



Diffuse Phosphorus Loss

Risk Assessment, Mitigation Options and Ecological Effects in River Basins

The 5th International Phosphorus Workshop (IPW5)

3-7 September 2007 in Silkeborg, Denmark

Goswin Heckrath, Gitte H. Rubæk and Brian Kronvang (eds.)



A A R H U S U N I V E R S I T E T

Faculty of Agricultural Sciences

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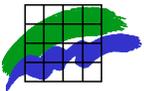
The 5th International Phosphorus Workshop (IPW5)

3-7 September 2007 in Silkeborg, Denmark

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Preface

The central role of diffuse phosphorus (P) losses in eutrophication of surface waters has long been recognized. Eutrophication impairs ecological quality and biodiversity of aquatic ecosystems, restricting the use of surface waters for drinking water abstraction and recreation. Diffuse P losses have thus become a major worldwide environmental concern. In Europe the Water Framework Directive (WFD) will oblige river basin authorities to oversee the improvement of ecological quality which in many river basins implies substantial reductions in agricultural P losses. The abatement of diffuse P losses and the choice of mitigation strategies will increasingly rely on the identification of source areas in landscapes that contribute most P to surface water bodies. River basin managers and local environmental authorities currently need tools to assist them in mapping critical source areas of P loss and models to predict the effects of the various mitigation options for reducing P losses. Many and diverse mitigation options for reducing P losses have been suggested. Their effectiveness depends on local conditions, as do the costs of implementation and side effects. Hence, there has been a growing interest in cost-benefit analyses to assist managers and policymakers in choosing the best mitigation options.

The previous International Phosphorus Workshops (1995 Wexford, 1998 Antrim, 2001 Plymouth, 2004 Wageningen) have greatly contributed to increasing our knowledge of the relations between agriculture and P losses, of P transfer from soil to water and of the effects of mitigation measures. The 5th International Phosphorus Workshop takes place in Silkeborg, Denmark, 3-7 September 2007 and is jointly organized by the National Environmental Research Institute and the Faculty of Agricultural Sciences, both from the University of Aarhus. The workshop follows up on the latest developments, focusing on strategies for abating P losses to the aquatic environment. The scope of the workshop is holistic and comprises P cycling and P loss from agriculture, tools for predicting and mapping the risk of P loss, effectiveness of different mitigation options, and the impact of P on the aquatic environment.

These proceedings include extended abstracts of both oral and poster presentations from the IPW5. We wish to thank all contributors for their high quality input and all participants for travelling to Silkeborg. We are very grateful for all the help we have received in organizing the workshop. Our very special thanks go to the workshop secretaries Anne Sehested and Margit Schacht for their patient labours with the proceedings and Anne-Dorthe Villumsen and Birgit Sørensen for their organizational efforts. We gratefully acknowledge the generous financial support from the University of Aarhus and the Danish Research Council for Technology and Production.

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How important is phosphorus in surface waters for complying with the EU Water Framework Directive?

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Introduction

Surface waters in Europe are affected by several anthropogenic pressures causing eutrophication, acidification, accumulation of toxic substances, physical alterations and degradation of littoral habitats. The EU Water Framework Directive (WFD) is a European policy response to combat these processes that lead to deterioration of ecological water quality. According to a recent WFD report, in many EU Member States more than 50% of their water bodies are at risk of not achieving good ecological status by 2015 (COM(2007) 128 final:

http://ec.europa.eu/environment/water/water-framework/implrep2007/index_en.htm).

In many water bodies, eutrophication caused by excessive nutrient loading is reported to be the sole or main reason for this risk.

Diffuse pollution, mostly originating from agricultural land, is the highest source of both phosphorus and nitrogen to surface waters in many countries. This may partly be caused by an increase in agricultural pollution, but also due to the reduction of point sources due to improved wastewater treatment. However, in the south and east of Europe only half of the population is connected to a wastewater treatment facility and only 30 to 40% of the wastewater is processed with secondary or tertiary treatment (EEA 2005). Thus, phosphorus and nitrogen originating from municipal wastewater still remain a problem in large parts of Europe.

Both phosphorus and nitrogen play a role in eutrophication; phosphorus is mainly the limiting factor in fresh waters, as nitrogen is it in marine waters. However, nutrient limitation may differ from this general pattern on a seasonal, interannual and spatial scale for both fresh and marine water bodies. Consequently, successful eutrophication control requires both phosphorus and nitrogen load reductions.

This paper summarizes recent results obtained by an EU co-funded research project REBECCA (Relationships between ecological and chemical status of surface waters). The project investigated and assessed many other pressures (such as hydro-morphology, organic pollution, toxic substances), but only relationships between nutrients and ecological status indicators are summarized here.

Phosphorus and lakes

It has long been known that phosphorus concentration correlates well with eutrophication indicators, e.g. chlorophyll (e.g. Vollenweider and Kerekes, 1980). Present study compiled data from more than 1000 European lakes and the results showed higher slopes for TotP-Chlorophyll relationship compared to earlier studies, yielding higher chlorophyll concentrations per unit phosphorus. Results also showed evidence of non-linearity around TotP concentration of $100 \mu\text{g l}^{-1}$ (see Figure 1).

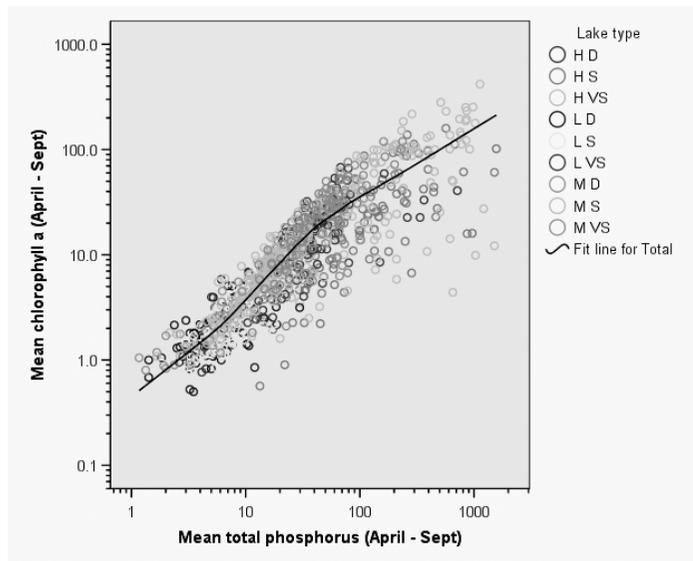


Figure 1. Total phosphorus – chlorophyll a relationship in European lakes. Legend refers to H=high alkalinity, M=moderate alkalinity, L=low alkalinity, D=deep, S=shallow, VS=very shallow. Regression line obtained from a LOESS fit of data points.

Significantly different TotP-chlorophyll relationships were found for lakes grouped by depth and alkalinity. Probably, as a result of light limitation, deep lakes had the lowest yield of chlorophyll per unit of TotP, low and moderate alkalinity shallow lakes the highest. Reduced TotN:TotP ratios were most pronounced in humic lakes, suggesting that in these lakes TotN rather than TotP was the best predictor of chlorophyll.

Phosphorus and rivers

In general, rivers are more often affected by numerous simultaneous pressures (e.g. by organic pollution, nutrient loading, hydromorphological alterations and toxic substances) compared to lakes and coastal waters. This fact caused much unexplained variability in all analyzed relationships between any pressures and ecological indicators in rivers. The best ecological indicators for eutrophication

(nutrient concentrations) was found to be benthic diatom indices, most often predictive power was slightly better for phosphorus than for nitrogen species. The high variability makes these indicators unlikely to be used on their own to design nutrient loading reductions and other mitigation measures, but more as one of the multiple biological indicators of nutrient stress.

Phosphorous and coastal waters

Generally, total nitrogen shows better correlation with biological indicators in coastal waters than phosphorus, except for low salinity coastal areas such as the Bothnian Bay. Many coastal areas shift from phosphorus limitation in spring to nitrogen limitation during summer and fall. Although nitrogen is the most important element for phytoplankton biomass, the availability of phosphorus has implications for the community and succession of phytoplankton. For example, the phytoplankton diversity (expressed by numerous indices) decreased with increasing nutrient levels, including the phosphorus concentrations.

Conclusions

To be compliant with the Water Framework Directive, water management and water pollution control have to be based on improving the ecological quality of water bodies, measured using different biological indicators. Thus, assessments and calculations of required mitigation measures, e.g. levels of nutrient load reductions, require knowledge and understanding of functional and often non-linear relationships between biological indicators and various pressures.

Improvement of ecological water quality in European waters requires reductions for many pressures, often simultaneously. However, in many countries and regions, eutrophication is the most important problem, and often the largest source of nutrients is agriculture. Due to the important role of phosphorus in surface water eutrophication, much can be achieved by significant reductions of phosphorus loads from agriculture.

Acknowledgements

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References

- European Environment Agency, 2005. The European Environment – State and Outlook 2005. Copenhagen.
- Vollenweider, R.A. & Kerekes, J., 1980. The loading concept as a basis for controlling eutrophication philosophy and preliminary results of the OECD programme on eutrophication. *Prog. Wat. Tech.* 12, 5-38.

Phosphorus mobilisation – importance of agricultural practice and soil properties

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Introduction

Mobilisation is the primary step in the process of diffuse phosphorus (P) transfer from soil and comprises solubilisation and detachment mechanisms driven by a combination of chemical, physical and biological-biochemical properties and processes (Haygarth *et al.*, 2005). Solubilisation of inorganic and organic P in agricultural soils is directly linked to P status which in turn is primarily determined by long-term inputs of P in the form of fertilisers and manures. On the other hand, detachment of particulate or colloidal P in the soil environment is closely related to chemical and physical properties that influence infiltration, drainage and erosion processes, which in turn are affected by the timing and intensity of cultivation. Accordingly, significant overall mobilisation of P is most likely to occur in long-established intensively managed agroecosystems with a combination of high P status and regular cultivation.

This overview will highlight recent advances in our understanding of P mobilisation processes in soil in relation to land use and management. The focus will be on specific key aspects of P mobilisation, including the influence of manure amendment on P mobility, detachment and subsurface transfer of P within the soil profile, and the influence of water management on P mobility. Consideration will be given to interactions between different types of soil and various aspects of agricultural practice on the potential for P mobilisation, and how these contribute to the control and mitigation of diffuse P transfer from soil.

Manure amendment and P mobility

It has been well documented that repeated application of manure to soils increases total and soluble P concentrations as well as P saturation indices in relation to the amount of P added. The addition of manure P also influences the forms of P found in soils as well as alter soil chemical properties which ultimately affect P solubility. Precipitation of P in manure amended soils has been shown to occur mainly as tricalcium phosphate and octacalcium phosphate and conversion of P to more stable

forms such as variscite and hydroxyl apatite are inhibited (Sato *et al.*, 2000; Sharpley *et al.*, 2004; Varinderpal-Singh *et al.*, 2006a). This can result in manure amended soils having a higher P availability, although not necessarily solubility compared with fertilizer amendments (Varinderpal-Singh *et al.*, 2006b; Leytem and Westermann, 2005). The application of manure results in increased soil organic carbon which in turn can inhibit precipitation of stable calcium P minerals by adsorption of organic acids onto calcium mineral surfaces (Leytem and Westermann, 2003; Inskeep and Silvertooth, 1998), while the addition of carbon to soils with manure application can stimulate short-term microbial activity and immobilization of P (Leytem *et al.*, 2005).

