Nutrient Cycling, Modelling and Management from Field to Catchment Scale

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Abstract

The changes in land use caused by antropogenic interventions, has increased the global pool of nutrients (Nitrogen and Phosphorus) in terrestrial and aquatic ecosystems. This report analyzes the behavior of the main nutrient biogeochemical cycles and the consequences of its alterations. It is remarkable the enhancement of some inputs and outputs to terrestrial and aquatic ecosystems and the increasing rate of nutrient loss from the soil to other reservoirs. From field to a catchment scale, the quality of fresh waters and its discharges to coastal environments have been deteriorated, emphasizing the increased trophic state and primary production with consequences such as acceleration of natural eutrophication processes of water bodies. Additionally, the effect of climate change in the balance of nitrogen and phosphorus is reviewed through the use of low complexity models to assess their inputs and outputs. Finally mitigation measures, to minimize nutrient losses from the soil, are explored and their efficiency is discussed. Even if many different ways to deal with the increasing input of nutrient to aquatic environments are presented, the main focus is put on the functioning of wetlands and buffer strips which have proven to be generally effective for the purpose.

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Chapter 1

Introduction

Today's population of around 7 billion is expected to increase to about 9 billion by 2050. By this time, another one billion tonnes of cereals and 200 million extra tonnes of livestock products will need to be produced every year.[9] This trend carries along an alteration on the organization of the territory in terms of land use (agriculture, forestation, livestock, mining, sewage, urbanization, industries) and a higher pressure on the environment in order to satisfy these demands. Especially the expansion of agricultural land is widely recognized as one of the most significant human alterations to the global environment.[5][3] Additionally, the intensification of agricultural systems in order to achieve a higher production efficiency and be able to satisfy the current food demand per hectare, implied the use of: higher-yielding crops, irrigation, mechanization and a higher use of fertilizers and pesticides. Jointly, these practices lead to adverse consequences to the environment such as increased erosion, lower soil fertility, reduced biodiversity, pollution of groundwater, eutrophication of aquatic systems and in a global scale, consequences on atmospheric constituents and climate.[5]

The heavy use of fertilizers to enhance plant growth and increase the production yield results in high amounts of untaken nutrients (specifically, nitrogen [N] and phosphorus [P]) on the topsoil. Then, they can easily leach into the soil or be transported by run-off, increasing the nutrients load in adjacent aquatic ecosystems. As the primary production of lakes is strongly limited by nutrient availability, these inputs would enhance lake productivity (trophic state) and cause cascading effects on the remaining trophic levels[4], and increase toxic algae blooming, frequency of anoxic events, and fish mortality.[7] Moreover, these changes entail high costs of water purification for human consumption, and losses on biodiversity, fishery and recreational activities.[8]

In order to come up with an appropriate mitigation plan against these adverse effects, awareness on the differences in the cycle and dynamics of phosphorus and nitrogen is crucial. From one side, N is ubiquitous in the environment and it is also one of the most important essential nutrients and central to the production of all crop plants. The most abundant form of N is elemental nitrogen (N_2) which is inert and not directly available for plant uptake. In order to be biologically available "natural" fixation of atmospheric N_2 needs to happen, which primarily takes place by lightning and biological processes. Once in "reactive" form as NO_3 and/or NH_4, it becomes very mobile and it can be rapidly incorporated into living tissue or leached into the soil; lastly, microbial mediated denitrification completes the cycle.[3] Moreover, nowadays the major sources of N to cropping systems and into the environment are: fertilizer inputs, organic wastes, manure, BNF (biological nitrogen fixation), atmospheric additions (N dissolved in wet and dry deposition) and nitrate contained in irrigation water.[3][5] On the other hand, once P is liberated from minerals during erosion processes and it is quickly sequestered into more recalcitrant phases (bound with iron and manganese oxyhydroxides). These P-bearing phases often constitute the main, long term storage pool for soil P and an atmospheric gas phase is non existent, thus ecosystems (plant and organisms) have a limited access to P which depends on its aqueous transfer. However, the modern terrestrial P cycle is dominated by agriculture and human activity; its load to rivers has doubled due to increase use of fertilizers, deforestation, soil loss, and sewage sources.[2] Additionally, P often functions as the limiting nutrient in aquatic systems and thereby determines phytoplankton abundance.[1] This implies that serious attempts should be launched to lower the phosphorus loading to lakes, and its achievement implies a reduction of agricultural inputs as a prerequisite.

The following chapters will elaborate on: trends in world rivers (one study case), in terms of nutrient concentrations, and modelling scenarios for nitrogen and, mitigation options in catchments to minimize nutrient losses with a special focus on the functioning of wetlands and buffer strips.

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Chapter II

Nutrient concentrations and trends in world rivers and modeling scenarios for nitrogen.

Adriana Rodríguez Fernández

Diffuse nutrient losses represent the major cause of decreased water quality in a large part of Europe's surface and groundwater bodies. Estimates made by the OSPAR Convention[11] indicates that as much as 64% of the nitrogen and 46% of the phosphorus inputs to the North Sea originate from diffuse sources of which agriculture is the dominating source.[13] Agriculture represents approximately 62% of nitrogen load to surface waters in Europe, (from a minimum of 18% in Portugal to a maximum of 97% in Denmark).

The concentration of nitrogen in lakes and reservoirs results from a dynamic equilibrium between inputs, outputs and different retention processes that remove nitrogen from the water or store it. In the case of nitrogen, retention processes are volatilization of nitrogen through denitrification of nitrate to N_2, NO_2 or N_2O and sedimentation of particulate nitrogen from the water phase to the lake or reservoir sediment.[9]

Nutrient losses from diffuse sources are now one of the main pressures on the ecological status of the aquatic environment in most countries worldwide. Agriculture is the dominant diffuse source, and the challenge of quantifying nutrient cycling from the field to coastal waters calls for the ability to describe and model the various physical and biogeochemical processes governing nutrient transport. Several nutrient models have been developed worldwide in an attempt to describe and quantify nutrient transfers from fields to the aquatic environment.[10]

Study Case: Rönne River – Sweden

Background

The trend in Europe is not significantly different *per se* from the global trend in terms of fertilizer use and hence water quality problems associated with such agricultural activities. Although point sources such as sewage discharges may contribute significantly to nutrient enrichment in some regions of the continent, diffuse sources, particularly agriculture, are still the major contributors.[12] The greater intensification of agriculture and higher productivity during the past 50 years has result in a significant increase in fertilizer use, particularly organic nitrogen use.[3][15] According to European Commission report, mineral nitrogen consumption in EU which was less than 1Mt in 1945 has considerably risen to a peak of over 11MT in 1985.[3] Similarly, intensive animal husbandry increased during the same period, contributing to a greater overall nitrogen load through manure. Nitrogen "pressure" from animal husbandry such as cows, pigs, poultry, and sheep on agricultural soils is approximately 8Mt per year based on 1997 data.[3]

