisory soil samples. Commun. Soil Sci. Plant Anal. 23: 645 — 657.

Sonneveld, C. 1990. Estimating quantities of water-soluble nutrients in soils using a specific 1:2 by volume extract. Commun. Soil Sci. Plant Anal. 21: 1257-1265.

Tunney, H., Breeuwsma, A., Withers, P.J.A. and Ehlert, P.A.I. 1997. Phosphorus fertilizer strategies: present and future. In: H. Tunney, O.T. Carton, P.C. Brookes and A.E. Johnston (ed). Phosphorus loss from soil to water. CAB International, Wallingford UK., p. 177-203.

Van Beek, C.L., Droogers, P. Van Hardeveld, H.A.,Van Der Eertwegh, G.A.P.H., Velthof, G.L. and Oenema, O. 2007. Leaching of solutes from an intensively managed peat soil to surface water. Water Air Soil Poll. 182: 291-301.

Van Dijk, W. 2003. Adviesbasis voor de bemesting voor akkerbouw en vollegrondsgroentengewassen. Publicatienummer 307. Praktijkonderzoek voor Plant en Omgeving. Lelystad. The Netherlands.

VLM 2010. Nitraatresidurapport 2010. Resultaten van de nitraatresidumetingen in Vlaanderen tot en met de staalnamecampagne van 2009. VLM, Brussels, 75 pp.

Vuorinen, J., and Mäkitie, O. 1955. The method of soil testing in use in Finland. Agrogeological Publications 63: 1-44.

6 Crop management for mitigation of nutrient losses to the aquatic environment

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6.1 Introduction

In this chapter we present the key successful crop management strategies for reducing N and P losses to the aquatic environment, with the main focus on the situation in North-Western Europe. We briefly describe (1) what crop management is and how it relates to other categories of mitigation options in this report, (2) the major traits of N and P cycling in soil and how they differ from each other, (3) interactions between crops and N and P cycling in soil and how to utilise this for retaining N and P and improving nutrient use efficiencies in the agricultural system, and (4) the interaction of crop management with nutrient mobilisation and transport pathways and how to utilise this for designing mitigation options.

Crop management encompasses the farmer's decisions on which crops and varieties to grow, when and how to sow or plant, how to handle crop residues, pests and weeds. Such decisions are often linked to decisions on manure and fertiliser handling and to soil management. For example the nutrient requirement of a certain crop on a given field, which is an essential part of a nutrient management plan (Chapter 5), is influenced by the expected yield and the crop rotation. Similarly, the chosen crop rotation governs the possibilities for soil tillage (Chapter 8). Additionally, crop rotation and crop management play an important role for maintaining an adequate soil organic matter content, good soil structure and aggregate stability facilitating infiltration and increasing soil strength (e.g. Watson et al., 2002), which in turn affect the nutrient dynamics and transport in the soil profile. Improved soil organic matter content can be achieved by straw incorporation and by introducing ley-arable crop rotations. By including deep rooting crops in the rotation macroporosity and infiltration in soil profiles can be enhanced.

Different crops and their different growth patterns also affect the evapo-transpiration from a field due to different rooting systems and differences in leaf area index development. This, in turn, may affect the whole field water balance and the active hydrological pathways through which nutrients may be lost from the field.

Photo 6.1
Different crop management strategies (Photo, Elly Møller Hansen, DJF)

Changing land use from arable or grassland to woody perennials will cause dramatic changes in the hydrology of the area (Bachmair et al., 2009). The infiltration capacity of former arable and pasture soil, which is planted with trees, is 20 and 60 times larger, respectively, than before (Carroll et al., 2004). Introduction of woody perennials will lead to the channelling of water into soil via the tree roots (Liang et al., 2009), to an increased organic matter content, and thus improved soil structure and porosity (Saha et al., 2007 and Olszewska and Smal, 2008). These changes may together lead to an improvement of the soil hydraulic conductivity in the top 25 cm by as much as a factor of 20 (Hofmann et al., 2005). Furthermore, perennial crops utilise water, whereby the soil water balance and water movement in and on the soil are directly affected. This can be exemplified by willow which has a very high water use, often using 100 mm more than grain crops per year (Persson, 1997; Jørgensen, 2000), and the low leaching level from willow is thus a combination of low nitrate concentrations and low percolation. Changes from arable land to systems with perennial/woody vegetation are mostly associated with a change in land use. Land use changes are described in detail in Chapter 10.

In this chapter we deal with the 'within-field' effect of crop management on nutrient losses on arable land (i.e. the effect of crop management on a given field under contrasting crop management strategies). Aspects dealing with how to reduce nutrient losses by optimizing the distribution of the mitigation options within the landscape, or targeted implementation of crop management options at certain risk areas in the catchment, are dealt with in Chapter 11. With reference to the components of the source - mobilization - transport - impact continuum defined by Withers and Haygarth (2007), crop management may affect the three first components (source, mobilization and transport). Introduction of catch crops is an example of altered nutrient mobilization. In this context we define *catch* crops as crops grown between two main crops with the purpose of immobilizing nutrients by crop uptake, and we define *cover* crops as crops introduced to protect the soil surface against losses through erosion and surface runoff.

Photo 6.2
Oil radish, a popular catch crop (Photo by Elly Møller Hansen DJF)

Crop management strategies for mitigating nutrient losses are typically introduced after nutrient and manure management have eliminated excessive nutrient supply to the fields and lowered the nutrient losses to a certain extent (see Chapter 5). If further reductions in nutrient losses are needed in addition to those obtained through nutrient management to achieve certain surface water quality standards in the receiving water body, changes in crop management can be the next group of mitigation options to be introduced. Such a situation is clearly illustrated by the source apportionment of nitrate leaching from Danish agriculture shown in Table 6.1. Between 1989 and 2003 several regulations related to nutrient management have been implemented in Denmark, e.g. restrictions on animal density, maximum rates for N fertilisation based on crop type and expected yield and demands relating to the utilisation on N in manure, requirements regarding minimum storage capacity for animal manure and bans on autumn and winter application of manure, requirements for fast incorporation of surface-applied manure (Kronvang et al., 2008). Due to these regulations in 2003 nitrate leaching had been reduced by almost 50% compared to the mid eighties and further reductions in N leaching in Danish agriculture have mainly to be achieved by improved crop management (Schou et al., 2007).

Table 6.1
Nitrate leaching from agricultural land in Denmark in 1989 and 2003. Between 1989 and 2003 several rules and regulations related to nutrient management has been implemented in Danish agriculture (see e.g. Kyllingsbæk and Hansen, 2007; Maguire et al., 2009). In 2003 the sources of N leaching have been approximated (Schou et al., 2007)

Year, and estimated source	**Nitrate leaching from agricultural land in Denmark (kg N ha^{-1})**
1989	109
2003	61
- Animal manure related	4
- Crop related*	45
- Natural/background	12

- *Nitrate leaching related to application of deep litter and manure deposited during grazing are included in crop related leaching losses, as these could not be separated during estimation*

Discharge and discharge patterns vary considerably throughout Europe, and it is inevitable that crop management as a strategy to mitigate nutrient losses has to be designed and adjusted according to the local soil, landscape and climatic conditions. In a northern coastal climate the period after harvest until the following spring is characterized by relatively high percolation rates and limited active crop growth due to low air temperatures. Since soil temperatures and soil water content often allow mineralization there is a risk of nitrate leaching during this period. In areas with a long growing season, catch crops can be sown after harvest of the main crop (i.e. Germany), but in areas where the post-harvest growing season is short (i.e. Sweden), under-sowing of a catch crop in cereals is preferable (Karlsson-Strese et al., 1998). In Denmark there is strong interest in a sort of compromise where seeds of crucifers are spread on the soil as catch crops two to three weeks before harvesting the main crop. In cold climates like in Norway with freezing/thawing periods there is a risk that nutrients immobilized in the catch or cover crop can be released after freezing (Sturite et al., 2007), which has to be taken into account when using catch and cover crops for mitigation of nutrient losses.

6.2 Catching N and mining P

Agricultural soils generally contain large pools of N, most of which is in organic form, but leaching losses of N are mainly associated with its inorganic form, nitrate. That is because nitrate is not retained on soil surfaces, leaving it highly mobile in soil and it may therefore easily be lost by leaching. Around 1-3% of N bound in organic forms may become mineralized and available to crop uptake or leaching within a growth period (Christensen, 2004). Transformation of N is associated with microbial-mediated turnover of organic matter during which N losses may occur. Gaseous N losses by N_2O may have an impact on global warming, while NH_3 volatilization may deteriorate nutrient status in restrained natural ecosystems and cause acidification by deposition.

Cycling of P in soil differs significantly from cycling of N. Inorganic P (in the sense of inorganic phosphate) exchange of gaseous phases with the atmosphere is non-existent; instead, phosphate binds rapidly and strongly to soil constituents. Long-term fertilisation with surplus P therefore leads to accumulation of P in the soil. However, the concentration of P in soil solution is modest because it is strongly buffered by the large pool of P retained on soil surfaces. The amount of P which can be lost by leaching of dissolved P is therefore typically very small compared to the amount of P which is retained in the soil. However, P may also be lost in particulate, colloidal and soluble forms, during rigorous run-off events e.g. pursuant to heavy rainfall (e.g. Gburek et al., 2005).

The pronounced differences in cycling of N and P in soil necessitate different management strategies for minimizing losses of the two nutrients. For instance, occasionally high concentrations of potentially leachable nitrate in soil can be reduced direct management strategies towards cropping systems with a high N uptake capacity (Figure 6.1). By such management strategy it may possible to reduce the leaching of nitrate considerably (Factsheet Rubæk and Jørgensen, 2010).

The situation is different for P. Even though the P uptake reduces the solution concentration of P in the vicinity of the root, this reduction is counteracted by desorption of a small fraction of the P retained on the soil particle surfaces. I.e. the soil solution P concentration is buffered by the large amount of P accumulated in the solid fraction. The concentration of dissolved P in soil solution is therefore less dynamic than for N, and even a well-established crop cannot lower and maintain a significantly lower P concentration in the soil solution. If leaching of dissolved P has to be lowered through crop management, it is only possible through P mining (Factsheet Chardon, 2010), this means by creating a situation where the P exported by the crops over several years is significantly larger than the P input with fertilizers and manures.

Certain crops (e.g. grass for silage) export large amounts of N and P from the field if they are offered optimum growing conditions and sufficient nutrient supply. Other crop types like woody energy crops (willow, poplar etc.) may result in low or modest nutrient export (Factsheet Jørgensen, 2010).

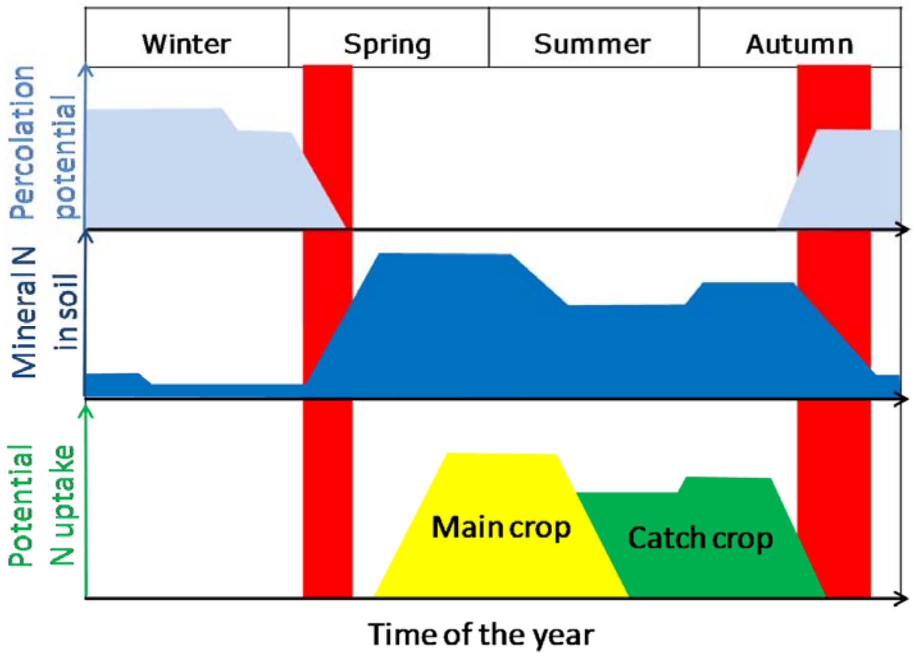

Figure 6.1

The seasonal dynamics of potentials for percolation, mineral N in soil (mineralization+ external inputs) and crop uptake in a "standard" year under northwest European conditions. The main crop (e.g. spring barley) is undersown with ryegrass acting as a nitrate catch crop. The vertical red zones indicate periods susceptible to elevated nitrate leaching losses. (After Christensen, 2004)

From an environmental point of view it is not only the export of nutrients by the crop which is important, the amount and form of the nutrients left in the soil after harvest is even more crucial for the potential losses of especially N. The amounts of potentially leachable N and P in the root zone are affected by the crop management strategy, fertilization practice, historical cropping intensity, soil type, and the climatic conditions. The concentrations furthermore depend on how well the crop utilises N and P in the various soil layers and on the timing of the crop growth and nutrient uptake (immobilization) in relation to mineralization processes (mobilization) and the discharge (transport) (Figure 6.1).

The length of the growth period where the crops take up nutrients varies considerably between crops. While some crops are harvested at maturity, others are harvested while they are in an active growth stage. This also has significant effects on how well the crop is able to utilise nutrients from added fertilisers and manures, and nutrients made available throughout the year by mineralisation of organic matter. For catch crops inserted after a main crop to catch and immobilise dissolved N in the soil, there will generally be a much better effect of introducing a catch crop after a crop that utilises nutrients poorly than after a crop with a high N utilisation. Table 6.2 gives an example of how much residual N is left in the soil after various crops. It thus gives information on the necessity of introducing catch crops after different crops, which are harvested at very different growth stages and have very different rooting systems and nitrogen use efficiencies. If climatic conditions allow, double cropping (two main crops grown within the same growing season) may also be introduced in order to increase productivity and optimize utilisation of N and P.

Table 6.2
An example of rooting depth and latent mineral N until rooting depth of several crops (Georges Hofman, pers. comm.)

Crop	**Rooting depth (cm)**	**Residual N_{min} at harvest (kg N/ha)**
Arable crops		
Grass	60	30
Wheat	90	30
Barley	90	30
Corn	90	<50
Sugar beets	90	30
Potatoes	60	50-75
Vegetables		
Spinach	30	50-75
Peas	30	<35
Beans	30	<35
Lettuce	45	50
Celery	60	50-75
Celeriac	60	40-60
Cauliflower	60	50-75
Carrots	60	<35
Cabbages	60	40
Brussels sprouts	90	<35

Grassland and other perennial crops cover the soil permanently throughout the year with well-established root systems, which are able to take up and immobilise nutrients when the climatic conditions are suitable for growth. However, when a perennial crop like grass is terminated, for example by ploughing, considerable nutrient quantities stored in the stubble and roots are incorporated into the soil and will become available due to mineralisation processes. Termination of a grassland crop therefore calls for careful planning of nutrient requirement and use of catch crops in the following years (Eriksen et al., 2004; Hansen et al., 2007).

The selection of variety and type of catch crop can be based on a number of criteria:
(1) Competition with the main crop. For undersown catch crops the competition with the main crop for nutrients and light can be an important issue.
(2) A large and fast nutrient uptake is important. It can be obtained by choosing crops with fast establishment after harvest of the main crop.
(3) Fast and deep root development enabling nutrient uptake from deeper soil layers (Thorup Kristensen, 2006). However in many soils, the properties of the deeper soil layers may limit rooting depth.
(4) old tolerance of the catch crop. The importance of frost and frost tolerance in cold climates has been demonstrated in South-Eastern Norway where the nitrogen uptake in a Ryegrass catch crop was 25-35 kg N/ha during autumn. During winter some of this immobilised nitrogen leached from the plants leaving 20-25 kg N/ha in the grass in spring. (Molteberg et al., 2004). Catch crops that were less cold tolerant may have caused even larger release of nitrogen from the catch crop.

6.3 Reducing nutrient mobility and modifying transport pathways

Soil erosion and surface runoff are considered important transport pathways, especially for dissolved and particulate P. A number of crop management options reduce the risk of nutrient mobilization by surface runoff and soil erosion (e.g. Morgan, 1995).

Principally, crop management options for reducing the erosion risk aim at:
(1) increasing water infiltration to reduce runoff volumes and erosivity,
(2) trengthening topsoil resistance to detachment of soil particles, and
(3) protecting the soil surface against erosive forces with plant or residue cover (e.g. Govers et al., 2004).

Most of these effects depend crucially on the crop type and growth stage.

A soil cover of vegetation or plant residues acts as a protective layer for the soil surface absorbing some of the energy of rainfall or overland flow. Soil cover thus reduces the physical degradation of the surface soil and hence the rate of surface sealing and crusting during rainfall, which in turn supports higher infiltration rates. By dissipating some of the runoff energy also soil detachment is reduced. Empirical evidence suggests that water erosion may decrease exponentially with increasing soil cover (Govers et al., 2004). Crops may also reduce erosion through a well-developed root system reinforcing the topsoil against detachment. As winter-sown cereals combine sparse soil cover and a weak root system they are often associated with erosion problems in temperate, humid climates where most runoff occurs during winter (e.g. Chambers et al., 2000). In contrast, catch crops and stubble fields have been shown to be effective in controlling erosion (Morgan, 1995).

In Norway, erosion could be reduced by 5% by growing cover crops (Ital. Ryegrass) compared to a situation with no cover crop (Vandsemb and Bechmann, 2004). The growing of a cover crop necessitates spring ploughing (no-till in autumn). Model prediction carried out by the ERONOR-model suggested a reduction in erosion of approx. 20% by growing cover crops (Lundekvam, 2002). The effect of cover crops depends on the development of the crop during autumn. A well-developed crop gives a higher reduction in erosion compared to a less developed crop.

The effect of cover crops on P losses is, however, less significant than the effect on soil erosion. This may partly be explained by increased losses of dissolved P released from the cover crop due to freezing during winter (Bechmann et al., 2005). Repeated freeze-thaw cycles increase the release of P from plant material. Cover crops may in this way redistribute nutrients from within the soil to the soil surface with an increased risk of transport to the stream via surface runoff. Losses of N and P via surface run-off after freezing of a Ryegrass catch crop amounts to 55% (±26) of the N and 54%(±23) of the P under Norwegian conditions (Sturite et al., 2007).

In conclusion

1. Catch crops can immobilise nitrogen during the winter runoff season in humid climates, and cover crops can protect the soil surface against nutrient losses via surface runoff and erosion.
2. P losses related to the high P status of a soil can be reduced by using crops to mine the soil of P over several years.
3. Crop management and crop rotation can actively improve soil properties in ways that may reduce nutrient losses.
4. Crop management strategies have to be designed and adapted to local farming conditions, soils and climate.
5. There are numerous possibilities to design and develop new crop management strategies, with improved nutrient utilisation and reduced losses.

List of factsheets
Chardon, W.J., 2010. Vegetative mining. [FS]
Rubæk, G.H. and U. Jørgensen, 2010. Catch crops and cover crops. [FS]
Jørgensen, U. 2010. Perennial energy crops substituting crop rotation.

References

Bachmair, S., Weiler, M. and Nützmann, G. 2009. Controls of land use and soil structure on water movement: lessons for pollutant transfer through the unsaturated zone. J. Hydrol. 369, 241-252.

Bechmann, M., Kleinman, P.J.A., Sharpley, A.N. and Saporito, L. 2005. Effect of freezing and thawing on fate of phosphorus in bare, manured and catch cropped soils. J. Environ. Qual. 34, 2301-2309.

Carroll, Z.L., Bird, S.B., Emmett, B.A., Reynolds, B. and Sinclair, F.L. 2004. Can tree shelterbelts on agricultural land reduce flood risk? Soil Use Manage. 20, 357-359.

Chambers, B.J, Garwood, T.W.D. and Unwin, R.J. 2000. Controlling soil water erosion and phosphorus losses from arable land in England and Wales. J. Environ. Qual. 29, 145-150.

Christensen, BT. 2004. Tightening the Nitrogen Cycle. In: P. Schjønning, S. Elmholt and B.T. Christensen (eds.) Managing Soil Quality: Challenges in Modern Agriculture. CAB International.

Eriksen, J., Askegaard, M. & Kristensen K. 2004. Nitrate leaching from an organic dairy crop rotation: the effects of manure type, nitrogen input and improved crop rotation. Soil Use Manage. 20, 48-54.

Gburek, WJ., Barberis, E., Haygarth, PM., Kronvang, B. and Stamm, C. 2005. Phosphorus mobility in the landscape. In: Sims, J.T. and Sharpley, A.N. (eds.) Phosphorus: Agriculture and the Environment. Agronomy Monographs nr 46. American Society of Agronomy Madison, USA. p. 941-979.

Govers, G., Poesen, J. and Goosens, D. 2004. Soil Erosion - processes, damages and countermeasures. In: P. Schjønning, S. Elmholt, B.T. Christensen. Managing Soil Quality. Challenges in Modern Agriculture. CABI Publishing, Wallingford, UK, p. 199-217.

Hansen, E.M., Eriksen, J. and Vinther, F.P. 2007. Catch crop strategy and nitrate leaching following grazed grass-clover. Soil Use Manage.23, 348-358.

Hofmann, B., Both, S., Tischer, S. and Christen, O. 2005. Beeinflussung physikalischer Bodeneigenschafen und des Kohlenstoffgehaltes im Boden durch Aufforstung von ehemals landwirtschaftlich genutzten Flaechen im Gebiet des 'Suessen Sees' Seeburg. Mitteilungen der Deutschen Bodenkundlichen Gesellschaft 107, 53-54..

Jørgensen, U. 2000. Combined energy crop production and ground water protection. DJF rapport Markbrug 29, 97-104.

Karlsson-Strese, E.-M., Rydberg, I., Becker, H.C. & Umaerus, M. 1998. Strategy for catch crop development II. Screening of species undersown in spring barley (Hordeum vulgare L.) with respect to catch crop growth and grain yield. Acta Agric. Scand., Sect. B. Soil and Plant Sci. 48, 26-33.

Kronvang, B., Andersen, H.E., Børgesen, C.D., Dalgaard, T., Larsen, S.E., Bøgestrand, J. and Blicher-Mathiesen, G. 2008. Effects of policy measures implemented in Denmark on nitrogen pollution of the aquatic environment. Environ. Sci. Policy 11, 144-152.

Kyllingsbæk, A., and Hansen, J.F. 2007. Development in nutrient balances in Danish agriculture 1980-2004. Nutr. Cycl. Agroecosyst. 79, 267-280.

Liang, W-L., Kosugi, K. and Mizuyama, T. 2009. A three-dimensional model of the effect of stemflow on soil water dynamics around a tree on a hillslope. J. Hydrol. 366, 62-75.

Lundekvam, H. 2002. Eronor/Usleno - Empirical erosion models for Norwegian conditions. Report no 6/2002. University of Life Sciences. ISBN: 82-483-0022-6. pp 40.

Maguire, R.O., Rubæk, G.H., Haggard, B.E. and Foy, B.H. 2009. Critical evaluation of the implementation of mitigation options for phosphorus from field to catchment scales. J. Environ. Qual. 38, 1989-1997.

Molteberg, B., Henriksen, T. and Tangsveen, J. 2004. Bruk av gras som fangvekster i korn, Grønn kunnskap 8/12, p 57. www.bioforsk.no.

Morgan, R.P.C. 2005. Soil Erosion and Conservation. 3rd ed.. Blackwell Publishing, Oxford, UK, p. 304.

Olszewska, M. and Smal, H. 2008. The effect of afforestation with scots pine (Pinus sylvestris L.) of sandy post-arable soils on their selected properties. I. Physical and sorptive properties. Plant Soil 305, 157-169.

Persson, G. 1997. Comparison of simulated water balance for willow, spruce, grass ley and barley. Nordic Hydrology 28, 85-98.

Saha, R., Tomar, J.M.S. and Ghosh, P.K. 2007. Evaluation and selection of multipurpose tree for improving soil-hydro-physical behaviour under hilly eco-system of north east India. Agroforestry Systems, 69, 239-247.

Schou, J.S., Kronvang, B., Birr-Pedersen, K., Jensen, P.L., Rubæk, G.H., Jørgensen, U. and Jacobsen, B. 2007. Virkemidler til realisering af målene i EUs Vandrammedirektiv. Udredning for udvalg nedsat af Finansministeriet og Miljøministeriet: Langsigtet indsats for bedre vandmiljø. Danmarks Miljøundersøgelser, Aarhus Universitet, 132 s. (NERI report no. 625).

Sturite, I., Henriksen, T.M. and Breland, T.A. 2007. Winter losses of nitrogen and phosphorus from Italian ryegrass, meadow fescue and white clover in a northern temperate climate. Agriculture, Ecosystems and Environment 120, 280-290.

Thorup-Kristensen, K. 2006. Effect of deep and shallow root systems on the dynamics of soil inorganic N during 3-year crop rotations. Plant Soil 288, 233–248.

Vandsemb, S. and Bechmann, M. 2004. Miljøeffekter av fangvekst i Hedmark. Jordforsk rapport18/2004. ISBN 82-7467-502-9. pp 21.

Watson, C.A., Atkinson, D., Gosling, P., Jackson, L.R. and Rayns, F.W. 2002. Managing soil fertility in organic farming systems. Soil Use Manage. 18, 239-247.

Withers, P.J.A. and Haygarth, P.M. 2007. Agriculture, phosphorus and eutrophication: A European perspective. Soil Use Manage. 23, 1-4.

7 Livestock management

W.J. Chardon, M. van Krimpen and O.F. Schoumans

7.1 Introduction

Problems with losses of N and P to surface waters are often found in regions with intensive agricultural and livestock production (Sharpley et al., 1994). Figure 7.1 shows the N and P input via manure within the EU-15 countries per total land surface. The figure shows a number of regions that had a much higher input than the average for EU-15, e.g. The Netherlands, Belgium, Brittany, Po valley Southwest England, Northern Ireland, Northeast Germany and Denmark.

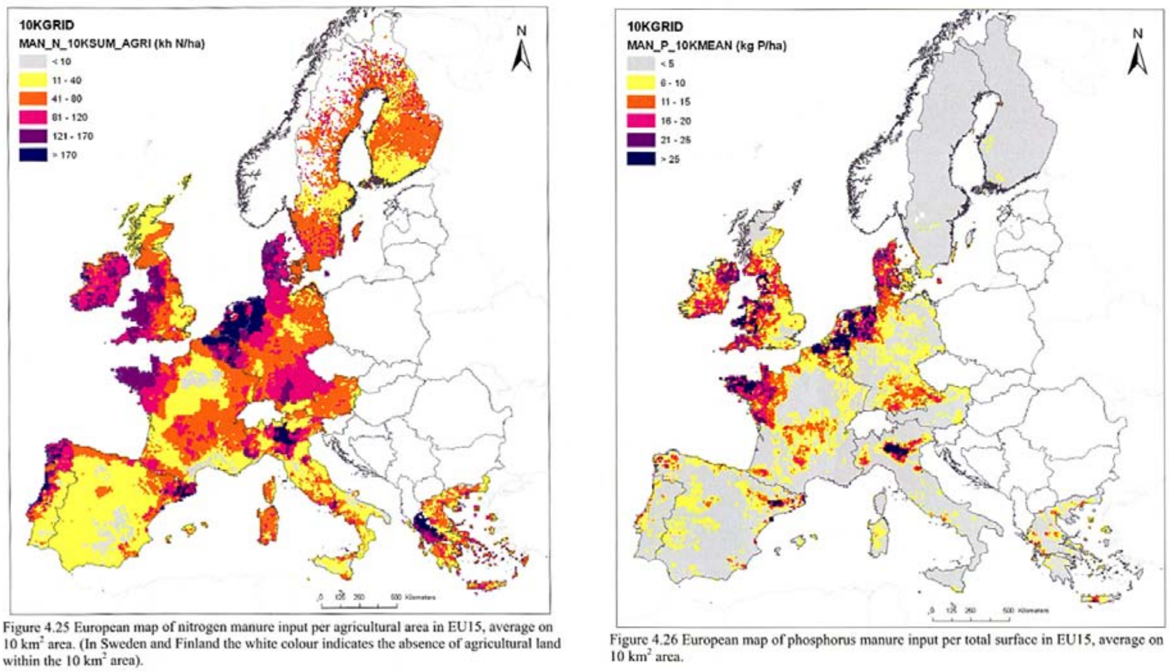

Figure 7.1
European map of input via manure of N (left) and P (right) per total surface in EU 15, average on 10 km^2 area (Grizzetti et al., 2007)

During the first decades after World War II, both in north-western Europe and in the USA animal agriculture became concentrated, with farms importing concentrated feeds and fertilizers, resulting in nutrient accumulation on the farms. This was often on poorer sandy soils and hilly areas that are less suitable for e.g. growth of crops (Rotz et al., 2005). Figure 7.2 shows, as an example, the inputs and outputs of the intensive Dutch agricultural system in the period 1970 - 2008 for phosphorus (Schoumans et al., 2010). The main inputs are imported feeds (mainly concentrated feeds) and fertilizers. The outputs in terms of agricultural products were in the past much lower than nowadays. The P surplus in 2008 was 60 million kg P_2O_5 which equals to about 30 kg P_2O_5 per ha. However, a huge part of the manure surplus is transported to Northern France and Eastern Germany.

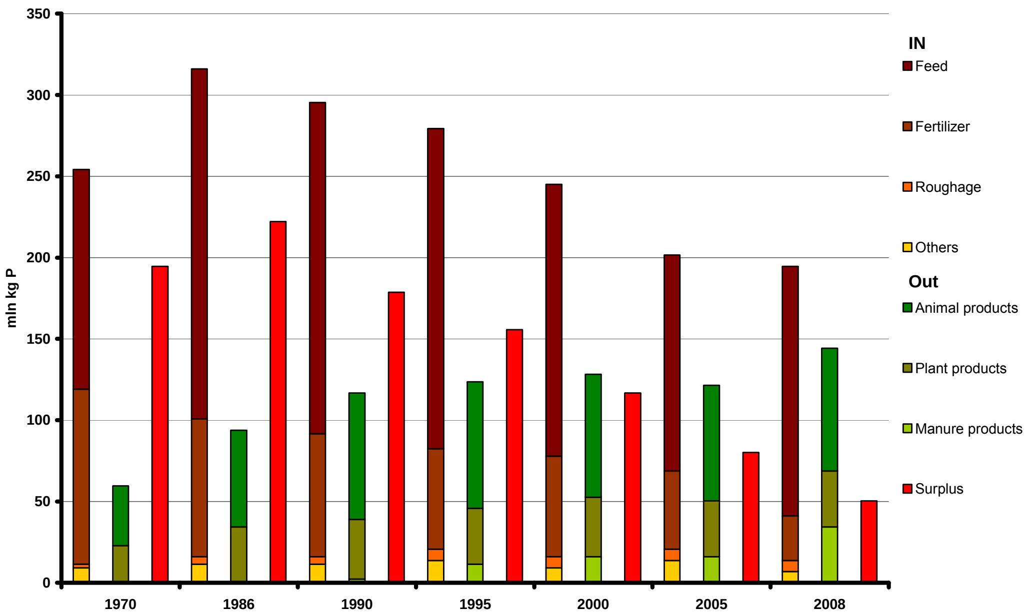

Figure 7.2
Phosphorus balance of the Dutch agriculture in the period 1970 – 2008 (Schoumans et al., 2010; source data CBS statistics The Netherlands)

It is common practice to apply manure on the basis of N requirement of crops. However, the N:P ratio (w:w) of manures and composts is, on average, ca. 2-4, (Eghball, 1998; Maguire et al., 2006) while the N:P uptake by major grain and hay crops is 4.5-9 (Eghball, 1998). Table 7.1 shows the range in values found for the N:P ratios of different types of manure. Due to the high amounts of N required by the crops, compared to P, too much manure is applied on agricultural land because the N:P ratio in manure is much lower. On the long-term this will lead to accumulation of P in soil. To date, in many regions with a high animal density the majority of soils sampled do not require further addition of P to support plant growth (Sharpley et al., 1994; Chardon and Schoumans, 2007).

Table 7.1
Range in N:P ratios found for different types of manure (Eghball, 1998)

Manure type	Minimum	Maximum
Cattle feedlot	2:1	3:1
Dairy	5:1	7:1
Swine	1.5:1	4:1
Chicken	2:1	4:1
Broiler	2:1	6:1

7.2 Overall production

The build-up of N and P in agricultural soils is strongly influenced by the balance of N and P on farm level. A build-up in soil increases the risk of substantial surface runoff and leaching to groundwater and / or surface water, when the soils are connected to the waters. In the past it was common practice to apply all manure that was produced on a farm onto the soils of the same farm. This has led to high levels of P in soils especially in the countries / regions with a high production of manure (Figure 7.1). Decreasing livestock density, or the overall (animal) production on a farm, decreases the amount of manure produced, and the pressure of application onto the soils. However, if it is possible and economically feasible to transport the manure to neighbouring farms, to other regions or to a manure processing industry, the pressure decreases also and the manure market will become in balance. Nevertheless, the higher the livestock density within a larger region, the more local economy will depend on the agricultural system. History shows that in several countries the dependency has led to regulations on manure applications that were less strict than they should have been from an environmental point of view. The economical and social impacts make it more difficult to restructure agriculture in the region in order to create a regional / national balance of N and P that creates less risk for pollution of surface water. Reducing meat consumption per capita could also lead to a lower overall production of manure (and greenhouse gasses), and reducing the impact of animal production on surface water quality (Factsheet: Isermann, 2010). However, worldwide an increase of total food consumption (and meat consumption) is expected, because the population will grow from 6 billion nowadays to 9 billion people in 2050.

