Phosphorus losses at the catchment scale within Europe: an overview

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Abstract

Quantifying P losses to surface waters at different scales and partitioning of the loads into P losses from point sources and diffuse sources are significant future challenges for river basin managers. The agricultural share of P losses to surface waters is, in many river basins, increasing and therefore becoming more important to quantify and analyse. The importance of phosphorus losses from agricultural land was analysed using monitoring data and two different models for 35 micro-catchments (<30 km^2) in the Nordic–Baltic region of Europe, 17 European macro-catchments (250–11 000 km^2) and 10 large European river basins (>50 000 km^2). Average annual phosphorus loss from agricultural land in the micro-catchments varied from 0.1 to 4.7 kg P ha^{-1} and showed no relationship with the short-term P surplus on agricultural land. The average annual total P loss from agricultural land showed equally large variation in the 17 macro-catchments (0.1–6.0 kg P ha^{-1}), but the range was less for the 10 larger river basins (0.09–2.0 kg P ha^{-1}). The annual P loss from the 35 micro-catchments was greatest in the micro-catchments characterized by soil erosion and a high proportion of surface run-off as in the Norwegian catchments. The same pattern was true for the 17 macro-catchments where the model-simulated total P loss from agricultural land was greatest in the catchments in northern and southern parts of Europe. The main diffuse pathways for total P loads in the 17 macro-catchments were simulated with the MONERIS model. On average, soil erosion and surface run-off was estimated to have contributed 53% (4.1–81%), groundwater 14% (0.2–41.7%) and tile drainage water 3% (0–14.0%).

Keywords: Catchments, river basins, agricultural phosphorus loss, retention, pathways

Introduction

Currently, there is increasing concern over the losses of phosphorus (P) from agriculture when assessing the problem of nutrient pollution of the aquatic environment. This is largely because the influence of P from larger point sources has decreased considerably in most countries following improved wastewater treatment (European Environment Agency, 2005; Kronvang *et al*., 2005a). Most concern has been directed towards the mobilization and transfer or delivery of dissolved and particulate P forms to surface waters from the P pool in agricultural soils which has been steadily increasing during the last four to five decades (Leinweber *et al*., 2002; Bundy *et al*., 2005). Thus, agricultural soils are becoming increasingly exposed to the risk of losing dissolved and particulate P forms by erosion and leaching via surface (overland flow) and subsurface (preferential flow) hydrological pathways (Grant *et al*., 1996; Sharpley & Rekolainen, 1997; Haygarth *et al*., 1998). Since the 1970s, numerous authors have reported the interactions between soil P and water at small scales on the basis of either laboratory, plot-scale or field-scale experiments (Sharpley, 1985; Grant *et al*., 1996; Laubel *et al*., 1999). The problems arising from incidental P losses from agricultural land after fresh additions of P in fertilizer and manure on fields have also recently been addressed (Withers *et al*., 2003).

Quantification of the main sources (different point sources and diffuse sources) of P at the catchment scale (source apportionment) has been routinely conducted using a simple mass-balance approach where the measured P load at the

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catchment outlet is subdivided into inputs from point sources and diffuse sources (e.g. Kronvang *et al*., 1993). However, this relatively simple approach can only provide end-users with an overall estimate of the magnitude of P sources at the catchment scale.

The need to identify specific contributing P-source areas in catchments has highlighted the importance of delivery in the P transport process (Gburek *et al*., 1996; Gburek & Sharpley, 1998; Heathwaite *et al*., 2000). This requires quantification of the importance of the different pathways for dissolved and particulate P forms, and an understanding of how fields some distance from a watercourse may still be hydrologically connected. Different catchment-scale approaches to quantifying P losses from agricultural land using process-based models (e.g. Groenendijk and Kroes, 1999), export coefficient models (e.g. Johnes *et al*., 1996), lumped statistical models (e.g. Kronvang *et al*., 2003) or risk mapping models (e.g. Heathwaite *et al*., 2003) have received increasing attention during recent years. The development of a more holistic understanding of mobilization, transfer, storage and transport dynamics of P at the catchment scale has been required by end-users such as catchment managers, who need to be able to implement cost-effective mitigation strategies for reducing agricultural P losses (Kronvang *et al*., 2005b).

Information from catchment-scale studies, whether experimental, routine monitoring or modelling studies, brings new evidence of the magnitude of agricultural P losses and the magnitude of different P pathways in different European catchments (Kronvang *et al*., 2005b). A review of the recent information on P losses from agricultural land in European catchments may provide valuable insight into questions such as: (i) are there any methodological problems that arise when assessing P losses from agricultural land at different catchment scales? (ii) what is the importance of scale and regional differences in agricultural P losses? (iii) what are the important factors influencing P mobilization and transport to surface waters?

This paper reviews the magnitude of P loss from a range of catchments at different scales in Europe with the purpose of estimating and discussing the importance of agricultural land as a P source, and the factors influencing agricultural P loss in different regions. We also assess and discuss the methodological problems arising from catchment-scale quantification of agricultural P losses, and the issue of P delivery from agricultural land to surface waters.

Materials and methods

Catchment data and models applied

Descriptive catchment information, and phosphorus concentration and load data from a range of European catchments and river basins were collected from various sources to review the current information on agriculturally derived phosphorus loads (Vagstad *et al*., 2001; Kronvang *et al*., 2003; Schreiber *et al*., 2003; European Environment Agency, 2005).

Phosphorus concentration and load data from 35 micro-catchments in the Nordic and Baltic region were examined against descriptive catchment data covering the period 1993–1997 (Table 1). The catchment areas are usually less than 20 km^2, and with a few exceptions, agricultural land constitutes more than 50% of the land use. Cereals are the dominant crop types in most of the catchments, although areas of root crops, vegetables and/or pastures are substantial in some of the catchments. Livestock densities are generally at a moderate level and usually below 1 Livestock Units (LU) ha^{-1}.

Descriptive catchment data and phosphorus concentration and load data from 17 larger national catchments in Europe are also used in the analysis of agricultural phosphorus losses (Table 2). These catchments varied in size from 250 to 11 000 km^2 and only one of the 17 catchments analysed was crossing a national boundary. In addition, the results from modelling agricultural P losses within 10 larger river basins, including the supra-national Danube basin, using the MONERIS model (Behrendt *et al*., 1999; Schoumans & Silgram, 2003) are included in the paper (Table 3). The catchments and river basins considered are situated in different parts of Europe and cover large variation in catchment size, climate, soil type and land use.

