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Soil texture effects on the transport of phosphorus from agricultural land in river deltas of Northern Belgium, The Netherlands and North-West Germany

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Abstract

The loss of phosphorus (P) from agricultural soils can lead to serious eutrophication of surface water. Within the large river delta areas in Northern Belgium, The Netherlands and North-West Germany, a number of representative soil textures are found (sandy, peaty and clayey soils), each with its own specific combination of pathways for P loss. Based on a literature review, this paper describes for these areas the main eutrophication issues and the factors affecting P loss from these agricultural soils. Aspects of P input, legislation and water quality are discussed. On sandy soils, intensive animal production systems have led to the localized build-up of P in the soil profile with subsequent leaching to shallow groundwaters, and seepage into surface waters. Regulations for production and application of manure on these soils have led to improvements in surface water quality in the last decade, but P concentrations are still too high to maintain satisfactory ecological standards in water. On peat soils, the mineralization of organic matter in the topsoil or in the subsoil can release large amounts of P, leading to high concentrations of P in surface water. However, large P losses can also occur in seepage water from peat deposits in marine sediments containing easily-soluble calcareous P minerals. On clays, incidental losses of P are dominant: surface runoff or preferential flow can lead to loss of freshly applied P, even from soils that are known to have a large retention capacity for P.

Keywords: Phosphorus, subsurface runoff, surface runoff, seepage, incidental losses, preferential flow

Introduction

The transfer of phosphorus (P) from agricultural soil to water bodies can lead to eutrophication, and may cause serious problems when the water is used for recreation, fishery or drinking, due to the abundant growth of algae and weeds (Sharpley & Rekolainen, 1997). In the past, loss of P to the environment was often generally ascribed to erosion of particles with P attached and subsequent removal in surface runoff (overland transport). In many regions in Europe and elsewhere with sloping landscapes (Figure 1) this is the dominant mechanism and hydrological pathway for P transfer (e.g. Baker *et al.*, 1975; Sharpley & Syers, 1979; Sharpley & Smith, 1990). In flat regions, surface runoff is less likely to occur and P transport through the soil profile plays a more dominant role, which can ultimately lead to enrichment of groundwater (Leinweber *et al.*, 1997; Koopmans *et al.*, 2007). The P concentration in surface water is also increased by lateral

groundwater flow via pipe drains or by natural drainage. Flat areas, therefore, might be considered to be high risk in terms of potential eutrophication associated with agriculture.

Surface water contamination with P is a major problem in lowland Europe (European Environment Agency, 1999; Pieterse *et al.*, 2003). Although environmental concern has led to increasing legislative pressure on farmers and some improvement in surface water quality, the EU Water Framework Directive will necessitate further measures to reduce P loss from agriculture to meet ecological targets (Kronvang *et al.*, 2005). In this study, the sources and pathways of P losses that are relevant to flat regions of Europe will be described. The focus will be on the delta areas in lowland Europe, because they vary widely in soil texture and P losses can be large. The processes determining the P losses in these delta areas will be illustrated for the sandy region of Belgium (River Schelde), The Netherlands (Rivers Maas and Rhine), and North-West Germany (Rivers Ems, Jade and Weser). The delta is flat apart from some low hills, remnant of erosion since the Tertiary era, or formed of glacial till during the