Detachment and subsurface transfer

In structured soils, P originating from the topsoil or from manure or residues on the soil surface may be lost via water flowing through macropores and by-passing vacant P sorptions sites in the subsoil (Heckrath *et al.*, 1995; Stamm *et al.*, 1998). Phosphorus losses through tile-drainage systems in these soils is therefore directly and immediately affected by manure application and tillage operations if drainage flow is initiated by heavy rainfall shortly after application (Schelde *et al.*, 2006). Surplus P added to soil is mainly found in the clay fraction in the topsoil (Rubæk *et al.*, 1999). Detachment of colloids from the topsoil in response to precipitation is a natural phenomenon, which is affected by both intrinsic and dynamic soil properties such as clay content, mineralogy, organic carbon content, ionic strength of the pore water and soil-water potential (Kjaergaard *et al.*, 2004, Seta and Karathanasis, 1996; Pojasok and Kay, 1990; Flury *et al.*, 2002). Tillage affects detachment of colloids by increasing dispersion and by changing the active flow volume allowing a larger contact area of the infiltrating water in the P rich topsoil. It has also been demonstrated that chemical properties such as electric conductivity is inversely related to detachment of colloids in leaching experiments on structured soils (de Jonge *et al.*, 2004a). At the same time the balance between soluble P release and particulate P detachment in overland flow from clay soils can be influenced by chemical properties such as ionic strength (Ulén, 2003). Furthermore, the capacity of topsoil and subsoil to attenuate mobilised P in fine textured and stoney soils is influenced by the presence and characteristics of preferential flow channels and consequently significant amounts of colloidal-P can be transported in tile drains (Sinaj *et al.*, 2002; Ulén, 2004; de Jonge *et al.*, 2004b). There is a need to further investigate and quantify the relative importance of P losses related to overland flow and P lost through leaching with macropore flow in structured soils.

Water management and P mobility

Water management in agricultural production can have profound effects on P mobilisation in the landscape. The three basic practices with the largest impacts are surface irrigation, tile drainage, and controlled drainage or subsurface irrigation. During an irrigation event, overland flow detaches, transports and deposits sediment

and P, together with P in vegetation and manure. Mundy *et al.* (2003) reported that flow-weighted P concentrations and loads were about 100% higher from pasture cut to 47mm above ground than pasture standing at 155mm. Irrigation can also induce leaching of P through the soil profile in coarse textured and stoney soils. Thus Condrón *et al.* (2006) found that irrigation of pasture improved the utilization of applied fertilizer P but also resulted in significant leaching of P through the soil profile. Installation of artificial drains significantly improves the structural stability of the soil, water quality in recipient streams may be adversely affected by the accelerated rate of nutrient transport, and the circumvention of critical storage areas such as buffer zones. Kinley *et al.* (2007) found that mean total P concentrations in tile drainage exceeded USEPA guidelines at 82% of the fields monitored. Dils and Heathwaite (1999) showed that total P concentrations in tile drain discharge were low ($< 100 \mu\text{g P L}^{-1}$) and stable during base flow periods ($< 0.5 \text{ L min}^{-1}$), but elevated P peaks exceeding 1 mg P L^{-1} were measured in drain-flow during high discharge periods ($> 10 \text{ L min}^{-1}$). Subsurface irrigation is also used to raise the water table close to the soil surface during certain times of the year. Sanchez-Valero *et al.* (2007) reported increased P loads in tile drainage from controlled drainage/subirrigation plots compared to free drainage plots, which were attributed to an increase in P solubility rather than by the addition of P from the subirrigation water. Other studies have found a decrease in P loading in drain outflow under controlled drainage which was related to the decrease in drain outflow rate (Wesstrom and Messing, 2007; Wahba *et al.*, 2001).

References

- Condrón, L.M., Sinaj, S., McDowell, R.W., Dudler-Guela, J., Scott, J.T. & Metherell, A.K., 2006. Influence of long-term irrigation on the distribution and availability of soil phosphorus under permanent pasture. *Aust. J. Soil Res.* 44, 127-133.
- Dils, R.M. & Heathwaite, A.L., 1999. The controversial role of tile drainage in phosphorus export from agricultural land. *Water Sci. Tech.* 39, 55-61.
- Flury, M., Mathison, J.B. & Harsh, J.B., 2002. In situ mobilization of colloids and transport of cesium in Hanford sediments. *Environ. Sci. Technol.* 36, 5335-5341.
- Haygarth, P.M., Condrón, L.M., Heathwaite, A.L., Turner, B.L. & Harris, G.P., 2005. The phosphorus transfer continuum; linking source to impact with an interdisciplinary and multi-scaled approach. *Sci. Total Environ.* 344, 5-14.
- Heckrath, G., Brookes, P.C., Poulton, P.R. & Goulding, K.W.T., 1995. Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk Experiment. *J. Environ. Qual.* 245, 904-910.
- Inskeep, W.P. & Silvertooth, J.C., 1988. Inhibition of hydroxyapatite precipitation in the presence of fulvic, humic, and tannic acids. *Soil Sci. Soc. Am. J.* 52, 941-946.
- de Jonge, L.W., Kjaergaard, C. & Moldrup, P., 2004a. Colloids and colloid-facilitated contaminants in soils: an Introduction. *Vadoze Zone J.* 3, 421-425.
- de Jonge, L.W., Moldrup, P., Rubæk, G.H., Schelde, K. & Djurhuus, J., 2004b. Particle Leaching and particle-facilitated transport of phosphorus at field scale. *Vadoze Zone J.* 3, 462-470.
- Kinley, R.D., Gordon, R.J., Stratton, G.W., Patterson, G.T. & Hoyle, J., 2007. Phosphorus losses through agricultural tile drainage in Nova Scotia, Canada. *J. Environ. Qual.* 36, 469-477.

- Kjærgaard, C., de Jonge, L.W., Moldrup, P. & Schjønning, P., 2004. Water dispersible colloids: Effects of measurement method, clay content, initial matric potential and wetting rate. *Vadose Zone J.* 3, 403-412.
- Leytem, A.B. & Westermann, D.T., 2003. Phosphorus sorption by Pacific Northwest calcareous soils. *Soil Sci.* 168, 368-375.
- Leytem, A.B. & Westermann, D.T., 2005. Phosphorus availability to barley from manures and fertilizers on a calcareous soil. *Soil Sci.* 170, 401-412.
- Leytem, A.B., Turner, B.L., Raboy, V. & Peterson, K.L., 2005. Linking manure properties to phosphorus solubility in calcareous soils: Importance of the manure carbon to phosphorus ratio. *Soil Sci. Soc. Am. J.* 69, 1516-1524.
- Mundy, G.N., Nexhip, K.J., Austin, N.R. & Collins, M.D., 2003. The influence of cutting and grazing on phosphorus and nitrogen in irrigation runoff from perennial pasture. *Aust. J. Soil Res.* 41, 675-685.
- Pojasok, T. & Kay, B.D., 1990. Assessment of a combination of wet sieving and turbidmetry to characterize the structural stability of moist aggregates. *Can. J. Soil Sci.* 70, 33-42.
- Rubæk, G.H., Guggenberger, G., Zech, W. & Christensen, B.T., 1999. Organic phosphorus in soil size separates characterised by phosphorus-31 nuclear magnetic resonance and resin extraction. *Soil Sci. Soc. Am. J.* 63, 1123-1132.
- Sanchez Valero, C., Madramootoo, C.A. & Stämpfli, N., 2007. Water table management impacts of phosphorus loads in tile drainage. *Ag. Water Manag.* 89, 71-80.
- Sato, S., Solomon, D., Hyland, C., Ketterings, Q.M. & Lehmann, J., 2005. Phosphorus speciation in manure and manure-amended soils using XANES spectroscopy. *Environ. Sci. Tech.* 39, 7485-7491.
- Seta, A. & Karathanasis, A.D., 1996. Stability and transportability of water-dispersible soil colloids. *Soil. Sci. Soc. Am. J.* 61, 604-611.
- Schelde, K., de Jonge, L.W., Kjærgaard, C., Laegdsmand, M. & Rubæk, G.H., 2006. Effects of manure application and plowing on transport of colloids and phosphorus to tile drains. *Vadose Zone J.* 5, 455-458.
- Sharpley, A.N., McDowell, R.W. & Kleinman, P.J.A., 2004. Amounts, forms, and solubility of phosphorus in soils receiving animal manure. *Soil Sci. Soc. Am. J.* 68, 2048-2057.
- Sinaj, S., Stamm, C., Toor, G.S., Condrón, L.M., Hendry, T., Di, H.J., Cameron, K.C. & Frossard, E., 2002. Phosphorus exchangeability and losses from two grassland soils. *J. Environ. Qual.* 31, 319-330.
- Stamm, C., Fluhler, H., Gächter, R., Leuenberger, J. & Wunderli, H., 1998. Preferential transport of phosphorus in drained grassland soils. *J. Environ. Qual.* 27, 515-522.
- Ulén, B., 2003. Concentration and transport of different forms of phosphorus during snowmelt runoff from an illite clay soil. *Hydrol. Proc.* 17, 747-758.
- Ulen, B., 2004. Size and settling velocities of phosphorus containing particles in water from agricultural drains. *Water Air Soil Poll.* 157, 331-343.
- Varinderpal-Singh, Dhillon, N.S., Raj-Kumar & Brar, B.S., 2006a. Long-term effects of inorganic fertilizers and manure on phosphorus reaction products in a Typic Ustochrept. *Nutr. Cycl. Agroecosy.* 76, 29-37.
- Varinderpal-Singh, Dhillon, N.S. & Brar, B.S., 2006b. Influence of long-term use of fertilizers and farmyard manure on the adsorption-desorption behaviour and bioavailability of phosphorus in soils. *Nutr. Cycl. Agroecosy.* 75, 67-78.
- Wahba, M.A.S., El-Ganainy, M., Abdel-Dayem, M.S., Gobran, A. & Kandil, H., 2001. Controlled drainage effects on water quality under semi-arid conditions in the Western Delta of Egypt. *Irrig. Drain.* 50, 295-308.
- Wesström, I. & Messing, I., 2007. Effects of controlled drainage on N and P losses and N dynamics in a loamy sand with spring crops. *Ag. Water Manag.* 87, 229-240.

Understanding spatial signals in catchments: linking critical areas, identifying connection and evaluating response

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Introduction

The critical source area (CSA) concept is embedded in much of our thinking about how we represent the risk of diffuse sources of nutrients, especially phosphorus (P), being delivered to watercourses. Examples range from simple P Index approaches (e.g. Gburek *et al.*, 2000; Heathwaite *et al.*, 2003a), to screening tools designed to work at large scales (e.g. Heathwaite *et al.*, 2003b; Anderson *et al.*, 2005), to process-based models of diffuse P delivery (e.g. Whitehead *et al.*, 2006). Recent work (Brazier *et al.*, 2006) has sought to explicitly address the uncertainties inherent in representing the delivery of nutrients to watercourses where there is limited data on which to base predictions.