The greater density of livestock population and manure applications has resulted not only in the direct deposition of nutrients into the ecosystem but also produced a strong volatilization of ammonia to the atmosphere and deposition back on soils and waters with values up to 50-60 kg N/ha/yr. Categorically, about 50% of the nearly 20Mt annual N input to EU agricultural soils comes from mineral fertilizer while the remaining 50% is attributed to air deposition, biological fixation and livestock manure spreading. According to the European Commission report (2002), the intensified agricultural activity has also resulted in the reduction of permanent grassland and disappearance of wetland areas, which could serve as buffer and sink zones for nutrients. This has subsequently caused increased erosion, high rate of run-off and more rapid flow of nutrients to the aquatic ecosystem and groundwater[3], with the most important consequences of surface water eutrophication and possible health effects. The increasing concentration of nitrate in drinking water and eutrophication of surface waters with adverse environmental and health effects triggered growing public concern, which has prompted the European Union for action to improve water quality since the 1970s. During the last two decades, new agricultural policies and environmental regulations have been enforced in many European countries with the objective of reducing agricultural diffuse source pollution, improve water quality and protect the aquatic habitat from eutrophication.[8] Limits have been established to agricultural fertilizer inputs in order to control water pollution caused by nitrates from agricultural sources through the nitrate directive issued in 1991.[5] A similar directive was issued concerning urban waste water treatment in the same year to tackle the problem of nitrate pollution. However, despite the EU Nitrate Directives and the increased use of agri-environmental measures, the agricultural sector remains the main source of diffuse pollutants without as much progress. Nitrogen surpluses from agriculture are rather constant and hence levels in rivers are still as high as they were in the early 1990s.[6][7][4]

In Sweden, similar to most EU countries, the load of nutrients on streams, lakes and coastal waters has increased dramatically after World War II mainly due to the intensification of agricultural activities. This has caused the reduction of wetland areas by 90% thereby stimulating the growth of eutrophication in surface waters and reduction of biological diversity. Nitrogen input to agricultural soils from livestock manure was estimated between 40 to 50 kg N/ha/yr in 1997, while the total nitrogen pressure including mineral fertilizer use, livestock manure spreading, atmospheric deposition and biological fixation was between 100 – 125 kgN/ha/yr.[3] Average N atmospheric deposition in Southern Sweden, including the study, has been estimated between 5 and 10 kg N/ha/yr[1][3]. The proposed research area, the Rönne River basin, known to have been suffering from such elevated nutrient loadings and subsequent eutrophication of surface waters within the watershed.[1] The problem has been persistent that raised a growing concern, due to: firstly, its socioeconomic and ecological importance in that it contains lakes such as Ringsjön, which is frequently affected by algal blooms. In addition to being a habitat for aquatic ecosystem, Lake Ringsjön is currently used as a spare drinking water source for people inhabiting the area. Secondly, the catchment drains through the Rönne River to the coastal water of the North Sea. Coastal waters, in general, are ecologically sensitive from inland pollution sources and in particular the coastal water of the North Sea is a subject of major concern in the EU that led the current study area to have been designated as “nitrate vulnerable zone”. As a result, the Rönne River basin has been used as a pilot catchment area for eutrophication control[1] using an integrated river basin management approach.

The study area, Rönne River basin, is the second largest catchment located in the county of Skåne, the most southern region of Sweden. The watershed is elongated in the southeast-northwest direction from 55.8ºLat/13.2ºLon at the southern east end to 56.4ºLat/12.0ºLon at the northern west end. Location map of the study area is shown in figure 1.

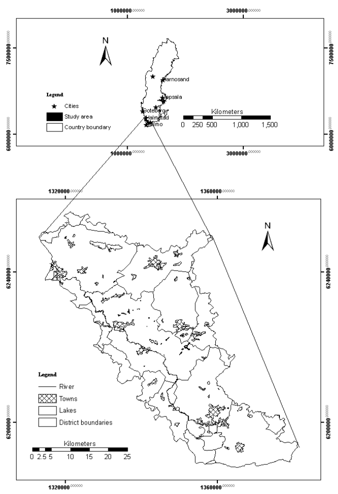


Figure 1. Location of Rönne River basin in Southern Sweden

The basin covers a catchment area of 1897 km^2 being drained by the Rönne River in the northwest direction where it finally enters the North Sea near the city of Ängelhom. Figure 2 shows a panoramic view of the Rönne River as it crosses agricultural fields (left) and enters the North Sea (right).[14]

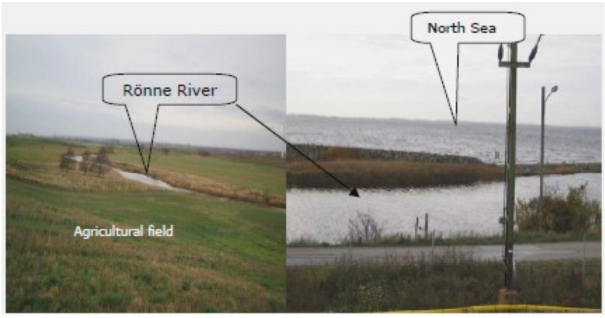


Figure 2: A panoramic view of Rönne River with respect to agricultural lands and The North Sea.

The area is relatively densely populated as compared to other parts of Sweden, a total of nearly 100000 people living in the catchment area out of which about 70000 or 70% are residents of urban areas.[14] The region has a history of elevated nutrient content due to mainly intensified agricultural activity resulting in nutrient leakage. Lake Ringsjön, which is currently serving as a spare drinking water source, has been suffering from such high nutrient loadings and frequently affected by eutrophication. Consequently, construction and creation of wetlands, ponds and buffer zones have been underway to complement improving water quality, reduce nutrient transport and increase biodiversity in intensively cultivated farmland. A map indicating land uses is shown in Figure 3.

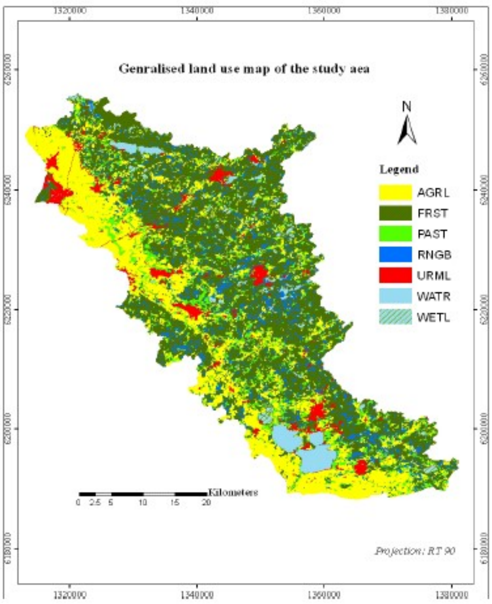


Figure 3: Generalized land use map used as model input.