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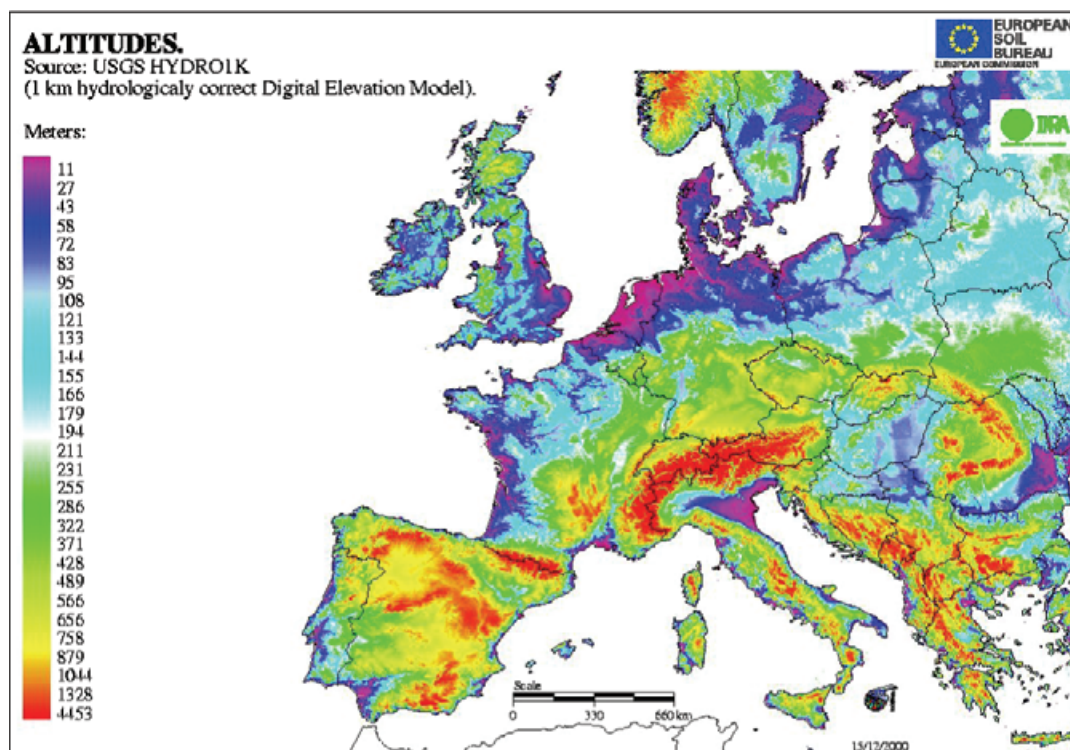


Figure 1 Land altitude in Europe, the purple colour denotes lowland areas.

Pleistocene and partly eroded again by the rivers. The main textures in the region are:

1. sandy soils, mostly acidic, but a small area of calcareous dune sand along the coast;
2. marine clay deposited along the coast;
3. river clay deposited on floodplains along the rivers;
4. peat soils formed in lower areas in former wetlands with stagnant water.

Figure 2 shows the regions in The Netherlands where these different soil textures are found. Three soil textural groups provide the focus for this study, because they characterize the different mechanisms leading to P losses: the groups are (i) sandy soils (Gleysols) where intensive animal production developed since 1950, (ii) peat soils (Histosols), and (iii) clay soils (Fluvisols) deposited along the coasts and rivers.

Soil characteristics

Sandy soils

Within Europe, Belgium and The Netherlands have the largest national P surpluses –defined as the input from all sources minus crop removal – of $> 20 \text{ kg P ha}^{-1} \text{ year}^{-1}$ (De Smet *et al.*, 1996; Steén, 1997). This large excess can be ascribed mainly to intensive animal production. Figure 3 shows animal densities in livestock units per hectare for the 15 EU countries in 1997,

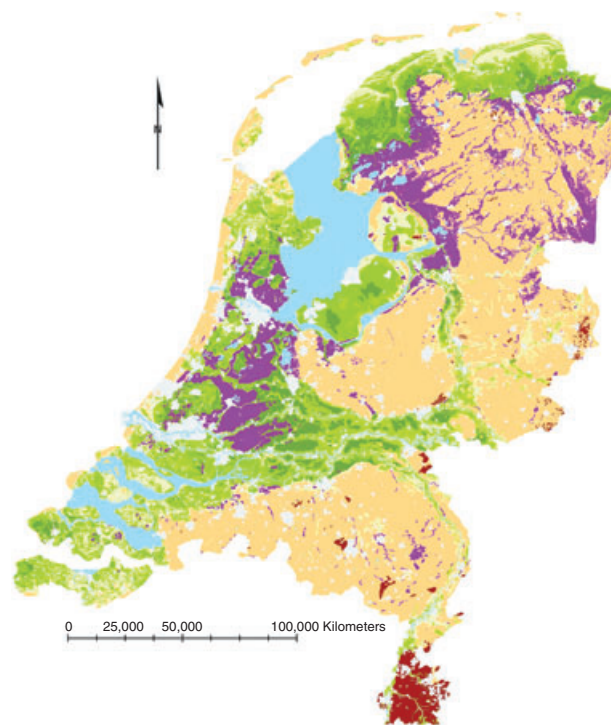


Figure 2 Soil textures in The Netherlands: sand, light brown; peat, purple; river clay, light green; sea clay, dark green (not discussed: loess, red).