Table 1 Characteristics of the 35 micro-catchments included in this study shown as the range in total catchment area, precipitation, phosphorus balance, average annual concentration of dissolved reactive phosphorus (DRP) and total phosphorus

Country (no. of sites)	Total area (km^2)	Precipitation (mm $year^{-1}$)	P balance (kg P ha^{-1})	DRP (mg L^{-1})	Total-P (mg L^{-1})
Estonia (2)	9.7–25.5	676–730	n.i.	0.01–0.16	0.04–0.36
Finland (4)	0.12–15.4	513–662	n.i.	0.01–0.06	0.11–0.68
Latvia (3)	3.7–9.6	569–730	−3 to +1	0.02–0.19	0.03–0.27
Lithuania (3)	1.7–13.6	501–580	−6 to −1	0.04–0.07	0.07–0.14
Sweden (10)	1.9–16.8	477–864	−9 to +12	0.05–0.14	0.06–0.28
Norway (7)	0.7–20.0	585–1230	+5 to +9	0.04–0.06	0.12–0.93
Denmark (6)	4.7–11.4	614–993	+3 to +13	0.005–0.1	0.07–0.23

n.i., no information.

Table 2 Descriptive information of the macro-catchments in Europe

Catchment	Size (km^2)	Average annual precipitation (mm)	Dominant soil types	Population density (inhabitants per km^2)	Agricultural land (%)	Surface water area (km^2)
Core catchments						
Vansjø-Hobøl, Norway (NOV)	690	810	Clay	20	17	45.7
River Yorkshire-Ouse, England (ENO)	3314	923	Loam	98	60	20.4
River Enza, Italy (ITE)	901	1000	Silt/sand	325	48	1.9
Non-core catchments						
Eurajoki, Finland (FIE)	1336	559	Clay/moraine	20	23	176.2
Rönne Å, Sweden (SWR)	1897	700	Loam/sand	49	33	58.1
Odense Å, Denmark (DEO)	486	740	Loam	124	71	7.2
Uecker, Germany (GEU)	2430	540	Sandy loam	82	64	63.2
Susve, Lithuania (LIS)	1165	675	Sandy loam/peat	18	62	2.4
Vechte, Germany/The Netherlands (NLV)	3970	730	Sand/peat	200	73	8.0
Lough Derk and Ree, Ireland (IRL)	10 797	1150	Clay	25	73	433
Attert, Luxembourg (LUA)	254	900	Silt/clay	50	51	0.8
Gurk, Austria (AUG)	2602	905	Sand/loam	90	35	33.1
Zelivka, Czech R. (CRZ)	1187	669	Loam	45	64	24.2
Kapos, Hungary (HUK)	3295	690	Loam	46	62	25.8
Vilaine, France (FRV)	10 533	773	Clay	103	40	18.8
Guadiamar, Spain (SPG)	1357	555	Luvisols	114	52	4.3

Data for Greece are not available.

Table 3 Characteristics of the 10 larger river basins investigated

River basin	Area (km^2)	Specific run-off (mm $year^{-1}$)	Max altitude (m)	Agricultural area (%)	Urban areas (%)	Population density (1000s)
Axios	24 438	187	2482	40.1	1.5	81.9
Elbe	148 270	159	1457	61.4	5.9	166.0
Po	73 761	681	4453	45.8	4.9	191.9
Rhine	185 300	430	3727	51.8	7.9	309.0
Vistula	194 424	185	2317	60.0	2.5	121.3
Danube	802 888	269	3652	53.7	3.9	102.3
Daugava	87 826	301	311	32.8	0.7	31.9
Ems	12 714	367	326	80.1	6.7	212.1
Odra	118 148	149	1390	62.0	4.0	134.0
Weser	46 091	371	1043	61.5	7.2	198.6

Models applied for calculation of agricultural phosphorus losses

The method generally referred to in this paper for estimating phosphorus transport is an interpolation method (Kronvang & Bruhn, 1996). It is assumed that concentrations of P in river water have been measured a number of times during a given year and that daily discharge values exist for the selected measurement site. The method then uses interpolated concentration values for days when P has not been measured. The P concentrations are measured at the days denoted by $t_i = 1, 2, ..., n$. Concentrations are denoted C, $i = 1, 2, ..., n$. Let t_0 and t_{n+1} be the start and the end of the year, respectively. The assumption is made that $c_0 = c_1$ and $c_{n+1} = c_n$. Then the P load is estimated by

$$\hat{L} = \sum_{i=0}^{n-1} \sum_{t_i < t \leq t_{i+1}} q_t \frac{c_i(t_{i+1} - t) + c_{i+1}(t - t_i)}{t_{i+1} - t_i}, \quad (1),$$

where $\sum$ denotes summation, i.e. $\sum_{i=0}^{n-1}$ denotes summation of values for the index in the interval 0 to $n-1$, and $\sum_{t_i < t \leq t_{i+1}}$ denotes summation of values for t in the interval t_i to t_{i+1}, but t_i is not included in the interval, t denotes a day between two measurement days, q_t is the daily discharge for day t. The assumption that $c_0 = c_1$ results in $c_{\text{interpolated}} = c_1$, for $t_0 < t \leq t_1$, and the assumption $c_{n+1} = c_n$ results in $c_{\text{interpolated}} = c_n$, for $t_n < t \leq t_{n+1}$.

Two different models, MONERIS and Source Apportionment (SA), were used for calculating P losses from agricultural areas in the 17 catchments. The MONERIS and SA models are lumped annual models that have been comprehensively described in Schoumans & Silgram (2003).