This paper will examine the implications of the delivery of pollutants from diffuse sources to water from the perspective of the measures needed to protect watercourses from these inputs. We will show how it is possible, using a parsimonious approach, to identify and prioritise landscape units (e.g. fields) where the consequences of land management activities are most readily transmitted to watercourses. In doing so, we will show how the CSA concept may be developed further by linking the delivery of diffuse pollutants to water to an understanding of the ecological response of the waterbody to such inputs (Lane *et al.*, 2006; Reaney *et al.*, submitted). Unless we can evaluate the implications of diffuse pollutants for the 'response' of the waterbody (e.g. the quality of habitat for fish), our understanding of the CSAs of diffuse pollutant risk will continue to remain isolated from our understanding of the ecological health of receiving waters.

The concept of relative risk – a parsimonious approach to diffuse pollution

There exists a circular argument in the way that models of diffuse pollution and field measurements relate to one another: complex process-based models need data for calibration but current technology is not able to supply the data at the appropriate spatial and temporal scales (Kirchner 2006). Consequently, we resort to interpolation, extrapolation or downright guesswork; this introduces uncertainties that are rarely dealt with explicitly and so constrains the quality of the models. The complexity of P delivery is a good example here (Beven *et al.*, 2005).

The question we pose is, if we do not have the tools to measure the delivery of diffuse pollutants to water at appropriate scales, is it possible to adopt a parsimonious approach that uses the best available technology and data but in a way that looks at the relative risk of a diffuse source pollutant reaching a waterbody in terms of its connection to that waterbody. The SCIMAP approach (www.scimap.org) is built on the CSA concept but uses 'minimum information requirement' (MIR) ground rules to represent diffuse pollution in a probabilistic framework (Lane *et al.*, 2006). It poses the question: what do we really need to know and what is the minimum information requirement to get there? It is based on the principle of the network index (Lane *et al.*, 2004).

New research has demonstrated a unique relationship between the signature of a catchment in terms of its fine sediment connectivity and fish habitat response (Lane *et al.*, submitted¹). And we have shown that the spatial structure of landscape connection may be controlled by a relatively small number of topographically-defined locations (Lane *et al.*, submitted²). Such locations are likely to be critical for the delivery of diffuse pollutants to water. We have shown that the approach works well for the delivery to water of pollutants such as fine sediment via surface runoff (Figure 1). New work will be presented that has developed the approach further to consider the connectivity between P sources in catchments and the P signal in receiving waters.

References

- Andersen, H.E., Kronvang, B. & Larsen, S.E., 2005. Development, validation and application of Danish empirical phosphorus models. *J. Hydrol.* 304, 355-365.
- Beven, K., Heathwaite, A.L., Haygarth, P.M., Walling, D.E., Brazier, R.E. & Withers, P., 2005. On the concept of delivery of sediment and nutrients to stream channels. *Hydrol. Proc.* 19, 551-556.
- Brazier, R.E., Schärer, M., Heathwaite, A.L., Beven, K., Scholefield, P., Haygarth, P.M., Hodgkinson, R., Walling, D.E. & Withers, P., 2006. A framework for predicting delivery of phosphorus from agricultural land using a decision-tree approach. *IAHS Publ.* 306, 514-523.
- Gburek, W.J., Sharpley, A.N., Heathwaite, A.L. & Folmar, G., 2000. Phosphorus management at the watershed scale. *J. Env. Qual.* 29, 130-144.
- Heathwaite, A.L., Sharpley, A.N. & Bechmann, M., 2003a. The conceptual basis for a decision support framework to assess the risk of phosphorus loss at the field scale across Europe. *J. Pl. Nut. & Soil Sci.* 166, 1-12.
- Heathwaite, A.L., Fraser, A.I., Johnes, P.J., Hutchins, M., Lord, E. & Butterfield, D., 2003b. The Phosphorus Indicators Tool: a simple model of diffuse P loss from agricultural land to water. *Soil Use & Managt.* 19, 1-11.
- Kirchner, J.W., 2006. Getting the right answers for the right reasons: linking measurements, analyses, and models to advance the science of hydrology, *Water Resources Research* 42 W03S04, doi:10.1029/2005WR004362.

- Lane, S.N., Brookes, C.J., Kirkby, M.J. & Holden, J., 2004. A network-index based version of TOPMODEL for use with high-resolution digital topographic data. *Hydrol. Proc.* 18, 191-201.
- Lane, S.N., Brookes, C.J., Heathwaite, A.L. & Reaney, S., 2006. Surveillant science: challenges for the management of rural environments emerging from the new generation diffuse pollution models. *J Ag. Econ.* 57, 239-257.
- Lane, S.N., Reaney, S. & Heathwaite, A.L. (submitted¹) Topographical control of landscape connectivity by surface flow. *Geophysical Research Letters*.
- Lane, S.N., Burt, T.P., Dickson, J., Dugdale, L., Heathwaite, A.L., Maltby, A. & Reaney, S. (submitted²) Hydrological connectivity and instream ecology: Surface hydrological connectivity influences the extent to which landscape factors impact on instream ecology. *Conservation Biology*.
- Reaney, S., Lane, S.N. & Heathwaite, A.L. (submitted) Inverse modelling of catchment connectivity and land management impacts upon juvenile salmonid fry. *Ecological Modelling*.
- Whitehead, P.G., Heathwaite, A.L., Flynn, N.J., Wade, A.J. & Quinn, P.F., 2006. Evaluating the risk of nonpoint source pollution from biosolids: integrated modelling of nutrient losses at field and catchment scales. *Hydrol. & Earth Systems Sci.* 10, 2-13.

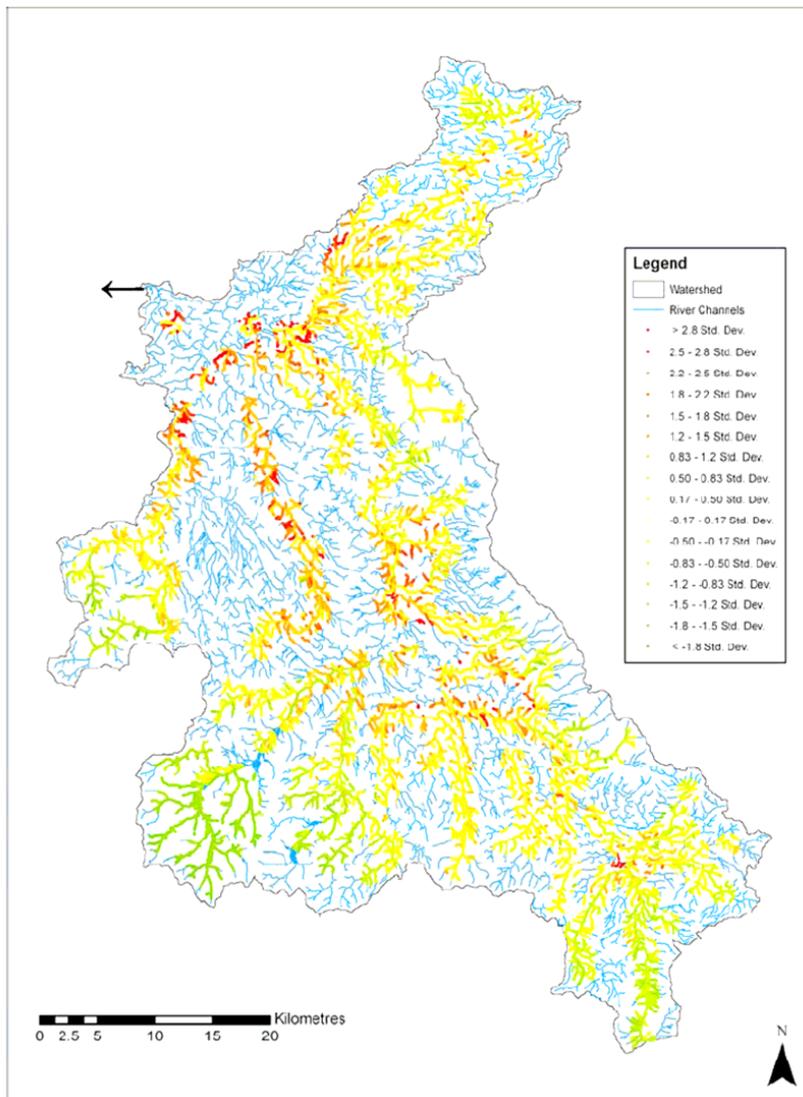


Figure 1. Delivery index predictions for the R. Eden Catchment, NW England. The drainage network is shown in blue. The delivery index expresses the likelihood of surface hydrological connectivity between source and receptor and is shown only for those river reaches where field data have shown that the instream habitat should be suitable for brown trout. The index is expressed as standard deviations from the catchment average: negative standard deviations indicate lower delivery index values than the catchment mean. The index has been shown to discriminate between locations where field records show brown trout fry were present or absent: a low delivery index is related to higher fry abundance (Lane *et al.*, submitted²).

Demonstrating phosphorus mitigation strategies can work at field and catchment scales

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Introduction

Studies have demonstrated some phosphorus (P) loss reduction following implementation of remedial strategies. For instance, Jokela et al. (2004) and Baker and Richards (2002) reported improved water quality in Lake Champlain and Erie, respectively, as a result of decreased P inputs following implementation of Best Management Practices (BMPs) in their catchments. However, there has been little coordinated catchment scale evaluation of P-based BMPs, to show where, when, and which work most effectively to minimize degradation. Research is needed to evaluate spatial and temporal variability in system response to BMP implementation. This will allow us to answer the critical questions; how long before we see an environmental response and where would we expect the greatest response?

Results and discussion

To remediate deteriorating Great Lakes water quality, BMPs were targeted to agricultural nonpoint sources. Between 1975 and 1995, in the Maumee and Sandusky River tributary catchments of Lake Erie, conservation tillage increased from virtually nothing to 50% of cropland (mainly no-till soybean and corn); 75,000 hectares (<5% of total farmland in the catchments) were taken out of production (i.e., Conservation Reserve Program), and applied fertilizer and manure P decreased (Baker and Richards, 2002). These measures translated into significant decreases in total (TP; 40%) and dissolved P (DP; 77%) concentrations averaged for catchment tributaries between 1975 and 1995. Overall, BMPs, decreased fertilizer and manure applications, which were the main factors affecting P reductions.

Even so, the question still remains as to whether P-based measures, will actually decrease soil and runoff P levels and how long will it be before significant decreases are seen, especially to levels below water quality thresholds? The effect of P-based manure applications on soil and runoff P was evaluated for an Othello silt loam (*Typic Endoaquults*) under a corn-soybean rotation that had received poultry litter for the last 20 years and as a result had high soil test P (~400 mg kg⁻¹ as Mehlich-3 P). Poultry litter applications were N-based, to meet crop N requirements (40 to 116 kg P ha⁻¹ yr⁻¹); P-based, to supply crop P uptake (20 to 58 kg P ha⁻¹ yr⁻¹); and soil test P

threshold, where no litter was applied as Mehlich-3 P was $>200 \text{ mg kg}^{-1}$. Although the loss of DP and TP in runoff increased each year since the three strategies were implemented, due to increased annual rainfall and runoff volumes, the effect of P-based and soil test P strategies on decreasing P loss compared to N-based was evident after three years (2002; Table 1).

Table 1. Runoff and P loss as a function of basing poultry litter applications on crop N requirement (N-based), crop P requirement (P-based), and soil test P as Mehlich-3 P for 0.1 ha plots in Coastal Plains region of Maryland.