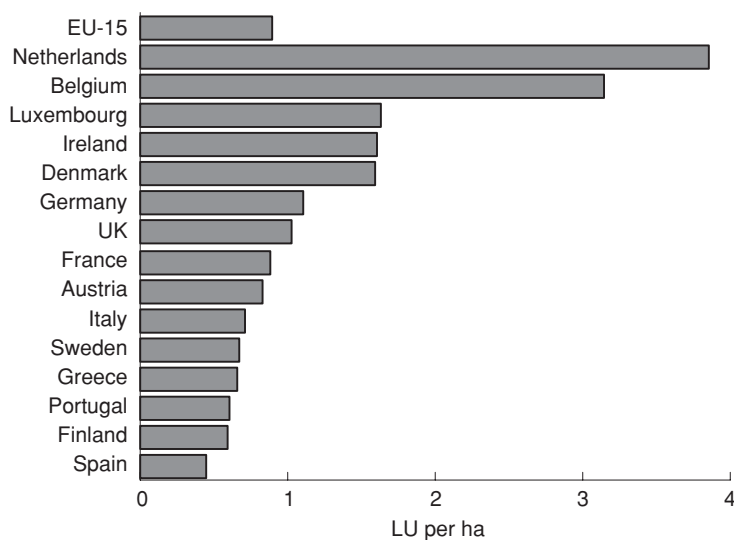


Figure 3 Animal densities in livestock units (LU) per hectare in 15 European countries in 1997 (data Eurostat; Oenema, 2005).

with The Netherlands and Belgium having a density of more than three times the EU average. In the northwest of Germany, the counties of Vechta and Cloppenburg are among the regions with the highest livestock densities in Europe (Leinweber, 1996). Intensive livestock production increased sharply from approximately 1950, stimulated by European price support mechanisms and the presence of the large harbours of Antwerp, Rotterdam and Hamburg, which made it possible to import cheap fodder. The intensive production mainly took place on the previously unfertile sandy soils in Flanders (west and north Belgium, De Smet *et al.*, 1996), in the middle, east and south of The Netherlands (Breeuwsma *et al.*, 1990), and in Lower Saxony and North Rhine-Westphalia in Germany (Leinweber *et al.*, 1997, 2002). On these soils mixed farming (livestock + crop production) on small farms with poor incomes was traditional, and intensive production became an attractive alternative. In Flanders land prices were high, which also stimulated the development of farms with large livestock densities (De Clercq *et al.*, 2001). Locally, the P surplus at the farm scale was often very high, which, in the absence of regulations to prevent the overuse of manure, led to large application of manure and build-up of P in the soil profile (Van Riemsdijk *et al.*, 1983; Breeuwsma *et al.*, 1995; De Smet *et al.*, 1996; Leinweber *et al.*, 1997). This build-up increases the risk of movement of P through the soil profile to shallow groundwaters and surface waters (Sims *et al.*, 1998; Koopmans *et al.*, 2007).

Due to continued large P surpluses prevailing both in Flanders and The Netherlands, the P status of agricultural soils is now (relatively) high. Table 1 shows the relative proportion of agricultural soils with the classification varying from low to very high for Belgium and The Netherlands. For both countries, more than 60% of the agricultural land has a P status satisfactory or higher, requiring no more P than crop offtake. Increasing the P

Table 1 Agricultural phosphorus status of soil in Flanders and The Netherlands with estimated percentage of total area within each category

Land use	Belgium ^a		The Netherlands ^b			
	Arable	Grass	Arable/Grass	Arable/Grass		
Status	Criterion	%	Criterion	%	Criterion	%
Low/moderate	< 12	4	< 19	19	< 20	4
Sufficient	12–18	13	19–25	20	20–30	29
Sufficient plus	19–30	33	26–40	38	30–60	39
High/very high	> 30	50	> 40	23	> 60	28

^aEstimation made for period 2000–2003; analytical method used NH₄-lactate, pH 3.75 (mg P 100 g⁻¹), for both arable land and grassland (Source: BDB, 2005). ^bEstimation made for period 1999–2000; method used NH₄-lactate, pH 3.75 (mg P₂O₅ 100 g⁻¹), for grassland and water extraction (soil:water 1:60, v/v, mg P₂O₅ L⁻¹) for arable land (Source: RIVM, 2004).

status would be beneficial to crop production on less than 20% of the area. This area is insufficient to effectively utilize the national P surplus.