From de report made for the EUROHARP Project Nº19[2] the catchment dominant land use was mostly arable land and mixed forest, with an extension of 617.6 km^2, the average amount of chemical fertilizers applied in 1999 was: 103 kg N/ha – 14 kg P/ha and animal manure: 42 kg N/ha – 8 kg P/ha.

Analysis of Nutrient Pressures

The point sources included in the model of the Rönne River basin were waste water treatment plants (WWTP), discharges from industrial plants, discharges from freshwater fish farms and discharges from scattered dwellings.

Source apportionment of Nutrient loads- Nitrogen models

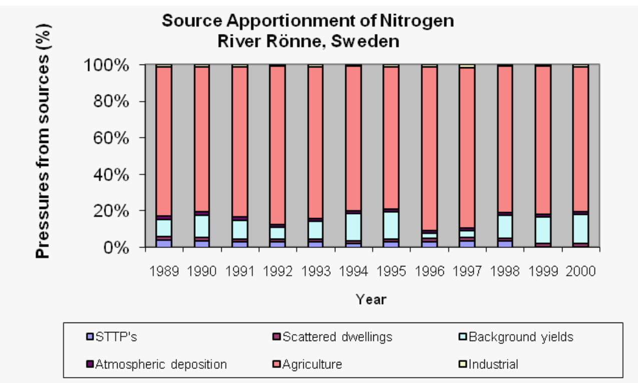
- Level of complexity: Low (Balance approach based on measurements)

Transport measured in a stream (T_V) = Emission from point sources (E_P) + emission from agricultural areas (E_L) + background emissions from all areas except water areas (E_B) + atmospheric deposition on surface water (A_O) – retention in surface waters (R)

With known T_V, E_P, E_B, A_O and R we can estimate the unknown variable E_L from:

$$\boxed{\mathbf{E_L = T_V - E_P - E_B - A_O + R}}$$

In order to know the source apportionment of N and P loadings in the Rönne River catchment we calculate the percentage contribution from all sources for each year (from 1989 to year 2000), shown in Figures 4 and 5.



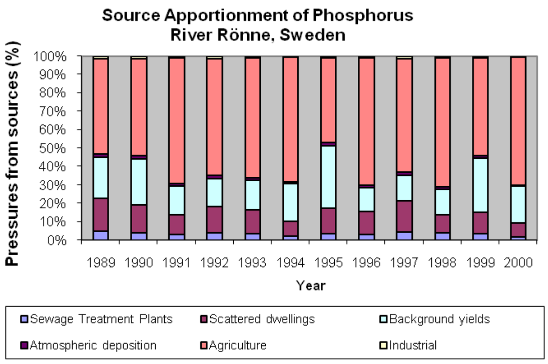


Figure 4. Source apportionment of N (%) Figure 5. Source apportionment of P (%)

From the above graphs it can be seen that the main source for both nutrient (N and P) comes from agricultural activities, for nitrogen the second source in relative importance is the load from background yields and in the last place the inputs from WTTP´s. For the apportionment of phosphorus, after the contributions of agriculture which are more variable than in the N case, the scattered dwellings and the background loads are the second in porcentual importance, nevertheless the inputs from WTTP´s are more relevant than in the N balance.

In order to calculate the gross total N and total P losses from agricultural land in this catchment, for the data given in the course we present the annual results in the following graphs (Figure 6 and 7).

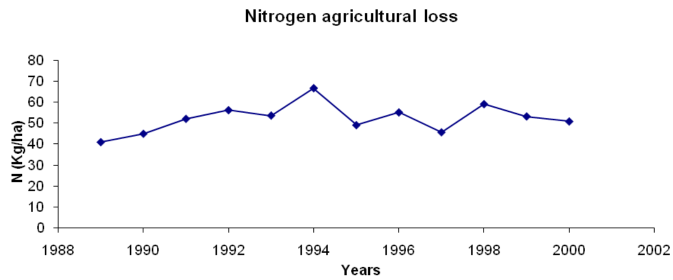


Figure 6. Nitrogen agricultural losses for the Rönne River catchment

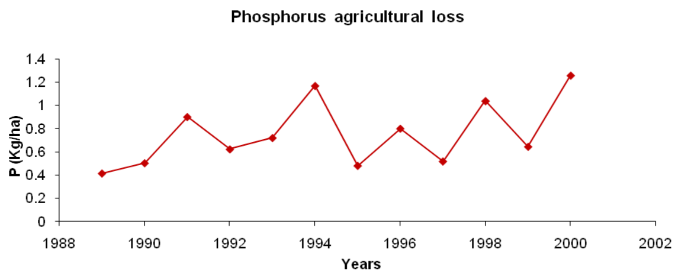


Figure 7. Phosphorus agricultural losses for the Rönne River catchment

As we compare the behaviour of the two nutrients in the time series (1989 – 2000), we can appreciate a higher variability in the P losses than the N ones, probably due to the different sources of each other and the influence of the precipitation in the point discharges into the river, among other causes, as the mayor pool of phosphorus is kept in soils and sediments.

Results for the N- modeling (Low complexity), with different scenarios of temperature, precipitation, surplus and land use (% agriculture) for 2 soil types (sandy and loamy) are shown in Figures 8 to 15.