The SA model is based on the assumption that the total P transport at a selected river measurement site (L_{river}) represents the sum of the components of the P discharges from point sources (D_P), the P losses from anthropogenic diffuse sources (LO_D) and the natural background losses of nutrients (LO_B). Furthermore, it is necessary to take into account the retention of P in surface waters (R). This may be expressed as follows:

$$L_{river} = D_P + LO_D + LO_B - R, \quad (2)$$

The aim of the SA model was to evaluate the contributions of specific point and diffuse sources of nutrients to the total riverine nutrient load, i.e. to quantify the nutrient losses from anthropogenic diffuse sources (LO_D) as follows:

$$LO_D = L_{river} - D_P - LO_B + R, \quad (3)$$

The model MONERIS (MOdelling Nutrient Emissions in RIver Systems) was developed for the investigation of the nutrient inputs via various point and diffuse pathways in German river basins (Behrendt *et al*., 1999). MONERIS is a steady-state distributed parameters model with an annual time step. The basis for the model is data on run-off and water quality for the river catchments studied and also a geographical information system (GIS), in which digital maps as well as extensive statistical information are integrated. While the point inputs from municipal wastewater treatment plants (WWTPs) and from industry are directly discharged into the rivers, the diffuse entries of nutrients into the surface waters represent the sum of various pathways which have been taken by the individual components of the run-off. As a consequence, four main pathways are considered in the model: (i) direct nutrient input on the water surface area by atmospheric deposition; (ii) nutrient input into the river systems by surface run-off; (iii) nutrient input via interflow which represents a fast subsurface flow component (flow in pipe drains and horizontal flow through the upper soil); (iv) nutrient inputs via base flow (groundwater) realized by the slow subsurface flow component. The MONERIS model simulates P discharges from point sources (WWTP and scattered dwellings) and the major pathways (groundwater, pipe drainage, soil erosion and surface run-off, and atmospheric deposition). The performance of the MONERIS model has been validated based on data from three of 17 catchments included in this paper [The Norwegian: NOV, the English (ENO) and the Italian (ITE) macro-catchments] (Schoumans *et al*., unpublished data).

Statistical methods applied

The phosphorus concentration and discharge relationship was modelled by the non-parametric and robust curve-fitting method LOWESS (Locally Weighted Scatterplot Smoothing, Cleveland, 1979). A Monte Carlo analysis of the estimation of annual total phosphorus (TP) and dissolved reactive phosphorus (DRP) load was conducted using daily concentration values from the River Odense during a 13-year period. Table 3 shows the number of days with sampling and analysis of phosphorus concentrations together with the true annual transport estimated with the linear interpolation methods (Kronvang & Bruhn, 1996). In the case of missing daily concentration data, linear interpolation was used to fill the gaps. A monthly (30-day sampling period) and fortnightly sampling programme was imitated, giving 35 annual transport estimates with monthly sampling and 15 annual estimates with fortnightly sampling.

Results

Phosphorus dynamics and estimation of riverine P loads

Great differences were observed in the seasonal variation in phosphorus concentrations between the larger European catchments (Figure 1). The seasonal pattern is most extreme in the river basins in the northern and southern parts of Europe experiencing either snow melt or intense rainfall.

The discharge and P concentration relationships (LOWESS) established showed a decrease in phosphorus concentration with increasing discharge in many of the macro-catchments (Figure 2). Such a phosphorus response is typical for rivers receiving large and constant discharges of phosphorus from point sources (dilution effect). In many cases, P concentrations can be very high at peak discharges, reflecting the importance of particulate P from erosion (soil erosion, stream bank and bed erosion).

Regional differences in weather and the hydrological regime in catchments together with local variations in P emissions from various point and diffuse sources have a great impact on the accuracy of estimating the riverine P loads, depending on the sampling protocol used at specific monitoring stations (Kronvang & Bruhn, 1996). As an example a Monte Carlo analysis was conducted based on a 12-year sampling period (1989–2002) with nearly daily total P concentrations in the River Odense, Denmark (485 km^2). A monthly or fortnightly sampling programme was assessed with the Monte Carlo analysis as this sampling frequency was generally used in the routine monitoring programmes run in our studied catchments. The outcome of the analysis resulted in a biased annual P load estimate (−3% to −2%) with a low precision (17–20%). Larger uncertainties (larger bias and less precision) of annual load estimates of total P could very well be the case in rivers showing greater seasonal variation and more extreme concentrations of P than the Odense river, like the rivers Uecker, Attert, Kapos and Guadiamar shown in Figures 1 and 2.