Some older text books suggested that P is immobilized in soils so leaching is not likely: 'Further evidence of the absorption of phosphate by soils is the almost complete absence of phosphate from drainage waters' (Hall & Robinson, 1945, p. 156). However, much earlier studies already recognized that P might be mobile in soils, and ascribed this to other forms of P than ortho-phosphate: 'soluble organic forms, as well as inorganic and organic colloidal forms, might move readily with water' (Stephenson & Chapman, 1931). The capacity of non calcareous soils to retain P can be related to their contents of amorphous Fe and Al, as

determined with an acid oxalate extraction (Schwertmann, 1964). This relationship was first investigated by Beek and co-workers for application of raw sewage water (Beek, 1979; Beek *et al.*, 1980). From the P, Fe and Al in the oxalate extract (respectively, P_{ox} , Fe_{ox} and Al_{ox} , $mmol\ kg^{-1}$) the degree of P saturation (DPS) of a soil can be calculated by:

$$DPS = \frac{100P_{ox}}{0.5(Al_{ox} + Fe_{ox})}. \quad (1)$$

Based on the results of oxalate extractions on samples from different soil horizons, a DPS (%) for a soil profile can be determined (Van der Zee *et al.*, 1990; Schoumans, 2000). In the literature, there has been some debate about the factor 0.5 in the denominator in equation (1) (see e.g. Beauchemin & Simard, 1999; also for examples of the use of DPS in various studies). Therefore, the simple P sorption index (PSI) is also used (Schoumans, 2000):

$$PSI = \frac{P_{ox}}{Al_{ox} + Fe_{ox}}, \quad (2)$$

Theoretically, DPS is not expected to exceed 100%. However, long-term use of manure causes the pH to rise, and previously acidic soils to become neutral or alkaline (Koopmans *et al.*, 2003; Lehmann *et al.*, 2005; Sato *et al.*, 2005), and the amount and proportion of Ca-P phases increase (Leinweber *et al.*, 2002). This implies that calcium phosphates become stable, and that under these conditions large accumulations of P can occur independently of Fe_{ox} and Al_{ox} . In samples from heavily manured fields, DPS values of 179% (Leinweber *et al.*, 1997) and of 208% (Koopmans *et al.*, 2003) have been recorded. Indications for the existence of brushite ($CaHPO_4 \cdot 2H_2O$) were found in sandy soils receiving heavy applications of pig manure (De Haan & Van Riemsdijk, 1986). Sato *et al.* (2005) identified a build up of $CaHPO_4$ after short-term poultry manure application, and of the more stable $\beta-Ca_3(PO_4)_2$ after prolonged application. No crystalline P-forms were found by Sato *et al.* (2005), so solubility of these Ca-P compounds remains relatively high. Lack of stabilization of Ca-P via crystallization can be due to the presence of Mg-P phases in heavily manured soil (Josán *et al.*, 2005). Discontinuation of manure application may result in a decrease of soil pH and an increased solubility of pedogenic Ca-P compounds (Lehmann *et al.*, 2005).

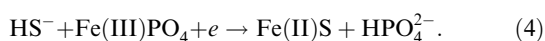
The concept of DPS and PSI has been used for estimation of local P saturation of soils and of remaining sorption capacity in many studies in Belgium (Lookman *et al.*, 1995, 1996; De Smet *et al.*, 1996; Schoeters *et al.*, 1997; VLM, 1997), The Netherlands (e.g. Van Riemsdijk *et al.*, 1983; Breuwsma *et al.*, 1986, 1990), and Germany (Leinweber *et al.*, 1997). In addition, DPS was shown to be correlated with simulated soil solution P (Lookman *et al.*, 1996), water extractable P (Chardon, 1994; Schoumans & Groenendijk, 2000), and leaching of P from lysimeters (Leinweber *et al.*, 1999), as predicted by modelling results (Schoumans, 1995).