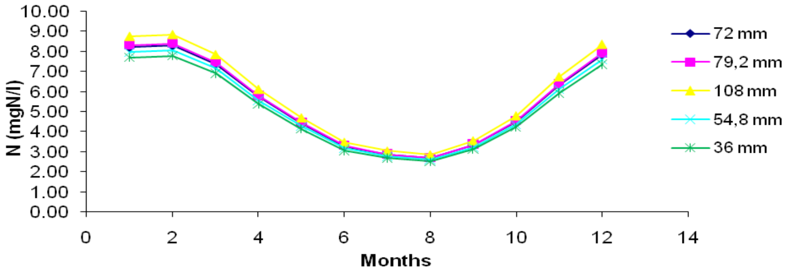


Figure 8. N losses & precipitation. Sandy soil

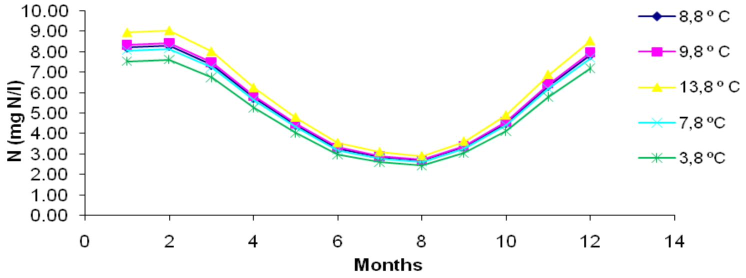


Figure 9. N losses & temperature. Sandy soil

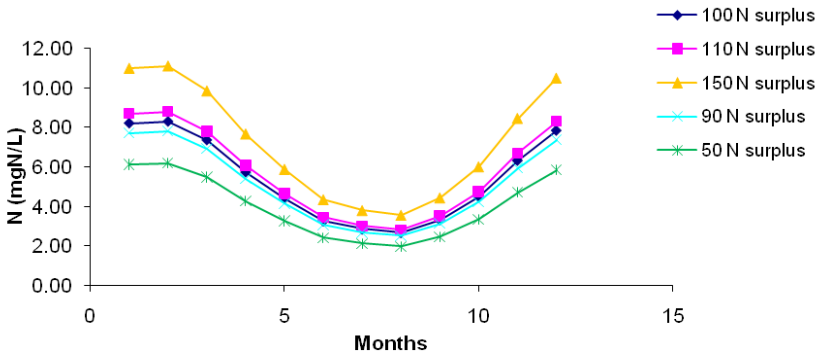


Figure 10. N losses &. N surplus. Sandy soil.

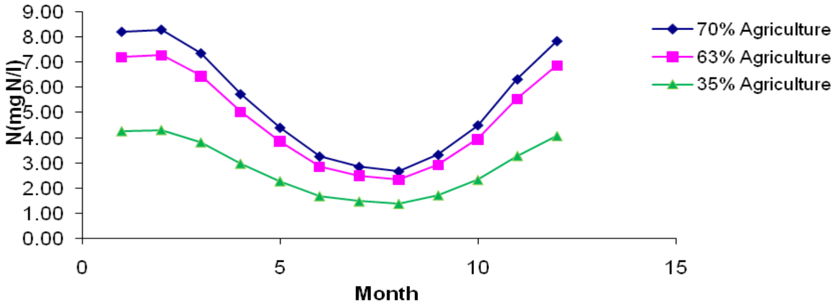


Figure 11. N losses & % agricultural land. Sandy soil

For sandy soils N losses are almost independent from temperature and precipitation changes, however, they have a high positive correlation with the N surplus and the proportion of agricultural land.

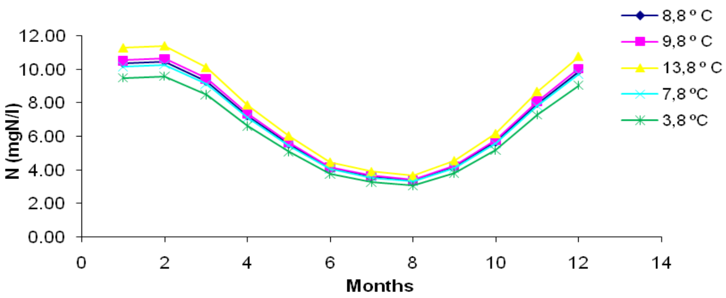


Figure 12. N losses & temperature. Loamy Soil.

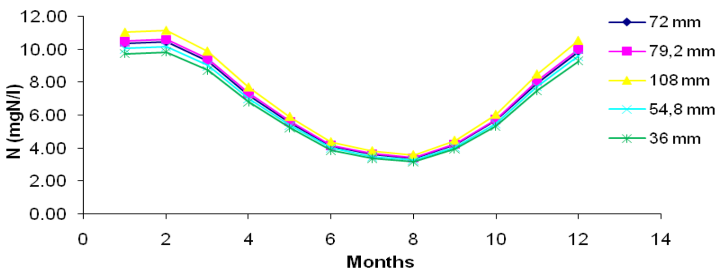


Figure 13. N losses & precipitation. Loamy soil.

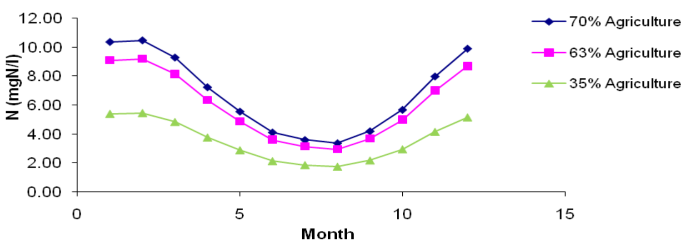


Figure 14. N losses & % agricultural land. Loamy soil.

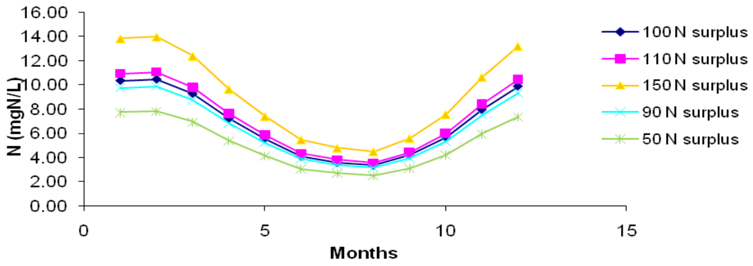


Figure 15. N losses & N surplus. Loamy soil.

We can observe the same trend in loamy soil than the sandy one, with dependence of the N surplus and the proportion of agricultural land.

In the following figure (Fig. 16) we compare the average N loss in sandy and loamy soils for each month from the monitored period.

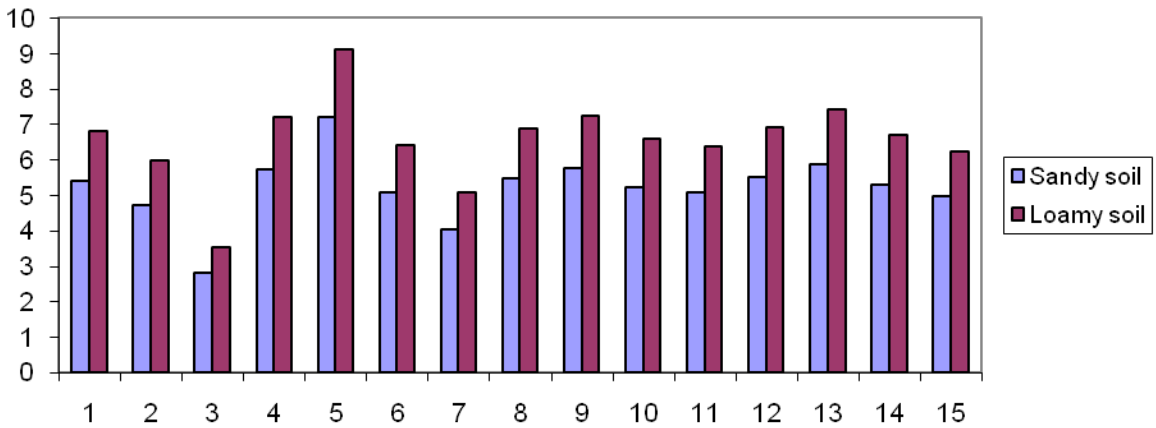


Figure 16. N losses (mg/L) in time (months) for sandy & loamy soil.

