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Measurement and Modelling of Phosphorus Transport from Arable Land

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Preface

The 'licentiatexamen' is an autonomous part of a doctoral programme, consisting of 80 credit points, which is equivalent to two years of full-time study beyond the degree needed for doctoral studies. This degree is awarded on the basis of both course work and a dissertation. This licentiate thesis is based on the following papers:

I. Persson, K. (2000). Jordbearbetningens påverkan på fosforförlusterna från en mjälalättlera i södra Dalarna. The impact of soil cultivation on phosphorus losses from a silty clay soil in southern Dalarna. *Ekohydrologi*. 52. SLU. In Swedish.

II. Ulén, B. and Persson, K. (1999). Field-scale phosphorus losses from a drained clay soil in Sweden. *Hydrological Processes*. 13, 2801-2812.

III. Persson, K., Andersson, L. and Ulén, B. (2001). Linking a hydrological river basin model and field-scale models for phosphorus transport. Manuscript.

Introduction

An elevated level of phosphorus (P) is often identified as the main cause of eutrophication in lakes (Schindler, 1977). Eutrophication is defined as a process where the natural aquatic environments are enriched with nutrients in such amounts that it results in changes in the community of organisms and a declined water quality, with an increased production of algae and macrophytes (Vollenweider and Kerekes, 1982). The degradation of the increased amount of biomass can lead to oxygen depletion of bottom waters and dead lake bottoms.
The relative contribution of P from agriculture increased when P removal was introduced in municipal wastewater treatment plants, and P transport from arable land is now the main source of P (Rekolainen *et al.*, 1997).

Phosphorus transport process

In the P transport process from soil to water, water flow is the driving force and may occur on the soil surface (surface run-off), above the plough-pan in the topsoil layer (inter flow), in the soil profile (preferential flow and matrix flow), and through drain tiles where such exist (Johnes and Hodgkinson, 1998). Surface run-off has been identified as one of the main contributors of P to surface water in certain areas in Sweden (Persson, 2000) and elsewhere (Sharpley and Rekolainen, 1997).
Although high concentrations in drainage water have been measured for 30 years (Ryden *et al.*, 1973), it was mainly the surface run-off of P from fields that was considered. This was due to the fact that subsurface soil is known to have a very high binding capacity for P (McBride, 1994). However, phosphorus can be transported through the subsoil down to drainage pipes. Such losses are episodic in their character (Ulén, 1995). In the Broadbalk experiment in England soluble reactive P (SRP) constituted the largest part of total phosphorus (TP) losses in the drainage pipes, ranging between 66% and 86%. Furthermore, a positive relationship between soil P level and concentration of P in the drainage water was found (Heckrath *et al.*, 1995).

The transport through soil can be divided into two major pathways: preferential flow and matrix flow. Preferential flow is very rapid and only involves a few per cent of the pore volume, thus P is transported through the soil with little opportunity of being adsorbed to soil particles (Stamm *et al.*, 1998). Preferential flow has been shown to be an important pathway in structured clay soils (Dils and Heathwaite, 1999; Djodjic *et al.*, 1999), a common soil type in Sweden. In matrix flow, water moves slowly through the larger part of the soil pores and dissolved P has ample time to equilibrate with soil minerals.

Measurement of phosphorus concentration and direct calculation of phosphorus transport

Phosphorus can be transported in several different forms. The most commonly measured are soluble reactive P (SRP), and particulate P (PP). SRP is commonly determined after filtration through filters with the pore diameter 0.45 μm and assumed to represent biologically available P (Haygarth *et al.*, 1997). PP is usually measured in the sediment caught on filters, with the same pore size.

If the SRP and PP are summed and subtracted from the TP, a small amount of other P is usually found. This could be P transported as Unreactive P (UP) also know as dissolved organic P (DOP), soluble organic P (SOP) or dissolved nonreactive P (DNRP) (Haygarth and Sharpley, 2000). Animal manure is an important source of UP, and up to 77% of TP leached from lysimeters treated with pig slurry has been shown to be in the form of UP (Chardon *et al.*, 1997). In this Dutch study the percentage of UP increased with soil depth, and below 50 cm depth it was the dominating P form even in soils receiving mineral fertiliser, indicating that UP had higher mobility in the soil than SRP (Chardon *et al.*, 1997).

Another important form is P bound to colloids, which are small, between 0.001 and 1.0 μm, and therefore can be transported very long distances with running water before settling. Colloidal bound P also remains in the photic zone of lakes and rivers for a long time and is accessible to algae (Mayer and Jarrel, 1995). Due to their small size the colloids pass through filters used to distinguish between dissolved and particulate P, and the amount of dissolved P can therefore be overestimated. A large part of TP in water can be colloidal bound. In soil surface run-off water, 15% of the P was in the range between 1000 molecular weight (MW) to 0.45 μm, and 14% below 1000 MW, in contrast to soil leachate where 87% was <1000 MW (Haygarth *et al.*, 1997). SRP measured as filtrate <0.45 μm may in reality be adsorbed to small particles, giving it totally different physical characteristics and the possibility to bypass the strongly P adsorbing subsoil and reach drainage pipes. Up to 48% of TP in the Tualatin river was colloidal bound. Furthermore, the concentrations of P and Fe in colloidal form were correlated, indicating that colloids were formed when iron rich water was oxidised and Fe II changed to Fe III (Mayer and Jarrel, 1995).

Calculation of P transport is often based on continuous measurement of water flow on an hourly or daily basis and discrete measurements of P concentrations. The concentration values are interpolated between the measuring times and the resulting concentrations are multiplied with the water flow. In the Swedish programme for monitoring experimental fields, manual samples are taken every two weeks. However, it has been shown that due to rapid changes in P concentration in the drainage water, this time-based sampling greatly underestimates the transport of P (Kronvang *et al.*, 1997; Ulén and Persson, 1999) (Table 1). A more reliable method is flow-proportional sampling where samples are taken when a certain amount of water has passed the gauging station. In a Danish investigation, time-proportional samples were taken every hour, but the TP transport was still underestimated by 18% compared to when flow-proportional sampling strategies was used (Smith and Pedersen, 1996).

Table 1. Drainage total P losses from flow-proportional and manual sampling at an experimental field situated in the Lake Bornsjön drainage area, Sweden.

Year	Discharge (mm)	Flow-proportional P samples Tot (kg/ha)	Manual samples Tot P (kg/ha)
1992/93	142	0,29	0,07
1993/94	267	0,41	0,09
1994/95	326	0,46	0,25
1995/96	38	0,05	0,01
1996/97	133	0,23	0,05
1997/98	170	0,28	0,24
Sum		1,72	0,71
Manual / flow proportional			0,41

Measurement of phosphorus transport via surface run-off (Paper I) and through tile drains (Paper II)

The surface run-off of P is highly weather dependent and varies from year to year. The loss of PP is strongly correlated to soil erosion (Kronvang *et al.*, 1997). Long time series are needed to obtain the good statistical material needed to be able to determine the different causes of P run-off. However, long time series are scarce in the literature. In paper I, experimental plots with different management practises were used, and the effect on soil erosion and P surface run-off was evaluated. The plots were run for 6 years and loss of TP varied between 0 - 1.25 $kg \cdot ha^{-1} \cdot year^{-1}$. The loss of PP dominated over SRP in all but one treatment (direct sown without soil cultivation) where SRP constituted 50% of the loss. This was probably due to release of SRP from frozen plant material left on the field. In some years, a large amount of dead vegetation was present. The best way to inhibit loss of PP was to keep the soil covered by vegetation during the winter. In this experiment, the loss of surface run-off P was shown to be best managed by keeping the soil covered by vegetation during winter, and by increasing the content of organic matter in the soil. Other investigations have shown that avoiding application of P on wet soils before expected rain (Catt *et al.*, 1998) is important.

The subsurface transport of P may vary nearly as much from year to year as the lateral transport. In Paper II, loss of subsurface TP via tile-drains in central Sweden was shown to vary between 0.05 - 0.46 $kg \cdot ha^{-1} \cdot year^{-1}$. The difference was due to difference in weather conditions such as rain, soil frost, snow melt and antecedent soil moisture. The P loss was very episodic and these episodes occurred during autumn and winter as well as during spring. Less than 1% of the soil surface area was hydraulically active as indicated by staining of soil monoliths. Few, but continuous, macropores were found. During the most extreme single event 18-19 April 1999, 0.10 kg TP was transported per hectare. Other periods were characterised by low transport and during 452 days 1995-1997, the same amount of P was lost from the soil to the water.

The use of models for calculating phosphorus transport

The use of models must always be governed by the purpose (Gustafsson *et al.*, 1982). Different model approaches are needed for different problems and conditions. The models can be divided into several groups depending on the construction of the model. The simplest is the regression model, where data from measurements are statistically analysed and used to make predictions (Gustafsson *et al.*, 1982). This type of model is static, i.e. it can not change with different conditions. P models of this kind exist and are used to map losses from different areas (Ulén *et al.*, 2001).

Another simple P model is the export coefficient model, where different coefficients are used for different P sources (Johnes, 1996). The sum of all individually calculated loads gives the transport from the area. With changing conditions the values of the coefficients can be modified and the model can be used for scenario calculations (Johnes and Hodgkinson, 1998). One drawback of coefficient models is that they deliver mean values, usually yearly and lack dynamics.

In contrast to these simple models the dynamic model may be time continuous, where time is simulated in steps. The result from the preceding time step is used as the starting condition for the next time step. The time period used for P simulation is often one day but other time steps are also in use. The CENTURY model used for long (hundreds of years) time periods, for example, uses a time step of one month (Metherell *et al.*,). The dynamic continuous models are further divided into physical and conceptual models. The physical model is based on physical and chemical knowledge of P turnover and movement in soil and plants and the water flow interaction. The only physical P model is the ANIMO model (Groenendijk and Kroes, ; Schoumans and Groenendijk, 2000), developed in the Netherlands. Physical models demand very large amounts of data regarding soil conditions and other parameters, making the model impractical for use in poorly investigated or heterogeneous areas. The most commonly used model type is therefore the conceptual model, which is a combination of physical connections and empirical knowledge. The conceptual model must be calibrated before use. Examples of conceptual P models are GLEAMS (Leonard *et al.*, 1987; Knisel, 1993), CREAMS (Knisel, 1980) and ICECREAM (Rekolainen and Posch, 1993; Rekolainen *et al.*, 1998). In order to be able to properly simulate very episodic P-transport, a development to more physical models may be necessary.

Linking of a hydrological river basin model and two field-scale models for phosphorus transport (Paper III).

With the increasing focus on the catchment scale perspective on environmental monitoring (Berglöf, 2000), it is of vital importance to be able to calculate transport of nutrients on a catchment scale. For nitrogen (N) transport calculation, the HBV-N model (Arheimer and Brandt, 2000) has been developed and tested in Sweden. The hydrological compartment is based on the HBV model (Bergström, 1995) and the nitrogen compartment on the soil nutrient model SOIL-N (Johnsson *et al.*, 1987). A similar catchment scale tool for P calculation (HBV-P) might be based on the HBV model and a phosphorus compartment. One of the demands on such a model should be that it is able to run scenarios with different climatological and soil management conditions. This excludes the use of regression models for P concentration from different land uses, such as the one that has been linked to HBV in Finland (Bilaletdin *et al.*, 1994) or the one developed in Sweden for the prediction of long-term P transport (Ulén *et al.*, 2001). The conceptual ICECREAM (Rekolainen and Posch, 1993; Rekolainen *et al.*, 1998) model was chosen as the soil P component of HBV-P (Figure 1). However, this model does not incorporate calculation of drainage particulate P (PP) transport, which can comprise a large part of the TP losses (Ulén and Persson, 1999). Therefore, the PARTLE model (Shirmohammadi *et al.*, 1998; Ulén *et al.*, 1998) was incorporated in the model for calculation of drainage PP losses.

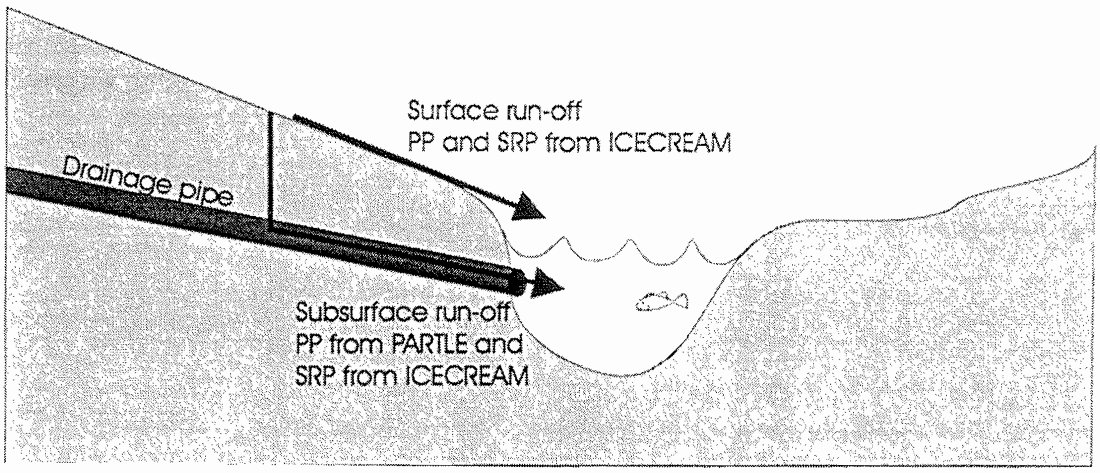

Figure 1. The P pathways from soil to water and models used for simulation of P transport along these pathways.

The results were promising, in most cases HBV-P performed better than ICECREAM on the three tested sites (Table 2), located on different soils and in different climate regions of Sweden. The performance and data demands of PARTLE were, however, less satisfactory. Pearson correlation for PP losses simulated with PARTLE and measured values were lower than the correlation between simulated dissolved P (SRP) losses and measured values.

Table 2. Performance of the HBV-P and ICECREAM models compared to measurements of SRP and PP losses.

	SRP				PP			
	Difference %		Pearson		Difference %		Pearson	
	HBV-P	ICE.	HBV-P	ICE.	HBV-P	ICE.	HBV-P	ICE.
Lanna, calibration (1993-1998)	-33	-30	0.34	0.29	-10	-27	0.25	0.35
Näsbygård, calibration (1990-1997)	+10	+1	0.71	0.59	-1	+10	0.48	0.44
Näsbygård, validation (1974-1989)	-41	-52	0.66	0.43	+31*	-11*	0.21*	0.11*

* = 1987-89.

HBV-P estimates of sediment transport had a very high correlation with the amounts monitored, whereas ICECREAM significantly underestimated the sediment transport (Table 3). However, it should be noted that HBV-P was calibrated, whereas ICECREAM was validated with standard values. A significant correlation was also found between measured sediment transport and particulate P transport, and also between particulate P transport and sediment transport with HBV-P. ICECREAM underestimated the loss of PP with surface run-off, apparently due to the low erosion. Dissolved P in surface run-off was only estimated with the ICECREAM model, obtaining a correlation of 0.87

Table 3. Correlation's between monitored and modelled sediment and particulate P transport at Hedemora research plot. * = correlation significant at the 0.10 level, ** = correlation significant at the 0.05 level.

	Sediment transport Measured	Sediment transport ICECREAM		Sediment transport HBV-P		PP transport ICECREAM	
		Pearson	Diff %	Pearson	Diff %	Pearson	Diff %
Sediment transport measured	-	0.76	-95	0.96**	-8	0.67	-
PP transport measured	0.90*	0.70	-	0.84*	-	0.74	-60

To be able to test the model on larger areas such as river sub-basins, access to a good conversion equation is necessary to convert the phosphorus measurements routinely carried out in Sweden to the resin-extractable soil P that is used by the P sub-model. The initial values of the soil P content are vital for accurate predictions of P transport.

A problem is the lack of fast preferential flow that has to be simulated. If the model could handle preferential flow it would probably not miss the large P loss episodes occurring during snowmelt and prolonged rain. In order to do this a more physical model approach may be needed.

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References

Arheimer, B. and M. Brandt (2000). "Watershed modelling of non-point nitrogen pollution from arable land to the Swedish coast in 1985 and 1994" *Ecological Engineering* 14: 389-404.

Berglöf, L. R. (2000)."Sötvatten, årsskrift från miljöövervakningen 2000". SLU.

Bergström, S. (1995). "The HBV model", in V. P. Singh (Eds) *Computer Models of Watershed Hydrology*. Water Resources Publications, Highlands Ranch, Colorado, 443-476.

Bilaletdin, Ä., K. Kallio, T. Frisk, B. Vehviläinen, M. Huttunen and J. Roos (1994). "A modification of the HBV model for assessing phosphorus transport from a drainage area" *Wat. Sci. Tech.* 30(7): 179-182.

Catt, J. A., K. R. Howse, R. Farina, D. Brockie, A. Todd, B. J. Chalmers, R. Hodgkinson, G. L. Harris and J. N. Quinton (1998). "Phosphorus losses from arable land in England" *Soil use and management* 14: 168-174.

Chardon, W. J., O. Oenema, P. del Castilho, R. Vriesema, J. Japenga and D. Blaauw (1997). "Organic phosphorus in solutions and leachates from soils treated with animal slurries" *Journal of environmental quality* 26: 372-378.

Dils, R. M. and A. L. Heathwaite (1999). "Phosphorus transport from diffuse agricultural sources: shallow subsurface pathways in grassland soils". IUGG 99 Symposium HS3, Birmingham, July 1999, IAHS, 57-64.

Djodjic, F., L. Bergström, B. Ulén and A. Shirmohammadi (1999). "Mode of transport of surface-applied phosphorus-33 through a clay and sandy soil" *Journal of environmental quality* 28(4): 1273-1282.

Groenendijk, P. and J. G. Kroes Modelling nitrogen and phosphorus leaching to groundwater and surface water. Description of ANIMO 3.5". Green world research, Netherlands.

Gustafsson, L., H. Lanshammar and B. Sandblad (1982). "System och modell". Lund, Studentlitteratur.

Haygarth, P. M. and A. N. Sharpley (2000). "Terminology for phosphorus transfer" *Journal of environmental quality* 29: 10-15.

Haygarth, P. M., M. S. Warwick and W. A. House (1997). "Size distribution of colloidal molybdate reactive phosphorus in river waters and soil solution" *Wat. Res.* 31(3): 439-448.

Heckrath, G., P. C. Brookes, P. R. Poulton and K. W. T. Goulding (1995). "Phosphorus Leaching from Soils Containing Different Phosphorus Concentrations in the Broadbalk Experiment" *Journal of environmental quality* 24: 904-910.

Johnes, P. J. (1996). "Evaluation and management of the impact of land use change on nitrogen and phosphorus load delivered to surface waters: the export coefficient modelling approach" *Journal of hydrology* 183: 323-349.

Johnes, P. J. and R. A. Hodgkinson (1998). "Phosphorus loss from agricultural catchments: pathways and implication for management" *Soil use and management* 14: 175-185.

Johnsson, H., L. Bergström and P.-E. Janson (1987). "Simulated nitrogen dynamics and losses in a layered agricultural soil" *Agric. Ecosyste. Environ* 18:333-356.

Knisel, W. G. (1980)."CREAMS: a field-scale model for chemicals, runoff and erosion from agricultural managment systems". USDA, Conservation Research Report 26.

Knisel, W. G. (1993)."GLEAMS: Groundwater loading effect of agricultural management systems, version 2.1". UGA-CPES-BAED University of Georgia, Publ. No. 5.

Kronvang, B., A. Laubel and R. Gran (1997). "Suspended sediment and particulate phosphorus transport and delivery pathways in an arable catchment, Gelbæck stream, Denmark" *Hydrological processes* 11:627-642.

Leonard, R. A., W. G. Knisel and D. A. Still (1987). "GLEAMS: Groundwater loading effects of agricultural management systems" *Transactions of the ASAE* 30(5): 1403-1418.

Mayer, T. D. and W. M. Jarrel (1995). "Assessing colloidal forms of phosphorus and iron in the Tualatin river basin" *Journal of environmental quality* 24: 1117-1124.

McBride, M. (1994). "Environmental chemistry of soils". New York, Oxford, Oxford university press.

Metherell, A. K., L. A. Harding , C. V. Cole and W. J. Parton Century Documentation. CENTURY Soil Organic Matter Model Environment Technical Documentation Agroecosystem Version 4.0". USDA-ARS, Great Plains System Research Unit, Technical Report No. 4.

Persson, K. (2000)."Jordbearbetningens påverkan på fosforförlusterna från en mjälalättlera i södra Dalarna". Swedish University of Agricultural Sciences, Division of Water Quality Management, Ekohydrologi 52.

Rekolainen, S., P. Ekholm, B. Ulén and A. Gustafson (1997). "Phosphorus losses from agriculture to surface waters in the Nordic countries", in H. Tunney, O. T. Carton, P. C. Brookes and A. E. Johnston (Eds) *Phosphorus loss from soil to water*. CAB International, Wallingford, UK, 77-94.

Rekolainen, S., I. Grönroos, A. Bärlund, A. Nikander and Y. Laine (1998). "Modelling the impact of management practices on agricultural phosphorus losses to surface waters of Finland". 3rd International Conference on Diffuse Pollution, 33-40.

Rekolainen, S. and M. Posch (1993). "Adapting the CREAMS Model for Finnish Conditions" *Nordic Hydrology* 24: 309-322.

Ryden, J. C., J. K. Syers and D. Haris (1973). "Phosphorus in runoff and streams." *Advances in agronomy* 25: 1-45.

Schindler, D. (1977). "Evolution of phosphorus limitation of lakes" *Science* 196: 260-262.

Schoumans, O. F. and P. Groenendijk (2000). "Modelling soil phosphorus levels and phosphorus leaching from agricultural land in the Netherlands" *Journal of environmental Quality* 29: 111-116.

Sharpley, A. N. and S. Rekolainen (1997). "Phosphorus in agriculture and its environmental implications", in H. Tunney, O. T. Carton, P. C. Brookes and A. E. Johnston (Eds) *Phosphorus loss from soil to water*. CAB International, Wallingford, UK, .

Shirmohammadi, A., B. Ulen, L. F. Bergstöm and W. G. Knisel (1998). "Simulation of Nitrogen and Phosphorus Leaching in a Structured Soil Using Gleams and a New Submodel, "Partle"" *Transactions of the AEA E* 41(2): 353-360.

Smith, O. and S. Pedersen (1996). "Flow proportional sampling - a method to estimate phosphorus transport". Phosphorus and sediment. Erosion and delivery, transport and fate of sediments and sediment associated nutrients in watersheds., Silkeborg, Denmark, NERI Technical report 178.

Stamm, C., H. Flühler, R. Gächter, J. Leuenberger and H. Wunderli (1998). "Preferential transport of phosphorus in drained grassland soils" *Journal of environmental quality* 27: 515-522.

Ulén, B. (1995). "Episodic Precipitation and Discharge Events and their Influence on Losses of Phosphorus and Nitrogen from Tile Drained Arable Fields." *Swedish Journal of Agricultural Research* 25: 25-31.

Ulén, B., G. Johansson and K. Kyllmar (2001). "Prediction model and long term-trend of phosphorus transport from arable land in Sweden" *Agricultural Water Management* **In press.**

Ulén, B. and K. Persson (1999). "Field-scale phosphorus losses from a drained clay soil in Sweden" *Hydrological Processes* 13(17): 2801-2812.

Ulén, B., A. Shirmohammadi and F. Djodjic (1998). "The use of Gleams model to predict phosphorus losses from drained clay soils" *Vatten* 54: 53-60.