Treatment ¹	2000	2001	2002	2003	2004
Rainfall, cm	7.7	37.1	32.2	64.3	108.3
Runoff, cm	0.05	1.25	4.00	4.50	8.00
<i>Soil test – Mehlich-3 P, mg kg⁻¹</i>					
N-based	401	477	480	512	558
P-based	401	433	450	463	488
Soil test P	401	410	394	366	320
Decrease, % ²	--	14	18	29	43
<i>Dissolved P runoff, g ha⁻¹</i>					
N-based	0.33	29	466	2050	3112
P-based	0.05	34	72	268	1063
Soil test P	0.07	19	52	144	517
Decrease, % ²	79	34	89	93	83
<i>Total P runoff, g ha⁻¹</i>					
N-based	2.37	185	2067	2509	3493
P-based	1.08	170	1361	1633	1386
Soil test P	1.35	124	1016	1300	689
Decrease, % ²	43	33	51	48	80

¹ P applied in poultry litter averaged 75, 35, and 0 kg P ha⁻¹ for N-based, P-based, and soil test P treatments.

² Percent decrease in runoff P loss from soil test P compared to N-based litter treatment.

In the fifth year of treatment, DP and TP losses were a respective 83 and 80% lower from the soil test P than N-based approaches (Table 1). Over the same time, surface soil (0 to 5 cm depth) Mehlich-3 P decreased with the soil test P threshold approach only (401 to 320 mg P kg⁻¹) and as a consequence, corn and soybean yields were not affected by any management approach (Table 1). This research shows that while implementation of P-based management can decrease runoff P, it took three years for these effects to be evident. Even five years after implementing nutrient management changes, both mean annual TP concentrations (1.85 and 1.07 mg L⁻¹ for P- and soil test P-based approaches) in runoff and surface soil (488 and 320 mg kg⁻¹ for P- and soil test P-based approaches) were still above respective environmental thresholds for flowing waters and soils (0.05 mg L⁻¹ for total P and 75 mg kg⁻¹ for Mehlich-3 P; Gibson et al., 2000).

In a Swedish study conducted in lysimeters containing a sandy soil over 3 yrs, it was found that increasing input of P with manure (up to 320 kg P ha⁻¹ during the period), unexpectedly decreased P leaching significantly (Bergström and Kirchmann, 2006). In contrast, leaching of N increased with increasing manure inputs. Similarly, Djodjic et al. (2004, found that in three of five soils, which had received different P inputs during 40 yrs, P leaching loads tended to decrease with increasing P inputs. This indicates that it may take quite a long time for a new fertilizer strategy to have any effect on water quality. Crop yields in the replacement treatment of these soils were lower than in the surplus treatment (Djodjic et al., 2005). However, use efficiency of surplus P applied was very low, indicating that only a small portion of the surplus P was used by the crop. Although these results are contradictory, to maintain optimum yields and limit P surpluses, balanced P inputs are the most prudent approach. However, additional management measures are also needed to reduce P losses.

Conclusions

The lag time between BMP implementation and water quality improvements can be several years. Despite our knowledge of controlling processes, it is difficult for the public to understand or accept this lack of response. When public funds are invested in remediate programs, rapid improvements in water quality are usually expected. Thus, assessment of effectiveness of P-based BMPs must consider re-equilibration of catchment and lake behavior, where nutrient sinks may become sources of P with only slight changes in catchment management and hydrologic response.

References

- Baker, D.B. & Richards, R.P., 2002. Phosphorus budgets and riverine phosphorus export in northwest Ohio. *J. Environ Qual.* 31, 96-108.
- Bergström, L. & Kirchmann, H., 2006. Leaching and crop uptake of nitrogen and phosphorus from pig slurry as affected by different application rates. *J. Environ. Qual.* 35, 1633-1968.
- Djodjic, F., Börling, K. & Bergström, L., 2004. Phosphorus leaching in relation to soil type and soil phosphorus content. *J. Environ. Qual.* 33, 678-684.
- Djodjic, F., Bergström, L. & Grant, C., 2005. Phosphorus management in balanced agricultural systems. *Soil Use Manage.* 21, 94-101.
- Gibson, G.R., Carlson, R., Simpson, J., Smeltzer, E., Gerritson, J., Chapra, S., Heiskary, S., Jones, J. & Kennedy, R., 2000. Nutrient criteria technical guidance manual: lakes and reservoirs. EPA-822-B00-001. U.S. Environmental Protection Agency, Washington, D.C.
- Jokela, W.E., Clausen, J.C., Meals, D.W. & Sharpley, A.N., 2004. Effectiveness of agricultural best management practices in reducing phosphorous loading to Lake Champlain. p. 39-53. *In* Manley, T.O., Manley, P.L. & Mihuc, T.B. (eds.). *Lake Champlain: Partnerships and Research in the New Millennium*. Kluwer Academic Publishers, Dordrecht, The Netherlands.
- Richards, P.R. & Baker, D.B., 2002. Trends in water quality in LEASEQ rivers and stream (Northwestern Ohio), 1975 – 1995. *J. Environ. Qual.* 31, 90-96.

Quantifying diffuse phosphorus (P) losses to the farm/sub-catchment scale: targeting methods and uncertainties for P loss mitigation

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Introduction

Phosphorus (P) inputs affect surface water quality and these effects change with scale. In this paper we examine the use of (i) empirical data and (ii) Bayesian networks to investigate the impacts of pastoral farming systems. We attempt to answer the question – which processes offer the highest probability of efficiently mitigating P losses and impacts?

In some ways, intensive pastoral farming systems are simple. Often they have only a few soil types, are located on flat to rolling topography, and are infrequently disturbed by grazing animals or mechanical traffic. The system is often simpler in Australasia: cows rotationally graze paddocks every 20-60 days depending on the time of year and similar amounts of P fertiliser are applied each year (provided nutrient budgeting is adhered to). Add to this that we know the deposition rates of dung for cattle, sheep and deer, and we can start to estimate the quantity of P returned to paddocks or waterways (if accessible) throughout the year. Given the similarity and routine of soil management, the potential of a paddock to lose P can therefore be categorized into inherent loss from the soil, P lost associated with dung deposition or fertiliser deposition, and P loss associated with animal traffic (treading) and grazing (defoliation).

Empirical approach

If we assume a similar topography, we can begin to estimate the relative importance each source of P loss for a given runoff volume via empirical data. At a small scale, P losses via overland flow and subsurface flow can be generated via simulated rainfall. However, relevance to field conditions depends on the rainfall intensity and duration used since we know these affect P losses and forms (dissolved vs. particulate). Data for pastoral dairy systems in the South Island of New Zealand have utilised a median rainfall intensity to generate overland and sub-surface flow and estimates of P losses from various sources. For example, the concentration of P loss in overland flow from ungrazed pasture and soil can be determined as = $[0.024 \text{Olsen P (mg kg}^{-1}) / \text{P retention (\%)}] + 0.02$, where P retention is that left behind after buffering with a known concentration of P at pH 4.5 (Saunders, 1964). For dung deposited on

pasture, P loss in overland flow declines exponentially with time, varying from 3-6 mg P L⁻¹ 1 day after deposition (depending on the initial P concentration in dung) to about 0.2 mg P L⁻¹ 6 days later. P loss in overland flow associated with P-fertiliser depends on its water solubility, but, like dung, declines exponentially with time since application for 30-60 days, after which P concentration is similar to that before application. Outside of this 30-60 day period, additional studies have examined the effect of typical animal treading rates (20-30 imprints m² for 24 h grazing) and defoliation. Treading tends to increase P losses exponentially via sediment disturbance beyond 24 h grazing time, while defoliation increases P losses for about 7 days compared to ungrazed pasture.

The positive or negative trends with soil and animal management means that the probability for one source to be dominant is slim, but possible if, for example, an overland flow event should occur soon after fertilisation. At the moment, using an empirical approach has enabled us to quantify or account for sources of P loss in uniform systems and determine the most efficient mitigation practice. An example is given in Table 1. Estimated loads are similar to the actual loads and modelling would indicate that simply switching to a poorly water soluble P-fertiliser would decrease loads significantly off these paddocks without much additional cost (reactive phosphate rock tends to be about 10% dearer than superphosphate). However in another year, decreasing soil Olsen P concentration would be more effective. This could be done in combination with an alternative fertiliser strategy and could save the farmer additional money by applying less P.

Table 1. Actual and estimated total loads (kg P ha⁻¹), and loads from various sources, of P lost from grazed paddocks in a dairy farm on the West Coast of the South Island of New Zealand.

Year	Actual Total	Estimated				Total
		Treading/defoliation	Soil	Dung	Fertiliser	
2002	7.7	12	25	19	45	7.8
2003	2.9	20	45	30	5	2.9

These empirical relationships form the basis of our understanding of P loss processes and are often incorporated into models for scientists or end users like OVERSEER Nutrient Budgets 2[®] (McDowell *et al.*, 2005). Of course this is only applicable to pastures that are grazed uniformly. Concepts, such as critical source areas that dictate that certain areas of a catchment contribute disproportionately more than others, suggest that empirical relationships will be influenced by scaling.

Bayesian networks

While linking field scale models such as OVERSEER to catchment scale outcomes is the holy-grail of nutrient research, unfortunately, there is often insufficient information

to validate such models and routing P through stream systems has proven difficult. It is widely recognised that in-stream processing occurs, but the multitude of processes are often represented by a simple decay, usually linear, function. Process based, Bayesian Networks offer the opportunity of combining the best features of both empirical and conceptual modelling to develop catchment scale models that integrate collective research and experience.

Bayesian networks are increasingly being used to study multi-factor problems, such as water quality. Bayesian networks represent uncertainty in knowledge and use probability theory to manage uncertainty by explicitly representing the conditional dependencies between the different knowledge components. Bayesian networks are, in essence, cause and effect diagrams with the relationships represented as probability tables. Consequently, Bayesian Networks provide an intuitive graphical visualisation (i.e. conceptual model) of the knowledge including the interactions among the various sources of uncertainty.

Bayesian networks have recently been used to investigate the effects of “Better Management Practices” on P exports from individual farms in south-eastern Australia where little empirical data exists. Developed using social survey techniques and calibrated using both empirical relationships from other areas and expert opinion, these networks show that, with current technology, some farmers have already eliminated the majority of P exports that are under their control. In other cases the networks suggest options for improvement and quantify P export reductions from one or more management changes. Importantly, the networks identify the key processes affecting nutrient exports on individual farms.

Conclusions

Bayesian networks are not data intensive and allow integration of qualitative information and knowledge with the types of quantitative information generally included in integrated models. Readily available data can be used with an associated confidence interval and this confidence interval can be combined and associated with the final model output. The potential for combining in Bayesian networks the empirical data underlying tools such as OVERSEER with qualitative information on in-stream processing and social information such as farmers’ attitudes towards adoption of “Better Management Practices” offers the realistic possibility of integrated, cross disciplinary, decision support tools.

References

- McDowell, R.W., Monaghan, R.M. & Wheeler, D., 2005. Modeling phosphorus loss from New Zealand pastoral farming systems. *N. Zeal. J. Ag. Res.* 48, 1-11.
- Saunders, W.M.H., 1964. Phosphate retention by New Zealand soils and its relationship to free sesquioxides, organic matter, and other soil properties. *N. Zeal. J. Ag. Res.* 8, 30-57.

Phosphorus dynamics in wetlands and riparian areas

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Introduction

Wetlands and riparian areas have received much attention due to their ability to retain nutrients or transform nutrients into less harmful substances thereby mitigating nutrient loadings on downstream recipients. This has led to a variety of initiatives such as re-establishment of former wetlands and shallow lakes, remeandering of rivers whereby riparian areas are hydraulically reconnected with the adjacent river. Furthermore, establishment of buffer strips, construction of different types of wetlands and ponds have been extensively used to reduce nutrient loads on downstream recipients.

