Nutrient Cycling, Modelling and Management from Field to Catchment Scale

Final Report

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Content

Abstract

Agricultural activities generate many pressures on the environment. Phosphorus and nitrogen leaching to aquatic ecosystems has many negative consequences, for example enhancing eutrophication and leading to water quality losses. Here we review world tendencies in agricultural systems and nutrient concentration in rivers from different countries. We also review some useful tools for nutrient management as well as the characteristics and efficiency of two landscape management strategies. It was noticed that the intensity of agriculture is rising. However, trends are not equally distributed. In many developed countries several management strategies have been successful at reducing the amounts of agricultural land, fertilizers consumption and nutrient losses to water bodies, while they are scarce or inexistent in developing countries. As the world population is growing and also the demand for food, to attempt to reach a sustainable development, is urgent to start designing and applying environmental policies at a regional scale. To be effective, these policies must be based on scientific knowledge and take into account each country's environmental and socio-economical situation.

Chapter I:

Introduction

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1.1-Introduction

Nutrient leaching (most importantly nitrogen and phosphorous) from terrestrial ecosystems to water bodies within a basin is a natural phenomenon, which represents important nutrient subsidies of key importance for aquatic biota (Polis *et al.*, 1997). However, nutrient leaching has increased substantially during the last century, and most importantly during the last 40 decades mainly as a consequence of industrialization and agriculture intensification with the consequent use of artificial fertilizers and manure to satisfy the growing demand for food production at a worldwide level (Schoumans *et al.*, 2011). Nutrient losses from land to water can be classified according to its mechanism in point and diffuse leaching sources. Point nutrient sources are those in which all the residuals effluents (with very high nutrient concentrations) of an industry or activity are channelized to a water body, leaching on a much localized way (Hogan, 2010). Diffuse sources are those in which nutrients are lost from the terrestrial systems under dissolved and particulate forms via soil erosion, or lixiviation via surface water runoff or underground water flow (Kronvang *et al.*, 2007; Hogan, 2010).

The loss of nitrogen and phosphorous from point industrial sources and diffuse leaching from agricultural lands cause an increase in primary producers biomass (phytoplankton, periphyton and aquatic macrophytes) in superficial water bodies, process well known as eutrophication (Smith *et al.*, 1999; Lampert & Sommer, 2007). Eutrophication process promote several losses of ecosystem services provided by aquatic ecosystems, such as the degradation of water quality by increasing the frequency and intensity of occurrence of toxic cyanobacteria blooms (most importantly on lentic systems), increased turbidity and oxygen reduction that could result in mortality events for benthic fauna, all of this is known as ecological degradation of water bodies (Kronvang *et al.*, 2008). Additionally it often leads to loss of aesthetic value of aquatic systems (for example, by an increase in turbidity and odor problems) and reduction of fish production and mortality events at an extreme level (Carpenter *et al.*, 1997; Chorus & Bartram, 1999; Lampert & Sommer, 2007). Given this, different management policies and technologies have focused (and achieved to a great extent) on a reduction of nutrient pollution caused by point sources, however, agricultural land is exposed to continuous increase in diffuse nutrient leaching. This process was listed as one of the main sources of pollution to aquatic ecosystems all around the world (Zhang *et al.* 2010; Jeppesen *at al.* 2011). Phosphorous diffuse leaching from agriculture represents an increasing proportion of total P loses, accounting for more than the 50 % of total P loses at the present (Andersen & Kronvang, 2004). This nutrient leaches mainly via surface water runoff and soil erosion process in a particulate form (Kronvang *et al.* 2007). On the other hand, nitrogen is the nutrient applied in the highest concentrations in agriculture on the world constituting the most important diffuse source of nutrient pollution (Jeppesen *et al*., 2011). The main leaching mechanism for nitrogen is via lixiviation and underground water flow, but also leaches in an important way via lixiviation from surface water runoff (Kronvang *et al*., 2007)

Total content of phosphorous and nitrogen in agricultural soils is increasing continuously, as agricultural activity intensifies, this increases the vulnerability of soils to nutrient leaching via erosion and lixiviation (Kronvang *et al*., 2005). The existence of this pollution source represents one of the main causes for ecological degradation of water bodies in the world at the present (Follett, 2008; Kronvang *et al*., 2008; Jeppesen *et al*., 2011). For this reason, with the objective of achieve a good ecological condition of water bodies and keep the important ecosystem services they provide, there is a strong need to reduce the diffuse nutrient leaching (Andersen & Kronvang, 2004; Kronvang *et al*., 2005).

In this report we aim to review worldwide tendencies in nutrient leaching from terrestrial to aquatic systems, mostly caused by agricultural activities. Specifically we review world tendencies in the use of fertilizers and nutrient levels in rivers during the last decades as well as review the main tools applied to management (such as source appointment and nutrient modelling) and policies applied aiming to reduce diffuse nutrient pollution.

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Chapter II:

Trends in world agricultural systems and source apportionment of nutrients

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2.1- Introduction

Agriculture represents the main diffuse source of water bodies' pollution around the world (Zhang *et al.* 2010; Jeppesen *at al.* 2011). The application of inorganic industrial fertilizers started in the early 1900s, becoming massive at the middle of the twentieth century (Gellings & Parmenter, 2004), when the technology available made it possible to produce industrial amounts of nitrogen fertilizers, through a process that resembles biological fixation of nitrogen but requires huge quantities of energy. Before that expansion, farmers used to appeal to other kind of methodologies to keep or improve soil fertility, such as crop rotation, application of organic fertilizer (manure or crop residues) or taking profit from biological fixation (Tilman, 1998; Fink *et al*., 1999). The only way to expand agriculture was the transformation of more land into crops or grassland (Wood *et al.*, 2000), something that still happens in many countries around the world (Harris, 1996). However, the emergence of the use of inorganic fertilizers and the dependence on them, promoted a drastic change in global agricultural systems due to the intensification of exploitation of a given area (Tilman, 1998). It also allowed modern agriculture to expand its limits and increase yields in an unprecedented way (Khush, 1999), accompanying and enabling to some extent the fast increase in world population (Tilman, 1998; Tilman *et al*., 2002; Evenson *et al*., 2003). The so called 'green revolution', which started in the early sixties, lead to high crop yield increases and attempted to provide food for the rising world population. This process involved the new varieties of crops like wheat and rise, more responsive to fertilizers and resistant to pathogens, features that allow farmers to obtain greater quantities of food per Hectare (Khush, 1999; Evenson *et al.*, 2003).

During the last decade, soybean and other transgenic crops are expanding through world, mostly in developing countries, carrying with them the change in agricultural systems and the intensification in the use of inorganic fertilizers and agrochemicals (Paruelo *et al*., 2006; Arbeletche & Carballo, 2008; OECD/FAO, 2011). Livestock, now the largest user of agricultural land, has also suffered from intensification, given the fact that the demand for meat and animal products is increasing and animals are being fed in densely populated confinements (Naylor *et al.*, 2005)

Nowadays, humanity's survival and the functioning of global economy rely on agroecosystems and the goods and services that they provide (food, fiber and fuels, Wood *et al.*, 2000). However, such an increase in ecosystems exploitation is exceeding their biocapacity (Harris, 1996; Wood *et al.*, 2000). As a consequence, we are experiencing now the negative results of the degradation of natural resources. For example, soil erosion, loss of biodiversity and excess of nutrients in aquatic systems could promote problems like losses in productivity, losses in resistance to pathogens and eutrophication respectively (Wood *et al.*, 2000). Moreover, this situation is intensifying in the upcoming years, as well as the production capacity of lands to feed a growing population will decline (Harris, 1996). Attempting to give solutions to this problem represents a challenge to managers and policy makers (Khush, 1999).

To manage agroecosystems properly and attempt to prevent (or make less intense) the negative effects of agricultural activities on environment it is necessary to know their current situation and identify the points that need special attention (Vitousek *et al.*, 2009). To that extent, it is worthwhile to make a land use characterization (Wood *et al.*, 2000). Nevertheless, due to the fact that information only about land use could not be accounting for the potential effects of agricultural activities, analyze the trends in the consumption and application of fertilizers, would be an extra indicator of the degree of intensity of agriculture (Wood *et al.*, 2000). The information obtained through this type of approach, as well as the identification of the contribution of agricultural activities to water bodies' nutrients (via source apportionment) can be helpful in understanding the main pressures of these practices on the environment , and thus, directing management strategies (Kronvang *et al.*, 2003; Kronvang *et al.*, 2007).

This chapter aims at reviewing the main trends in world agricultural systems. Specifically we compare the trends in the proportion of agricultural land (arable and grassland), the livestock production and the use of fertilizers for different countries and the world. Additionally we attempt to recognize the main sources of nitrogen and phosphorus for Odense River in Denmark, applying performing a source apportionment.

2.2- Methodology

In order to analyze the trends in world agricultural systems we utilized the Organization for Economic Co-operation and Development environmental data compendium (OECD, 2008). This data set contains information and statistics about the agricultural activities that result in certain environmental degradation, such as transformation of land for agricultural purposes, irrigation, livestock and consumption of fertilizers. We compared information about those topics from Denmark, Japan, Mexico, New Zealand, US, Uruguay and the world during the period that goes from 1980 to 2005. In some cases, we also included information about EU and OECD countries as a whole. All data analyses were performed in Microsoft Excel and SigmaPlot version 11.1.

Particularly, we analyzed the tendency (fitting of a linear function) of changes in the amount of agricultural areas (km^2), considering arable and grasslands separately, the apparent consumption (Tonnes) of nitrogen and phosphorus synthetic fertilizers and the livestock (cattle and pigs). Additionally, we carried out an analysis of the gross balance of nitrogen (quantities of fertilizers divided by the areas of agricultural land) and phosphorus fertilizers for the countries mentioned above (including EU and OECD) but comparing the periods 1990-1992 and 2002-2004.

To know about the contribution (percentage) of different sources to nutrient loading in Odense river catchment (Denmark), we performed a source apportionment of net total nitrogen and phosphorus, with data for the period 1989-2000. This analysis assumes that the concentration of nutrients at a given monitoring point of a river is the sum of point and diffuse anthropogenic sources and natural background losses (Kronvang *et al*., 2003).