We can conclude that in loamy soil N losses are higher than sandy soil for all the monitored months, this could be explained by the content of clay in these soils and its behavior related with its physicochemical structure.

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Chapter III

Mitigation options in catchments –functioning of wetlands and buffer strips

Natalie Corrales Martin

Introduction

During the past 100 years, increased population density and intensified agriculture have led to enhanced nutrient input and deteriorated water quality in many water courses.[14] In the case of nitrogen (N), agriculture is the main source of water pollution. Total nitrogen (TN) loss through runoff in areas with intensive agriculture is more than 5, and often more than 10 times higher, than in forested areas. For phosphorus (P), eventhough the contribution of households and industries is still important, in many countries the contribution of urban wastewater, industrial effluents, and phosphate in detergents, has been reduced more than 50 % since 1980. As a result, the relative contribution of intensive agriculture to the nutrient loads to surface waters has increased up to 50 % of the total P loss in many parts of Europe. Moreover, in the Nordic countries nutrient losses seem to be related with the intensity of agriculture practice, given that in the five areas with highest amounts of nutrient loss per ha about 40-50 % of the area is cultivated.[10]Furthermore, the enrichment of the agricultural land and its potential nutrients losses to surface water are dependent on the farming system (farm area and the production type e.g. dairy, pig, poultry, and arable farms) within the catchment.

The amount of mobile nutrients depends on: the quantity and form of applied nutrients, the uptake of nutrients by the crop during the year, the current nutrient status of the soil, weather conditions, water availability, and the internal absorption capacity of the soil Besides, precipitation and upward seepage are the main sources that force water flow through and over the soil and transport nutrients within the field or out of the field to surrounding fields or surface waters. Moreover, not all these nutrients will be delivered to the surface waters and this will depend on the connectivity of the field with the receiving surface water and the buffering capacity of the landscape, such as adsorption or absorption by the soil, of mainly of P, and denitrification.[10]

The consequences of nutrients input on aquatic systems (lake, brook, river, estuary, coastal areas) in terms of trophic structure, physical and chemical conditions, and use of water triggers an urgent need for mitigation measures to restore and prevent damage. Mitigation measures must intend to control and 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.

Mitigation options

Nutrient supply

Balance

The behaviour of P in soils differs completely from nitrate, and a remarkable difference is that NO_3-N is more mobile than P given that the first is not sorbed on the matrix soil. Only a very small part of the total P is present under the form of $H_2PO_4^-$ or HPO_4^{-2}. Instead P is sorbed at the surface of clay minerals or organic matter bound to Fe- and Al-hydroxides, and Ca^{2+} or $CaCO^3$. Thus, the recommendations for soil with very low P content will be to apply more than the output by the crop, while at a high level will be to apply less than the output. For N the recommendation based on soil testing is much more complicated because, beside the N in soil organic matter, there is no real accumulation given that its inorganic form is highly mobile and leaches easily. Only about 1-3% of N bound in organic forms may become mineralized and available to crop uptake or leaching within a growth period.[4]

In other to carry up a proper management of nutrients in agricultural systems, it is necessary to count on a reliable estimate of the nutrient content and release of the applied animal manure or other organic byproducts. Then the amount of P applied with organic sources can be subtracted from the calculated needed P fertilization. The efficiency coefficient for P in manure is assumed to be 100%, while for N it could reach 70% depending on the C/N ratio, the way of application, pH, and climatic conditions. 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.[11]

Delivery

A fertilization practice adapted to the needs of the crop in terms of amounts and a better placement by row or band application has proven to minimize N losses. Positive results can be expected especially for crops with large distances between the rows and for those which are earthed up soon after fertilization, due to a better nutrient availability with lower residual mineral N at harvest and lower leaching losses. In addition, NH_3 volatilization is reduced to almost nil since the fertilizers are incorporated to a depth of a few cm, and nitrification of NH_4-N is delayed given that there is a high salt concentration nearby the fertilisers, which reduces possible losses by leaching because NH_4-N is adsorbed at clay and humus components. Moreover, 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.[11]

The chemical form of the nutrients in the fertilizer is also important, in terms of its nature (e.g. N in form of NO_3 or NH_4) and the dissolution rates (e.g. fast and slowly available N and P). In this line, gradual nutrient fertilization is a practice more and more frequently used by farmers. Lastly, it is also a good agricultural practice to avoid manure and fertilizer application before predicted high or long rainfall events.[11]

Agricultural practices

Crop type

The great differences in N and P dynamic in the soil, requires different management strategies to minimize the losses of the two nutrients from the field. The presence of high concentrations of potentially leachable nitrate in soil can be reduced by direct management strategies, through sowing crops with a high N uptake capacity. On the contrary, for P, whereas uptake reduces its solution concentration in the vicinity of the roots, the concentration is buffered by the large amount of P accumulated in the solid fraction. Thus, if leaching of dissolved P has to be lowered through crop management, it is only possible through P mining; this means that the P exported by the crops over several years should be significantly larger than the P input with fertilizers and manures.[9]

Catch crops

Perennial crops cover the soil permanently throughout the year with well-established root systems, taking up and immobilising nutrients. However, when a perennial crop like grass is terminated the nutrients are incorporated into the soil and become available though mineralisation processes.[9] Then alternative ways, such as catch crops, are needed in order to prevent nutrients, especially N, from leaching. A review of the literature, on the effect of catch crops, shows an average annual reduction of 48% in NO_3 leaching.[15] There are certain considerations when choosing the catch crop: 1) competition with the main crop for nutrients and light can be an important issue for under sown catch crops. 2) Crops with fast establishment after harvest of the main crop which can keep a large and fast nutrient uptake. 3) Fast and deep root development enabling nutrient uptake from deeper soil layers. 4) Cold tolerance of the catch crop is especially important in agricultural systems placed in cold climates, because during winter some of the immobilised N leaches from the plants and this is more accentuated on less cold tolerant types.[9]