Vollenweider, R. and J. Kerekes (1982)."Eutrophication, monitoring, assessment and control. Final report". OECD cooperative programme monitoring of inland waters (eutrophication control). Environmental directorate .

JORDBEARBETNINGENS PÅVERKAN PÅ FOSFORFÖRLUSTERNA FRÅN EN MJÄLALÄTTLERA I SÖDRA DALARNA.

The impact of soil cultivation on phosphorous losses from a silty clay soil in southern Dalarna.

Kristian Persson

Abstract: Phosphorous losses with surface runoff are affected by how the soil is cultivated. In a plot study, located in central Sweden, eight different management practices were tested on a silty clay soil with a 10% slope. The different treatments were; conventional autumn plowing, spring plowing with and without catch crop, no tillage, deep cultivation (40 cm), direct drilling, ley alternating with autumn wheat, and addition of organic material.

The two spring-plowed treatments had the lowest losses of total phosphorous the catch crop did not develop sufficiently to affect the losses. The largest losses were from the autumn-plowed and the deep-cultivated plots, beeing almost twice those of the spring-plowed plots. The loss of phosphate-phosphorous were largest from the direct-drilled plot, where it contributed with half of the loss, compared to 10-20% in the other treatments. The treatments that gave the best ratio between phosphorous loss and harvest were the two spring-plowed plots and the one with addition of organic matter.

Målsättning

Målet med försöket är att studera hur fosforförluster med ytavrinningen beror av olika jordbearbetningsmetoder på en erosionskänslig jord. Även skördeutfallet och fosforupptag i grödan undersöktes. För kväveomsättning och kväveläckage redogörs i en separat report (Lindén, Rydberg och Stenberg, 1997).

Material och metoder

Försöksfält, odlingsåtgärder

Försöket startade 1993. Försöksfältet är beläget i södra Dalarna, strax utanför Hedemora (Figur 1). Fältet har ca 10% lutning ned mot ett biflöde (Västerbydike) till Mässingsboån. Fältet är delat i åtta led med olika behandling (Tabell 1). Växtföljden har varit korn, korn, havre, korn, havre, korn. I Tabell 2 redovisas gödslingen. Det direktsådda ledet har vissa år (1994 och 1997) haft stora problem med ogräs. Dessa år behandlades ledet med 3 liter glyfosfat per ha på hösten. Efterföljande vintrar var marken täckt med död ogräsvegetation. Som fånggröda i led F utnyttjades engelskt rajgräs de första fyra åren men eftersom det etablerades dåligt utnyttjades klöver 1997 och 1998. I Led G har grödan vartannat år varit höstvete vartannat vall. Det organiska material som tillfördes led H motsvarade 6 ton / ha och utgjordes av gräsklipp från en omställningsmark. Materialet harvades ned i markytan på hösten.

Tabell 1. Jordbearbetning, bearbetningsdjup (cm) och tidpunkt.

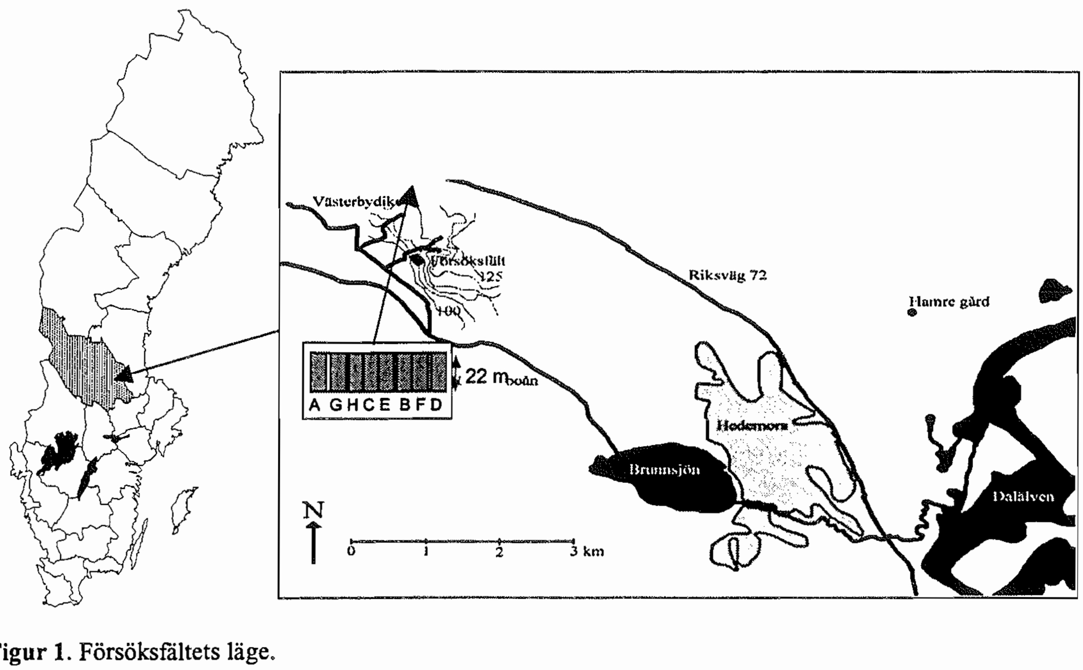
Led	behandling	Djup		Djup		Djup
A	Konventionell plöjning, höst	21	Harvning 3 ggr, vår	5		
B	Konventionell plöjning, vår	21	Harvning 3 ggr, vår	5		
C	Plöjningsfri odling, tallriksharvning, höst	10	Kultivering 2 ggr, höst	15	Harvning, 4 ggr, vår	5
D	Direktsådd, vår					
E	Djupkultivering varje år, 3 ggr höst	15	Djupkultivering, höst	40		
F	Insådd fånggröda, vårplöjning	21	Harvning 3 ggr, vår	5		
G	Vintergrön, Vall / höstsäd, höstplöjning	21	Jordfräsning, höst	5	Harvning, 3 ggr, höst	5
H	Tillförsel av org. mat., harv. 3 ggr, höst	5	Kultivering 2 ggr, höst	5		

Tabell 2. Gödsling.

		1993	1994	1995	1996	1997	1998
led A, B, C, E, F, H							
Giva	N kg/ha	90	90	90	90	90	90
	P kg/ha	14	14	18	18	18	17
	K kg/ha	23	36	36	36	36	30
led D							
Giva	N kg/ha	90	118	90	90	90	90
	P kg/ha	14	21	18	18	18	17
	K kg/ha	23	40	36	36	36	30
led G							
Giva	N kg/ha	100	108+48	101	75	101	100+38
	P kg/ha	22	36	-	25	-	33
	K kg/ha	42	72+36	-	50	-	67+29

Jorden

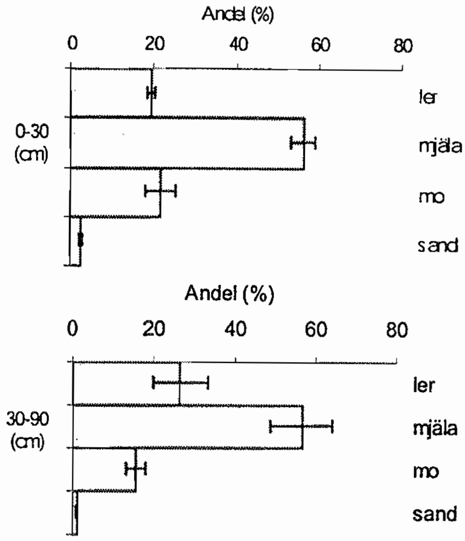
Jorden är en mjälalättlera (Figur 2) med låg kolhalt (Tabell 3) och följaktligen låg organisk halt. Mjälajorden är känd för att vara mycket erosionskänslig. I området har jordens organiska halt minskat kraftigt på senare år på grund av minskad vallodling (Olsson, 1996). Vid försökets början hade jorden en likartad aggregatstabilitet i alla led (Tjell, 1994). Fosforhalten



Figur 1. Försöksfältets läge.

Tabell 3. Ytlagrets (0-5 cm) pH-värde mätt i ren vattenlösning , fosfor-, kaliumvärde, totalkol- och totalkvävehalt

		Medelhalt	Standardavvikelse
pH	(H_2O)	5,13	0,10
P-AL	(mg / 100 g ts)	3,35	0,28
Tot C	(% av ts)	1,43	0,13
Tot N	(% av ts)	0,15	0,02



Figur 2. Textur i matjord (0-30 cm) och alv (30-90 cm).

var 3.35 mg / 100 g jord motsvarade klass II i matjordskiktet (0-30 cm). Halten var något lägre i Mässingsboåns avrinningsområde som helhet där den var 2,9 mg / 100 g jord. pH var ganska lågt, drygt 5, vilket är lägre än vad som uppmäts i Mässingsboåns avrinningsområde som helhet där den var 6,1 (Olsson, 1996).

Ovanjordiskt växtmaterial och fosfor bortfört med kärnskörd

Ovanjordiskt växtmaterial mättes sent i oktober och tidigt i maj under perioden 1993-1996. Varje prov bestod av 5 delprov från vardera 2 kvadratiska ytor på 0,25 m^2. Materialet delades i grönt och övrigt. Prover togs på samma sätt vid skörd. Vattenhalt och renhet analyserades och fosforinnehållet i kärnskörden 1994-1997 bestämdes med ICP-teknik efter uppslutning i koncentrerad svavelsyra.

Klimat

Nederbörd mättes med SMHI's standardmätare vid Hamre gård, 5 km från försöksplatsen. Även luft- och marktemperatur mättes här.

Ytavrinning och fosfor i ytvattnet

Rutorna var 22 m långa och avgränsade uppåt med en plogfåra och plastkant. Uppsamling för ytvattenprov startade hösten 1994 och skedde med hjälpt av 0,5 m breda Gerlachtråg (Gerlach, 1967). 2-4 st parallella tråg grävdes ner i nedre kanten på varje ruta. Vattenproven samlades upp varje vecka då ytavrinning inträffad och mängden vatten mättes. Parallella vattenprov från samma led slogs ihop till ett samlingsprov. Analyser utfördes av totalfosfor (tot-P), partikulär fosfor (part-P) mätt som skillnaden mellan tot-P på filtrerat och ofiltrerat prov, fosfatfosfor (PO_4-P) mätt på filtrerat prov och suspenderat material (susp). Analyserna är utförda enligt svensk standard (1990) med undantag av att filtret hade porstorlek 0,2 µm istället för 0,45 µm, för att bättre kunna skilja av det finpartikulära materialet. Analyserna utfördes på ackrediterat laboratorium vid avdelning för Vattenvårdslära, SLU.
Jorden är mycket lättrörlig och både erosion och frysning ledde till att uppsamlingskärlen rörde sig. Ytvattenavrinningen har därför mätts genom att ytvattnet samlats upp i ett gummiklätt dike, varefter mängderna registrerades med hjälp av vippkärl. Detta gjordes på 2

Tabell 4. Jämförelse av de olika ledens skördeutfall då höstplöjt led A har satts till 0. Signifikanta skillnader enligt t-test.

Led	A	B	C	D	E	F	H
A Höstplöjt	-						
B Vårplöjt	-0,371	-					
C Plöjningsfri	-2,219	-1,155	-				
D Direktsådd	3,706**	2,334	4,019**	-			
E Djupkult.	-1,133	-0,202	1,597	-3,062*	-		
F Fånggröda	0,473	1,744	1,901	-1,957	1,092	-	
H Org. matrl.	-6,613***	-3,450***	-5,414***	-5,263***	-7,154***	-3,739**	-

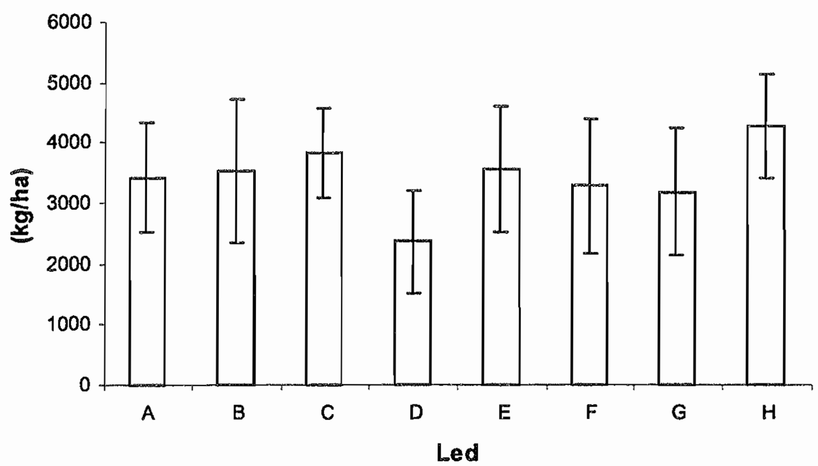
*p<0,1 **p<0,05 ***p<0,01

av rutorna; E som är höstbearbetad och B som är vårbearbetad. Dessa har antagits representera leden A, C och H respektive D, F och G med avseende på ytavrinnande vatten. Ämnestransporterna har beräknats genom att avrinningen multiplicerats med halterna mätta i Gerlachtrågen. Årstransporter redovisas i agrohydrologiska år: första juli till sista juni.

Resultat

Skörd

Direktsådd (D) gav lägst skörd och var signifikant lägre än led som höstplöjts, eller som varit plöjningsfria, djupkultiverats eller tillförts organiskt material (Figur 3). Skörden i det direktsådda ledet blev bara hög ett år (1995) då grödan såddes tidigare än i de andra leden. Övriga år har ogräs och sen uppkomst minskat skörden. Höstsådd (A) har i medeltal gett ungefär samma skörd som vårsådd med (F) eller utan fånggröda (B). Led G med höstvete gav dålig skörd 1993. Samtidigt med vete på hösten såddes vallen in och konkurrensen med gräset blev för stor. Vid nästa vetegröda, 1995, såddes därför inte vallen förrän på våren. Detta ledde dock till en mycket dålig vallutveckling och nysådd av vall fick göras våren 1996. 1997 gav en dålig höstveteskörd trots en god utveckling på våren. Vid beräkning av skillnader mellan de olika leden har skörden från höstplöjt led satts till 0 och skillnaden i den procentuella avvikelsen från övriga led har beräknats för samtliga sex år (Tabell 4) Led H med tillförsel av organiskt material hade signifikant högre skördar än alla andra behandlingar. Andra skillnader var inte signifikanta (P<0,05).



Figur 3. Medelvärde (staplar) med standardavvikelse (linjer) av skördar (kg/ha) 1993-1998. Led G är baserat på de tre år då höstvete odlades.

Tabell 5. Ovanjordiskt dött och levande växtmaterial under vinten (kg/ha). Medelvärden från mätningar sent i oktober och början av maj 1993-1996.

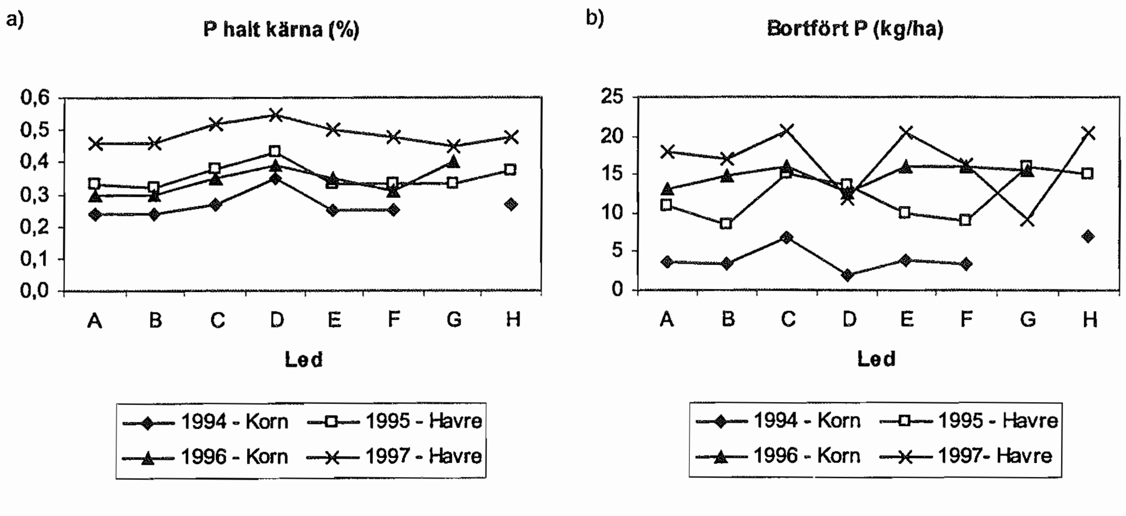
	Dött	Levande
A Höstplöjning	0	0
B Vårplöjning	1600	50
C Plöjningsfri	360	7
D Direktsådd	2000	120
E Djupkultivering varje år	400	7
F Vårplöjning & fånggröda	1700	260
G Höstvete	30	130
G Vall	3500	110
H Plöjningsfri & organiskt mtrl.	770	0

Fosfor i kärnskörd.

Led D (direktsådd) hade de högsta halterna av fosfor i kärnan. Eftersom skörden var liten var det totala bortförseln av fosfor förhållandevis låg (Figur 4). Högst bortförsel hade led C (plöjningsfri) med i medeltal 14,6 kg/ha. Även led G (vall/höstvete) hade stor bortförseln. 1994 var mängden bortförd fosfor från alla led var mycket låg (i medeltal 4,3 kg/ha) då skörden var dålig. 1995 var medelbortförseln 12,2 kg/ha, 1996 14,9 kg/ha och 1997 16,7 kg/ha. Givan var vanligen 14 eller 18 kg P/ha (Tabell 2). Fosforn i kärnskörden motsvarade i genomsnitt 31%, 68%, 83% respektive 93% av den tillförda fosforn åren 94-97.

Växtmaterial på markytan under vintern

Växtmaterial på markytan bör leda till mindre erosion än om marken är bar. Konventionell höstplöjning innebär att det varken fanns levande eller dött material på markytan (Tabell 5), medan vårbearbetade led hade mycket växtmaterial varav det mesta var dött. Fånggrödan hade ofta haft en dålig etablering och dålig tillväxt på hösten och gav därför inte mycket erosionsskydd. Växtsäsongen är för kort för att fånggrödan ska hinna utvecklas.



Figur 4. a) Halten fosfor i kärnskörden. **b)** Mängd fosfor som bortfördes med skörd.

Ytavrinning

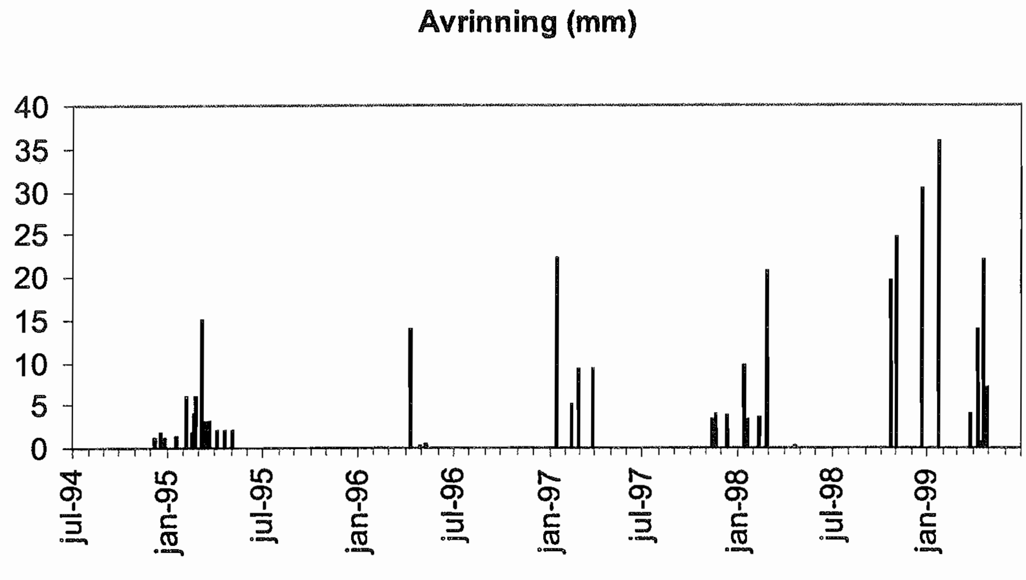
Ytavrinningen (Figur 5) varierade mycket olika år och var episodisk. Störst avrinning var det vintern 1998/99 då det rann 157 respektive 185 mm i led B och E. Medelavrinningen för alla år var 63 respektive 62 mm.

Fosforkoncentrationer och fosforförluster med ytvatten

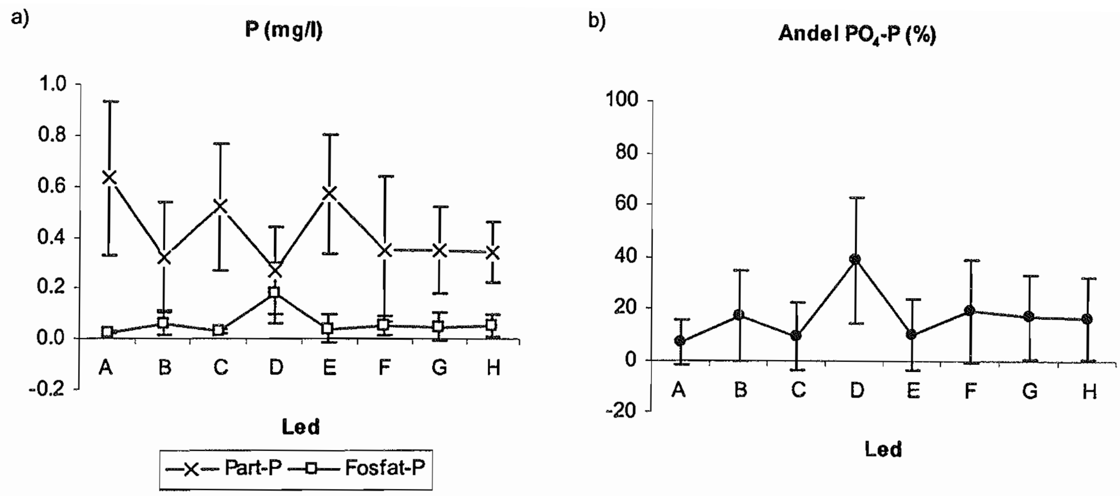
År med liten ytvattenavrinning det svårt att se skillnader i halter eller avrunna mängder fosfor. Led A hade de högsta halterna av partikulärt bunden fosfor i ytavrinningsvattnet de tre första åren medan led E hade de högsta halterna de två sista åren. Led D, G och B hade i medeltal de lägsta halterna (Figur 6a). Förlusten av partikulär fosfor ökar mer med ökad halt suspenderat material i ytavrinningsvattnet i de vårbearbetade leden än i de höstbearbetade (Figur 7).

Fosfatfosfor utgjorde i medeltal mellan 7 och 39% av den totala fosforn (Figur 6b). Den uppvisade också ett annat läckage mönster mellan de olika leden än den partikulära fosforn. Led D avvek från de andra leden genom en betydligt större andel fosfatfosfor (45%) än de övriga leden där den låg mellan 4 och 18%. Vid ett tillfälle under hösten 1996 var 88% av tot-P i led D i form av fosfat-P. Lägst förluster av fosfatfosfor hade led A och C. Dessa led hade å andra sidan höga förluster av partikulärt bunden fosfor.