Based on that assumption, the diffuse nutrient loss from agricultural activities in a catchment could be estimated using the following equation:

$$[LO_{AGR} = L_{RIVER} - D_P - L_{OB} + R - LO_{AT} - LO_{SD}]$$

In that equation, L_{RIVER} represents the nutrient loading measures in rivers' water, the term D_P represents the point sources, L_{OB} is the background natural nutrient loss, R is the retention of nutrients into aquatic systems, LO_{AT} represents the atmospheric deposition and $LO_{SD\ the\ nutrient}$ loss due to scattered dwellings in the catchment (Kronvang *et al*., 2003).

2.3- Results

The amount of arable and grassland land generally diminishes, being Mexico the only exception to the observed trend in both cases. It also could be noted a difference in the extension of arable land between United States (with areas higher than 1,600,000 km^2) and the rest of the studies countries (Figure 2.1, Table 2.2) which present areas of arable land below 400,000 km^2. Grasslands occupy much higher areas than arable lands, ranging from less than 500,000 km^2 in Japan, Denmark and New Zealand to values around 1,000,000 km2 and 2,500,000 km2 in Mexico and the United States respectively (Figure 2.1)

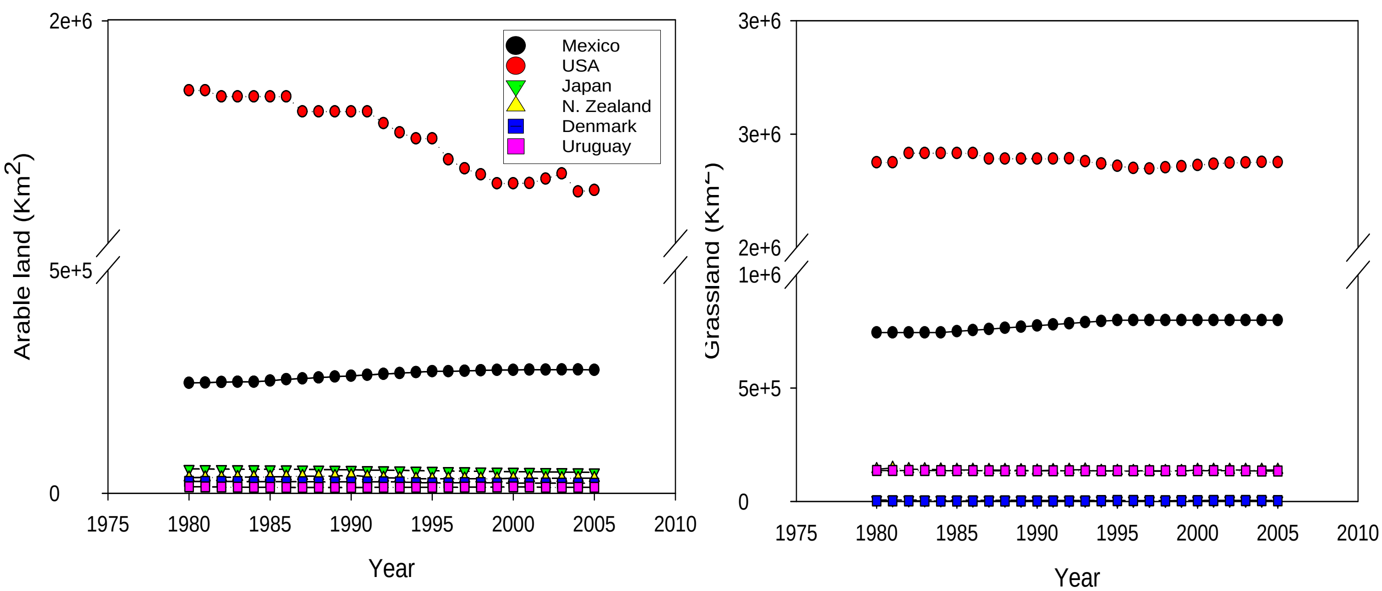

Figure 2.1. Tendencies in agricultural land use change for the period 1980-2005 in different countries around the world. Arable land (left,km^2); B) Grassland (right, km^2).

Livestock, more specifically cattle and pig farming showed a variable response during the period 1985-2005 (Figure 2.2). At a global scale, both pigs and cattle tended to increase.

Among the countries with the same tendency were Uruguay, New Zealand and the United States, being the last one the country with the highest amounts of livestock. While in Denmark and Japan the quantity of both pigs and cattle declined, in Mexico, only pigs have shown this pattern (Figure 2.2, Table 2.1).

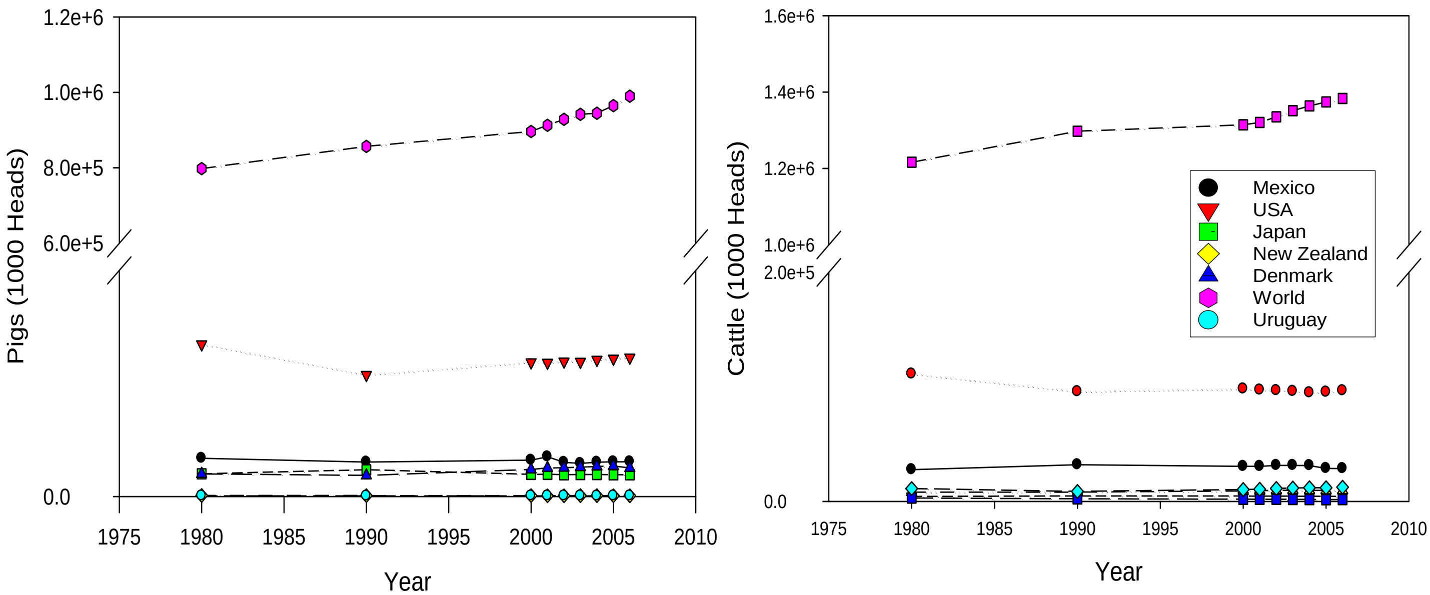

Figure 2.2. Trends in livestock during the period 1985-2005 for the world and different countries. Pigs (left, 1000 Heads); Cattle (right, 1000 Heads).

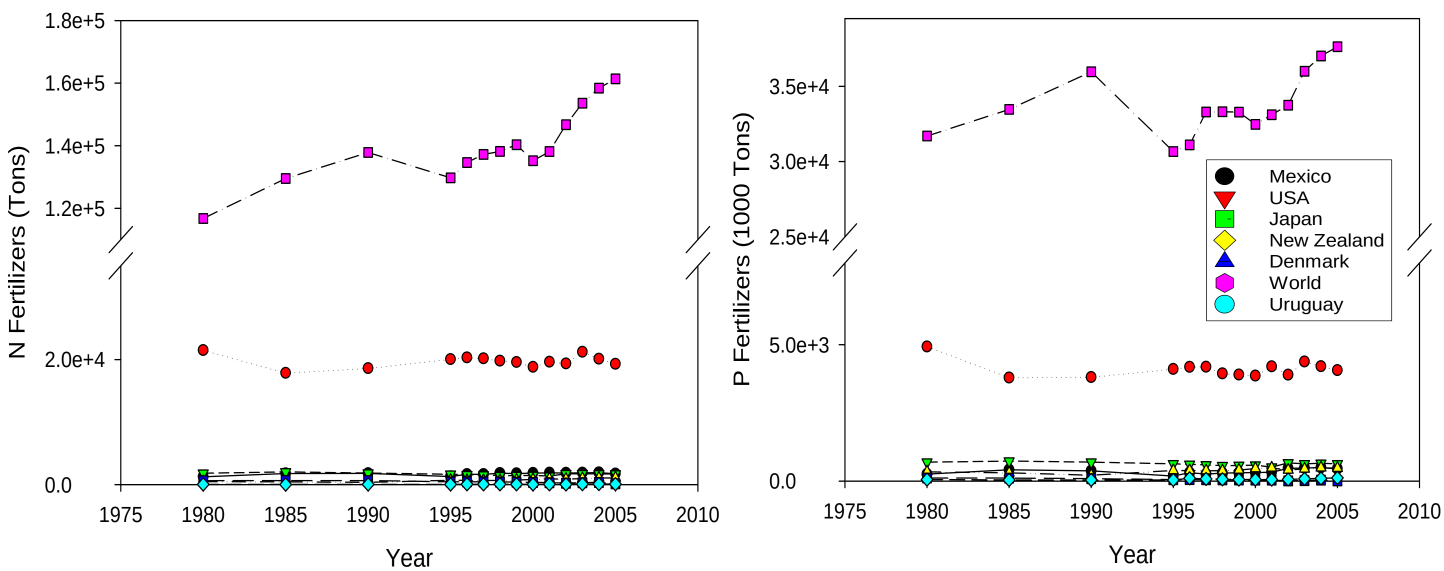

Figure 2.3. Trends in apparent consumption of nitrogen (left) and phosphorus (right) commercial fertilizers around the world and different countries. The period 1985-2005 is considered.