Peat soils

It was recognized long ago that peat soils (also called organic soils or Histosols) can release much P to surface waters, especially when taken into agricultural production and drained. Hortenstine & Forbes (1972) measured $30\ mg\ L^{-1}$ of inorganic P in soil solution at 30 cm depth of a mucky peat soil in Florida (USA). Reddy (1983) estimated P export from organic soils in Florida of $16\text{--}168\ kg\ P\ ha^{-1}\ year^{-1}$, Kuntze & Scheffer (1979) reported an export of $4\text{--}15\ kg\ P\ ha^{-1}\ year^{-1}$ for raised-bog soils in Germany, and Van Beek *et al.* (2004) measured $5\ kg\ P\ ha^{-1}\ year^{-1}$ export from a pasture on peat in The Netherlands. The high P export is ascribed to rapid mineralization when organic soils are drained and oxygen can enter the soil (Hortenstine & Forbes, 1972), and to the absence of free Al- and Fe-hydroxides in some peat soils (Kuntze & Scheffer, 1979). Low P sorption capacity of peat soils is ascribed to organic acids blocking sorption sites (Daly *et al.*, 2001). In The Netherlands, peat soils were drained on a large scale in the past; the peat was dried and used for fuel. In the east of the country, so-called reclaimed peat soils were formed, combining high organic matter content with a low content of amorphous Al- and Fe-hydroxides, leading to a relatively high mobility of P (Chardon & Van Faassen, 1999; Koopmans *et al.*, 2001). In the west of the country, water from the river Rhine stagnated during the Pleistocene and large peat deposits were formed, often mixed with layers of sea clay that contain easily-soluble calcareous P minerals. After removal of the peat, large inland lakes were formed, which later were partly transformed into 'polders' by pumping water from the lake. Most polders are situated from 1 to 6 m below average sea level (Van Rees Vellinga *et al.*, 1981), leading to large amounts of seepage water that during winter continuously has to be pumped away into surrounding storage and discharge canals. Concentrations of $3\text{--}4\ mg\ L^{-1}$ can be found in the surface water of polders, which can be ascribed to seepage of water with high concentration of P, originating from decomposing peat in the aerated subsoil (Van Rees Vellinga *et al.*, 1981). However, it can also be caused by seepage of soluble P from marine clay sediments located directly below the peat layers. For 93 groundwater samples in seepage areas in the provinces of Noord- and Zuid-Holland, Rozemeijer *et al.* (2005) found a median value of $1.8\ mg\ P\ L^{-1}$, with values ranging from 0 to $5.6\ mg\ L^{-1}$; for 77 groundwater samples from marine clay soils in the coastal region the median was 0.95 and the range $0\text{--}3.3\ mg\ P\ L^{-1}$. Low concentrations in the seepage water can be explained by a high Fe:P ratio in the water, leading to immobilization of P upon oxygenation of dissolved Fe^{2+} , or to precipitation as Ca-P after CO_2 -degassing and pH rise of groundwater seeping from the clay (Griffioen, 1994, 2006). These high P concentrations in seepage water combined with the decomposition of peat are the main factors responsible for the high concentrations in surface water

found in many places in the coastal provinces of Zeeland and Noord- and Zuid-Holland, with median values of 0.5–1.0 mg P L⁻¹ for the period 1985–2000 (Portielje *et al.*, 2002).

During summer, when evaporation exceeds rainfall, water from the River Rhine is used in peaty areas and in the polders to prevent the peat from drying out. However, the quality of this water differs from the water originally present in peat soils, and especially a high concentration of sulphate in the river water can cause (internal) eutrophication. When sulphate is reduced under anaerobic conditions and sulphide reacts with iron to form pyrite, P bound to the iron is released (Koerselman *et al.*, 1993; Lamers *et al.*, 1998), according to:



In Germany, attempts have been made to restore fenlands, and to take peat soils out of agricultural production (set-aside). One of the measures taken is raising the groundwater level, in order to prevent further oxidation of the peat. However, it was found that this led to increased P concentrations in the soil solution and near-surface groundwater, especially when the redox potential is low and P is released from Fe(III) compounds due to reduction to Fe(II) (Kalbitz *et al.*, 1999).

River clay soils

Along the floodplains of the main rivers in The Netherlands (Rhine, Waal and IJssel) clay was periodically deposited in the past (Holocene). Because of compaction of the subsoil and increasing volumes of river water, dikes had to be built along the rivers to prevent flooding. As a consequence, during the winter and spring, the level of the river water is often higher than the level of the surface water in the surrounding land, which causes seepage of river water. Additional seepage occurs from the higher sandy regions; when this seepage water contained iron, soil layers with a high iron content were formed. The seepage results in high groundwater levels during spring and winter, and temporal flooding of the soil surface when excess surface water cannot be pumped away fast enough. The main agricultural use of these soils is permanent grassland, which is better able to survive extended periods with high water levels provided the soil temperature is low. In order to improve transport of excess rain water, often 20–30 cm-deep trenches are made; the soil that is removed is deposited in between the trenches. Thus, height differences of 30–40 cm are created within the field, and excess rainwater can flow away via surface runoff.