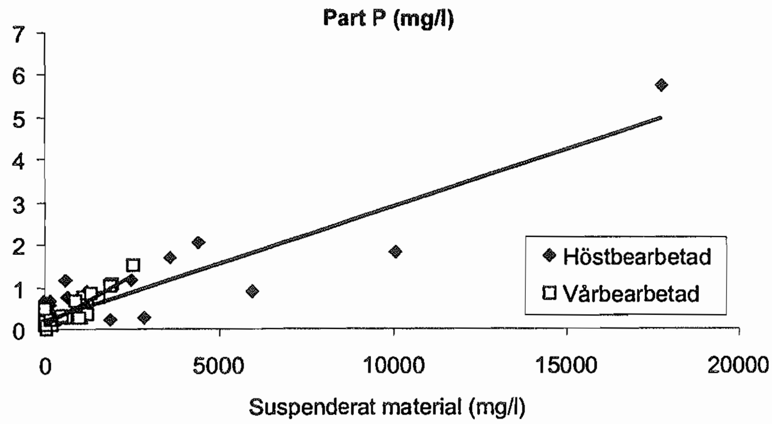
Förlusten av fosfor varierar kraftigt mellan olika år (Tabell 6). År 95/96 var förlusterna små medan de var stora 98/99 och i viss mån också 94/95. Högst ytvatten förlust skedde från led A, E, C och D. I medeltal (Figur 8) var förlusterna av partikulär fosfor mellan 0,2 och 0,4 kg P/ha och för fosfatfosfor mellan 0,2 och 0,5. Led D, direktsådd, hade en förlust på 0,14 kg P/ha .



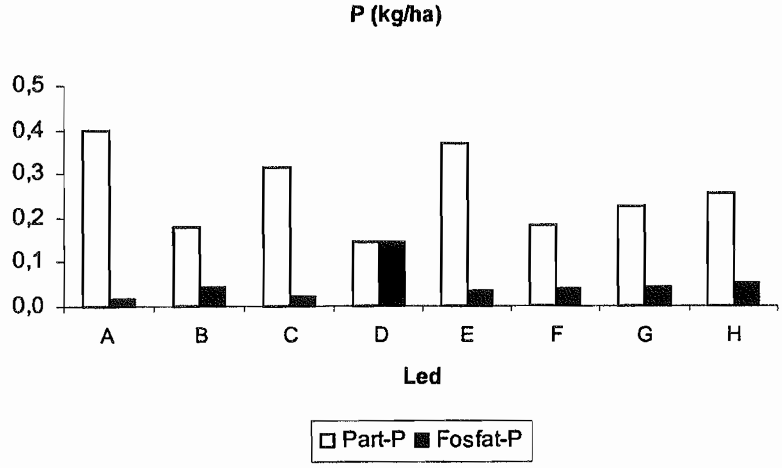
Figur 5. Ytavrinning per vecka i led B (mm).



Figur 6. a) Medelvärden 1993-1998 av halten av de olika fosforfraktionerna i ytavrinningsvattnet. **b)** Medelvärden 1993-1998 av andelen fosfat-P i % av total-P. Linjerna är standardavvikelse.



Figur 7. Samband partikulärt bunden fosfor och halten suspenderat material i ytavrinningsvattnet i höstbearbetade (R^2 =0,83) och vårbearbetade (R^2 =0,84) led.



Figur 8. Medelvärden 1994-98 av part-P och fosfat-P förlust med ytavrinning.

Tabell 6. Förluster av olika fosforfraktioner med ytavrinning (kg/ha).

	Tot P					
Led	94/95	95/96	96/97	97/98	98/99	medel
A	1,18	0,04	0,25	0,14	0,68	0,46
B	0,36	0,03	0,06	0,17	0,54	0,23
C	0,46	0,00	0,11	0,15	1,10	0,36
D	0,42	0,08	0,32	0,11	0,65	0,32
E	0,53	0,03	0,13	0,17	1,25	0,42
F	0,43	0,03	0,10	0,08	0,60	0,25
G	0,63	0,05	0,07	0,13	0,55	0,29
H	0,46	0,04	0,06	0,16	0,93	0,33
	Part P					
Led	94/95	95/96	96/97	97/98	98/99	medel
A	1,04	0,03	0,24	0,10	0,57	0,40
B	0,34	0,02	0,05	0,09	0,39	0,18
C	0,43	0,00	0,05	0,12	0,98	0.32
D	0,24	0,07	0,08	0,07	0,26	0,14
E	0,50	0,02	0,06	0,13	1,14	0,37
F	0,40	0,02	0,06	0,03	0,39	0,18
G	0,60	0,03	0,06	0,03	0,40	0,22
H	0,42	0,02	0,04	0,10	0,68	0,25
	PO4 P					
Led	94/95	95/96	96/97	97/98	98/99	medel
A	0,01	0,00	0,00	0,01	0,06	0,02
B	0,02	0,01	0,01	0,06	0,11	0,04
C	0,02	0,00	0,03	0,01	0,04	0,02
D	0,11	0,02	0,23	0,02	0,34	0,14
E	0,02	0,00	0,07	0,01	0,07	0,03
F	0,02	0,01	0,02	0,03	0,12	0,04
G	0,02	0,02	0,01	0,05	0,10	0,04
H	0,03	0,02	0,01	0,02	0,18	0,05

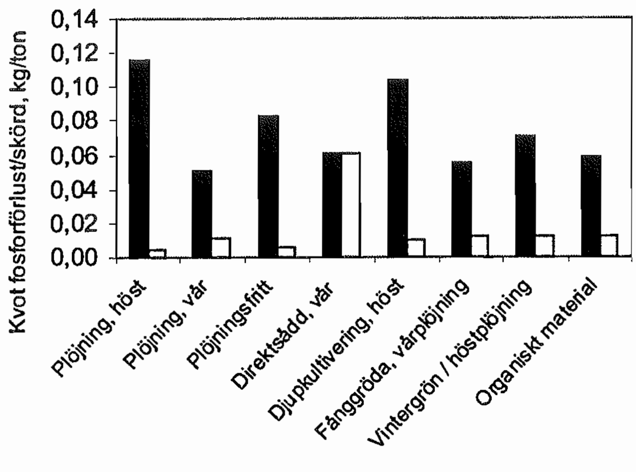
Genom att räkna ut hur mycket skörd man får per kg förlorad fosfor kan man jämföra de olika metodernas förluster (Figur 9). De vårplöjda leden A och F, direktsådd D och ledet med tillförsel av organiskt material H hade lägst kvot av partikulär fosfor förlust och skörd. Kvoten skörd och fosfatfosforförlust var däremot lägst för de höstplöjda A och djupkultiverade E leden, vilket kan bero på att fosforfattig jord har förts upp till ytan med minskad fosfatförlust som följd. Direktsått led hade den i särklass högsta kvoten fosfatfosfor förlust och skörd, vilket beror på hög förlust och dålig skörd.

Diskussion

De olika odlingsåtgärdena har kraftigt påverkat förlusten av fosfor under perioden som helhet. Höstbearbetade led hade större förlust av fosfor än vårbearbetade led. Erfarenheterna från andra försök är blandade. I en dansk undersökning var ytavrinningsförlusterna av fosfor från höstplöjda fält mindre än från höstsådda led. Vattnet samlades i gropar och infiltrerade på de höstplöjda fälten medan markytan var slät och avrinningen blev större i de höstsådda (Schjønning *et al.* 1995). Vårbearbetade led hade en stark koppling mellan suspenderat material och partikulär fosfor. Som förklaras av att aggregaten är mindre och binder relativt mer fosfor i de vårbearbetade leden medan höstbearbetningen för upp jord med låg fosfor halt till ytan. Norska undersökningar visade till skillnad från de danska att vårplöjda fält hade betydligt mindre förluster än höstplöjda (Ludvigsen, 1995). En finsk undersökning visar att ytavrinningen av vatten var högre på vall än på höstplöjda fält, men att förlusten av partikulär fosfor var högre från de plöjda fälten. Anledningen är att marken på de plöjda fälten är mycket känsligare för erosion än den vallbevuxna. Förlusten av löst fosfatfosfor var dock högre från vallen (Turtola, 1999). Frostskadad vegetation kan vara en källa till fosfatfosfor förluster (Timmons *et al.*,1970, Miller *et al.*, 1994). Fånggrödan var dock mycket dåligt utvecklad och påverkade inte förluster av fosfor i försöken i Dalarna. Den rikliga förekomsten av dött organiskt material i det direktsådda ledet kan varit en källa till fosfor och är troligen orsaken till den mycket höga förlust av fosfatfosfor som ledet uppvisar. I ett mildare klimat och utan användning av ogräsbekämpningsmedel skulle troligen förlusterna vara mindre. På vintergrön mark i Halland (Ulén, 1997) medförde fånggröda ett tillskott på mindre än 0,02 kg PO_4-P /ha i direktsått led men i det här försöket i Dalarna var tillskottet i genomsnitt 0,11 kg PO_4-P /ha.

Slutsatser

De olika jordbearbetningsmetoderna i de olika leden påverkade fosforns rörlighet och skördeutfallet. Bäst beträffande förlust av fosfor var vårplöjning av marken och sämst var konventionell höstplöjning. Båda gav ungefär samma skörd. Bäst ur skördesynpunkt var om jorden tillfördes organiskt material. Sämst ur skördesynpunkt var direktsådd gröda och från det ledet förlorades också mest fosfatfosfor med ytvattnet.



Figur 9. Förhållandet mellan ytvattenförlust av fosfor och skörden, medelvärden för åren 1993-1998.

Referenser

Gerlach, T. 1967. Hillslope throughs for measuring sediment movment. Rev. Geomorph. Dynamique 4:173.

Lindén, B., Rydberg, T. & Stenberg, M. 1998. Jordbearbetningssystem på en mjälalättlera i södra Dalarna: Inverkan på grödornas kväve och fosforutnyttjande. Stencil. Institutionen för markvetenskap, SLU.

Ludvigsen, G. H. (1995). Jordmonnovervåkning i Norge 1992-1996, Rapport från programmet 1994. Jordforsk rapport 81/95. Ås, Norge

Miller, M. H., Beauchamp, E. G. & Lauzon, J. D. 1994. Leaching nitrogen and phosphorus from the biomass of three cover crop species. J. Environ. Qual. 23:267-272.

Olsson, K. 1996. Markkaraktärisering av ett jordbruksområde i Dalarna. Examensarbete, Avdelningen för marklära, SLU.

Schjønning, P., Sibbesen, E., Hansen, A. C., Hasholt, B., Heidmann, M. B. & Nielsen, J. B. 1995. Surface runoff, erosion and loss of phosphorus at two agricultural soils in Denmark. SP report no 14.

Svensk standard 1990. Kemiska vattenundersökningar. Katalog över svensk standard. Standardiseringskomissionen i Sverige, 588 s.

Timmons, D. R., Holt, R. F. & Latterell, J. J. 1970. Leaching of crop residues as a source of nutrients in surface runoff. Water. Resour. Res. 6:1367-1375.

Tjell, D. 1994. Fosforförluster från åkermark via yterosion och inre erosion. Seminarier och examensarbeten nr 23. Avdelningen för vattevårdslära, SLU.

Turtola, E. 1999. Phosphorus in surface runoff and drainage water affected by cultivation practices. Agricultural Research Centre of Finland, Institute of Crop and Soil Science, Jokioinen, Finland.

Ulén, B. 1997. Nutrient losses by surface run-off from soils with winter cover crops and spring-ploughed soils in the south of Sweden. Soil & Tillage research 44:165-177

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Field-scale phosphorus losses from a drained clay soil in Sweden

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Abstract:

The objective of this study was to determine and discuss field-scale phosphorus losses via subsurface tile drains. A total phosphorous (Tot-P) export, which averaged 0·29 kg ha^{-1} $year^{-1}$, was measured over a six-year period from the 4·43 ha drainage system of a Eutric Cambisol in Central Sweden. The main part (63%) was in particulate form (PP) while the remainder was either in phosphate form (PO_4-P) or in other dissolved or colloidal forms. A very small area, less than 1% of the soil surface, was demonstrated to be hydraulically active by using a staining technique in soil monoliths taken from the field. The stained macropores were few, but were continuous downward, and were relatively evenly distributed among the eight 7 dm^2 areas that were investigated. The transport from the field mainly occurred in episodes during which the relationship between phosphorus concentration and discharge was characterized by hysteresis loops. On average, half of the yearly P transport occurred in 140 hours. Compared with flow-proportional and frequent sampling, manual and fortnightly sampling underestimated the transport of Tot-P and suspended solids (SS) by 59 and 42%, respectively, during the six years studied. Amounts of different phosphorus forms exported through the tile drains were very similar to those reported from other clay soils in Northern Europe and North America. Copyright © 1999 John Wiley & Sons, Ltd.

KEY WORDS phosphorus losses; drainage water; hysteresis loops; macropores

INTRODUCTION

Phosphorus (P) export via drain tiles may reach the receiving water course without any retention. Clay soils which are usually artificially drained constitute a large part of the arable land in Nordic countries and leaching from such soils accounts for much of the phosphorus (P) loss to surface water (Rekolainen *et al.*, 1997). Much of the P loss via subsurface pathways is predominately episodic (Ulén, 1995). High P concentrations in subsurface drainage were identified in the early 1970s by Ryden *et al.* (1973) but have not, until recently, attracted a great deal of attention (c.f. Heckrath *et al.*, 1995; Turtola and Paajanen, 1995). P may be transported in several forms; particulate, colloidal and dissolved P forms (Haygarth *et al.*, 1998).

Macropores are commonly regarded as preferential pathways responsible for P movement through the unsaturated zone in agricultural clays soils. Thus, unsaturated flow may occur along distinct paths within the macropores, especially when triggered by ponded conditions and saturation of the mesopores (c.f. Quinsberry and Phillips, 1976; Steenhius and Muck, 1988). In this way, particulate and soluble P from the soil surface may be transported through the soil column into the tile drains, resulting in subsurface P export to receiving streams (Øygarden *et al.*, 1997; Grant *et al.*, 1996). Characterization of the macroporosity (i.e. distribution, shape, length, frequency and number) may improve understanding of phosphorus transport through the soil (Sims *et al.*, 1998). Pores have been classified into four groups, based on

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formation processes and geometric shape, by Beven and Germann (1982). However, other P transport paths may also occur. Residual deposits of particulate P can accumulate in the vicinity of tile drains, and be mobilized during flow events (Stamm *et al.*, 1998). Moreover, leaching of dissolved phosphate in some locations may be triggered by high groundwater levels or flooding causing reduced conditions in the soils (Sallade and Sims, 1997; Martin *et al.*, 1997). Thus the macro- and meso-porosity of a field may not always be clearly related to P leaching.

During individual drain water or stream events, P concentrations at a given discharge may vary greatly between the rising and the recession stage, i.e. the hysteresis effect. The major types of hysteretic relationships in rivers were classified by Williams (1989) and hysteresis loops have been used to reveal the varying relationships between discharge and P water concentrations on a catchment scale (Pacini and Harper, 1995; Kronvang *et al.*, 1997). However, in a catchment these loops may simply reflect the fact that, during the rising stage, suspended solids and dissolved phosphate in the water may originate primarily from near-stream sources, often with a high proportion of arable land. During the recession stage, a major part of the water comes from far-stream sources, with different soil used for other purposes. Therefore, the hysteresis loops observed may simply reflect the heterogeneity of the catchment. In drainage water, different sources of P in a single field may also give a mixed response pattern of P concentrations. However, analysis of the smaller field scale might give a clearer indication of the dynamic interplay between hydrology and P concentrations than the larger catchment scale.

The aims of the present field-scale experiment were (1) to evaluate the number and shape of macropores by using a staining technique, (2) to measure the forms (PP, PO_4-P and other dissolved P) and amounts of P exported from the tile drain system; and (3) to classify and evaluate the PP and PO_4-P hysteresis loops documented in major transport events. Measurements of P export were undertaken during a six-year period (1992–1998). The focus was on P leaching, since P was considered to be the factor with the largest environmental impact on the recipient lake.

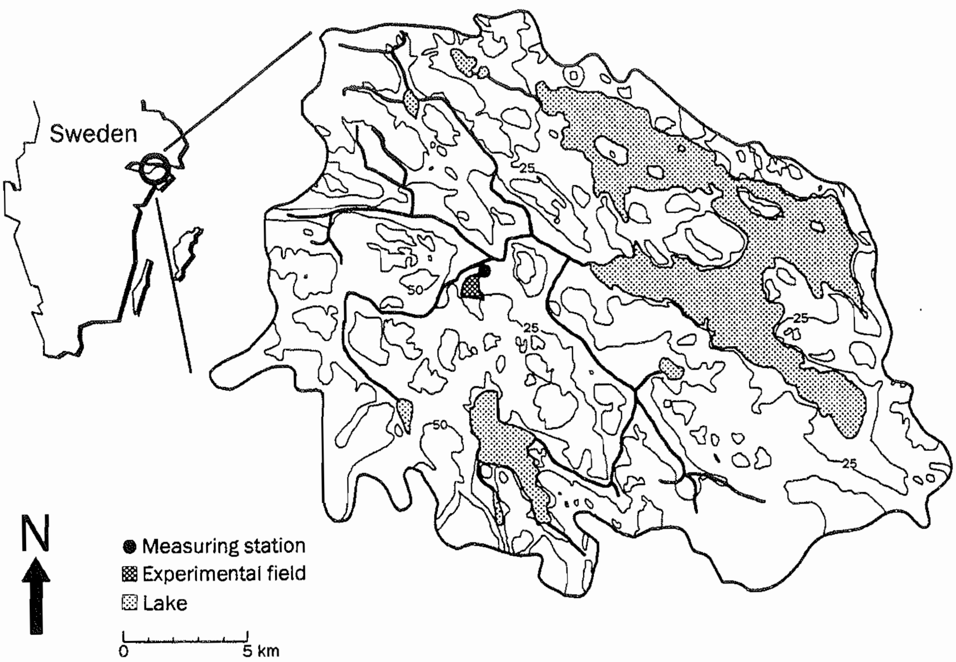

Figure 1. Location of the experimental field together with a contour map of the Lake Bornsjön drainage area. Altitudes (m) of some main contours are shown

MATERIALS AND METHODS

The experimental field

The field-scale experiments were carried out at a site 30 km south of Stockholm and 20 m above sea level (Figure 1). Drainage water from the study area flows into Lake Bornsjön, which is the reserve drinking water reservoir for the city. The 4·43 ha of arable land has a system of tile drains which were successively installed in the late 1940s and 1950s. The drain tiles have topsoil and some gravel backfill. They are situated at a depth of 1–1·2 m with irregular spacing. The main drainage line is 25 cm in diameter. In 1987 this was intercepted, and minor excavation was undertaken in order to install the measuring station at the end of the drainage system.

The field is flat, class 1 according to FAO–UNESCO (1988). The soil was classified as a *Eutric Cambisol* according to the same classification system. At a depth of 0·22 cm there is a distinct boundary between the topsoil and the grey horizon underneath. Under the second horizon at 0·43 m the soil is olive-grey under moist conditions and there are streaks of gley. Large aggregates with clear sharp-edged prismatic structures have been observed at a profile depth of 0·2–1 m. Several coarse, as well as fine, macropore cracks have been observed at a depth of 0·43–1 m. The bedrock is a veined garnet-gneiss with meta-argillite continuing much plagioclase, cordierite and biotite (Stålhös, 1968).

Soil P, extractable with an ammonium lactate solution and HCl (Egnér *et al.*, 1960), shows a moderate P status (Table I). Surface soil pH is relatively low (6·6) which is typical for many Swedish soils (Eriksson *et al.*, 1997). The ionic composition of the drainage water (Table II) is characteristic of this soil type, and is rich in calcium and magnesium.

Soil monolith studies

When measuring P losses from soil monoliths from the field, the occurrence of hydraulically active macropores in the soils was also studied. Eight undisturbed soil monoliths, each with a surface area of 6·83 dm^2, were taken in plastic (PVC) sewage pipes using a technique described by Persson and Bergström (1991). Because of the difficulty of sampling the compact subsoil, shallow monoliths were collected. Thus, the monoliths contained either the topsoil alone or the topsoil and c.20 cm of the subsoil. The monoliths were placed in pipes below ground level at a lysimeter installation and were exposed to natural weather conditions and drained under gravity. This has been described in detail elsewhere (Ulén *et al.*, 1998). In the early summer 1998, after five years of leaching measurements, the monoliths were brought into the laboratory and were slowly saturated with water for seven days. This was done from the bottom end (c.5–10 cm per day) in order to avoid air entrapment as much as possible. After saturation, the monoliths were allowed to drain under gravity for 3 days. Brilliant-blue with a concentration of 2 g l^{-1} was used for staining and was applied to the

Table I. The variation of texture, loss on ignition, pH, ammonium lactate and HCl-extracted P within a representative soil profile sampled in June 1996

Soil Depth (cm)	Clay (%)	Silt (%)	Sand, gravel (%)	Loss on ignition (%)	pH	P-Al	P-HCl
						(mg per 100 g dry soil)	
0–15	58·6	37·2	4·2	4·4	6·6	4·9	113
15–30	65·0	33·0	2·0	3·1	6·6	5·3	109
30–50	62·5	36·9	0·6	2·5	7·0	1·8	84
50–90	62·0	37·4	0·6	1·8	7·2	2·7	72

Table II. Mean volume weight ionic composition (meq l^{-1}) of drainage water 1988–1994

Na^+	K^+	Ca^{2+}	Mg^{2+}	Cl^-	SO_4^{2-}	HCO_3^-	NO_3^-
0·70	0·06	2·12	0·92	0·35	1·19	2·12	0·13

Table III. Crop, harvested yield, fertilization and number of tillings

Year	Crop	Harvest ($t\ ha^{-1}$)	Fertilization		Harrowing*	Cultivating	Disc harrowing
			Date	Amount ($kg\ P\ ha^{-1}$)			
1992	Winter wheat	5·4	—	—	2	2	1
1993	Grass seeding Green fallow	—	—	—	—	—	—
1994	Green fallow	—	—	—	—	—	—
1995	Unsuccessful Winter wheat	—	20/08/95	15	—	2	1
1996	Barley	4·0	01/05/96	14	2	3	—
1997	Barley	4·0	23/05/97	14	2	2	—
1998	Oats						

* Harrow with levelling board + roller.

soil surface with drip tubes at a rate of 7·7 mm hour $^{-1}$. The application was stopped when the stained wetting front arrived at the bottom of the monoliths. These were then cut into 10 or 20 cm sections and each transect was examined visually for the areal extent and continuity of the stained flow paths.

Drainage, weather conditions and cultivation

Drainage discharge from the experimental field was continuously measured with an open V-notch weir equipped with a pressure transducer and datalogger, as well as a mechanical water level recorder. Hourly precipitation and air temperature were measured during the entire experiment at a weather station situated in the lower part of the field.

During the initial three years of the study the land was in fallow and after an unsuccessful winter wheat crop (Table III), cereal crops receiving normal applications of fertilizer were grown. Conventional tillage methods of harrowing and sowing were used.

Water sampling and analysis

Tile drainage water from the experimental field was sampled flow-proportionally with an ISCO sampler controlled by a datalogger. Flow was calculated every minute, together with the volume of water that had passed the V-notch weir. Each sample consisted of 10 consecutive subsamples that corresponded to 0·2 mm of drainage. In total, more than 500 samples were taken during the period studied. In addition, manual samples were taken every two weeks at the same place, since this is the sampling technique used in the Swedish programme for monitoring experimental fields.

In all flow-proportional samples, total phosphorus (Tot-P) and suspended solid (SS) concentrations were determined. Particulate P (PP) was determined as the difference between Tot-P in unfiltered and filtered water. Dissolved phosphate (PO_4-P) was measured in filtered water. The remaining P, calculated as the difference between Tot-P and the sum of PP and PO_4-P, was simply called 'other P' and may have contained fine colloidal and dissolved P forms, as discussed by Haygarth *et al.* (1998).