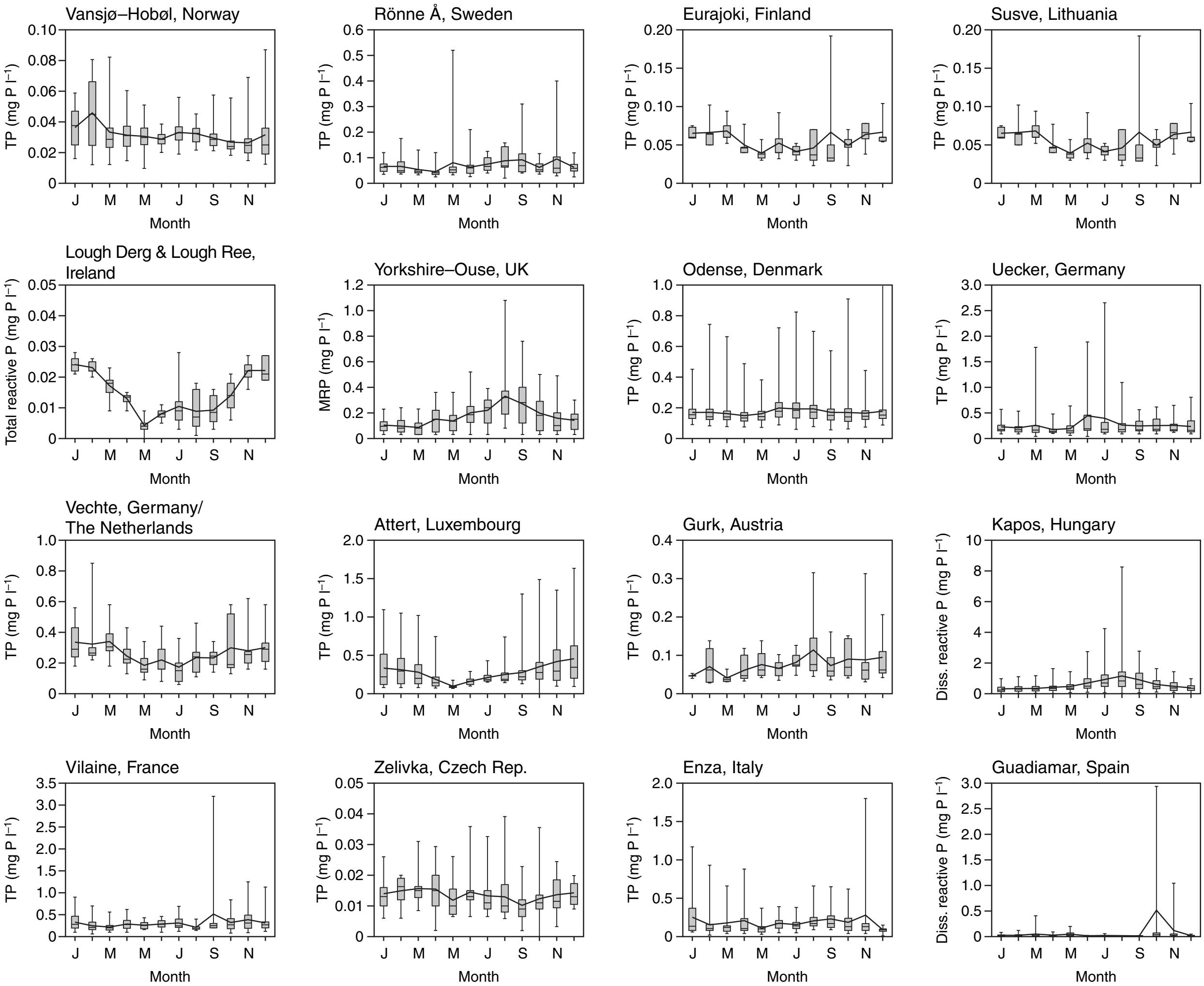

Figure 1 Box–Cox plot of phosphorus concentrations at the outlet stations of 16 river basins during the sample period (see Table 2).

Phosphorus losses in Nordic and Baltic agricultural micro-catchments

The TP and DRP concentrations at the outlets of the 35 micro-catchments and the estimated soil surface P balances for agricultural land are shown in Table 1. The measured mean annual TP and DRP losses in the 35 catchments during the period 1993–1997 are shown in Figure 3. Results are presented per unit area of agricultural land, corrected for the assumed P loss from non-agricultural land. The average annual TP loss shows great variability, with an overall range in TP losses from less than 0.1–4.7 kg P ha^{-1} during the 5-year period. The observed P losses in the Norwegian catchments are generally higher than those observed in the other countries, although substantial variations also occur among the different Norwegian catchments (Figure 3).

In addition to the large spatial variability, the results also show remarkably high temporal variability (seasonal and annual). Table 4 presents examples of the annual variability in P losses during the period 1993–2002 in a selected number of catchments where cereals are the dominant crop. Results show that the ratio between the highest and the lowest observed annual P losses of the individual catchments ranges from approximately two to 15 during the 10-year measurement period, much of which is explained by variation in flow (Figure 4).

The possible explanations for the large ranges in total P loss from agricultural land are not straightforward, although some of the cases might be explained by high levels of soil erosion and a relatively high proportion of overland flow – the case in most of the Norwegian catchments. A large base flow component in the Danish catchments shows that groundwater contributes a considerable part of the total annual run-off, which often implies a lower level of P concentration and less variability in P losses than in cases where surface run-off and preferential flow are more important pathways for P transfer.

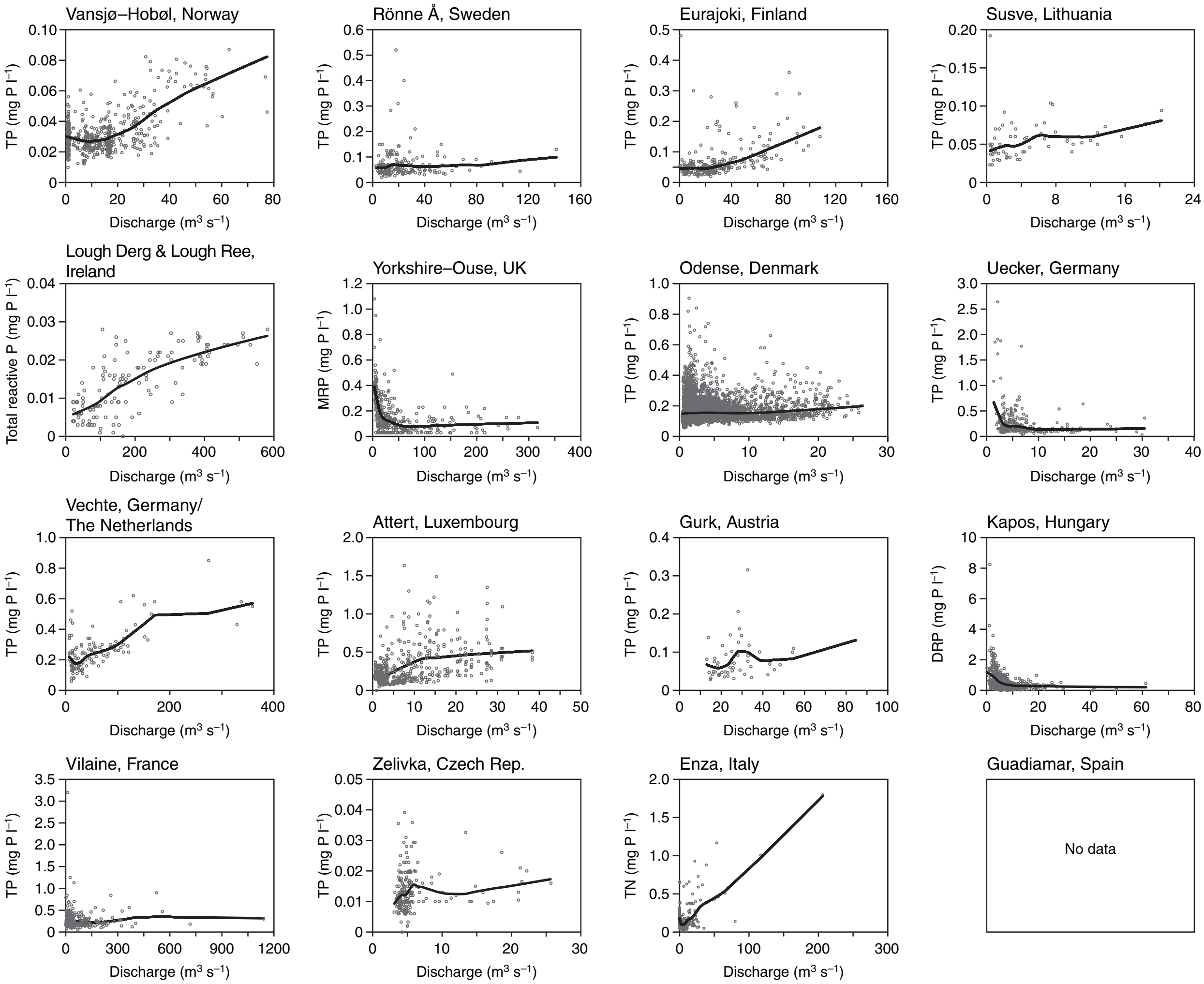

Figure 2 Relationships between discharge and phosphorus concentrations at the outlet station in 15 river basins during the sampling period (see Table 2).