In this context phosphorus provides a good example because wetlands and riparian areas in its widest sense are used to solve or mitigate several environmental problems related to agriculture such as enhanced leaching of phosphorus due to enhanced use of fertilizer, erosion from sloping fields, inappropriate tillage practise, bank erosion and enhanced phosphorus transport in running waters due to heavy stream and river maintenance.

The effectiveness of wetlands and riparian areas in retaining phosphorus is strongly related to the form of phosphorus and the processes involved. This paper goes through some of the most important phosphorus retention mechanisms taking place in riparian corridors.

Restored wetlands and shallow lakes

Denmark offers a good example concerning P retention in restored systems. A wetland restoration programme was initiated as part of The Second Action Plan on The Aquatic Environment. The aim was to restore 16,000 ha of wetlands and shallow lakes between 1998 and 2003. At present approximately 7,000 ha have been restored, but the programme has been prolonged and new projects are still appearing. Although the action plan focused on nitrogen reduction, net releases or leaching of phosphorus was not allowed as a consequence of the restoration project (Hoffmann and Baattrup-Pedersen, 2007). The wetland monitoring programme also

included phosphorus and results are shown in Table 1. The mass balances are made from simple input output measurements based on monthly sampling of nutrients and water discharge measurements. Six of the eight lakes retain phosphorus in the first year following restoration, while seven out of nine wetlands retain phosphorus. Generally, there is not enough detailed information about the mechanisms responsible for the phosphorus retention in the restored wetlands but indirectly the monitoring data points to sedimentation as the most obvious P retention process as months with large discharge events combined with inundation of riparian areas showed significant P-retention while several low flow months did not show P-retention (Hoffmann et al., 2006).

Table 1. P-retention in restored wetlands and shallow lakes (Hoffmann et al., 2006).

Wetlands	P-retention kg P ha⁻¹ year⁻¹	Retention efficiency %
Ulleruplund, irrigated meadow	-0.43	-88
Gammelby Bæk, irrigated and inundated fen and meadow (uncertain calculation)	-0.4 - 20	-7 - 75
Egebjerg Enge, inundated meadow	0.13	6
Karlsmosen, irrigated and inundated fen and meadow	8.1 – 9.0	53-60
Snarelose, irrigated fen-meadow	2.6	18
Lindkær, irrigated fen-meadow	-0.5	-11
Geddebækken, irrigated fen-meadow	0.5	21
Nagbøl Å, remeandered, irrigation	0.9	11
Hjarup Bæk, remeandered, irrigation	12	42
Shallow lakes	P-retention kg P ha⁻¹ year⁻¹	%
Årslev Eng sø	-1.3	-4
Ødis Sø	-2.3	-192
Wedellsborg	16	(91)
Skibet	3.0	43
Sliv Sø	2.9	23
Nakkebølle	2.7	35
Gødstrup Enghave	0.9	26
Hals Sø	0.7	-

Flooding and sedimentation

The amount of sediment deposited during overbank flooding events is generally high and may account for a substantial amount of the total annual sediment load. Kronvang et al. (1999) reported that 5.6 – 23.9 % of the total suspended sediment export from the river Gjern basin was entrapped on the floodplain during short overbank flood events (i.e. 1-30 days). Phosphorus is sorbed to suspended particles, especially the smaller sized particles in the silt and clay fractions, and these plate-like structures may be carried far away from the river channel on the floodplain before they settle again in slow flowing waters. Table 2 shows a few results from European inundation studies. P deposition rates are significant, also on restored floodplains, and storage efficiency may range from 4.0 – 7.0 % of total river transport (Kronvang

et al., 2007). Studies of soil cores sampled from flooded floodplains indicate that retention of particle bound P to some extent is permanent, although French studies have shown that liberation of dissolved phosphorus takes place during periods following periods without flooding, e.g. first autumn flood (Brunet and Astin, 1998, 2000).

Table 2. P-deposition rates from different European studies.

Type	Country	P-deposition g P m ⁻² yr ⁻¹	Method	Reference
20 floodplains	UK	1.3-11.6	Cs-137	Walling, 1999
Floodplain	DK	11.8	Traps	Kronvang et al., 2002
Floodplain 10 years	DK	8.2	Estimated	Kronvang et al., 2002
Floodplain	France	9.0	Cs-137	Fustec et al., 1995
Floodplain	France	12.7	Budget	Brunet and Astin, 1998
3 Restored Floodplains	DK	1.2-3.6	Traps	Kronvang et al., 2007

Buffer zones

(Dillaha et al., 1989; Uusi-Kämpä et al., 2000). Several factors seem to influence the efficiency, and especially the width of the buffer is important (Kronvang et al., 2005; Syversen, 2005), but also vegetation type, management practice (mowing), amount of surface water runoff and soil texture are key factors (Kronvang et al., 2005; Syversen, 2005; Uusi-Kämpä, 2005).

Planning restoration of riparian corridors

Planning restoration of riparian corridors need a holistic perspective taking various risks into account. Examples of risks in connection with changing the use and function of the riparian corridor are:

- Overload of the riparian corridor with nutrients which may facilitate eutrophication and unwanted impacts on the biodiversity.
- Creation of too many upstream wetlands may lead to downstream erosion because the river will try to retrieve its capacity for sediment transport.
- Risk of P-release from former agricultural fields, which has received a high surplus for many years (e.g. Kjaergaard et al., 2007).
- Contamination of buffer zones due to overloading of nutrients.

To overcome some of these risks, restoration of the riparian corridor will call for targeted management options as e.g. mowing, grazing, removal of sediment surplus or technical measures as e.g. dykes, controlled drainage.

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References

- Brunet, R.C. & Astin, K., 1998. Variation in phosphorus flux during a hydrological season: the river Adour. *Wat. Res.* 32(3), 547-558.
- Brunet, R.C. & Astin, K., 2000. A 12-month sediment and nutrient budget in a floodplain reach of the river Adour, southwest France. *Regulated Rivers. Research & Management* 16, 267-277.
- Dillaha, T.A., Reneau, R.B., Mostaghimi, S. & Lee, D., 1989. Vegetative filter strips for agricultural nonpoint source pollution control. *Trans. ASAE* 32, 513-519.
- Fustec, E., Bonte, P., Fardeau, J.C., Khebibeche, L., Chesterifoff, A. & Carru, A.M., 1996. La rétention des MES et des polluants associés dans les zones inondables. In *Piren-Seine Rapport 1996/II, Thème 'corridor Fluvial'*, Laboratoire de Géologie Appliquée, Université Pierre et Marie Curie, Paris: 3-3-3-16.
- Hoffmann, C.C. & Baatrup-Pedersen, A., 2007. Re-establishing freshwater wetlands in Denmark. *Ecological Engineering* 30, 157-166.
- Hoffmann, C.C., Baatrup-Pedersen, A., Amsinck, S.L. & Clausen, P., 2006. Overvågning af Vandmiljøplan II vådområder 2005 (Surveying wetlands established under The Second Action On The Aquatic Environment). *Danmarks Miljøundersøgelser. - Faglig rapport nr. 576 fra DMU*, 128 pp.
- Kjaergaard, C., Hoffmann, C.C. & Greve, M.H., 2007. Phosphorus forms and phosphorus release as affected by organic lowland geochemistry. *This issue*.
- Kronvang, B., Andersen, I.K., Hoffmann, C.C., Pedersen, M.L., Ovesen, N.B. & Andersen, H.E., 2007. Water Exchange and Deposition of Sediment and Phosphorus during Inundation of Natural and Restored Lowland Floodplains. *Water, Air, & Soil Pollution* 181, 115-121.
- Kronvang, B., Laubel, A., Larsen, S.E., Andersen, H.E. & Djurhuus, J., 2005. Buffer zones as a sink for sediment and phosphorus between the field and stream: Danish field experiences. *Water Sci. Technol.* 51(3-4), 55-62.
- Kronvang, B., Falkum, Ø., Svendsen, L.M. & Laubel, A., 2002. Deposition of sediment and phosphorus during overbank flooding. *Verh. Internat. Verein. Limnol.* 28, 1-5.
- Kronvang, B., Hoffmann, C.C., Svendsen, L.S., Windolf, J., Jensen, J.P. & Dørge, J., 1999. Retention of nutrients in river basins. *Aquatic Ecology* 33, 29-40.
- Syversen, N., 2005. Effect and design of buffer zones in the Nordic climate: The influence of width, amount of surface runoff, seasonal variation and vegetation type on retention efficiency for nutrient and particle runoff. *Ecol. Eng.* 24(5), 483-490.
- Uusi-Kämpä, J., Braskerud, B., Jansson, H., Syversen, N. & Uusitalo, R., 2000. Buffer zones and constructed wetlands as filters for agricultural phosphorus. *J. Environ. Qual.* 29, 151-158.
- Uusi-Kämpä, J., 2005. Phosphorus purification in buffer zones in cold climates. *Ecol. Eng.* 24(5), 491-502.
- Walling, D.E., 1999. Linking land use, erosion and sediment yields in river basins. *Hydrobiologia* 410, 223-240.

Critical evaluation of mitigation options for phosphorus from field to catchment scales

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Introduction

Since recognizing P as a major pollutant of surface waters, much research has been done on a wide variety of mitigation options. Using this research as a basis, many regions of the world have now started to implement strategies at the field and catchment scales to minimize P losses. However, approaches used to address P losses vary greatly by country and catchment, and in the US by state. Many of the regulations target animal producers due to manure management issues. Water quality is regulated in the US under the Clean Water Act, and much of the implementation of strategies to improve water quality has been the result of lawsuits citing failure to meet Clean Water Act standards. In Europe, the EU Water Framework Directive now sets criteria for “good water quality” in rivers, lakes and coastal waters. Plans to fulfill these criteria are presently being prepared in EU countries. Exactly how effective the mitigation options will be remains to be seen, as changes in practices take many years to yield results in terms of water quality improvements.

Examples of mitigation options that have been implemented

a) Denmark

Agricultural P loss has become the major contributor to the eutrophication of many Danish lakes and estuaries, as the loss of P from point sources has been considerably reduced due to the environmental regulations initiated in the 1980's (Poulsen og Rubæk, 2005). At that time little was known about the agricultural contribution of P to surface waters and the regulations addressing agriculture therefore focused on e.g. nitrogen and handling of manure. However, knowing what we know today, many of these regulations of agriculture also had significant impacts on P. Law-enforced regulations like restrictions on animal density on agricultural land, norms for how much N (in manure and fertilizer) can be given to each crop, rules for calculation of available N in manures and mandatory nutrient budgets for control improved the distribution and utilization of N as well as P in manures and fertilizers. Even though a surplus of P still can be added in areas with the highest animal densities, an upper limit of the P surplus was indirectly introduced with these

regulations. Furthermore, the demand for 9 months storage capacity for animal manure made it possible to shift from autumn application towards spring application. This resulted in further improvements of nutrient utilization and most probably also a reduction in the manure-P losses through surface runoff and runoff through macropores to tile drains in the wet winter season. Manure application methods introduced to reduce ammonia volatilization (application with trail hoses and direct injection into the soil and demands that manure applied on bare soil has to be incorporated within 6 hours after application) probably decreased these losses even further.