All phosphorus determinations were undertaken in accordance with European Standard EN 1189 (European Committee for Standardization, 1996) but some of the pre-treatments differed from standard procedures. Thus, the filters used had a 0·20 μm pore size which is smaller than the commonly used Swedish Standard of 0·45 μm. This change was made because the smaller pore size filters trapped the fine clay and coarser colloidal particles that were present. Concentrations of SS were determined by weighing the material on the filters. During 1992–1996, dissolved phosphate [PO_4-P(c)] was also determined in water centrifuged for 20 minutes at a rate of 3000 rpm, the method used in the Swedish monitoring programme for experimental fields.

Table IV. Average air temperature, precipitation and drainage for the Oxelby experimental field

	Air temperature (°C)				Precipitation (mm)				Drainage (mm)			
Year Month	07/08	09/12	01/04	05/06	07/08	09/12	01/04	05/06	07/08	09/12	01/04	05/06
1992/93	17·8	3·9	1·3	12·2	7	186	88	100	0	103	36	3
1993/94	14·4	3·2	−1·3	10·8	161	181	120	146	34	120	108	5
1994/95	17·3	5·1	0·6	11·6	180	289	260	113	1	155	138	12
1995/96	16·8	3·0	−2·6	10·9	56	156	34	90	4	36	17	2
1996/97	15·7	3·8	0·2	11·8	99	158	110	206	1	24	32	78
1997/98	18·3	4·3	0·8	11·2	34	163	123	62	4	34	116	16
Average 1961/90	16·3	5·0	−0·5	13·0	134	169	132	85	7*	67*	79*	18*

*Average 1988–1998.

The transport of each P form and SS was calculated from water volumes and concentrations during each sampling interval, and was then summed to estimate the annual transport during one agrohydrological year (i.e. 1 July–30 June). Transport was divided by the total drainage volume to obtain a measurement for the mean concentration of phosphorus in the total volume of water.

RESULTS AND DISCUSSION

Weather and discharge

The months from September to December during the six-year period were colder and wetter than during the preceding 30 years (Table IV). The year 1994–1995 was unusually wet and rates of water loss from the field were high, while the following year was dry compared with the average precipitation. During the entire experiment, the winter period (January–April) was warmer than normal, except during 1996. This resulted in more precipitation in the form of rain than snow. On average, 80% of the precipitation falls as rain (National Atlas of Sweden, 1995).

Occurrence of macropores and P transport in monoliths

On average, the brilliant-blue 'waterfront' in the soil monolith experiment appeared at the base of the monolith when a water volume equivalent to c. 3% of the topsoil volume has passed through the soil columns. Larger amounts (10%) of water were bound in the deeper profiles. Few stained pores were present at a depth of 10 cm, but nearly all of them continued through the entire monolith depth. Clearly, the clay studied had the property of forming deep cracks. The prismatic character of the soil structure resulted in angular paths downwards. In contrast to the findings of Edwards *et al.* (1988), clay cracks, but not earthworm burrows, seemed to be of importance, and the frequency of the pores was very much less. The stained pores belonged to the medium-sized pore class according to the Beven and Germann (1982) classification, and were described as cracks in clay soils formed by shrinkage or by freezing–thawing cycles. They all had a narrow, rectangular shape and a mean area of 6 mm^2. For all the monoliths, representing more than 0·5 m^2, less than 1% of the surface area consisted of stained macropores at a depth of 30 cm. This shows that the hydraulically active area was a very small part of the total soil area.

The macropores were just visible at the end of the experiment but a seasonal difference in macroporosity is likely. However, based on these results, the relationship between macropores and P transport was indicated to be complex. A few macropores in the topsoil could be associated with greater leaching (monoliths 5 and 8, Table V). Since the monoliths were shallower than the drainage depth they were not meant to give accurate results for leaching throughout a deeper soil profile. For instance, surface-leached mineral P could absorb in the subsoil, and measured amounts (Table V) should only be regarded as a worst case scenario, delivering

Table V. Monolith depths and number and total area of continuous and stained macropores, together with annual leaching of total phosphorus during the period 1993–1998

Lysim. no	Depth (cm)	Macropore		Tot-P ($kg\,ha^{-1}\,year^{-1}$)
		Number	Area (mm^2)	
1	30	5	31	1·01
2	50	2	26	0·18
3	30	3	7	0·64
4	30	3	25	0·94
5	30	2	11	1·38
6	50	1	4	0·24
7	60	2	8	0·05
8	30	2	12	0·74

more P than the actual field. Mean total P leaching amounts (0·94 $kg\,ha^{-1}\,year^{-1}$ from a depth of 30 cm and 0·42 $kg\,ha^{-1}\,year^{-1}$ from a depth of 50 cm) were also much higher than the actual tile drain losses from the field. The P drainage from the topsoil of different monoliths varied by a factor of two, indicating a spatial variation of the P loss from the soil.

Forms of P leached and P concentrations

PP concentrations in drainage water from the experimental field were highest (average 0·101 $mg\,l^{-1}$), while PO_4-P concentrations were lower (average 0·037 $mg\,l^{-1}$), and the remaining P (colloidal or dissolved organic P) was found in the lowest concentrations (average 0·022 $mg\,l^{-1}$). In order to reduce the number of points, concentrations of PP and PO_4-P were grouped into five drainage classes (0–0·125, 0·125–0·250, 0·250–0·375, 0·375–0·500 and 0·500–0·625 mm $hour^{-1}$). Using simple regression, PP concentration was found to be controlled by discharge (Figure 2) whereas the PO_4-P concentration showed a weaker correlation with discharge.

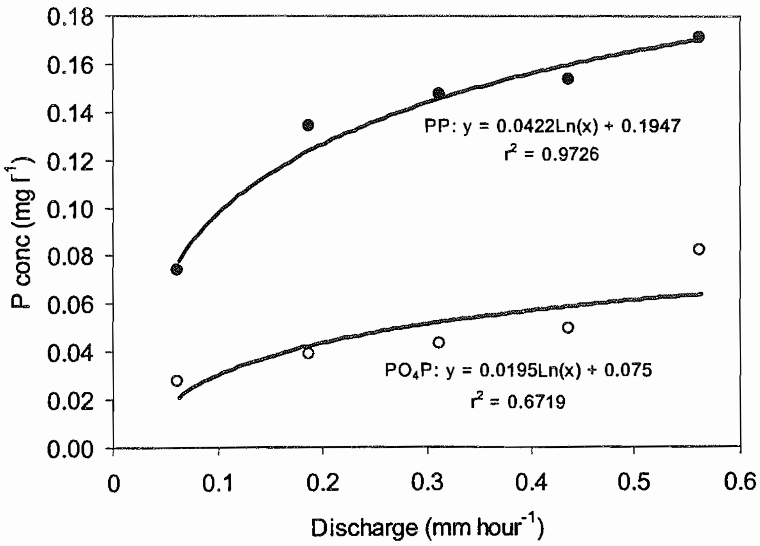

Figure 2. Grouped concentrations of particulate phosphorus (PP) and phosphate phosphorus (PO_4-P) as a function of discharge of five drainage classes

Table VI. Event drainage data and associated hysteresis response type for phosphorus concentration

Period	Year	Total drainage (mm)	Max drainage (mm hr^{-1})	Average drainage (mm hr^{-1})	Response type (PP)	Response type (PO_4-P)
2/11–5/11	92	27·2	0·66	0·27	3	3
12/11–15/11	92	25·5	0·65	0·27	1	2
25/11–28/11	92	12·1	0·48	0·13	3	3
16/9–20/9	93	27·4	0·66	0·23	3*	2*
13/10–17/10	93	12·5	0·54	0·09	3	2
4/12–6/12	93	19·2	0·57	0·16	2	2
16/12–22/12	93	31·9	0·56	0·19	2	1
13/1–19/1	94	21·8	0·49	0·13	2†	2†
3/3–12/3	94	11·0	0·36	0·05	3	2
23/3–26/3	94	10·3	0·28	0·13	2	2
29/3–5/4	94	19·3	0·31	0·12	2	2
5/4–13/4	94	13·1	0·33	0·06	2	2
17/9–26/9	94	41·4	0·65	0·20	3†	3†
2/10–10/10	94	32·3	0·60	0·17	3†	3†
8/12–17/12	94	22·9	0·61	0·11	2	2
27/12–12/1	95	17·4	0·22	0·04	2	2
6/2–8/2	95	5·5	0·23	0·11	3	2
17/3–30/3	95	34·4	0·63	0·11	3	2
18/4–27/4	95	21·1	0·49	0·10	2*	1*
27/4–4/5	95	17·7	0·46	0·10	2	3
10/5–18/5	95	15·1	0·52	0·08	2	2
23/11–2/12	95	17·0	0·29	0·09	2	2
5/4–9/4	96	11·8	0·37	0·10	3	3
3/12–15/12	96	20·0	0·35	0·07	3	2
9/5–18/5	97	22·5	0·58	0·10	3	3
16/6–25/6	97	34·3	0·72	0·15	3	3
26/12–30/12	97	5·3	0·20	0·05	2	1
31/12–10/1	98	34·4	0·32	0·13	2†	2†
9/2–13/2	98	23·5	0·47	0·20	2	2
16/2–27/2	98	15·2	0·31	0·05	2	2

1 Single simple flush with no or unclear hysteresis.
2 Simple flush with P peak occurring simultaneously or after discharge peak.
3 Simple flush with P peak occurring before discharge peak.
* Three consecutive events.
† Two consecutive events.

P hysteresis curves

The water samples from the experimental field were collected primarily to permit accurate calculation of P export. Consequently, they were sampled on a flow-proportioned basis and sampling was not initiated by the start of precipitation or by saturated conditions in the soil. One should also keep in mind that measured concentrations represented mixed samples collected over a 2 mm discharge period, during which the rate of discharge may have fluctuated. Since these conditions may obscure some of the detail of the PP and PO_4-P hysteresis loops, they were only categorized in three response types, one of which was clearly observed for all of the episodes according to Table VI. Except for those showing no, or unclear, hysteresis (type 1), many winter and spring events were of type 2, where the concentrations on the recession limbs of the hydrographs were higher than those on the rising limbs. Hysteresis in these events was counterclockwise. This may occur during snowmelt when sediment with a high proportion of P is available in surface soil, but is not transported until the discharge is high. An example of such a loop is shown in Figure 3a.

A third type of loop occurs when the P peak precedes the discharge peak, producing a clockwise hysteresis loop. This may reflect either only a small amount of P being available or long-lasting and/or intense drainage.

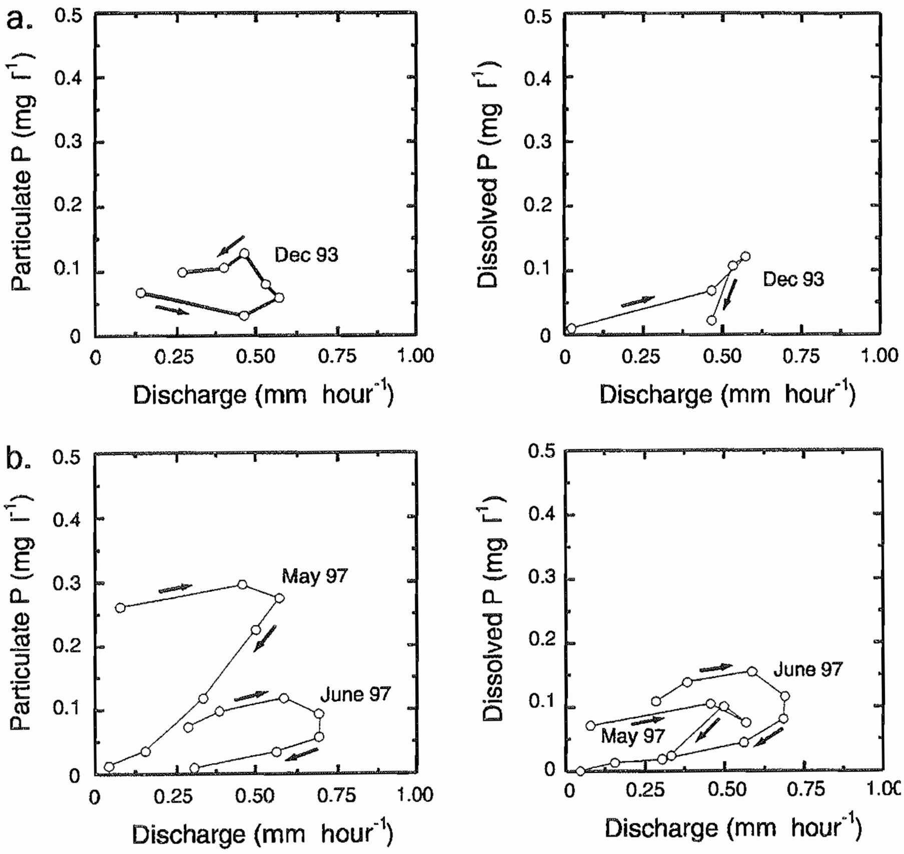

Figure 3. Hysteresis curves during (a) December 1993 and (b) May and June 1997

Thus, a sustained discharge peak after long-term precipitation often tended to deplete the SS available to transport P, and clockwise PP loops appeared. Consequently, the ratio of PP to SS was relatively low during such events (average 0·08%), compared with 0·10% during counterclockwise loops. The volume of subsurface water may be of great importance during such recession limbs when P-rich surface water may be diluted during the later stages of the event. Clockwise loops, when the amount of easily eroded material was limited, were frequent during summer and autumn. When two events occurred in close succession, the second PP loop might have a smaller maximum concentration (Figure 3b), probably due to lower amounts of particulate material being available for transport. In contrast, PO_4-P concentrations were higher during the second loop, most likely as a result of the fertilizer application on 23 May 1997.

In total, clockwise and counterclockwise loops were of equal frequency. In contrast, in larger rivers, clockwise loops have been more frequently reported than counterclockwise ones (Williams, 1989). At the larger river catchment scale, long-lasting events may be more frequent, favouring clockwise loops.

Phosphorus loads and duration

Table VII presents estimates of P loss calculated using both flow-proportional and manual sampling of the experimental field. The measured transport of Tot-P and PO_4-P estimates on the basis of manual sampling was only 41 and 34%, respectively, compared with the transport calculated using more frequent flow-proportional sampling. This stresses the importance of a good sampling strategy for accurate calculation of P export from this type of soil. Events with P fluxes larger than $0{\cdot}5\ 10^{-3}$ kg Tot-P ha^{-1} hour^{-1}

Table VII. Drainage sediment and nutrient losses calculated from flow-proportional and manual sampling and the relationship between the estimates based on the two different sampling strategies

Year	Discharge (mm)	Tot-P	PP	PO_4-P	Other P	PO_4-P(c)*	SS
		(kg ha^{-1} year^{-1})					
Flow-proportional samples							
1992/93	142	0·29	0·18	0·09	0·02	0·09	92
1993/94	267	0·41	0·25	0·09	0·07	0·17	151
1994/95	326	0·46	0·30	0·09	0·07	0·16	221[1]
1995/96	38	0·05	0·02	0·02	0·01	0·02	25
1996/97	133	0·23	0·14	0·07	0·02	0·08	109
1997/98	170	0·28	0·19	0·04	0·05	—	207
Sum		1·72				c.0·58	805
Manual samples							
1992/93	147	0·07	—	—	—	0·02	32
1993/94	267	0·09	—	—	—	0·03	40
1994/95	326	0·25	—	—	—	0·07	228
1995/96	38	0·01	—	—	—	0·00	6
1996/97	133	0·05	—	—	—	0·02	35
1997/98	170	0·24	—	—	—	0·05	125
Sum		0·71				0·19	466
Manual/flow prop.		0·41				0·34	0·58

*Period April–June not included.

Table VIII. Reported mean annual subsurface P losses from artificially (0·9–1 m deep) drained clay soils (clay, silty clay or clay loam)

Number of years studied	Tot-P	PP	PO_4-P	(Tot-P − PP)	Location of site	Reference
	(kg ha^{-1} year^{-1})					
6	—	—	—	0·16	Ontario	Bolton *et al.* (1970)
3	0·28	—	0·07	—	Indiana	Bottcher *et al.* (1981)
2	—	0·14	—	—	Ontario	Culley *et al.* (1983)
3	—	—	0·08	—	England	Hawkins and Scholefield (1996)
6	0·28	0·14	0·07	0·15	Sweden	This study

accounted for 50% of the total transport. The duration of such high loads are limited (Figure 4), and on average was only 140 hours year^{-1}. Longer episodes of high-rate P transport occurred during the first three years, while the duration was shorter for the other three years, especially during 1995–1996.

Average annual losses of P in different forms from this soil are consistent with other findings from clay soils with deeper tile drains (Table VIII). This is probably a matter of chance in view of the very different climatic conditions and agricultural management practices. These factors are known to greatly affect subsurface P export (Culley *et al.*, 1983). When comparing a wet year in the present study (1994–1995) with a dry one (1995–1996), Tot-P export was found to be nine times as high in the former, and the PP export was even higher during the wet year. A relationship between seasonal precipitation and Tot-P losses was also evident, at least during the autumn (Figure 5).

The soil was ploughed annually during the last three years of the investigation and the macropores should consequently have been disrupted, but the soil had not been tilled during the first three years. In spite of this, mean concentrations of PP and PO_4-P were exactly the same (0·10 and 0·04 mg l^{-1}, respectively) during both periods. Hence, cultivation did not decrease PP or PO_4-P concentrations. This is in contrast to the findings of Gaynor and Findlay (1994) who found increased PP and especially PO_4-P concentrations during

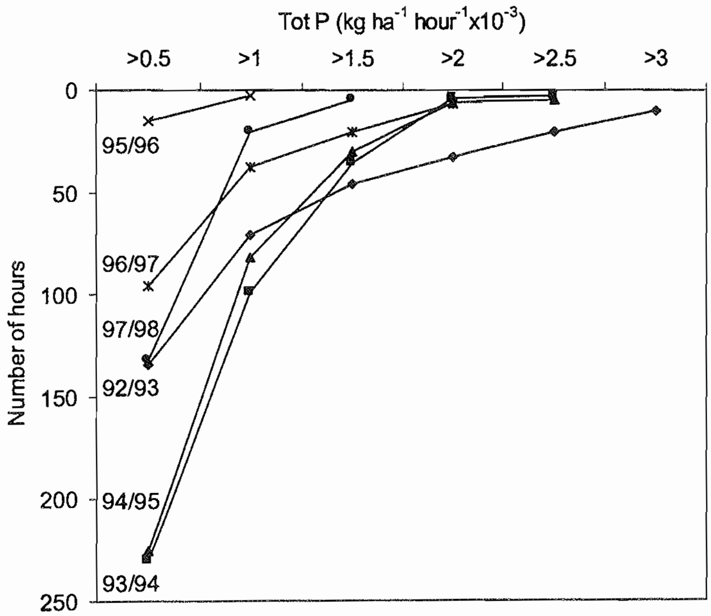

Figure 4. Duration of total phosphorus transport

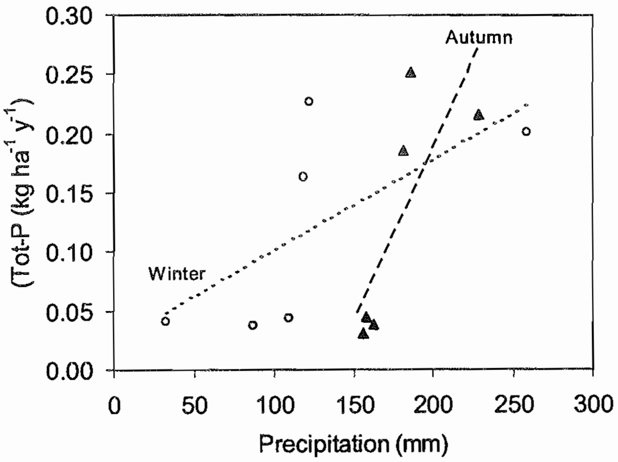

Figure 5. Total phosphorus (Tot-P) loss versus precipitation during autumn (1 September to 31 December) and winter (1 January to 30 April)

conservation, compared with conventional tilling of a clay loam. However, as pointed out by Grant and Phillips (1979), macropore flow still occurs after ploughing and may not have been much affected by tilling this particular clay soil with its clear aggregate structure.

CONCLUSIONS

Substantial drainage losses of phosphorus (P) were recorded for a structured heavy clay soil in Central Sweden.

No simple relationship was found between macroporosity and Tot-P drainage losses as evaluated using soil monoliths. However, spatial variation of P transport through the soil was indicated.

Analyses of the relationship between P concentration and discharge from the drainage system showed clockwise as well as counterclockwise hysteresis loops with an equal frequency. Clockwise loops appeared mainly during the summer and autumn and counterclockwise loops during the winter and spring.

The importance of an appropriate sampling programme for determining the P losses was emphasized.

Subsurface phosphorus losses from this soil were unexpectedly similar to findings from other clay soils. In this study the export was highly affected by weather conditions. However, it indicated that cultivation does not influence drainage P concentrations.

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REFERENCES

Beven, K. and Germann, P. 1982. 'Macropores and water flow in soils', *Wat. Resour. Res.*, **18**, 1311–1320.

Bolton, E. F., Aylesworth, J. W., and Hore, F. R. 1970. 'Nutrient losses through tile drains under three cropping systems and two fertility levels on a Brookstone clay soil', *Can. J. Soil Sci.*, **50**, 275–279.

Bottcher, A. B., Monke, E. J., and Huggins, L. F. 1981. 'Nutrient and sediment loadings from a subsurface drainage system', *Trans. ASAE*, **24**, 1121–1226.

Culley, J. L. B., Bolton, E. F., and Bernyk, V. 1983. 'Suspended solids and phosphorus loads from a clay soil: I. Plot studies', *J. Environ. Qual.*, **12**, 493–498.

Edwards, W. M., Norton, L. D., and Redmond, C. E. 1988. 'Characterization macropores that affect infiltration into nontilled soil', *Soil Sci. Soc. Am. J.*, **52**, 483–487.

Egnér, H., Riehm, H., and Domingo, W. R. 1960. 'Untersuchungen über die chemische Bodenanalyse als Grundlage für die Beurteilung des Nährstoffzustanden des Böden. II Chemische Extraktions-methoden zur Phosphor- und Kaliumbestimmung, *Annal. Landwitsch. Hochs. Swedens*, **26**, 199–215.

Eriksson, J., Andersson, A., and Andersson, R. 1997. 'Current status of Swedish arable soils', *Report 4778*. Swedish Environmental Board, Stockholm.

European Committee for Standardization, 1996. 'Water Quality. Determination of phosphorus. Ammonium molybdate spectrometric method, European standard EN 1189'. European Committee for Standardization, Brussels. 18 pp.

FAO-UNESCO 1988. *Soil Map of the World. Revised Legend.* World Soil Resources Reprot 60, FAO, Rome. 119 pp.

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

Grant, W. T. and Phillips, R. E. 1979. 'Consequences of water movement in macropores', *J. Environ. Qual.*, **8**, 149–152.

Grant, R., Laubel, A., Kronvang, B., Svendssen, L.-M., and Fugelsang, A. 1996. 'Loss of dissolved and particulate phosphorus from arable catchments by subsurface drainage', *Water Res.*, **30**, 2633–2642.

Hawkins, J. M. B. and Scholefield, D. 1996. 'Molybdate-reactive phosphorus losses in surface and drainage waters from permanent grassland', *J. Environ. Qual.*, **25**, 727–732.