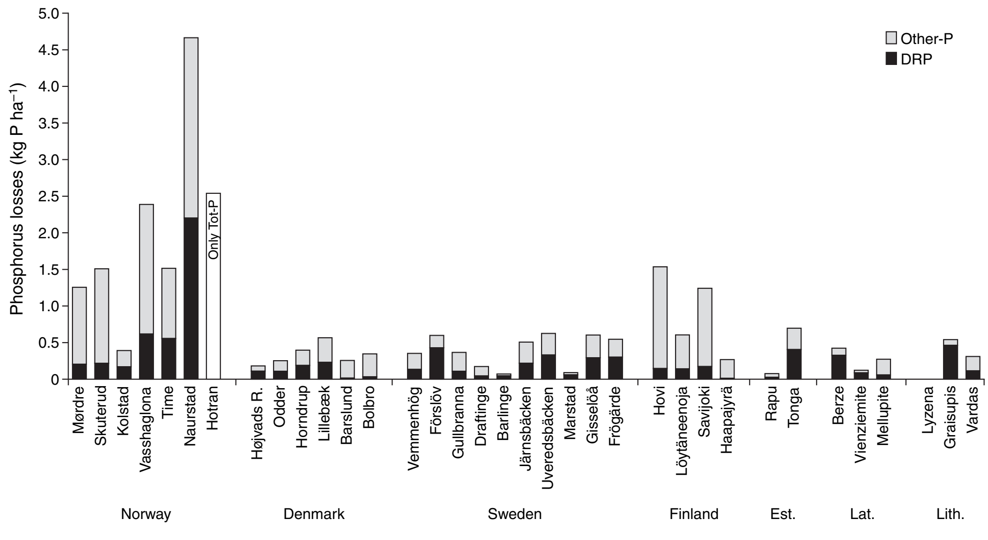

Figure 3 Measured total P and dissolved reactive P in 35 small agricultural catchments in the Nordic–Baltic region during 1993–1997, shown as P losses per hectare of agricultural land (redrawn from Vagstad *et al.*, 2001).

Table 4 Average, annual measured total P loss from catchments in Norway, Denmark, Sweden and Latvia (1993–2002)

Catchments	Average annual loss (kg P ha^{-1})	Minimum annual loss (kg P ha^{-1})	Maximum annual loss (kg P ha^{-1})	Max/min ratio	CV
Denmark					
Horndrup	0.48	0.15	0.96	6.4	0.47
Lillebæk	0.63	0.16	0.71	7.2	0.48
Højvands Rende	0.21	0.03	1.15	15.7	0.62
Latvia					
Mellupite	0.17	0.09	0.27	3.0	0.43
Berze	0.31	0.12	0.52	4.2	0.46
Norway					
Mørdre	1.59	0.89	3.96	4.4	0.54
Skuterud	2.58	0.92	5.88	6.4	0.62
Kolstad	0.56	0.21	1.26	6.0	0.63
Sweden					
Ulverød (S)	0.94	0.40	2.30	5.8	0.61
Marstad (S)	0.12	0.09	0.16	1.8	0.33
Gisseløå (S)	0.51	0.33	0.82	2.5	0.41

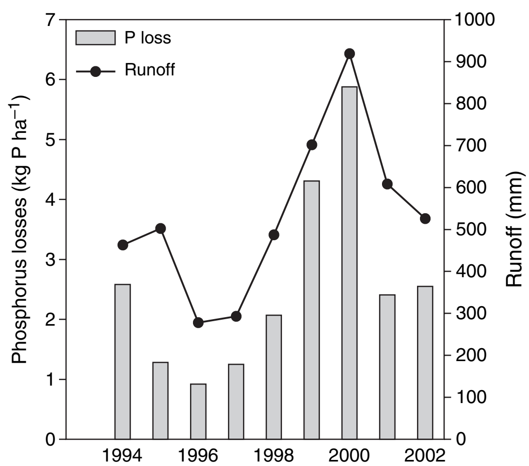

Figure 4 Annual P losses versus annual run-off in the Skuterud catchment, Norway.

The fraction of DRP loss as a proportion of the TP loss is highly variable from one catchment to another, ranging from less than 10% to more than 90% (Figure 3). This indicates substantial variation in terms of possible biological impacts in receiving waters. In this respect, it is also worth mentioning that the N/P-ratio varied from around 10 to 300, bearing in mind the possible linkages between nutrient compositions and algal community development in surface waters (Kronvang *et al.*, 2005a).

Phosphorus soil balances are frequently used as an indicator of water quality impacts related to agriculture (e.g. OECD, 2001). However, this study of micro-catchments did not reveal any significant relationship between measured total P losses and calculated P balances at the soil surface (Figure 5). This result is not surprising, bearing in mind that the type of process determining the P losses at the micro-catchment scale tends to be at the field scale where the soil P sorption capacity buffers the impact of increased P surplus (Withers *et al.*, 2001).

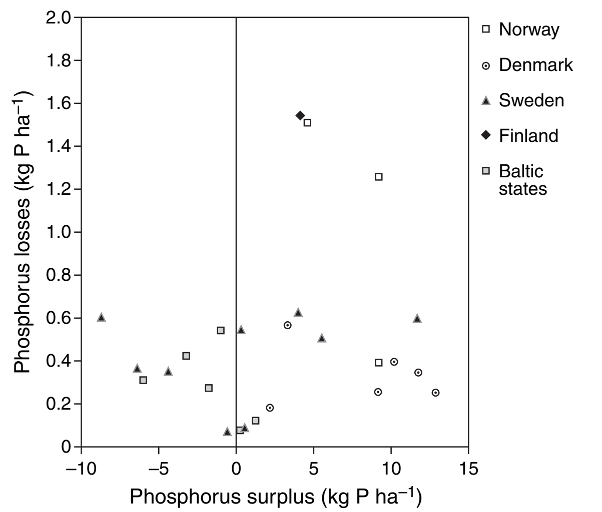

Figure 5 Relationship between measured P losses (1993–1997) and soil surface P balances in the 35 study catchments (redrawn from Vagstad *et al.*, 2001).