The amount of inorganic fertilizers (containing both nitrogen and phosphorus) consumed, as well as livestock production, when analyzed globally, tended to rise over time. Once again, United States was the country utilizing the highest quantities of these nutrients in agriculture. In this case, although the linear fit was not significant, it was noticed a tendency to increase for nitrogen and to decrease for phosphorus. Mexico, New Zealand and Uruguay showed increments in fertilizers consumption. On the other hand, Denmark and Japan slowly diminished the amount of nitrogen and phosphorus utilized during the period (Figure 2.3, Table 2.1).

Table 2.1. Linear fit equations for the six variables assessed: Arable land (km^2), Grassland (Km^2), Pigs (1000 Heads), Cattle (1000 Heads), Nitrogen and Phosphorus inorganic fertilizers (Tons). M: Mexico; US: United States; J: Japan; NZ: New Zealand; D: Denmark; W: World; U: Uruguay. N: no significant fitting and * significant fitting at a p-level=0.005.

	Arable land	Grassland	Pigs	Cattle	N fertilizers	P fertilizers
M	1398x-2521282*	2764x-4729768*	-46x+108736 N	44x-57696 N	17x-33419*	5x-10862 N
US	-64579x+129791363*	-1816x+6001254*	6614x-12306722*	5710x-10087737*	3x+13629 N	-11x+27953 N
J	-339x+726701*	-78x+160143*	-120x+300359 N	-489x+1075706*	-13x+28552*	-4x+9825*
NZ	-174x+381861*	-211x+560053*	153x-295664*	70x-132229*	26x-53058*	7x-15498*
D	-186x+396480*	72x-142573*	-120x+300359*	-51x+104765*	-18x+37759*	-4x+8681*
W	---	---	6614x-12306722*	5710x-10087737*	1381x-2619516*	132x-230341 N
U	-2.6x+18698*	-54x+244858*	153x-295664*	69x-127116 N	2x-4165*	1x-3620*

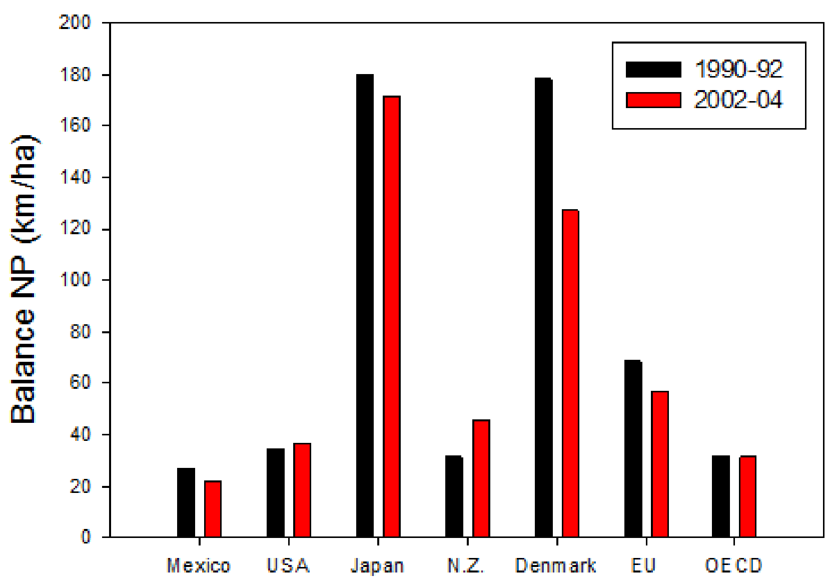

Figure 2.4. Gross nutrient balance (nitrogen and phosphorus nutrients applied by area of agricultural land, Kg Nutrients/ Hectares of agricultural land) for countries belonging to different regions in the periods 1990-1992 and 2002-2004. USA: United States; N. Z.: New Zealand; EU: European Union; OECD: Organization for Economic Co-operation and Development.

The gross balance of nutriets pointed out Japan, Denmark and EU as the countries/groups with the highest application of fertilizers per unit of area. However, there was a decrease in that balance during the period 2002-04 compared to 1990-92, especially in Denmark. In contrast, New Zealand and the United States slightly increased their balances (Figure 2.4).

Finally, the source apportionment indicated Agriculture as the main diffuse source of both nitrogen and phosphorus in Odense River. With regard to nitrogen, the agriculture, followed by the background losses, showed little variation along time, being the decline noticeable in the year 2000 (Figure 2.5 A).

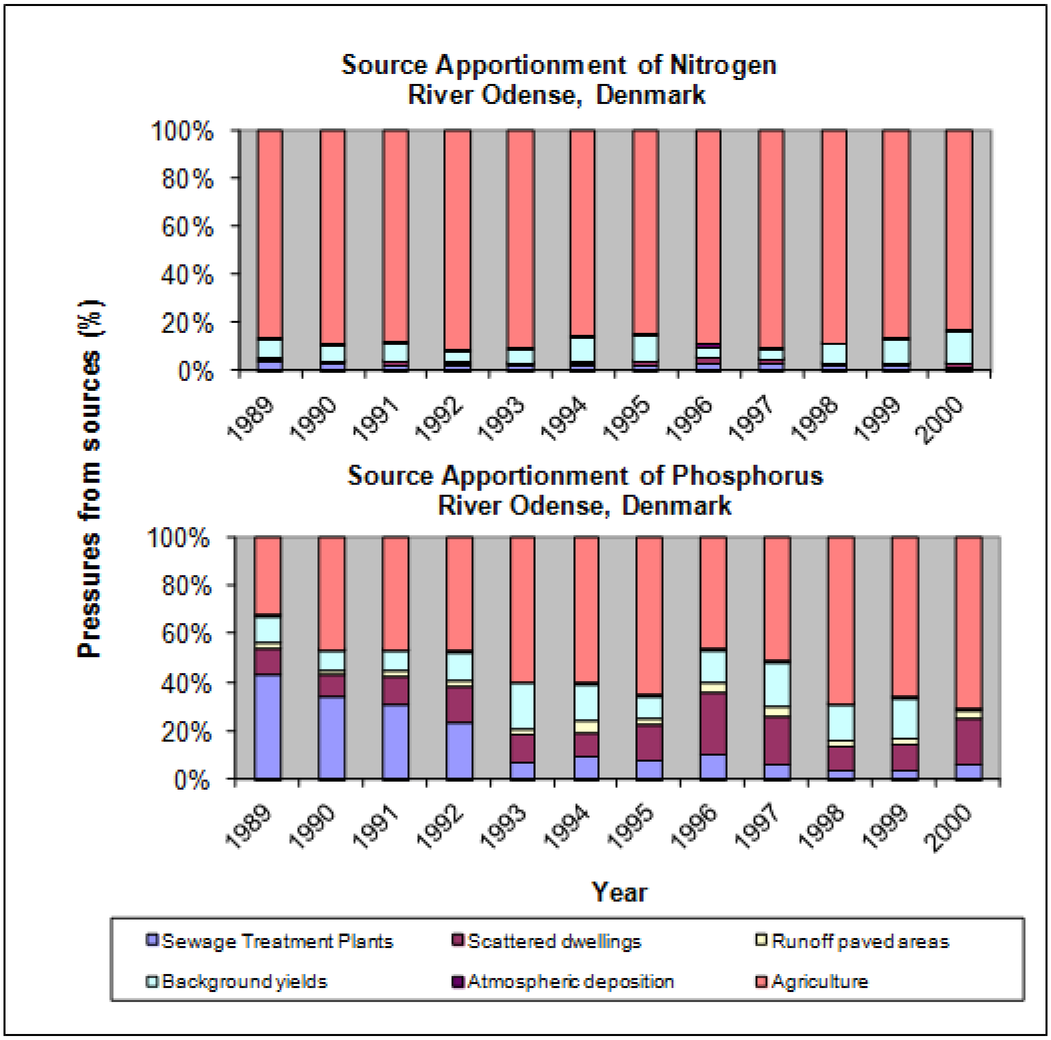

Figure 2.5. Source apportionment of nitrogen (top) and phosphorus (bottom) during the period 1989-2000 in Odense River, Denmark.

In the case of phosphorus, while the percentage of pressure produced by sewage treatment plants diminished through time, the opposite happened with phosphorus coming from agricultural losses (Figure 2.5 B).

2.4- Discussion

During this data analysis, the general world trends in land use and agricultural systems could be assessed. Moreover, the high relative importance of agriculture as source of nutrients in a river (Odense) was identified.

The general trends for the world showed a diminishment in agricultural land extension, both in arable land and grassland. Nevertheless, there exists great variability among different regions of the globe, probably related to the socio-economic conditions prevailing (Kronvang *et al.*, 2008). Among certain countries, most of them undeveloped, there is still a tendency for agriculture to extend, however, it happens with high costs for environment because many natural environments are being degraded to transform land into arable land or grassland for livestock (Harris, 1996; Wood *et al.*, 2000; OECD/FAO, 2011). On the other hand, in many regions, mostly the developed ones, which are more concerned about the negative effects of agriculture on environment and where management policies are more widespread (Tinker, 1997), are experiencing the retraction of agricultural land (Wood *et al.*, 2000), although this pattern could not be identified by means of linear fit utilized here. One explanation to this could be a nonlinear diminishment in agricultural extension due to different responses to management measures through time and diverse socio-economic situations.

With regard to inorganic fertilizers, it was seen that even though some countries are attempting to reduce their application (e.g. Schoumans & Chardon, 2011; Kronvang *et al*., 2008), the amounts of phosphorus and nitrogen consumed are still high and increasing in many places (Gellings & Parmenter, 2004). The expansion of agricultural land (for example leading to use poor soils for crops) to poor soils and the need to obtain goods and services from areas that are becoming smaller and less productive through time could result in a rise of fertilizers consumption. This fact, together with the rising observed for pigs and cattle, indicates that agricultural practices are intensifying (OECD/FAO, 2011).