7.3 Feeding

As mentioned above, the lower N:P ratio in manure than in crop uptake can lead to accumulation of P in soils when manure is applied based on N requirements. Lowering the P content in animal diets, or improving dietary P efficiency, will increase the N:P ratio of manure, reducing the risk of accumulation in soils (Maguire et al., 2005). It will help to reduce a surplus on the balance of P of a farm or region, and decrease the necessity of transport or processing of manure. The main methods that were investigated for reducing dietary P content (Maguire et al., 2005; Factsheets: Bannink, 2010; Van Krimpen and Jongbloed, 2010) include:

i. feeding according to requirement in the growth phase of the animal;
ii. adding phytase to feed for swine, poultry and fish, to increase the ability to digest phytate-P;
iii. feeding corn with a lower content of phytate-P;
iv. use feed that is better digestible.

In the Netherlands the total P content in feeds for growing-finishing pigs has decreased by more than 2.5 g/kg (33%) in the period 1973 to 1996. Also, during the same period the feed conversion ratio has improved substantially. This has led to a reduction in P excretion from 1.62 to 0.67 kg P/pig (Jongbloed and Lenis, 1998). Besides feeding, optimal management with regard to housing and health status of the pigs and feeding strategy, which may improve feed conversion ratio, will be beneficial for the environment (Van Krimpen and Jongbloed, 2010).

7.4 Grazing management

During grazing of e.g. cows, sheep or deer both N and P are deposited on the soil on concentrated spots via urine and dung pats. Locally, grazed pasture can be the major source of P entering surface waters in lowland catchments (McDowell et al., 2007; Bourke et al., 2008). Grazing usually takes place on grassland only, but in cooler areas like New Zealand with restricted growth of grass in the winter, animals are put in fields where forage crops have been grown (McDowell et al., 2005). Dung pats can cause high concentrations of P in runoff water (Chardon et al., 2007), and urine patches are local sources of N (Haynes and Williams, 1993). The potential for P loss via runoff from grazed pastures is greatest during, or soon after grazing and declines exponentially with time thereafter (McDowell et al., 2007). Also, physical damage to the soil due to treading can reduce the water infiltration rate and pasture growth, which may increase the risk of overland flow and sediment transfer (McDowell et al., 2005). Maintaining a good vegetative cover will promote the entrapment of particles that are otherwise lost with runoff (Toor et al., 2005).

Restriction of grazing, e.g. to a few hours a day instead of day-round, was found to strongly decrease P loss via runoff (McDowell et al., 2005). Limiting the grazing season, especially in autumn and winter when N uptake by the sward is limited, can reduce N leaching significantly (Cuttle et al., 2007, p. 31-32). Since also the risk of physical damage to the soil is higher when the soil is wet, the grazing intensity should preferably be reduced during such periods. In Denmark, production of hay and silage on plots instead of grazing, combined with low-emission application of manure, strongly decreased N loss (Factsheet: Jorgensen et al., 2010).

Direct access of animals to surface water and destruction of riparian vegetation in buffer zones should be avoided if possible. Especially in arid areas, or during periods of warm weather, water exerts a 'magnetic force' over livestock, strongly influencing grazing patterns (Agouridis et al., 2005; Factsheet Dorioz and Gascuel-Odoux and Chapter 11). Offering shade outside riparian areas and water sources other than surface water can significantly reduce loss of soil particles, N, P and *E. coli* (Byers et al., 2005).

7.5 Point sources at farm scale

Drinking troughs, feeding places and shelter areas will attract cattle during grazing, which will lead to an uneven redistribution of nutrients via deposition on grazed grassland. Such places can be considered as 'hot spots' at the farm scale. Livestock standing will also lead to an increased soil compaction and run-off potential, creating areas of greater risk of P loss to water (Tunney et al., 2007). Therefore, drinking and feeding places could be moved at regular intervals so both the uneven loading and physical damage can be reduced (Cuttle et al., 2007, p. 35).

Where solid manure is stacked in the field or outside of buildings, the heap should not be sited over field drains or close to a watercourse. Water leaking from manure can contain very high concentrations of N and P (McDowell et al., 2005; Chardon et al., 2007), this can reach surface water either directly or via artificial drains. Ideally, the heaps should be placed on concrete so the effluent can be collected; this is already prescribed by regulations in e.g. Switzerland and The Netherlands (Factsheets Haygarth, 2010 and Newell-Price, 2010).

In conclusion

1. Reducing the N and/or P content of animal feed, or increasing the uptake availability, can greatly reduce a nutrient surplus of a farm and the risk of losses due to over-application.
2. Transport of manure from a farm with a nutrient surplus is costly; manure separation can reduce these costs if the liquid fraction with a relative low P content can be applied on agricultural land.
3. Streams should be protected against direct access of grazing animals in order to prevent pollution of the water with nutrients and bacteria.
4. Grazing during autumn and winter, when nutrient uptake by the vegetation is small, should be limited
5. Point sources on farm scale can be an important for nutrients in surface water. Although data on these losses are limited they should receive more attention.

List of factsheets (FS)

Bannink, A. 2010.. Reducing content of N and P in dairy nutrition. [FS]
Dorioz, J.M. and Gascuel-Odoux, C., 2010. Manage interaction between livestock and rivers or streams. [FS]
Haygarth, P.M. 2010. Site solid manure heaps away from watercourses and field drains. [FS]

Isermann, K. 2010. Optimize / reduce overall stocking rates on livestock farms. [FS]
Jørgensen, U., Rubæk, G.H. and Vertès, F. 2009. Harvest of grasslands for silage or hay instead of cattle grazing. [FS]
Newell Price, J.P. 2010. Site solid manure heaps on concrete and collect the effluent. [FS]
Van Krimpen, M. and Jongbloed, A. 2010. The impact of nutrition on reduction of phosphate excretion in pigs. [FS]

References

Agouridis, C.T., Workman, S.R., Warner, R.C. and Jennings, G.D. 2005. Livestock grazing management impacts on stream water quality: A review. J. Am. Water Resour. Assoc. 41, 591-606.

Bourke, D., Jeffrey, D., Dowding, P., Kurz, I. and Tunney, H. 2008. Eutrophication from agricultural sources - the impact of the grazing animal on phosphorus, nitrogen, potassium and suspended solids loss from grazed pastures - phosphorus dynamics in grazed grassland. ERTDI Report Series No. 77, EPA and TEAGASC.

Byers, H.L., Cabrera, M.L., Matthews, M.K., Franklin, D.H., Andrae, J.G., Radcliffe, D.E., McCann, M.A., Kuykendall, H.A., Hoveland, C.S. and Calvert, V.H. 2005. Phosphorus, sediment, and Escherichia coli loads in unfenced streams of the Georgia Piedmont, USA. J. Environ. Qual. 34, 2293-2300.

Chardon, W.J., Aalderink, G.H. and Van der Salm, C. 2007. Phosphorus leaching from cow manure patches on soil columns. J. Environ. Qual. 36, 17-22.

Chardon, W.J. and Schoumans, O.F. 2007. Soil texture effects on the transport of phosphorus from agricultural land in river deltas of Northern Belgium, The Netherlands and North-West Germany. Soil Use Manage. 23 (suppl. 1), 16-24.

Cuttle, S.P., Macleod, C.J.A., Chadwick, D.R., Scholefield, D., Haygarth, P.M., Newell-Price, P., Harris, D., Shepherd, M.A., Chambers, B.J. and Humphrey, R. 2007. An inventory of methods to control diffuse water pollution from agriculture (DWPA). User Manual. Prepared as part of Defra Project ES0203, IGER and ADAS, UK.

Eghball, B. 1998. Phosphorus and nitrogen based manure and compost application. Manure Matters 9(2). http://www.p2pays.org/ref/16/15497.htm.

Grizzetti, B., Bouraoui, F. and Aloe, A. 2007. Spatialised European nutrient balance. JRC36653 report, 98 pp.

Haynes, R.J. and Williams, P.H. 1993. Nutrient cycling and soil fertility in the grazed pasture ecosystem. Adv. Agron. 49,119–199.

Jongbloed, A.W., and Lenis, N.P. 1998. Environmental concerns about animal manure. J. Anim. Sci. 76, 2641-2648.

Maguire, R.O., Brake, J.T. and Plumstead, P.W. 2006. Phosphorus in manure: Effect of diet modification. Encyclopedia of Soil Science, 1: 1, p. 1285-1287.

Maguire, R.O., Dou, Z., Sims, J.T., Brake, J. and Joern, B.C. 2005. Dietary strategies for reduced phosphorus excretion and improved water quality. J. Environ. Qual. 34, 2093-2103.

McDowell, R.W., Drewry, J.J., Muirhead, R.W. and Paton, R.J. 2005. Restricting the grazing time of cattle to decrease phosphorus, sediment and *E-coli* losses in overland flow from cropland. Aust. J. Soil Res. 43, 61-66.

McDowell, R.W., Nash, D.M. and Robertson, F. 2007. Sources of phosphorus lost from a grazed pasture receiving simulated rainfall. J. Environ. Qual. 36,1281-1288.

Rotz, C.A., Taube, F., Russelle, M.P., Oenema, J., Sanderson, M.A. and Wachendorf, M. 2005. Whole-farm perspectives of nutrient flows in grassland agriculture. Crop Science 45, 2139-2159.

Schoumans, O.F., Rulkens, W., Ehlert, P.A.I., Oenema, O. 2010. Phosphorus recovery from animal manure; Technical opportunities and agro-economical perspectives. Wageningen, Alterra-report, Alterra. The Netherlands.

Sharpley, A.N., Chapra, S.C., Wedepohl, R., Sims, J.T., Daniel, T.C. and Reddy, K.R. 1994. Managing agricultural phosphorus for protection of surface waters: Issues and options. J. Environ. Qual. 23, 437-451.

Toor, G.S., Sims, J.T. and Maguire, R.O. 2005. Grazing management. [BMP SERA-17].

Tunney, H., Kurz, I, Bourke, D. and O'Riley, C. 2007. Eutrophication from agricultural sources: The impact of the grazing animal on phosphorous loss from grazed pasture. Synthesis report. ERTDI Report 66, TEAGASC, ISBN: 1-84095-236-9.

8 Soil management

M. Bechmann[1] and B. Ulén

8.1 Introduction

A number of soil management methods are evaluated for their effect on soil and nutrient losses, including different soil tillage methods and soil amendments. Soil amendments include organic matter and chemical additives. Soil tillage methods include direct drilling (no-till), shallow cultivation and ploughing.
Cropping systems without ploughing are receiving great attention in Europe both for economic reasons, including possible reduction in labour and energy consumption, and for soil improvements (Holland, 2004). In cereal cropping systems, soil tillage is a major factor contributing to increased risk of soil erosion, transport of soil particles and losses of particulate P by water (Lundekvam and Skøien, 1998). Furthermore, the risk of transport of soil particles and PP loss depends on the slope and the soil texture, silty soils being more vulnerable to soil erosion than clay and sandy soil types. Changes in soil tillage will have highest effect on soil erosion on high risk areas.

The purpose of soil management is to improve the production potential for a certain crop. Soil management systems to reduce nutrient losses have to be adapted to the actual production system with a minimum of reduction in yields. Minimum tillage systems relevant for cereal production systems may include different depth and intensity of soil management and as well different timing of the soil management. Some definitions are given in the textbox below.

Soil tillage methods:
1. No tillage/direct drilling: Tillage leaving more than 30% of the soil covered with plant residues.
2. Shallow cultivation: Soil tillage to <10 cm depth. No inversion
3. Ploughing: Soil inversion at 20-25 cm dept
   a. Contour ploughing: Ploughing along the contours

In general, deeper and more intense soil management results in higher erosion risk. The risk of soil erosion is highest during autumn and winter and hence direct drilling in autumn or undisturbed stubble, compared to mouldboard ploughing in autumn, is an important option to mitigate soil losses in arable cropping. The presence of crop residues and an intact root system may act as a filter for soil particles in these systems. The effect of soil tillage on losses of particulate P (PP) follow the lines of soil erosion, but the relationship between soil tillage and losses of nutrients are much more complex.

8.2 Soil tillage systems

Direct drilling
A soil tillage system with direct drilling or shallow cultivation excludes inversion of the whole topsoil. Such methods have a mulching effect, since crop residues are left on the soil surface or in the uppermost soil layer. The term mulching also refers to covering the soil surface with material imported from beyond the field, e.g. straw and leaves, as commonly used in horticulture. In this context, direct drilling means that the crop is sown

in a single operation with or without shallow cultivation by separate tines or discs operating in front of the drill tines. By definition, direct drilling requires that more than 30% of the soil is covered with plant residues. Plough-less tillage is practised to decrease erosion and surface runoff losses of PP in many European countries (e.g. Baker and Richards, 2002). Direct drilling has great potential for reducing soil erosion (Figure 8.1) and losses of particulate bound phosphorus (PP) from unstable, erodible clay loams, silty and clay soils under cereal cropping in Scandinavia. Similar results are found for direct drilling of other crops, e.g. sugarbeet crops in Austria (Strauss et al., 2004). Direct drilling also has a potential to reduce concentrations of DRP compared to autumn ploughing. This is shown in Table 8.1 for establishment of winter wheat in Norway. However, as discussed below, the effect of direct drilling on DRP losses depends on several factors.

Figure 8.1
Water quality in surface runoff from field lysimeters with winter wheat sown with shallow cultivation (1), direct drilling (2) and ploughing (3) (Grønsten et al., 2007)

Direct drilling poses an increased risk of dissolved reactive phosphorus (DRP) losses since more plant material is left on the soil surface which may release P (Figure 8.2). Additionally, the amount of surface runoff may increase with direct drilling compared to tillage and hence, at the same concentrations, losses of phosphorus will be higher. Furthermore, surface application of P fertilizer combined with no-till, carried out during consecutive years, leads to P being accumulated in the top layer of the soil, thereby increasing the risk of DRP losses in surface runoff (Table 8.2) (Factsheets: Bechmann et al., 2010; Ulén et al., 2010e). Without total topsoil inversion, fertiliser and manure incorporation into the soil is limited. A stratified layer of P builds up on or near the soil surface (Logan et al., 1991), increasing the risk of P release and subsequently, transport of DRP via surface runoff (Sharpley and Smith, 1994) and tile drain water (Gaynor and Findlay, 1995). Studies from the northern Mississippi area of the US concluded that even though losses of total P were considerably reduced when soil tillage was omitted, losses of DRP were eight-fold higher with no-till compared to conventional ploughing (McDowell and McGregor, 1984). Addiscott and Thomas (2000) strongly recommend interrupting periods of plough-less tillage with conventional ploughing in order to dilute P concentrations in the uppermost part of the soil. They also suggest that soil inversion and mixing of the soil increase adsorption of DRP to soil particles and reduce DRP in horizontal and lateral water movements. DRP has a higher ecological impact than PP due to its higher bioavailability and therefore direct drilling should preferable be allocated to erosion-prone sites identified using the critical source areas (CSA) concept.

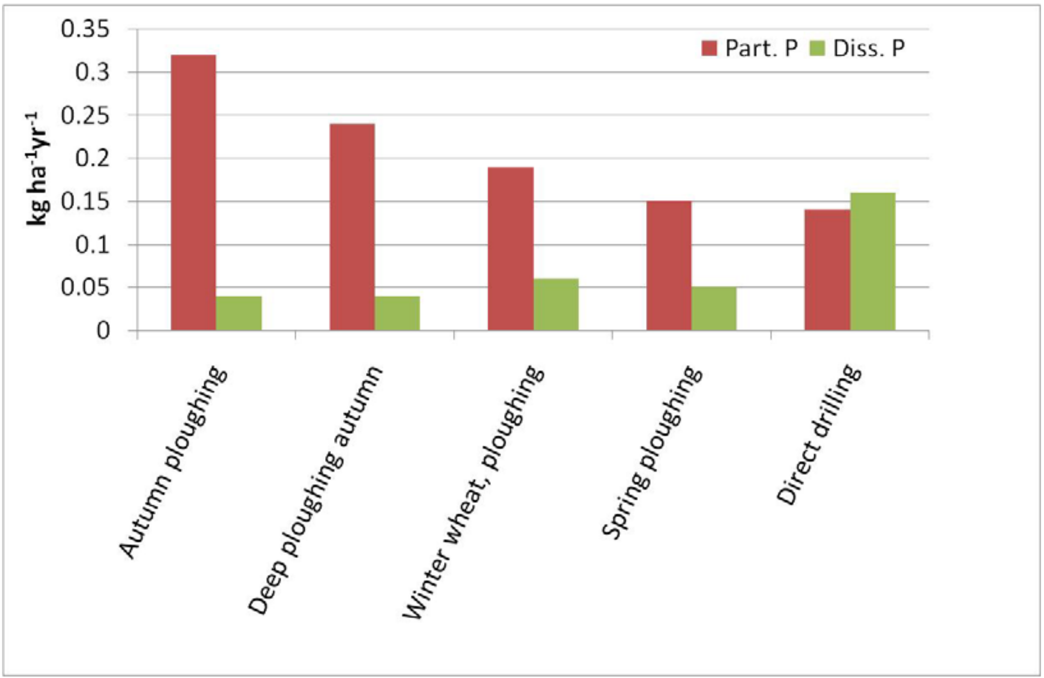

Figure 8.2
Soil management effects on losses of particulate P and dissolved reactive P (Ulen and Kalisky, 2005)

In northern France, tillage compared to reduced tillage or no tillage in a long term field trial showed similar N leaching (Oorts et al., 2007). Nevertheless, no tillage or shallow tillage tends to concentrate organic matter in the surface layer. After a long period of reduced or no-tillage, ploughing is expected to activate the N mineralization in the soil surface layer (Silgram and Shephard, 1999) and to induce higher risk of N leaching. This process is similar to ploughing of grassland.

With direct drilling, the risk of infestation by pests and weeds and the need for chemical control must be considered. Scientific evidence about the possible effects of increased use and leaching of herbicides on aquatic wildlife remains inconclusive. Furthermore, on heavy clays in Scandinavia, crop establishment in spring may suffer because of soil structural problems when these soils are not ploughed in autumn.

Shallow cultivation
Shallow cultivation involves no inversion of the total topsoil soil layer. Tillage usually occurs to a depth of 5-10 cm and leaves the soil covered with some crop residues. Based on the field experiments reviewed in Scandinavia (Table 8.2; Ulén et al., 2010), it can be concluded that shallow cultivation compared to ploughing reduces erosion and PP losses, though with less effect than direct drilling (Table 8.1). However, some studies show that losses of DRP may increase in shallow tillage systems compared to autumn ploughing, especially via surface runoff (Table 8.2). Therefore, shallow tillage should be targeted to CSA, where losses of PP dominate. Furthermore, shallow cultivation is appropriate on soils where structural problems have already been alleviated and where the erosion risk is high (Factsheets: Ulén et al., 2010i, l).

Table 8.1
Mean concentrations (2002-2007) of suspended solids (SS), total phosphorus (TOTP) dissolved reactive phosphorus (DRP) and the differences between TOTP and DRP for three soil management strategies before sowing of winter wheat at a site with moderate and a site with high erosion risk in Norway (Grønsten et al., 2007)

	SS	**TOTP**	**DRP**	**PP = TOTP-DRP**
		(mg L^{-1})		
Moderate erosion risk				
1 Direct drilling	250	0.55	0.38	0.17
2 Shallow cultivation	420	0.80	0.50	0.30
3 Ploughing	630	1.31	0.85	0.46
High erosion risk				
1 Direct drilling	330	1.05	0.64	0.41
2 Shallow cultivation	1800	1.39	0.82	0.57
3 Ploughing	2000	2.37	0.74	1.63

Spring ploughing
Ploughing involves inversion of the whole topsoil, for instance by mouldboard ploughing. In areas with high risk of runoff during autumn and winter, spring ploughing instead of autumn ploughing for a spring crop is an important method for reducing erosion and PP losses. On erodible soils, spring ploughing compared to traditional autumn ploughing has been shown to decrease soil erosion by up to 80 % (Lundekvam, 2007) and decreases total P losses by 60-80%. On less erodible soils, the effect of spring ploughing on total P losses is lower and may even be negative in surface runoff (Kværnø and Bechmann, 2010). Additionally, as for direct drilling, increased losses of DRP during winter may occur from spring ploughed soils. This has been demonstrated in several Scandinavian field experiments and the negative effect has to be combined with the positive effect of reduced erosion and PP (Table 8.2), depending on the situation in the area of receipt.

Spring ploughing can be applied to most soils in Scandinavia, but should be targeted to CSAs. However, on heavy clays, crop establishment in spring may suffer because of soil structural damage when these soils are not tilled in autumn.

Table 8.2
Experimental field results on soil tillage as a measure to reduce losses of dissolved reactive phosphorus (DRP) and particulate-bound phosphorus (PP) from clay (>30% clay) or clay loam soil (25-40 % clay) with surface runoff or subsurface lateral movements to drains. Results are expressed as relative values where P losses from conventional ploughing in mid-autumn are set to 1

	DRP	**PP**
Tillage without ploughing		
Direct drilling in autumn	3.5[c]	0.3[c]
Direct drilling in spring	4.0[f]	0.4[f]
Deep cultivation in autumn	1.4[c] 1.0[f]	1.0[c] 0.8[f]
Shallow cultivation, early autumn	1.9[d] 1.3[c] 0.5[f] 4.4[a]	0.5[d] 0.6[b] 0.8[c] 0.9[f]
	1.9[a] 1.4[g] 1.0[g]	0.4[a] 0.9[a] 1.3[g] 0.9[g]
Shallow cultivation, late autumn	1.7[d]	1.2[d]
Shallow cultivation, spring		0.2[b]
Tillage with ploughing		
Ploughing early autumn	0.7[e]	0.9[e] 0.7[b]
Ploughing in spring	1.4[c] 1.8[g] 1.1[g] 1.2[c] 1.1[f]	0.9[c] 0.7[g] 0.9[g] 0.6[b] 0.5[c] 0.5[f]
Contour ploughing	1.0[c]	0.2[b] 0.8[c]

Values based on [a] Koskiaho et al., 2002 (PP is presented as the difference between TOTP and DRP); [b] Lundekvam, 1993 (all TOTP values estimated to be PP); [c] Puustinen et al., 2005 (runoff above plough pan is here presented as surface runoff); [d] Stenberg et al., 2009 and unpublished (all TOTP values is estimated to be PP); [e] Ulén, 1997 (short slopes); [f] Ulén and Kalisky, 2005; [g] Uusitalo et al., 2007

The combination of spring ploughing and cultivation of catch crops has not been thoroughly evaluated with respect to DRP losses. In Scandinavia, the practice of avoiding tillage during autumn is regarded as an important measure against N leaching (Factsheet Ulén et al., 2010j). Ploughing in early autumn results in increased microbial activity and N mineralisation which constitute a significant risk of elevated N leaching especially when no crop is growing during winter. Studies on sandy soils in Denmark and southern Sweden demonstrated that N leaching was clearly reduced when spring ploughing was practiced instead of autumn ploughing (Djurhuus and Olsen, 1997; Stenberg et al., 1999).

Contour ploughing
Contour ploughing means ploughing, cultivating and seed drilling perpendicular to slopes and along contours (Factsheet Ulén et al., 2010a). Contour ploughing is applicable for all areas with risk of surface runoff and erosion. The importance of rill erosion needs to be stressed. Eroded particles in surface runoff may be filtered through grass and allowed to settle in depressions in fields with undulating topography. Therefore, grass-covered waterways in combination with contour ploughing are desirable.

8.3 Improved soil structure

Avoiding soil compaction.
Soil compaction is an extended problem in Europe (Batey, 2009). Soil management in relation to certain crops (e.g. potato) are more likely to cause soil compaction than others. We recognise two types of soil compaction, topsoil compaction and subsoil compaction. The latter includes compaction of both the plough pan and any unloosened subsoil. In Scandinavia, tillage to avoid soil compaction is possibly an important measure in reducing PP losses and to some extent DRP losses, both via surface and subsurface flow from fine-textured

soil. However, the effect is hard to quantify and may only be conceptually evaluated (Factsheets Ulén et al., 2010d, h).

Deep ploughing and subsoiling.
Conventional ploughing is done at around 20 cm dept in Scandinavia. Deep ploughing is soil inversion to a deeper depth than conventional ploughing. Subsoiling means inversion the soil to a depth deeper than the subsoil (even down to 1 m in the Netherlands) and it may provide mixing of P rich topsoil with P deficient subsoil and thereby reducing availability of P to leaching (Table 8.2). Deep ploughing should be complemented with other options such as loosening the plough pan with deep-rooted crops in order to break up any compacted subsoil or adding lime above the subsoil. Such combined measures are possibly highly relevant for reducing PP losses from compacted soils e.g. clay soil in Scandinavia where soil erosion probably will decrease as a result of improved infiltration.

Avoid tramlines
Tramlines are semi-permanent wheelways for farm machinery to travel down during spraying and fertilising operations without causing wheeling damage to the rest of the field, a practice sometimes referred to as controlled traffic. Thus tramlines are important vectors of runoff, causing increased mobilisation of sediment and phosphorus (Withers et al., 2006; Silgram et al., 2007). The solution could be either to delay the establishment of tramlines until crop cover has been established (or alternatively until spring), or to shallow cultivate them using a simple goose-foot tine (Factsheet Ulén et al., 2010n). Avoiding tramlines is applicable for winter cereals in most arable farming (sandy-loamy-silty clayey soils) and particularly on light soils in areas with higher winter rainfall and snow cover. When spraying pesticides on such crops, low ground-pressure vehicles should be used.

8.4 Soil amendments

Addition of organic material from manure, sludge and compost
Low soil organic matter levels can give rise to soil structural problems and increased risk of all kinds of erosion especially for clay and silty soils. The opposite, to maintain or enhance the content of soil organic matter, may maintain good soil structure, and in turn enhance the infiltration, retention and movement of water through the soil (Rasmussen, 1999). By a regular addition of manure or other organic by-products, soil organic content may be improved. However, such addition is also succeeded by organic acids known to appear during decomposition of organic matter. The organic acids are known to form complexes with iron (Fe) and aluminium (Al) and reduce the number of sites for P sorption. Organic N added by repeated manures or organic by-products applications enhance organic N stock or/and influence soil mineralisation rates according the organic matter quality (Chang and Janzen, 1996). Therefore, addition of organic material may also contribute to increased risk of nutrient loss in surface runoff and should be incorporated shortly after addition. Incorporation involves soil tillage and hence, the timing of addition of organic material in relation risk of runoff is crucial. These changes in soil N content have to be quantified in order to adapt fertilisation practices so as to limit N leaching.

Addition of lime and other chemicals
Calcium and pH are factors that control the chemical mobility of P in terms of precipitation and chemical complexation mechanisms. No-till management systems result in an uneven distribution of pH in the soil profile with higher pH in topsoil and lower pH in the subsoil. In addition, lime granules, local materials (e.g. waste products from industries) and even sludge from waste water treatment plants may be used as stabilising agents. Certain forms of lime ('quick lime'; CaO) highly improves the soil structure immediately. Therefore P losses via surface runoff and drainage and in both dissolved and particulate form may be reduced after liming. In addition, P is directly precipitated by the added lime. In Finland, gypsum is tested for its effects on P

leaching. Alum sludge is produced in water treatment plants when the raw water is treated with aluminium sulphate to precipitate organic and inorganic material. Such sludge has been proven to have a good effect on soil structure in Norwegian experiments (Øgaard et al., 2007) and also increase the sorption capacity (Factsheet Chardon and Dorioz, 2010). Other industrial and municipality by-products may, in addition, act as stabilisators such as mesalime from paper-mill industry, drinking water treatments residuals and fly-ash and biogas slurry from energy production.

8.5 Adverse effects of soil management

No-till or shallow tillage without soil inversion has been shown to have very good effects on reducing P losses from most sloping clay loam and clay soils especially in areas of moderate and high erosion risk. Since soil erosion is an extended problem in many European countries (Boardman and Poesen, 2006) it may probably have a potential in many other countries as well. However, no-till systems should be carefully evaluated for the following adverse effects:

a) Accumulated surplus P fertilizer and organic matter on or near the soil surface pose a clear risk of P release and therefore an increased DRP in surface and subsurface water. This is especially the case in climates with repeated frost events.
b) Accumulated organic matter and N in upper soil layers with increased risk of N release to surface runoff.
c) Tillage without ploughing may result in decreased crop yields and more weeds.
d) Tillage without ploughing may increase the risk of *Fusarium* infection which may result in an increased use of pesticides and pesticide leaching.
e) Tillage may have a negative effect on soil biodiversity.

Spring tillage reduces erosion and PP losses and will reduce most of the problems occurring under no-till. However, the positive effects of no-till on soil structure will be deleted by annually repeated spring-ploughing. Furthermore, the positive effect of no-till on soil biodiversity will also disappear (Postma-Blaauw et al., 2010).

In conclusion

1. Direct drilling and shallow cultivation reduce erosion and total phosphorus losses from high risk areas compared to inversion of the soil by ploughing.
2. Spring ploughing compared to autumn ploughing reduce erosion risk and P losses during winter.
3. Improved soil structure by deep loosening of soil contribute to increased infiltration and less erosion and P losses.
4. Soil amendments contribute to improved soil structure, but adding organic material may contribute to increased nutrient losses.
5. Accumulated surplus P fertilizer and organic matter on or near the soil surface pose a risk of P release and therefore an increased DRP in surface water.

List of factsheets

Bechmann, M., and T. Krogstad, 2010. Soil tillage methods to reduce erosion. [FS]
Bechmann, M., T. Krogstad, B. Ulén, 2010. On clay and silty soils - direct-drilling in early autumn instead of ploughing for winter crops. [FS]
Chardon, W.J. and J.M. Dorioz, 2010. Phosphorus immobilizing amendments to soil. [FS]
Ulén, B., M. Bechmann, T. Krogstad, 2010a. Cultivation along soil contours and across slopes. [FS]
Ulén, B., M. Bechmann, T. Krogstad, 2010b. Maintenance and enhancement of soil organic matter content. [FS]
Ulén, B., M. Bechmann, T. Krogstad, 2010c. Mulching. [FS]
Ulén, B., M. Bechmann, T. Krogstad, 2010d. On clay and organic soils - tillage conditions to avoid soil compaction. [FS]

Ulén, B., M. Bechmann, T. Krogstad, 2010e. On clay and silty soils - direct drilling (no tillage) in spring compared to spring ploughing. [FS]
Ulén, B., M. Bechmann, T. Krogstad, 2010f. On clay and silty soils - establishment of in-field buffer strips. [FS]
Ulén, B., M. Bechmann, T. Krogstad, 2010g. On clay and silty soils - soil cultivation to get an uneven winter soil surface. [FS]
Ulén, B., M. Bechmann, T. Krogstad, 2010h. On compacted clay and silty soils - biological loosening or deep ploughing to destroy the plough pan and deep macropores. [FS]
Ulén, B., M. Bechmann, T. Krogstad, 2010i. On medium to heavy soils - shallow cultivation in late autumn instead of autumn ploughing for spring crops. [FS]
Ulén, B., M. Bechmann, T. Krogstad, 2010j. On sandy soils - spring rather than autumn cultivation for spring crop establishment. [FS]
Ulén, B., M. Bechmann, T. Krogstad, 2010k. Ploughing to reduce stratification and shallow macropores. [FS]
Ulén, B., M. Bechmann, T. Krogstad, 2010l. Shallow cultivation in early autumn instead of ploughing for winter crops. [FS]
Ulén, B., M. Bechmann, T. Krogstad, 2010m. Soil tillage and crop establishment practices to decrease nutrient losses to water - introduction. [FS]
Ulén, B., M. Bechmann, T. Krogstad, 2010n. Tillage to avoid tramlines. [FS]
Ulén, B., M. Bechmann, T. Krogstad, 2010o. Use of soil stabilizers. [FS]

References

Addiscott, T.M. and Thomas, D. 2000. Tillage mineralization and leaching: phosphate. Soil Tillage Res. 53, 255-273.

Baker, D. and Richards, R.P. 2002. Phosphorus budgets in riverine phosphorus exports in north-western Ohio watershed. J. Environ. Qual. 31, 96-108.

Batey, T. 2009. Soil compaction and soil management - a review. Soil Use Manage. 25, 335-345.

Boardman, J. and Poesen, J. (eds.) 2006. Soil Erosion in Europe. Wiley & Sons, Ltd, London.

Chang C. and Janzen, H.H. 1996. Long-term fate of nitrogen from annual feedlot manure applications. J. Environ. Qual. 25, 785-790.

Djurhuus, J. and Olsen, P. 1997. Nitrate leaching after cut grass/clover leys as affected by time of ploughing. Soil Use Manage. 13, 61-67.

Gaynor, J.D. and Findlay, W.I. 1995. Soil phosphorus loss from conservation and conventional tillage in corn production. J. Environ. Qual. 24, 734-741.

Grønsten, H.A., Øygarden, L. and Skjevdal, R. 2007. Nutrient runoff, phosphorus reduced tillage, soil erosion and winter wheat. Bioforsk report vol 2 nr 60. 2007. ISBN 978-82-17-00231-4. 65 pp. (in Norwegian).