Haygarth, P. M., Hepworth, L., and Jarvis, S. C. 1998. 'Forms of phosphorus transfer in hydrological pathways from soils under grazed grassland', *Eur. J. Soil. Sci.*, **42**, 65–72.

Heckrath, G., Brooks, P. C., Poultron, P. R., and Goulding, K. W. T. 1995. 'Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbank experiment', *J. Environ. Qual.*, **24**, 904–910.

Kladivko, E. J., van Scoyoc, G. E., Monke, E. J., Oates, K. M., and Pask, W. 1991. 'Pesticide and nutrient movement into subsurface tile drains on a silt loam soil in Indiana', *J. Environ. Qual.*, **20**, 264–270.

Kronvang, B., Laubel, A., and Grant, R. 1997. 'Suspended sediment and particulate phosphorus transport and delivery pathways in an arable catchment, Gelbaek stream, Denmark', *Hydrol. Process*, **11**, 627–642.

Martin, H. W., Ivanhoff, D. B., Graetz, D. A., and Reddy, K. R. 1997. 'Water table effects on histosol drainage water carbon, nitrogen and phosphorus', *J. Environ. Qual.*, **26**, 1062–1071.

National Atlas of Sweden 1995. 'Climate, Lakes and Rivers', in Raab, B., and Vedin, H. (Eds), *The Swedish Environmental Protection Agency*, SNA, Stockholm.

Pacini, N. and Harper, D. 1995. 'Transport and fate of sediments and phosphorus in an equatorial watershed: the influence of geology, climate and land use', in Kronvang, B., Svendsen, L. M., Sibbesen, E. (Eds), *Sediment and phosphorus — erosion and delivery, transport and fate of sediments and sediment-associated nutrients in watersheds.* Proceedings of a workshop 9–12 October 1995, Silkeborg, Denmark. National Environmental Research Institute, Technical Report No 178. pp 104–105.
Persson, L. and Bergström, L. F. 1991. 'A drilling method for collection of undisturbed soil monoliths', *Soil Sci. Soc. Am. J.*, **55**, 285–287.
Quinsberry, V. L. and Phillips, R. E. 1976. 'Percolation of simulated rainfall under field conditions', *Soil Sci. Soc. Am. J.*, **40**, 484–489.
Rekolainen, S., Ekholm, P., Ulén, B., and Gustafson, A. 1997. 'Phosphorus losses from agriculture to surface waters in the Nordic countries', in Tunney, H., Carton, O. T., Brookes, P. C. and Johnston, A. E. (Eds), *Phosphorus Loss from Soil to Water.* CAB International, Oxon. pp. 77–94.
Ryden, J. C., Syers, J. K., and Haris, D. 1973. 'Phosphorus in runoff and streams', *Adv. Agron.*, **25**, 1–45.
Sallade, Y. E. and Simms, J. T. 1977. 'Phosphorus transformations in the sediments of Dalaweare's, agricultural drainways: II Effect of reducing conditions on phosphorus release', *J. Environ. Qual.*, **26**, 1579–1588.
Sims, J. T., Simard, R. R., and Joern, B. C. 1998. 'Phosphorus loss in agricultural drainage: Historical perspective and current research', *J. Environ. Qual.*, **27**, 277–293.
Stamm, C., Flühler, H., Gächter, R., Leuenberger, J., and Wunderli, H. 1998. 'Preferential transport of phosphorus in drained grassland soils', *J. Environ. Qual.*, **27**, 515–522.
Steenhuis, T. S. and Muck, R. E. 1988. 'Preferred movement of nonabsorbed chemicals on wet, shallow sloping soils', *J. Environ. Qual.*, **17**, 376–384.
Stålhös, G. 1968. *Map of the Solid Rocks of the Stockholm Region.* SGU serie Ba, Stockholm.
Turtola, E. and Paajanen, A. 1995. 'Influence of improved subsurface drainage on phosphorus losses and nitrogen leaching from a heavy clay soil', *Agric. Wat. Mgmt.*, **28**, 295–310.
Ulén, B. 1995. 'Episodic precipitation and discharge events and their influence on losses of phosphorus and nitrogen from tile drained arable fields', *Swedish J. Agric. Res.*, **25**, 25–31.
Ulén, B., Shirmahammadi, A., and Bergström, L. F. 1998. 'Phosphorus transport through a clay soil', *J. Sci. Health*, **A33**, 67–82.
Williams, G. P. 1989. 'Sediment concentration versus water discharge during single hydrologic events in rivers', *J. Hydrol.*, **111**, 89–106.
Øygarden, L., Kvaerner, J., and Jensen, P. D. 1997. 'Soil erosion via preferential flow to drainage systems in clay soils', *Geoderma*, **76**, 65–86.

Linking a hydrological river basin model and field-scale models for phosphorus transport

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Abstract

The European Union's (EU) Water Framework Directive stipulates that nutrient transport shall be monitored and calculated for water districts. There is presently no phosphorus (P) transport model available on a catchment scale adapted for Nordic conditions, with a considerable impact of soil frost, snow accumulation and melt, and with shallow groundwater, and with a need for input data and driving variables that are available within most water districts. This paper presents the first step in the creation of such a model. In the field-scale P model ICECREAM, the hydrology compartment was replaced with the catchment-scale hydrological model HBV. To be able to calculate particulate P (PP) losses through drainage tiles the sub-model PARTLE was incorporated in the model. The resulting HBV-P model will, in the future, be expanded with sub-models for leaching from forests and other land, and processes that occur in rivers and lakes, as well as contribution from point sources. In Finland another approach has been tried, where a simple catchment scale P model has been added to the HBV model (Bilaletdin *et al.*, 1994).

The results are promising, in most cases HBV-P performed better than ICECREAM on the three tested sites, located on different soils and in different climate regions of Sweden.
The performance and data demands of PARTLE were, however, less satisfactory. Pearson correlation for PP losses simulated with PARTLE and measured values (0.11 to 0.48) were lower than the correlation between simulated dissolved P (SRP) losses and measured values (0.29 to 0.71). The loss of P as both SRP and PP through drainage tiles is extremely episodic and large transport can occur during prolonged rain periods and/or if snow melt takes place on frozen soil. This is something the P-module of ICECREAM cannot handle well and that needs to be dealt with in the next step of model development.

Keywords: phosphorus, modelling, agriculture, ICECREAM, PARTLE, HBV

Introduction

In line with the European Union's (EU) Water Framework Directive, the countries within the EU, including Sweden, will be divided into water districts, which among other things will be responsible for control and operational supervision of nutrient transport. In order to carry out this task, tools for estimates of river-basin transport, source apportionment and division between natural and human-induced temporal variability are needed. For Swedish conditions, models for river basin nitrogen (N) transport have been developed and tested, whereas such models not are available for phosphorus (P) transport. The model system used for nitrogen, HBV-N (Arheimer and Brandt, 2000), is based on the hydrological river basin model HBV (Bergström, 1995), and the soil-leaching model SOIL-N (Johnsson *et al.*, 1987). Similar work has now been initiated for P-modelling, where the field-scale models ICECREAM (Posch and Rekolainen, 1993; Rekolainen and Posch, 1993; Rekolainen *et al.*, 1998) and PARTLE (Shirmohammadi *et al.*, 1998) have been linked to the HBV model. This work is the first step in the development of a dynamic river basin model for source apportionment, and subsequent analysis of remedies for reduction of P transport from land-based sources to the sea. A prerequisite for the scaling up of field models by integrating them with a river basin model is that all model components use hydrological and climatologic routines available in the river basin model.

In our work, we are aiming to link the field models ICECREAM and PARTLE to the river basin model HBV. The HBV model is conceptual and calculates run-off from a catchment, divided into sub-catchments, using a daily time step. To enable a scaling up of the ICECREAM and PARTLE models to be used on a catchment scale, their hydrological/climatologic routines have been substituted with similar routines available in the river basin model. Correspondingly, hydrological/climatologic routines that are vital for ICECREAM and PARTLE, but not available in the river basin model, have been added to the HBV model in a form suitable for application on a river basin scale that only requires easily obtainable input data. In addition to work on substituting the hydroclimatological routines, work has also been initiated to substitute the ICECREAM routines for calculation of soil erosion and enrichment with routines more suitable for use on a catchment scale. This work will be followed by development of routines for stream and lake processes, and for inclusion of P losses from non-agricultural land, and from point sources to the HBV-P model.

The aims of this paper are to test:
(I) the effect on simulated hydroclimatological variables from substituting routines within the field scale models with routines from the river basin model;
(ii) the effect on simulated concentrations and transport of different P fractions in surface run-off and leaching to tiles if the hydroclimatological routines in the field models are substituted by the routines in the river basin model;
(iii) if weaknesses in the field-scale models can be detected;
and to
(iv) recommend future work on scaling up of the field-scale models.

P losses from agricultural land

Phosphorus has rather complicated soil chemistry. The total amount of P in the soil can be divided into pools, with P bound to soil particles in various ways, and with different behaviour depending on their chemistry. P can be adsorbed to the surface of minerals. Equilibrium between dissolved and adsorbed P is quickly established. However, P is often strongly bound at the surface and to the mineral grains, thus making it unavailable for plants and for dissolution to the soil water. P can also be bound to organic matter of different composition and with varying binding strength.

There has been an increasing awareness that in addition to soil-surface transport of P bound to eroded soil particles, P can be transported from the field along several different pathways. These include surface run-off of dissolved P and particulate P, subsurface transport of both dissolved and particulate P by macropore flow (Stamm *et al.*, 1998), and slow transport of soluble P through the soil matrix. P might also be released under anaerobic conditions in the soil profile due to reduction of iron from Fe^{3+} to Fe^{2+}, which does not bind P (Lindsay, 1979).

Anaerobic conditions are linked to high groundwater levels, which also favour high surface run-off and tile drain discharge, causing large losses of P. The subsurface pathways connect the soil surface and the sub-drainage system of the soil, from which P is rapidly transported to connecting watercourses (Ryden *et al.*, 1973). Loss of P both with surface run-off and through drainage tiles is highly episodic (Grant *et al.*, 1996). For instance 45% of the P transport in tile drains during a three-year period in an experimental field with clayey soils in southern Sweden emanated from only two events. Both events were characterised by heavy rain on newly frozen soil (Ulén, 1995).

In addition to knowledge of processes that determine P turnover and release from the soil, surface and subsurface water flow is the driving force for P transport. A model of P transport from agricultural fields must thus include both components describing P turnover and release, and water flow below and on top of the soil surface.

Even though the basic understanding of different processes that are involved in P transport from agricultural fields are known, there is a lack of knowledge about their extend and duration. When modelling P transport, it is therefore inadvisable to solely rely on physical descriptions of the systems, i.e. calibration and verification against measured concentrations and transported loads are needed. It should also be considered that physically based models need substantial amounts of soil and other physical parameters, and in addition, detailed information about a large number of driving variables, which limits the possibility of using this sort of models as an integrated part of a river basin model. An alternative is to use conceptual models based on a combination of known physical conditions and empirical knowledge, which require less detailed information about soils, land-use and other physiographic parameters, and of driving variables

Particle-bound P transport is usually modelled with some form of empirical sediment loss routine, which is often based on the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978). The component used for surface run-off is usually derived from the SCS-curve number techniques developed by the US Dept. of Agriculture, with parameters adapted for North American conditions. These routines are used, for example, in the CREAMS (Knisel, 1980) and GLEAMS (Knisel, 1993) model concepts, which are the most widely used model for sediment and P transport from agricultural fields. Overall, the number of existing field-scale P models is limited. Field-scale models of P transport have only been applied in a small number of European countries (Schoumans and Chardon, 2000). The WAVE model, developed at the University of Leuven, has been used in Belgium. It is primarily a N-simulation model that has been extended to simulate P. Both organic and inorganic P are modelled. (Schoumans and Chardon, 2000). The ICECREAM model, which is an adjustment of GLEAMS for Nordic conditions, has been used in Finland (Rekolainen and Posch, 1993). The MORPHO model that only simulates inorganic P and loss of dissolved P has been applied in Germany (Schoumans and Chardon, 2000). In the Netherlands, the ANIMO model, which is a physically based model for N simulation that has been extended to P, has been developed and used. ANIMO describes inorganic P reactions and organic P cycles. It has been developed for sandy non-calcareous soils in flat areas, where subsurface run-off is the dominating pathway (Schoumans and Groenendijk, 2000). In Sweden, the GLEAMS model has been used (Shirmohammadi *et al.*, 1998; Ulén *et al.*, 1998).
The use of different models in different countries makes it difficult to regionally compare modelled P transport from fields. Because the models have different approaches to the problem, their results differ and they may simulate different pathways of P transport. In most cases, however, the choice of model is based on an adaptation to local conditions that drive P transport from fields.

The advantages of including field-scale modelling of P transport in a river basin model, instead of using standard values or simple regression models is substantial, since it allows simulation of scenarios of, for example, the effect of changing the amount of fertiliser applied and method of tilling.

Models used

ICECREAM was used to simulate dissolved and particulate P from surface run-off, and dissolved P from the root-zone. PARTLE was used to simulate losses of particulate P from macropores in the root-zone (Figure 1). The models were integrated in such a way that hydroclimatological (i.e. waterflow, snowmelt and soilfrost) routines in the field-scale models were substituted by those in HBV. In addition, some additional routines that were needed by ICECREAM were added to the HBV model. The results from the integrated model, HBV-P, with regard to transport of water and P, were then compared with the results when using the hydrolimatological routines available in the ICECREAM model. The models used are described below, with emphasis on the differences in model structures between ICECREAM and HBV. All models are conceptual, and use a daily time step. PARTLE was developed as a sub-model to GLEAMS, and needs hydrological input from the model it is linked to.

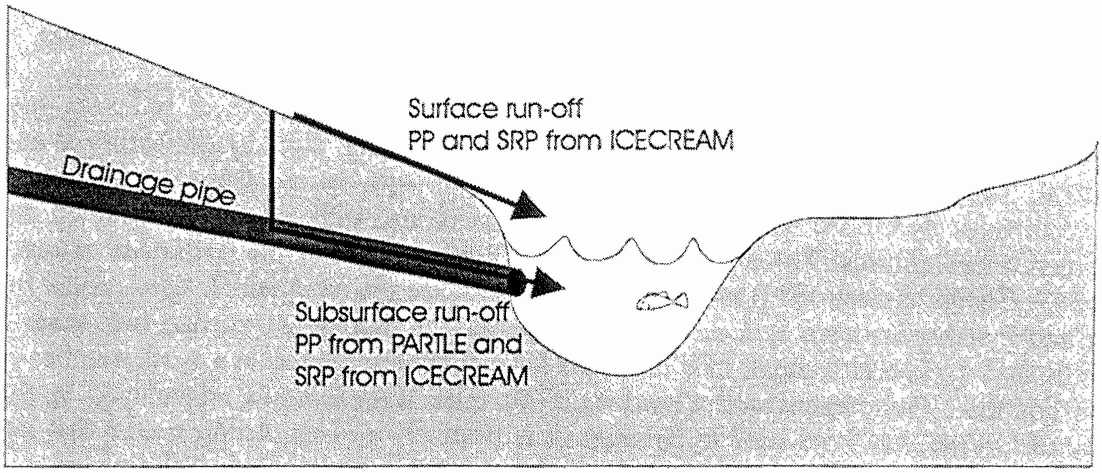


Figure 1. The different P models and simulated pathways of P loss.

ICECREAM

The model consists of sub-models covering hydrology, erosion, plant nutrients and pesticides. P and N cycles are basically calculated independently of each other, with the exception of calculations of degradation of organic matter, which involves both P and N amounts. The P cycle includes mineralisation, immobilisation, chemical fertilisation and manure application, crop uptake, and losses by erosion, run-off and leaching through the root-zone. P is stored in five pools (Figure 2): (i) P bound in very stable minerals; (ii) P bound to less stable minerals; (iii) labile P, which is the central pool form where leaching and plant uptake takes place; (iv) fresh organic P; and (v) organically bound P, a large, rather stable pool. The main deviations from the CREAMS and GLEAMS models concern adaptations of the erosivity factor in the Universal Soil Loss Equation, estimated from Finnish rainfall data (Posch and Rekolainen, 1993), and implementation of a new snow accumulation and snow melt model, the use of an adjustable albedo for evapotranspiration calculations, the implementation of a plant growth model for calculation leaf area index and soil loss ratio (Rekolainen and Posch, 1993).

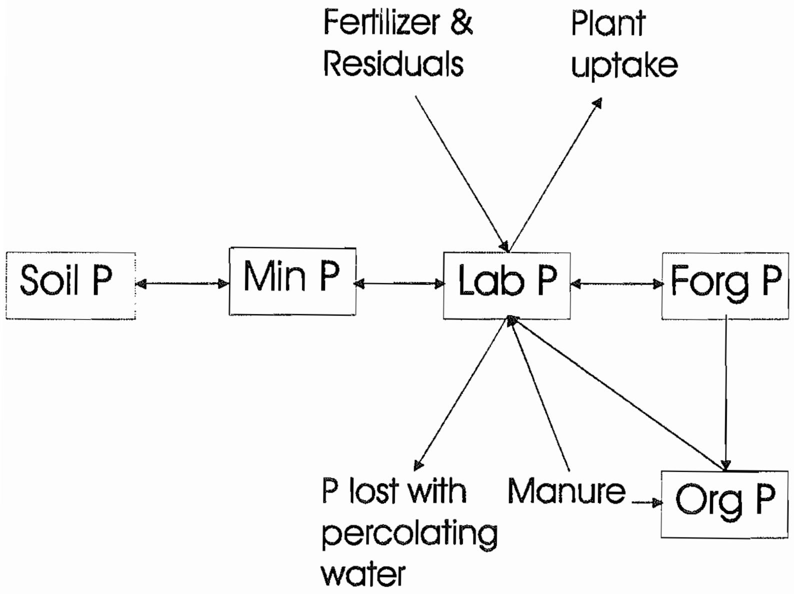


Figure 2. The P sub-routine in ICECREAM. Strongly bound inorganic P (SoilP), less strongly bound inorganic P (Min P), P in equilibrium with soil-water (Lab P), fresh organic P (Forg P), humus-bound P (Org P).

PARTLE

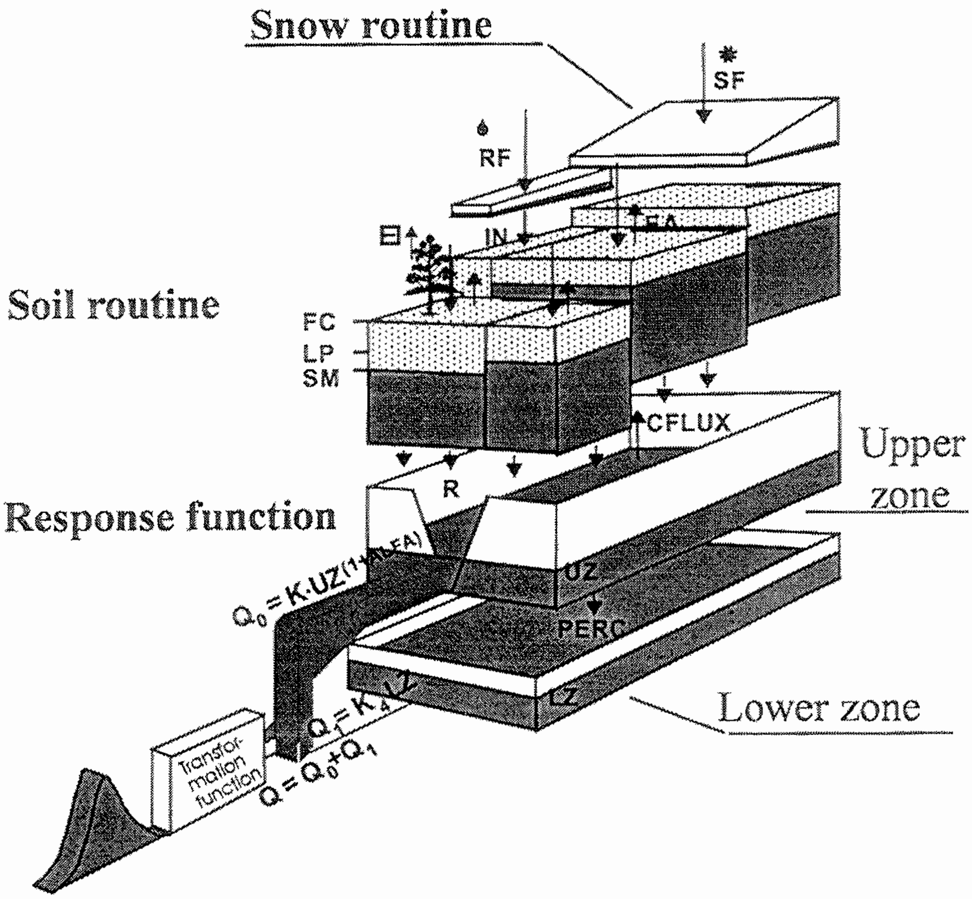
The PARTLE submodel assumes that particulate P (PP), attached to small, suspended soil particles, is transported from the topsoil through macropores with high hydraulic conductivity to tile drains. Daily sediment yield through the macropores is calculated by multiplying daily percolation with measured yearly mean concentration of suspended solids in the drainage water (Shirmohammadi *et al.*, 1998; Ulén *et al.*, 1998). The relationship between PP and suspended solids has been shown to be strong (Grant *et al.*, 1996). Concentrations of suspended solids was multiplied with soil surface phosphorous concentrations (simulated with, e.g. ICECREAM) and a correction factor for the soil's hydraulics and hydrology. The latter includes hydraulic conductivity for both macro and matrix water flow and the ratio of annual rainfall to the annual drain outflow (Shirmohammadi *et al.*, 1998; Ulén *et al.*, 1998).

$$PP = SC * PERC * PC * (K_{mac} / K_{mat}) / (1 - R) * f_{conversion} \quad (1)$$

PP	= PP loss though subsurface drains		(kg ha^{-1})
SC	= Average annual sediment conc. in the subsurface drains	Input	(mg/l)
PERC	= Daily percolation	Modelled	(cm)
PC	= LabP conc. in top layer	Modelled	(μg/g)
K_{mac}	= Macropore hydraulic conductivity	Input	(cm/h)
K_{mat}	= Matrix hydraulic conductivity	Input	(cm/h)
R	= Ratio of annual percolation to annual precipitation	Input	-

HBV

HBV is a hydrological model for calculation of river discharge (Figure 3) (Bergström, 1995). The model is conceptual and calculates run-off from a river basin, divided into sub-basins, using a daily time step.



SF = Snowfall	R = Run-off
RF = Rain	CFLUX = Capillary Flux
IN = Infiltration	UZ = Upper Zone
EA = Actual Evaporation	LZ = Lower Zone
EI = Interception Evaporation	PERC = Percolation
SM = Soil Moisture	K, K4 = Recession Parameters
FC = Field Capacity	ALFA = Recession Parameter
LP = Limit for potential evaporation	Q_0, Q_1 = Runoff Components

Figure 3. Schematic structure of the HBV-96 model in a one basin application

COMPARISON OF HYDROCLIMATOLOGICAL ROUTINES IN ICECREAM AND HBV.

In ICECREAM and HBV, it is possible to divide the soil profile into several layers. In this work we have used three layers, representing 0-0.01 m, 0.01-0.3 m, and 0.3-0.7 m below the soil surface. In the HBV model, there are options to use different routines. The routines described below are those used in the present work.