Cover crops

Soil erosion and surface runoff are considered important transport pathways, especially for dissolved and particulate P. 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. Empirical evidence suggests that water erosion may decrease exponentially with increasing soil cover.[8] In Norway, erosion could be reduced by 5% by growing Italian Ryegrass as cover crops. Moreover, the effects of cover crops depend on the development of the crop during autumn; the more developed the crop the higher the reduction on soil erosion.[9]

However, the effect of cover crops on P losses is less significant than the effect on soil erosion, which may be partly explained by increased losses of dissolved P released from the cover crop due to freezing during winter.[9]

Direct drilling - Ploughing

A soil tillage system with direct drilling or shallow cultivation excludes inversion of the whole topsoil. Direct drilling means that the crop is sown in a single operation, this practice 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 particulate bound phosphorus (PP) in many european countries; such as from unstable, erodible clay loams, silty and clay soils under cereal cropping in Scandinavia. However, direct drilling increases the risk of dissolved reactive P (DRP) losses since more plant material is left on the soil surface. Furthermore, surface application of P fertilizer combined with no-till during consecutive years, also leads to P accumulation in the top layer of the soil. Thus, Addiscott and Thomas (2000) recommend interrupting periods of plough-less tillage with conventional ploughing in order to dilute P concentrations in the uppermost part of the soil and increase adsorption of DRP to soil particles. These measures would reduce DRP in horizontal and lateral water movements and minimize the impact on surface water given that DRP has a higher ecological impact than PP due to its higher bioavailability.[3]

Mitigation measures in the landscape

The heterogeneity of the landscape has often been considered as a state favourable for trapping diffuse pollutants emitted by farm fields, leading to the definition of the "buffer capacity of a catchment".[17] The buffer capacity is defined as the ratio between nutrient output from the sources and the amount delivered to the water bodies. It is mainly attributed to landscape structures, called 'buffers' which intercept, filter and retain water and associated nutrient flows, such as "wetlands" (WL) and "buffer strips" (BS). However, in this section also artificial "ponds" and "grass waterways" are included as landscape structures that contribute to the heterogeneity of the catchment and function as traps to pollutants. Nutrients may be variably stored without transformation or transformed within buffers, according to their nature, degree of biogeochemical reactivity and interactions with plants or micro-organisms.[7]

Ponds

Creating sediment ponds at the edges of the fields block the water flow rate and allow particles, and associated nutrients to settle, reducing suspended material and nutrient losses to the surface waters.[12] This measure will be effective in flat areas, but in hilly areas the effectiveness will depend on the amount of water flow and the amount of water that can be hold by the pond. Well-constructed ponds can potentially remove 65-75 % of the sediment and 25-33 % of total P entering the pond. 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. A serious disadvantage of this measure is that the field will be wet for a longer period and the farmers will be prevented from cultivating in the mean time. [3]

Grass waterways

Grass waterways are channels of grass within fields that are constructed in order to get a more condensed and controlled water flow with low velocities. The vegetation in the waterways acts 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, and vegetation. Some limitations of grass waterways as mitigation measures are that they need specific additional treatment with farm equipment and there is a loss of land for crop production.[3]

Wetlands functioning

WLs contribute up to 40% of the renewable ecosystem services, even though they only cover 1.5% of the earth surface.[5] In this section it will be discussed their ability to counteract floodings; improve water quality through the removal of N, P and carbon (C); support biodiversity and their vulnerability toward invasive species; and efficiency as C sink. Cowardi *et al.* (1979) hold that WLs constitute transitional systems located between terrestrial and aquatic ecosystems, and they present three main characteristics. 1) The soil is saturated with water or covered by a layer of water throughout a part of the year. 2) The type of soil differs from that of the adjacent upland area. 3) These sites are covered by a type of vegetation adapted to the reductive characteristics of the soil, called hydrophytes and macrophytes.[6]

In a study case carried out in the center-west region of US, WLs were capable of retaining and eventually decreasing the impact of water downstream. The evidence suggests that when 10% to 20% of the WL is kept on the landscape, the risks for flooding and eutrophication decrease. The ability of small, widely distributed WLs to abate flooding depends on the amount of storage relative to the volume of floodwater, as well as the WT capacity for evapo-transpiration and infiltration.[18]

Water quality declines when runoff increases the delivery of sediment and nutrients (N and P) to streams, lakes, WLs, and estuaries and carries toxic materials downstream. Kadlec y Knight (1996) showed clear evidence that WLs can be used to treat urban wastewater, but this system is not readily transferable to agricultural lands because agricultural WLs receive more pulsed flows, less organic matter, and more sediment than urban wastewater WLs.[18] However, positive results were obtained by Vellidis *et al*. who studied the effect of a forested riparian WL on water quality in the coastal plain of Giorgia. This dairy WL receives nutrient inputs through manure application, based on the use and removal of N, bringing up considerable concerns over P transport through runoff. It was observed that even if TN removal (mainly through denitrification) was remarkable, the retention for TP was higher (66% for TP and 59% for TN). These results indicate that a forested riparian WL is efficient in the retention of P, and its presence in areas where manure application to land is practiced, they can perform as an efficient tool to reduce its input to water courses.[16] Besides, the hydrological conditions in WLs is a major aspect for its correct functioning, it is necessary to increase residence time, decrease spatial variability of the fluxes and decrease flow velocity within the WLs. High heterogeneity of water flow within the WLs may avoid the site where denitrification could be higher.

WLs have a great potential for supporting biodiversity, the presence of water, high plant productivity, and other habitat qualities attracts high numbers of animals and animal species, many of which depend entirely on WLs. The Pantanal, which spans parts of Brazil, Bolivia, and Paraguay, supports 260 species of fish, 650 species of birds, and a high concentration of large animals. However, in terms of vascular plant their diversity is especially high in WLs that do not receive much surface water runoff.[3] WLs fed by surface water from agricultural and urbanized watersheds tend to have many invasive species. The capture of great loads of sediments and high inputs of nutrients promote the formation of monocultures of Typha and similar aggressive plants which displace native species.[18] Various authors note that lowlying lands support high proportions of exotic species. This position in the landscape interacts with dispersal routes and disturbances to facilitate plant establishment. If dispersal routes are a strong factor, then it is understandable that riparian habitats are especially prone to invasion, as claimed by Stohlgren *et al.* and Tickner *et al.*. Moreover riparian WLs are subject to flood pulses and mechanical disturbance (bare soil) which facilitate invasions, further resulting in a decrease on the quantity and quality of native plant species.[19] Therefore complex feedback mechanisms should be considered when designing WLs to maximize biodiversity. For example, while cattails might remove the most nutrients, they can replace native species in the process. [18] Furthermore WLs that become dominated by invasive plants tend to support fewer native animal species, and ultimately more invasive animals will likely be attracted. [19]