Phosphorus losses in European macro-catchments

The average annual run-off, phosphorus concentrations and export coefficients showed great differences between the 17 European catchments (Table 5). A multiple regression analysis was performed between the export coefficient of TP (P_{ex}) and percentage agricultural land (A), area of surface water in the catchment (S) (km^2), population density in the catchment (P) (inhabitants per km^2), total catchment area (CA) (km^2) and run-off (R) (mm). A significant relationship (P = 0.0034) was established between the phosphorus export from the catchments and three of the explanatory variables:

$$P_{ex} = 0.0011R + 0.0020P - 0.0015S(N = 16; r^2 = 0.65).$$

The main factors governing phosphorus export were catchment hydrology (total run-off), population density in the catchment as a measure of point source pollution and area of surface water in the catchment as a proxy for phosphorus retention in rivers, lakes and reservoirs (Bechmann & Stålnacke, 2005).

A riverine load apportionment of the net TP export subdivided into the gross P loss from point sources, agricultural land and other diffuse sources (P loss from non-agricultural land) to surface waters and the P retention in surface waters

Table 5 Average annual run-off, total P concentration and total P export from each of the 17 macro-catchments

Catchments	Sampling period	Average annual run-off (mm)	Average annual phosphorus concentration (mg P L^{-1})	Average annual phosphorus export (kg P ha^{-1})
Core catchments				
Vansjø-Hobøl, Norway	1989–2000	507	0.032	0.17
River Yorkshire-Ouse, England	1990–2000	410	0.180	0.55
River Enza, Italy	1991–2000	355	0.230	1.00
Non-core catchments				
Eurajoki, Finland	1989–2000	236	0.068	0.18
Rönne Å, Sweden	1989–2000	285	0.071	0.13
Odense Å, Denmark	1989–2001	290	0.172	0.52
Uecker, Germany	1995–2000	92	0.267	0.137
Susve, Lithuania	1997–2000	144	0.057	0.089
Vechte, Germany/The Netherlands	1993–2000	304	0.243	0.90
Lough Derk and Ree, Ireland	1999–2001	552	0.014	0.10
Attert, Luxembourg	1999–2002	521	0.296	1.28
Gurk, Austria	1991–1999	347	0.071	0.28
Zelivka, Czech R.	1993–2000	157	0.014	0.022
Kapos, Hungary	1990–1996	58	0.591	0.22
Vilaine, France	1990–1999	218	0.305	0.60
Pinios, Greece	1988–1996	537	0.58	4.10
Guadiamar, Spain	1992–1997	54	–	0.18

(rivers and lakes) was conducted using on 16 of the macro-catchments two models (MONERIS and SA). The average results of the two models show that agriculture was the dominant P source in 12 of the 16 catchments analysed, with point sources dominating in the remaining four catchments (Figure 6). On average, the share of the gross TP export was 32% (23–36%) for agriculture, 25% (13–33%) for point sources, 22% (17–26%) for other diffuse sources and 22% (18–26%) for retention within the 16 macro-catchments.

The model estimated average annual specific TP loss from agricultural land varied greatly between the catchments (0.1–6.0 kg P ha^{-1}) (Figure 7). The average model-simulated TP loss from agriculture was the largest in the Mediterranean region catchments of Europe (GRP: Greece; ITE: Italy) and the colder Nordic region (NOV: Norway; FIE: Finland) (Figure 7). The only exception from this pattern was the large estimated TP loss from agricultural land for the catchment in Luxembourg (LUA) (Figure 7). The range in the specific TP loss from agricultural land in each catchment estimated with the two models indicates that differences between the two outputs seem to vary from catchment to catchment.

Phosphorus losses in supra-national European river basins

Agriculture is also one of the largest contributors to phosphorus pollution of the aquatic environment in

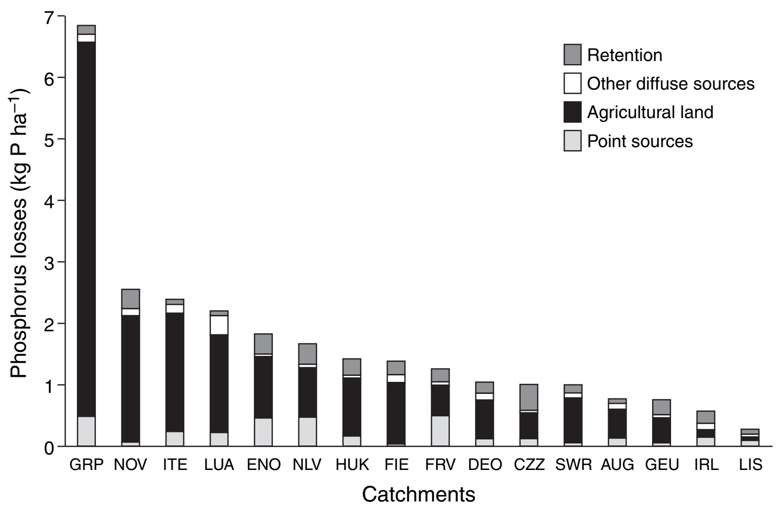

Figure 6 Source apportionment of total P export and total P retention for 17 European catchments based on the source apportionment method.

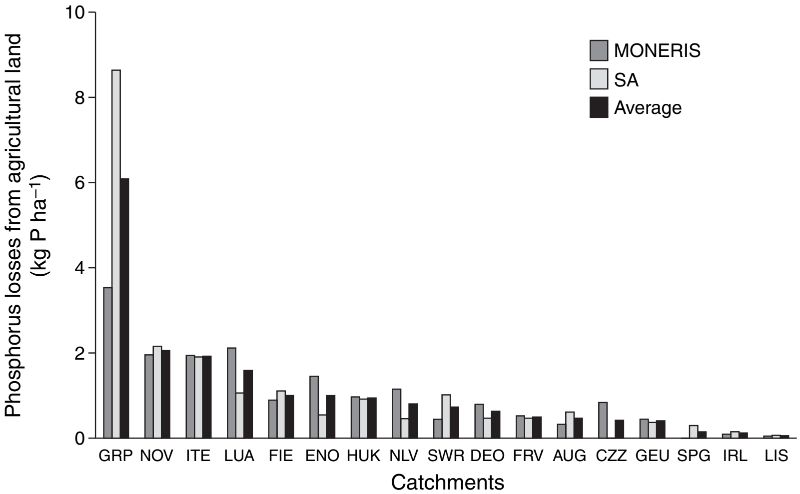

Figure 7 Agricultural total P losses from 17 European catchments calculated with the MONERIS model, the source apportionment model and the average of the two models.

supra-national river basins in Europe as seen from a partitioning of total P loads in 10 large European river basins using the MONERIS model (Figure 8). However, the dominant sources of P changes, from river basin to river basin, with the agricultural share varying between 13 and 83% of total P loads (Figure 8). The agricultural share of the total P loss is highest in the rivers Ems (83%) and Weser (61%) and lowest in the rivers Axios (13%), Odra (17%), Rhine (23%) and Po and Daugava (24%). The specific TP loss from agricultural land varies greatly between the river basins (0.09–2.0 kg P ha^{-1}).