The gross balance of inorganic fertilizers (nitrogen and phosphorus together) is very variable, probably depending on each country economic development (Vitousek *et al*., 2009), land use and the policies regulating fertilizers' trade. In this case, it was observed an effect of the area dedicated to agriculture among countries on the gross balance. For example, in countries like the United States, where the amounts of agricultural land are higher than in other, this balanced resulted lower than the observed where the arable and grassland resulted to be smaller (mainly due to the country surface), as Uruguay, Denmark, Japan and Holland. Moreover, the balance decreased in 2002-04 compared to 1990-92 in Denmark and EU, a trend that may be related to the strict normative and the many management actions carried out during the last decades (Kronvang *et al*., 2008; Shoumans & Chardon, 2011). On the other hand, the OECD countries did not show any change through time. The fact explaining that trend could be the diversity among the members. This organization not only includes developing countries (such as the ones belonging to the EU) but also developing or emerging countries (like Chile and Brazil, respectively). As a consequence, diversity in socio-economical situation, development strategies and environmental policies can be leading to a compensation of nutrient losses diminishment in some countries (mostly developed) with other's increasing fertilizers consumption (e. g. China, Brazil and India, OECD/FAO, 2011).

The source apportionment of nutrients for Odense River through the period 1989-2000, constitutes an example of the response of an aquatic system to management measures to diminish losses from different kind of activities. The decrease in the relative contribution of both nitrogen and phosphorus coming from sewage treatment stations is consistent with the efficient regulation of pollutant point sources, as it was shown by many studies in Denmark and other European countries (Kronvang *et al.*, 2003; Kronvang *et al.*, 2005; Kronvang *et al.*, 2008;). What is more, phosphorus, which represents the main nutrient in sewage and industrial discharges (Smith *et al.*, 1999), showed a stronger pattern than nitrogen (Kronvang *et al.*, 2003).

Along this chapter it was identified a tendency to the intensification of agricultural activities and their pressures on the environment. Moreover, as it was seen, this situation is predicted to exacerbate in the coming years. Given that scenario, to prevent an irreversible degradation of natural resources and attempt to reach the way to sustainable development, the implementation of management measures to avoid the impacts of agriculture is vital.

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Chapter III:

Nutrient concentrations and trends in World Rivers and modelling scenarios for nitrogen

Ivan Gonzalez-Bergonzoni

3.1- Introduction

The mechanisms and forms in which the nutrient transport from land to water occur are very diverse and depend on the nutrient input form and amount, biological activity on the soil, soil type, predominant hydrology pathway, and several climate variables such as precipitation patterns and temperature (Follet, 2008; Leinweber *et al*. 2002; Jeppesen *et al.*, 2011).

Nitrogen is the nutrient leaked in higher proportions from land to water, being the main transported form, the inorganic soluble NO_3 (Follet, 2008). This nutrient form is an abundant product of nitrification and also frequently solubilized from minerals and added artificially by the use of fertilizers, it transports directly with the water flow, constituting the most important N leaking pathway (Follet, 2008). Additionally, mineralized ammonia can be found bound to clay and small particles and thus, may transport attached to particulate material. Surface water runoff can be an important transport pathway, although this depends largely on the soil type, landscape and climate characteristics (Follet, 2008). However, despite of the variable importance of this transports pathway, in general nitrogen leaching via infiltration to underground water flow, accounts for the highest N transport from land to water bodies (Kronvang *et al*., 2008).

Transport of P from the land to water in a basin usually occurs from the P pool on soil, via solubilization or detachment. The solubilization implies passage from solid to dissolved phase of P (PO_4)) and the time between this process and the reverse of the same (sorption) is within seconds or minutes (Leinweber *et al.*, 2002). As soluble P is often limiting for primary producers and microorganisms, it is incorporated into their biomass very fast and then released as dead biomass as organic P (Leinweber *et al*., 2002). On the other hand, detachment is the process by which P is moved from the land attached to solid surfaces (most commonly fine clay particles with high sorption capacity) by the main effect of water erosion (e.g. stream flow, surface runoff, rain drops). This process is recognized as the main leaking mechanism for P. Depending mainly on soil type, water from precipitation can flow on the surface (runoff) or subsurface (groundwater (upper and deeper)) routes, being the surface runoff the most important path for P leaking. However this also depends on the erosion processes on the soil (e.g existence of macropores, worm holes, cracks, amount of vegetation cover) (Leinweber *et al.*, 2002).

The actual increase in nutrient transport from land to water has evoked the need for a series of management actions aiming to reduce such pollution source. On this way the development of modelling tools to predict nutrient leaking from land to water, under diverse scenarios, complemented by monitoring of nutrient concentrations throw time, seem of key importance to achieve good management policies.

This chapter will is organized in two main sections; firstly aiming to review the evolution of nutrient transport (P and N) in World Rivers in the last twenty years. Finally getting introduce to the use of a simple but useful modelling tool for nitrogen transport, evaluating nitrogen inputs to aquatic ecosystems under different climate and land use scenarios.

3.2- Nutrient concentration level evolution in World Rivers

3.2.1-Methodology

For the analyzing of nutrient concentration trends in World Rivers, data of total nitrogen and total phosphorous concentrations (in milligrams per liter) was extracted from 9 rivers in OECD database, provided by the course professors. The studied regions were: USA (River Misisipi); Hungary (River Duna); Finland (River Toniojoki); Denmark (River Suså); Germany (River Rhein); France (River Loire); UK (River Thames); Spain (River Ebro); Canada (River Saskatchewan) and Mexico (River Lerma). The monitoring period consisted of a time series from 1980 to 2005, although there were certain blank years for some of the systems, data plotting and trend analyzing was arrayed in Sigma Plot 11. In the case of the total phosphorous concentrations the rivers were sorted arbitrarily between low and higher P concentrations to achieve a better visualization of results. We performed no statistics on this as were focused on observing large scale trends more than strictly modelling the behaviour of nutrient concentrations throw time.

Data from Uruguay River was also analysed separately as belong to a different time series. This belongs to a monitoring program on the context of a pulp mill impact assessment report and data was available but averaged between years 2005-2007 and 2008 to 2010. For this reason only a comparison between these two periods was possible, and was done by a box plot graph showing maximum, minimum and average value for each period. Statistics on this comparison were arrayed via non parametric Mann Whitney U test at an α level of 0.05.

3.2.2- Results

The analysis of Total N and P concentrations from 1980 to 2005 from 10 large rivers at different regions of the globe has shown interesting and different nutrient dynamics across time and regions.

Total Nitrogen levels in rivers (Figure 3.2.1) varied between approximately 0.5 mg/l in River Toniojoki (Finland) and Saskatchewan (Canada) to around 8 mg/l in the polluted river Thames (England) (Figure 3.2.3), However if we look at the trends within the 25 years period we can observed a decreasing tendency for the rivers from UK, Germany, Denmark, Finland and Hungary. In contrast the rivers from France, Spain, USA, Canada and Mexico tend to increase their nitrogen concentrations during such period (Figure 3.2.1). The strongest decrease seems to be that of river Susa (Denmark) as shown by its steepest slope of the linear trend analysis (but considering linear adjustment just as a trend analysis and not strictly as a model).

The long term series analysis for total phosphorous revealed the highest fluctuations among rivers and time, median values for that period ranging from 0.02 mg/l in the Finnish oligotrophic River Toniojoki to 1.7mg/l in Polluted River Lerma (Mexico) (Figure 3.2.3). In the general trend analysis (Figure 3.2.2) we can observe notorious regional differences in the P concentration behaviour during the study period. While all European rivers showed a marked decrease in P levels during the last 25 years, both Mexico and USA systems continue to raise their total P levels (Figure 3.2.2).

Interestingly, when examining the mean values and ranges for every river in the whole study period both for total N and total P (Figure 3.2.3) it becomes clear that Phosphorous dynamics tend to be more variable than Nitrogen concentrations. The 25 years variability in total N rarely surpasses 1 mg/l, being the highest for the Danish river, whose total Nitrogen decrease was the most important. Highest P variability corresponded to those with highest levels, and highest decrease and increases (rivers Thames (2mg/l variation) and Lerma (5 mg/l variation) respectively) (Figure 3.2.3).

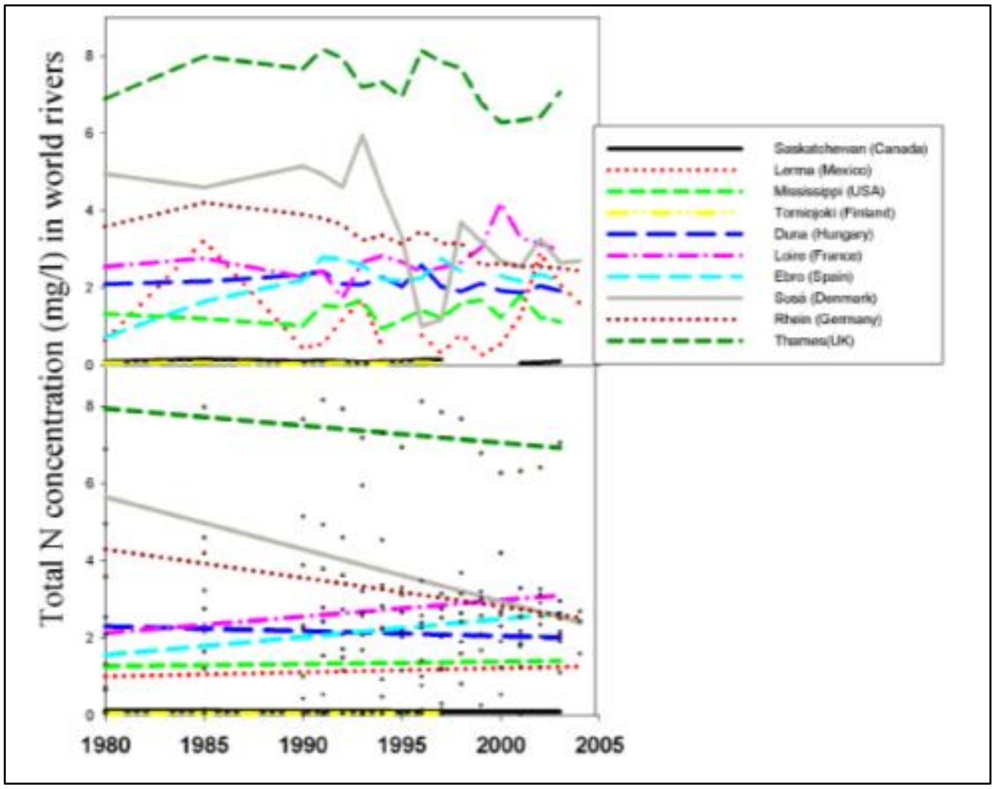

Figure 3.2.1. Total nitrogen concentration levels in 10 world rivers between 1980 and 2005 (above) and linear trends in the concentration of total nitrogen in the same study period (below).