Holland, J.M. 2004. The environmental consequences of adopting conservation tillage in Europe: reviewing the evidence. Agri. Ecosys. Environ. 103, 1-25.

Koskiaho, J., Kallio, K. and Puustinen, M. 2002. Reduced tillage: Influence on erosion and nutrient losses in a clayey field in southern Finland. Agric. Food Sci. Finland 11, 37-50.

Kværnø, S. & Bechmann, M. 2010. Transport av jord og næringsstoffer i overflate- og grøftevann. Sammenstilling av resultater fra rutefelter og småfelt i Norge. Bioforsk report 5/30. (In Norwegian, English summary).

Logan, T.J., Lal., R. and Dick, W.A. 1991. Tillage systems and soil properties in North America. Soil Till. Res. 20, 241-270.

Lundekvam, H. 1993. Runoff, erosion and nutrient losses under different cultivation systems and soil types in Akershus/Østfold Report Nordic Agriculture Reserach, Suppl 16, 124-141.

Lundekvam, H. and Skøien S. 1998. Soil erosion in Norway. An overview of measures from soil loss plots. Soil Use Manage. 14, 84-89.

Lundekvam, H. 2007. Plot studies and modelling of hydrology and erosion in southeast Norway. Catena 71, 200-209.

McDowell, L.L. and McGregor, K.C. 1984. Plant nutrient losses in runoff from conservation tillage corn. Soil Till. Res. 4, 79-91.

Oorts, K., Laurent, F., Mary, B., Thiebeau, P., Labreuche, J. and Nicolardot, B. 2007. Experimental and simulated soil mineral N dynamics for long-term tillage systems in northern France. Soil Till. Res. 94, 441-456.

Øgaard, A.F., Grønsten, H.A., Sveistrup, T.E., Bøen, A., Kværnø, S. H. and Haraldsen, T.K. 2008. Potentielle miljøeffekter av å tilføre avløpsslam ti jordbruksarealer. Resultater fra feltforsøk I korn, 1. forsøkår 2007. Bioforsk Rapport Vol 3 No 59. 43 pp.

Postma-Blaauw, M.B., de Goede, R.G.M., Bloem, J., Faber, J.H. and Brussaard, L. 2010. Soil biota community structure and abundance under agricultural intensification and extensification. Ecology 91, 460 - 473.

Puustinen, M., Koskiaho, J., and Peltonen, K. 2005. Influence of cultivation methods on suspended solids and phosphorus concentrations in surface runoff on clayey sloped fields in boreal climate. Agric. Ecosys. Environ. 105, 565-579.

Rasmussen, 1999. Impact of ploughless soil tillage on yield and soil quality: a Scandinavian review. Soil Till. Res. 53, 3-14.

Sharpley, A.N. and Smith, S.J. 1994. Wheat tillage and water quality in the southern plains. Soil Till. Res. 30, 33-38.

Silgram, M. and Shepherd, M.A. 1999. The Effects of Cultivation on Soil Nitrogen Mineralization. Advances in Agronomy, 65, 267-311.

Silgram, M., Jackson, B., Quinton, J., Stevens, C. and Bailey, A. 2007. Can tramline management be an effective tool for mitigating phosphorus and sediment loss? In: Heckrath, G., Rubaek, G. & Kronvang, B. (eds). Fifth International Phosphorus Workshop 3-7 September 2007, Silkeborg, Denmark p. 287-289.

Stenberg, M., Aronsson, H., Lindén, B., Rydberg, T. and Gustafson, A. 1999. Soil mineral nitrogen and nitrate leaching losses in soil tillage systems combined with a catch crop. Soil Till. Res. 50, 115-125.

Strauss, P. and Smid, G. 2004. Einfluss von Saattechnik und Zwischenfrucht auf den Oberflächenabfluss und die Bodenerosion im Zuckerrübenbau. Schriftenreihe BAW, 20, 91-109.
Ulén, B. 1997. Nutrient losses by surface run-off from soils with winter cover crops and spring ploughed soils in the south of Sweden. Soil Till Res. 44, 165-177.

Ulén, B. and Kalisky, T. 2005. Water erosion and phosphorus problems in an agricultural catchment – Need for natural research for implementation of the EU Water Framework Directive. Environ. Sci. Policy 8, 477-484.

Ulén, B., Aronsson, H., Bechmann, M., Krogstad ,T., Øygarden, L. and Stenberg M. 2010. Soil tillage methods to control phosphorus loss and potential side-effects – a Scandinavian review. Soil Use Manage. 26, 94-107.

Uusitalo, R., Turtola, E. and Lemola, R. 2007. Phosphorus losses from a subdrained clayey soil as affected by cultivation practices. Agric. Food Sci. 16, 352-364.

Withers, P.J.A., Hodgkinson, R.A., Bates, A. and Withers, C.M. 2006. Some effects of tramlines on surface runoff, sediment and phosphorus mobilization on an erosion-prone soil. Soil Use Manage. 22, 245-255.

Withers, P.J.A., Hodgkinson, R.A., Bates, A. and Withers, C.L. 2007. Soil cultivation effects on sediment and phosphorus mobilization on surface runoff for three contrasting soil types in England. Soil Till. Res. 93, 438-451.

9 Water management within agricultural land

O.F. Schoumans, I.G.A.M. Noij, A. Lo Porte, W.J. Chardon and M. Bechmann

9.1 Introduction

Nutrient losses from farming practices are mainly caused by transport of the excess of water over and/or through agricultural land. Since phosphorus (P) is the limiting nutrient in most European inland waters (Carpenter, 2008), the focus of the water management measures is on P. However, the influence of the P mitigation options on nitrogen (N) losses to groundwater surface water and air is also discussed. In some cases there will be a negative effect on nitrogen losses, so called pollution swapping; see Section 4.3). With respect to P losses erosion of detached soil particles by overland flow and the transport of soluble soil components via surface runoff and further transport to the surface waters is one of the most important pathways of the pollution of surface waters in hilly and mountain areas (Heathwaite et al., 2005). According to Borda et al. (2010) the magnitude of soil particle loss in runoff appears to be greatly influenced by texture (Lado et al., 2004), type of clay (Curtin et al., 1994), organic carbon (Rhoton et al., 2002) and soil surface characteristics (Udawatta et al., 2004). On the other hand in flat or low hilly areas phosphorus leaching through the soil or via artificial drainage is an important pathway of the pollution in surface water (Ulén and Mattsson, 2003; Nelson et al., 2005; Chardon and Schoumans, 2007). With respect to nitrogen the N-losses are mainly caused by leaching through the soil via deeper groundwater layers to the surface water and the contribution of N-losses by surface runoff and erosion is relative low. The main reason is that mineral nitrogen (mainly nitrate) is relative mobile compared phosphate. Phosphate normally accumulates in the top soil because of chemical binding. Both organic nitrogen and phosphorus are also accumulating in the ploughed layer in the organic matter fraction of the soil. Hence, organic N and P can be transported as soluble organic material or as organic particles by erosion and runoff.

Since water transport is the main driving force of nutrient loss, many measures deal with changing or blocking the water flow to reduce the contribution of agricultural pollution. Soil management measures (Chapter 8) are often indirectly related to reduce a specific pathway of nutrient transport (mainly runoff/erosion).

Runoff following rainfall may transport particles due to soil detachment. Since nutrients are often attached to sediments, nutrient losses can occur from the field to other fields or directly to adjacent surface water. The amount of particle bound nutrients in runoff water is highly correlated to the amount of sediments transported and particle size of the transported material, because smaller particles (clay particles versus sand particles) have a high specific surface area and therefore contain more adsorbed nutrients. The amount of phosphate bound to soil particles is much higher then the amount of ammonium (and nitrate is negligible). In case organic particles are transported the amount of transported nitrogen is higher than the amount of phosphorus, because the N:P ratio of organic fractions in the soil (ploughed layer) is always more than 1. A C:N:P ratio of 100:10:1 for soil organic matter is generally accepted although values range considerably (Tilsdale et al., 1990; Stevenson and Cole, 1999).

Special attention has to be given to high rainfall events immediately after surface manure or fertilizer application. Nutrient losses can be very high under these conditions (Heathwaite et al., 1997).

Furthermore, nutrients (phosphate and ammonium) can be desorbed from the soil and transported with the overland water flow (surface runoff). The soluble *inorganic* nutrients (mainly phosphate and ammonium) in runoff water depend highly on the amount of adsorbed phosphate and ammonium in the soil. Since the amount of phosphate that is adsorbed to the soil is much higher than the amount of ammonium, there is a good relationship between the amount of inorganic P in runoff water and the soil P status of the ploughed layer (Allen et al., 2006). The magnitude of inorganic P losses by leaching seems to be greatly influenced by the phosphate saturation degree (Van der Zee, 1988).

The soluble organic nutrients in soil solution depend on factors like the organic matter content, physical and chemical type of organic material, affinity to adsorb to the soil and the pH of the soil (McDowell and Koopmans, 2006; Wilson and Xenopoulos, 2009). However, less attention is given to the description of the soil mechanisms and modelling approaches to describe organic nutrient transport because compared to erosion (particle P transport) the contribution is low. Regarding organic nutrient losses the focus is on organic soils (Larsen et al., 1958; Duxbury and Peverly, 1978; Reddy, 1983; Cogger and Duxbury, 1984; Gjettermann et al., 2007; Waldron et al., 2008) and the impact of manure application on runoff losses (Gessel et al., 2004; Sharpley et al., 2004; Van Es et al., 2004; Kleinman et al., 2005; Schelde et al., 2006; Wortmann and Walters, 2006; Chardon et al., 2007; Koopmans et al., 2007; Smith et al., 2007; Allen and Mallarino, 2008; Vadas et al., 2009).

If the main pathways of nutrient losses (e.g. overland flow, subsurface leaching, deep groundwater flow or vertical transport through the top soil in combination with lateral flow via artificial drains) and the main nutrient components in soil solution (particulate material, colloids, inorganic soluble nutrients, organic soluble nutrients) are well known than the right measures can be applied to reduce the nutrient losses. Via water measures, which mainly focus on changing the length of the pathway of the water flow, the nutrient concentrations in the water flow will be reduced by adsorption (P), denitrification (NO_3) or physical processes (sterical effects; sedimentation). However, it is important to notice that changing the water flow, will also change the water conditions in soil and therefore, will have an impact on the chemical (sorption and precipitation) and biological processes like, mineralization / immobilization of organic matter and nitrification/denitrification. As a result, pollution swapping can occur.

In this chapter different water management measures are described in a general way. The water management measures focus directly on changing or blocking the water flow, and consequently affecting nutrient losses. The water management measures are mainly grouped according to pathways:

1. Reduce nutrient loss by overland flow to surface water
2. Reduce nutrient loss by subsurface flow to surface water
3. Reduce nutrient loss by artificial drainage to surface water
4. Reduce nutrient loss via controlled surface water level
5. Reduce nutrient loss by adapted irrigation methods

In the following sections the water management measures (9.3 - 9.8) and the interaction between the water content and nutrient concentration are discussed in detail to gain better understanding of the processes (section 9.2).

9.2 Influence of the water content on the nutrient processes in soils

The water content in soil should be sufficient in relation to crop demand. The yield will decrease, or even plants will die, if the water content is too high or too low for a longer period. Furthermore, the availability of nutrients will change because of the importance of chemical and biological processes that occur. In relation to the nitrogen and phosphorus cycling the most important processes which are influenced by the water content (besides crop uptake) are:

- Denitrification
- Oxidation / reduction

Denitrification

Due to manure or nitrogen fertilizer applications the nitrogen concentration will increase. Manure contains organic N and mainly ammonium N. Fertilizer N can be in form of NH_4 or NO_3 or both. As a result of decomposition of organic matter NH_4 comes available. The NH_4 of mineralized organic matter or applied fertilizer will be nitrified into NO_3 by means of biological processes. Nitrate is very soluble and can easily leached out of the ploughed layer. However, the nitrate concentration in soil solution will be reduced if the water content (water filled pore space, WFPS, which is defined as the ratio between the actual soil moisture content and the content at saturation) increases because the following reaction will take place as a result of biological activities:

$$5C_6H_{12}O_6 + 24NO_3^- \rightarrow 30CO_2 + 18H_2O + 12N_2\uparrow + 24OH^-$$

Figure 9.1 shows the relationship between the water filled pore space of the soil and the denitrification capacity (Heinen, 2003). If the soil water content becomes fully saturated all available nitrate will be volatilized (mainly as N_2 and partly as N_2O). So a wet soil will decrease the nitrate leaching to groundwater and surface water. However, if the soil is wet the drainage of ammonium and soluble organic substances will increase compared to a less wet soil, because the impact of a rainfall event will lead to more water drainage to the surface water.

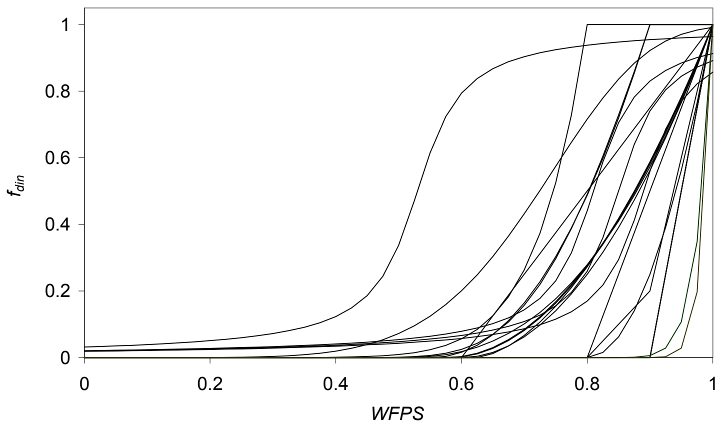

Figure 9.1

Influence of the water filled pore space (WFPS) on the denitrification factor (f_{den}) (at a value of zero no denitrification takes place and at a value of 1 full denitrification will occur; different lines are used in different nitrogen models) (after Heinen, 2003)

Oxidation/reduction

Compared to nitrate the opposite reaction occurs for phosphate. Phosphate is highly bound to iron- and aluminium(hydroxides) in non-calcareous soils. If the water content increases Fe^{3+} will be reduced to Fe^{2+} and the adsorbed phosphate to Fe^{3+} becomes available because Fe^{2+} is very soluble:

$$Fe(H_2PO_4)_3\ (s) + e^- \rightarrow Fe^{2+} + 3H_2PO_4^-$$

Al(hydr)oxides are still stable, because Al^{3+} is not reduced. In calcareous soils, where phosphate is mainly adsorbed to lime ($Ca(Mg)CO_3$), the influence of reduction is less important.

So, as a result of denitrification processes and the oxidation/reduction processes we can understand why the nitrate concentration will decrease and the phosphate concentration will increase in non-calcareous soils as a result of the water content (reduced circumstances) of the soil. Figure 9.2 shows the overall impact of the rewetting a sandy soil on the nitrate, iron and phosphate concentration (Schoumans and Kohlenberg, 1997). At first the nitrate drops fast (within one day) and also the Eh (redox potential) decreases. After a couple of days the Fe concentration increases and also the phosphate adsorbed to the iron is released. Based on this information it is better to understand in which way water measures (following sections) will influence the nutrient losses to groundwater and surface water.

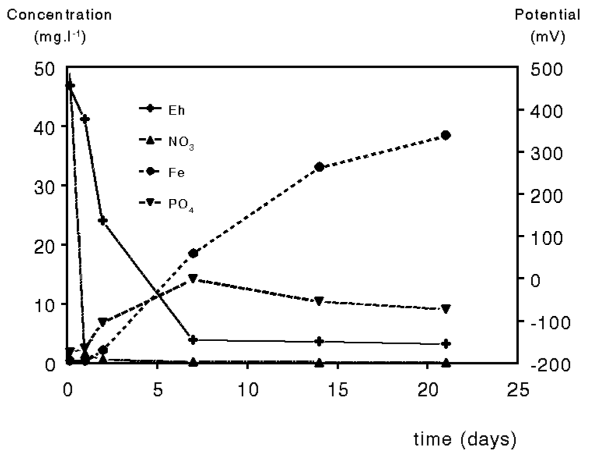

Figure 9.2
Influence of time in a complete wet sandy soil on the nitrate (▲), iron (•) and phosphate (▼) concentration and the redox potential (+) (after Schoumans and Köhlenberg, 1997)

9.3 Overland flow

Blocking or changing the overland water flow by constructions can be achieved via different kinds of measures:

i. Construct ponding systems / increase soil levels along ditches
ii. Construct grassed waterways
iii. Install sedimentation boxes

As the water moves across the field, it is important to decrease the flow rate, and facilitate an even infiltration into the soil, because soil detachment will be reduced and particles in the flow will deposit in lower parts of the field. Furthermore, denitrification occurs that will reduce the nitrate losses via overland flow. Lowering the water flow rate and increasing the length of pathway of the overland water flow will reduce the nutrient losses. In the lower parts of the field also the dissolved P can infiltrate into the soil again.

Creating *sediment ponds* at the edges of the field can also reduce suspended material and nutrient losses to the surface waters (Brown et al., 1981). Those sediment ponds remove suspended material from the overland water flow because the water flow rate will be reduced (blocked) allowing particles, and associated nutrients, to settle. Furthermore, denitrification is stimulated (wet conditions). Such a blocking system can be achieved by increasing the soil level along the ditches (also known as ponding) and is visualized in Figure 9.3. The ditch or stream may also be designed broader to have a long floodplain. Especially, in flat areas such a measure will be effective. In hilly areas the effectiveness will depend on the amount of water flow, more specific, on the amount of water that can be hold by the ponding. A serious disadvantage of the measure will be that the field will be wet for a longer period and the farmers will not be able to cultivate such fields during wet circumstances, which can occur in early spring time (sowing time). Moreover, anoxic conditions in the ponded soil may increase P leaching making the situation even worse (as demonstrated in Figure 9.2).

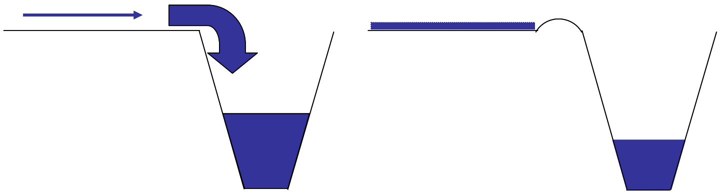

Figure 9.3
Schematic visualization of the impact of measure 'ponding' on the water flow (left old situation and right new situation)

The most experiences of the impact of this measure can be obtained from research under irrigation conditions (Bjorneberg et al., 2002). Well-constructed ponds can potentially remove 65-75 percent of the sediment and 25-33 percent of total P entering the pond. Total P reduction will depend on the relative amounts of dissolved and particulate P in the pond inflow (Bjorneberg and Leytem, SERA-17 letter). However, some of these findings are from situations with low P losses. In flat sandy areas in the Netherlands reductions in nutrient loads to the surface water have been calculated of 12-41 % for P and 0 to 11% for N (Noij et al., 2008).

Another option is to construct grassed waterways. Grass waterways are channels of grass within fields (e.g. arable land) that are constructed in order to get a more condensed and controlled water flow with low velocities. The application of such a measure is shown in Figure 9.4. Compared to arable land grassland is rather effective to reduce the water flow rate and therefore more effective in reducing nutrient losses by runoff/erosion. Furthermore, less soil will be eroded and less gullies will be formed. As a result constructing grass waterways within e.g. arable land, will have a positive effect on reducing nutrient losses by erosion. An outlet is often installed at the base of the drainage way to stabilize the waterway and prevent a new gully from forming. This vegetation within the waterway may act as a filter, absorbing or taking up some of the chemicals and nutrients in runoff water. As a result, nutrient losses caused by soil erosion and runoff will be reduced. The effectiveness of the grassed waterways depends on soil characteristics, land slope/topography, the vegetation, area for establishment and the correct construction and maintenance (SERA-17) (Pfost and Caldwell, 1993; Stone, 1994;.Fiener and Auerswald, 2009; Hunt et al., 2007; Kenwick et al., 2009; Shelton et al., 2009; Zhou et al., 2009). There are also some limitations. Grass waterways need sufficiently sized land areas and specific additional treatment (working around it with farm equipment). There is also a loss of acreage for crop production.

Figure 9.4
Two pictures of the application of grassed waterways

The third option deals with intake of surface water into the artificial subsurface drainage system via intake-wells to avoid gully erosion and P losses (Aspmo, 1989). During runoff events in an undulating landscape, surface runoff often accumulates in depressions and form concentrated flow and gully erosion. To target this problem intake-wells for surface water are placed in the depressions with a backfill that helps collect the water efficiently. The aim is to avoid the concentrated flow of water from having so much energy that it will cause erosion in the depression. The density of intake-wells should therefore correspond to the length of the depression. The intake-wells could be combined with a grassed waterway for filtering sediments from the water before they enter the subsurface drainage system. A disadvantage of these intake-wells is that they constitute a direct connection from field to surface water and if sediments are not filtered efficiently, they will be transported directly from the field to the stream. Therefore, the importance of connectivity of a field compared to risk of gully erosion should be evaluated before installing intake-wells in a field.

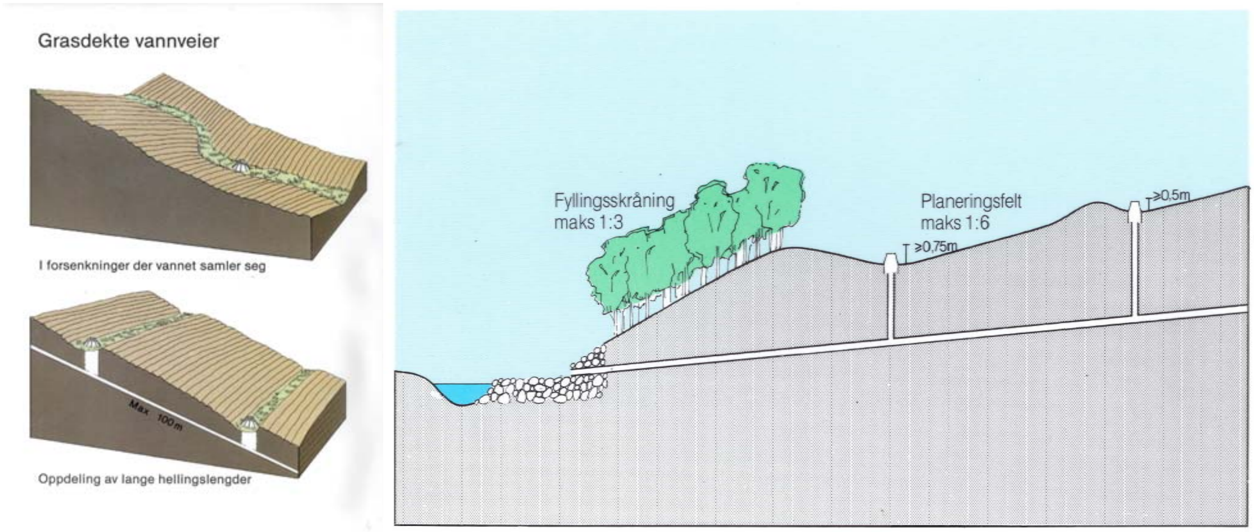

Figure 9.5
Picture of the application a sediment box in fields (left: in the downward pathflow and as a terrace; right: cross section. Aspmo, 1989)

9.4 Subsurface flow

The phosphorus transport through subsurface drainage is often supposed to be small or negligible because of its low mobility in soils due to high phosphate sorption capacity of the subsoil (buffering capacity) (Heathwaite and Dils, 2000) and (Hansen et al., 2002). However P transport could also occur by subsurface drainage if P levels in the soil are high and shallow groundwater tables frequently occur, like in flat areas (Chardon and Schoumans, 2007). Moreover, in areas with tile drains any mobilised dissolved nutrients as well as mobilised particulate bound nutrients can be transported via tile drains, especially when macropores or cracks are available. In areas where nutrient losses are mainly caused by subsurface drainage water, also water management measures can be constructed to reduce P losses. We distinguished three approaches for blocking or changing subsurface water flow and we will discuss the advantages and disadvantages:

i. Removal of trenches and ditches
ii. Allow field drainage systems to deteriorate
iii. Install artificial drains

Removal of trenches (and/or ditches) will directly reduce the fast transport ways of nutrient losses to the surface water, because the water flow have to flow over a longer distance to reach the surface water (ditches, brooks, rivers). By allowing field drainage of a trench or ditch to deteriorate such an effect will occur at longer terms. So the impact on the long term will be comparable although the time frame will be different. However, during the deterioration of the trench (ditch) also the water quality is improved because the residence time of the water flow in the ditches is increased. (Olli et al., 2009). The first option is visualized in Figure 9.6.

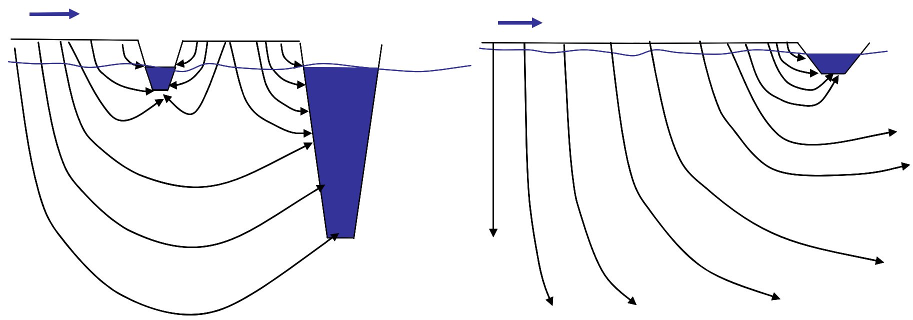

Figure 9.6
Schematic visualization of the impact 'removal of ditches' on the water flow

The general idea of this measure is to increase the pathway of water flow through the soil by reducing the drainage of the precipitation surplus via trenches and ditches. Removing trenching or ditches will have a maximum effect to block this pathway of subsurface leaching completely. Since trenches and ditches are constructed in rather wet areas and were necessary to ensure water discharge of the precipitation surplus to the surface water, the impact of this measure will be that more frequently high groundwater levels (rewetting of the area) will occur. It is important that runoff and erosion events are not increased as a side effect due to too wet conditions (long time of water saturation of the soil). So, the construction of this type of measures will have an impact on a larger area then the fields itself. Therefore such measures should be evaluated at least at sub-catchment level in the first place. The measure will be most effective when applied in field situations where transport by subsurface flow of soluble (in)organic nutrient components contributes to a large extend especially to the P losses to surface water, since phosphate is highly adsorbed to the soil. Such conditions occur in rather flat areas, deltas or brook/river stream areas with shallow groundwater levels. In those rather wet areas often many trenches and ditches were constructed to drain the excessive amount of water from the fields to the surface water. One of the major disadvantages is that the fields will be longer wet. As a result the farmers will not be able to cultivate such fields during wet circumstances, which can occur in early spring time (sowing time). Hence, the impact of this measure will be comparable to the ponding measure (Chapter 9.3). In flat areas in the Netherlands reductions of 12-41% have been calculated for P loads to the surface water for sandy areas, 0 to 11% for N loads to the surface water (Noij et al., 2008). On drained clays soils there was no effect and this measure is not possible in peat areas. Also in practice there is some evidence that deterioration of ditches had a positive effect on the improvement of the water quality in some studies.

An important option to reduce P losses of subsurface drainage to trenches and ditches is to install artificial drains in the soil, because the length of the pathway of the water flow is increased and phosphate can be sorbed in the subsoil, as visualized in figure 9.7. The general idea of this measure is to reduce shallow groundwater levels by building / construct artificial drains in soils. Since phosphorus is mainly adsorbed in the top soil (most situations less than 0.5 m) high P losses can occur if groundwater levels reach the enriched P layers in the top soil. By implementing artificial drains at greater depth (e.g. 0.8 - 1.2 m) the groundwater will not reach the enriched layers anymore and the water surplus (precipitation surplus) will be drained to surface waters at greater depth. The inorganic P equilibrium concentrations at depths one meter below surface will be rather low. Since also the organic P concentrations are often higher in the root zone than at 1 meter below surface also the P losses caused by organic P losses will be reduced, but still can significant source of P losses (Turner and Haygarth, 2000; Ulén and Mattsson, 2003). An additional reduction of P losses can be obtained after digging for tile drains the backfill is mixed with reactive material like reactive iron- or aluminum(hydr)oxides or lime (CaO) depending on the pH of the soil. McDowell et al. (2008) tested the use of a mixture of melter slag and basic slag as backfill material of field drains. Concentrations of DRP in drainage outflow were reduced with 73% when compared with the control, and for TP this was 51%. When the same slag mixture was placed in a sock at a drain outlet, reductions were 87 and 70%, respectively. Hanly et al. (2008) tested tephra, an Al-rich allophane material as backfill material of a drain. The average TP loss from the backfill treatments was 47% lower than the control treatment. Jaynes et al. (2008) compared a conventional drainage consisting of a free-flowing pipe installed 1.2 m below the soil surface with a tile with denitrification walls beside it, where trenches were excavated parallel to the tile and filled with woodchips serving as additional carbon sources to increase denitrification. NO_3 loss from the drain with walls was 55% lower than control. An additional positive effect will be the reduction of surface runoff by improved soil structure.

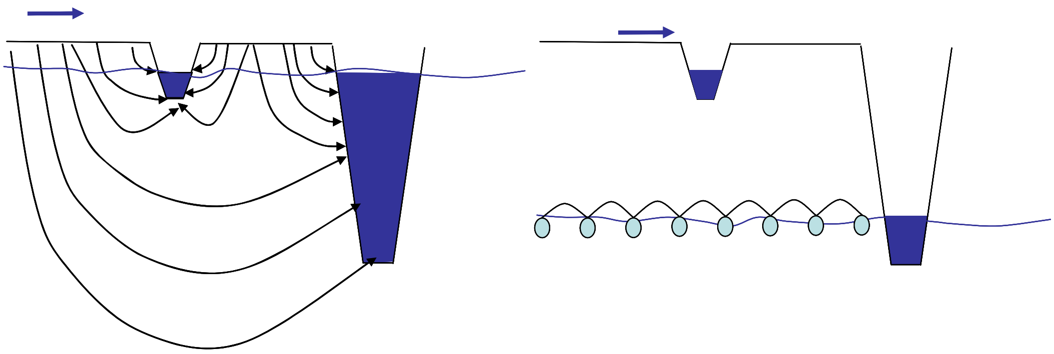

Figure 9.7
Schematic visualization of the impact of measure 'implementing artificial drains' on the water flow

The effectiveness of implementation artificial drains will have the highest impact in flat areas where shallow groundwater levels frequently occur. In flat areas of the Netherlands reductions of 74% have been calculated for sandy plots, but the N losses increase with 100% because the denitrification decreased (Noij et al., 2008; Van Boekel, 2008). In clay soils the N and P losses are both reduced, resp. 51% and 30%. This measure is not recommended for peat areas, because peat material will be oxidized as a result of the decrease of the water level. Some field studies addressed a reduction in P loss from 90% to 95% when drains at 2.5 m below surface were used in sandy soils. These drains are placed in permanent groundwater in combination with controlled groundwater level (Schoumans and Kruijne, 1995; Schoumans et al., 1995).

9.5 Tile drainage

In some areas the excess of water (precipitation surplus) is drained from the field via tile drainage. Tile drainage contributes to reduction of runoff by lowering the groundwater table and changing the groundwater flow (Section 9.2 and 9.3). We distinguished two measures in relation to artificial drainage:
i. Increase depth of tile drainage in combination with controlled drainage
ii. Let tile drainage water irrigate meadows (interrupt artificial drainage)

The pathway of the groundwater flow can be further manipulated by changing tile drain depth via controlled drainage. In areas with artificial drains, a controlled drainage system can be constructed as visualized in Figure 9.8. In the top of this figure the reference situation is illustrated, that is the influence of tile drainage on the groundwater level. At the bottom of this figure the impact of lowering the tile drains in combination with controlled groundwater level is visualised. Especially, in case the tile drains are installed deeper, i.e. below the groundwater table (see Figure 9.8; bottom), the reduction in P losses will also increase (provided the subsoil has sufficient phosphate sorption capacity). Since, the top soil will be less wet during the year the nitrate concentration leaching out to the groundwater from the root zone will increase. However, below the groundwater level denitrification will occur and consequently, the drained amount of water from deeper layers will contain less nitrate.

The measure will be most effective when applied under field conditions where high water soluble P components are transported to the surface water via artificial drainage. This can be the case in artificial drained areas with high P accumulation in the soil at greater depth. The effectiveness of increasing the depth of artificial drains in combination with controlled groundwater levels will have the most impact in situations with high P content of the soil and a rather shallow artificial drainage system. In flat sandy fields in the Netherlands reductions of 45% (P) and 9% (N) have been calculated (Noij et al., 2008; Van Bakel et al., 2008).