A combination of increased awareness of the potential environmental risks and the fear of future regulation inspired pork producers to begin implementing new feeding strategies, with less mineral P supply, even before this was encouraged though a new tax on feed phosphates implemented in 2004. The P excretion from pigs for slaughtering declined from 1.05 kg P per animal in 1985 to 0.72 kg P per animal in 2000, due to new feeding recommendations and phasing out feed phosphates with low P availability. It declined further to 0.62 kg P/animal in 2002 due to the substitution of inorganic feed phosphates with phytase to increase phytate digestibility (Poulsen and Rubæk, 2005).

All in all, 20 years of environmental regulation of Danish agriculture have resulted in a decline in the national P surplus from 23 kg P ha⁻¹ in 1985 to 13 kg ha⁻¹ in 2002, and an increase in the utilization P from 32% in 1985 to 52 % in 2002. In spite of this, the agricultural contribution of P to surface waters has remained constant (~0.5 kg P ha⁻¹). Reasons for this might be: (1) the considerable delays in many of the expected effects, (2) it is a too short time period to track temporal trends in agricultural contribution to the P load in surface waters or (3) that the negative effect of the P surplus at the moment overwhelms the positive effect of improved nutrient management. New initiatives are therefore looked for to reduce P loss from Danish agriculture. The most recent and the probable future initiatives include: (1) introduction of 10 m buffer zones along streams and lakes, (2) research and development regarding low-P feeding strategies and regarding targeted mitigation in areas having high risk for P loss (3) improved management of streams and extensification of the cultivation in areas neighboring surface water, (4) processing of animal manures, such as slurry separation, utilization for energy production in biogas or combustion plants.

b) Chesapeake Bay

The Chesapeake Bay is a huge estuary on the eastern sea board of the US, with a watershed of 64,000 square miles (166,000 km²) stretching across 6 states (New York, Pennsylvania, Maryland, Delaware, Virginia and West Virginia). Due to concerns about rapidly decreasing water quality in the Chesapeake Bay, the Chesapeake Bay Commission was formed and set nutrient and sediment reduction

targets to the Chesapeake Bay to meet Clean Water Act standards. The Commission recently published a report which outlined six cost effective strategies to reduce nutrient inputs to the Bay, four of which related to agricultural practices aimed at P reductions (CBC, 2004):

- 1) Diet and feed adjustments to decrease dietary and hence manure P. This has been mandated for poultry in Maryland, which drove implementation both there and in neighboring states due to the integrated nature of the poultry industry. Data available in Delaware shows that the P concentration in poultry litter has decreased 30-40% as a result of improved diets. Full implementation could prevent 100 Mg P yr⁻¹ entering the Bay with no anticipated costs.
- 2) Nutrient management plans (NMPs), which represent the most common approach to control P in the watershed. The NMPs prescribe the rate and timing for fertilizer and manure applications to eradicate excess applications. Full implementation of NMPs could prevent 363 Mg P yr⁻¹ from entering the Bay at a cost of \$62 per kg.
- 3) Enhanced nutrient management, where farmers apply 15% less nutrients than recommended for full yield and receive compensation for yield reductions. Full implementation could prevent 363 Mg P yr⁻¹ from entering the Bay at a cost of \$210 per kg.
- 4) Conservation tillage that minimizes soil disturbance and maintains cover crops and crop residues to minimize soil erosion. This was estimated to be the most attractive option, with the potential to prevent 1,177 Mg P yr⁻¹ entering the Bay at no additional cost.

Implementation of these strategies has varied by state, but between 1985 to 2002 it is estimated that P loads to the Bay decreased from 12,318 Mg P yr⁻¹ to 8,863 Mg P yr⁻¹, with further reductions (to 5,818 Mg P yr⁻¹) needed to meet Clean Water Act standards. Additional efforts are underway to implement other agricultural best management practices, with much regional variability. These include fencing animals out of streams to avoid direct deposition of manure and stream bank erosion, dietary P reductions in non-poultry such as swine and dairy, and alternate uses of manures such as burning for energy production.

References

- CBC (Chesapeake Bay Commission), 2004. Cost effective strategies for the bay: 6 smart investments for nutrient and sediment reductions.
www.chesbay.state.va.us/Cost_Reports.htm
- Poulsen, H.D. & Rubæk, G.H., 2005. Fosfor i dansk landbrug. Omsætning, tab og virkemidler mod tab. DJF rapport - Husdyrbrug 68, 163-182.
<http://web.agrsci.dk/djfpublikation/djfpdf/djfh68.pdf>

Dynamic watershed-scale phosphorus models: their usages, scales, and uncertainties

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Introduction

Much progress has been made in the last three decades in modeling pollutant transport. Today, there are many well-established computer models that are being used at different spatial and temporal scales to describe water, sediment, and phosphorus (P) transport. When these models are used properly, they can provide new information, but validation of models is still a problem. In this review, we describe how dynamic watershed-scale P models are being used in the US and Europe, the challenge presented by different temporal and spatial scales, and the uncertainty in model predictions.

How models are being used

Models of diffuse P transport are being used in a number of important ways. In the US, the federal Clean Water Act requires that water quality standards be set by states and water bodies monitored to determine if they are meeting these standards. For water bodies that do not meet these standards, a total maximum daily load (TMDL) of the pollutant of concern must be set and a plan implemented to reduce the current loads to meet the TMDL. Models are used to determine the current maximum daily load and annual average load by interpolating among sparse data sets that usually provide, at best, weekly-to-monthly samples of P concentration, much of which is taken under base flow conditions. Models are also used to estimate the percent contribution from non-point sources and how that contribution is divided up among different non-point sources. However, there is a great deal of uncertainty in quantifying this distribution among non-point sources. The final step is to use models to develop different scenarios for reducing the current daily or annual load to the TMDL target, taking into account costs and the time that will be required to achieve the load reduction. Throughout the process, models with modern graphical displays, are being used as a tool for communicating with stakeholders in meetings and via websites. Outside the TMDL arena, models are playing a role in regional efforts to reduce P loading to critical water bodies such as the Chesapeake Bay and the New York City water supply system. Models have even been used as evidence in court cases such as the conflict between Arkansas and Oklahoma over P loading to drinking water reservoirs.

In Europe, the EC-Water Framework Directive (WFD; Directive 2000/60/EC) is one of the most important international driving forces to improve water quality, since 20% of all surface waters are seriously threatened with pollution and 60% of European cities overexploit their groundwater resources. Models will play a key role in the implementation of the various directives affecting P applications and losses. Indeed the WFD requires Member States to perform an impact analysis, often done through modeling of various degrees of sophistication. Models will also play a key role in the evaluation of the river basin management plans that Member States have to develop in order to reach good chemical and ecological status by 2015. In the mean time large efforts and resources are being put into developing monitoring networks to assess the actual status of the water bodies and evaluate the efficiency of the measures taken by the different Member States. Models are being used to optimize the design of these networks and interpolate between data. The WFD requires Member States to manage at the watershed level and models are being used to understand processes and pathways operating at such large scales and also to provide an integrated response both in terms of resources (soil and water) and disciplines (ecology, hydrology, economics, etc.). Many types of models are being used and these were reviewed in the EU-EUROHARP-project (Schoumans and Silgram, 2003).

Model scale issues

In general, models can be differentiated in terms of the spatial and temporal scales for which they are best suited. In recent years, there has been a trend to scale up models from small catchments to large contributing watersheds of critical lakes, reservoirs, and estuaries. Most of these models are based on an implicit assumption that small-scale processes can be scaled up by using effective parameter values obtained through calibration (Kirchner, 2006). This has led to over-parameterized models that cannot be used to test hypotheses regarding non-point sources of P or transport processes using the monitoring data that is typically available. There is a need to develop models designed for the large watershed scale with fewer parameters and to design monitoring programs to test these models. However, selecting the most important processes and transport routes for the simplified models will require new thinking. Furthermore, lumping or eliminating processes and pathways may limit the ability of the model to describe other scenarios. As models move to large-scale watersheds, in-stream processes become more important and the current P models differ substantially in how they describe these processes. Temporal scales are also important and one role models may play is in showing the long response time that may be required to see improvements in water quality where P is the pollutant.

Model uncertainty

One of the reasons for a degree of skepticism among peer scientists and the public in regard to the use of models is the lack of any measure of the certainty (confidence limits) on model predictions. Despite the progress in developing new methods for quantifying model uncertainty, these methods are seldom used by modelers. Tools for measuring model uncertainty, by whatever method as long as the method is clearly identified, must become an integral part of models and be readily available for model users (Pappenberger and Beven, 2006). Progress is being made along these lines in that both SWAT and HSPF now have autocalibration tools available through interfaces.

The future

There are a number of ways in which dynamic P models can be improved, beyond those already mentioned. Progress is being made in identifying critical source areas and this must be incorporated into models. A manure pool, separate from other soil P pools, is required for P when manure is applied to pasture. Better modeling of leaching of all P forms is needed (Schoumans and Chardon, 2003). Long-term nested watershed monitoring experiments for the calibration and validation of models is essential. More process-based modeling and monitoring of the effect of BMPs and combinations of BMPs at the local and watershed-scale is essential. In the near future, questions will arise regarding the impact of climate change on water quality. From that point of view, many model processes need to be reconsidered, for example the snow melting process in the mountains and oxidation/reduction processes during long rewetting conditions in low lands. Looking longer range, we need to ask a number of questions:

- What should a 'learning framework' for P modeling and prediction look like? Should we go about building models in a completely different way?
- How can we improve the way we evaluate models with limited data? Can we use 'soft information' or other ways of constraining models?
- Will any emergent technologies help us improve our ability to model and if so at what scales?

References

- Kirchner, J.W., 2006. Getting the right answers for the right reasons: Linking measurements, analyses, and models to advance the science of hydrology. *Water Resources Research*. 42, W03S04.
- Pappenberger, F. & Beven, K.J., 2006. Ignorance is bliss: Or seven reasons not to use uncertainty analysis. *Water Resources Research*. 42. W05302.
- Schoumans, O.F. & Chardon, W.J., 2003. Risk assessment methodologies for predicting phosphorus losses. *Journal of plant nutrition and soil science* 166(4; aug), 403-408.
- Schoumans, O.F. & Silgram, M., 2003. "Review and Literature Evaluation of Quantification tools for the assessment of Nutrient Losses at catchment scale" Oslo (Norway): NIVA, 2003 (EUROHARP Report 1-2003) - ISBN 82-557-4411-5 - p. 120.

Farmers and mitigation options: economic and practical constraints

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Introduction

The Water Framework Directive (WFD) requires that programmes of measures are established to control point and diffuse sources of pollutants in river basin districts to ensure the successful attainment of good ecological status and maintenance of high ecological status in EU waters. The WFD also requires that a cost-effectiveness analysis is undertaken prior to implementation to ensure uptake of least-cost combinations of measures. This process will require the integration of cost-effective mitigation options to control phosphorus (P) loss from diffuse sources, since agriculture has been identified as a significant contributor to eutrophication in many EU waters. Measures are likely to include a combination of regulation, advice and incentive and uptake of mitigation options will depend on farmer motivation and acceptability including the extent of any economic and practical constraints. Recent initiatives in England and Wales to provide targeted advice and support to farmers to encourage voluntary change in farming practices and help them cope with the WFD have provided some useful insights into the main constraints and barriers to uptake.