Snow accumulation and melt. The subroutine in ICECREAM is similar to that of the HBV model. In both models, daily air temperature is used as the driving variable. A threshold temperature interval is used to determine a linear decrease of the amount of precipitation that falls in the form of rain or snow. Snow melt is calculated by a simple degree-day method. The main difference compared to the HBV model is that in the ICECREAM model, evaporation from snow is assumed to take place.

Surface run-off. ICECREAM computes the daily surface run-off using the CREAMS approach. The curve number is selected from a table, and depends on soil, crop, land management, and drainage conditions. When the soil is frozen, a higher curve number is applied. The daily amount of surface run-off is a function of soil moisture and the amount of water reaching the ground. The original HBV model does not include a surface run-off routine. In connection to this work, a surface run-off routine similar to that used in ICECREAM was added to the HBV model. The routine in HBV used in this study, in contrast to ICECREAM, only included one curve number for unfrozen conditions, independent of whether the plot was harvested or not.
Sediment load. Estimates of sediment loads with ICECREAM are based on a modification of the Universal Soil Loss Equation (USLE). For the HBV model, a simple daily erosion model was used, where HBV estimates of surface run-off were used for calculation of erosivity and daily erosion:

$$\text{Erosivity} = (\text{precipitation}+\text{melt})^{a} * b \qquad (2)$$
$$\text{Erosion} = \text{erosivity} * \text{surface run-off} * c \quad (\text{kg ha}^{-1})$$

Where ^{a} was set to 1.18, b to 0,6036, and c calibrated against measured sediment loads.

Evapotranspiration. In ICECREAM, transpiration and soil evaporation are calculated for each soil layer by a model presented by Ritchie (1972). This model is driven by mean daily temperature and solar radiation (or cloudiness). In addition, parameter values for albedo for snow, bare soil, and vegetation must be given. In the HBV model, evapotranspiration is not divided into transpiration and soil evaporation. The evapotranspiration routine is driven by air temperature. Alternatively, if available, monthly standard values can be used. It is set to zero if the air temperature is below 0°C. The P part of the ICECREAM model only requires soil evaporation for a routine of P evaporation between layers. In the model integration, soil evaporation was set to zero when driving the P routines by the HBV model.
Percolation. In ICECREAM, percolation from each layer is calculated with consideration to saturated conductivity, field capacity, wilting point, and maximum soil water storage. In HBV, percolation parameters consists of field capacity and a parameter that controls the amount of percolation generated from water infiltration at a certain soil moisture deficit (Bergström, 1995). In addition to percolation from the three soil compartments, the model includes a response function that transforms percolated water into run-off (Figure 3).
Soil temperature. In ICECREAM, soil temperature at different layers is computed according to Williams *et al.* (1990). The routine is based on running averages of air temperature, with corrections for snow cover, soil moisture, and biomass. Also HBV includes a soil temperature routine based on accumulated air temperature, a small temperature contribution from deeper soil layers, soil moisture content, and snow accumulation (Lindström *et al.*, 1996).
Soil frost. In periods when soil frost occurs, the amount of calculated surface run-off increases significantly since a higher value of the curve number parameter CN2 is used. In the original ICECREAM model (Rekolainen and Posch, 1993), soil frost occurs in the period in autumn when a threshold temperature sum is reached, and remains until spring next year, when the absolute sum of accumulated temperatures reaches a threshold. However, in southern Sweden, soil frost may come and go several times during the winter. To make ICECREAM more flexible, the original soil frost routine was substituted by the assumption that soil frost occurred when the simulated soil temperature in the upper soil layer was below 0°C. The HBV routine includes a routine for calculation of the depth of soil frost (Lindström *et al.*, 1996), and in the calculations, soil frost is assumed to occur as soon as the calculated soil temperature is below zero.

Sites used for model calibration and validation

Three sites (Figure 4) with varying physiographic and climatological conditions were used for model calibration and validation. At two of the sites, Lanna and Näsbygård, the models were calibrated against dissolved and particulate P in tile drain flow and at Näsbygård they were also validated. At the third site, Hedemora, surface runoff and sediment transport were modelled with the HBV-P and ICECREAM models, whereas, dissolved and PP in surface runoff were modelled with the original routines of the ICECREAM model. At all sites monitoring was carried out by the Division of Water Quality Management at the Swedish University of Agricultural Sciences (SLU).

The Lanna site (Table 1) has a soil characterised by macropore flow classified as *Uderitic Haploboroll*. The field is divided into seven plots which each have separate system tiles with 13.5 m spacing. With regard to soil characteristics the field is very heterogeneous and the soil parameters vary considerably over small distances and the models were set up for the plot that was determined to have average leaching. The field is very flat and normally no surface run-off occurs. The soil is permeable, and most of the water infiltrates the soil, where it percolates to the tile drains (Djodjic *et al.*, 2000). Climatological data were obtained from the Swedish Meteorological and Hydrological Institute (SMHI), from a station in Längjum, situated 15 km from the field.

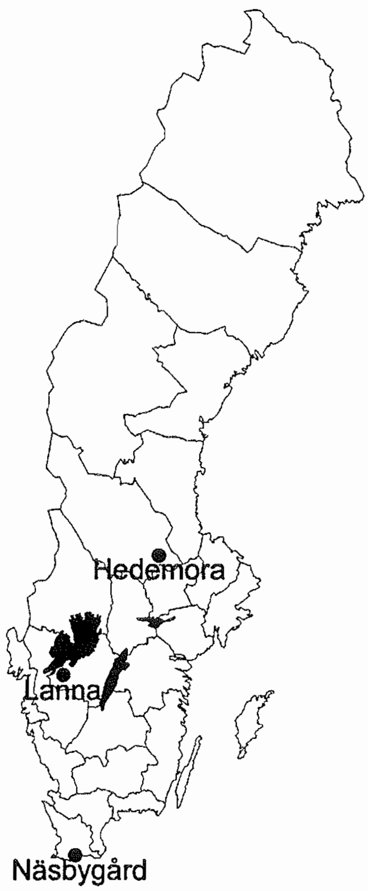


Figure 4. Map of Sweden with the location of the 3 sites used marked with black circles.

Monitoring of nutrient flow from the field at Näsbygård (Table 1) was initiated with the aim of studying long-term trends in nutrient transport (Brink *et al.*, 1978; Gustafson, 1987). The monitored drainage water is a mixture of direct percolation to the tile drain system and short-distance surface run-off that infiltrates the soil. Temperature and cloudiness data were obtained from a SMHI station situated 10 km from the field, and precipitation data from a SMHI station 6 km from the field. The water chemical sampling was done every fortnight. This may have underestimated the P loss, in comparison to newly installed flow proportional sampling. The use of sampling strategies with sampling at a certain time interval have been shown to underestimate P loss by more than 50% compared to flow proportional sampling (Grant *et al.*, 1996; Ulén and Persson, 1999).

The Hedemora experimental research plot was selected with the aim of studying surface run-off and erosion. A substantial part of the water flow consists of surface run-off. For Swedish conditions, the plot has a significant soil erosion rate, due to a combination of erosion sensitive soils and steep slopes (Table 1). The plot is divided into eight smaller plots with different tilling treatments. The plot used in this study was autumn tilled, i.e. the most common practice in this region (Persson, 2000). For the period 1994-1997, climatological data were available from a SMHI station, situated 7 km from the research plot, whereas for 1998-1999, climatological data from another station, situated further away from the research plot had to be used, since the first station had been closed down.

Table 1. Locations and characteristics of sites used for model calibration and validation

	Lanna	Näsbygård	Hedemora
Location	58°N 13°E	55°N 13°E	60°N 16°E
Time period	1993-1998	1974-1996	1994-1999
Drainage system	System drained Experimental plot	System drained Experimental field	Surface run-off Experimental plot
Area	0.4 ha	38 ha	0.01 ha
Dominant soil type	Clay	Loam	Silt
Average slope (%)	<1	3.87	10
Average precipitation (mm year^{-1})	612 (556)*	692 (556)*	610 (617)*
Average temperature (°C)	5.4 (5.9)*	7.7 (7.1)*	5.8 (4.2)*
Calculated annual average of soil frost days	74 (HBV-P) 83 (ICECREAM)	50 (HBV-P) 60 (ICECREAM)	122 (ICECREAM)
Mode of water sampling	Flow proportional and frequent	Manually every 14 day	Weekly partly accumulated
Type of water flow	Tile drain flow	Tile drain flow	Surface run-off
Water flow measurement	Pump	V-notch and level recording	Tilting vessels
Time resolution	Hour	Hour	Week
Average Q/P	0.35**	0.35**	0.15***

* = 30 years (1961-1990) average from nearby SMHI weather station. ** = Q in tile drains. *** = Q in surface run-off

The P concentrations between measurements were linearly interpolated and the transport was then calculated by multiplication with water flow, that were measured on an hourly basis. This means that any changes in P concentration between the measurements are not reflected in the calculated transport, which can underestimate the P transport.

Model calibration and validation

ICECREAM was calibrated manually in two steps: first the hydrological parts of the model were set up, then the P parts. In the first step, simulated percolation from the lowest soil layer at Lanna and Näsbygård was manually calibrated against monitored discharge from the tile drains. All parameters in the hydrological sub-models can theoretically be estimated in the field. Such field estimates were available for the Lanna site. However, model fits were significantly improved after calibration of the parameters shown in Table 2. The calibration was based on measurements of physical soil properties and literature values that were manually adjusted to obtain the best possible fit between monitored and modelled water flow. The main parameters to be adjusted were hydraulic conductivity (Rc) and field capacity (Fc). Experimentally estimated values of the parameters and calibrated estimates are shown in Table 2. Finally, the P component of the model was calibrated. This was done by adjusting the initial pools of soil P (Table 3). Measurements of initial pools were not available since the model uses resin extractable P and measurements in Sweden are usually based on ammonium lactate. Work is at present being done to enable conversions to be made between these estimates. For surface run-off and P transport with surface run-off, standard parameter values based on type of soil and vegetation etc. were used, i.e. there was no calibration. The P-routine in ICECREAM was not recalibrated when the hydroclimatological routines were substituted with the ones in HBV-P.

Simulations with PARTLE were driven by ICECREAM model estimates of daily percolation and daily PO_4-P concentrations in the surface soil layer. The model is sensitive to the correction factor for the soil's macropore hydraulic conductivity and matrix hydraulic conductivity. This factor was determined by adjusting both the hydraulic conductivities and minimising the error against measured PP drainage discharge. This was calibrated when PARTLE was driven by the

ICECREAM's hydrological component, i.e. PARTLE was not re-calibrated when the hydrological components of ICECREAM were substituted by those in HBV-P.

Table 2. Calibrated parameter values describing soil water characteristics in ICECREAM. Rc = saturated hydraulic conductivity, Fc = field capacity, Solpor = soil porosivity, Br15 = soil water content at wilting point. Measured values in brackets

Parameter	Lanna	Hedemora	Näsbygård
Rc (mm/h)	8 (32)	1	1.5
Fc (m^3/m^3)	0.23 (0.26)	0.36	0.22
Solpor (m^3/m^3)	0.48 (0.47)	0.47	0.27
Br15 (m^3/m^3)	0.10 (0.17)	0.20	0.09

Table 3. Initial P pools (kg ha^{-1}). Forg P = Fresh organic P, Lab P = P in equilibrium with soil water, Org P = organic bound P and Soil P = mineral bound P

	Lanna	Hedemora	Näsbygård
Forg P	0.0001	0.0005	0.0005
Lab P	0.01	0.01	0.01
Org P	0.10	0.295	0.045
Soil P	0.08	0.049	0.05

For the HBV-P model, estimates of run-off generation (Q) from the HBV-P model were calibrated against available records of discharge from tile drains in Lanna and Näsbygård. For Hedemora it was calibrated against river discharge in the nearby Mässingsboån (58 km^2). All model calibrations were made using an automatic calibration routine (Lindström, 1997). The field capacity in HBV-P corresponds to the plant-available water, i.e. water content between field capacity and wilting point. The value of this parameter was set according to the estimates from the calibration of ICECREAM, i.e. to Fc-Br15 and this value was not adjusted by further calibration (Table 2). Threshold temperatures (TT) and the melting factor (CFMAX) in the snow routine, the soil parameter BETA, and the response function parameters ALFA and KHQ were mainly calibrated (Figure 3). Parameter values used are shown in Table 4.

Table 4. Calibrated parameter values used in the HBV-P model. For explanation se text.

Parameter	Lanna	Hedemora	Näsbygård
TT	-0.29	-0.01	0.80
CFMAX	3.88	3.27	6.12
BETA	3.73	2.17	3.22
ALFA	0.13	0.50	0.86
KHQ	0.19	0.05	0.18

Surface run-off was calculated, without calibration, with the ICECREAM model, using curve numbers representing existing soil and vegetation conditions (CN=78 for unfrozen soil before harvesting, CN=86 for unfrozen soil after harvesting, CN=98 for harvested, frozen soil). Calculations of surface run-off with the HBV-P model was based on CN=84 for unfrozen soil, and CN=98 for frozen soil.

Näsbygård was the only one of the three research sites to have time series of P concentration and water flow from the tile drains of sufficient length to include both a calibration and a validation period. In addition, surface run-off of P, simulated with ICECREAM for Hedemora was not based on calibration, but used literature estimates of parameters.

Results

Simulation of water flow

It should be noticed that for HBV-P, it is the modelled streamflow generation (Q), and not the recharge from the soil moisture zone (R in Figure 3) that was used for model calibration, and also as input to the ICECREAM model. However, over longer time periods this does not affect the total volumes of water; its effect is on the temporal distribution of the water flow, with a dampening of the marked peaks of simulated percolation obtained with the HBV-P model (Figure 3).

Table 5. The precipitation (P), measured tiledrain flow (Q) and the ratio of Q to P, during agrohydrological years. The average is calculated from the complete agrohydrological years in the measurement periods.

Year*	Precipitation (P) (mm)	Drainage water flow (Q) Measured (mm)	HBV-P (mm)	ICECREAM (mm)	Q/P
Lanna					
1993	202	90	99	144	0.45
1993/94	581	223	212	141	0.38
1994/95	756	340	347	349	0.45
1995/96	478	127	146	176	0.27
1996/97	588	188	218	195	0.32
1997/98	623	214	207	181	0.34
1998	445	289	183	145	0.65
Average	605	218	226	208	0.35
Näsbygård calibration period					
1990	246	44	60	53	0.18
1990/91	756	185	207	179	0.24
1991/92	506	204	171	125	0.40
1992/93	679	173	234	290	0.26
1993/94	986	450	442	386	0.46
1994/95	853	321	338	334	0.38
1995/96	499	59	108	42	0.12
1996	304	21	54	42	0.07
Average	713	232	250	226	0.33
Näsbygård validation period					
1974	252	118	97	56	0.47
1974/75	752	308	260	341	0.41
1975/76	696	16	97	125	0.02
1796/77	640	232	229	125	0.36
1977/78	570	191	199	148	0.33
1978/79	660	220	240	160	0.33
1979/80	662	170	212	148	0.26
1980/81	874	519	374	337	0.59
1981/82	804	376	317	239	0.47
1982/83	731	275	206	272	0.38
1983/84	719	197	249	229	0.27
1984/85	616	265	229	140	0.43
1985/86	717	291	286	182	0.41
1986/87	692	242	191	130	0.35
1987/88	724	429	301	191	0.59
1988/89	657	250	130	173	0.38
1989	376	4	66	62	0.01
Average	701	265	235	196	0.38

* Agrohydrological year, i.e. 1 July - 30 June.

The ratios of drainage to precipitation were 0.35 for Lanna (Table 5) and 0.33 and 0.38 for Näsbygård during the calibration and the validation period, respectively, which is within the interval given by SMHI of 0.3-0.4 for the period 1961-1990 for those areas (Brandt and Grahn, 1998). The ratio of surface run-off to precipitation in Hedemora was 0.15 (Table 6).

Table 6. The precipitation (P), measured surface run-off (Q) and the ratio of Q to P, during the winter period, October to May. The average is calculated from the complete agrohydrological years in the measurement periods.

Year	Precipitation (P) (mm)	Surface water flow (Q) Measured (mm)	HBV-P (mm)	ICECREAM (mm)	Q/P
94/95	494	54	-	74	0.11
95/96	228	10	-	17	0.05
96/97	358	28	-	42	0.08
97/98	383	31	-	64	0.08
98/99	448	185	-	82	0.41
Average	382	62	-	56	0.15

The cumulated flow volumes over agrohydrological years (i.e 1/7 - 30/6 the following year) are shown in Figure 5. Total accumulated flow volumes with HBV-P were higher than those obtained with ICECREAM. For the validation period at Näsbygård, the flow volume estimated with ICECREAM was significantly underestimated (Table 7). Pearson correlation coefficients were higher for the HBV-P model during the calibration periods, and significantly higher during the validation period in Näsbygård compared to the correlation obtained with ICECREAM (Table 7). However, ICECREAM performed better than HBV-P in simulation of 1996 and 1997 spring peaks at Lanna.

Table 7. Performance of the HBV-P and ICECREAM models compared to measurements of discharge.

	Deviation from measured (%)		Pearson correlation (2-tailed)	
	HBV-P	ICECREAM	HBV-P	ICECREAM
Lanna, calibration (1993-1998)	-4	-9	0.64	0.59
Näsbygård, calibration (1990-1997)	+11	0	0.80	0.67
Näsbygård, validation (1974-1989)	-10	-25	0.78	0.57

No calibration was used with either ICECREAM or HBV-P surface run-off estimates. Calculations were based on standard values of curve numbers selected from soil and vegetation characteristics. Since for the HBV-P model no deviation was made between curve numbers before, and after harvesting, as long as the soil not was frozen, selected curve numbers for the HBV-P application were higher than for the ICECREAM application (see the model calibration section). Surface run-off could only be validated against accumulated surface run-off data from five time intervals at Hedemora (Figure 6a). When correlating the amount of collected surface run-off during the 5 periods with simulated volumes, a correlation coefficient of 0.96 was obtained for HBV-P and 0.90 with ICECREAM. In Table 8 the correlation between measured and simulated sediment and PP transport are shown.

Table 8. Correlation's between monitored and modelled sediment and particulate P transport; * significant ($P<0.10$) and ** $P<0.05$ respectively

	Sediment transport Measured	Sediment transport ICECREAM	Sediment transport HBV-P	PP transport ICECREAM
Sediment transport measured	-	0.76	0.96**	0.67
PP transport measured	0.90*	0.70	0.84*	0.74

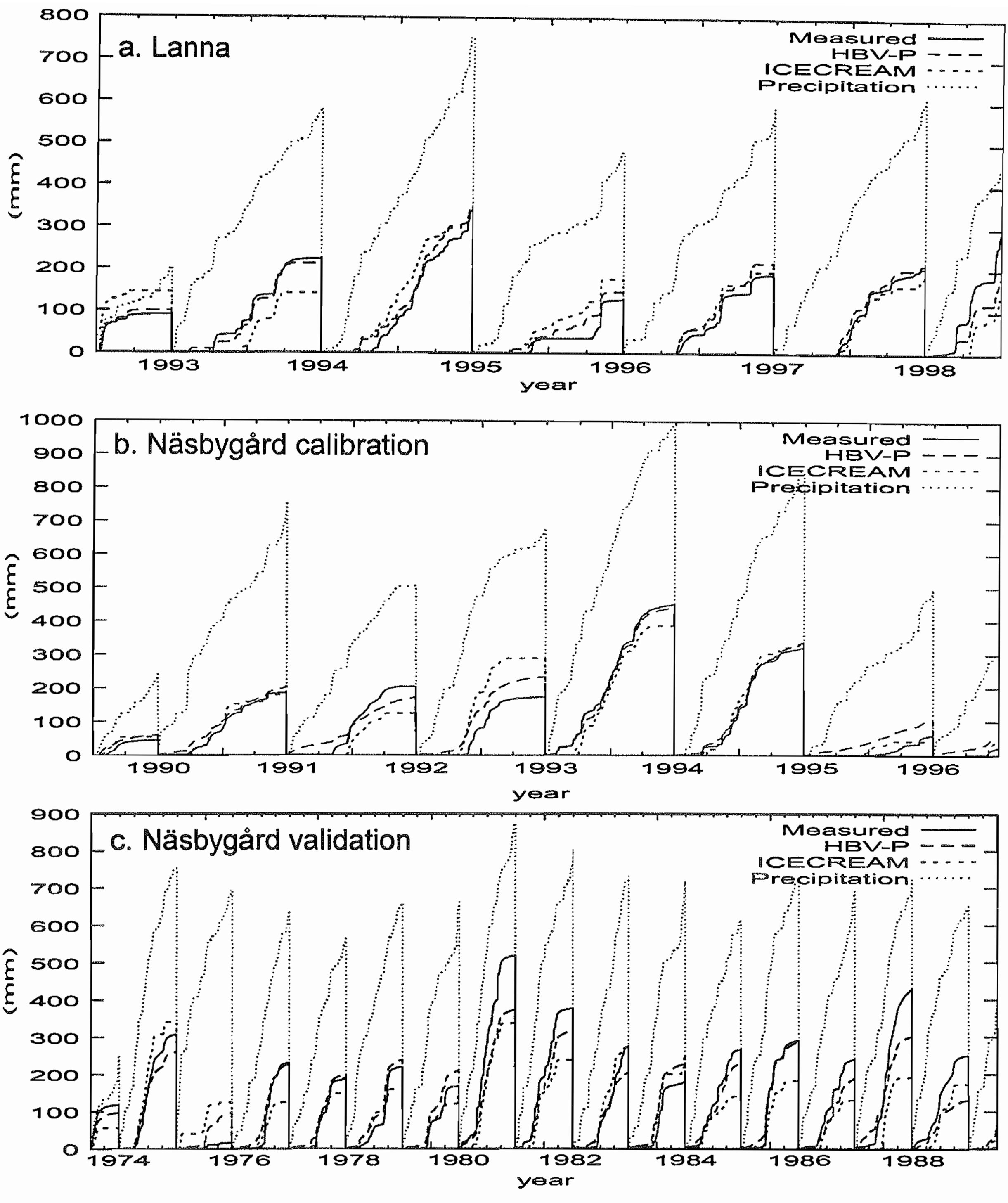


Figure 5. Precipitation (mm) and measured and simulated discharge in agrohydrological years a) Lanna, calibration period b) Näsbygård, calibration period c) Näsbygård, validation period

Simulation of sediment and P transport with surface run-off

The samples of surface run-off used for estimates of total volumes and for chemical analyses of concentrations of suspended solids, and dissolved and particulate P were monitored at irregular intervals. For one period (95/02/24 – 1995/03/06), the concentrations of sediments (17788 mg/l) were extremely high, combined with a monitored run-off of 15 mm, giving a sediment load for this period that corresponded to 74% of the total load between 1994 and 1999, and a loss of 2668 kg ha^{-1} for this 11 days period. This period was excluded for the calibration of the sediment routine in the HBV-P model.

Neither the non-calibrated sediment routine in ICECREAM nor the calibrated routine in the HBV-P made any estimates of the magnitude of the monitored estimate during the excluded 11-day period, for which HBV-P estimated a loss of 18 kg ha^{-1}, and ICECREAM a loss of 0,08 kg ha^{-1}. In Figure 6b, this period is excluded. For the other periods, HBV-P estimates of sediment transport had a very high correlation with the amounts monitored, whereas ICECREAM significantly underestimated the sediment transport (Figure 6b and Table 8).

A significant correlation was found between measured sediment transport and particulate P transport, and also between particulate P transport and sediment transport with HBV-P. However, since no routines for enrichment have been added to the HBV-P model, estimates of particulate P transport based on the HBV-P model were not made. ICECREAM underestimated the loss of PP with surface run-off, apparently due to the underestimation of the erosion.