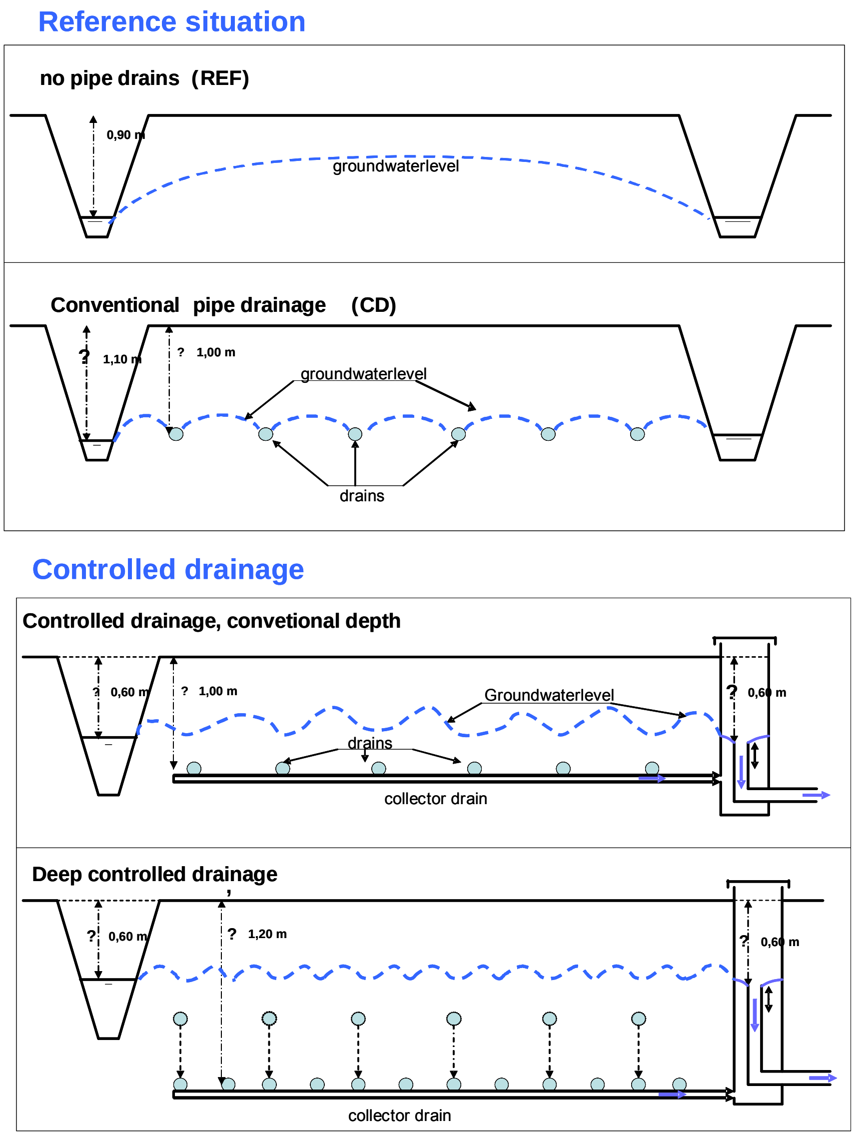

Figure 9.8

Schematic visualization of the impact of 'implementing deeper artificial drains and controlled drainage' on water flow compared to a reference situation

A second option is to change the pathway of the water flow via tile drains completely into overland flow and irrigate meadows or riparian areas. This means that the tile drainage should be forced up on meadows by cutting them at the edge of riparian areas instead of running underneath directly to the stream channels and ditches. In flat areas the water should be pumped up via a collected drain and in hilly areas this can be done by natural forces (let the drain come out at the surface in a certain low land area. However, in this last case the water flow have to be reduced via a perforated distribution pipe/tube across the meadow or riparian area.

The general idea of this measure is that polluted water with nutrients (particulate and soluble) is filtered by a meadow or a riparian area (Tanner et al., 2005; http://www.unl.edu/nac/aug94/rip-crop-2.html). The effectiveness of filtering of phosphorus will depend on the efficiency of the riparian area or the meadow (Chapter 11 and 12). Further distribution of the artificial drainage water over land is very important in order to reduce the flow rate in order to increase sedimentation of particles and denitrification of nitrate. At the long term the meadow can become saturated with P and the efficiency will be reduced. By removing the upper part of the top soil (few cm) the reduction capacity can be improved again.

In conclusion

1. Controlling the water flow from fields to surface water in one of the most important options to reduce nutrient pollution to the surface water, because nutrients from agricultural land are mainly transported to surface water via water movement.
2. Since the composition of different forms of nutrients (NH_4, NO_3, organic N, ortho-P, particulate P and organic P) in drainage water highly depends on the pathways of the water flow to surface water, it is important to have quantitative information of the distribution of nutrient losses via different pathways in order to select the most efficient measure.
3. Measures that change the (in field) water flow pathways, will also have an impact on the water content of the soil and therefore influence the occurrence and rate of biological-chemical processes. This can cause some pollution swapping.
4. The in field overland water flow can be directly changed by means of creating ponds or grassed waterways or via installing sediment boxes.
5. The subsurface nutrient losses from agricultural land can be changed via changes in the drainage system (trenches, ditches and tiles drains). Specific controlled tile drainage systems seems to have some interesting perspective, since the depth of the water discharges can be regulated even below the lower groundwater levels.

List of factsheets

Delgado, A., 2010a. Nutrient loss with surface irrigation. [FS]
Delgado, A., 2010b. Tailwater recovery on irrigated fields for water and nutrient cycling. [FS]
Delgado, A., 2010c. Use of controlled drainage for reducing the amount of water leaving a field. [FS]
Schoumans, O.F., 2010a. Construct grassed waterways. [FS]
Schoumans, O.F., 2010b. Construct ponding systems / increase soil levels along ditches. [FS]
Schoumans, O.F., 2010c. Increase depth of artificial drainage in combination with controlled drainage. [FS]
Schoumans, O.F., 2010d. Install artificial drains. [FS]
Schoumans, O.F., 2010e. Install sedimentation boxes. [FS]
Schoumans, O.F., 2010f. Let tile drainage water irrigate meadows / interrupt artificial drainage. [FS]
Schoumans, O.F., 2010g. Removal of trenches and ditches / allow field drainage systems to deteriorate. [FS]
Turtola, E., 2010. Improving sub-surface drainage systems. [FS]

References

Allen, B.L., A.P. Mallarino, J.G. Klatt, J.L. Baker and M. Camara, 2006. Soil and surface runoff phosphorus relationships for five typical USA midwest soils. J Environ Qual. 35, 99-610.

Allen B.L. and A.P. Mallarino, 2008. Effect of liquid swine manure rate, incorporation, and timing of rainfall on phosphorus loss with surface runoff. Journal of Environmental Quality 37, 125-137.

Aspmo, R. 1989. Informasjonskampagne, Rapport nr. 2. Handlingsplan mot landbruksforurensning. ISBN 82-7467-010-8 (In Norwegian).

Bjorneberg, D.L., D.T. Westermann and J.K. Aase. 2002. Nutrient losses in surface irrigation runoff. Journal of Soil and Water Conservation 57, 524-529.

Borda, T., P. J. A. Withers, D. Sacco, L. Zavattaro, and E. Barberis, 2010. Predicting mobilization of suspended sediments and phosphorus from soil properties: a case study from the north west Po valley, Piemonte, Italy. Soil Use and Management, 26, 310–319.

Brown, M.J. J.A. Bondurant and C.E. Brockway. 1981. *Ponding surface drainage water for sediment and phosphorus removal*. Transactions of the ASAE 24, 1478-1481.

Carpenter, S.R. 2008. Phosphorus control is critical to mitigation eutrophication. Proceedings of the National Academy of Sciences of the United States of America, 105, 11039–11040.

Chardon W.J., G.H. Aalderink and C. van der Salm, 2007. Phosphorus leaching from cow manure patches on soil columns. Journal of Environmental Quality 36, 17-22.

Chardon W.J. and O.F. Schoumans, 2007. Soil texture effects on the transport of phosphorus from agricultural land in river deltas of Northern Belgium, The Netherlands and North-West Germany. Soil Use and Management 23, 16-24.

Cogger C. and J.M. Duxbury, 1984. Factors affecting phosphorus losses from cultivated organic soils. J. Environ. Qual. 13, 111-114.

Curtin, D., C.A. Campbell, R.P. Zentner, and G.P. Lafond, 1994. Long-term management and clay dispersibility in two Haploborolls in Saskatchewan. Soil Science Society of America Journal, 58, 9–62.

Duxbury J.M. and J.H. Peverly, 1978. Nitrogen and phosphorus losses from organic soils. J. Environ. Qual. 7, 566-570.

Fiener P. and K. Auerswald, 2009. Effects of hydrodynamically rough grassed waterways on dissolved reactive phosphorus loads coming from agricultural watersheds. Journal of Environmental Quality 38, 548-559.

Gessel P.D., N.C. Hansen, J.F. Moncrief, M.A. Schmitt, 2004. Rate of fall-applied liquid swine manure: Effects on runoff transport of sediment and phosphorus. Journal of Environmental Quality 33, 1839-1844.

Gjettermann B., M. Styczen, S. Hansen, O.K. Borggaard, H.C.B. Hansen, 2007 Sorption and fractionation of dissolved organic matter and associated phosphorus in agricultural soil. Journal of Environmental Quality 36, 753-763.

Hanly, J.A., M.J. Hedley, and D.J. Horne. 2008. Evaluation of tephra for removing phosphorus from dairy farm drainage waters. Aust. J. Soil Res. 46, 542–551.

Hansen N.C., T.C. Daniel, A.N. Sharpley, K.L. Lemunyon, 2002. The fate and transport of phosphorus in agricultural systems. Journal of Soil and Water Conservation 57, 408-417.

Heathwaite A.L. and R.M. Dils R.M., 2000. Characterising phosphorus loss in surface and subsurface hydrological pathways. Sci. Total Environ. 251, 523-538.

Heathwaite, A.L., P. Griffiths, P. M. Haygarth, S. C. Jarvis and R. J. Parkinson, 1997. Phosphorus loss from grassland soils: implications of land management for the quality of receiving waters. Freshwater Contamination (Proceedings of Rabat Symposium S4, April-May 1997). IAHS Publ. no. 243, 1997.

Heathwaite, A.L., P.F. Quinn, and C.J.M. Hewett, 2005. Modelling and managing critical source areas of diffuse pollution from agricultural land using flow connectivity simulation. Journal of Hydrology, 304, 446-461.

Heinen, M., 2003. A simple denitrification model? Literature review, sensitivity analysis and application. Report 690, Alterra, Wageningen.

Hunt S.L., D.M. Temple and G.J. Hanson, R.D. Tejral, 2007. Evolution of vegetated waterways design, 2007 ASABE Annual International Meeting, Technical Papers, Minneapolis, MN. http://www.unl.edu/nac/aug94/rip-crop-2.html.

Jaynes, D.B., T.C. Kaspar, T.B. Moorman, and T.B. Parkin. 2008. In situ bioreactors and deep drain-pipe installation to reduce nitrate losses in artificially drained fields. J. Environ. Qual. 37, 429-436.

Kenwick R.A., M.R. Shammin and W.C. Sullivan, 2009. Preferences for riparian buffers. Landscape and Urban Planning 91, 88-96.

Kleinman P.J.A., M.S. Srinivasan, A.N. Sharpley and W.J. Gburek (2005) Phosphorus leaching through intact soil columns before and after poultry manure application. Soil Science 170, 153-166.

Koopmans G.F., W.J. Chardon and R.W. McDowell, 2007. Phosphorus movement and speciation in a sandy soil profile after long-term animal manure applications. Journal of Environmental Quality 36, 305-315.

Lado, M., M. Ben-Hur, and I. Shainberg, 2004. Soil wetting and texture effects on aggregate stability, seal formation, and erosion. Soil Science Society of America Journal, 68, 1992–1999.

Larsen J.E., R. Langston, G.F. Warren, 1958. Studies on leaching of applied P in organic soils. Soil Science Society of America Proceedings 22, 109-117.

McDowell R.W., G.F. Koopmans, 2006 Assessing the bioavailability of dissolved organic phosphorus in pasture and cultivated soils treated with different rates of nitrogen fertiliser. Soil Biology and Biochemistry 38, 61-70.

McDowell, R.W., A.N. Sharpley, and W. Bourke. 2008. Treatment of drainage water with industrial by-products to prevent phosphorus loss from tile-drained land. J. Environ. Qual. 37, 1575-1582.

Nelson N.O., J.E. Parsons, R.L. Mikkelsen, 2005. Field-scale evaluation of phosphorus leaching in acid sandy soils receiving swine waste. Journal of Environmental Quality 34, 2024-2035.

Noij, I.G.A.M., W. Corré, E.M.P.M. van Boekel, H. Oosterom, J. van Middelkoop, W. van Dijk, O. Clevering, L. Renaud and P.J.T van Bakel, Kosteneffectiviteit van alternatieve maatregelen voor bufferstroken in Nederland. Alterra report 1618, Wageningen, The Netherlands.

Olli G., A. Darracq A. and G. Destouni, 2009. Field study of phosphorous transport and retention in drainage reaches. Journal of Hydrology 365, 46-55.

Pfost, D.L. and L. Caldwell. 1993. Maintaining Grassed Waterways. University of Missouri Extension. Report No. G1504.

Reddy K.R., 1983. Soluble phosphorus release from organic soils. Agric. Ecosyst. Environ. 9, 373-382.

Rhoton, F.E., M.J. Shipitalo and D.L. Lindbo, 2002. Runoff and soil loss from midwestern and southeastern US silt loam soil as affected by tillage practice and soil organic matter content. Soil and Tillage Research, 66, 1–11.

Schelde K., L.W. De Jonge, C. Kjaergaard, M. Laegdsmand , G.H. Rubæk G.H., 2006. Effects of manure application and plowing on transport of colloids and phosphorus to tile drains. Vadose Zone Journal 5, 445-458.

Schoumans O.F. and R. Kruijne, 1995. Onderzoek naar maatregelen ter vermindering van de fosfaatuitspoeling uit landbouwgronden; meting van de fosfaatuitspoeling uit fosfaatverzadigde zandgrond met en zonder hydrologische maatregel, Wageningen, SC-DLO, 1995. Rapport 374.1, 111 blz.

Schoumans, O.F. and L. Köhlenberg, 1997. Invloed van veroudering van ijzerhydroxide en anaërobe omstandigheden op de fosfaatconcentratie in fosfaatverzadigde lagen. Staring Centrum, rapport 508. Wageningen. The Netherlands.

Schoumans O.F., R. Kruijne and D.T. van der Molen, 1995. Vermindering fosfaatuitspoeling; mogelijkheden bij fosfaatverzadigde gronden. Landschap 12 (1995), 6, 63-73.

Sharpley A.N., R.W. McDowell and P.J.A. Kleinman, 2004. Amounts, forms, and solubility of phosphorus in soils receiving manure. Soil Science Society of America Journal 68, 2048-2057.

Shelton D.P., R.A. Wilke, T.G. Franti and S.J. Josiah, 2009. Farmlink: Promoting conservation buffers farmer-to-farmer. Agroforestry Systems 75, 83-89.

Smith D.R., P.R. Owens, A.B. Leytem and E.A. Warnemuende, 2007. Nutrient losses from manure and fertilizer applications as impacted by time to first runoff event. Environmental Pollution 147, 131-137.

Stevenson, F.J. and M.A. Cole, 1999. Cycles of soil (second edition). John Wiley and Sons, Inc. Book. Hoboken, USA.

Stone, R., 1994. Grassed Waterways. Ontario Ministry of Agriculture and Food. Order #94-039.

Tanner C.C., M.L. Nguyen and J.P.S. Sukias, 2005. Nutrient removal by a constructed wetland treating subsurface drainage from grazed dairy pasture. Agriculture, Ecosystems & Environment 105, 145-162. http://www.unl.edu/nac/aug94/rip-crop-2.html.

Tilsdale, S.L., Nelson, W.L., Beaton, J.D. 1990. Soil fertility and fertilizers. Fourth edition. Maxwell Macmillan International Editions, New York, 754 pp. ISBN 0-02-946760-8.

Turner B.L. and P.M. Haygarth, 2000. Phosphorus Forms and Concentrations in Leachate under Four Grassland Soil Types. Soil Sci. Soc. Am. J. 64, 1090-1099.

Udawatta, R.P., P.P. Motavalli, P.P. and H.E. Garrett, 2004. Phosphorus loss and runoff characteristics in three adjacent agricultural watersheds with claypan soils. Journal of Environmental Quality 33, 1709-1719.

Ulén B. and L. Mattsson, 2003. Transport of phosphorus forms and of nitrate through a clay soil under grass and cereal production. Nutr. Cycl. Agroecosyst. 65, 129-140.

Vadas P.A., L.W. Good, P.A. Moore Jr and N. Widman, 2009. Estimating phosphorus loss in runoff from manure and fertilizer for a phosphorus loss quantification tool. Journal of Environmental Quality 38, 1645-1653.

Van Bakel, P.J.T. van, E.M.P.M. van Boekel, I.G.A.M. Noij, 2008. Modelonderzoek naar effecten van conventionele en samengestelde, peilgestuurde drainage op de hydrologie en de nutriëntenbelasting. Alterra report 1647, Wageningen. The Netherlands.

Van Boekel, E.M.P.M., 2008. Kosteneffectiviteit van alternatieve maatregelen voor bufferstroken in Nederland. Alterra, Alterra-report 1618 Appendix 4, Wageningen, The Netherlands.

Van der Zee, 1988. Transport of reactive contaminants in heterogeneous soil systems. Dissertation Agricultural University, Wageningen, The Netherlands.

Van Es H.M., R.R. Schindelbeck and W.E. Jokela, 2004. Effect of manure application timing, crop, and soil type on phosphorus leaching. Journal of Environmental Quality 33, 1070-1080.

Waldron S., H. Flowers, C. Arlaud, C. Bryant and S. McFarlane, 2008. The significance of organic carbon and nutrient export from peatland-dominated landscapes subject to disturbance. Biogeosciences Discussions 5, 1139-1174.

Wilson H.F. and Xenopoulos M.A., 2009. Effects of agricultural land use on the composition of fluvial dissolved organic matter. Nature Geoscience 2, 37-41.

Wortmann C.S. and D.T. Walters, 2006 Phosphorus runoff during four years following composted manure application. Journal of Environmental Quality 35, 651-657.

Zhou X., M. Al-Kaisi and M.J. Helmers, 2009. Cost effectiveness of conservation practices in controlling water erosion in Iowa. Soil and Tillage Research 106, 71-78.

10 Change in land use and land use patterns

C. Gascuel-Odoux, B. Christen, J.-M. Dorioz, P. Moreau, L. Ruiz, D. Trévisan and F. Vertes

10.1 Introduction

The European land use and land cover have highly evolved during the last decades, mainly due to urbanisation and specialisation of the agricultural areas. Corine Land Cover programs are powerful tools to evaluate the nature and the intensity of these changes. From 1990 to 2000 each European country have their land use changed (up to more than 10% of the surface area in Portugal; Figure 10.1). In most European countries the area of agricultural land decreased in favour of natural areas, but the variability is high between countries. There is an influence of climatic region. For northern and temperate countries, woodlands and natural areas have been extended from agricultural areas and wetlands re-created. For Mediterranean countries, the agricultural withdrawal is counterbalanced by a conversion from natural dry land to agricultural lands due to the better possibilities to make use of irrigation. Urbanisation and human density have increased roads and trails networks, splitting up agricultural areas, mixing closely fields and development areas. Agricultural areas have changed to artificial areas. Specialisation and high productivity of agricultural activities has driven to field enlargement, hedgerow removal and agricultural tile drain and ditches network development.

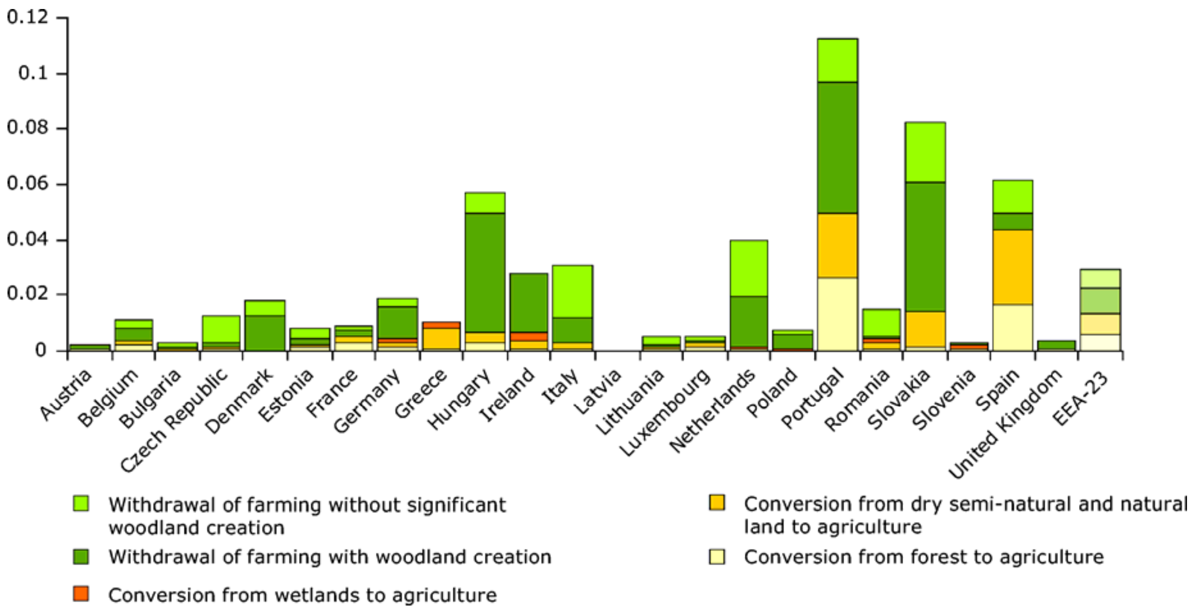

Figure 10.1

Land use changes in the period 1990 to 2000 expressed as fraction of the land area of the countries (Source EEA: http://www.eea.europa.eu/)

Catchment agronomy as emerging areas
Due to these driving forces, changes have affected agricultural and rural landscapes which are made up of various elements related to environmental and human factors, structured by a mosaic of farmers' fields and semi-natural areas. These trends modify the way nutrients are applied, transferred, transformed and stored, and finally exported and their impact in receiving aquatic ecosystem. Diffuse pollution due to agriculture is taking place in a complex and hierarchical matrix of fields and natural areas interacting with a hydrographical network. The agricultural landscape can be understood as a spatially heterogeneous mosaic (Forman and Godron, 1981) reflecting interrelations between anthropogenic processes (farm management, rural development, land conversion) and natural processes. A chain of inputs, management, storages, transfer and transformation of nutrients acts at short distance (1 to 10 km), in and between, environmental systems (soil, groundwater, atmosphere and ecosystem) and technical systems (fields, farms, groups of farms). Spatial interactions occur between natural processes such as N deposition, N and P soil fixation and transfer, denitrification and P release in some wetlands, and technical processes such as fertilizer and manure applications, crop succession, soil tillage (See Cellier et al. in press, for a review on nitrogen; Wang et al., 2004, for a review on phosphorus). Because of these interrelations between environmental and technical processes, the concept of landscape presents a practical dimension for planning, management, conservation and development of territories (Deffontaines et al., 1994, 1995; Rapport et al., 1998). Finally, the mitigation options at landscape scale can contribute to develop a new field of knowledge and engineering, the "catchment agronomy", coupling farm and crop management within the catchment area, as a functional level of the landscape concepts in hydrology. These options can be applied to medium size catchment (10-50 km^2) where fields and interfaces can be clearly identified.

The catchment buffer capacity
Some areas in agricultural landscapes can be considered as nutrient sources and others as sinks or bio-reactors. Interfaces between fields and streams can soften or short-cut nutrient fluxes. As an example, hotspot of N deposition near animal housing, or P accumulation in soil due to manure spreading or intensive grazing close to the farmstead, can act as source. Infiltration of overland flow at the foot of a hedgerow is a sink for water and solutes. A wetland where denitrification and P release can occur can be considered as a biogeochemical reactor. These spatial features and their function vary in time due to landscape characteristics (extension of saturated areas, agricultural practices), surface characteristics (soil surface conditions) and meteorological conditions. The spatial patterns of fields within the landscape, including grassland and arable lands vary in time according to the management of cropping systems and the farm-scale allocation of land use. This dynamics acts on a large range of temporal scales, events, seasons, duration of crop succession. These spatio-temporal dynamics and the involved patterns can contribute to modulate the nutrient export by modifying what Viaud et al. (2004) have called the buffer capacity of the catchment, i.e. the ratio between nutrient output and input, by trapping them by different processes along the different flow pathways.

The basis of mitigations options at landscape scale.
Such options aim at improving such modulation of sink/source. This latter is particularly active in hydrological systems where lateral flows are dominant, i.e. surface flow and subsurface flow hydrological systems. Catchment agronomy consists in decreasing N and P export by: 1) decreasing the nutrient surplus on catchment; 2) increasing the heterogeneity of the landscape, i.e. to optimize the relative position of source and sink and attenuate the effect of nutrient hotspots especially during critical periods; 3) introducing buffer areas as closer as possible to the sources (P), but also to vulnerable areas which can be streams, lake or wetlands; 4) managing sustainably stocks created by mitigation options to avoid release of P from buffer areas by example, side effect or interactions between nutrients (N, P, DOC).

In this chapter, we focus on land use and cover, while the mitigation on interfaces will be presented and discussed in Chapter 11. Land use and cover are related to the ratio of agricultural lands versus woodland /urban area, as well as the ratio of arable land versus grassland, the cropping systems of crop land, the stocking rates and the duration of grassland in lay-arable rotations. The spatial patterns of fields will regulate the nutrient balance and export at catchment scale. Figure 10.2 shows that the N loss of catchments depends on the part of agricultural areas within the catchment. When this part is low, under 50%, the N loss remains low. Above this 50% the N-loss can increase remarkably. Kronvang et al. (2005) reported a gradual increase of the total-N, NO_3 and NH_4 concentration in relation to proportion of agricultural land and for phosphorus above 60% agricultural land the concentration did not increase anymore (Figure 10.3)

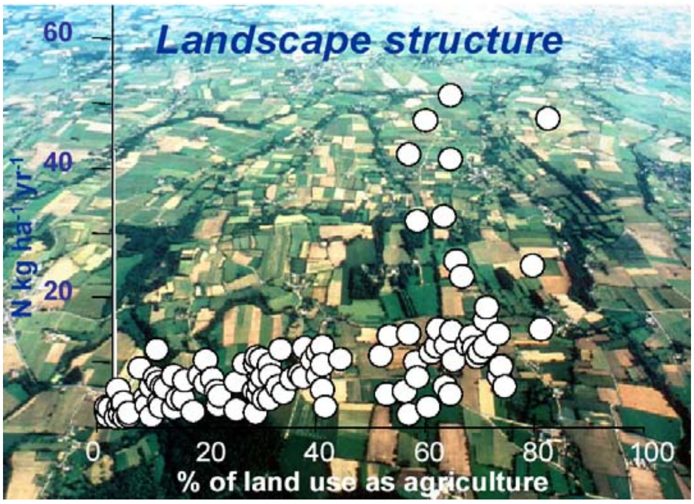

Figure.10.2

N loads (kg.ha-1.year-1) for different catchments, versus their land use as agriculture (%) (oral communication, EGU 2008). Adapted from Strayer and Pinay (2003), Burt and Pinay (2007) and Lefebvre et al. (2007)

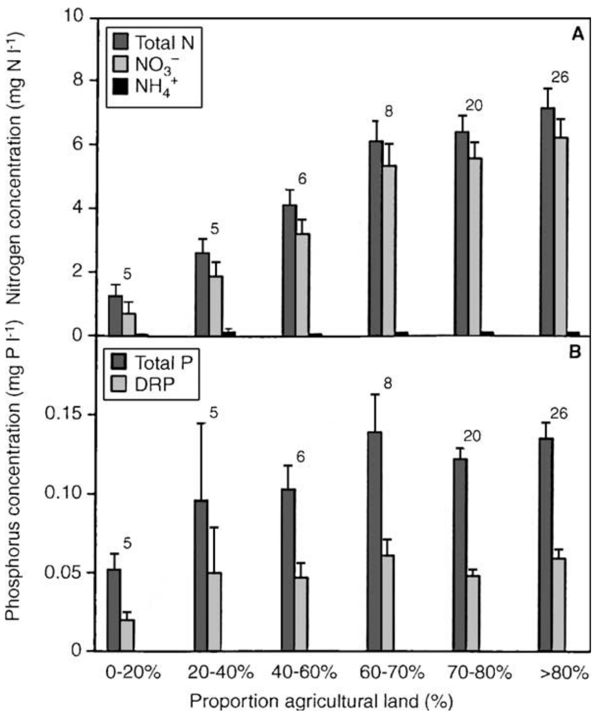

Figure 10.3

Average annual flow-weighted concentrations of total N, nitrate and ammonium (A) and total phosphorus and dissolved reactive phosphorus (B) in streams draining catchments with an increasing proportion of agricultural land. Number of locations is written above the bars in the figure (From Kronvang et al., 2005)

The effect of land use has been analysed at field scale in the chapter on crop management; here it is analysed at catchment level. Despite a strong link between the two chapters, this chapter will show that the catchment cannot be considered as a simple sum of fields because of agricultural and environmental features, constraints and interrelations in space and time.

Technical and environmental constraints and their effect on landscape use and organization
As an example the effect of farm constraints on land use location is discussed. Thenail (2002) and Thenail et al. (2009), studying the interactions between farm functioning and landscape pattern in north-eastern Brittany (western France), has emphasized the strong spatial pattern of the crop mosaic of dairy farms, which is to a large extent determined by distance to farmstead (Figure 10.4). Land use is organized into approximate concentric circles around the farmstead: pastures grazed by dairy cows are located as close to the farmstead as possible, because dairy cows move daily from the farmstead into the fields. Dairy cows density near the farmstead is reinforced when milking robots are used. A second circle consists of fields used for cash crops and forage. The outer circle consists of permanent grasslands grazed by heifers, extensive lands or wood-lands, which require little management. This applies to many locations in north-western Europe. By contrast, crop allocation in crop farming systems or intensive breeding farming systems are expected to be less controlled by the distance to farmstead. More generally, the degree of spatial organisation varies according to the farming systems. Where grasslands are mainly permanent meadows, the catchment organization depends on forage systems that are conceived by farmers to provide various kinds of forage quality to livestock, depending of the objectives and the calendar of the dairy production. Several meadows and grass manage-ment practices (e.g. fertilization, periods of hay cutting, number of successive grazing) are defined through the space, with consolidate assemblage of various forage functions, such as examples 'hay in the barn' where the needs for forage quantity prevail over that of forage quality, 'hay for the milk' where instead high forage value is searched, 'extensive grazing areas' for heifers (Fleury et al., 2001).

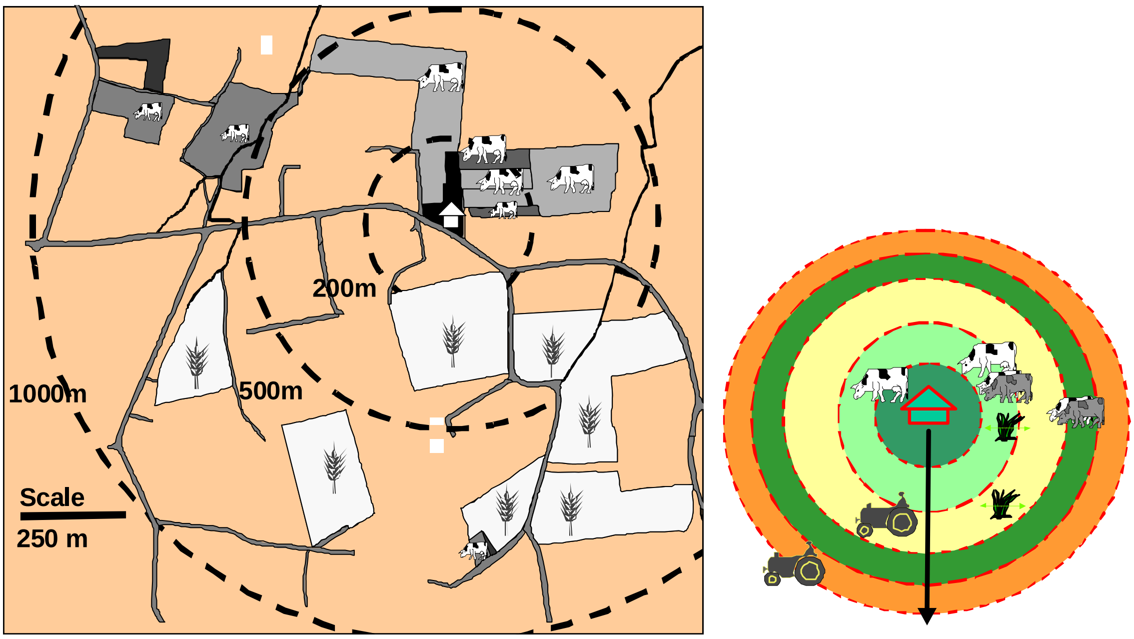

Figure 10.4
Theoretical organization of the crop mosaic according to the distance to the farmstead in dairy farms in Brittany (western France)(left) and to the constraints (right) (adapted from Thenail, 2002)

Similarly, environmental constraints can act on land use locations. Crops requiring a lot of water for growing, such as maize, are often preferentially located in the bottom domain. Reversely crop harvested late in autumn and catch crop sowing cannot always be easily achieved in these domains due to a lower soil trafficability. Cerdan et al. (2002) have shown that a little portion of crop-fields contribute effectively to loads of suspended matter and nutrients into the stream, according to the flow pathways (Fig. 10.5). Water and pollutants that flow at the soil surface, generated by low infiltration rates in soil due to soil surface conditions (roughness, soil cover by plant and crusting) are generally re-infiltrated in the bottom domain, so that the heterogeneity of the soil surface conditions determines the runoff at the outlet of the catchment. This catchment scale mitigation can be applied to surface flow dominant hydrological systems, where the cropping systems vary in space and determine a various spatial organisation of soil surface conditions (Papy and Boiffin, 1988).