Awareness and attitudes

One of the main barriers to uptake of measures is a general lack of understanding by farmers and their advisers of diffuse pollution issues, specific water quality problems within their area and how farming practices can increase P transfer to levels that might be environmentally damaging. This is accompanied by a lack of personal responsibility for their individual farm management actions and lack of collective responsibility for water quality in the catchment. Surveys suggest that 'concern for the environment' is a major motivator for farmers to adopt more sensitive management practices. An increased awareness and knowledge of the P loss process should therefore lead to more considerate attitudes towards the need for voluntary management change and provide consultants with greater confidence in advising farmers to implement appropriate mitigation options (Carter et al., 2006). Allied to this is a requirement by water regulators and conservation bodies to raise general awareness of water quality issues at the local level. One can't expect farmers to adopt voluntary change if they don't know what the problem is.

Good communication with farmers is therefore essential for improving understanding of water quality issues and persuading farmers to take action in those high-risk areas which are causing a problem. Although more resource demanding and costly, one-to-one advice and on-farm small group discussion focusing on practicalities and

demonstration has been most successful in England and Wales. The necessity to implement basic soil resource protection under Cross-Compliance in order to receive the EU Single Income Payment (SIP) has also greatly helped farmers engage on diffuse pollution issues. Legislation remains a powerful motivator.

Accurate targeting of sources

Rural catchments may contain a complex range of P sources including not only surface and sub-surface runoff from farmed fields but also road runoff, farmyard runoff and point source discharges from village sewage treatment works (STW) and septic tanks. An example of the array of potential sources in a small (<10 km²) rural catchment (Kivernoll, River Wye, UK) is shown in Fig.1. The agricultural land use is considered to be high risk due to either a build-up of soil P associated with poultry manure applications or high sediment loss risk associated with winter cereals, or where both high soil P and erosion risk occur together as in the potato crop. However, there are at least 7 points where roads cross the stream network and a number of ‘pipes’ discharging directly into the stream from sources that are far more concentrated in soluble P than land runoff.

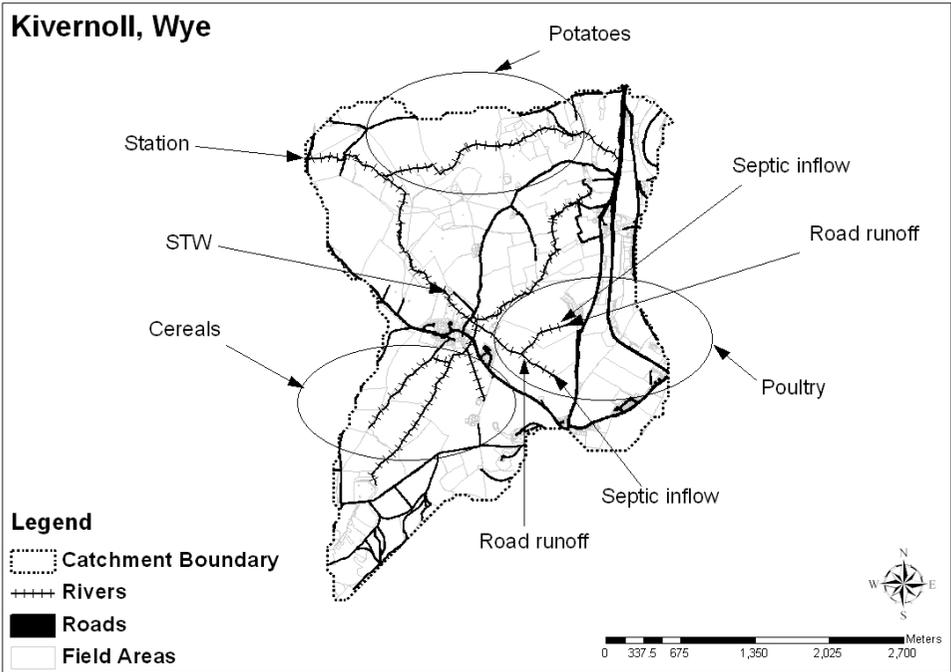


Figure 1. A variety of phosphorus sources occur within a small (10 km²) agricultural subcatchment (Kivernoll) of the River Wye Herefordshire.

Clear separation of the relative contribution of the different sources to eutrophication is problematical without intensive monitoring but clearly necessary for accurate

targeting of control measures. Farmers are therefore not always to blame for water quality problems in small rural catchments and misguided targeting of options will not only be ineffective in solving the problem but further undermine the confidence that farmers have in water regulators. A lack of farmer confidence in so-called expert solutions has been highlighted as an important barrier to uptake of mitigation options.

Some practical constraints

Many surveys have highlighted a range of practical constraints associated with the adoption of mitigation options. These include a lack of time or skilled labour to make the necessary management changes, potential conflicts with the requirements of other schemes or regulations (e.g. Nitrate Vulnerable Zones) and a lack of expertise where new management techniques or land uses are recommended. Some options are just not practical to implement every year (e.g. incorporation of P fertiliser before sowing in a busy autumn) or create additional weed, disease and pest burdens (e.g. leaving cloddy seedbeds and in-field buffer strips) or require a major change in management structure (phase feeding of livestock). Some practical issues are not immediately apparent. Fencing off river banks from livestock clearly requires provision of separate water supply facilities but may also lead to weed ingress and increased lamb mortality where banks provide shelter.

Some economic constraints

The cost of diffuse pollution (e.g. P loss) control on the farm is the most important barrier to farmer uptake of mitigation options. The major economic constraints include any adverse effects on farm profitability (for example by taking high-risk land out of production or reducing stocking rates) and the often substantial costs of containing the pollution on the farm (e.g. providing additional manure storage or constructing a wetland). Whilst some financial assistance is usually available through countryside stewardship schemes to encourage uptake of the less costly options, farmers still have to bear a significant proportion of the costs of the more expensive options. Surveys suggest that farmers are not willing to implement those options that create an unacceptable economic burden.

Conclusions

Farmers need clearer guidance on diffuse pollution issues and how their farming practices might be contributing to local water quality problems. In turn this requires better apportionment of sources and further research to fully understand their ecological impacts. With greater understanding of the problem, and their acknowledged concern for the environment, UK experience suggests farmers are amenable to change. Voluntary uptake of P mitigation options is more likely to be acceptable if the practical and economic difficulties are adequately identified and overcome. Persistent problem farms may also require increased regulation.

Reference

Carter, A.D., Groves, S., Bailey, R. & MacLeod, J., 2006. Environment sensitive farming – practical advice for land managers. In: Gairns, L., Crighton, K., Jeffrey, B. (eds.), *Agriculture and the Environment VI - Managing Rural Diffuse Pollution*. Proceedings of the SAC and SEPA Biennial Conference, Edinburgh, 5-6 April 2006, pp. 137-143.

Prioritising mitigation methods for diffuse pollution from agriculture by estimating cost and effectiveness at the national scale

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Pressures for meeting legislative requirements such as the European Union's Water Framework Directive are forcing governments to make priorities for mitigation of diffuse pollution, even where existing knowledge can be extremely uncertain. In this paper we examine approaches that are being adopted at the national scales to help guide governments towards policies that may help them meet EU water quality targets and examine both the uncertainties and benefits of the process. Two examples are used. The first concentrates on mitigating agricultural phosphorus loss from England and Wales. In the second example, we show how the phosphorus approach was expanded to involve a wider range of diffuse pollutants across a range of agricultural systems. In both these examples, model farm systems were constructed so that both the economic cost and the effectiveness of mitigation (in terms of mass of pollutant abated) were estimated, in order to provide the potential for determining the effectiveness of policy instruments for different bioclimatic regions.

Potential mitigation options for a typical range of farm systems in England and Wales were collated to assess their potential effectiveness (in reducing mass of phosphorus (P) transferred) and potential cost (in terms of GB pounds sterling £) to the farming industry. This was achieved by plotting 'cost curves' for prioritising cost and effectiveness, to help planners and scientists gauge where priority areas for further investment and research are required. A simple conceptual model framework, incorporating a number of assumptions identified 15 methods for mitigating inputs of phosphorus to agricultural systems, 19 methods for preventing mobilisation of phosphorus and 6 methods for controlling the transport of phosphorus to streams. In all cases, cost and effectiveness were determined. The potential for largest reduction in phosphorus *inputs* was with grassland and horticulture, reflecting the preferential accumulation of phosphorus at the surface in grassland soils and the use of short season succession cropping in horticultural systems. In one example, estimated potential reductions in phosphorus *mobilisation* by using cover cropping with spring planted roots and vegetables, reduced estimated losses from 3.9 to 2.8 kg P ha⁻¹. Reductions in phosphorus transfer associated with *transport* mitigation were larger than those associated with input and mobilisation methods and ranged

up to 2.2 kg P ha⁻¹ (a reduction from 2.7 to 0.5 P ha⁻¹). These reductions were the largest overall and achieved by installing buffer zones and were also very cost effective (£3-5 kg⁻¹ P saved). When combining methods in 'cost curve' analysis, the cumulative P transfer reductions attainable by the methods selected were ca. 0.2 kg ha⁻¹ for upland systems, 0.6 kg ha⁻¹ for outdoor pigs, 0.9 kg ha⁻¹ for intensive dairy systems and 2.2 kg/ha⁻¹ for arable systems.

In a more recent project from the UK, the approach used for phosphorus was expanded to embrace a wider range of pollutants, including nitrate, sediment and faecal indicator organisms. The purpose of doing this was to determine a set of common mitigation options and to be able to assess their cost and mitigation potential in a common and integrated manner. This has provided the UK government with a useful tool for highlighting where diffuse pollution may be tackled *a priori* and in a most cost effective manner. The work has also helped guide policy decisions towards packages of mitigation methods that might be used to make policy instruments. On the face of it, this task can be achieved with relative ease within an appropriate theoretical modelling framework. However, it should be noted that the evidence base to underpin this framework was found to be sometimes of low substance as pollutant, farm system and area-specific evidence to develop and test the process can be difficult to find.

The approach of adopting mitigation and cost modelling frameworks for helping policy planners target priorities for mitigating diffuse water pollution is a useful 'top down' exercise. It can yield useful information for planning priorities where and how best options might be most effectively targeted and for least cost but of greatest potential benefit. The approach also highlights potential diffuse water pollution 'trade-offs', and with future development could link to impacts on gaseous emissions. However, in conducting this policy relevant exercise, it is also evident that there is a notable absence of robust, locality-specific evidence for how mitigation methods work in the field under a range of conditions. We conclude that these modelling studies are required at the science policy interface and are a first step in the important activity of establishing what we know (and what we do not know) about the performance and cost (and affordability and likelihood of implementation). The future applied research agenda can in part be set to address the most sensitive uncertainties.

Acknowledgements

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Phosphorus and ecological conditions in freshwaters in a climate change perspective

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Introduction and results

Climate change may have a profound effect on phosphorus (P) processes in lakes and lowland streams. Higher temperature enhances phosphorus release from sediment due to enhanced oxygen consumption and consequently redox conditions, which in turn result in release of iron-bound P. In streams, lower oxygen concentrations during the night as a result of higher respiration and lower oxygen saturation levels induced by higher temperatures may further amplify this process. Moreover, higher temperatures will increase the turnover rate of organic matter and hence also the bio-availability of nutrients. Changes in precipitation affect phosphorus losses to streams and lakes.