An analysis of total P emissions and agriculturally derived P emissions for the contributing areas of the countries within the Danube river basin is shown in Table 6 using the MONERIS model for the period 1998–2000. Even though the area-specific total P losses within the basin vary only by a factor of two (0.67–1.24 kg P ha^{-1} $year^{-1}$), the area-specific losses from agriculture vary by a factor of four (0.14–0.57 kg P ha^{-1} $year^{-1}$) (Table 6). The agricultural share of the total P emissions, in each country contributing to the basin, increases as the area-specific total P emission decreases. This shows that the relative share of agricultural P in larger river basins is to a large extent affected by the point source share, which in turn is a result of population density, industrial activities and wastewater treatment. As wastewater treatment is more efficient in the north-western countries discharging into the Danube river basin, the agricultural share of total P emissions is also generally higher in these countries (Table 6).

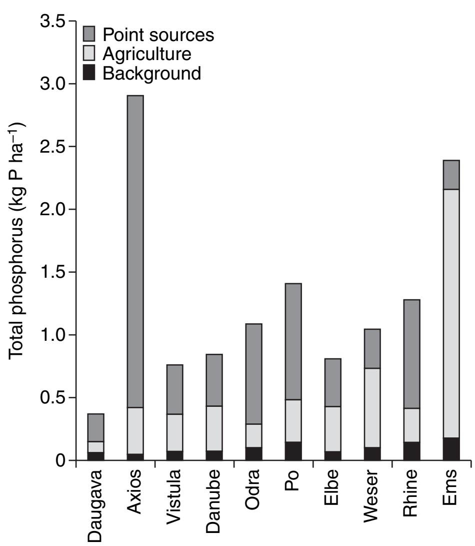

Figure 8 Load partitioning of total P export from 10 large river basins in Europe.

Apportioning sources and pathways of phosphorus in catchments

The outcome of MONERIS model calculations of the share of point sources and the major pathways for the TP loads within the 17 macro-catchments in Europe are shown in Figure 9. The average contribution of TP from WWTP and scattered dwellings is, respectively, 18.2% (range 4.7–38.5%) and 10.0% (range 1.6–38.7%). The average contribution of TP from the different diffuse pathways is 53.4% for surface run-off and soil erosion, 14.4% for groundwater, 2.8% for pipe drainage and 1.3% for atmospheric deposition on surface waters. However, large variations between the 17 catchments in the contribution of TP from different hydrological pathways were observed (Figure 9). Thus, the TP loss via soil erosion and surface run-off contributed 4–81%, the TP loss via groundwater contributed 0.2–42%, the TP loss via pipe drainage water contributed 0–14% and the atmospheric deposition of TP on surface waters contributed 0.1–4% (Figure 9).

Table 6 Total phosphorus emissions and phosphorus emissions derived from agricultural land from the countries draining to the Danube transboundary river basin during the period 1998–2000

	Area drained (km^2)	Total emissions (t P year^{-1})	Area-specific emissions (kg P ha^{-1})	Agricultural emissions (t P year^{-1})	Area-specific agricultural emissions (kg P ha^{-1})	Agricultural share of total emissions (%)
Germany	56 630	4759	0.84	2606	0.46	55
Austria	80 850	7126	0.88	2825	0.35	40
Czech Republic	21 690	2112	0.97	1240	0.57	59
Slovakia	47 210	4012	0.85	2129	0.45	53
Hungary	92 770	6991	0.75	2036	0.22	29
Slovenia	16 410	2029	1.24	638	0.39	31
Bosnia-Herzegovenia	34 630	3297	0.95	1186	0.34	36
Croatia	37 600	3454	0.92	1600	0.43	46
Yugoslavia	88 490	9311	1.05	2364	0.27	25
Romania	222 330	16 007	0.72	8003	0.36	50
Bulgaria	55 190	5214	0.94	2150	0.39	41
Moldova	12 330	827	0.67	567	0.46	69
Ukraine	33 930	2429	0.72	1420	0.42	58
Other countries	2820	213	0.76	40	0.14	19
Total	802 890	67 781	0.84	28 804	0.36	42

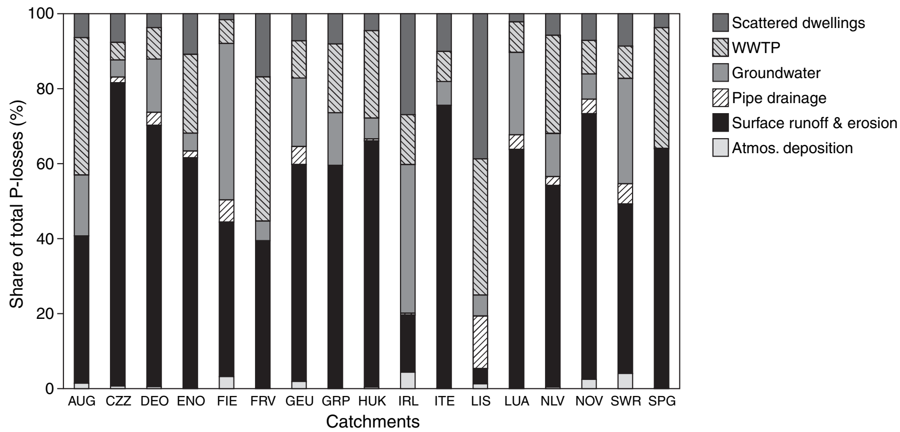

Figure 9 Proportion of sources and pathways for the total P loss from the 17 macro-scale catchments in Europe simulated with the MONERIS model.