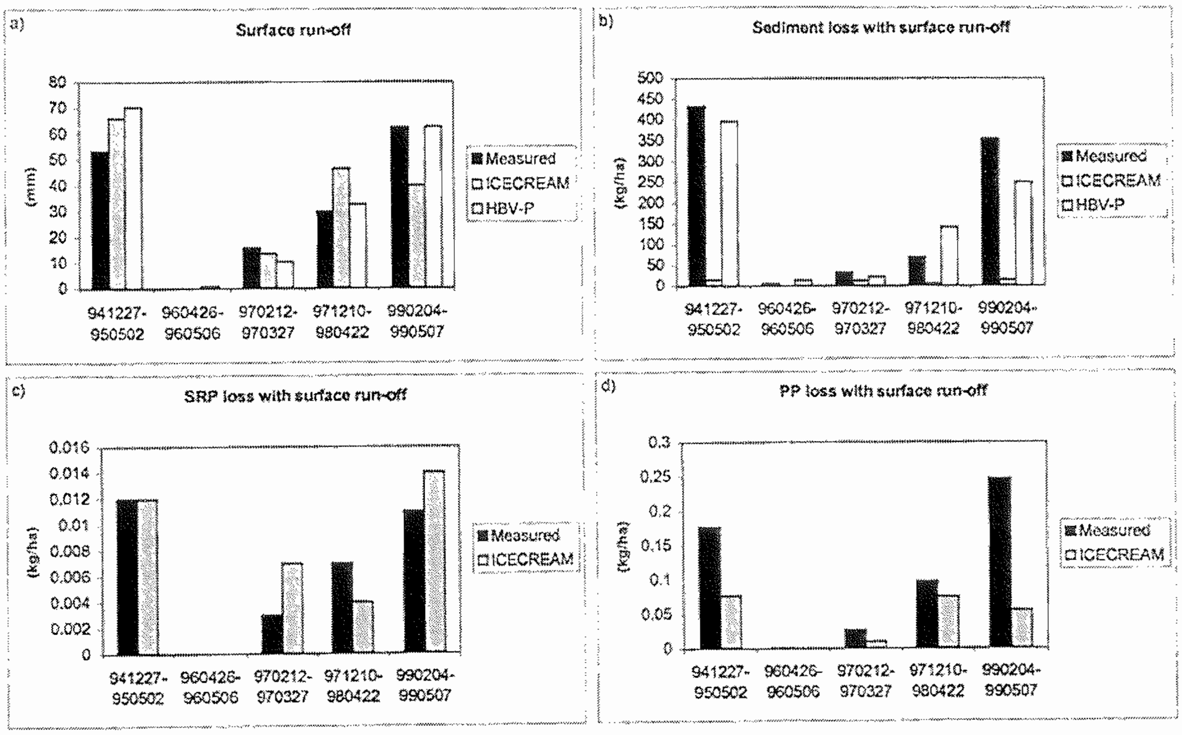


Figure 6. Hedemora experimental plot, five episodes during the period 1994-99. a) The precipitation and surface run-off, measured and simulated with ICECREAM and HBV-P. b) Soil loss with surface run-off c) The soluble phosphorus transported with surface run-off. d) Particle phosphorus transported with surface run-off.

Dissolved P in surface run-off was only estimated with the ICECREAM model (Figure 6c), obtaining a correlation of 0.87 ($p = 0.052$). For monitored P in surface run-off during the five time intervals, 0-10% consisted of dissolved P, compared to 5-41% for the ICECREAM modelled P in surface run-off.

Simulation of dissolved P leaching via tile drains

Both models overestimated the soluble reactive P (SRP) loss in Lanna by 0.06 kg ha^{-1}, which was probably caused by too large initial P pools in the soil. Both models clearly overestimated the loss in the first period. However, in the agrohydrological year 1993/94 both models missed the large loss that occurred mainly during 3 episodes (Figure 7a). All three episodes were triggered by rain for several days and on the two later occasions the soil was frozen and covered by melting snow. The SRP losses during the agrohydrological years 1994/95 and 1997/98 were both largely overestimated.

During the calibration period for Näsbygård (1990-1996) both models performed well, with a total error of 0.04 kg ha^{-1} for ICECREAM and 0.03 kg ha^{-1} for HBV-P (Figure 7b). However, in the validation period ICECREAM underestimated the loss by 0.67 kg ha^{-1} and HBV-P by 0.54 kg ha^{-1}. During 7 agrohydrological years (Figure 7c) the loss of SRP was severely underestimated. Most of the large episodes during these periods were caused by heavy rain for several days and / or snowmelt on frozen soil. The SRP loss was overestimated for the agrohydrological year 1975/76, but that can be explained by infrequent sampling, once every month instead of fortnightly, that missed the short episodes.

The SRP loss was very episodic and 17.8% to 55.8% of the total losses of SRP occurred in episodes greater than 0.002 kg ha^{-1} (Table 9). Both models had considerable problems with this. The SRP losses from HBV-P were better correlated with the measured values than the values from ICECREAM (Table 10).

Table 9. Phosphorus loss from episodes with peak values higher than 0.002 kg ha^{-1} and percentage of total loss. Measured and simulated values for Lanna and Näsbygård

	Lanna		Näsbygård 1990-97		Näsbygård 1974-1989	
	P (kg ha^{-1})	%	P (kg ha^{-1})	%	P (kg ha^{-1})	%
Soluble P loss						
Measurement	0.108	55.8	0.053	17.8	0.523	40.1
HBV-P	0.003	1.0	0.000	0.0	0.017	2.2
ICECREAM	0.035	13.9	0.000	0.0	0.000	0.0
Particulate P loss						
Measurement	0.203	62.4	0.080	31.3	0.115	39.7
HBV-P	0.057	19.5	0.002	0.8	0.069	18.1
ICECREAM	0.063	26.6	0.011	3.9	0.066	25.7

Table 10. Performance of the HBV-P and ICECREAM models compared to measurements of SRP and PP losses

	SRP				PP			
	Difference %		Pearson		Difference %		Pearson	
	HBV-P	ICE.	HBV-P	ICE.	HBV-P	ICE.	HBV-P	ICE.
Lanna, calibration (1993-1998)	-33	-30	0.34	0.29	-10	-27	0.25	0.35
Näsbygård, calibration (1990-1997)	+10	+1	0.71	0.59	-1	+10	0.48	0.44
Näsbygård, validation (1974-1989)	-41	-52	0.66	0.43	+31*	-11*	0.21*	0.11*

* = 1987-89.

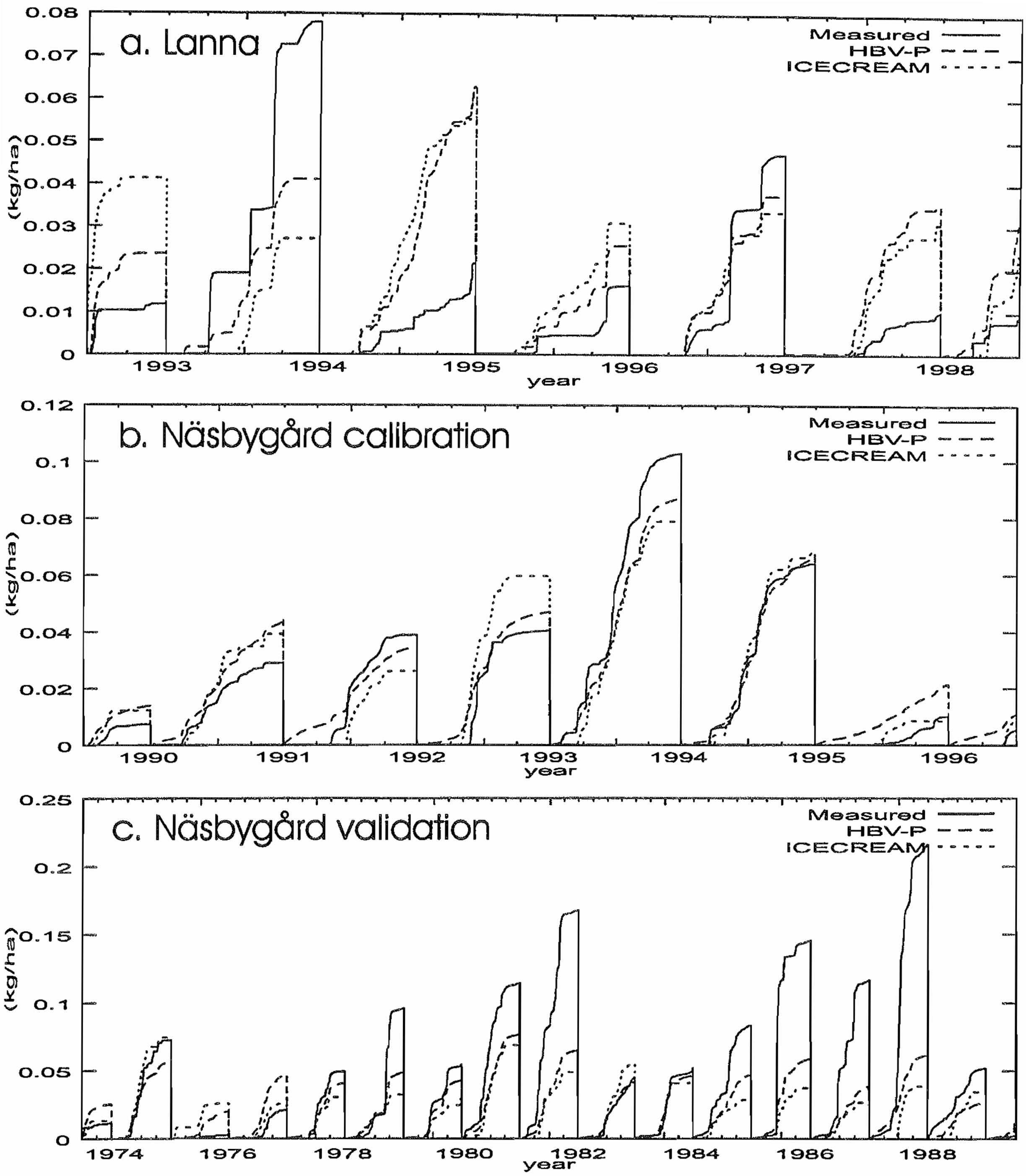


Figure 7. Losses of dissolved P, measured and simulated, in agrohydrological years: a) Lanna calibration period, b) Näsbygård calibration period, c) Näsbygård validation period.

Simulation of PP losses via tile drainage

The transported amount of PP from Lanna was relatively well described by PARTLE in both HBV-P and ICECREAM (Figure 8a) but the correlation between measured and simulated amounts was low (Table 10) meaning that the dynamics was poorly described. HBV-P underestimated the PP loss over the whole period by 0.03 kg ha^{-1} and ICECREAM by 0.09 kg ha^{-1}. However, the agrohydrological year 1996/97 was greatly underestimated with only half of the measured loss modelled.

At Näsbygård (Figure 8b) during the calibration period, HBV-P underestimated the PP loss by 0.002 kg ha^{-1} while ICECREAM overestimated it by 0.025 kg ha^{-1}. Both models overestimated PP losses the agrohydrological year 1992/93.

It was not possible to simulate PP losses during most of the validation period at Näsbygård due to lack of measurements of suspended solids in the drainage water (Figure 8c). However, the loss of PP was measured. The very high loss for 1981/82 was the result of a single run-off event lasting for 10 days, caused by simultaneous rain and snowmelt on thawing soil. That single event transported as much PP as the total amount during the preceding 3 years.

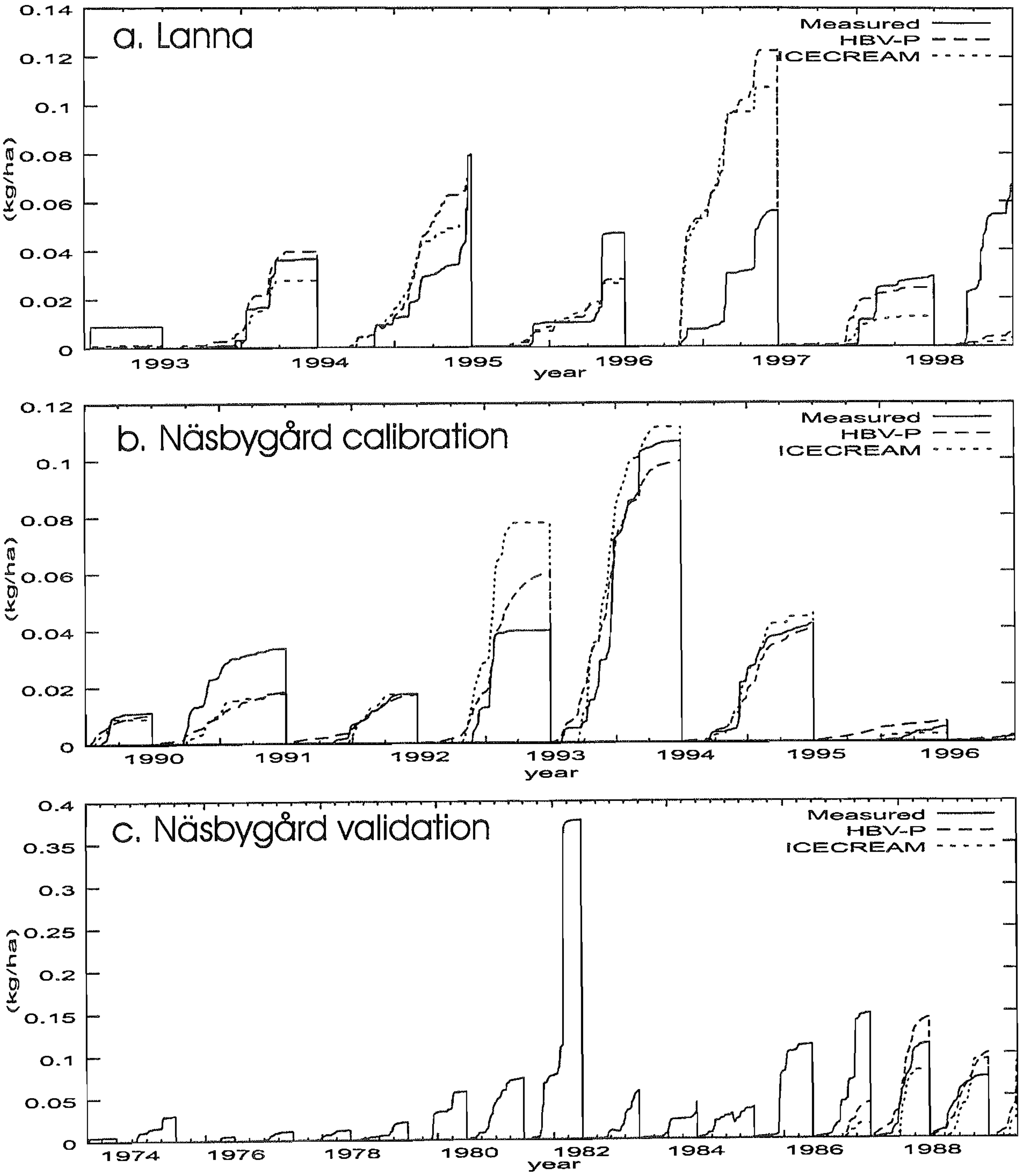


Figure 8. Losses of particulate P, measured and simulated, in agrohydrological years: a) Lanna calibration period, b) Näsbygård calibration period, c) Näsbygård validation period.

The loss of PP was even more episodic than the loss of SRP and up to 62.4% was lost during episodes larger than 0.002 kg ha^{-1}. PARTLE managed to produce up to 26.6% of the total PP losses during such peaks (Table 9). The correlation between measured and simulated losses was much lower than for the loss of SRP with drainage water (Table 10).

Discussion

A big advantage of the HBV-P model compared to the ICECREAM model is the possibility of using an automatic calibration process for the water flow, which simplifies the calibration enormously. We hope that on-going work on finding a conversion equation between lactate-extracted P and resin-extracted P will be completed soon, thereby much simplifying the calibration of the initial P pools.

Both surface run-off models worked satisfactory, although the simpler HBV-P model actually gave slightly better results. The surface run-off erosion simulated with ICECREAM was much to low which shows that some model calibration might be necessary. The regression line between measured erosion and measured PP loss had almost the same slope as between simulated erosion and simulated PP loss indicating that ICECREAM would have simulated the correct amount of lost PP if the simulated sediment loss had been on the right magnitude.

The lack of episodic SRP losses is due to the design of the P sub-model in ICECREAM. The P sub-model results in a constant P concentration in drainage water (Figure 9) and the amounts are regulated by the amount of drainage of water instead. In the field the concentration is often higher during high flows, accenting the episodic nature of P losses. It is probably possible to correct this without making too large a change to the model. One way of doing it could be to change the way transportation of dissolved P through the soil is simulated. At present the dissolved P is added to the labile P in the underlying layer: instead the dissolved P could be left in the dissolved pool and slowly equilibrated with the labile P, giving it a chance to be transported further before it is bound to the soil. Another problem is the lack of fast preferential flow that has to be simulated. If the model could handle preferential flow it would probably not miss the large P loss episodes occurring during snowmelt and prolonged rain. The use of PARTLE to compensate for the lack of preferential flow is unsatisfactory. In order to simulate the loss of PP with PARTLE the yearly mean value of suspended solids in the drainage water must be known: that makes it impossible to simulate on a drainage basin scale or in fields where suspended solids have not been monitored. Furthermore, PARTLE can not calculate the loss of dissolved P through macropore flow, which is as important as PP losses. The only solution to this is to incorporate a completely new way of handling preferential flow.

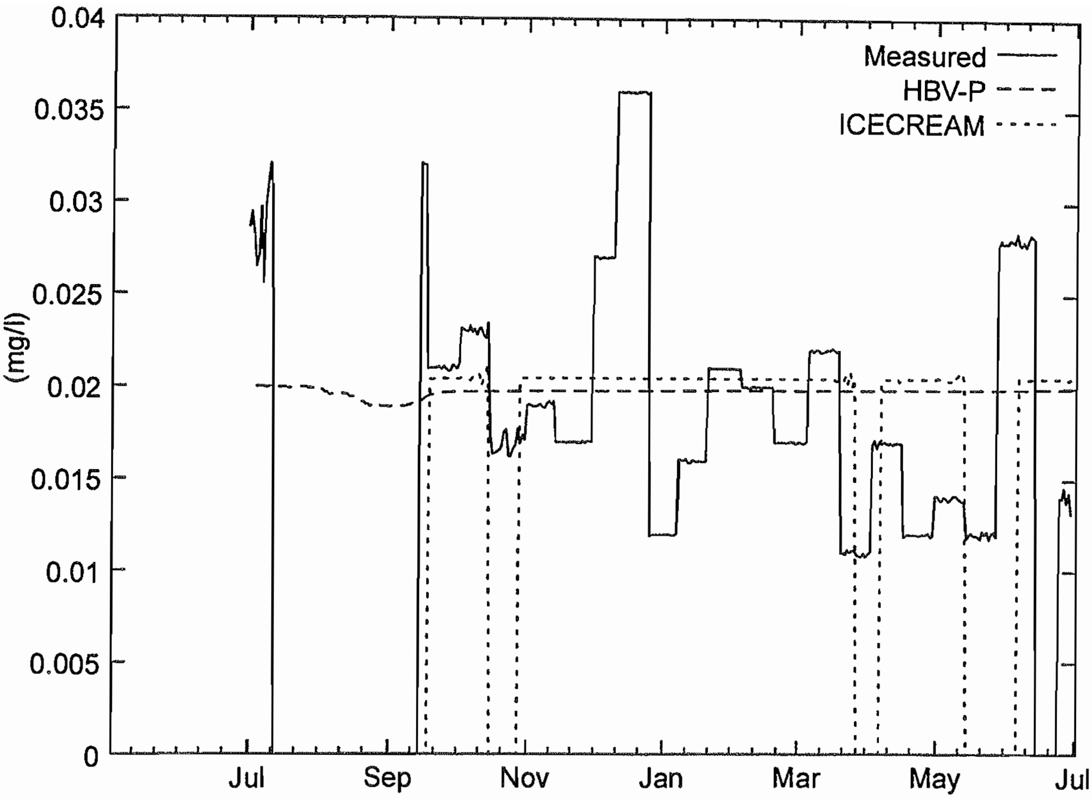


Figure 9. The concentration of soluble P in the drainage water during the agrohydrological year 1994/95 at Näsbygård.

The P sub-model of ICECREAM can only handle inorganic dissolved P, but when looking at the measurements in the drainage from Lanna it is clear that a large part (21%) of the lost P is not in the form of SRP or PP. As only TP, SRP and PP were measured, it is impossible to say what form it was in, but it could have been dissolved organic P (DOP).

Conclusions

(I) Substituting hydrological routines with more simple routines available in the hydrological catchment model HBV, which is adopted for Nordic conditions, will not deteriorate model performance.

(II) The new combined model based on HBV and the P sub-model from ICECREAM works at least as well as the ICECREAM model for calculating phosphorus losses from small fields.

(III) The P model must handle preferential flow in order to be able to satisfactorily simulate the P loss from arable land. Furthermore, the model needs to be modified so that it generates drainage water with shifting P concentration, as have been measured at the experimental plots. PARTLE has a substantial limitation in its use on a catchment scale since it depends on input of annual losses of sediment through the drains, which means that a model of sediment losses through drains needs to be added to the model, alternatively some standard values have to be applied, or alternatively other models should be included into the catchment model

(IV) To be able to use the model on larger areas such as river sub-basins, access to a good conversion equation is necessary to convert the phosphorus measurements routinely carried out in Sweden to the resin-extractable soil P that is used by the P sub-model.

References

Arheimer, B. and M. Brandt (2000). "Watershed modelling of non-point nitrogen pollution from arable land to the Swedish coast in 1985 and 1994" *Ecological Engineering* 14: 389-404.

Bergström, S. (1995). "The HBV model", in V. P. Singh (Eds) *Computer Models of Watershed Hydrology*. Water Resources Publications, Highlands Ranch, Colorado, 443-476.

Bilaletdin, Ä., K. Kallio, T. Frisk, B. Vehviläinen, M. Huttunen and J. Roos (1994). "A modification of the HBV model for assessing phosphorus transport from a drainage area" *Wat. Sci. Tech.* 30(7): 179-182.

Brandt, M. and G. Grahn (1998)."Avdunstning och avrinningskoeficient i Sverige 1961-1990 Beräkningar med HBV-modellen". SMHI, Hydrologi 73.

Brink, N., A. Gustafson and G. Persson (1978)."Förluster av växtnäring från åker. Losses of nutrients from arable land". Water Quality Management, Swedish University of Agricultural Sciences, Ekohydrologi 1.

Djodjic, F., B. Ulén and L. Bergström (2000). "Temporal and spatial variations of phosphorus losses and drainage in a structured clay soil" *Water Research* 34(5): 1687-1695.

Grant, R., A. Laubel, B. Kronvang, H. E. Andersen, L. M. Svendsen and A. Fuglsang (1996). "Loss of dissolved and particulate phosphorus from arable catchments by subsurface drainage" *Wat. Res.* 30(11): 2633-2642.

Gustafson, A. (1987). "Simulation of water discharge rates from a clay-till soil over a ten year period" *Journal of hydrology* 92: 263-274.

Johnsson, H., L. Bergström and P.-E. Janson (1987). "Simulated nitrogen dynamics and losses in a layered agricultural soil" *Agric. Ecosyste. Environ* 18: 333-356.

Knisel, W. G. (1980)."CREAMS: a field-scale model for chemicals, runoff and erosion from agricultural managment systems". USDA, Conservation Research Report 26.