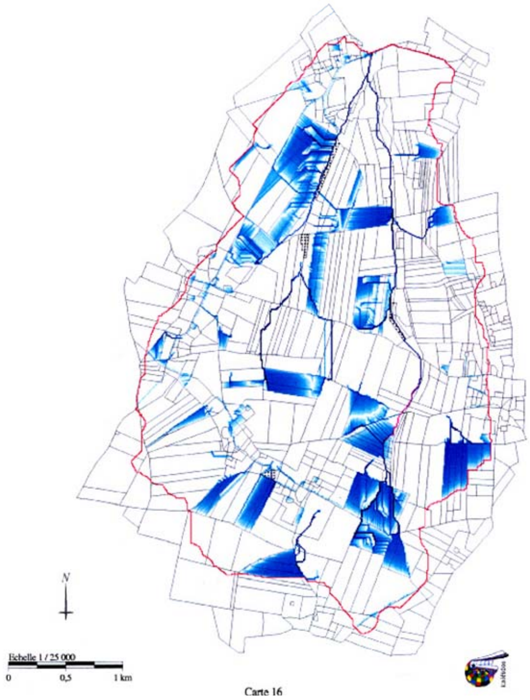

Figure 10.5
The surface runoff of a low number of plots contribute to the stream flow because of a re-infiltration of the surface runoff along the flow pathways due to the heterogeneity of the soil surface conditions and the infiltration rate within the catchment (Adapted from Cerdan et al., 2002)

10.2 Strategies for mitigation

Landscape mitigation measures can be divided into three main strategies (Table 10.1):

1. 1. Change agricultural land use patterns by relocating land use or crop**,** to increase the buffer capacity of the catchment.
2. 2. Change cropping systems and agricultural practices**,** to develop agroforestry and improve nutrient storage in soils, and to avoid hot spots of nutriment emission.
3. 3. Change agricultural land use by 'extensification' options on land use or crop systems, to decrease nutrient input or increase nutrient storage in soils, considering for P a sustainable storage, mainly by incorporating nutrients in soil organic matter or increasing duration of plant cover.
4. 4. Land use changes form agriculture to non-agriculture use by afforestation of agricultural lands, to decrease the nutrient budget of the catchment, simply by diluting the nutrient budget of the agricultural lands.

Table 10.1
Landscape mitigation strategies: changes, measures and processes

Strategy	Change	Measures	Processes
Change agricultural land use patterns	Catchment buffer capacity: - location of sinks and sources - flow pathway and connectivity from field to field	Alternate grass lands and arable lands in the landscape, particularly alternate crop with different seasonality.	Nutrient trapped along the flow pathways (P)
		Locate crops with a high nutrient crop uptake in bottom lands Introduce catch crops, energy crops without nutrient input Avoid certain crops in hilly areas	Nutrient uptake by crops located down slope (N) Increase N export by plants (N, P) Decrease surface runoff (P)
Change locally agricultural practices	Avoid hot spots responsible for nutrient emission	Change locally practices: soil tillage, seed-bed preparation (direction, …), ploughing near ditches, soil compaction by wheels tracks on plots close to the stream	Disconnect critical sources areas from receiving waters
Change land use: agricultural extensification and agro-forestry development	Nutrient budget over the agricultural areas of the catchment	Introduce crop rotation with more years with grass (see crop management)	Decrease of nutrient budget by nutrient input decreasing and nutrient storage in soils increasing
		Develop mix cropping system: mixing perennial and annual crops	Plant cover
		Set aside for a couple of years	Nutrient storage in soils increasing
Land use change from agriculture to non-agriculture use. Diversification, protecting sensitive areas	Nutrient budget over the catchment	Afforestation of agricultural land New protected wetlands Set aside definitively	Dilution of nutrient budget of agricultural areas

10.2.1 Agricultural land use patterns

Change agricultural land use patterns by relocating land use or crop can increase the buffer capacity of the catchment by reasoning the location of sinks and sources and by decreasing the flow connectivity. This option can act on the surface flow as well as the subsurface flow. It cannot be considered as a curative solution, as explained by Vertès et al. (2009). It implies that fields are fertilized and managed according to good agricultural practices. Two processes are discussed in the literature.

Firstly, water, sediments and nutrients in surface flow, produced upslope, can be infiltrated, deposited or trapped downhill on a field which presents a higher infiltration rate. The flow pathways, the soil and hydrological conditions and the properties of the transported elements have to be taken into account (Cerdan et al., 2002; Aurousseau et al., 2009). This option is particularly efficient when lands with low and high infiltration rates alternate along the hill (Souchere et al., 2005). The generated surface runoff can infiltrate all along the hill before reaching the stream and the set of the plots acts as successive buffers along the slope. This option is efficient if down the hill areas are unsaturated, and therefore, more efficient in infiltration during dry than wet season. It may concern short periods when different infiltration rates along the slope can be observed, determined by soil surface conditions (vegetation cover, roughness, soil structure) and thus, by the interaction between agricultural operations and climate. This is important to identify these short critical periods to implement such options.

Secondly, nutrient can be taken up by the crops from shallow groundwater with high concentrations of nitrates (Beaujouan et al., 2001). Nitrate draining in shallow groundwater and migrating by subsurface flow can be taken up by the crops growing downslope. If the groundwater is close enough to the surface it can interact with the root zone of the crop. The mitigation option consist in cultivating down the hill crops which can uptake a higher or delayed (winter / spring) nitrate amount compared to those located upslope (Beaujouan et al., 2001). It is an issue only where a significant lateral flow through the soil is present with considerable N transport. Generally, in the region where excess of manure exist, the fertilisation is already high in the whole area; as a consequence the uptake will be low or nil downslopes.

At farm level, new decision rules for the organization of the allocation of the crops on the farm are needed, since the location of the crops on the farm is generally decided according technical constraints (subset of plots, crop succession, animal and machinery access etc.) and catchment features such as soil and slope position. At the landscape scale, a collective planning of the crops within the catchment has to be implemented by the farmers of the catchment, particularly between neighbours (Martin et al., 2004; Jouanon et al., 2006). Moreover, field access and land use and cover have to be revised to optimize such collective planning.

Land use re-location is only applicable on moderate and relatively uniform slope conditions, with surface sheet flow and low flow velocity for surface flow as well as subsurface flow. The measure is relevant for all areas affected by surface runoff and erosion, probably under all climatic conditions. This effectiveness may only concern short periods along the year, for nutrients concerned by surface flow (the infiltration rate highly varies in time), as well for nutrients concerned by subsurface flow (the interaction between the root zone and the groundwater is seasonal). The effect on particulate P starts as soon as it is established.

The effectiveness of such options is mainly evaluated by hydrological modelling including a detailed description and functioning of the agricultural landscape (Jetten et al., 1996; Cerdan et al., 2002; Aurousseau et al., 2009).Catchment experiments can not be easily used to evaluate the effectiveness of such options while testing the effect of crop locations would take many years. The modelling shows a high effectiveness of such options on surface runoff and erosion, thus on particulate P, similarly to vegetative buffer strip, and a low effectiveness on nitrate budget.

A relevant crop allocation can present some positive side effects. It increases the slope stability by armouring the soil with plant roots which reduce the erosion and P loss. The filters can also prevent the damages due to the transfer of sediments to roads or urban areas. Benefits include not only water quality improvement but also aesthetic values and ecological benefits. The landscape heterogeneity can contribute to increase the biodiversity.

10.2.2 Change locally agricultural practices

According to Heathwaite et al. (2005) any zone can be classified as a critical source area (CSA) if 'there is a significant source of nutrient input and flow from that zone is in direct connection with the receiving waters'. A set of mitigation options aiming at avoiding hot-spots and critical sources areas to reduce P losses and organic carbon have been identified. They correspond to measures that maintain locally a high soil infiltration capacity, avoiding inappropriate practices of soil tillage, crop harvesting or management of inter-crop (Vansteelant et al., 1997; Dorioz and Trevisan, 2008). Critical situations have been identified (Wang et al., 2004; Dorioz and Trevisan, 2008). As an example, surface runoff are frequent on wheel tracks or compacted soil surface which directly connect crop lands to stream networks (Cerdan et al., 2002; Heathwaite et al., 2005). All these measures have been presented in the crop and soil management chapters. They have particularly to be implemented in areas adjacent to the stream.

In addition to mitigation options that aim to disconnecting livestock on grassland from river networks, catchment agronomists are also concerned by the optimization of the spatial distribution of agricultural functions and practices within the catchment. A better organization of herd displacements can reduce a sporadic contamination of superficial waters (Wilcock et al., 1999) which depend strongly on the forage systems. Non-permanent defined grazing or hay-meadows areas are positive. They allow the farmers to use mobile-milking machineries, mobile infrastructure management (trails that connect farmsteads to peripheral forage resources; bridges and riverbank managements, watering system implantation) that can efficiently erase hot spots of grassland areas (Meals and Braun, 2006).

Several methods and models can help to delimitate critical sources areas, by coupling geographical information systems and hydrological modeling to locate within the catchment the circumstances and the frequency of events where high nutrient inputs coincide with surface runoff (Heathwaite et al., 2005; Gascuel-Odoux et al., 2009; Trevisan et al., 2010). Such modeling deals with: 1) the analyze of agricultural practices that generate the spatial distribution of pollutants or nutrients during the agricultural schedule; 2) the determination of areas giving the probabilities of runoff generation; 3) the routing of generated runoff along slopes and flow paths, and its consequences in terms of connection to river system or dilution by upslope contributions. Such developments are sustained by strong transdisciplinary collaborations between agronomists and hydrologists.

10.2.3 Agricultural land use: agro-forestry and/or extensification

Changes of farming system start from optimisation of current practices to a complete change of the production system. Change of crop, rotation or fertilization strategy could be presented as a single measure or as a part of a systemic and integrative modification of production system.

Three levels of changes can be considered:
1. arable land can be converted into 'extensive' grassland or more drastically in hay fields or even unfertilized cut swards, to modify drastically the nutrient budget of the agricultural lands; such drastic option is often used in stream or subsurface water collection areas used to produce drinkable water;
2. mix cropping systems, including agro-forestry mixing annual and perennial crops can be developed;
3. more moderate changes can also be considered in this option, such as conversion into grazed grassland, a longer duration of the crop rotation, including low surpluses on nutrient balances for crops and grasslands, a longer duration of catch crops, a better adjustment between nitrogen mineralisation and plant uptake which are described in the crop management chapter.

All combination can be selected but the farmers have to be strongly associated to the selection of the mitigation measures due to the high constraints they can induce on the farm systems (Deffontaines et al., 2004).

Land use extensification aims to modify the nutrient budget, inputs and land cover on agricultural and catchment areas. It may correspond to a slight decrease of productivity of crops, or to an extension of grasslands, which are often considered by environmentalists as water quality friendly (Vertès et al., 2009). This ability is attributed to i) their permanent soil cover combined with a well developed root system which both improves soil structure and infiltration of water, and acts as infiltration areas, ii) their moderate nutrient level and free pesticide management which enable them to dilute and regulate diffuse pollution, and iii) their location within the catchment, and their margins which often locate them close to the streams, surrounded by hedgerows, and banks. Grasslands are in fact quite diverse and complex: intensively grazed grasslands can impair water quality (nitrate, phosphate, and overall faecal contaminants); bad management in riparian areas, such as animals drinking direct in stream water, can occur in all types of grazing systems and damage aquatic ecosystems (e.g. sediments). Therefore a real improvement can only come from well managed grasslands, including their place in ley-arable rotations (Vertès et al., 2007). Extension of grasslands means a lot of changes in dairy farming systems, particularly in socio economic conditions.

Agro-forestry mixing annual and perennial crops can improve nutrient management by 1) a higher immobilisation of N in a higher stable organic matter than in annual crops and 2) a continuous uptake of the nutrients by the perennial crops, avoiding high N residue and leaching. Similarly to extension of grasslands, agroforestry means a lot of changes in farming systems, particularly in socio economic conditions.

The effectiveness of such options is mainly evaluated by agro-hydrological modelling. While the models can be easily validated at plot scale, integrating and validating them at catchment scale is not often possible. The efficiency of agricultural land use extensification has been tested by modelling in Denmark (Dalgaard et al., 2002; Dalgaard, 2009), in France (Durand et al., 2004). Kersebaum et al. (2003) have tested a passage from observed intensive system to extensive system changing crop succession. Vinten and Dunn (2001) have tested switching from vegetable rotations to vegetable and cereal rotations or to cereal and grass rotations. Volk et al. (2009) tested a combination of reduction of livestock units per hectare on pasture, a reduction of the mineral and organic fertilizer rates, an application of conservational and eco-farming practices. But few models couples farm and environmental constraints. Such options were also experienced successfully by practical application (Vittel mineral water protection area; Deffontaines et al., 1994; Gras and Benoit, 1998). In this last case, the drastic changes have been accepted widely because it was also funded by Vittel. That underlines the role of the socio-economic acceptability.

The response time of a fertilisation measure downstreams in a catchment can be long due to water and solute transit time in groundwater and organic matter stabilisation. The effectiveness depends on the physiographic settings of the catchment. Payraudeau et al. (2007) have shown that the effect of a lower level of fertilisation really depends on the surface area of the wetlands. Therefore the effect of such options can locally start as soon as they are established, but really observed at catchment scale after few years or decades, depending of soil and hydrological systems and of the range of decrease in the nutrient budget. Land use extensification is always applicable in all physiographic settings. The socio-economic dimension is the major one regarding applicability.

10.2.4 Land use change from agriculture to non-agriculture use

Afforestation is a drastic measure which therefore can induce a quick and highly visible response of the nutrient export of the catchment. The nutrient budget of woodlands or more generally, non agricultural lands, is much smaller (few $kg.ha^{-1}$) than the agricultural ones (few tenth of $kg.ha^{-1}$) for N as well as P. The cost of such measure, i.e., by buying the land, cannot be considered as high as expected compared to the costs of a more sustainable agriculture. The main difficulties come from societal issues since implementing such options require land exchanging or buying, in order to solve property problems which necessarily takes time.

Therefore, such options have to be dedicated to highly vulnerable areas. Some coastal bays such as present in French Brittany (invasive green macro algae) as well as in the Baltic Sea area (phyto plankton blooms) or catchment providing drinkable water could be concerned by such high level measures, because of the low target in N and P in these areas.

The effectiveness of such option depends on the interaction between climate, current storage of nutrients in soil and groundwater. As previously mentioned the effect of such measure on a given catchment can be mainly evaluated by hydrological modelling including a detailed description and functioning of the agricultural landscape.

Decision support tools can facilitate the identification of the most appropriated solutions for land and watershed managers, farmers, professional and technical organizations, in an operational way (De Vries et al., 2005). As an example, 'Territ'eau' is a holistic tool to improve the rural territory management from a water quality perspective, developing education and training, operational tools in order to help stakeholders in decision making (Gascuel-Odoux et al., 2009).

In conclusion

1. Mitigation at landscape level is based on a new field of knowledge and engineering, the 'catchment agronomy', the landscape patterns resulting from interrelations between environmental and technical processes.
2. The part of agricultural land in the landscape is a key factor of nutrient (N) loads in surface water. Under 50% of agricultural land, the nitrate loads and concentrations remain generally low.
3. The landscape heterogeneity attenuates the surpluses on nutrient balance (N) and export (P) at catchment scale by optimizing the probability of sources to be buffered by sinks, attenuating the effect of nutrient hotspots, especially during critical periods.
4. Changes in agricultural land use patterns by relocating land use or crop is effective on erosion and P emission in water bodies. It requires collective planning at catchment level.
5. Changes in agricultural land use by ‘extensification’ on land cover or crop systems, or by afforestation of agricultural land is needed for a long term effect, and thus, is a condition for sustainability, but present a socio-economic dimension regarding applicability.

List of factsheets

Gascuel-Odoux, C., D. Trevisan, Y. Le Bissonnais, 2009. Land use re-location. [FS]

References

Aurousseau, P., Suividant, H., Tortrat, F., Gascuel-Odoux, C. and Cordier, M.O. 2009. A plot drainage network as a conceptual tool for the spatialisation of surface flow pathways for agricultural catchments. Computer and Geosciences, 35, 276–288.

Beaujouan, V., Durand, P. and Ruiz, L. 2001. Modelling the effect of the spatial distribution of agricultural practices on nitrogen fluxes in rural catchments. Ecological Modelling, 137(1): 93-105.

Boiffin, J., Eimberk, M. and Papy, F. 1988. Influence des systèmes de culture sur les risques d’érosion par ruissellement concentré. I. Analyse des conditions de déclenchement de l’érosion. Agronomie 8, 663-673.

Burt, T.P. and Pinay, G. 2005. Linking hydrology and biogeochemistry in complex landscapes. Progress in Physical Geography 29, 297–316.

Cellier, P., Bleeker, A., Breuer, L., Dalgaard, T., Dragosits, U., Drouet, J.L., Durand, P., Duretz, S., Hutchings, N., Kros, H., Loubet, B., Oenema, O., Olesen, J., Merot, P., Theoblad, M., Viaud, V., de Vries, W. and Sutton, M., in press. Dispersion and fate of nitrogen in rural landscapes. Cambridge, UK: Cambridge University Press, Chapter 7.

Cerdan, O., Souchère, V., Lecomte, V., Couturier, A. and Le Bissonnais, Y. 2002. Incorporating soil surface crusting processes in an expert-based runoff model: Sealing and Transfer by Runoff and Erosion related to Agricultural Management. CATENA, 46(2-3): 189-205.

Dalgaard, T., Heidman, T. and Mogensen, L. 2002. Potential N-losses in three scenarios for conversion to organic farming in a local area of Denmark. Eur. J. Agron. 16, 207-221.

Dalgaard, T. 2009. Landscape Agroecology: managing interactions between agriculture, nature and socio-economy. In: Nagel et al. (eds.) Proceedings from the Multi-level processes of integration and disintegration. Green Week Scientific Conference. International Congress Centre, Berlin, Jan 14-15 2009. MARGRAF publishers.

Deffontaines, J.-P., Brossier, J., Benoît, M., Chia, E., Gras, F. and Roux, M. 1994. Agricultural practices and water quality. A research development project. Systems studies in agriculture and rural development. In: Brossier, J., de Bonneval, Laurence, Landais, E. (eds) Coll. Science Update: pp 31-61.

Deffontaines, J.P., Thenail, C. and Baudry, J. 1995. Agricultural systems and landscape patterns – how can we build the relationship? Landscape and Urban Planning 31, 3-10.

De Vries, W., Kros, J. and Velthof, G. 2005. Integrated evaluation of agricultural management on environmental quality with a decision support system. In: Zhu, Z., K. Minami & G. Xing (Eds). 3rd International Nitrogen Conference, October 12-16, 2004. Nanjing. China, Science Press, pp. 859-870

Dorioz, J.M. and Trevisan, D. 2008. Le transfert du Phosphore dans les bassins agricoles : ordres de grandeur, mécanismes, maîtrise. Ingénieries, 27-47.

Durand, P. 2004. Simulating nitrogen budgets in complex farming systems using INCA: calibration and scenario analyses for the Kervidy catchment (W. France). Hydrology and Earth System Sciences 8, 793-802.

Fleury, P., Dubeuf, B., Jeannin, B. and Bonsaquet, R. 2001. Fonctionnement fourrager des exploitations des alpes du nord : conseil technique agricole et environemental. In Les Prairies de fauches et les pâtures des alpes du nord, GIS alpes du Nord, Chambéry.

Forman, R.T.T. 1997. Land mosaics. The ecology of landscapes and regions Cambridge University Press. Cambridge, UK.

Forman, R.T.T. and Godron, M. 1981. Patches and structural components for a landscape ecology. Bioscience 31, 733-740.

Gascuel-Odoux, C., Massa, F., Durand, P., Merot, P., Baudry, J., Thenail, C. and Troccaz, O. 2009. A framework and tools for agricultural landscape assessment facing to the preservation of the water quality. Environmental Assessment, 43, 921-935.

Gras F. and Benoît M. 1998. "Influence des systèmes de culture et des pratiques agricoles sur la qualité de l'eau minérale de Vittel. Le programme de recherches AGREV." Comptes Rendus de l'Académie d'Agriculture de France (FRA), 5, 166-168.

Heathwaite, A., Quinn, P. and Hewett, C., 2005. Modelling and managing critical source areas of diffuse pollution from agricultural land using flow connectivity simulation. Journal of Hydrology, 304, 446-461.

Jetten, V.G., Boiffin, J. and De Roo, A.P.J. 1996. Defining monitoring strategies for runoff and erosion studies in agricultural catchments: a simulation approach. Eur. J. Soil Sci. 47, 579–592.

Jouanon, A., Souchere, V., Martin, P. and Papy, F. 2006. Reducing runoff by managing crop location at catchment level, considering agronomic constraints at farm level. Land degradation & Development 17, 467-478.

Kersebaum, K.C., Steidl, J., Bauer, O. and Piorr, H.P. 2003. Modelling scenarios to assess the effects of different agricultural management and land use options to reduce diffuse nitrogen pollution into the river Elbe. Physics and Chemistry of the Earth 28, 537-545.

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

Martin, P., Joannon, A., Souchère, V., and Papy, F. 2004. Management of soil surface characteristics for soil and water conservation: the case of a silty loamy region (Pays de Caux, France). Earth Surf. Process. Landforms 29, 1105–1115.

Meals, D.W. and Braun, D.C. 2006. Demonstration of methods to reduce E. coli runoff from dairy manure Apllication sites. Journal of Environmental quality, 35, 1088-1100.

Lefebvre, S., Clement, J.C., Pinay, G., Thenail, C., Durand, P. and Marmonnier, P. 2007. N-15-nitrate signature in low-order streams: effects of land cover and agricultural practices. Ecological Applications 17, 2333-2346.

Payraudeau, S., van der Werf, H.M.G. and Vertès, F. 2007. Analysis of the uncertainty associated with the estimation of nitrogen losses from farming systems. Agricultural Systems 94, 416-430.

Papy, F. and Boiffin, J. 1988. Influence des systèmes de culture sur les risques d'érosion par ruissellement concentré. II. Évasion des possibilités de maîtrise du phénomène dans les exploitations agricoles. Agronomie 9, 745-756.

Rapport, D.J., Gaudet, C., Karr, J.R., Baron, J.S., Bohlen, C., Jackson, W., Jones, B., Naiman, R.J., Norton, B. and Pollock, M.M. 1998. Evaluating landscape health: integrating societal goals and biophysical process. J. Environ. Manage. 53, 1-15.

Souchere, V., Cerdan, O., Dubreuil, N., Le Bissonnais, Y. and King, C. 2005. Modelling the impact of agri-environmental scenarios on runoff in a cultivated catchment (Normandy, France). Catena 61, 229-240.

Strayer, D.L., Beighley, R.E., Thompson, L.C., Brooks, S., Nilsson, C., Pinay, G. and Nairman, R.J. 2003. Effects of land cover on stream ecosystems: Roles of empirical models and scaling issues Ecosystems 6, 407-423.

Thenail, C., 2002. Relationships between farm characteristics and the variation of the density of hedgerows at the level of a micro-region of bocage landscape. Study case in Brittany, France. Agricultural Systems 71, 207-230.

Thenail, C., Jouannon, A., Capitaine, M., Souchère, V., Mignolet, C., Schermann, N., Di Pietro, F., Pons, Y., Gaucherel, C., Viaud, V and Baudry, J. 2009. The contribution of crop-rotation organization in farms to crop-mosaic patterning at local landscape scales. Agriculture, Ecosystems and Environment 131, 207-219.

Trevisan, D., Cena, F. and Dorioz, J.M. 2005. Incidence d'Aménagements agro-environnementaux sur le fonctionnement hydrologique et la qualité de l'eau du bassin versant du Mercube. Gis Alpes du Nord-Chambre d'Agriculture de Haute Savoie, 7p.

Trevisan D, Dorioz J.M., Poulenard J., Quetin P., Prigent Combaret C. and Merot P. 2010. Mapping of critical source areas for diffuse fecal bacterial pollution in extensively grazed watersheds. Water Res. 44, 3847-3860.

Vansteelant, J., Trévisan, D., Perron, L., Dorioz, J. & Roybin, D. 1997. Conditions d'apparition du ruissellement dans les cultures annuelles de la région lémanique. relation avec le fonctionnement des exploitations agricoles. Agronomie 17, 68-82.

Vertès F., Hatch D., Velthof G., Taube F., Laurent F., Loiseau P. and Recous S. 2007. Short-term and cumulative effects of grassland cultivation on nitrogen and carbon cycling in ley-arable rotations. Invited paper In 'Permanent and temporary grassland: Plant, Environment and Economy', De Vliegler A. & Carlier L., (eds.), 14th symposium of the European Grassland federation, Gent (B), 3-5 sept; 2007, Grassland Science in Europe, 12, 227-246.

Vertès, F., Trévisan, D, Gascuel-Odoux, C. and Dorioz, J.M. 2009. Developing tools to improve grassland management in two French grassland systems (intensive and extensive), to comply with Water Framework Directive, Tearmann, Irish J. Agri-Environmental Res. 7, 161-174.

Viaud, V., Merot, P. and Baudry, J. 2004. Hydrochemical buffer assessment in agricultural landscapes: from local to catchment scale. Environ. Manage. 34, 559-573.

Vinten, A.J.A. and Dunn, S.M. 2001. Assessing the effects of land use on temporal change in well water quality in a designated nitrate vulnerable zone. Sci. Total Environ. 265, 253-268.

Volk, M., Liersch, S. and Schmidt, G., 2009. Towards the implementation of the European Water Framework Directive? Lessons learned from water quality simulations in an agricultural watershed. Land Use and Policy 26, 580-588.

Wang, D., Dorioz, JM., Trevisan, D., Braun, D., Windhausen, L. and Vansteelant, J. 2004. Using a landscape approach to interpret diffuse phosphorus pollution and assist with water quality management in the basins of Lake Champlain (Vermont) and Lac Léman (France). In Lake Champlain: Partnership and research in the New Millenium, T. Manley et al., Ed, Kluwer Academic/Plenum Publishers, 159-189.

Wilcock, R.J., Nagels, J.W., Rodda, H.J.E, O'Connor, M.B., Thorrold, B.S. and Barnett, J.W., 1999. Water quality of a lowland stream in a New Zealand dairy farming catchment New Zealand Journal of Marine and Freshwater Research 33, 683-696.

11 Landscape management

J.-M. Dorioz, C. Gascuel-Odoux, M. Stutter, P. Durand and P. Merot

11.1 Introduction

In this chapter we introduce the importance and design of 'landscape buffers' as a key aspect of field to catchment scale diffuse pollution management. Rural landscapes are heterogeneous mosaics, consisting of a complex matrix of agricultural fields and non-agricultural lands such as forests or wetlands, with some patches corresponding to farm infrastructure, all these components being interconnected by linear structures such as roads, field margins, riparian strips and hydrographic networks. The nutrient movement through the landscape to the outlet of the corresponding catchment, results in a set of interacting and cascading processes whose workings lead to storage (mainly in soils), emission from sources, hydrologic transport, retention (storage in sinks, and transformations) and finally export to receiving and sensitive water bodies (Wang et al., 2004).

Farm fields and some farm infrastructures usually represent the main sources of diffuse pollutants for water bodies. The mitigation options regarding land use changes (often at farm level) have been presented in Chapter 10. Other, interwoven but less agriculturally-managed landscape entities, can be: (i) 'interfaces', meaning structures inserted between a source and a water body and acting either as sinks which buffers the transfer of diffuse pollutants, or as transfer media ensuring hydraulic connection; (ii) neutral components, providing a flow of water, diluting the concentration of pollutants at the outlet of the catchment and sometimes serving to flush sediments during storm flows (Jordan Meille et al., 1998). All these components are distributed in space and time according to natural landscape characteristics, mainly hydrological and morpho-pedological settings, and also agricultural practices, which are driven by farm objectives and constraints. They include remnant fragments of the semi-natural landscape (e.g. established riparian woodlands, wetlands, groves) and altered or man-made management features (e.g. hedges, grassed buffers).

The heterogeneity of the landscape has often been considered as a state favourable for trapping diffuse pollutants emitted by farm fields, leading to the concept of buffers and the definition of the 'buffer capacity of a catchment' (Haycock et al., 1997; Viaud et al., 2004; Wang et al., 2004). The buffer capacity is universally defined as the ratio between nutrient output from the sources (fields) and the amount delivered to the water bodies. It is mainly attributed to a few landscape structures, called 'buffers' which intercept, filter and retain water and associated nutrient flows. Nutrients may be variably stored without transformation or transformed within buffers, according to their nature, speciation, degree of biogeochemical reactivity and interactions with plants or micro-organisms. For N and C, such transformations in landscape buffers (e.g. denitrification in wetlands) may result in losses from the catchment in gaseous forms leading to a potential pollutant swapping (Stevens and Quinton, 2009). For P there is no significant loss but changes to speciation (dissolved-particulate; mineral-organic etc.) can occur, and sometimes facilitate a further release of a part of the stored total-P (Dorioz et al., 2006). Ultimately, interactions of contaminants with these landscape buffers either limit the amount delivered to watercourses, or alter the timing or location of the delivery. The latter can be beneficial if key sensitive periods and places for ecology are protected (e.g. fish spawning periods).

This chapter focuses on the identification, efficiencies and management of such landscape features interacting with contaminant fluxes and consequently having a potential buffer function, both because of their specific structure (vegetation-soil) and of their location at the interface between pollution sources (farm infrastructures or emitting fields) and water bodies. These 'buffer interface' features are diverse and correspond to natural landscape features (wetlands, riparian areas, banks) as well as manmade structures (buffer strips or field margins) or intermediate cases (hedgerows). Their role and efficiency depends on the local factors controlling the retention processes (internal organisation and properties of the buffer), the position within the watershed, and the landscape context, and determines the overall buffer capacity of the catchment.

11.2 The effects of landscape buffers on nutrient transfers

P, N dynamics at the landscape scale
Diffuse nutrient transfers are highly variable in space and time. Different sources of N and P can be associated with varying modes and timings of delivery (how, how much and when). Thus nutrient inputs may be continuous or cumulative (stream connectivity with NO_3 enriched groundwater, annual/seasonal (associated with fertilisation or biomass recycling), or stochastic (from infrequent erosive storm events). That dynamic is governed, in interaction with seasonal precipitation trends, by the nature and location of the sources, the hydrological connectivity between the sources and water bodies, the nature of flow (convergent or sheet surface flow, soil matrix flow, piped/ditched flow) and on the interaction of flow with buffer features.

Buffer interception is an important component of this landscape control on diffuse nutrient transfer. The effect results in four main processes:
- Storage and trapping of water and/or sediment and nutrients within the buffer;
- Water and dissolved nutrient uptake by vegetation and biota;
- Biogeochemical transformation (such as sorption, denitrification);
- Dilution.

Once the buffer has retained contaminants or nutrients, other processes may act to transform them. An important action for this is the increased retention time that allows biogeochemical processing. Individual buffer efficiencies are highly variable and depend on their structures (soils, vegetation) and their locations which control the intensity of these complementary processes.

Nitrogen and phosphorus respond differently to interactions with landscape features. Even if the sources are the same overall, the relationships (the ratios) between annual inputs, soil stocks and fluxes transferred to water, the preferential flow paths for these transfers and the effects of landscape buffers are highly different in intensity and in spatial and temporal distribution. Inspired by the reviews of Haag and Kaupenjohann (2001) and Sharpley et al. (2001). Figure 11.1 compares these key transfer steps of N and P at the landscape level.

As nitrate is very mobile and easily transferred in the landscape, a buffer interface is efficient regarding nitrate only if it combines a low flow velocity, a high biological activity and a source of carbon, as it is the case in wetlands, hedges and ponds. Thus, the elements of long residence time, appropriate soil moisture conditions and stimulation of microbial activity, are present. Riparian areas and wetlands are particularly efficient at nitrogen removal. Topographic indices are good indicators to delineate these types of areas and, therefore, to evaluate their extension and buffering effect at catchment level (Beaujouan et al., 2001). Long term or irreversible removal is achieved by denitrification and immobilisation in perennial biomass or refractory organic matter.

In contrast, phosphorus can be easily trapped, even during short and relatively rapid transfer, via sorption of dissolved-P and physical retention of particulate-P. Consequently, the buffering action is distributed differently in space and time than for nitrogen processing. In the case of particulate-P, and more generally of sediment and particle-bound pollutants such as ammonium and organic nitrogen, two complementary major processes control the working of landscape buffer: infiltration which reduces overland flow (phenomena which also result in the retention of dissolved forms of P) and filtration due to vegetation cover and roughness which reduce flow velocity; these two processes interact to decrease the sediment transport capacity of overland flow and promote particle deposition. The efficiency of the resultant trapping depends on a number of local features (soil permeability, root structure, vegetation structure, and dimensions) and on the position within the watershed (which drives flow dynamics, nature and particle size of incoming material). Dissolved-P is more sensitive to contact time, kinetics and soil chemistry (organic matter, Fe-oxides etc). Finally at the landscape scale, slope and vegetation gradients are good indicators of buffer potentialities for P retention. It should also be remembered that only a small proportion of this retention is really long-lived (decades): P and sediments carrying P are mainly stored in soil surface layers (0-5 cm) and this storage compartment of the buffer may be easily saturated (primarily a physical saturation reducing permeability).