Data from water quality in Uruguay is usually restricted to different state division's internal reports usually not available to open public. For this reason the only data we could access to come from an environmental impact assessment from a Pulp Mill industry located in Uruguay River facilitated by the pulp mill authorities. This data was shown reported as median values for three years periods, only allowing comparison between 2005-2007 and 2008-2010 periods for total N and total P (Figure 3.2.4) . In this analysis it became evident that both Nitrogen and Phosphorous are facing an increase phase in the Uruguay River (Mann Whitney $p<0.05$). Additionally the higher variability of total P concentration with respect to total N also became evident (ranging from approx. 10 micrograms/ liter to more than 200 in the 2005-2007 period), as happened in the 10 rivers analysis (Figure 3.2.4). If we match this concentration values with the observed for the several world rivers analysed we can see that total N values position Uruguay River among the medium concentration's Rivers (e.g River Loire, Duna and Mississippi), while Phosphorous concentrations place it similar to the lowest P concentration Rivers, in particularly similar levels to the Rivers from Finland and Canada.

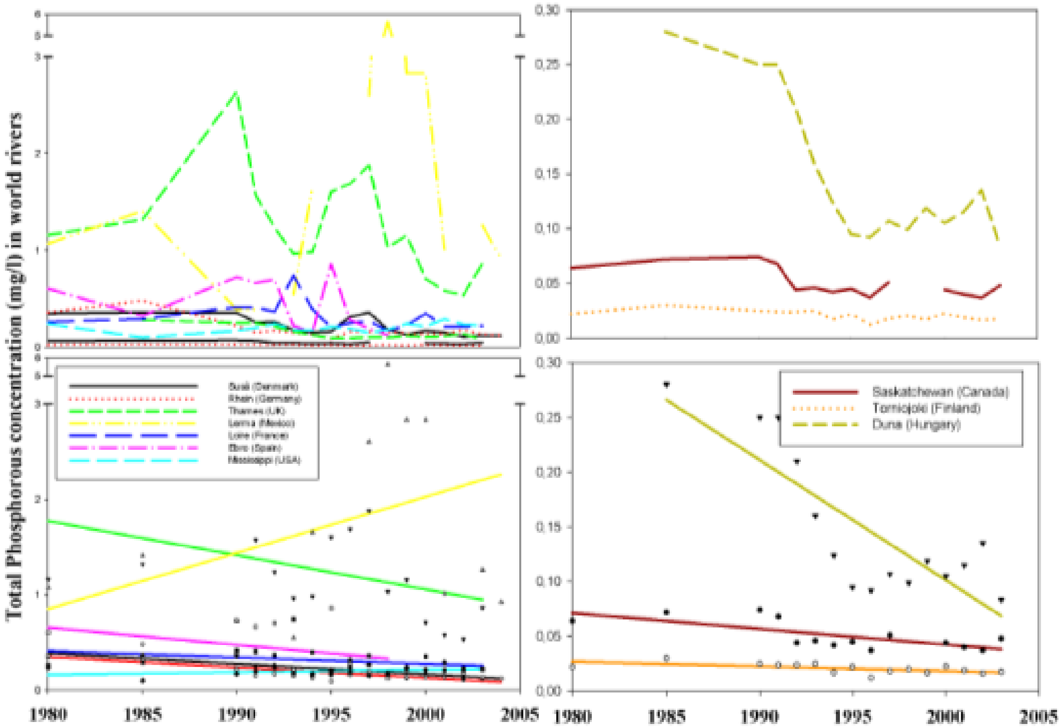

Figure 3.2.2. Total phosphorous concentration levels in 10 world rivers of medium and high P levels (left) and low P levels (right) between 1980 and 2005 (above) and linear trends in the concentration of total P in the same study period (below).

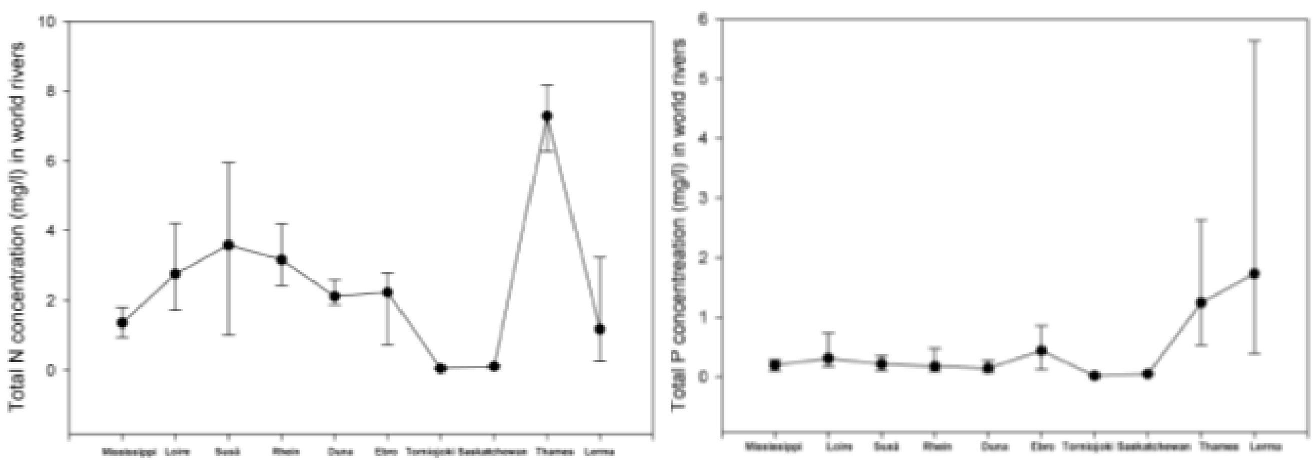

Figure 3.2.3. Median values and minimum maximum ranges for total Nitrogen concentration (Left) and total Phosphorous concentrations (right) for every studied river.

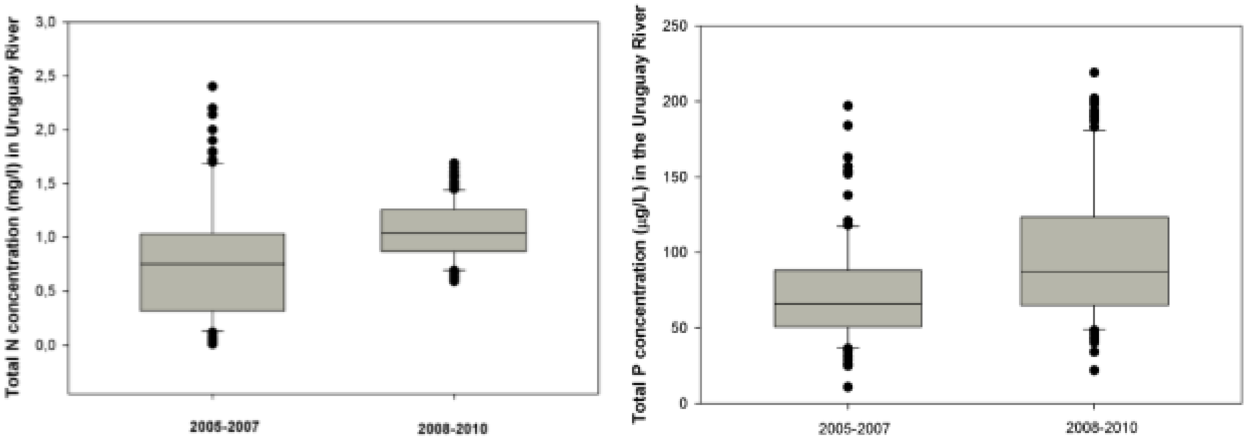

Figure 3.2.4. Total nitrogen (left) and total phosphorous concentration (right) values for periods 2005-2007 and 2008-2010 in Uruguay River. Data is shown as Box plots graphs showing the media and including outlier values significant differences were found between periods for both total N and total P.

3.2.3- Discussion

During the nutrient concentration levels analysis we could observe that while all the European countries tend to decrease their phosphorous levels in rivers and most of them tend to decrease nitrogen levels as well (Shoumans & Chardon, 2011), this pattern is the not clear or the opposite in most rivers of the American continent. This is probably (at least partly) related with the existence of regional environmental policies such as the European Water Framework Directive putting high pressure on water quality improvement for all the European region, and the lack of such strong organization and pressure on other countries.

The reduction in total phosphorous values along Europe is usually attributed to the improvement in effluent quality at punctual sources (O'Shea, 2002; Shoumans & Chardon, 2011). Punctual sources are easier to control given the singularity and easy identification of sources (identified industries) when compared to the nitrogen emissions coming mostly from diffuse sources (as was seen in Chapter II) (O'Shea, 2002). Related to this, the two river systems showing higher P charge during the whole period (River Thames and Lerma) were those flowing throw the dense urban and industrialized areas (Prevailing punctual sources), while for the nitrogen levels highest charges were also found in agricultural countries such as Denmark or France (prevailing diffuse sources).

The analysis of the scarce data existent for Uruguay allowed us to characterize it as a River system with medium to low nutrient levels (when compared to rivers from industrialized countries), but in which nutrient levels are in an increasing phase. This is not rare if we consider the observed in chapter one about the import of artificial fertilizers and the increase in the percentage of arable land in Uruguay, all of which, together with the evident lack of nutrient reduction policies, suggest nutrient levels in Uruguay River will continue to increase in the coming years as agriculture keeps increasing in extension and intensity.