Knisel, W. G. (1993)."GLEAMS: Groundwater loading effect of agricultural management systems, version 2.1". UGA-CPES-BAED University of Georgia, Publ. No. 5.

Lindsay, W. L. (1979). "Chemical equilibria in soils". Chichester, John Wiley & Sons Ltd.

Lindström, G. (1997). "A Simple Automatic Calibration Routine for the HBV Model" *Nordic Hydrology* 28(3): 153-168.

Lindström, G., M. Gardelin, B. Johannson, M. Persson and S. Bergström (1996)."HBV-96 - En areellt fördelad modell för vattenkrafthydrologin". SMHI, SMHI Report RH 12.

Persson, K. (2000)."Jordbearbetningens påverkan på fosforförlusterna från en mjälalättlera i södra Dalarna". Division of Waterquality Management, SLU, Ekohydrologi 52.

Posch, M. and S. Rekolainen (1993). "Erosivity factor in the Universal Soil Loss Equation estimated from Finnish rainfall data" *Agricultural Science In Finland* 2(4): 271-279.

Rekolainen, S., I. Grönroos, A. Bärlund, A. Nikander and Y. Laine (1998). "Modelling the impact of management practices on agricultural phosphorus losses to surface waters of Finland". 3rd International Conference on Diffuse Pollution, 33-40.

Rekolainen, S. and M. Posch (1993). "Adapting the CREAMS Model for Finnish Conditions" *Nordic Hydrology* 24: 309-322.

Ritchie, J. T. (1972). "A model for predicting evaporation for a row crop with incomplete cover" *Water Resources Research* 8(5): 1204-1213.

Ryden, J. C., J. K. Syers and D. Haris (1973). "Phosphorus in runoff and streams." *Advances in agronomy* 25: 1-45.

Schoumans, O. F. and W. J. Chardon (2000)."Modelling P losses by leaching in Europe". , Cost action 832. Working group 2: Phosphorus losses at field scale.

Schoumans, O. F. and P. Groenendijk (2000). "Modelling soil phosphorus levels and phosphorus leaching from agricultural land in the Netherlands" *Journal of environmental Quality* 29: 111-116.

Shirmohammadi, A., B. Ulen, L. F. Bergstöm and W. G. Knisel (1998). "Simulation of Nitrogen and Phosphorus Leaching in a Structured Soil Using Gleams and a New Submodel, "Partle"" *Transactions of the AEA E* 41(2): 353-360.

Stamm, C., H. Flühler, R. Gächter, J. Leuenberger and H. Wunderli (1998). "Preferential transport of phosphorus in drained grassland soils" *Journal of environmental quality* 27: 515-522.

Ulén, B. (1995). "Episodic Precipitation and Discharge Events and their Influence on Losses of Phosphorus and Nitrogen from Tile Drained Arable Fields." *Swedish Journal of Agricultural Research* 25: 25-31.

Ulén, B. and K. Persson (1999). "Field-scale phosphorus losses from a drained clay soil in Sweden" *Hydrological Processes* 13(17): 2801-2812.

Ulén, B., A. Shirmohammadi and F. Djodjic (1998). "The use of Gleams model to predict phosphorus losses from drained clay soils" *Vatten* 54: 53-60.

Williams, J. R., C. A. Jones and P. T. Dyke (1990). "The EPIC model", in A. N. Sharpley and J. R. Williams (Eds) *EPIC--Erosion/Productivity Impact Calculator 1. Model Documentation. U. S. Department of Agriculture, Agricultural Research Service, Technical Bulletin*, 3-92.

Wischmeier, W. H. and D. D. Smith (1978). "Predicting rainfall erosion losses: a guide to conservation planning", United States. Science and Education Administration. Indiana. Agricultural Experiment Station.

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5	1979	Gunnar Fryk och Sven-Åke Heinemo. Självrening av lakvatten från kompost på sand och mo. *Self-purification of leachate from compost on sand and fine sand.*
		Nils Brink. Växtnäringsförluster från skogsmark. *Losses of nutrients from forests.*
		Nils Brink. Utlakning av kväve från agroekosystem. *Leaching of nitrogen from agro-ecosystems.*
		Nils Brink. Ytvatten, grundvatten och vattenförsörjning.
6	1980	Arne Gustafson och Mats Hansson. Växtnäringsförluster i Skåne och Halland. *Losses of Nutrients in Skåne and Halland.*
		Nils Brink, Sven L. Jansson och Staffan Steineck. Utlakning efter spridning av potatisfruktsaft. *Leaching after spreading of potato juice.*
		Nils Brink och Arne Gustafson. Att spå om gödselkväve. *Forecasting the need of fertilizer nitrogen.*
		Arne Gustafson och Börje Lindén. Lantbruksuniversitetet satsar på exaktare kvävegödsling.
7	1980	Nils Brink och Börje Lindén. Vart tar handelsgödselkvävet vägen. *Where does the commercial fertilizer go.*
		Barbro Ulén och Nils Brink. Omgivningens betydelse för primärproduktionen i Vadsbrosjön. *The importance of the environment for the primary production in lake Vadsbrosjön.*
		Arne Gustafson. Jordbruket och grundvattnet.
		Nils Brink. Utlakning av växtnäring från åkermark.
		Nils Brink. Vart tar gödseln vägen.
8	1981	Nils Brink. Försurning av grundvatten på åker. *Acidification of groundwater on arable land.*

Rikard Jernlås och Per Klingspor. TCA-utlakning från åker. *Leaching of TCA from arable land.*

Arne Joelsson. Ytavspolning av fosfor från åkermark. *Storm washing of phosphorus from arable land.*

Arne Gustafson, Sven-Olof Ryding och Barbro Ulén. Kontroll av växtnäringsläckage från åker och skog. *Control of losses of nutrients from arable land and forest.*

9 1981 Barbro Ulén och Nils Brink. Miljöeffekter av ureaspridning och glykolanvändning på en flygplats. *Environmental effects of spreading of urea and use of glycol at an airport.*

Gunnar Fryk. Utlakning från upplag av malda sopor. *Leachate from piles of shredded refuse.*

10 1982 Arne Gustafson och Arne S. Gustavsson. Växtnäringsförluster i Västergötland och Östergötland. *Losses of nutrients in Västergötland and Östergötland.*

Barbro Ulén. Växtnäringsförluster från åker och skog i Södermanland. *Losses of nutrients from arable land and forests in Södermanland.*

Arne S. Gustavsson och Barbro Ulén. Nitrat, nitrit och pH i dricksvatten i Västergötland, Östergötland och Södermanland. *Nitrate, nitrite and pH in drinking water in Västergötland, Östergötland and Södermanland.*

Lennart Mattsson och Nils Brink. Gödslingsprognoser för kväve. *Ferilizer forecasts.*

11 1982 Barbro Ulén. Vadsbrosjöns närsaltbelastning och trofinivå. *The nutrient load and trophic level of lake Vadsbrosjön.*

Arne Andersson och Arne Gustafson. Metallhalter i dräneringsvatten från odlad mark. *Metal contents in drainage water from cultivated soils.*

11, forts. Arne Gustafson. Växtnäringsförluster från åkermark i Sverige.

Barbro Ulén. Erosion av fosfor från åker. *Erosion of phosphorus from arable land.*

Rikard Jernlås. Kväveutlakningens förändring vid reducerad gödsling.

12 1982 Nils Brink och Rikard Jernlås. Utlakning vid spridning höst och vår av flytgödsel. *Leaching after spreading of liquid manure in autumn and spring.*

Gunnar Fryk och Thord Ohlsson. Infiltration av lakvatten från malda sopor. *Leachate migration through soils.*

Nils Brink. Measurement of mass transport from arable land in Sweden.

Arne Gustafson. Leaching of nitrate from arable land in Sweden.

13 1983 Nils Brink, Arne S. Gustavsson och Barbro Ulén. Yttransport av växtnäring från stallgödslad åker. *Surface transport of plant nutrients from field spread with manure.*

Rikard Jernlås. TCA-utlakning på lerjord. *Leaching of TCA on a clay soil.*

Arne Gustafson och Gunnar Torstensson. Växtnäringsförluster vid Öjebyn. *Losses of nutrients at Öjebyn.*

Arne Gustafson och Gunnar Torstensson. Växtnäringsförluster vid Röbäcksdalen. *Losses of nutrients at Röbäcksdalen.*

Rikard Jernlås och Per Klingspor. Nitratutlakning och bevattning. *Drainage losses of nitrate and irrigation.*

14 1983 Arne Gustafson, Lars Bergström, Tomas Rydberg och Gunnar Torstensson. Kvävemineralisering vid plöjningsfri odling. *Nitrogen mineralization in connection with non-ploughing practices.*

Rikard Jernlås. Rörlighet och nedbrytning av fenvalerat i lerjord. *Decomposition and mobility of fenvalerate in a clay soil.*

Nils Brink. Jordprov på hösten eller våren för N-prognoser. *Soil sampling for nitrogen forecasts.*

Nils Brink. Närsalter och organiska ämnen från åker och skog. *Nutrients and organic matters from farmland and forest.*

Nils Brink. Gödselanvändningens miljöproblem.

15 1984 Nils Brink, Arne S. Gustavsson och Barbro Ulén. Växtnäringsförluster runt Ringsjön. *Nutrient losses in the Ringsjö area.*

Arne Gustafson och Gunnar Torstensson. Fånggröda efter korn. *Catch crop after barley.*

Arne Gustafson och Gunnar Torstensson. Växtnäringsförluster från åker i Nybroåns avrinningsområde. *Losses of nutrients from arable land in the Nybroån river basin.*

Arne Gustafson och Gunnar Torstensson. Växtnäringsförluster i Vagle. *Losses of nutrients at Vagle.*

Arne Gustafson och Gunnar Torstensson. Växtnäringsförluster i Offer. *Losses of nutrients at Offer.*

16 1984 Arne Gustafson, Arne S. Gustavsson och Gunnar Torstensson. Intensitet och varaktighet hos avrinning från åkermark. *Intensity and duration of drainage discharge from arable land.*

17 1984 Jenny Kreuger och Nils Brink. Fånggröda och delad giva vid potatisodling. *Catch crop and divided N-fertilizing when growing potatoes.*

Nils Brink och Arne Gustavsson. Förluster av växtnäring från sandjord. *Losses of nutrients from sandy soils.*

Arne Gustafson och Gunnar Torstensson. Växtnäringsförluster i Boda. *Losses of nutrients at Boda.*

Nils Brink. Vattenföroreningar från tippen i Erstorp - ett rättsfall.

18 1984 Barbro Ulén. Påverkan på yt- och dränerings- ocg grundvatten vid Ekenäs. *Influence on surface water, drainage water and groundwater at Ekenäs.*

Barbro Ulén. Nitrogen and Phosphorus to surface water from crop residues.

19 1985 Arne Gustavsson och Nils Brink. Förluster av kväve och fosfor runt Ringsjön. *Losses of nitrogen and phosphorus in the Ringsjö area.*

Nils Brink och Kjell Ivarsson. Förluster av växtnäring från lerjordar i Skåne. *Losses of nutrients from clay soils in Skåne.*

Arne Gustavsson, Berit Tomassen och Börje Wiksten. Växtnäringsförluster från åker på Uppsalaslätten. *Nutrient losses from arable land in the region of Uppsala.*

Christina Lindgren, Margaretha Wahlberg och Arne Gustavsson. Dricksvattenkvalitet i Uppsala regionen. *Drinking water quality in the region of Uppsala.*

Jenny Kreuger. Rörlighet hos MCPA och Diklorprop. *Mobility of MCPA and dichlorprop.*

Barbro Ulén. Ytavrinningsförluster av cyanazin. *Losses with surface run-off of cyanazine.*

20 1985 Jenny Kreuger. Rörlighet hos MCPA och diklorpropp på sandjord. *Mobility of MCPA and dichlorprop in a sandy soil.*

Kjell Ivarsson och Nils Brink. Utlakning från en grovmojord i Halland. *Losses of nutrients from a sandy soil in Halland.*

Barbro Ulén. Åkermarkens erosion. *Erosion of phosphorus from arable Land.*

Arne S. Gustavsson. Förluster av kväve och fosfor runt Ringsjön.

Arne Gustafson. Växtnäringsläckage och motåtgärder

Nils Brink. Bekämpningsmedel i åar och grundvatten.

21 1986 Birgit Loeper. Toxicitetstest för pesticider med protozoer. ***Toxicity test for pesticides using protozoa.***

Nils Brink, Arne Gustafson och Gunnar Torstensson. Odlingsåtgärders inverkan på kvalitet hos yt- och gundvatten.

Barbro Ulén. Lakning av fosfor ur jordar. *Leaching of phosphorus from soils.*

Nils Brink och Gunnar Torstensson. Vådan av proteingödsling. Värdera miljön. *Risk of fertilizing for increased protein. Evaluate the environment.*

Jenny Kreuger. Bekämpningsmedel. Utlakning från åkermark.

22 1987 Arne Gustafson. Water Discharge and Leaching of Nitrate.

23 1987 Lars Bergström. Transport and Transformations of Nitrogen in an Arable Soil

24 1987 Arne Gustafson och Gunnar Torstensson. Fånggröda efter skörd. *Catch crop after harvest.*

Arne Gustafson och Gunnar Torstensson. Läckage av växtnäring från åker i Nybroåns vattensystem. *Leaching of nutrients from arable land in the Nybroån river basin*

Solweig Ellström och Nils Brink. Stallgödslad och konstgödslad åker läcker växtnäring. *Fields spread with manure and fertilizer leach plant nutrients.*

Nils brink. Kväveläckage vid försök med nitrifikationshämmare.

Nils Brink. Kväve och fosfor från stallgödslad åker.

Nils Brink. Kväve och fosfor från konstgödslad åker.

25 1987 Nils Brink och Klaas van der Meulen. Losses of Phosphorus and Nitrogen to Lake Ringsjön.

Nils Brink. *Regional vattenundrsökning söder och öster om Ringsjön. Water nutrient status to the south and east of Lake Ringsjön.*

Petra Fagerholm. Vattenkvalitet och jordbruksdrift inom Ringsjöområdet. *Water Quality and agriculture in the area of Lake Ringsjön.*

Nils Brink. Nitrifikationshämmare eller svält mot kväveläckage. *Nitrification inhibitors or starvation against nitrogen losses.*

Nils Brink, Jenny Kreuger och Gunnar Torstensson. Näringsflöden från åkermark. *Nutrient fluxes from arable land.*

26 1988 Arne Andersson och Arne Gustafson. Deposition av spårelement med nederbörden. *Bulk deposition of trace elements in precipitation.*

Arne Andersson, Arne Gustafson och Gunnar Torstensson. Utlakning av spårelement från odlad jord. *Removel of trace elements from arable land by leaching.*

Barbro Ulén. Fosforerosion vid vallodling och skyddszon med gräs. *Phosphorus erosion under ley cropping and a grass protective zone.*
Gustafson, A. & Torstensson, G Växtnäringsläckage efter vallbrott.

Ellström, S. Avrinning och växtnäringstransport från åkermark.

27 1990 Lisbet Lewan. Insådd fånggröda: Effekter på utlakning av växtnäringsämnen. *Undersown Catch Crop - Effects on leaching of plant nutrients.*

Lisbet Lewan och Holger Johnsson. Insådd fånggröda: Effekter på utlakning av kväve. *Undersown catch crops - effects on leaching of nitrogen.*

Solweig Wall Ellström. Avrinning och växtnäringsförluster från JRK:s stationsnät på åkermark. *Discharge and nutrient losses from arable land.*

28 1992 Gunnar Torstensson, Arne Gustafson, Börje Lindén, och Gustav Skyggesson. Mineralkvävedynamik och växt näringsutlakning på en grovmojord med handels- och stallgödslade odlingssystem i södra Halland. *Mineral nitrogen dynamics and nutrient leaching in a sandy soil in southern Halland with cropping systems fertilized with commercial fertilizers and manure.*

29 1992 Barbro Ulén. Närsaltsförluster från mindre avrinningsområden inom jordbrukets recipientkontroll i Sverige. *Nutrient losses from small catchment areas in the recipient control of agriculture in Sweden.*

Markus Hoffman. Avrinning och växtnäringsförluster från JRK:s stationsnät agrohydrologiska året 90/91 samt långtids-översikt för 1977/90. *Discharge and nutrient losses from arable land in 1990/91 and review of the years 1977/90*

Markus Hoffman. Odlingsåtgärder och vattenkvalitet - en studie på sju fält i Malmöhus län. *Cultivation practices and water quality - a study on seven fields in Malmöhus county.*

30 1993 Börje Lindén, Arne Gustafson, Gunnar Torstensson och Erik Ekre. Mineralkvävedynamik och växtnäringsutlakning på en grovmojord i södra Halland med handels- och stallgödslade odlingssystem. *Mineral nitrogen dynamics and nutrient leaching in a sandy soil in southern Halland with cropping systems fertilized with commercial fertilizers and manure, and with or without ryegrass catchcrop.*

31 1993 Gunnar Torstensson, Arne Gustafson och Börje Lindén. Kväveutlakning på sandjord - motåtgärder med ny odlingsteknik. *Leaching of nitrogen from sandy soil - counter measures with new technique.*

32 1993 Markus Hoffman och Solweig Wall Ellström. Avrinning och växtnäringsförluster från JRK:s stationsnät för agrohydrologiska året 1991/92 samt långtidsöversikt. *Discharge and nutrient losses from arable land in 1991/92 and a long term review.*

33 1993 Börje Lindén, Helena Aronsson, Arne Gustafson och Gunnar Torstensson. Fånggrödor, direktsådd och delad kvävegivastudier av kväveverkan och utlakning i olika odlingssystem i ett lerjordsförsök i Västergötland. *Catch crops, direct drilling and split nitrogen fertilization - studies of nitrogen turnover and leaching in crop production systems on a clay soil in Västergötland.*

34 1993 Gunnar Torstensson, Arne Gustafson, Helena Aronsson och Artur Granstedt. Ekologisk odling - utlakningsrisker och kväveomsättning. Ecological Agriculture - Leaching risks and Nitrogen Turnover. *Ecological agriculture - leaching risks and nitrogen turnover.*

35 1993 Erik Kellner. Årstidsbunden kvävebelastning och denitrifikation i dammar - en enkel modellansats. *Seasonal nitrogen fluxes and denitrification in ponds - simple model approach*

36 1995 Markus Hoffmann och Solweig Wall Ellström. Avrinning och växtnäringsförluster från JRK:s stationsnät för agrohydrologiska året 1992/93 samt en långtidsöversikt. *Discharge and nutrient losses from arable land in 1992/93 and a long term review.*

37 1995 Katarina Kyllmar och Holger Johnsson. Växtnäringsförluster till vatten från ett jordbruksområde på Gotland 1989/94.

38 1995 Katarina Kyllmar, Göran Johansson och Markus Hoffmann. Avrinning och växtnäringsförluster från JRK:s stationsnät för agrohydrologiska året 1993/94 samt en långtidsöversikt. *Discharge and nutrient losses from arable land in 1993/94 and a long term review.*

39 1996 Holger Johnsson och Markus Hoffmann. Normalutlakning av kväve från svensk åkermark 1985 och 1994.

40 1996 Katarina Kyllmar och Holger Johnsson. Typområden på jordbruksmark (JRK). Avrinning och växtnäringsförluster för det agrohydrologiska året 1994/95.

41 1997 Bo Wejfeldt och Arne Gustafson. Utesuggor och kväveutlakning. Resultat från ett fältförsök i Halland.

42 1997 Katinka Hessel, Jenny Kreuger och Barbro Ulén. Kartläggning av bekämpningsmedelsrester i yt-, grund- och regnvatten i Sverige 1985-95. Resultat från monitoring och riktad provtagning.

43 1997 Göran Johansson och Katarina Kyllmar. Observationsfält på åkermark. Avrinning och växtnäringsförluster för det agrohydrologiska året 1994/95 samt en långtidsöversikt. *Discharge and nutrient losses from arable land in 1994/95 and a long term review.*

44 1998 Katarina Kyllmar och Holger Johansson. Växtnäringsförluster till vatten i Typområden på jordbruksmark (JRK) 1984-1995.

45 1998 Kristina Mårtensson och Katarina Kyllmar. Växtnäringsförluster till vatten från fyra jordbruksområden i Västra Götalands län 1993-97. Utvärdering av mätningar och inventeringar utförda inom miljöövervakningsprogrammet "Typområden på jordbruksmark" i Järnsbäckens, Öxnevallabäckens, Vikensbäckens och Forshällaåns avrinningsområden.

46 1998 Katinka Hessel, Helena Aronsson, Börje Lindén, Maria Stenberg, Tomas Rydberg och Arne Gustafson. Höstgrödor – Fånggrödor – Utlakning. Kvävedynamik och kväveutlakning på en moränlättlera i Skåne.

47 1998 Mårtensson Kristina och Katarina Kyllmar. Växtnäringsförluster till vatten från två jordbruksområden i Örebro län 1994-1997

48 1998 Kyllmar Katarina och Holger Johnsson. Typområden på jordbruksmark (JRK) Avrinning och växtnäringsförluster för det agrohydrologiska året 1995/96.

49 1999 Johansson Göran, Kyllmar Katarina och Johnsson Holger. Observationsfält på åkermark. Avrinning och växtnäringsförluster för det agrohydrologiska året 1995/96 samt en långtidsöversikt.

50 1999 Hessel Tjell Katinka, Aronsson Helena, Torstensson Gunnar, Gustafson Arne, Linden Börje, Stenberg Maria och Rydberg Tomas. Mineralkvävedynamik i handels- stallgödslade odlingssystem med och utan fånggröda. Resultat från en grovmojord i södra Halland, perioden 1990-1998.

51 1999 Lindén Börje, Engström Lena, Aronsson Helena, Hessel Tjell Katinka, Gustafson Arne, Stenberg Maria, Rydberg Tomas. Kvävemineralisering under olika årstider och utlakning på en mojord i Västergötland. Inverkan av jordbearbetningstider, flygödseltillförsel och insådd fånggröda.

52 2000 Persson Kristian, Jordbearbetningens påverkan på fosforförlusterna från en mjälalättlera i Södra Dalarna.
Ulen Barbro, Johansson Göran och Kyllmar Katarina. Fosforläckage från elva observationsfält under tjugoett år.
Ulen Barbro och Kreuger Jenny. Bekämpningsmedelsrester i vatten 1985-1999. Riktade provtagninggar och monitoring samlade i en databas.

53 2000 Kyllmar Katarina och Johnsson Holger. Typområden på jordbruksmark (JRK). Avrinning och växtnäringsförluster för de agrohydrologiska åren 1996/97 och 1997/98.

54 2000 Kreuger Jenny. Övervakning av bekämpningsmedel i vatten från ett avrinningsområde i Skåne. Årsredovisning för Vemmenhögsprojektet 1998 samt en kortfattad översikt.

55 2000 Carlsson Carina, Kyllmar Katarina och Johnsson Holger .Typområden på jordbruksmark (JRK) Avrinning och växtnäringsförluster för det agrohydrologiska året 1998/99.

56 2000 Torstensson Gunnar, Gustafson Arne, Bergström Lars och Ulén Barbro. Utredning om effekterna på kväveutlakning vid övergång till ekologisk odling

57 2001 Torstensson Gunnar och Håkansson Magnus. Kväveutlakning på sandjord - motåtgärder med ny odlingsteknik. Miljöanpassad stallgödselanvändning och odling i realistiska odlingssystem. Resultat från en grovmojord i Södra Halland, perioden 1991-1999.