Although WLs store vast quantities of C in vegetation and especially in their soils, they also contribute more than 10% of the annual global emissions of the greenhouse gas methane (CH_4) and can also be a significant source of CO_2 under some conditions. To what degree WLs function as net sinks or sources of greenhouse gases appears to depend on interactions involving the physical conditions in the soil, microbial processes, and vegetation characteristics. Smith et al. conclude that CO_2 release from the soil increases exponentially with increasing temperature and decreases both with soil saturation and drought. Thereby when natural WLs are drained for cultivation or peat mining, large quantities of stored organic C decompose and are lost to the atmosphere as CO_2. In addition, methane (CH_4), a potent greenhouse gas with a warming potential 23 times greater than CO_2, is formed in soils under anaerobic conditions and low-redox potential, resulting from prolonged water logging which occur in both natural and managed WLs.[20]

A great variety of WLs types exist, and they are determined by their composition, maturity and location. Moreover becoming a source or a sink of C will be partly conditioned by those characteristics. In this line, Glatzel et al. found that the high decomposability of new peat in a restored peatland resulted in very slow C sequestration and net emissions of both CO_2 and CH_4 in the short term.[20] In contrast, coastal WLs may offer excellent potential for C sequestration.[13] Lastly, forested WLs also sequester C effectively, hence the restoration (from cultivation) of forested WLs in floodplains and its hydrology would very likely contribute to C sequestration and indeed to biodiversity support, water quality improvement, and flood abatement functions.[20]

Buffer strips functioning

Riparian (BS) can contribute to water preservation at local level, as a selective barrier to surface runoff, erosion, pollutants and organic matter export. In this section it will be discussed BS efficiency in terms of its shape and its removal capacity for N (nitrate, ammonium, and organic N), sediments, P (dissolved, particulate and total), dissolved organic carbon (DOC), pesticides, herbicides and bacteria. There are five general ways through which (BS) reduce non point source water pollution from cropland: 1) reducing surface runoff from fields, 2) filtering surface runoff from fields, 3) filtering groundwater runoff from fields, 4) reducing riverbank erosion, and 5) filtering pollutants from stream water. [16] Furthermore, BS can also have an effect at hydrological levels, decreasing the peak flow (through infiltration) contributing water to the groundwater recharge or through evapo-transpiration. The infiltration, as a result of dense rooting brought about by perennial vegetation, is the main key to the slowing of flow velocity and further sedimentation, which is the major physical process occurring within BSs. [7]

In terms of BS efficiency for N, in Marano, Italy, Borin and Bigon reported a 90% reduction in NO_3, leaving a 5 m grass buffer with an additional line of trees. Peterjohn and Correll found similarly high rates of N removal in Maryland, with large reductions in the N content of overland flow: a 79% reduction in NO_3, 73% reduction in ammonium, and 62% reduction in organic N. Combining results for both surface runoff and groundwater, the BS retained 89% of the N entering the system, which is much higher than the 8% retained by the same area of cropland. However, in relation to BS, literature shows results ranging from an increase of almost 20% in NO_3 to a decrease in N load of up to 99%, with a mean reduction of 35%.[15] Making comparisons between different studies is difficult because of variation in the buffer width, species composition, buffer area to field area ratio, soil type, and runoff conditions, all of which influence the ability of the BS to remove pollutants.

BSs provide deposition opportunities for sediment, and can function as a trap even under concentrated flow conditions. Abu-Zreig et al. found that filter length rapidly increased the proportion of sediment trapped up to a length of 10 m, then this increase tails off. Furthermore, Syversen found that a forested buffer zone trapped significantly more particles than a grassed one.[15] Even though P removal is very closely related to sediment removal, BS are generally less effective at removing P than sediment, potentially because a large fraction of it is associated with fine clay, which is more difficult to sediment and retain. Hoffmann et al. (2009) have reviewed efficiencies of riparian buffers for TP retention and quantify the reduction of outputs as 41% to 93%.[7] Moreover, at a watershed scale, Reed and Carpenter found that the shape and continuity of the buffer was more closely related to the P retention than the length of the buffer.[15] However, the retention of dissolved reactive phosphorus (DRP) in riparian buffers is low, several studies found increases of over 50%, and in particular flooding has been observed to provoke its increment.[7][13][15] In comparisons between forested, grassed buffers, and mixed vegetation buffers, the cutting and removal of vegetation appeared to be the key difference between a reduction and increase in DRP; thereby being the leaching from decaying vegetation the most likely source.[15] It is important to notice that total suspended solids, N and P are subjected to different fates in the BS: suspended solids are trapped by the grass, N can be abated by plant absorption and microbial activity leading to denitrification and dissolved P can be uptaken by plants and microbes.[16]

There is some evidence for increases in dissolved organic C (DOC) reaching waterways where forested buffers are present. In a study in Maryland, Peterjohn and Correll found a 2.9 fold increase in DOC, and a second study in North Carolina found an increase in DOC in shallow and deep groundwater under forested BS.[15] Moreover, increased levels of DOC in groundwaters have a number of potential impacts; increase of denitrification and higher N_2O production, lower pH and under saturated conditions the methane (CH_4) production can increase.

BS retention capacity also applies to pesticides, herbicides and fecal bacteria. Lacas *et al.* and Arora *et al.* thorough a review about the effectiveness of BS for trapping pesticide runoff, presented that they intercepted between 13% and 100%, and 11% and 98%, respectively. Despite this wide range, in a majority of studies pesticide retention in BS is high. Moreover, BS can also reduce herbicides by 55% and 95% in the shallow water table, which will be directly dependent on its chemical properties. Lastly, Coyne *et al.* reported a 59% average reduction in fecal bacteria leaving an BS in a catchment with manure amended soils, and Young et al. reported a 70% reduction. On the contrary, there is also the potential for BS to become a reservoir for sediment bound bacteria.[15]

The effectiveness of the BS depends on many factors, including vegetation composition which can be composed of native vegetation intentionally left intact as well as re-established vegetative buffers, soil type, subsurface drainage characteristics, temperature, slope, relative sizes of the filter strips and runoff areas, topography, activities on the cropped land, volume of runoff, and the nutrient loading rates.[15] Even narrow buffers (~1 m) with appropriate vegetation and correct management have some potential for limiting pollution of surface and slightly wider buffers (>5 m) bring increasing nutrient retention and transformation benefits, as well as increasing the possibilities for taller vegetation such as trees, to create root stabilisation of banks and beneficial detritus input (leaves) to streams. The widest buffers (>20 m) bring benefits associated with habitat availability and potential zones for flood water storage.[7]