As a consequence of these different behaviours of P and N, the ratio between annual agro-ecological inputs and annual exports to water bodies, is highly different between these nutrients. For nitrate, usual values range from 0.5 to 0.9, whereas for insoluble phosphorus it is much less, often below 0.05. For example, in Brittany, a region of Western France strongly impacted by diffuse pollution, the relative emission is about 0.7 for nitrate and less than 0.01 for total-P (Aurousseau, 2002). As no loss pathway exists for P, this small delivery ratio actually reflects the increasing saturation of soils and buffer zones with reactive-P, and it is unlikely that this can be infinitely sustained (Dorioz et al., 2006; Stutter et al., 2009).

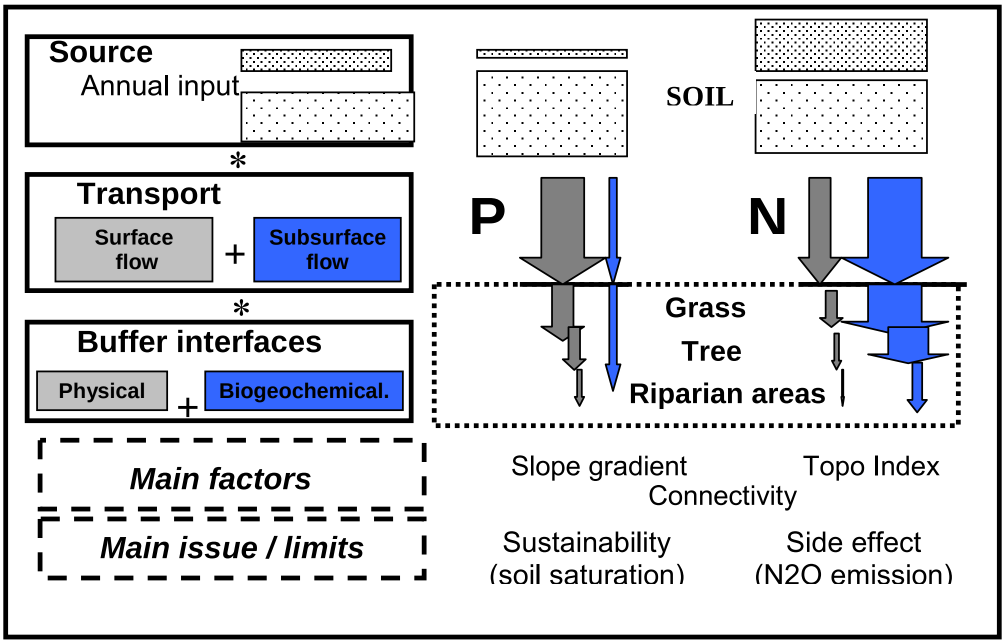

Figure 11.1

Main stages of nutrients dynamics at the landscape scale: comparison between P and N, regarding soil stocks (squares), fluxes according to flow paths (arrows) and effect of main landscape interface on fluxes

Design and assessment of buffers to maximise capacity

Where new buffers are being planned for landscapes they should be correctly designed for site-specific conditions (Correll, 2005). Decisions on design are heavily influenced by local experiences of land managers who can assess some aspects of their functioning visually. This expert guess should be also based upon and supported by proper scientific references (studies into buffer effectiveness, frameworks etc.).

Such references and operational tools are not easy to provide because of the extreme variability of the landscape, regarding morpho-pedological features, local topography, shape, position within the catchment, vegetation structure and soil type. The consequence of this multiplicity of factors and parameters being that each study of an individual buffer tends to be more or less site-specific and scale-specific (Dillaha and Inamdar, 1997). Furthermore, water quality parameters being considered are more or less contaminant specific (e.g. concentrations for NO_3, fluxes of all or of bioavailable forms, for P). The methods and duration of monitoring also tend to be different. All this makes the comparison between study cases difficult and limits our capability to derive from individual experimental studies, not only the buffering capacity of the diverse types of interface within a catchment, but also the overall buffering capacity of a set of interfaces at catchment level. However, all the studies agree on one important point: the major condition for optimal functioning of all kinds of buffer interfaces is the uniformity and regularity of pollutant input flows (Bidois, 1999; Uusi-Kamppa et al., 2000; Haag and Kaupenjohann, 2001). In most cases where particular buffer zones are found partially or totally inefficient, the reason can be attributed to certain conditions of flow entering the buffers, either spatial (concentrated flow, shortcuts), temporal (flooding events, high flow velocity, snowmelt), or heterogeneous hydrological conditions. This suggests that the position of the interface buffer within the catchment has to be considered as one of the major drivers of their efficiency. Such an effect of landscape position as a modifier of function has been demonstrated for wetlands (Wang et al., 2004; Merot et al., 2006). In this respect we should consider the placement of buffers in 'hot spot' locations such as in bottom field corners, and factors that may undermine buffers such as subsurface drains, and select carefully the choice of buffer features for certain situations. An example of the latter point would be selection of grassed buffer strips for sheet flow, but bunds and constructed wetlands for convergent flow paths.

In summary there are several key issues in the assessment of buffers that contribute to the uncertainty in capacity and design of a range of buffer types:

i. a lack of buffer assessment at the catchment scale (e.g. Verstraeten et al., 2006) where multiple buffers are judged with respect to landscape connectivity and incorporating control (non-buffered) catchments;
ii. a lack of longer duration studies for proper assessment of aspects of buffer seasonality (particularly in source delivery and restoration of buffering properties between floods periods, due to biological dynamics, physical processes, management) and long-term function (Dorioz et al., 2006);
iii. improper consideration of the combined action of multiple pollutants (N/P interactions; P organic matter, effects of pesticides and metals, see Vidon et al., 2010).

Some limitations of landscape buffers

They can exist in situation where their sustainability is uncertain or may have possible side effects. Sustainability depends on the processes which can limit the further mobility of accumulated pollutants and sediments, essentially sorption, biogeochemical transformations and stabilisation of deposits by plant cover which can interact with maintenance operations such as re-seeding and mowing (Dillaha and Inamdar, 1997; Uusi-Kamppa et al., 2000; Dorioz et al., 2006). Biological uptake can be at the origin of further seasonal remobilisation and release of a fraction of the trapped total-P, eventually with an increase of bioavailability. In the case of soluble pollutants such as nitrates, the major processes involved in long-term retention are essentially biological or microbiological ones: plant uptake, microbial assimilation and denitrification. Although they depend primarily on biological factors (vegetation, microbial communities and biomass etc), these processes are strongly controlled by physical and chemical conditions: temperature, pH and redox conditions, water residence time, water table depth and fluctuation (Gilliam et al., 1997)

Side effects can exist in N and P antagonistic dynamics or in transfer systems, potentially impacting other compartments of the environment. A well known example of the antagonistic effect on nutrients is given by soil anoxia which decreases nitrate but can generate dissolved-P (Bidois, 1999). P release has been observed in wetlands receiving excessive amounts of NO_3 from cultivated areas (Paludan, 1995). Conversely, denitrification processes can be stopped at any stage of the reduction process and therefore, induce gaseous emission of nitrogen oxides. This is often described in literature on wetlands.

Socio-economics, policy and buffer cost-effectiveness

Increasing the areas of buffer features within catchments has costs to the farmer in terms of land taken out of productivity, set up and maintenance costs. Buffers may be promoted by a variety of means; through regulation, education and incentives. There are several important pieces of recent and upcoming legislation, predominantly relating to riparian buffers for improving water quality:

- Water Framework Directive - This has no explicit buffer regulations, but buffers appear in 70% of RBMPs. Buffers are defined as areas without cultivation, grazing or agro-chemicals.
- Pesticides Framework Directive - There is an obligation to provide 'appropriately sized' buffer zones where pesticides cannot be used and these must be in National Action Plans by 2012.
- Good Agricultural and Ecological Conditioning (with a direct link to the Single farm Payment) - No application of fertilisers is allowed near watercourses. There is an obligation to provide buffer zones along watercourses by 2012 (but 'buffers' and 'watercourses' are poorly defined).

Other countries have additional systems of regulation and incentivisation. In Scotland, for example, there is a three-tier system, where:

i. A minimum level of good practice is required by regulation. The minimum statutory requirements for riparian buffer strips state 'no land shall be cultivated for a crop that is within 2 metres of any surface water or wetland' (Scottish Regulations 'General Binding Rules for Agriculture, GBR 20, April 2008'; http://www.legislation.gov.uk/ssi/2008/54/pdfs/ssi_20080054_pdf).
ii. A national campaign of awareness raising compliments the baseline regulation and provides one to one advice, including on buffer features, in identified problem catchments.
iii. A system of financial payments exists to offset the costs of installing buffer strips (under the Scottish Rural Development Program), equal to £450 (~530 euro) per ha.

The latter payment for Scotland is similar to that for Sweden of 400 euro per ha. It is therefore desirable that governments and farmers (as subsidisers) achieve the most cost-effective solutions to placement of buffers. Methods are developing to bring together aspects of socio-economics and biophysical modelling to achieve these goals and help design buffer features at landscape scales (e.g. Qi and Altinakar, 2011; Balana et al., 2010). One aspect of this economic evaluation is judging buffers against other mitigation options in the rural environment and these could be farm based activities or alternate measures such as treatment of domestic septic tanks.

A further aspect is in optimising the use and design of buffers to achieve multiple benefits. Using an ecosystem approach buffers may be evaluated for a wide range of benefits to increase their attractiveness and likelihood of implementation. Such multiple benefits (in addition to diffuse pollution abatement) include:

- Biodiversity gains, both improving terrestrial habitats (e.g. beetles and pollinators beneficial to crops; Meier et al., 2005) and aquatic habitat (by shading to regulate stream temperatures);
- Societal gains, through improving the public image of agriculture and concretely the access to stream and rivers in otherwise intensively managed farmland (e.g. buffers incorporating footpaths);
- Offsetting water treatment costs, for example buffers that successfully treated nitrate may reduce the requirement for expensive water treatment downstream;
- By incorporating alternative crops into wider buffer features, for example production of hard wood tree species for timber, or energy crops.

It is generally recognised that buffers are considered beneficial but the metrics to quantify these wider benefits are difficult. Policy-makers realise that a system of payments by 'effects' not 'installation' for nations' limited agro-environmental budgets would be beneficial but there are difficulties in adopting the necessary framework to achieve this.

11.3 The landscape interfaces

Not all the interfaces within a landscape are buffers. Their state is crucial: they act as buffers if they present a proper structure and are properly managed. If not, they ensure a direct hydraulic connection of a source, or group of sources, to a water body. This type of propagation of diffuse pollution can include some cascading effects. Different types of interfaces can be distinguished regarding the object or landscape component they 'join' together and the fluxes which flow through them (table 11.1).

Table 11 1
Different types of interfaces and corresponding options for water quality control

Interface type	Strategy of mitigation	Hydrology	Landscape interfaces to deal	Expected Impacts Reductions of:
farm & surface water	Minimize the volume of dirty water produced Disconnect farm infrastructures from surface water	Surface water	Farm infrastructures (storage facilities etc) Farm environment	- organic matter loads and associated contaminants - total P
livestock & surface water	Introduce "barriers" between livestock and surface water	Surface water	Physical barrier: Fence off, Bridges	- sediment - bank erosion - particulate-P - fecal contaminants
Surface water boundaries	Create and manage vegetated buffers strips	Surface runoff	Vegetated strips	- erosion and - associated pollutants - total P
	Maintain, manage or restore riparian wetlands	Surface runoff and subsurface runoff	Wetlands	- water quantity - NO3 (denitrification) - particulate-P
Fields boundaries	Re-site gateways and paths	Surface runoff Surface flow connectivity	Trails, roads and access for animals and machinery to fields	- erosion and associated pollutants -total-P
	Field boundaries	Surface runoff	Vegetated Strips, Hedges, Terraces etc	- erosion and associated pollutants - total-P

Interfaces between farm water and surface water
Such interfaces regulate the surface flow connectivity between farm infrastructures, which produce waste water, and surface water. Mitigation options aim mainly to disconnect diverse storage facilities from surface water networks which prevent or reduce the transfer of water polluted by organic matter and nutrients.

Interfaces between livestock locations and movements, and surface water
Direct access of animals to the stream can significantly damage river banks and local aquatic ecosystems. Direct pollution with organic and inorganic nutrients and faecal contaminants is possible. Mitigation options aim to transform these interfaces into physical barriers between grazing animals and rapid hydrological pathways.

Surface water boundaries
The interfaces located along permanent streams and ditches control the direct inputs to surface water. If they have a buffer effect, they are called riparian buffers. They vary according to the hydrological conditions: 1) under saturated conditions, there are the riparian wetlands or wet meadows; the objective of water quality management is to maintain, manage or restore them; 2) under unsaturated conditions, the objective is to create and manage vegetated buffers strips. In all cases, it would be largely beneficial to maintain and manage vegetated strips on the river banks.

Field boundaries
Such interfaces control the connectivity of surface runoff and subsurface flows from plot to plot. These interfaces are very diverse: simple field boundaries or margins more or less vegetated, hedges and hedgerow, terraces etc., but all are outlets of plots. Properly managed or redesigned these interfaces can become classical vegetative buffer strips or areas, and so contribute to 1) attenuation of fluxes emitted from farm fields, an effect which partly depends on the soil and vegetation type and the possible interaction with shallow groundwater; 2) limiting flow concentration and resulting cascading effects on soil erosion.

Operationally, before any management of these interfaces, it is crucial to have a precise delineation of the functional hydrographic network, including natural rivulets and artificial ditches. This initial step is particularly important if the river springs are diffuse and vary with season, while the stream course and hydrographic network can expand in winter conditions. The choice of the starting point of the stream network is generally based on unclear criteria leading to underestimation of the length of the functional hydrographic network (Aurousseau et al., 1996; Merot et al. 2005). Three criteria have be efficiently considered: 1) current and local observations such as microtopography, or aquatic indicators such as plants, sediment; 2) seasonal observations, such as the precise locations of any springs, or water flowing during rainless periods (local experience of land managers is essential here); 3) past observations, which can be indicated by ancient maps or oral memory, especially when rivulets have been channelized. Cautiously, we should link aspects of expert on-site observations into flow paths, with catchment and hillslope modelling, to provide 'landscape design' tools for decision making when locating buffers.

11.3.1 Interface between farm and surface water

To attenuate the transfer of nutrients resulting from precipitation and surface runoff on farm infrastructures it is desirable to: 1) minimize the volume of waste water produced at farm level or regulate the period of loss due to water storage (e.g. by tightening slurry storage regulations); 2) separate if possible farm roof from steading runoff (from farm yards and roads); 3) move waste storage facilities far from surface water drainage networks 4) introduce buffers (such as a farm pond or filter strips; see photo 11.1) which disconnect farm infrastructures, mainly consisting of impervious surfaces, from sewerage and field drains or surface water (see Stadelmann and Blum, 2005).

Taken from: Rural Sustainable Drainage Systems (SuDS)
Technical Specification, Avery et al. 2009

Photo 11.1
A constructed farm wetland in England being used to treat field runoff from cropland (left; Avery et al., 2009); Retention lake to treat waste waters produced by a farm infrastructure (in Switzerland, picture from J. Blum; right)

11.3.2 Interface between livestock and surface water

Traditionally, dairy cows or other livestock spend a large part of the year on pastures far away from barns. Free and direct access for livestock to rivers, streams and ditches for drinking or crossing is common in many countries or regions, and especially in extensive pasturing or rangeland areas. This has many impacts on nutrient and sediment budgets of rivers: 1) direct input of nutrients and microbial contaminants due to defecation and urination into the watercourse; 2) erosion of river banks and re-suspension of sediment (Lefrancois et al., 2007), and thus generation of a flux of total-P and particulate-P. River bank erosion can allow direct flow connection with hillslope erosion.

A management targeting the protection of water quality aims to transform the interfaces between livestock and surface waters (Meals, 2004), into physical barriers between grazing animals and rapid hydrological pathways. This means establishing or modifying land infrastructure of grazing areas to avoid or limit direct access for livestock to stream networks, and to exclude high animal density -even temporary ones- in areas located very close to the watercourse. This can be achieved by: 1) fencing off rivers and streams (photo 11.2); 2) organizing livestock stream and river crossing through specific bridges or paths; 3) relocating gateways of pastures away from watercourses and if possible, from down to up slope. These three measures are applicable to grazing lands and livestock farms.

- *Fencing* creates a protective area of a minimal width (1 to 3 m) along the stream network (photo 11.2). This area has to be managed and new watering systems allowing cattle to drink without entering the streams are necessary. This measure cannot be applied in lands with a high density of ditches and streamlets. Fencing is required by law in Denmark. Under some agri-environment schemes, payments are available for the cost of providing off-stream water drinkers.
- *Bridging* the necessary crossing areas often goes with reducing the number and size of livestock (see photo 11.2). In some circumstances, e.g. uplands already mentioned, strengthening the stream bed of the remaining, necessary, crossing areas (with gravel, geotextile, etc) could be a compromise.
- *Gateways* are one of the critical areas of grazing fields. Around gateways temporary high livestock density generates soil compaction and poaching which represent a high risk of surface runoff and erosion when situated near a water course or stream. The pollution during rainfalls decreases when the gateways are not directly connected to the flow pathways, consequently moving the gate is a simple way to reduce pollution. Re-setting gateways and organizing better crossings for livestock, which implies reorganization of some aspects of the travel lines or animal travel pathways.

Photo 11.2
Fenced riparian buffer strips in Scotland encompassing gated animal crossing points (left); Fenced riparian buffer strips in Scotland with a public footpath as part of the buffer (right)

These measures aim to reduce sediment, total-P, probably total-N (via NH_4, organic N), and faecal contaminants during both low and high flow periods. Significant improvement of water quality, and a noticeable reduction of faecal contamination, has been obtained at catchment level when combined with general stream bank erosion control. In a modelling study of a dairy subcatchment of the River Humber, UK, Hampson et al. (2010) found a 60% reduction in Faecal Indicator Organisms as a result of riparian fencing. Other benefits are noticed in terms of ecosystem quality such as improvement of water-related wildlife habitat. It is often expected that aquatic biodiversity tends to increase (macroinvertebrate) and that there is an improvement in fish reproduction as spawning places are typically inhibited by excess amounts of sediment. However, there have been several recent studies which have been unable to observe improvements in macro invertebrates, and found uncertain effects on nutrients in the systems. In Virginia, USA, Ranganath et al. (2009) found, even for long-term 10-50 year fencing, that bank stability was improved but that macro invertebrate assemblages did not differ from control reaches as they were more influenced by upstream conditions than localised livestock exclusion. This supported earlier work in New Zealand by Parkyn et al. (2003) who commented that our expectations of riparian restoration should be subject to the acknowledgement of timescales for success and a need for restoration to be spatially targeted at the catchment scale.

Improvement of livestock health should also be considered and possible negative side effects mentioned. Livestock has a tendency to walk along fences and to create paths which become bare with a risk of concentrated runoff. Fencing increases the fragmentation of landscape (large fauna). Farmers sometimes complain about an uncontrolled development of some rodent populations. Management of vegetation along the fences has to be managed in an environmentally friendly manner.

11.3.3 Potential buffer capacity of surface water boundaries

Surface water boundaries are interfaces which, under certain conditions of structure, can act as a buffer, limiting nutrient transfer from the fields to water bodies. Two types of such riparian buffers can be distinguished regarding their hydrological conditions, either riparian buffer strips, in unsaturated conditions (deep or shallow groundwater), or riparian wetlands, in saturated conditions (very shallow groundwater, hilly and convexo-concave slope catchment).

Riparian buffers strips
In unsaturated conditions (deep or shallow groundwater systems), vegetative buffers, often called “vegetative filter strips” or “buffer strips” are usually constructed devices along reaches of hydrographic networks, with a strip feature. Implemented vegetated buffer strips have been widely studied and represent a quite well known,

tested and calibrated 'model'. By comparison with these references, it is possible to evaluate the buffer effect of some other types of less-standardized interfaces. Riparian buffer strips can contribute to the water preservation at local level, as a selective barrier effect to surface runoff, erosion, pollutant and organic matter export. They can also have an effect at catchment level, modifying the flow regime in the stream, particularly by a decrease of the peak flow which is determined by a combination of surface, subsurface and evaporation processes, and thus the erosion directly related to discharge such as river bank erosion.

Efficiency

Riparian strips are generally considered to offer an efficient protection against total-P: much of the P transported to watercourses is bound to particles and sedimentation is the main physical process occurring within buffer strips. Hoffmann et al. (2009) have reviewed efficiencies of riparian buffers for total-P retention and quantify the reduction of outputs (as a percentage of the inputs) as 41 to 93%. The same order of magnitude is given by Dorioz et al. (2006) for grass filter strips. According to Hoffmann et al. (2009) sedimentation is active along several flow paths and may account for P retention rates of up to 128 kg P $ha^{-1}.yr^{-1}$, while plant uptake may temporarily immobilize up to 15 kg P $ha^{-1}.yr^{-1}$. Retention of dissolved-P in riparian buffers is low, often below 0.5 kg P $ha^{-1}.yr^{-1}$ and several studies even show a significant release of dissolved-P (i.e. up to 8 kg P ha^{-1} yr^{-1}).

Functioning

The buffering effect in vegetative buffer strips results from a group of phenomena which are triggered during runoff periods and are the consequences of the hydrological and biogeochemical properties of the zone. Buffer strips have an effect on surface runoff flowing downslope over a rougher and more porous surface than upslope, causing it to slow down and infiltrate into the soil. These changes are linked to continuous soil coverage by plants, hence a greater resistance to surface flow which induces the decrease in flow velocity, and to a denser and sometimes deeper root system, which improves soil structure and increases the permeability of the surface soil layer. The infiltration, as a result of dense rooting brought about by perennial vegetation, is the main key to the slowing of flow velocity and deposition of particles. The coarser sediments and the main part of the total sediment are deposited in the upstream part of the buffer strip. Infiltration leads also to the injection of water, and of the dissolved forms carried, into the soil mass. The vegetation can be grass or a mixture of natural vegetation including grass, trees and bushes: the efficiency of the filter depends more on plant coverage of soil than of the vegetation type. However trees and shrubs are more stable over the long term and connote a more sustainable landscape (Michaud Irda, pers. comm.)

Infiltrated water can be stored, contributed to the groundwater recharge or evapo-transpirated. The consolidation, between periods of rainfall, of the deposited sediments is a critical but often neglected aspect of the buffer effect (Dorioz et al., 2006). Stabilisation is due to entrapment by fine roots in growth, and/or re-aggregation of fine particles in larger water stable aggregates. The dissolved pollutants and particularly dissolved-P are actively retained by soil constituents and biota which limits their displacement to the deeper layer of soil. Intercepted total-P is recycled, taken up and transformed by plants and microorganisms but accumulation tends to increase soil's P content of the filter. Microbial decomposition of organic pollutants is also active in the root zone.

Key factors

Retention of total-P in riparian buffers is mainly controlled by sedimentation processes, and the efficiency has long been considered by extension services and policy makers to depend mainly on the width of the buffer (or at best width weighted by slopes). But, there are no clear relationships between trapping of nutrients and buffer width. There have been a number of reviews in recent years concerned with the effects of width on retention of sediment (Schmitt et al., 1999; Liu et al., 2008), P (Hoffmann et al., 2009) and a combination of nutrients, sediment, FIOs and pesticides (Collins et al., 2009). Figure 11.2 compiles N and P retention data from the review by Collins et al. (2009). The degree of scatter in the percentage of total-P and nitrate retained by riparian buffers at a given width is the product of various factors, including the nature of contributing sources, slope, soil type, vegetation and, importantly, local flow and hydrological soil conditions. There are also the issues of uncertainty in true effectiveness, given that the majority of these data were generated by short-term (even single rainfall event) plot studies. The message is that the scientific evidence for effectiveness, in terms of nutrient trapping, makes policy decisions on buffer widths rather difficult. This is also true regarding the communication of the benefits of buffers to farmers and other land managers. The minimum for total-P would be to consider the nature of the surface runoff inputs. In some French regions recommendations take this suggestion into account, and indicate a width from 4 to 8 m with diffuse surface runoff and 10 to 15 m for concentrated surface runoff.

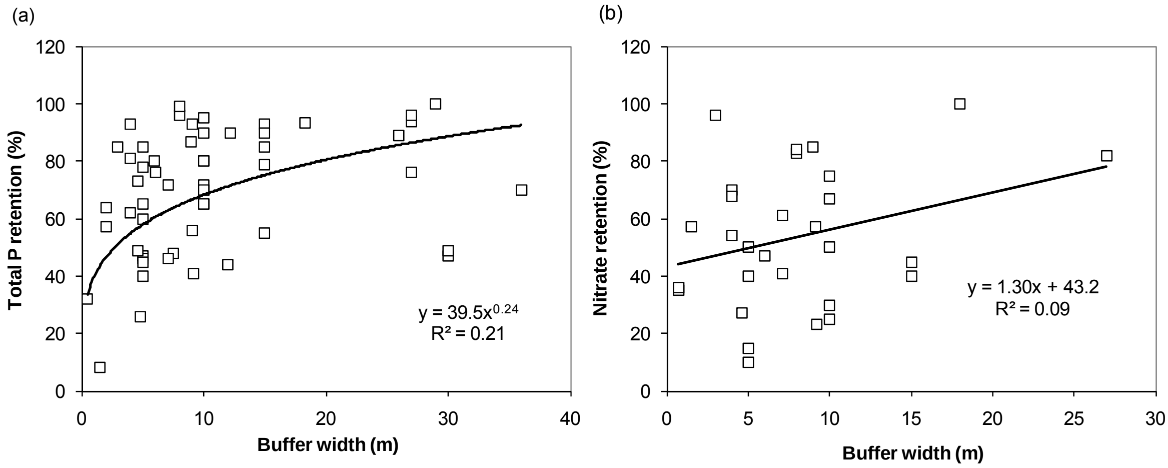

Figure 11.2
Documented retention efficiencies for (a) total-P and (b) nitrate as a function of riparian buffer width from the review by Collins et al. (2009)

Riparian buffers can have a number of additional functions (Borin et al., 2010), the action of which becomes more likely at increasing buffer widths (Figure 11.3). Even narrow buffers (~1 m) with appropriate vegetation and correct management have some potential for limiting pollution of surface waters by reducing application drift of fertilisers, pesticides or manure, trapping the coarser fraction of sediment, preventing bank erosion, and eliminating the practice of tilling and planting up to the extreme limit of properties. Of course, in this case, the effect on the dissolved and fine sediment fraction and associated pollutants in water runoff, is negligible. Conversely, if improperly managed they can be a critical area for erosion (furrows, trampling, bank damaged by tilling) or manure transfer from livestock. Slightly wider buffers (>5 m) bring increasing nutrient retention and transformation benefits, as well as increasing additional benefits. These are related to the possibilities for taller vegetation such as trees, to create root stabilisation of banks and beneficial detrital (e.g. leaf litter) input to streams. The widest buffers (>20 m) bring benefits associated with natural functioning riparian woodlands and floodplains such as habitat and potential zones for flood water storage.

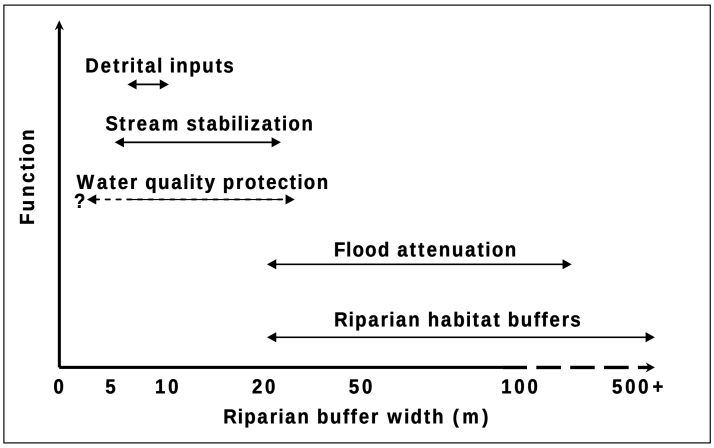

Figure 11.3

The functions that can be expected of riparian buffers according to their widths

Management of vegetation is another important factor to consider. Biological cycling is proposed to be the cause of greater P leaching from narrow, grassed, unmanaged riparian buffers than from adjacent cropland observed by Stutter et al. (2009). It may be that vegetation management is a critical factor in manipulating buffer conditions to remove stored P, increase buffer lifespan and prevent P leaching losses. Enhancing plant uptake of these solubilised nutrients would have dual benefits; by removing pore water nutrients that would otherwise leach (Lee et al., 2000) and providing a possible loss pathway via vegetation removal. Loss rates of 4-15 kg P ha^{-1} yr^{-1} have been documented through biomass removal (Hoffmann et al., 2009). Buffer biomass harvesting is a recommended strategy for agri-environmental schemes in Finland. However, the practicalities of manually cutting vegetation in narrow linear features make this task difficult and allowed grazing of buffers could provide issues for local degradation of soil surface state, bank erosion and FIO contamination.

Riparian wetlands
In saturated conditions (shallow groundwater systems), small riparian wetlands are often inserted in agricultural landscapes and neglected in national and regional wetland inventories. They are located in the lowlands of headwater catchments, scattered in the rural landscape and often associated with wet meadows. They are the inescapable interface between groundwater, for which they are the non-point outlet, and the water bodies, and often interface between intensively cultivated hillslopes and plateaus and the water bodies. The awareness of the wetlands functional role is increasing having progressively disappeared in intensive farming landscapes. The difficulty with integrating wetlands in management is exemplified by the European Water Framework Directive, where wetlands are not recognised as water bodies, but have to be included either in the groundwater bodies, or in the surface water bodies. They are now included more and more in regional or national inventories. French regional inventories are often a follow up of Natura 2000 and/or are associated with basin management programs (example in Savoie, http://www.patrimoine-naturel-savoie.org/inventaire-zh.php).

Wetland efficiency at reducing nitrate pollution has been extensively studied in natural (Fisher and Acreman, 2004; Machefer and Dise 2007) and artificial wetlands (Vymazyl et al., 2006; Kadlec et al., 2009). Their narrow width is adapted to function as a biogeochemical buffer (Mitsch and Gosselink, 1993). Beaujouan et al. (2002) and Sabater et al. (2003) have demonstrated that the length of contact between the wet zone and the contributing area of nitrate, i.e. the cultivated hillslope -and not the surface area of the wetland- is a major factor in their efficiency regarding nitrogen abatement (Figure 11.4).

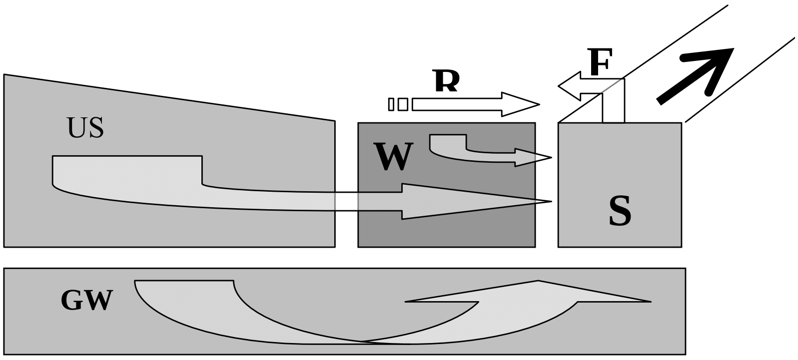

Figure 11.4

The different exchange between unsaturated zone (US, Groundwater (GW), wetlands (W) and stream (S), including R (surface runoff) and F (flooding). The buffer processes and effect highly depends on the exchange of water and matted between these compartments and the flow pathways and conditions (adapted from Merot et al., 2009)

Three conditions are required for denitrification: nitrate income fluxes; availability of C (heterotrophic activity of the bacteria); anoxic conditions; biological activity of specific bacteria regarding denitrification process. These conditions vary seasonally with hillslope conditions as well as micro-local conditions. Management can be advised to maintain or improve these conditions: increase or direct income fluxes from the hillslope to the wetlands (i.e. redirect drainage water for example); control stream water discharge to increase residence time; decrease spatial variability of the fluxes and decrease flow velocity within the wetlands (i.e. stream management), and harvesting of vegetative biomass to limit P stocks.

Despite a real denitrification in riparian wetlands, their effect on N removal at catchment scale stays low and highly variable (Sebilo et al., 2003), limited to medium events which allow sufficient water residence time for denitrification and wetland boundaries. High heterogeneity of water flow within the wetlands may avoid the site where denitrification could be increased. Their effectiveness increases from North to South European countries (Billen et al., in press), due to temperature conditions and microbiological activities.

Small wetlands can also trap particles and thus total-P. But under anaerobic conditions, reductive dissolution of some ferric compounds carrying P can be an important mechanism of seasonal dissolved-P release (Khalid et al., 1974). Anaerobic conditions can also limit the biotransformation of pesticides at an intermediate degradation step, producing the accumulation of metabolites. In some cases, the use of wetlands for nutrient retention or removal cannot be recommended as nutrients may have a negative effect on natural flora and fauna. Thus, if the preservation of the wetlands is positive anyway and acts as buffer, their management and artificial transformation have to consider some uncertainties and negative side effects.