During a 3-year experiment, flows of P and N were monitored on a farm with a heavy clay soil, known to be P-fixing (Van der Salm *et al.*, 2006). The experimental field was pipe

drained, and due to the heavy clay subsoil no seepage or water flow to deeper soil layers occurred. Water from tile drains and trenches left the field via ditches and the flow was monitored continuously. The water was sampled regularly, after each 0.12 mm rainfall surplus. The transport of P via the ditch increased from year to year (1.1, 2.5 and 7.9 kg TP ha⁻¹ year⁻¹, respectively; average 3.8 kg), with trenches contributing 67% to P transport to the ditch. The loss of P was strongly influenced by so-called ‘incidental losses’ (Withers *et al.*, 2003). In this case it was the occurrence of surface runoff: water transport via trenches shortly after application of manure or fertilizer P in spring. Loss of P on a drained clay soil after manure application was also found by Hodgkinson *et al.* (2002). During summer shrinkage cracks formed in the clay, which were responsible for P transport to drains by preferential (bypass) flow (Simard *et al.*, 2000). Thus, although in the field experiment the soil was known to be P-fixing, there was a large loss of P due to cracks in summer and water transport via trenches shortly after application of P (Van der Salm *et al.*, 2006).

Surface water quality and legislation

Agriculture’s contribution to surface water quality

In Flanders (Belgium), a monitoring network for surface waters comprises approximately 1700 locations that are sampled about 15 times a year and where, among other parameters, dissolved total P is measured. Since 1990, a considerable reduction was found in total P concentrations, and the percentage of locations that met the local water quality standard increased (VVM, 2004); however, since 2000, no further improvement has been recorded (Figure 4a). For 2003, it was estimated that households contributed 49% to the total-P load to surface water in Flanders, agriculture 39%, and industry 12%. For 2004 it was calculated that, compared to 1992, the contribution of households decreased by 35%, agriculture by 4%, and industry by 78% (MIRA, 2005). Thus, it can be concluded that the large improvement in the water quality shown in Figure 4a can be mainly ascribed to improved treatment of wastewater from households and industry, and only a small part to changes in agricultural practices.

In The Netherlands, surface water quality data were collected for the period 1985–2000, from sampling stations in locations that were classified as strongly influenced by agriculture (Portielje *et al.*, 2002). For the sandy regions, 80% of the average concentrations of total P for this period were between 0.1 and 0.3 mg L⁻¹. There was a trend of decreasing median values of total P between 1985 and 1995, after that stabilization or even a slight increase was found (see Figure 4b) (Portielje *et al.*, 2004). For 2002, it was estimated that agriculture contributed 58% to the total-P load to surface water in The Netherlands, households 32%, industry 6%, and soils in natural unfarmed areas 4% (RIVM, 2004).

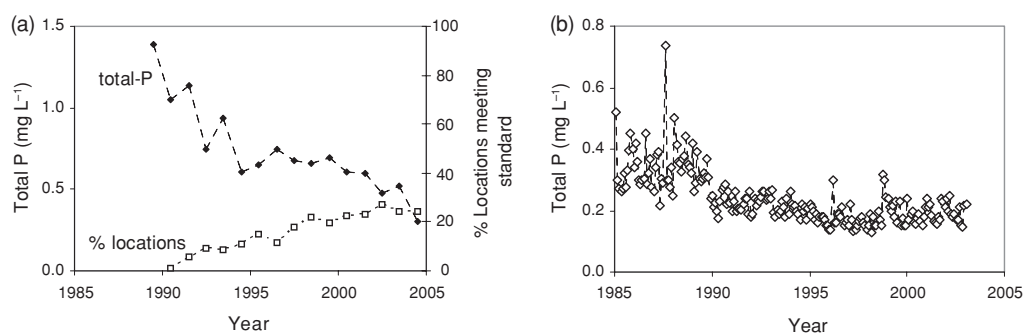


Figure 4 Trend in surface water quality for (a) annual median concentrations of total P in surface water in Flanders, Belgium, with % of sampling locations that meet local water quality standards (after VVM, 2004), and (b) monthly median concentrations of total P in surface water influenced by agriculture in The Netherlands (after Portielje *et al.*, 2004). Note different scales on the y-axes.