Runoff flow on flat landscapes tends to occur along narrow flow paths, so that only very small portions of a BS actually intercept runoff from the field edge. Vegetated swales that extend into the field along shallow gullies may prove more effective at retaining runoff P than a uniform width BS because of the increased contact between the vegetated soils in the swale and runoff. Moreover, the effectiveness of the BS inevitably varies with time, on two scales. Within a year, the BS probably functions to retain P both while under snow and later when there is active growth, in this way varying its relative effectiveness with season. On another time scale of perhaps decades, as the BS accumulates P, it may eventually become a source, unless specifically managed to avoid this problem. Removal of vegetation seems the only effective management practice to remove P from the VBS.[13]

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Chapter IV

Discussion

Human population growth has placed ever-increasing demands on both aquatic and terrestrial ecosystems, and one-third to one-half of the lands' surface has already been transformed. We have dramatically changed the environment by land clearing, agriculture intensification, forestry, animal husbandry and urbanization, altering hydrological and biogeochemical cycles of mayor elements and no less important by changing global climate. It is important to point out that agriculture practices already cover 11 percent of the world`s land surface for crop production. It also makes use of 70 percent of all water withdrawn from aquifers, streams and lakes. Moreover, land transformation represents the primary driving force in the loss of natural habitats, increasing extinction rates, changing the trophic structure and composition of native species.

Given the great input of nutrients to the environment associated to diffuse sources, particularly from agricultural practices, it becomes crucial to study and analyze closely the nutrient balance for every farm system. In order to elaborate a plan of action with effective mitigation measures to minimize nutrient losses from the land, it is important to be aware of the different inputs and outputs of nutrients and their dynamic in the interconnected ecosystems.
From one side, the use of nutrient modelling of different complexity is essential to quantify the net and gross balance from an ecosystem. In this report, some low level models for nitrogen were used in order

to know the main sources in a Sweden river catchment, and to analyze different climate scenarios and loadings. It is shown the relevance in these balances of the nutrient surplus in agricultural practices (both inorganic and organic fertilizers) and the increment of the arable land areas in time. Modelling is essential for trade commodities intermediaries and farmers themselves, in order to develop sustainable agricultural practices from an agro-environmental point of view.

On the other hand, nitrogen and phosphorus mitigation measures should be planned jointly. Even though it is a common measure for both to lower the amount added to the fields, for other actions it should be present that both nutrients have distinct differences in their dynamic and cycles which make their responses antagonist in some cases; then pollution swapping can take place. As an example, while nitrogen losses as nitrate into aquatic systems can be minimized by denitrification processes which occur under anaerobic conditions, the consequent low redox potential releases phosphate bound to soil particles, then increasing dissolve phosphorus inputs into adjacent water bodies. This can result in nitrogen becoming the limiting nutrient, contrary to the common assumption that phosphorus limits productivity in aquatic systems.

Moreover, the elaboration of a plan of action should be ensured by a legal framework which compiles an economic, social and environmental point of view. This is an important consideration, since most of the measures will directly impact farmers and landowners whose property is an economic mean and in most cases are already carrying on productive activities. The outcomes from modeling in terms of nutrient balances from field to catchment scales and location of hot spots for nutrient losses is very important data for policy makers; given that it helps the process of identification of vulnerable areas, the creation of management policies and the design of an appropriate legal framework. Moreover increasing taxes on fertilizers has proven to be effective in some countries to control the amount and frequency applied by farmers. Lastly, every mitigation plan must come along with a monitoring program which documents the resulting impact of the actions taken on the field. It is imperative to include precipitation and flow rates in order to know nutrients loadings referred to a given water flow, this is very important to identify loadings referred to seasonal dependent flows (which includes run off, evapotranspiration, evaporation, infiltration, and precipitation patterns). Most of the articles cited in this report presented data about nutrient balances in field experiments, but the lack of information on fluxes, could eventually lead to mistaken or incomplete conclusions.

Today, modeling is of great importance as predicting the impacts of global climate change has become a priority. In this report, diverse climate scenarios with different temperature and precipitation conditions were tried in the less complex model used, but the expected changes could not be appreciated, due to a lack of data in relation to the required ones for a complete trial (as it is needed: precipitations, topography, soil types, vegetal composition, and various other parameters). The current increasing global mean temperature and alterations in the hydrological cycle (patterns of precipitation and frequency of extreme events) have marked consequences on global climate with important impacts on nutrient cycles. Higher temperature will increase microbial activity, thereby denitrification also will increase, and primary production will be enhanced as well as toxic cyanobacteria blooms. In addition, more precipitation scenarios will increase runoff with a higher energy flow, which will transport a higher amount of nutrients to the adjacent aquatic systems. Increasing runoff will impact especially on the amount of sediments and phosphorus losses from agricultural systems. Nevertheless, if there is a high content of organic matter on the soil, it will also result in a higher transport of nitrogen. However, in terms of nitrogen the loss will generally happens through a higher leaching rate into the soil, and the final input into aquatic systems will depend on the inter connectivity between the terrestrial and the aquatic environment; proximity to surface water and access to groundwater.

Furthermore, it is interesting to notice that many mitigation measures to minimize nutrient losses have undesired side effects; such as, release of green house gases (CH_4, NO, N_2O, CO_2), increasing the sources of nutrients (e.g. wetlands and buffer strips that release dissolved phosphorus and nitrate), offer an opportunity for invasive species to spread, all of which can happen in buffer strips and wetlands. The last two side effects should be evaluated for each case, but generally they could be managed through non selective or selective harvesting. However, the release of green house gases, in most cases, seems to be very difficult to control. Thus, it would be very useful to implement a monitoring program which includes an estimation of green house gasses release, in this way a complete evaluation on the impact, of mitigation measures for nutrient management, on the environment can be accomplish.

Conclusions

1) Food security is a hot topic worldwide, faced with an increasingly population growth. Changes in land use for agriculture practices and livestock are rising the over-pressure on natural resources and environment functioning.

2) Improvements on agriculture yield production will have to come with sustainable practices which make effective use of the land and water resources without more alterations over the ecosystems.

3) Mitigation measures can also have undesired impacts on the environment in terms of becoming sources of greenhouses gases and exporting nutrients to adjacent ecosystems; thus they should be managed to prevent nutrient losses and monitored in order to evaluate its negative effects.

4) A legal framework together with a proper monitoring program, and fluid communication with farmers and landowners, is essential to accomplish a reduction in nutrient losses. This can be reinforced by taxes to fertilizers and economical incentives to adopt sustainable agricultural practices such as: creating and/or maintaining wetlands and/or buffer strips.

5) Modeling is a very useful tool and back up which conveys in a clear and solid manner, the situation around nutrient losses from terrestrial to aquatic systems, and a very important tool to prevent increasing loadings of nutrients with harmful consequences worldwide.