Passeport (2010) has reviewed studies on nitrate retention in wetlands and compared efficiencies of artificial and natural wetlands. This review shows a nitrate removal percentage of 44% and 73%, on an average concentration basis, and 46% and 69%, on an average mass basis, for artificial and natural wetlands respectively. The higher efficiency of the natural wetlands is explained by a higher density and longer duration of the vegetation cover, and probably and consequently, a lesser carbon limitation. Otherwise, efficiency is higher when located at the outlet of small watersheds, where concentrations are less diluted. Therefore, artificial wetlands are particularly recommended to intercept drained catchment for which water is already canalized through easily identified tile drains and drainage ditches.

Constructed Farm Wetland (CFW) is a type of artificial device aimed to treat runoff from fields, or specifically more polluted water from farm steadings (an example is given by Photo 11.2). In the UK surface flow receiving, multi-cell CFWs are becoming increasingly promoted as a cost effective method of treating light to moderately contaminated waters (Carty et al., 2008). These features may be placed into field corners or wider riparian buffer strips to trap considerable amounts of sediment. However, the timescales of their operation are highly variable since they may be rapidly filled with sediments and become ineffective. At worst they can become a source of dissolved contamination. For example, sorbed P may be released from CFW under conditions of anoxia. Additionally, being ponds, they remain full and are of limited effect in reducing the hydrographic peak for flood alleviation. In that respect a more 'leaky' barrier providing a temporary wetland feature which empties slowly after being filled may provide a better option.

11.3.4 Create, maintain or manage field boundaries as buffers

A field boundary is composed of three parts: right and left sides (or top and bottom sides), plus an 'inter side' area. These three elements constitute the interface and have to be considered in terms of property, constitution, and management to assess the eventual buffer capacity of a given boundary. They can be described by their internal features, topographic structure (flat area, topslope to bottomslope elevation range, height of the bank if it exists, width), type of vegetation (plant cover, trees, bushes, grassland, weeds, natural or planted), soil permeability, roughness and soil surface state (micro-morphology including wheel tracks or livestock effects), orientation (regarding the slope direction) and their specific management practices. A field boundary has also to be characterized by adjacent fields (crop, slope, field management system), their potential for diffuse or concentrated surface runoff, and finally by their organisation within the landscape. At the landscape or catchment level, these boundaries constitute a network of similar linear entities, which can be described by: 1) the connectivity between elements, their directions regarding the slope, their positions regarding the streams; 2) the physical environment (in terms of groundwater, soil, bedrock); and 3) the variability of management practices applied on these pieces of land.

Regarding these characteristics, field boundaries are very diverse. They can be constructed and managed to be a filter strip and consequently designed according to their site specificity. Sometimes they are just a technical space between two farmers or a technical outlet. In both cases, their potential to act as a filter for surface runoff depends on their structure and properties, processes and parameters being similar to those of riparian buffer strip on unsaturated areas (already described). Their effects on subsurface flow depend on the local or hillslope conditions and on the vegetation type (hedges, trees, grass). The root system of the trees can take up water and chemical elements from shallow groundwater (up to a few meters deep), and thus have an effect on the subsurface flow in some cases, whereas the grass system cannot. Conversely, if the shallow groundwater is close to the soil surface, the buffer effect is reduced, or may even be a source of pollution, due to the generation of saturated overland flow and erosion during storms. These conditions are related to the characteristics of the field boundary, as well to the hillslope, and above all, the root systems of the vegetation of the border and the depth and dynamic of the water table.

Gateway relocation
Gateways are high risk areas: they represent a break in the boundaries of the fields and are critical areas for surface runoff in all kinds of field, and not only in pastures. Re-location of gateways, if possible, from down to up-slope can avoid or limit direct connections between high risk areas and associated fields to water courses, and contribute to breaking the hydrological connectivity from topslope to bottomslope. Ruts from tractor wheels tend also to converge and can channel surface water to these areas. At the landscape level gateways, livestock and tractor pathways represent a network of preferential flow pathways that hydrologically connect fields situated up-slope to down-slope and finally to the water course. Consequently moving the gate is a simple way, to decrease local and global hydrological connectivity, thus reducing this source of pollution. Moving the gateways is applicable to all kind of sloping lands. It is commonly relevant and there are few cases that would limit the adoption of this measure. It is applicable to all farming systems.

Hedges and hedgerow planting
A hedge is an interface between two fields planted with trees, with or without a bank. As for all the other types of field boundaries, this landscape feature can be extremely varied, in its own structure (width, elevation range, etc), in vegetation cover (type and management of trees) and in location and connectivity in the landscape. It contributes to breaking the hydrological connectivity of the landscape, regarding surface and subsurface flow and to removing both dissolved and particulate nutrients.

Different functions relative to the water quality are associated with hedgerows: 1) avoidance of surface runoff, erosion, suspended sediment, and therefore all the associated pollutants (e.g. phosphorus, FIO); 2) removal of nitrate by denitrification and plant uptake; 3) prevention of drift and increase of the local deposition of NH_3 emission.

A network of hedgerows (such as 'bocage') modifies the flow regime of streams and rivers (Viaud et al., 2004), decreasing the peak flow by modifying surface, subsurface, and the inter-storm flow, by modifying evapo-transpiration on the catchment, and therefore can contribute to decreasing the erosion of the river bank. Thus the hedgerow has an effect on water quality, when they are located perpendicular to the runoff (surface flow pathways), when the trees interact with the water table (subsurface flow).

Nitrate removal from 50 to 80% can be locally observed in the surface of the groundwater, in autumn and for hedges that interact with a shallow groundwater. The efficiency of nitrate removal depends on the contribution of this part of the groundwater to the stream flow, compared to deeper levels of the groundwater. The level of N removal generally stays low and uncertain, depending of the interaction between the water table and trees. This may represent a few % at the year level

Regarding total-P an effect can be expected related to that of hedgerows on surface runoff but many uncertainties concern P dynamic and cycling in such a system. The long term effect of P accumulation and thus risk of P release, and the location of P storage in biomass of trees, and thus, recycling through dead leaves, are poorly documented.

In conclusion
1. Buffer areas act by storing and trapping water and/or sediment and nutrients, uptaking dissolved nutrient by vegetation and biota, biogeochemical transformation (such as sorption, denitrification) and dilution. They have to present a proper structure and properly managed to act as buffers.
2. Buffers are diverse in situation and nature: at the interfaces between steading farm water and surface water, between livestock locations and surface water, at the surface water boundaries, at the field boundaries. They control the connectivity of water and matter by surface and subsurface flows, from plot to plot up to the water bodies.
3. For nitrate, usual values of attenuation (output / net input) range from 0.5 to 0.9, whereas for phosphorus it is much less, often below 0.05.
4. As no decomposition pathway exists for P, saturation of soils and buffer zones with reactive-P can increase, and it is unlikely that this can be infinitely sustained. As N oxides emission exists during denitrification processes, a buffer capacity for N can partly reflects side effects.
5. Buffer capacity is difficult to assess at local and moreover, at catchment level: multiple buffers act on catchments; longer duration studies are few numerous; multiple pollutants (N/P interactions; P and organic matter, effects of pesticides and metals) can induce antagonisms.

List of factsheets
Dorioz, J.M. and C. Gascuel-Odoux, 2010. Manage interaction between livestock and rivers or streams. [FS]
Dorioz, J.M., C. Gascuel-Odoux, J.P. Newell-Price, 2010. Re-site gateways away from high risk areas. [FS]
Gascuel-Odoux, C. and J.M. Dorioz, 2010a. Delineate the functional hydrographic network. [FS]

Gascuel-Odoux, C. and J.M. Dorioz, 2010b. Field boundaries and their potential buffer functions – an overview. [FS]
Gascuel-Odoux, C. and J.M. Dorioz, 2010c. Hedges and hedgerow planting. [FS]
Gascuel-Odoux, C, and J.M. Dorioz, 2010d. Maintain and /or manage riparian wetlands. [FS]
Gascuel-Odoux, C, J.M. Dorioz, T. Krogstad and M. Bechmann, 2010. Create and manage vegetated buffers at field boundaries. [FS]
Newell Price, J.P., 2010. Minimize the volume of dirty water produced. [FS]

References

Aurousseau, P., Suividant, H., Tortrat, F., Gascuel-Odoux, C. and Cordier, M.O. 1996. A plot drainage network as a conceptual tool for the spatialisation of surface flow pathways for agricultural catchments. Computer and Geosciences 35, 276-288.

Aurousseau, P. 2002. Les flux d'azote et de phosphore provenant des bassins versants de la Rade de Brest. Comparaison avec la Bretagne. Oceanis 27(2), 137-161.

Avery, L.M., Booth, P., Stutter, M.I., Vinten, A.J.A. and Langan, S.J. 2009. Rural sustainable drainage solutions (SuDS): Technical specification. Contract report for the UK Environment Agency.

Balana, B., Lago, M., Vinten, A.J.A., Slee, W., Baggaley, N., Castellazzi, M., Giullem, E., Futter, M. and Stutter, M. 2010. Cost-effective targetting of buffer strips for phosphorus mitigation in Lunan catchment : the case of Rescobie Loch. In: K. Crichton and R. Audsley (eds.). Proceedings of the Joint SAC/SEPA Biennial Conference: 2010, Climate Water and Soil: Science, Policy and Practice, April 2010, Edinburgh, UK.

Beaujouan, V., Durand, P., Ruiz, L., Aurousseau, P. and Cotteret, G. 2002. A hydrological model dedicated to topography-based simulation of nitrogen transfer and transformation: rationale and application to the geomorphology-denitrification relationship. Hydrol. Processes 16, 493-507.

Bidois, J. 1999. Aménagement de zones humides ripariennes pour la reconquête de la qualité des eaux : expérimentation et modélisation. PhD thesis Thesis, Universite de Rennes I, 214 pp.

Billen, G., Silvestre, M., Grizzetti, B., Leip, A., Garnier J., Voss, M., Howarth, R., Bouraoui, F., Behrendt, H., Lepisto, A., Kortelainen, P., Johnes, P., Curtis, C., Humborg, C., Smedberg, E., Kaste, O., Ganeshram, R., Beusen, A. and Lancelot, C. 2011. Nitrogen flow from European regional watersheds to coastal and marine waters. In: The European Nitrogen Assessment, Eds. Billen, G., Bleeker, A., Erisman, J.W., Grennfelt, P., Grizzetti, B., Howard, C., Sutton, M., van Griensven, A.,: Cambridge: Cambridge University Press, Chapter 13, in press.

Bleeker, A. and Erisman, J.W. 1998. Spatial planning as a tool for decreasing nitrogen loads in nature areas. Environmental Pollution 102, 649–655.

Borin, M., Passoni, M., Thiene, M. and Tempesta, T. 2010. Multiple functions of buffer strips in farming areas. European Journal of Agronomy 32, 103-111.

Carty, A., Scholz, M., Heal, K., Gouriveau, F. and Mustafa, A. 2008. The universal design, operations and maintenance guidelines for farm constructed farm wetlands (FCW) in temperate climates. Bioresource Tecnology 99, 6780-6792.

Collins, A.L., Hughes, G., Zhang, Y. and Whitehead, J. 2009. Mitigating diffuse water pollution from agriculture: riparian buffer strip performance with width. CAB Reviews. Perspectives in Agriculture, Vetinary Science, Nutrition and Natural Resources. 4, No, 39.

Correll, D.L. 2005. Principles of planning and establishment of buffer zones. Ecological Engineering 24, 433-439.

Dillaha, T. and Inamdar, S. 1997. Buffer zones as sediment traps or sources. In: N. Haycock, T. Burt, K. Goulding and G. Pinay (Editors), Buffer zones: Their processes and potential in water protection. Quest Environmental, Hartfordshire, UK, pp. 33-42.

Dorioz, J.M., Wang, D., Poulenard, J. and Trévisant, D. 2006. The effect of grass buffer strips on phosphorus dynamics A critical review and synthesis as a basis for application in agricultural landscapes in France. Agriculture, Ecosystems & Environment 17, 4-21.

Fisher, J. and Acreman, M.C. 2004. Wetland nutrient removal: a review of the evidence. Hydrology and Earth System Sciences 8, 673-685.

Gilliam, J., Parsons, J. and Mikkelsen, R. 1997. Nitrogen dynamics and buffer zones. In: N. Haycock, T. Burt, K. Goulding & G. Pinay (eds), Buffer zones: Their processes and potential in water protection. Quest Environmental, Harfordshire, UK, pp. 54-61.

Groffman, P.M., Gold, A.J. and Jacinthe, P.A. 1998. Nitrous oxide production in riparian zones and groundwater. Nutrient Cycling in Agroecosystems 52, 179-186.

Haag, D. and Kaupenjohann, M. 2001. Landscape fate of nitrate fluxes and emissions in Central Europe - A critical review of concepts, data, and models for transport and retention. Agriculture Ecosystems & Environment 86, 1-21.

Hampson, D., Crowther, J., Bateman, I. , Kay, D., Posen, P., Stapleton, C., Wyer, M., Fezzi, C., Jones, P. and Tzanopoulos, J. 2010. Predicting microbial pollution concentrations in UK rivers in response to land use change. Water Research 44, 4748-4759.

Haycock, N.E., Burt, T.P., Goulding, K.W.T. and Pinay, G. 1997. Buffer zones: Their processes and potential in water protection. Quest Environmental, Hardfordshire, UK.

Hill, A.R. 1996. Nitrate removal in stream riparian zones. J. Environ. Qual. 25, 743-755.

Hoffmann, C.C., Kjaergaard, C., Uusi-Kämppä, J., Bruun Hansen, H.C. and Kronvang, B. 2009. Phosphorus retention in riparian buffers: review of their efficiency. J. Environ. Qual. 38, 1942-1955 .

Jordan Meille L., Dorioz J.M. and Wang D. 1998 , Analysis of the export of diffuse phosphorus from a small rural watershed. Agronomie 18, 5-26.

Kadlec, R.H. 2009. Comparison of free water and horizontal subsurface treatment wetlands. Ecological Engineering 35, 159-174.

Khalid, R.A. and Patrick, W.H. 1974. Phosphate release and sorption by soils and sediments: Effect of aerobic and anaerobic conditions. Science 186, 53–55.

Lee, K-H., Isenhart, T.M., Schultz, R.C. and Mickelson, S.K. 2000. Multispecies riparian buffers trap sediments and nutrients during rainfall simulations. J. Environ. Qual. 29, 1200-1205.

Lefrancois, J., Grimaldi , C., Gascuel-Audoux, C. and Gilliet, N. 2007. Suspended sediment and discharge relationships to identify bank degradation as a main sediment source on small agricultural catchments. Hydrol. Processes 21, 2923-2933.

Liu, X., Zhang, X. and Zhang, M. 2008. Major factors influencing the efficacy of vegetated buffers on sediment trapping: A review and analysis. J. Environ. Qual. 37, 1667-1674.

Machefert, S.E. and Dise, N.B. 2004. Hydrological controls on denitrification in riparian ecosystems. Hydrology and Earth System Sciences, 8, 686-694.

Meals, D.W. 2004 Water quality improvement following riparian restoration in Lake Champlain Partnership and Research in the new millennium; ed. T. Manley et al., Kluver Academic/Plenum publisher, pp 81-95.

Meier, K., Kuusemets, V., Luig, J. and Mander, U. 2005. Riparian buffer zones as elements of ecological networks: Case study on Parnassius mnemosyne distribution in Estonia. Ecological Engineering 24, 531-537.

Merot, Ph., Hubert-Moy, L., Gascuel-Audoux, C., Clement, B., Durand, P., Baudry, J. and Thenail, C. 2006. A method for improving the management of controversial wetland. /Environmental Management 37, 258-270.

Mitsch, W. J., and Gosselink, J.G. 1993. Wetlands. Van Nostrand Reinhold, New York, 722 pp.

Owens, P.N., Duzant, J.H., Deeks, L.K., Wood, G.A., Morgan, R.P.C. and Collins, A.J. 2007. Evaluation of contrasting buffer features within an agricultural landscape for reducing sediment and sediment-associated phosphorus delivery to surface waters. Soil Use and Management 23 (Suppl. 1), 165-175.

Paludan, C. 1995. Phosphorus dynamics in wetland sediments. (Fosfordynamik i sedimenter fra vadonrader). Ph.D. Thesis University of Aarhus, Biological Institute. National Environmental Research Institute, 106 pp.

Parkyn, S.M., Davies-Colley, R.J., Halliday, N.J., Costley, K.J. and Croker, G.F. 2003. Planted riparian buffer zones in New Zealand: Do they live up to expectations? Restoration Ecology 11, 436-447.

Passeport, E., 2010. Efficiency of an artifical wetland and a forested buffer for pesticide pollution mitigation in a tile-drained agricultural watershed. PhD thesis, AgroParisTech, 177pp.

Patty, L., Real, B. and Gril, J.J. 1997. The use of grassed buffer strips to remove pesticides, nitrate and soluble phosphorus compounds from runoff water. Pestic. Sci. 49, 243–251.

Qi, H. and Altinakar, M.S. 2011. Vegetation buffer strips design using an optimisation approach for non-point source pollutant control of an agricultural watershed. Water resources management Water Resour. Manage. 25, 565–578.

Ranganath, S.C., Hession, W.C. and Wynn, T.M. 2009. Livestock exclusion influences on riparian vegetation, channel morphology, and benthic macroinvertebrate assemblages. J. of Soil and Water Conservation 64, 33-42.

Sabater, S., Butturini, A., Clement, J.C., Burt, T., Dowrick, D., Hefting, M., Maitre, V., Pinay, G., Postolache, C., Rzepecki, M. and Sabater, F., 2003. Nitrogen removal by riparian buffers along a European climatic gradient: Patterns and factors of variation. Ecosystems 6, 20-30.

Sebilo, M., Billen, G., Grably, M. and Mariotti, A. 2003. Isotopic composition of nitrate-nitrogen as a marker of riparian and benthic denitrification at the scale of the whole Seine River system. Biogeochemistry 63, 35-51.

Schmitt, T.J., Dosskey, M.G. and Hoagland, K.D. 1999. Filter strip performance and processes for different vegetation, widths, and contaminants. J. Environ. Qual. 28, 1479–1489.

Sharpley, A.N., McDowell, R.W. and Kleinman, P.J.A. 2001. Phosphorus loss from land to water: integrating agricultural and environmental management. Plant and Soil 237, 287-307.

Stevens, C.J. and Quinton, J.N. 2009. Policy implications of pollutant swapping. Physics and Chemistry of the Earth 34, 589-594.

Stadelmann, P. and Blum, J. 2005. 20 Jahre Einsatz für einen gesunden See. Dienststelle Umwelt und Energie, Luzern.

Stutter, M.I., Langan, S.J. and Lumsdon, D.G. 2009. Vegetated buffer strips can lead to increased release of phosphorus to waters: a biogeochemical assessment of the methods. Environmental Science and Technology 43, 1858-1863.

Stutter, M.I., Langan, S.J., Stockan, J. and Vinten, A.J.A. 2010. Managing stream riparian areas for multiple environmental benefits: the role of buffer strips. In: (eds. K. Crichton, R. Audsley) Proceedings of the Joint SAC/SEPA Biennial Conference: 2010, Climate Water and Soil: Science, Policy and Practice, Apr 2010, Edinburgh, UK.

Uusi-Kämppä, J., Braskerud, B., Jansson, B., Syversen, N. and Uusitalo, R. 2000. Buffer zones and constructed wetlands as filters for agricultural phosphorus. Journal of Environmental Quality 29: 151-158.

Verstraeten, G., Poesen, J., Gillijns, K. and Govers, G. 2006. The use of vegetated filter strips to reduce river sediment loads: an overestimated control measure. Hydrological Processes 20, 4259-4267.

Viaud, V., Durand, P., Merot, P., Sauboua, E. and Saadi, Z. 2005. Modeling the impact of the spatial structure of a hedge network on the hydrology of a small catchment in a temperate climate. Agricultural Water Management 74, 135-163.

Viaud, V., Merot, P. and Baudry, J. 2004. Hydrochemical buffer assessment in agricultural landscapes: From local to catchment scale. Environmental Management 34, 559-573.

Vidon, P. 2010. Riparian zone management and environmental quality : a multi-component challenge. Hydrological Processes 24, 1532-1535.

Vidon, P., Allan, C., Burns, D., Duval, T.P., Gurwick, N., Inamdar, S., Lowrance, R., Okay, J., Scott, D. and Sebestyen, S. 2010. Hot spots and hot moments in riparian zones: potential for improved water quality management. J. Am. Wat. Res. Assoc. 46, 278-298.

Vymazal, J., Greeway, M., Tondrski, K., Brix, H. and Mader, U. 2006. Constructed wetlands for wastewater treatment. Wetlands and Natural Resource Management 190, 69-96.

Wang, D., Dorioz, J.M., Trevisan, D., Braun, D.C., Windhausen, L.J. and Vansteelant J.Y. 2004. Using a landscape appraoch to interpret diffuse phosphorus pollution and assist with water quality management in the basin of Lake Champlain (Vermont) and Lac Léman (France), p. 159-189. In T.O. Manley P.L. Manley and TB Mihuc (eds.) Lake Champlain in the New Millennium. Kluwer Academic Plenim Publisher New York.

12 Surface water management

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12.1 Introduction

Surface water management is commonly employed to retain sediment and nutrients transported to surface waters from agricultural fields upstream. The management is based on fundamental principles such as hydraulic retention meaning increasing the residence time of water, interaction between sediment and water and biogeochemical processes like denitrification and sorption. Surface water management also strive to re-establish lost biomes as wetlands and lakes and increase the ecological quality of streams and rivers. In this chapter we will analyse existing forms of surface water management and show results from regional or national case studies.

Photo 12.1
An example of a larger scale surface water management project where the river and floodplain was restored. The Danish river Skjern after re-meandering the river channel during the period 2001-2003 that originally was straightened during the mid 1960ies for reclamation of the floodplain for intensive agricultural production (see Pusch et al., 2009)

12.2 Types of surface water management

The area of wetlands, ponds and lakes in Europe has declined markedly within the last decades, and in many European countries the wetland loss exceeds 50% of the original historical area (Jones and Hughes, 1993).

Most wetlands were drained to increase farm land and it has been one of the largest degradation processes on aquatic ecosystems during the last century. Draining was commonly accomplished by straightening and channelizing streams and rivers to facilitate a higher flow capacity in the channels (Hansen, 1996). This large scale and intense draining has lowered the water storage capacity of the ecosystems. The decline of aerial surface waters was coupled with reduction in nutrient retention capacity especially because of decrease in denitrification and sorption.

In lieu of the environmental degradation following the drainage many restoration or rehabilitation programmes were initiated in recent years around the world. These programmes include wide array of surface water management options. On the basis of these programmes there is strong evidence that the various mitigation options are working with clear impact on the nutrient cycling.

12.3 Experience with surface water management

River maintenance

Streams have always been influenced by society. Thousands of kilometres of streams have been straightened - often to the detriment of animals and plants. Hard-handed maintenance in the form of weed cutting by machinery and excavation of stream bed and banks has further deteriorated the conditions. A stop of stream maintenance improves the physical conditions on a short and long term and may increase the nutrients retention capacity of the stream channel (Svendsen et al., 1995).

Undisturbed streams in Europe normally sustain many diverse habitats (Iversen et al., 1995). However, the variety of animal and plant species in most streams has declined dramatically throughout the last century (Iversen et al., 1995; EEA, 1999). This decline is partly due to pollution of the streams with wastewater from agriculture, urban centres and industry. The comprehensive European effort to mitigate the waterways has gradually solved the problem of wastewater, and the quality of the stream water has been improved (EEA, 2007). The stream water is once again clean enough to support large and diverse population of animals and plants. However, this goal will not be reached only by attaining the goal of clean stream water. In addition, the stream itself needs to contain diverse physical variations to create enough aquatic niches. If the stream lacks the natural variations created by the shift between riffles and pools, the aquatic life in the stream will have lesser suitable habitats. Moreover, uniform streams where the weed is cut periodically exhibit lower retention capacity for nitrogen and phosphorus due to limited denitrification and depositional zones (Table 12.1).

Table 12.1

Mean total nitrogen and total phosphorus retention in different parts of a stream reaches as measured during the summer period (June-August) in a Danish river system Gjern Å (after Jeppesen et al., 1999)

Retention ($g\ m^{-2}$)	Type of substratum on stream bed		Dead river zone – edge habitat		
	Gravel	Sand	*Batrachium pcl.*	*Sparganium emers.*	*Sparganium erectum*
Nitrogen	4	9	20	20	32
Phosphorus	4	8	18	18	29

Not only the presence of aquatic vegetation, but also the type of vegetation is important for nutrient retention. Drainage ditches and wetlands covered with floating biomass (e.g. Lemna) show a higher denitrification compared to a situation with submerged macrophytes (Veraart et al., 2010). On the other hand, due to low oxygen concentrations under floating plants, this may result in an increase of iron-bound phosphorus release.

Streams have traditionally been maintained in order to ease the quick drainage of water from the fields. This has been achieved by cutting all vegetation several times per year and by removing gravel, stones and other physical barriers. In that way, the streams have been kept in canal-like state to the detriment of animals and plants and with low capacity of nutrient retention. With limited or no maintenance, the physical conditions of the streams will change. The vegetation functions as biological agents and if left undisturbed may change the physical conditions (Sand-Jensen, 1997). Vegetation in the stream bed and banks will quickly contribute to narrowing the channel profile and promote the deposition of sediment and particulate phosphorus. Over time, the narrowing of the channel will increase meandering and improve the contact with the surrounding riparian areas. A change in stream maintenance is a relatively cheap method and should therefore be used to a great extent in connection with the rehabilitation of physical conditions in streams and wetlands. In addition, if the regular maintenance is stopped the plant community in the streams will become richer and more diverse (Baattrup-Pedersen et al., 1998). Thereby, the capacity of the channel will increase denitrification and sedimentation of particulate P. The increase in nutrient retention capacity has been shown from *in situ* studies in stream channels of particulate P deposition and denitrification in macrophyte dominated stream channels (Svendsen et al., 1995; Kronvang et al., 2005; Veraart et al., 2010).

The stop in weed cutting in streams is applicable where the weed is mowed once or several times per year. The surrounding land should also be converted to hold higher water content because artificial drainage of the land results in higher nutrient losses. Such measures will increase the N and P retention capacity for a number of years until the channel has reached a steady state condition. The denitrification capacity of the channel will increase during a number of years following the change to a more diverse and larger stand of macrophytes facilitating surfaces and more diverse in-stream habitats including micro-zones with low current velocities, high organic content and low oxygen. The stop of stream maintenance in the form of weed cutting will have a positive impact on the ecological conditions in the stream (Sand-Jensen et al., 2006). This measure is of high relevance in many areas as stream maintenance is often conducted. It can be targeted to stream channels where the surrounding land exhibit extensive agricultural production.

River restoration

Improvement in morphological and substratum diversity of watercourses that have been straightened and channelized for drainage of agricultural land can be achieved by carrying out different types of stream restoration. Active restoration of streams is a quick and direct way of achieving the required physical improvement of the channel, and restoring the interaction with the adjacent riparian areas through inundation during high flow periods. A study of 13 catchments in Western Europe showed that residence time in the surface water system is the main factor for nutrient retention. This can be promoted by restoring floodplains and reconnecting inundation areas (De Klein and Koelmans, 2010).

During the last century numerous watercourses have been straightened and channelized or otherwise manipulated throughout Europe (Vought, 1995; Iversen et al., 1995; Hansen and Iversen, 1998; Kronvang et al., 2009a). As a result there are now only few watercourses resemble natural streams and many watercourses were classified as heavily modified waterways by the EU Water Framework Directive. The outcome of watercourse restorations should preferably resemble the natural conditions of the stream as much as possible and should only require a minimum of future maintenance. The last two decades have shown a growing interest in restoring watercourses and river valley ecosystems for the benefit of wildlife worldwide. Many river restoration projects have already been undertaken achieving wide environmental benefits (Kronvang et al., 2009b). Today, biodiversity is given high priority, thus restoration of rivers and their adjacent areas has been identified as one of the measures to maintain or even improve European biodiversity. At the same time there is increasing awareness that reinstating naturally functioning river-floodplain systems may bring catchment management benefits, particularly by increasing flood-storage capacity, giving increased nutrient retention and ameliorating low flows. Sustainable management and restoration of river and floodplain ecosystems may also reduce river maintenance costs and provide better facilities for amenity and recreation.

Photo 12.2
A Type 3 restoration of the River Brede in Denmark where the river channel was re-meandered in the mid 1990ies

A classification system was established which differentiates between **'Types'** and '**Methods'** by the European Centre of River Restoration (ECRR). Each restoration project is subdivided thereby 3 **types** according to the overall objectives of the project. This subdivision is based on the extent of rehabilitation within the watercourse system:

- **Type 1: Rehabilitation of watercourse reaches -** encompasses projects whose objective is local improvement of shorter reaches. The methods used under type 1 will typically result in *better habitats* locally, both in the watercourse and in the 2 meter cultivation-free border zone.
- **Type 2: Restoration of continuity between watercourse reaches -** encompasses projects aimed at ensuring free passage along watercourse systems. The methods employed under type 2 are those that reconnect reaches and restore *free passage* and continuity between a watercourse's component reaches and between the watercourse and its immediate surroundings.
- **Type 3: Rehabilitation of river valleys -** encompasses projects affecting both the watercourse and its whole river valley. The methods employed under type 3 are those that ensure that the watercourse and river valley function as an *ecological and hydrological entity*. The impact affects the watercourse and its surrounding riparian area.

The measure of watercourse restoration can be used where the watercourse is mapped to have a low physical quality utilising some sort of physical habitat index. Watercourses being heavily modified by straightening and channelization can be restored on reaches for improving the physical habitat (Type 1) or obstructions that are blocking for a free migration for fish such as salmonids can be removed (Type 2). Watercourse restoration includes a long list of possible methods that can be applied (Factsheet Kronvang, 2010b). The ultimate active restoration method is to remeander a formerly straightened and channelized reach of a stream or river. Such a measure has been widely applied in many Northwestern European countries (Friberg et al., 1994; Vivash et al., 1998; Friberg et al., 1998). Other methods include restoration of a channel by filling in spawning gravel for trout or salmon, removal of all obstructions to re-establish the free passage up and downstream in a river system or alternatively re-establish free passage by a riffle or a bypass stream.

The effectiveness of watercourse restoration is especially high in type three projects that include reestablishment of hydrological contact with the adjacent riparian areas. Analysis of restoration projects in 26

European rivers revealed that that measures on a larger scale are most effective in terms of restoring biodiversity. More comprehensive measures and tackling catchment-wide problems are required for a recovery of the stream ecosystem (Jähnig et al. 2010). In addition, this will in most areas increase the nitrogen and phosphorus retention capacity (Kronvang et al., 2009). Re-meandering of a former stream will create a larger area of stream bed and an increase in water retention time therefore increasing the possibility of nitrogen removal through denitrification and sedimentation of nitrogen and phosphorus in dead zones of the stream channel (Table 12.1). As an example, increasing the time of travel in a catchment with 50% may result in 15-20% more N and P retention (De Klein, 2010).

Effects of restored watercourses for some indicators may be seen immediately (e.g., physical habitat diversity, nutrient retention), whereas for other indicators it may take many years before the final outcome is observed (macrophyte community). Restored watercourses will have an initial disturbance phase where higher erosion and sediment loads are anticipated. A restored river channel may also have a higher in-channel retention capacity that may increase the potential for seasonal flooding.

Restoration of watercourses can be used in natural water bodies (NBW) and heavily modified water bodies (HMWB) where the survey of the ecological quality shows that it is not sufficiently high (minimum good for natural water bodies and best attainable for HMWB's). Restoration of watercourses is a highly successful measure because many methods are well developed and tested. The administrative handling of a project can take time depending on the national and regional regulations, thus long-term monitoring and performance evaluation is advisable.

Wetland restoration

The interest in wetland restoration programmes are growing globally in recent years (Litaor et al., 2004; Hoffmann et al., 2009). Restoration of riparian wetlands is conducted on low-lying often former organic soils that at some time in recent history were drained mostly for agricultural production purposes. They are human-made wetlands created on areas where the physical and chemical composition of the soils has been changed due to many years of draining and farming. Restored wetlands are in most cases established with the principal aim to retain nutrients loss from upstream agricultural fields by denitrification and sorption, and sedimentation. The restored wetland is established as groundwater or surface water wetlands adjacent to streams and rivers or as estuarine wetlands along the coast line. Nitrate-N from adjacent agricultural fields leaching to groundwater in restored groundwater dominated wetlands will often be reduced due to anaerobic zone where denitrification takes place and an uptake by vegetation of both inorganic N and P is prevalent. Surface water dominated wetlands receive both dissolved and particulate bound nitrogen and phosphorus forms that can be retained by sedimentation, denitrification (Gophen, 2000), sorption (Litaor et al., 2005, 2008; Sade et al., 2010) and biological uptake (Hoffmann et al., 2009; Kronvang et al., 2009a). Generally wetland soils offer favourable conditions for denitrification due to high carbon and low oxygen levels.

Restoration of wetlands can be used in riparian areas where a former wetland historically was situated but where the area now is being drained via ditches, tile drainage, pumping or simply channelization of watercourses. Restoration of wetlands can take place if farming in the area is ceased, become unprofitable or changed from intensive crop rotation to grazing or hay making in riparian wetlands during summer (Hambright and Zohary, 1998).