3.3- Approach to nitrogen modelling

3.3.1- Methodology

To get an introduction to modelling and facilitate a better understanding of factors affecting nitrogen inputs to an aquatic ecosystem we used an empirical model based on data from 84 streams across Denmark, provided by the course professor. On this way, by simply modifying the model's parameters we predicted the changes in nitrogen input to aquatic systems under several different soil type, land use and climate scenarios.

The scenarios tested were the following: scenario A) sandy soil type (80% sandy soils) and drained area = 10% and scenario B) Loamy soil type (20% sandy soils) and drained area = 90%. With the combination of the following land use and climate changes: 1) Agricultural land reduction of 10% and 50%; 2) N-surplus increase or decrease of 10% and 50%; 3) Mean monthly precipitation increase or decrease of 10% and 50%; and finally 4) Mean monthly temperature increases or decrease of 1 ^{o}C or 5^{o}C .

The results were plotted using sigma plot 11 and compared to analyse the changes to occur given the different possible scenarios, as needed to discuss and create nutrient management policies.

3.3.2- Results

The modelling of nitrogen output to the recipient stream arrayed similar results in the predicted behaviour of nutrient outputs under the different land use, management and climate scenarios proposed, for both kind of agricultural systems (contrasting in soil type and drained area). However, in the less drained sandy soil case (scenario A) total nitrogen outputs tend to be less than in the loamy and drained soil type (scenario B), varying from maximum around 8 mg/l to in February to a minimum of 3 mg/l in August in the first case and from10 mg/l to around 4mg/l on the same season in the loamy soil (Figure 3.3.1, Table 3.3.1).

Nitrogen output reduction was here achieved by agricultural land and N surplus reduction as well as by a less precipitation and colder regime, while increase of increase of these same parameters caused the opposite (Table 3.3.1, Figure 3.3.1). Along the different scenarios tested for both kinds of agro systems, most of them throw out similar results nitrogen output varying in less than 2 mg/l during winter and less than one during summer (Figure 3.3.1), overall most cases ranging from 4.7 to 5.8 mg/l total N year average output in scenario A and from 5.9 to 7.28 mg/l total N in scenario B (Table 3.3.1). Despite this, three of the test trials stand apart from this, resulting in larger reductions or increases of the nutrient output. These cases are the increase in 50 % of the actual N surplus on land, resulting in the maximum total N output of 7.21 and 9.09 mg/l average per year in scenario A and B respectively (Table 3.3.1). Not surprisingly the opposite change (reduction in 50 % of N surplus) brings as a consequence a high reduction in total N output, reducing from the 5.3 and 6.8 mg/l average per year in basal conditions to 4.03 and 5.08 for scenarios A and B respectively (table 3.3.1, Figure 3.3.1). However, the highest magnitude of reduction is only achieved by agricultural land reduction of 50 % total N average values per year decreasing to 2.8 and 3.5 mg/l for scenario A and B respectively.

Table 3.3.1. Average year values for total nitrogen output from land to water for two contrasting soil type and drainage systems under variation of several land use, management and climate characteristics. In red the extreme changes observed, minimum (50% agricultural land reduction) and maximum (50% N surplus increase) N output (mg/l).

Parameters modified	Average year N output (mg/l) Scenario A	Scenario B
Baseline conditions	5,39	6,80
10% agricultural land reduction	4,73	5,96
50% agricultural land reduction	2,80	3,53
10% N surplus increase	5,71	7,21
50% N surplus increase	7,21	9,09
10% N surplus reduction	5,09	6,41
50% N surplus reduction	4,03	5,08
10% Precipitation increase	5,46	6,89
50% Precipitacion increase	5,75	7,25
10% Precipitation decrease	5,32	6,71
50% Precipitacion decrease	5,06	6,38
1°C temperature increase	5,49	6,92
5°C temperature increase	5,78	7,28
1°C temperature decrease	5,30	6,68
5°C temperature decrease	4,95	6,24

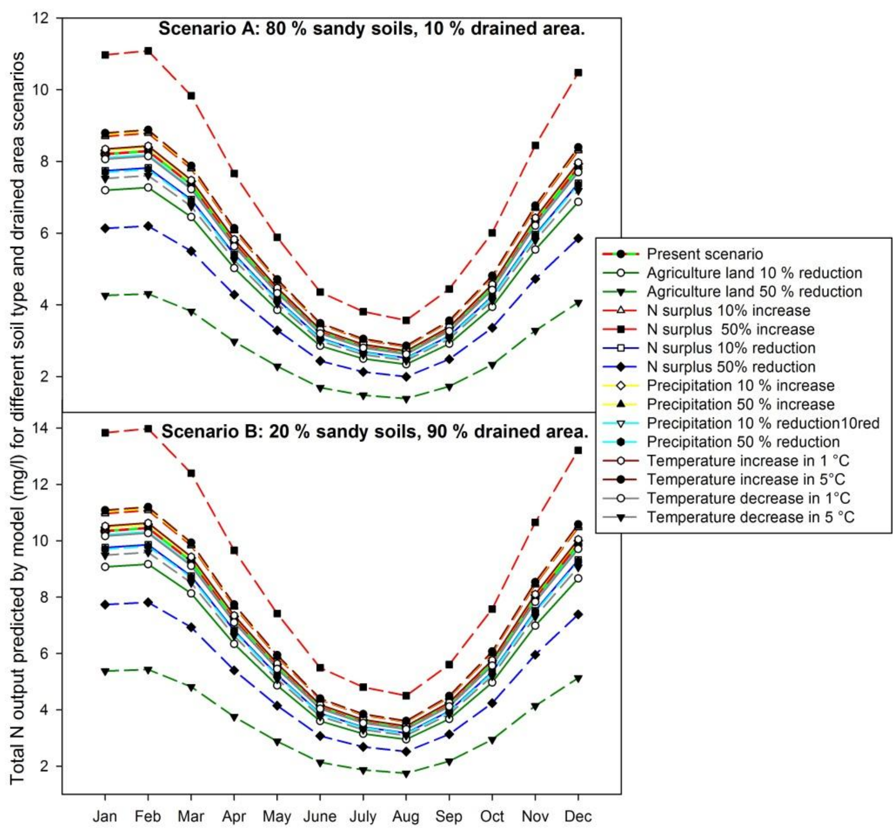

Figure 3.3.1. Total nitrogen output to stream thorough the year, predicted by an empirical nitrogen leaking model for two contrasting soils type scenarios: a sandy soil scarcely drained (above) and a highly drained loamy soil (below). Different trials arrayed included changes in land use, management and climate characteristics.

3.3.3- Discussion

During this introduction to the use of nitrogen model we demonstrated the great potential of such management tool to reduce N output, although nutrient loses to aquatic ecosystems varied between the two contrasting soil types and hydrology. Sandy soil here presents a lesser amount of nitrogen leaking, most likely due to the higher infiltration capacity to deeper underground water, reducing the nutrient flow towards upper sub superficial layers and surface runoff to water bodies (Follet, 2008).On this same way, artificial drains increase the amount of water (and thus solved nutrients) that follow directly to the aquatic systems (Follet, 2008), for such reason diminishment of this will likely cause a reduction in N output and this should be considered in management decisions.

Transcending differences among soils and drained areas, percentage of agricultural land and N surplus decrease cause an important decrease of N pollution, application of any of these measures would translate into an effective management action. However, the most effective N output reduction here (reduction of agricultural land on 50%) seems unreachable in the present reality, due to the higher need to produce food (Schoumans & Chardon, 2011). For this reason we remark that N surplus reduction should be the main focus of the management actions, although never discarding agricultural land reduction even at a small degree if possible.

The fact that rain pattern and temperature also be affecting this process becomes essentially important on the context on the global climate changes to occur in the next years (IPCC, 2007). Consideration of these aspects on a large scale nutrient reduction policy seems essential to achieve the desired environmental improvements regarding nutrient pollution.

3.3.4- References

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Chapter IV:

Mitigation options in catchments: functioning of buffer strips and wetlands

Anahí López

4.1- Introduction

In today's world in terms of increased loss of soil nutrients, management of surface runoff is commonly used to increase retention of sediment and nutrient transport to surface water from cropfields upstream. In the last, and for the coming years, the demands of EU Water Framework Directive (WFE) will require measures to obtain at least good ecological quality in surface water bodies which often are facing eutrophication as a major threat to ecosystem structure and function (Kronvang *et al.*, 2005; Hoffmann *et al.*, 2011). This management is based on fundamental principles such as water retention, increasing the residence time of water, the interaction between water and sediment and biogeochemical processes such as denitrification (reduction of NO_3^- to nitrogen gaseous form like N_2O and N_2 (Hernandez & Mitsch, 2007)), and sorption (Kronvang *et al.*, 2011). This operation can be done by landscape management, among others. For this, it may be useful to design "landscape buffers" such as buffer strips and wetlands (Dorioz *et al.*, 2011). Careful management of riparian zones offers a strategy to diminish the flow of a variety of impacts of land use, while allowing it to continue making use of the land (Cooper *et al.*, 1995). These areas provide a variety of ecosystem services, defined these as attributes of ecosystems consumed and/or used to produce human benefit. Defined in this way, ecosystem services include ecosystem organization (structure), operation (processes) and outputs, since all are used directly or indirectly by humanity (Turner *et al.*, 2008). A fundamental ecosystem service that these regions provide is the retention of contaminants such as suspended sediment, carbon, and pesticides in various forms (Stevens & Quinton, 2009). Another of the pollutants that are able to retain these areas are the nutrients, meaning retention of nutrients (nitrogen and/or phosphorus) the storage of excess through the same biological processes, biochemical and geochemical biomass (living and dead) and mineral compounds of buffer strips and wetlands. In the storage of nitrogen and phosphorus, nutrient retention service has the effect of improved water quality downstream of these areas (Loiselle *et al.*, 2006, Turner *et al.*, 2008).