Discussion

The TP loss from different European catchments derives from both point sources and diffuse sources, the latter becoming increasingly important as better methods for treatment of point sources are implemented (European Environment Agency, 2005; Kronvang *et al.*, 2005a). Thus, the model-simulated agricultural share of total P export dominated in 12 of the 17 European macro-catchments and in three of the 10 larger river basins analysed. However, the outcome of routine monitoring programmes using infrequent sampling often result in biased estimates of the P transport, which seems generally to be underestimated and with a relatively poor precision, as shown by authors for streams in different parts of Europe (Rekolainen *et al.*, 1991; Kronvang & Bruhn, 1996). The uncertainty in the estimation of P transport can be of great significance when trying to estimate agricultural P losses by a simple mass-balance approach (e.g. load partitioning) or a calibrated model as the bias (underestimation) will be primarily associated with the agricultural share of the TP loss.

The developed multiple regression model for the TP export from 17 European macro-catchments reveals that run-off, population density and the area of surface waters in the catchment are important factors governing the resulting TP export. Surprisingly, the proportion of agricultural land does not contribute significantly to the regression model. This could be due to the importance of TP retention in surface water in the macro-catchments, which was also shown to be an important component of P fluxes at the catchment scale (Figure 6). Other authors have shown that TP retention is of great significance when establishing P budgets for larger catchments (Kronvang *et al*., 1999, 2005a; Kneis *et al*., 2005).

Storage of P in riparian buffer zones and in surface water sediments may also play an important role in delaying the catchment response when attempting to control P losses at the larger catchment scale (Jeppesen *et al*., 2003; Kronvang *et al*., 2005a). Following implementation of general or targeted mitigation measures, the response of catchments will be suppressed due to a P resilience, created by releases of dissolved P from sediments in rivers, lakes and reservoirs, or delivery of dissolved and particulate P (PP) from riparian zones through leaching and stream bank erosion(Laubel *et al*., 2003; Kronvang *et al*., 2005b).

Agriculturally dominated micro-catchments provide the best estimate of P losses from agricultural land because no other major anthropogenic sources are present and P retention is minimal over a period of several years. We found wide differences in the annual TP losses at this catchment scale when quantifying P losses from a range of Nordic and Baltic micro-catchments (0.1–4.7 kg P ha^{-1}). The observed differences in TP losses from agricultural land was not related to the surplus of P within the catchments (Figure 5), although such a straightforward relationship has been advocated by some authors (e.g. OECD, 2001). Evidently, other factors must be important for the observed differences in these TP losses. Such factors undoubtedly include variation in source area, mobilization risk and connectivity between the field and the watercourse. Numerous research projects have also demonstrated that dissolved and particulate P forms are influenced by a range of hydrological processes linked to leaching, preferential flow, overland flow and groundwater (Haygarth & Jarvis, 1999; Haygarth & Sharpley, 2000; Kronvang *et al*., 2005b). The high TP losses from agricultural land found in Norwegian and Finnish micro-catchments, and in macro-catchments from the same countries and southern Europe, suggest that soil erosion and surface run-off losses of P forms may be a dominating pathway in these regions, due to large flows in spring caused by melting snow in Northern catchments, and high rainfall events in the Mediterranean region.

Our breakdown of the important TP loss pathways in the 17 macro-catchments using the MONERIS model corroborates these findings (Figure 9). Soil erosion and surface run-off was the dominant diffuse TP pathway in the 17 macro-catchments (53.4%) followed by groundwater (14.4%) and pipe drainage water (2.8%). However, these results are modelled and until now it has not been possible to validate such model findings as we still lack empirical information at the catchment level on the delivery of P (Heathwaite *et al*., 2005; Gburek *et al*., 2005). Combining field and model studies of P transfer from agricultural land in a nested approach covering field, micro- and macro-catchments in different climatic and geological regions of Europe will be a way of solving this problem (Haygarth *et al*., 2005). Small micro-catchments include the main processes governing P mobilization, source areas and hydrological pathways, but they are less governed than larger catchments by P transformation and retention processes in surface waters including P buffers like lakes, reservoirs, wetlands, etc. (Kronvang *et al*., 2005a). Clearly, the scale of study is important in understanding which factors need to be included in assessing the contribution of agriculture to catchment P loads and what control actions are required.

Conclusions

Eutrophication is still one of the main problems in European waters and at the same time the relative importance of diffuse P losses from agriculture is increasing due to improvements made to reduce point source emissions. Thus, efficient strategies to control the agricultural P losses within catchments are likely to become one of the key issues in the implementation of the River Basin Management Plans linked to the EU Water Framework Directive (WFD). Results from the catchment-scale monitoring programmes in Europe are providing essential results that may influence future research and, management approaches and strategies:

1. Very large temporal variations in P concentration and P losses, both within and between years, were found in the catchment data sets analysed. This may have great importance for the accuracy of the annual estimate of P load as shown for the Odense river where the annual TP and DRP loads were biased (underestimated) and the load estimated with relatively poor precision depending on the sampling frequency adopted.
2. Phosphorus losses from agricultural areas were shown to vary greatly from one Nordic/Baltic micro-catchment to another (0.1–4.7 kg P ha^{-1} $year^{-1}$). Similarly, large variations were found from one macro-catchment to another within Europe (0.1–6.0 kg P ha^{-1} $year^{-1}$), although the variation between estimates for supra-national river basins was less pronounced (0.09–2.0 kg P ha^{-1} $year^{-1}$).
3. No relationship could be developed between TP loss and P surplus on agricultural land in the 35 Nordic and Baltic micro-catchments investigated. However, a relationship was established for the average annual total P loss from the 17 macro-catchments for which the explanatory variables were annual run-off, population density and the area of surface water. The latter explanatory variable in the regression model show that retention of TP in surface waters including lakes, reservoirs and rivers is important for the resulting TP export from a catchment. The retention of TP in the 17 macro-catchments varied by a factor of six (0.07–0.42 kg P ha^{-1} of catchment area).
4. The bio-available component of the P losses varied greatly within the 35 micro-catchments studied. As the EU Water Framework Directive is impact and action oriented, methods enabling a more precise characterization and

assessment of the site-specific and potential ecological impacts of the P losses should be given priority. This will ensure more targeted approaches to reduce the impacts of agricultural P losses.

5. The diffuse pathways of TP were dominated by soil erosion and surface run-off (53%), groundwater (14%) and pipe drainage water (2%) in the 17 macro-catchments but with great intra-catchment variation.

The results show that more research and monitoring of P mobilization, transfer pathways, storage and export at all scales are needed preferably applying a nested catchment approach and covering different European regions.

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