Photo 12.3
Example of a naturally inundated riparian wetland in Denmark

The effectiveness of restored wetlands for reducing nitrate, sedimentation of P or uptake of nutrients in vegetation is normally high. Experience from Denmark following the effects of restored riparian wetlands shows a net removal of nitrogen amounting to 200 kg N per hectare restored wetland (Table 12.2). The effect of the restored wetlands increases with increasing nitrogen loads and increasing water residence time (Table 12.2). Another direct effect of restoring wetlands is the stop in agricultural production leading to cessation of N leaching and N emission from these areas.

Experience with sedimentation of particulate P also shows that restored riparian wetlands can have a high retention capacity of 10-100 kg P per hectare inundated wetland (Kronvang et al., 2009a). However, the P retention is more certain for particulate P than for dissolved P as some restored riparian wetlands experiences a net leakage of dissolved P due to iron bound pools of former agricultural P in soils that is released under anaerobic conditions (Hoffmann et al., 2009). Experience from the Danish wetland monitoring programme shows that phosphorus during the first post-restoration period can both be net retained and net released (Table 12.3). The net release of phosphorus is due to leaking phosphate from former agricultural soils as iron bound P is desorbed under anaerobic conditions in the new water logged areas. Such a net release will cease after some time depending especially on the amounts of iron bound P in the soils.

Table 12.2
Nitrogen removal in the restored wetland and reduction in nitrogen leaching from changed land use estimated from a one year monitoring and mass-balance of restored wetland sites in Denmark. (After Hoffmann and Baattrup-Pedersen7)

Wetland project area	Measured N-removal	Reduction in N-leaching due to changed land use	Total change in nitrogen loss
		(kg N ha/yr)	
Egebjerg enge	53	-	53
Kappel	14	25	39
Geddebækken	90	35	125
Horne Mølleå	220	35	255
Karlsmosen	337	35	372
Lindkær	191	35	226
Snaremose ”Sø”	256	35	291
Ulleruplund	133	37	170
Gammelby Bæk	83	22	105
Nagbøl Å	163	24	187
Hjarup Bæk	170	30	200

Table 12.3
Outcome of phosphorus mass-balances for restored wetlands in Denmark after the first post-restoration year (After Hoffmann and Baattrup-Pedersen, 2007)

Wetlands	P-retention kg P ha-1 year-1	%
Ulleruplund, irrigated meadow	-0.43	-96
Gammelby Bæk, irrigated and inundated fen and meadow (uncertain calculation)	-20.4	-82
Egebjerg Enge, inundated meadow	0.13	6
Karlsmosen, irrigated and inundated fen and meadow	8.1 – 9.0	53-60
Snaremose, irrigated fen-meadow	2.6	18
Lindkær, irrigated fen-meadow	-0.5	-11
Geddebækken, irrigated fen-meadow	0.5	21
Nagbøl Å, remeandered, irrigation and inundated	0.9	11
Hjarup Bæk, remeandered, irrigation and inundated	12	42

Effects of restored wetlands will increase with time as vegetation cover increases. Restored riparian wetlands will also capture or degrade other substances as organic carbon, sediment, iron, pesticides, heavy metals, and others. The wetland may also have effects on pesticide degradation and emission of green house gases (Kayranli et al., 2010). Restored wetlands can be used where low-lying areas were formerly drained and where the soil surface has lowered due to mineralization of peat layers and physical shrinking. Since restored wetlands may include numerous land owners the administrative handling and project planning can take a long time. One example from Denmark is the restoration of ca. 2000 ha of low-lying areas around the river Skjern which took nearly 25 years to accomplish from the day the Danish Parliament decided to restore the river and wetland (Pedersen et al., 2007).

Photo 12.4
A naturally functioning river and floodplain system in Denmark

Lake re-establishment
Lakes are areas being permanently covered with standing water and consist of both open water areas and the shallow lake shore zone with higher vegetation (e.g. *Phragmites*). Very small lakes with an area below 1,000 m^2 are named ponds ditto. If new lakes or ponds are created where they existed before we are dealing with re-establishing of lakes. An important aim with re-establishing former drained lakes is to increase the retention potential for nutrients in order to reduce the loading downstream. Another aim is to improve the biodiversity and recreational value of surface waters.

The phosphorus loading of lakes is in the form of dissolved or particulate P. Particulate P will be deposited in lakes due to sedimentation, whereas part of the dissolved inorganic P will be taken up by the biomass of phytoplankton or macrophytes produced in the lake (Søndergaaard, 2007). Higher trophic levels in the lake (zooplankton and fish) can build in part of dissolved inorganic P and part of the P will deposit in the lake sediments as organic P or inorganic P. Sooner or later part of the deposited particulate P will be released from the lake sediments due to desorption processes or mineralization of organic material (Søndergaard, 2007). Another part of the deposited P will be unavailable and will permanently immobilized. The amount of P re-circulated and immobilized depends on local conditions in the catchment and the nature of the re-established lake. The factors and processes involved are: i) periods of anaerobic conditions in lake sediments; ii) ratio of iron to P in sediments; iii) residence time of water in lakes; and iv) depth of lake; inlet P concentration (Kronvang et al., 2005; Hejzlar et al., 2007).

Re-establishment of lakes will significantly increase the retention of phosphorus and nitrogen as lakes are naturally in steady state condition (constant nutrient loading). The most important factor for nutrient retention in lakes is the residence time for water in the lake - the higher residence time the greater the retention of nutrients (Windolf et al., 1996; Søndergaard et al., 2003; Hejzlar et al., 2007; Hejzlar et al., 2009). The retention of nutrients is also dependent on the amount of nutrients delivered to the lake. Clear water lakes have proven better in retaining nutrients than turbid lakes possibly due to more macrophytes and oxygen in the clear water lakes than in the turbid and eutrophicated ones. Clear water lakes will also have a larger recreational and natural value than eutrophicated and turbid lakes.

Photo 12.5
The re-established lake Bølling in Denmark formelly drained for agricultural purposes

Knowledge on the retention of nutrients in lakes builds on many years of mass balances calculations for a great number of different lake types and from many different empirical models (Kronvang et al., 2005). The nutrient retention varies both intra- and inter-annually as well as from lake to lake depending on nutrient loading, water residence time, lake depth and eutrophication state. Danish evidence on the efficiency of re-established lakes is given in Table 12.4 and 12.5. The monitoring of these re-established lakes show a retention of total N amounting to between 100 to 250 kg N per hectare per year and a retention of total P amounting to 1-3 kg P per hectare per year.

Table 12.4
Mass-balances for total nitrogen retention in re-established Danish shallow lakes and wetlands under the Action Plan II. Data is from the post-restoration year monitoring programme. (After Hoffmann et al., 2006)

Restored lakes and wetlands	Total wetland area (ha)	Lake area of total wetland area (ha)	Lake depth (m)	N-loading (tons N/yr)	N-retention in total wetland area (kg N/ha/yr)	Efficiency (%)
Slivsø	203	160	2.2	137	244	36
Aarslev Engsø	210	100	1	266	252	20
Nakkebølle Ind.	110	85	1	31.9	125	43
Gødstrup Engsø	90	55	0.5	16.9	100	53
Hals Sø	53	42	1.7	2.6	33	67
Ødis Sø	40	26	1.5	10.4	183	70
Skibet Enge	40	26	0.6	10.2	125	49
Wedellsborg H.	27	11	0.3	4.06	117	78

In principle a re-established lake will start to retain nutrients from the onset. However, a period with net release of P from the former rewetted soils can occur if they contain high amounts of 'old' agricultural P and are low in iron content. Another factor that may be important is a release of P from the former terrestrial vegetation and a hydrolysis of easily decomposable organic matter in the buried soils under water in the new re-established lake. On a longer time scale the re-established lakes will retain nutrients like natural lakes and the retention potential for nitrogen and phosphorus can be calculated from lake nutrient models. Re-established lakes will generally increase the biodiversity of the area and will often have a large stand of birds which are of great recreational value. Re-established lakes will increase the recreational value of an area and this will often show up in increased pricing of the houses and land area lying down to the lake. Management will not be needed in re-established larger lakes above a certain minimum size (10 ha), whereas smaller lakes as ponds may need management in the form of excavation of bottom sediments to prevent that the pond disappear due to sedimentation.

Table 12.5

Mass-balances for total phosphorus retention in re-established Danish shallow lakes and wetlands under the Action Plan II. Data is from the post-restoration year monitoring programme (After Hoffmann et al., 2006)

Restored lakes and wetlands	Total wetland area (ha)	Lake area of total wetland area (ha)	Lake depth (m)	P-loading (tons P/yr)	N-retention in total wetland area (kg P/ha/yr)	Efficiency (%)
Vilsted Sø	913	472	1	3.7	-0.01	-265
Slivsø	203	160	2.2	2.6	2.9	23
Aarslev Engsø	210	100	1	6	-1.3	-5
Nakkebølle Ind.	110	85	1	0.86	2.7	35
Gødstrup Engsø	90	55	0.5	0.31	0.9	26
Hals Sø	53	42	1.7	0.05	0.7	67
Ødis Sø	40	26	1.5	0.05	-2.3	-192
Skibet Enge	40	26	0.6	0.28	3	43
Wedellsborg H.	27	11	0.3	0.48	16.2	91
Rødding Sø*	34	21.1	1.1	0.1	-6	-192

*P-loading to the lake can be higher.

Chemical restoration of lakes

Phosphorus-enriched sediments can release phosphorus to the water through a process known as internal loading (Søndergaard et al., 2007). When sediments are contributing phosphorus to the lake, the managers can use nutrient inactivation techniques to remove phosphorus from the water column (called precipitation) and to retard its release from the sediments (called inactivation). The aim of this practice is to prevent eutrophication or rehabilitate those bodies of water considered eutrophic due to high concentrations of soluble phosphorus binding and settling it in the river bed (Svendsen et al., 1995). Lake managers use aluminum, iron, or calcium salts for phosphorus inactivation of lake sediments. The chemical substance that is most commonly used is aluminum sulphate (alum) $Al_2(SO_4)_3$ that is frequently used as a flocculating agent in the purification of drinking water and in waste water treatment plants. The addition of alum helps in reducing soluble reactive P (SRP) concentration in surface water following three mechanisms (Factsheet Lo Porto et al., 2010).

Nutrient inactivation is only appropriate where internal loading is a significant phosphorus source. If most phosphorus comes through external sources, alum treatment will not be effective. For appropriate nutrient inactivation projects, the length of treatment effectiveness varies with the amount of alum applied and the depth of the lake. The size and volume of the pond or lake must be accurately determined together with sediment and influent water quality and volume to ensure accurate dosage of the chemical agent.

Several studies have evaluated the effectiveness and longevity of treatments on several lakes in the USA concluding that alum treatment in shallow lakes for phosphorus inactivation is effective in most of the cases (c.f. Welch and Cooke, 1995). Applications in stratified lakes were highly effective and long lasting (> 80%) (Factsheet Lo Porto et al., 2010).

Constructed wetlands

Constructed wetlands are areas designed with the aim to optimize the removal of nitrogen and storage of phosphorus. As opposed to natural wetlands which are often situated at low-lying areas constructed wetlands can be made both on elevated areas where wetlands would not occur naturally.

Constructed wetlands are established with the principal aim to retain nutrients loss from neighbouring agricultural fields through processes like denitrification, sedimentation and sorption. The constructed wetland is either established in small ditches and brooks or as an end of tile drain pipe control. In all cases, nutrient enriched water from agricultural fields flow through constructed wetlands for nutrient load reduction before entering surface waters downstream (Fig. 12.1). There are numerous configuration of constructed wetlands that may include small sedimentation basins, infiltration basins with horizontal or vertical flow through the artificial substrate for sorption of P, shallow vegetative filters for storage of fine particles enriched in P and uptake of dissolved P and small basins with material that increases the P sorption potential (Braskerud et al., 2005).

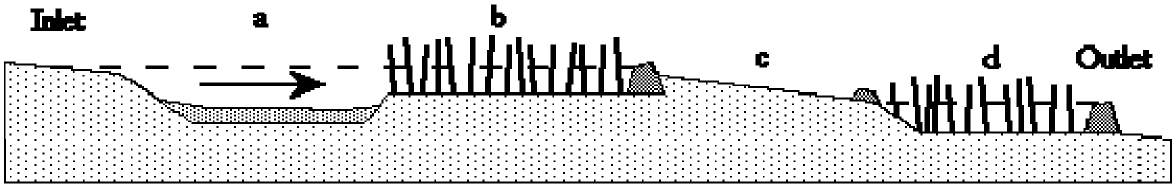

Figure12.1
Components used in Norwegian constructed wetlands: (a) sedimentation pond, (b) vegetation filter, (c) overflow zone covered with vegetation or stones and (d) outlet basin. Often low dams separate CW-components. Depths were originally 1 m in a, 0.5 m in b, 0 m in c and 0.5-0.8 m in d

The measure can be used in many different locations and constructions where nutrient enriched water leaves agricultural fields through e.g. tile drainage pipes, ditches or small brooks. Fine particles are often lost as larger aggregates from fields having a high settling velocity (Litaor et al., 2006). However, they will break up when transported in streams. As a result, wetlands should be constructed close to the problem area. Constructed wetlands can be made up of a combination of small surface sedimentation basins and infiltration basins with horizontal flow through the soil matrix for sorption of P. Another type of constructed wetland is installed in open ditches or small brooks and posses an initial sedimentation pond area for capturing sediment associated phosphorus (Braskerud et al., 2005). A constructed wetland may include overflow zones for oxygenation of water and sedimentation of fine particles under small runoff situations, and shallow (0.5 m) vegetative filters for sedimentation of P enriched particles under higher runoff (Braskerud et al., 2005). Vegetation and algae may take up dissolved P in the warm part of the year - quite insignificant however.

The effectiveness of constructed wetlands is normally high for nutrient removal and storage although most experience is from surface water systems in Norway, Sweden and USA. Experience with constructed wetlands established in small brooks shows an annual P retention of 1-50 g P m^{-2} yr^{-1} of constructed wetland (Table 12.6) (Braskerud et al., 2005). Usually the absolute and relative retention performance increases as the load increases. However, the P retention is more certain for particulate P than for dissolved P as some constructed wetlands experiences a net leakage of dissolved P.

Table 12.6 shows the average amount of phosphorus retained in a number of constructed wetlands in the south-eastern part of Norway. The table describes the retention, the catchment area and the percentage of agricultural land. The loading into the wetlands is not measured at all sites and it is therefore not possible to present the efficiency for the wetlands. A study of wetlands that constitutes 0.1-0.4 % of the contributing catchment area show retention efficiency of 45-75% for soil particles, 21-44% for phosphorus and 3-15% for nitrogen. There is a large variation in retention of phosphorus for the wetlands presented in Table 12.6. The differences in retention is mainly owing to differences between catchments (agricultural area and hydraulic loading), and variation in phosphorus loading to the wetlands. Aker, Buer and Holtan are wetlands with very low specific retention of phosphorus, which is mainly due to very low contributions of phosphorus to these wetlands.

Effects of establishing constructed wetlands will increase as vegetation cover increase until ca 50 % vegetative coverage (Braskerud, 2001). Depending on the erosion rate (filling of the wetland) constructed wetlands may last for 10-50 years before excavating. A decrease in N-retention performance has been monitored in some wetlands due to mineralization of organic-N. The P-binding capacity may also be filled up in the long run (Braskerud et al., 2005).

Photo 12.6
Example of a constructed wetland in Norway

Table 12.6
Retention of phosphorus in a number of constructed wetlands, including establishment, catchment area, and agricultural area (after Braskerud, 2001; Bach et al., 2003)

Wetland name	Established	Catchment area	Agricultural area	Wetland in % of catchment	Specific retention of P	Total retention of P/ha catchment area
	(yr)	(ha)	(%)	(%)	(g/m2 wetland area/yr)	(kg/ha/yr)
Berg	1990	148	17	0.06	51	0. 31
Kinn	1990	50	27	0.07	58	0. 4
Flataekken	1994	103	14	0.08	27	0. 23
Grautholen1	1993	22	99	0.21	71	1.5
Grautholen2	1993	22	99	0.38	46	1.76
Lågerød	1997	177	45	0.23	45	1.06
Aker	1995	16	75	0.37	2	0.06
Buer	1995	28	31	0.31	3	0.11
Haslestad	1996	7	100	0.14	60	0.86
Holtan	1995	13	52	0.33	4	0.16
Lund	1995	28	91	0.17	20	0.32
Råstad S	1996	25	65	0.18	22	0.41
Stein	1995	40	93	0.09	53	0.48
Sundby	1995	174	53	0.09	12	0.1
Vølen 1	1996	99	50	0.08	73	0.56
Vølen 2	1996	25	93	0.18	11	0.2
Average					34	0.53
Min						0.06
Max						1.76

Constructed wetlands will also capture excess sediments and iron coming from agricultural fields, and remove N through denitrification and biological uptake. However, for wetlands in streams the retention time will often be too short for significant nitrate removal. Retention of organic-N through sedimentation and oxidation of ammonia to nitrate may be relevant. Constructed wetlands may also contribute in the degradation of some pesticides.

Constructed wetlands can be used where nutrient enriched water from agricultural fields can be forced to pass through the facility. The ratio between the size of the constructed wetland and the contributing catchment area should be higher than a certain threshold (Norway: 0.2 %, Braskerud, 2001) as a certain hydraulic retention time is important for the different processes to take place. Biomass in the constructed wetland should not be harvested from the vegetative filters because the aquatic plants mitigate resuspension of sediment under storm runoff. Moreover, the sedimentation ponds may have to be emptied periodically due to limited storage capacity.

In conclusion

Management of surface water to increase nutrient removal and storage processes are often applied in River Basin Management Plans because they are cost-effective for both N and P.

1. Changes in river maintenance and river restoration can both assist in improving stream ecology and increase nutrient retention efficiency of running waters.
2. Re-establishment of formerly drained lakes in river systems will increase water residence time and increase nutrient retention.
3. Wetland restoration (surface water and groundwater wetlands) in river systems will increase water residence time and increase nutrient retention especially in larger streams.
4. Constructed wetlands can be used as a targeted mitigation measures for nutrient retention in drainwater outlets and smaller ditches and streams.

List of factsheets

Grizzetti, B. and F. Bouraoui, 2010. Streambank and shoreline protection. [FS]

Isermann, K., 2010. Protection aims for the hydrosphere: critical levels and critical loads of the nutrients C, N, P and S in surface waters and groundwater. [FS]

Kronvang, B., 2010a. Reestablishing of lakes. [FS]

Kronvang, B., 2010b. Restoration of watercourses with reestablishment of inundated wetlands. [FS]

Kronvang, B., 2010c. Restored riparian wetlands. [FS]

Kronvang, B., 2010d.. Stop for stream maintenance: weed cutting. [FS]

Kronvang, B. and A. Lo Porto, 2010. Restoration of stream with reestablishment of inundated riparian wetlands. [FS]

Kronvang, B., B. Braskerud and A. Lo Porto, 2010. Constructed wetlands. [FS]

Lo Porto, A., B. Kronvang and B. Braskerud, 2010. Lake and pond treatment by nutrient inactivation. [FS]

References

EEA, 1999. Environment in the European Union at the turn of the century Environmental Assessment Report No. 2, pp. 446.

EEA, 2007. Europe's Environment – The fourth assessment. European Environment Agency, Copenhagen, 411 pp.

Baattrup-Pedersen, A., Riis, T. & Hansen, H.O. 1998. Weed cutting and stream plants in Danish streams. Vand and Jord, 5, 136-139 (In Danish).

Braskerud, B.C. 2001. The influence of vegetation on sedimentation and resuspension of soil particles in small constructed wetlands. J. Environ. Qual. 30, 1447-1457.

Braskerud, B.C., Tonderski, K.S., Wedding, B., Bakke, R., Blankenberg, A.G., Ulén, B. and Koshiaho, J. 2005. Can constructed wetlands reduce the diffuse phosphorus loads to eutrophic water in cold temperate regions? J. Environ. Qual. 34, 2145-2155.

Conservation in the 1990s: A global perspective. Proceedings of an IWRB Symposium. Florida, 12–19 November 1992, IWRB Special Publication No. 26, Slimbridge, United Kingdom, pp. 164–169.

De Klein, J.J.M. and Koelmans, A.A. 2010. Quantifying seasonal export and retention of nutrients in lowland rivers at catchment scale. Hydrological Processes, in press.

Friberg, N., Kronvang, B., Hansen, H.O. and Svendsen, L.M. 1998. Long-term habitat-specific response of a macroinvertebrate community to river restoration. Aquatic Conservation 8, 87-99.

Friberg, N., Kronvang, B., Svendsen, L.M., Hansen, H.O. and Nielsen, M.B. 1994. Restoration of a channelized reach of the river Gelså, Denmark: Effects on the macroinvertebrate community. Aquatic Conservation 4, 289-296.

Gophen, M. 2000. Nutrient and plant dynamics in lake Agmon wetlands (Hula Valley, Israel): a review with emphasis on Typha domingensis (1994-1999). Hydrobiologia 441, 25-36.

Hansen, H.O. (Ed.) 1996. River Restoration - Danish experiences and examples. Ministry of Environment and Energy, National Environmental Research Institute, ISBN: 87-7772-279-5, 98 pp.

Hansen, H.O. and Iversen, T.M. 1998. The European Centre for River Restoration (ECRR). - In: Hansen, H.O. and Madsen, B.L. (Eds.): River restoration '96 - Proceedings - Session lectures. International conference arranged by the European Centre for River Restoration. - National Environmental Research Institute, Denmark.

Hambright, K.D. and Zohary, T. 1998. Lakes Hula and Agmon: destruction and creation of wetland ecosystems in northern Israel. Wetlands Ecology and Management 6, 83-89.

Hejzlar, J., Samalova, K., Boers, P. and Kronvang, B. 2007. Modelling Phosphorus Retention in Lakes and Reservoirs. In: Kronvang, B, Faganeli, J and Ogrinc, N (red.), The Interactions Between Sediments and Water, Dordrecht: Springer p. 123-130.

Hejzlar, J., Anthony, S., Arheimer, B., Behrendt, H., Bouraoui, F., Grizzetti, B., Groenendijk, P., Jeuken, M.H.J.L., Johnsson, H., Lo Porto, A., Kronvang, B., Panagopoulos, Y., Siderius, V., Silgram, M., Venohr, M., Zaloudík, J. 2009. Nitrogen and phosphorus retention in surface waters: an intercomparison of predictions by catchment models of different complexity. Journal of Environmental Monitoring 11, 584-593.

Hoffmann, C.C, Baattrup-Pedersen, A., Amsinck, S.L. and Clausen, P. 2006. Overvågning af Vandmiljøplan II, Vådområder 2005, Technical report from NERI, Aarhus University, No. 576, 126 pp. (In Danish).

Hoffmann, C.C. and Baattrup-Pedersen, A. 2007. Re-establishing freshwater wetlands in Denmark. Ecological Engineering 30, 157-166.

Hoffmann, C.C, Kjaergaard, C., Uusi-Kämppä, J., Hansen, H.C.B. and Kronvang, B. (2009). Phosphorus retention in riparian buffers: Review of their efficiency. J. Environ Qual. 38, 1942-1955.

Iversen, T.M., Hansen, H.O. and Madsen, B.L. 1998. Introduction. Aquatic Conservation. Marine and Freshwater Ecosystems 8, 3-4.

Iversen, T.M., Kronvang, B., Hoffmann, C.C., Søndergaard, M. & Hansen, H.O. 1995. 'Restoration of aquatic ecosystems and water quality'. In Møller, H.S. (Ed), Nature restoration in the European Union, Proceedings of a Seminar, Denmark 29-31 May 1995, Ministry of Environment and Energy, Denmark, 63-69.

Jeppesen, E., Søndergaard, M., Kronvang, B., Jensen, J.P., Svendsen, L.M. and Lauridsen, T.L. 1999. Lake and Catchment Management in Denmark. - Hydrobiologia 395/396,,419-432.

Jones, T.A. and Hughes, J.M.R. 1993. Wetland inventories and wetland loss studies: A European perspective. In: Moser, M.E., Prentice, R.C. and van Vessem, J. (editors). 1993. Waterfowl and wetland conservation in the 1990s - a global perspective. IWRB Sp. Publ. 26, 248 pp.

Jähnig, S.C., Brabec, K., Buffagni, A., Erba, S., Lorenz, A.W., Ofenböck, T., Verdonschot, P.F.M. and Hering, D. 2010. A comparative analysis of restoration measures and their effects on hydromorphology and benthic invertebrates in 26 central and southern European rivers. Journal of Applied Ecology 47, 671-680.

Kayranli, B., Scholz, M., Mustafa, A. and Hedmark, Aa. 2010. Carbon storage and fluxes within freshwater wetlands: a critical review. Wetlands 30, 111–124.

Kronvang, B., Hejzlar, J., Boers, P., Jensen, J.P., Behrendt, H., Anderson, T., Arheimer, B., Venohr, M., Hoffmann, C.C. and Nielsen, C.B. 2005. Nutrient Retention Handbook: Software Manual for EUROHARP-NUTRET & Scientific Reveiw on Nutrient Retention, Norwegian Institute for Water Research (NIVA) (EUROHARP Report 9-2004; NIVA report SNO).

Kronvang, B., Hoffmann, C.C. and Dröge, R. (2009a) Sediment deposition and net phosphorus retention in a hydraulically restored lowland river-floodplain in Denmark: combining field studies with laboratory experiments. Marine and Freshwater Research 60, 638-646.

Kronvang, B., Thodsen, H., Kristensen, E., Skriver, J., Wiberg-Larsen, P., Baattrup-Pedersen, A., Pedersen, M. L. and Friberg, N. 2009b. In: IVth ECCR International Conference on River Restoration 2008. Proceedings. CIRF Centro Italiano per la Riqualificazione Fluviale, p. 207-222.

Litaor, M.I., Reichmann, O., Auerswald, K., Haim, A., and Shenker, M. 2004. The geochemistry of phosphorus in peat soils of a semiarid altered wetland. Soil Sci. Soc. Am. J. 68, 2078-2085.

Litaor, M.I., Reichmann, O., Haim, A., Auerswald, K. and Shenker, M. 2005. Sorption characteristics of phosphorus in peat soils of a semi-arid altered wetland. Soil Sci. Soc. Am. J. 69, 1658-1665.

Litaor, M. I., G. Eshel, O. Reichmann, M. Shenker 2006. Hydrological control of phosphorus mobility in altered wetland soils. Soil Sci. Soc. Am J. 70, 1975-1982.

Litaor, M. I., G. Eshel, R. Sade, A. Rimmer, M. Shenker. 2008. Hydrogeological characterization of an altered wetland. Journal of Hydrology 349, 333– 349.

Pedersen, M.L., Andersen, J.M., Nielsen, K. and Linnemann, M. 2007. Restoration of Skjern River and its valley: Project description and general ecological changes in the project area. Ecological Engineering 30, 131-144.

Pusch, M., Andersen, H.E., Bäthe, J. Behrendt, H., Fischer, H., Friberg, N., Gancarczyk, A., Hoffmann, C.C., Hachol, J., Kronvang, B., Nowacki, F., Pedersen, M.L., Sandin, L., Schöll, F., Scholten, M., Stendera, S., Svendsen, L.M., Wnuk-Glawdel, E., and Wolter, C. 2009. Rivers of the Central European Highlands and Plains.

In: Rivers of Europe. Eds. Klement Tockner, Urs Uehlinger, Christopher T. Robinson. Academic Press Incorporated, p. 525-576

Sade, R., Litaor, M.I. and Shenker M. 2010. Evaluation of groundwater and phosphorus transport in fractured altered wetland soils. 2010. J. Hydrology 393, special issue 1-2, 133-142.

Sand-Jensen, K. 1997. Macrophytes as biological engineers in the ecology of Danish streams. In: K. Sand-Jensen and O. Pedersen (Eds.) Freshwater biology – Priorities and development in Danish research. G.E.C. Gad, Copenhagen. p. 74-101.

Sand-Jensen, K., Friberg, N. and Murphy, J. (Eds.). 2006. Running Waters –Historical development and restoration of lowland Danish streams. National Environmental Research Institute, Aarhus University, Denmark, 159 pp. (ISBN 978-87-7772-929-4).

Svendsen, L.M., Kronvang, B., Kristensen, P. and Græsbøl, P. 1995. Dynamics of phosphorus compounds in a lowland river system: importance of retention and non-point sources. Hydrological Processes 9, 119-142.

Søndergaard, M. 2007. Nutrient dynamics in lakes - with emphasis on phosphorus, sediment and lake restoration, Danmarks Miljøundersøgelser, Aarhus University.

Søndergaard, M., Jensen, J.P. and Jeppesen, E. 2003. Role of sediment and internal loading of phosphorus in shallow lakes. Hydrobiologia 506, 135-145.

Vivash, R., Ottosen, O., Janes, M. and Sørensen, H.V. 1998. Restoration of the rivers Brede, Cole. Skerne: A joint Danish and British EU-LIFE project - The river works and related practical aspects. Aquatic Conservation 8, 197-208.

Veraart, A.J., De Bruijne, W.J.J., De Klein, J.J.M., Peeters, E.T.H.M. and Scheffer, M. 2010. Effects of aquatic vegetation structure on denitrification. Biogeochemistry, in press.

Vought, L.B.M. 1995. Restoration of streams in the agricultural landscape. In: Eiseltova, M. & Biggs, J. (Eds), Restoration of stream ecosystems, IWRB Publication, 37, Slimbridge, Gloucestershire, UK, 18-29.

Welch, E.B. and Cooke, G.D. 1999. Effectiveness and longevity of phosphorus inactivation with alum. J. Lake Reserv. Manage. 15, 5-27.

Windolf, J., Jeppesen, E., Jensen, J.P. and Kristensen, P. 1996. Modelling of seasonal variation in nitrogen retention: a four-year mass balance study in 16 shallow lakes. Biogeochemistry 33, 25-44.

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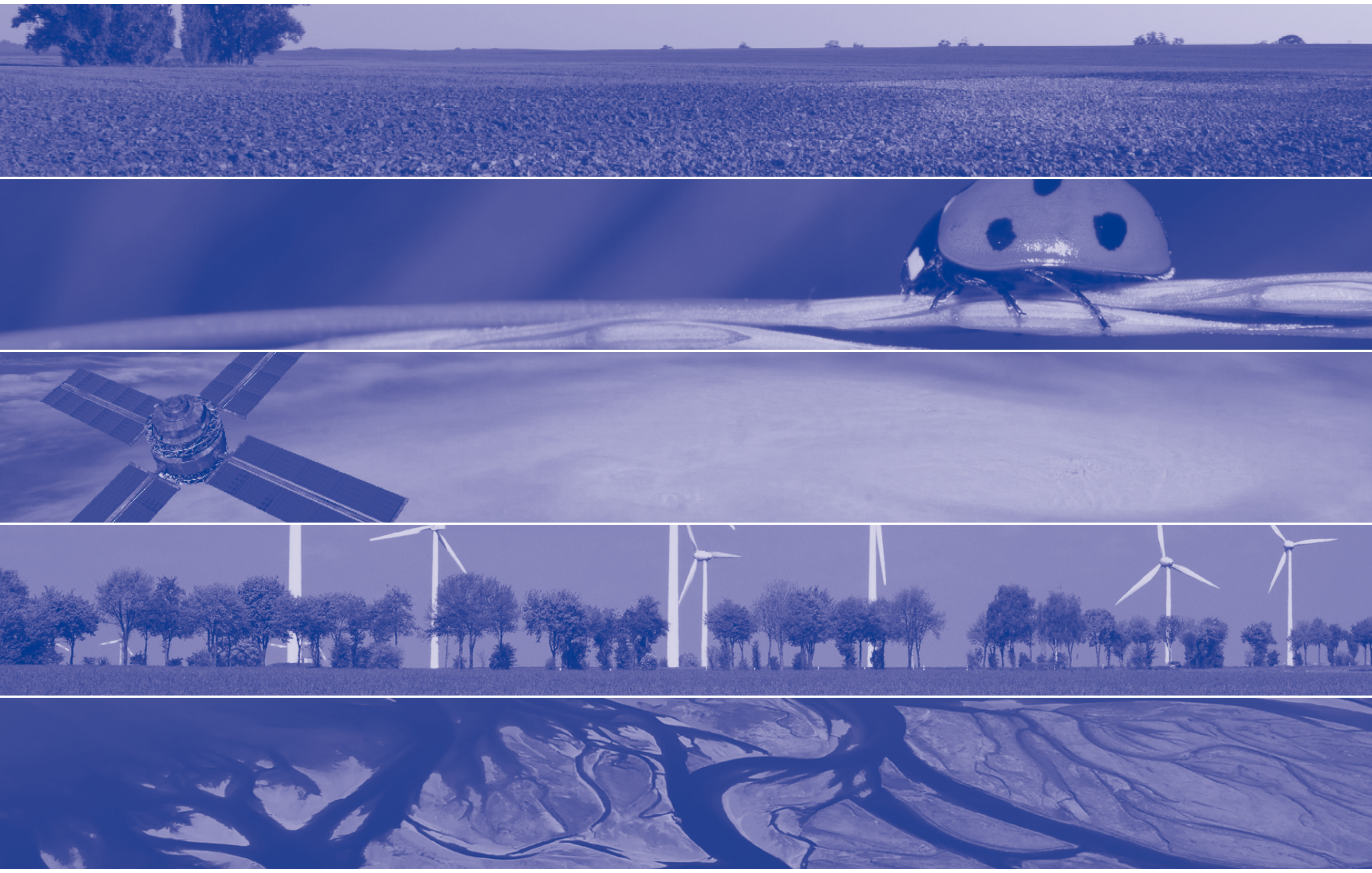

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