4.2- Buffer Strips

Buffer strips (BS) are vegetated riparian zones that act as filters between fields and water bodies, which intercept and treat water from agricultural fields, thereby serving as a tool to reduce diffuse agricultural pollution (Borin *et al.*, 2010). This effect results from four fundamental processes: the capture and storage of water and/or sediment and nutrients from the buffer zone, the uptake of water and dissolved nutrients by vegetation and biota, biogeochemical transformation (such as sorption and denitrification) , and dilution (Dorioz *et al.*, 2011; Sheppard *et al.*, 2006). The vegetation of Buffer strips consists of grass may be natural or a mixture of vegetation including grass, trees and bushes (Borin *et al.*, 2010, Zhang *et al.*, 2010; Dorioz *et al.*, 2011).

These areas may modify, incorporate, dilute or concentrate substances before they enter the aquatic ecosystem. The buffer strips contribute to the preservation of local water, acting as a selective barrier effect on surface runoff, pollutants and export of organic matter (Dorioz *et al.*, 2011). It is for these reasons that these areas have been used and are used as a viable and useful tool for the restoration and management of rivers and streams (Osborne & Kovacic, 1993). Several studies have been conducted worldwide to verify the efficiency of these areas in the mitigation of pollutants from the terrestrial environment. A study conducted in Italy during the period 1998-2002 (Borin *et al.*, 2010) resulted in an increasing trend in the concentrations of all forms of nitrogen (total, nitrate and ammonium) as it passes through the BS, but total N losses were reduced from 17.3 kg ha-1 to 4.5 kg ha-1 in terms of mass balance. By contrast, P concentrations were not modified in the case of soluble P or decreased (P total), generating an 80% reduction in the loss of P. It is worth noting that the retention of P in the buffer strip is mainly controlled by sedimentary processes (Dorioz *et al.*, 2011). In this study, decreased nutrient loss as a result of a 78% reduction in surface runoff in areas with BS compared with areas that had no BS was observed. However, in a review conducted by Sheppard *et al.* (2006), it was found that the buffer strips were effective in reducing P loss by runoff only in 50% of the study, whereas 18% of cases appear as a source of P. This may be because, as demonstrated by theoretical models developed by Zhang *et al.* (2010), the capacity retention of nutrients is dependent on the width of buffer strips, and concluded that the removal efficiency increases rapidly as the width of the buffer zone grows (Fig. 4.1.). The functions can be anticipated of riparian buffers conferring to their widths is shown in Fig. 4.2. However, it is believed that with appropriate vegetation and proper handling, narrow buffer strips may have some potential to reduce pollution to surface waters, reducing the entry of nutrients, pesticides or manure, trapping coarse particulate organic matter and preventing the bank erosion (Dorioz *et al.*, 2011).

In the before mentioned study conducted by Zhang *et al.* (2010), a reduction in N loss of 68.3% and 71.9% for P was obtained as a result. The buffer capacity also depends on the type of vegetation that composes the BS and the slope, increasing the efficiency when the slope increases from 0 to 10%, and becoming less effective when the slope increases further.

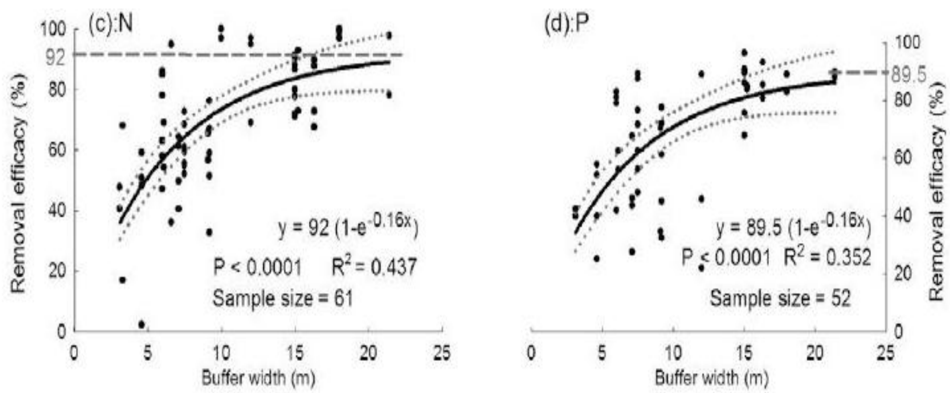

Figure 4.1. N and P removal efficacy vs. Buffer width. Black line are data. (From: Zhang *et al.*, 2006).

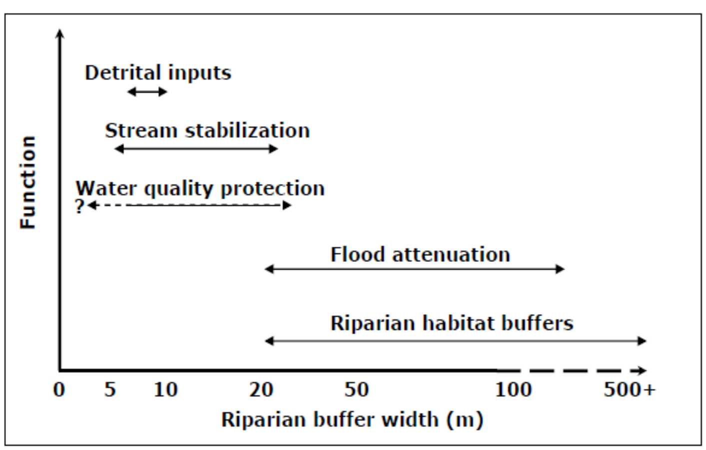

Figure 4.2. Function of BS according to their width. (From: Dorioz *et al.*, 2011).

Due to the high soil moisture and nutrient availability, buffer strips are areas of high productivity that are suitable for tree planting that, in addition to the elimination of pollutants and the uptake of CO2 (greenhouse gas), can be harvested, providing a source of renewable fuels such as wood and whose pulp can be used for paper production (Borin *et al.*, 2010). Furthermore, medium and small channels, where strips of timber vegetation located parallel to the canal can be used, it was noted that this area is able to moderate the temperature, reduce sediment input, provide an important source of organic matter to lotic communities and can also stabilize the riverbanks (Osborne & Kovacic, 1993). If the BS are formed by grassland, they can reduce the transfer of pesticides (88% in studies by Sheppard *et al.* (2006)) to water bodies in runoff from agricultural fields, as it was noted that this type of vegetation is the most effective (Borin *et al.*, 2010). Finally, other potential benefits of BS is that they can serve as a suppressor of weeds, plague management, supply sources for livestock forage in fields that are crop-livestock and can improve soil structure (Stevens & Quinton, 2009).

With the above, we conclude that for the implementation of a BS as a mitigation measure, it is necessary to consider the utility that wants to be given to this area to be able to elucidate the most appropriate type of vegetation and width to ensure a high effectiveness in the retention of contaminants. It is also extremely important to know the local conditions (substrate type, slope, biodiversity, adjacent land uses, etc.).

4.3- Wetlands

Wetlands are ecotones or transition zones between terrestrial and aquatic systems where the groundwater is usually near the surface or the land is covered by shallow water as well as being an area generally anaerobic. Wetlands are known for their ability to remove sediment, nutrients and other pollutants from water (Zedler & Kercher, 2005).

The intense agricultural production, and consequent water drainage, implemented with the aim of increasing the arable land and urban development has a major effect on the loss of natural wetland areas (Hoffmann & Baattrup-Pedersen, 2007), resulting in wetlands occupying only 9% of the land surface (Zedler & Kercher, 2005). The recognition of the importance of these regions and their steady decline, has led scientists and the general public to pay greater attention to restoration and management plans (King *et al.*, 2009). The restoration of wetlands, in most cases is done with the main objective of reducing nutrient loss from agricultural fields upstream water bodies (increasing the retention of P and N), through sedimentation, denitrification and sorption, in addition to increase biodiversity (Kronvang *et al.*, 2011; Hoffmann & Baattrup-Pedersen, 2007; Hoffmann *et al.*, 2011).

It has been observed in several studies that water flows facilitate the exchange of materials between water bodies and flood plains, so that the restoration of these river systems is an essential step in the restoration of wetlands, as the flood pulse and nutrient availability change the oxygen capacity of soils with potential for denitrification (Hernandez & Mitsch, 2007). Most studies in Europe on N removal in wetlands have shown high rates of removal, and high efficiency, mainly due to the reduction of nitrate by denitrification (Jeppesen *et al.*, 2011).

Two key factors for denitrification (main removal process for N loss of wetlands) are precipitation and temperature, as these influence nitrogen inputs and the metabolism of denitrifying bacteria (Hoffmann & Baattrup-Pedersen, 2007).

Experiences in Denmark have shown that the effect of restored wetlands on the retention of N reached a value close to 200 kg N per hectar of restored wetland. The effect of wetlands increases with increasing discharges of N and increases the residence time of water. In the case of P, wetlands showed the same retention capacity of 10-100 kg per hectar P of inundated wetland. However, the retention of P is more effective for particulate P than for dissolved P, because in the anaerobic conditions that are in wetland soil, P is released from its binding with iron (Kronvang *et al.*, 2011). It's common to find side effects in N and P antagonistic dynamics in transfer systems, potentially impacting other compartments of the environment. As well-known example of the antagonistic effect on nutrients is given by soil anoxia which decreases nitrate to gaseous nitrogen, but can generate dissolved-P. P release has been observed in wetlands receiving excessive amounts of NO_3 from cultivated areas (Dorioz *et al.*, 2011).

In another study conducted in Denmark by Hoffmann *et al.* (2011), it was demonstrated that there is a linear positive relationship between nutrient removal and the volume of runoff water (Fig. 4.3.). In this study, conducted in four restored wetlands in four different rivers, it was noted that there is great efficiency in the removal of N, not with the P, as some of the study sites appeared to be a source of P. It was further observed that in a restored system, the inflow of P was mostly in the form of SRP much, whereas the outflow was mostly in the form of organic P, which may be because what comes out is the uptake-P in organisms (algal biomass). A study in U.S.A. demonstrated too that wetlands can retain 59% of NT (total nitrogen), 78% of NO_3 (nitrate), NH_4 (ammonia) 52% and 66% of PT (total phosphorous) (Vellidis *et al.*, 2003).