Legislation

In Flanders, legislation was first based on fertilizer applications, allowing a certain amount of P to be applied, with limits that could be lowered over time (De Clercq *et al.*, 2001). Special regulations were developed for P-saturated areas, where a further restriction of P fertilizer application has been implemented. During an inventory, the DPS was estimated on approximately 7475 km² of sandy agricultural soils. The area with a DPS exceeding 40% (defined as 'P-saturated' in Flanders) was 73 km², and so-called 'risk areas' (DPS between 30 and 40%) covered 610 km² (VLM, 1997, p. 90). It was also decided not to allow additional production of P and N in animal manures in Flanders, with an obligation for farmers to process or export part of their surplus manure. The proportion that has to be processed or exported increases both with time and with the total amount of manure produced on a farm, and levies have to be paid when targets are not met (De Clercq *et al.*, 2001).

In The Netherlands, legislation was input-oriented initially, with limits set to the annual application of manure in terms of kg P ha⁻¹. At first limits were moderate, but they became stricter with time, with the main goal of stopping the increase in livestock numbers nationally. A manure spreading ban was introduced between 15 September and 1 February, increasing the need for manure storage facilities (Neeteson *et al.*, 2001). To prevent NH₃ volatilization, injection of

manure is compulsory on grassland, and on arable land manure must be incorporated within 24 h. These measures will probably also reduce the risk of incidental P losses. In 1990, legislation based on DPS was proposed (Van der Zee *et al.*, 1990). In brief, it was proposed to forbid further P application to any field when the DPS of the profile exceeds 25%, a level at which the soil is deemed 'saturated'. However, calculations in 1992 showed that, according to this criterion, approximately 400 000 ha of the acidic sandy soils were P-saturated, corresponding to approximately 20% of the total agricultural area in The Netherlands (Reijerink & Breuwsma, 1992). If on these soils manure application had been forbidden, the manure surplus would have been excessive, not only in the regions where intensive animal production is concentrated but also nationwide. For this reason the DPS-based legislation was not implemented on a national scale. From the mid-1990, regulations became based on the farm surplus, for which a mineral accounting system (MINAS) was developed. Above a certain level, the farmer had to pay a levy (Neeteson *et al.*, 2001). However, in 2003, the Court of Justice of the EU decided that the MINAS cannot achieve the goals set by the EU Nitrate Directive, because farmers may pay the levy and continue over-application of manure. Thus, a new regulatory system will be effective from 2006 onwards, under which the MINAS balance approach will be replaced by application standards for manure plus mineral P fertilizers, as enforced by the EU. It is assumed that by 2015

Table 2 Summary of the main problems causing phosphorus loss from agricultural soils with relevant pathways for major soil textures in river delta soils of north-western Europe

Soil textures	Main problem causing P loss	Pathways for P loss
Sandy soils	Excessive inputs and build-up of soil P	Subsurface runoff
Peat soils	Mineralization of organic matter, or easily soluble P minerals in marine clay sediments just below peat layers	Subsurface runoff, or seepage through the topsoil to surface water
Clay soils	Incidental loss of freshly applied P	Surface runoff, or preferential flow

these standards will be equated to crop offtake, so from that year no further enrichment of Dutch soils with P will be permitted.

Conclusions

Soil texture and drainage status in flat delta areas of Europe (Northern Belgium, The Netherlands and North-West Germany) differ widely and influence P loss to surface water in different ways. Table 2 summarizes the main problems causing P loss from agricultural soils with the relevant pathways, for the major soil textures. On sandy soils intensive animal production developed since 1950, with P import substantially exceeding export, has led to local accumulation of P in the soil profile and subsequent P leaching. This caused locally elevated concentrations of P in surface water, and the accumulated P will indefinitely be a long-term source of environmental problems. However, on other soil textures, even higher concentrations of P in surface water can be found: mineralization of peat, both in the topsoil and in the subsoil, releases large amounts of P. Furthermore, upward seepage through marine sediments located beneath the peat layers can cause high P concentrations because of the presence of easily-soluble phosphorus minerals in the sediments. In The Netherlands, this has led to much higher P concentrations in surface waters of regions with peaty soils than in regions with sandy soils that are P-saturated because of manure application. On heavy clay soils along the main rivers, surface runoff occurs as a result of low saturated hydraulic conductivity, and preferential flow down cracks formed during dry periods. Both processes may lead to significant loss of P, even from soils that have a large P-sorption capacity.

Acknowledgements

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