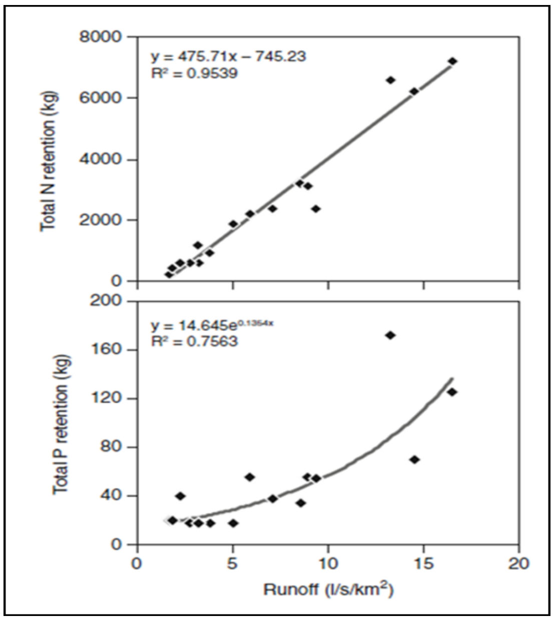

Figure 4.3. Karlmosen Fen River (Denmark). Relationship between total nitrogen (N) and total phosphorous (P) and runoff. (From: Hoffmann *et al.*, 2011).

When it comes to sediments, it has been observed that organic matter is decomposed into restored wetlands with subsurface flow, for aerobic and anaerobic microbial processes as well as by sedimentation and filtration of particulate organic matter, so that the entry into the bodies of water is diminished, avoiding the turbidity of the same (Vymazal & Kröpfelová, 2008).

This ecosystem provides many services to society (in addition to the ones already mentioned) including habitat diversity, carbon stocks, wood production and increased water quality (King *et al.*, 2009). It was also observed that the diversity of macrophytes determines the effectiveness of giving ecosystems services wetlands to society as sustainable food production, recreational opportunities and purification of water pollutants holding capacity. These services depend heavily on how well the wetland performs ecosystem services and the retention of nutrients and primary production (Engelhardt & Ritchie, 2001). Another ecosystem service provided by wetlands is the reduction of human pathogens from wastewater by sedimentation of them in these regions (Karim *et al.*, 2004). Not all wetlands perform all functions equally, so it is necessary to have a comprehensive restoration strategy. For example, large wetlands are often home to many species of birds, while the small wetlands can lodge large plant taxa. Wetlands retain nutrients upstream, while downstream wetlands in key positions in watersheds can eliminate up to 80% of nitrates. Priority should be given to the types of wetlands for restoration, the total area that is needed, and the best location (Zedler, 2003).

Some problems associated with the use of restored wetlands are such that the operation thereof has not been studied yet, so there is not much information available to the posing of new restoration projects. All the biogeochemical processes that are essential for understanding the functioning of wetlands are not well known. Therefore, these zones remove nitrogen but with subsequent emission of greenhouse gases (product of denitrification). To verify the effectiveness of these ecosystems in one of its main functions such as the removal of contaminants (mainly nutrients), it is necessary to conduct long-term studies, increasing the monitoring time after the restoration of wetlands (Hoffmann *et al.*, 2011).

4.4- References

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Chapter V:

Discussion

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5.1- Discussion

During this review we got introduced into the actual knowledge about nutrient cycles and pollution in aquatic ecosystems at a worldwide scale. Specifically by reviewing and assessing the evolution of land use, the importance of diverse nutrient pollution sources to aquatic systems and different management approaches. These trends were analyzed for regions with diverse socio-economic scenarios and environmental policies.

Agricultural land use intensity is increasing at a global scale, as evidenced by the rising amount of artificial fertilizers being applied, over equal amounts of productive land. Odense River nutrient source apportionment showed that agriculture represents the highest proportion of the nutrients released to the environment. This is coherent with other studies where agriculture often represents the main nutrient leaking pathways (e. g. Grizzetti *et al*., 2005; Withall *et al*., 2007). This translates into a heavier pressure on environment, as seen in Chapter III where the higher nutrient levels on Rivers were those from agricultural countries. Nutrient enrichment results in loss of ecosystem services, for example via loss of water quality and biodiversity (Carpenter *et al*., 1997; Chorus & Bartram, 1999; Lampert & Sommer, 2007). On this way eutrophication process has been reported as one of the most important and widespread environmental problems around the world. Recent surveys showed that 54% of lakes in Asia are eutrophic; in Europe, 53%; in North America, 48%; in South America, 41%; and in Africa, 28% (ILEC, 1993).

Due to the need to preserve essential ecosystem services and also maintain high production standards, a diverse set of management tools have been studied and applied (Schoumans & Chardon, 2011). On this way diverse nutrient modelling tools allow an accurate prediction and thus evaluation of different management strategies previous to implementation (Schoumans *et al*., 2009). These have been created based on main (land use and climate) factors influencing nutrient losses from land to water. On this way, changes in nitrogen levels were observed to depend on an important way on temperature and precipitation patterns, as well as on soil type (See chapter III).

During the last years several measures to reduce nutrient leaking applied at different levels have proved to be successful at different cases (Schoumans & Chardon, 2011). At a landscape scale, two management actions have been found to be particularly efficient in the retention of nutrients and other pollutants from agricultural fields to water bodies, these are the construction of riparian buffer strips and the wetland restoration (see chapter IV for a review). However the implementation of such management tools is mainly restricted to developed countries (Wood *et al*., 2000), in which nutrient pollution problems arose first and nutrient concentrations in aquatic ecosystems tend to be higher.

Along this review, an up surging aspect that caught up our attention was the existence of notorious differences around environmental policies applied among different regions. While in developed countries the environmental policies seems strong and with a long history (e.g. Schoumans & Chardon, 2011); on several underdeveloped countries actions on this topic seems to be less common (Tinker, 1997). In many developing countries management actions held by the government are not the priority and rather seem a distant reality. While in Europe and many industrialized regions pressures for improvement of water effluent quality in punctual nutrient sources have accomplished a diminishment in their importance (Jeppesen *et al*., 2007), in the undeveloped countries they often remain as a problem (Wood *et al*., 2000). A similar trend was observed for diffuse sources, where management practices have been reducing nutrient concentrations in Europe (Schoumans & Chardon, 2011), situation not reflected in less developed countries (Gellings & Parmenter, 2004). Contrary to the situation in many industrialized countries, there has been an increase in arable land surface in South American countries. This expansion was mainly prompted by the widespread use of soybean transgenic crops and the change in agricultural systems, for example the implementation of no tillage techniques and permanent crops. It also involves a higher application of inorganic fertilizers, pesticides and the loss of pristine areas, such as wetlands, natural grasslands or native forests (Paruelo *et al*., 2006; Arbeletche & Carballo, 2008). A similar pattern of expansion was observed in other developing countries outside South America (Harris, 1996). This kind of change in land cover exceeds the local or regional scale, with severe consequences worldwide, constituting an important part of global change phenomena (Vitousek, 1994; Paruelo *et al*., 2006).

To show a particular case, we briefly expose the contrasting situation of management of one of the leading countries with respect to environmental policies (Denmark) and a similar South American country (in landscape and land use) like Uruguay.

Denmark has been decreasing their nutrient emissions for the last 20 years. Meanwhile in Uruguay data is scarce and fragmented (e.g. for aquatic ecosystems, Conde *et al*., 2002), but a continuous heavy increase could be estimated (see chapter III). As follows we try to mention the possible underlying causes of these situations. First of all, the start point for management actions is the generation and diffusion of knowledge about this topic, making a diagnosis of the country status. On this aspect, while Demark has been researching and monitoring their systems and making public the results for long period (Kronvang *et al*., 2008), in Uruguay research has been very scarce and locally situated, while the results reach a very narrow audience. Probably (at least partly) due to this lack of general status diagnosis, Uruguay has not implemented any governmental level of policies regarding nutrient pollution. What is more, there is not regional pressure for environmental improvement in this aspect as happens in Europe with the European Water Framework Directive and the Nitrates Directive (Schoumans & Oenema, 2011). The former results in a series of measures that have achieved a reduction of nutrient pollution during the last years. For example, we can compare some measures being applied in Denmark to the situation in Uruguay. Firstly tax policies to nutrient pollution sources differ substantially. While in Denmark fertilizers and industrial activities are imposed with taxes (e.g. Kronvang *et al*., 2008), in Uruguay as a measure to promote productive development, fertilizers are exerted of all kind of taxes (Law number 13.663). Additionally, several large scale industries (important point sources of nutrients) are often benefitted by the exertion of taxes (e.g. Law 17. 202, investment treatment with Finland). On the same way, while landscape restoration is a common measure in Denmark and the European countries (Wood *et al.*, 2000, Kronvang *et al*., 2008), this is only focused on private enterprise, at a point scale in Uruguay. Moreover, state permissions for industries and agriculture to expand at the expense of native landscape are common.

Global population is increasing and it is expected to exceed the 9 million people in 2050 (United Nations, 2007). As a consequence, the demand for food (and other products like fiber and biofuels) derived from agriculture is predicted to continue rising and enhancing its intensification (Wood *et al*., 2000). If this phenomenon is not accompanied by a continuous improvement and application of ecosystem management measures the environmental damage caused by agriculture will turn more severe and reducing agroecosystems capability to produce goods and services (Tilman *et al*., 2002). What is more, if contextualize this situation into a global warming scenario (IPCC 2007) the effects of eutrophication will be enhanced (Wood *et al.,* 2000; Moss *et al*., 2011; Jeppesen *et al.*, 2011). Therefore, both land use and climate based management tools (as complex nutrient models) would turn essential to the application of efficient environmental management policies.

5.2- References

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Conclusions

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Conclusions

- The intensification of agriculture has unchained a continuous increase in nutrient pollution to aquatic ecosystems at a global scale.

- Research and application of landscape management policies aiming at reducing agricultural pollution have proved to be effective around Europe.

- However, the European experience at this subject is not reflected at a worldwide scale, where pollution processes continue increasing.

- To sum up, intensive research considering regional differences (e.g. climate, socio-economical development, and geology) would allow the application of region dependant management measures attempting to reduce nutrient pollution.