Winter precipitation and runoff is expected to increase in Denmark (Table 1) and the diffuse P loss is consequently expected to increase in such northern, temperate coastal streams, not least during winter and possibly be reduced in warm temperate streams. Despite reduced P loadings in arid systems, P concentrations may increase due to lower water tables in both lakes and streams and faster removal of the available oxygen in the water.

Table 1. Simulated change in average annual precipitation, runoff and diffuse P loss from the control period 1961-1990 to the scenario period 2071-2100 for Danish catchments. The climate simulations are based on the regional HIRHAM model with an a2 emission scenario, runoff is simulated with a precipitation runoff model and diffuse P loss with an empirical model.

	Precipitation (range)	Runoff (range)	Diffuse P loss
Change from control to scenario period	55 mm (24-74 mm)	18 % (11-33 %)	8 % (3-17 %)

For lakes it is expected that climate change will lead to enhanced eutrophication of nutrient-enriched lakes and higher risk of prolonged blooming of cyanobacteria. Shifts in the fish community structure towards small and abundant plankti-benthivorous fish enhance predator control of zooplankton, which in turn increases phytoplankton production and sedimentation, and thus the risk of P release from the

sediment. The critical P loading to obtain a good ecological state in lakes therefore likely has to be lower in a future warmer climate.

The effect of changes in P concentrations in streams is more debatable as physical and biological effects of changes in water table and flow may override the nutrient effect and therefore vary substantially among stream systems. Droughts, floods and the following erosion and sedimentation processes will be increased in both frequency and magnitude as a consequence of climate change and these processes will in many river systems play a key role for the P transport at river basin scale. Moreover, human interventions in the form of increased drainage, building of flood control structures and restoration of river, lakes and floodplains can in many cases be as important for P and freshwater ecology as climate change impacts.

Continuous monitoring to assess phosphorus dynamics and ecological status in the River Kennet, UK

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Introduction

The links between ecological status of lowland rivers and diffuse and point-source nutrient inputs are of key scientific importance under the EU Water Framework Directive (WFD) (EU, 2000). Many previous studies on the sources and effects of phosphorus in rivers have been based on weekly, storm event or even monthly sampling. However phosphorus concentrations are sensitive to a wide range of short-term dynamics and diurnal cycles that are not captured by conventional sampling programmes. Hence phosphorus loads may be underestimated and valuable information about the variability in P concentrations and links with in-stream processes and ecology may be missed. In this study, we use *in-situ* 'continuous' (hourly) measurements of total reactive phosphorus (TRP; unfiltered molybdate-reactive P), chlorophyll, dissolved oxygen (DO), conductivity, turbidity and flow to examine dynamics in P concentrations and sources and ecological status for the River Kennet, a lowland chalk river in south east England.

Methods

TRP measurements were taken at hourly intervals and analysed by an *in-situ* Nutrient Probe Analyser (NPA, Systea, Italy). Dissolved oxygen, pH, conductivity, turbidity and chlorophyll measurements were also taken at hourly intervals and analysed using a YSI6600 multiparameter probe. pH was used to calculate excess carbon dioxide pressure (Neal *et al.*, 1998), and diurnal variations in dissolved oxygen were used to estimate rate of photosynthesis and respiration, by the methods of Williams *et al.* (2000). These parameters were examined alongside hourly river flow measurements.

Influence of point and diffuse sources on TRP dynamics

Seasonal, diurnal and storm event patterns in TRP concentrations were observed. Daily average TRP concentrations were typically below the 'threshold' concentration of 100 µg/l (delineating the boundary between 'Moderate' and 'Good' ecological status (UKTAG, 2006)) for most of the year, with daily average TRP concentrations of c. 60-80 µg/l. However daily average TRP concentrations increased to between 120-150 µg/l during the summer, and in October and November 2005. There was a close temporal linkage between the time series of daily average TRP concentrations and river flow: TRP decreased as river flow increased, as a result of dilution of point-source P inputs.

Diurnal variability of TRP was typically 30-50 $\mu\text{g-P/l}$, but reached as much as 150 $\mu\text{g-P/l}$ over the summer months (Figure 1). Daily TRP maxima occurred at around 14:00h. A double diurnal pattern was seen during the summer months, with a secondary peak occurring at 02:00h. This reflects typical patterns in domestic water usage and hence the diurnal signal in TRP is most likely due to the daily variations in effluent discharged from a sewage treatment works (STW) located approximately 3km upstream of the monitoring station. Sharp increases in TRP concentrations above the diurnal concentration patterns corresponded with storm events in the flow hydrograph, as well as peaks in turbidity and either peaks or dips in conductivity. These event-related increases in TRP concentration are attributed to diffuse sources, either from P-associated sediment transport from the land surface, and/or in-stream resuspension of sediment.

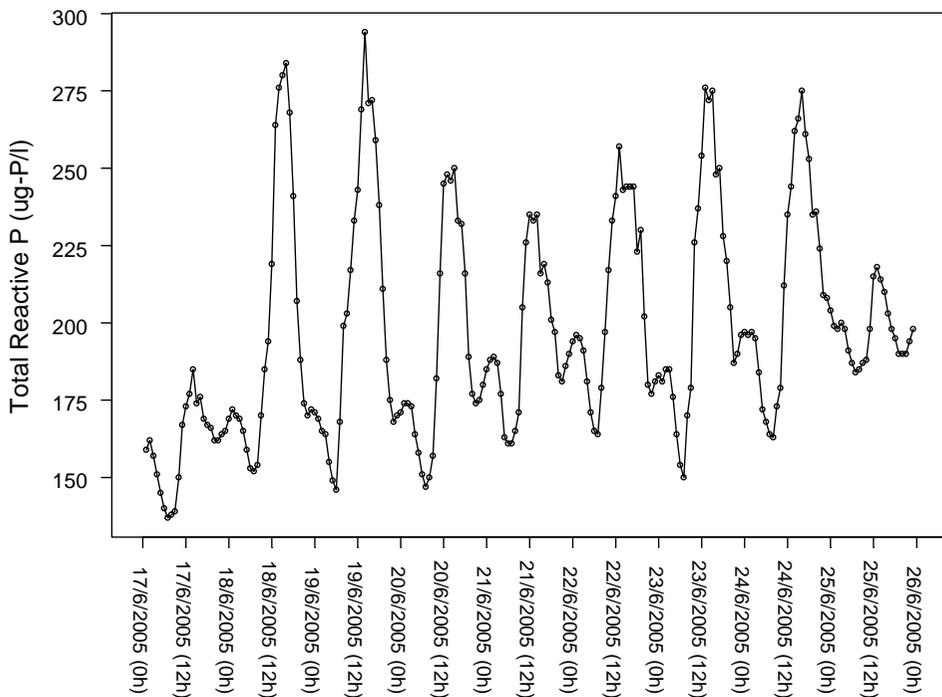


Figure 1. Diurnal patterns in Total Reactive Phosphorus (TRP) from hourly in-situ measurements during June 2005.

Phosphorus and ecological status

There was no clear link between periods of elevated TRP concentrations and peaks in chlorophyll concentration or photosynthesis rates. A spring phytoplankton bloom, corresponding with elevated chlorophyll concentrations occurred between March and May. Rates of photosynthesis and respiration also started to increase around March,

but remained at elevated levels until August and October respectively (well beyond the period of high chlorophyll concentrations). This indicates that in-stream productivity during the summer months was not primarily driven by phytoplankton, but that other autotrophs such as periphyton and macrophytes play an important role in overall river primary productivity.

Conclusions and wider comments

Diurnal cycling and storm event associated variations in TRP concentrations were observed, which would have been missed by a routine weekly sampling regime. These hourly TRP data, in conjunction with other water quality measurements, enable us to examine the sources and dynamics in TRP in the River Kennet under varying hydrological conditions. The major source of phosphorus under base flow appears to be from sewage effluent, with varying inputs from diffuse sources during storm events. The results of this monitoring indicate that TRP concentrations are not a major control on phytoplankton growth or on overall primary productivity in the River Kennet. Indeed, the data indicate that in-stream productivity in the Kennet is more closely related to light levels and that, after a spring phytoplankton bloom, the primary productivity in the Kennet is linked to growth of other aquatic plants (such as periphyton and/or macrophytes). This study shows that continuous monitoring of phosphorus and associated parameters can provide valuable information about the dynamics and sources of nutrients and general river ecological status. We highlight the need to integrate hydrochemical monitoring with wider biological monitoring, including phytoplankton, periphyton and macrophytes, when dealing with such complex and dynamic aquatic systems.

References

- EU (European Parliament and the Council of the European Union, 2000), 2000. Directive 2000/60/EC. Establishing a framework for community action in the field of water policy. Official Journal L 327, 22/12/2000, p. 1-73.
- Neal, C., House, W.A. & Down, K., 1998. An assessment of excess carbon dioxide partial pressures in natural waters based on pH and alkalinity measurements. *Sci. Tot. Environ.* 210/211, 173–86.
- Williams, R.J., White, C., Harrow, M.L. & Neal, C., 2000. Temporal and Small Scale Spatial Variations of Dissolved Oxygen in the Rivers Thames, Pang and Kennet, UK. *Sci. Tot. Environ.* 251/252, 477–495.
- UKTAG (Technical Advisory Group on the Water Framework Directive), 2006. Environmental Standards and Conditions (Phase 1). www.wfduk.org

Periphyton biomass response to changing phosphorus concentrations in a nutrient-impacted river: a new methodology for P target setting

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Introduction

Much effort is currently being focused on reducing the phosphorus and nitrogen loadings in nutrient-impacted freshwaters, to improve environmental status and reduce the risk of eutrophication. However, the resulting step-decreases in phosphorus concentrations often have little effect on river biology (Kelly and Wilson, 2004), suggesting that P concentrations are not limiting or co-limiting algal growth in many rivers. It is important to focus remediation efforts on rivers that will show an observable environmental improvement.

Current in-stream nutrient amendment methodologies, such as nutrient diffusing substrata and stream nutrient enrichments, are only capable of producing increases in river P concentrations. Therefore, they are only able to investigate the effect of algal nutrient limitation in relatively non-impacted rivers. The objective of this study was to develop a methodology that could simultaneously assess the effects of increasing and decreasing soluble reactive P (SRP) concentrations on algal biomass. This ability to reduce phosphorus concentrations (P-stripping) allows nutrient limitation to be investigated in rivers where P is currently in excess.

Experimental design

Experiments were conducted in 10 streamside flumes, fed directly by the mesotrophic River Frome, Dorset, UK, during the summer of 2005. The SRP concentration of the incoming river water in each flume was either increased (by phosphorus addition), decreased (by P stripping using FeSO_4 additions), or left unaltered (control), producing SRP concentrations ranging from $32 \mu\text{g L}^{-1}$ to $420 \mu\text{g L}^{-1}$. After nine days, the periphyton biomass on slate substrates in each flume were quantified (using chlorophyll-a analysis), to determine the effect that these varying P concentrations had on periphyton growth rates.

Results

An 80% increase in ambient SRP concentration ($200 \mu\text{g L}^{-1}$) did not increase the quantity of epilithic algal biomass, showing that the River Frome was not P limited at

109 $\mu\text{g SRP L}^{-1}$ (Figure 1). In the P-stripped flumes, algal biomass declined as the SRP concentration fell below $\sim 90 \mu\text{g L}^{-1}$, with a $\sim 60\%$ reduction in algal biomass at $\sim 40 \mu\text{g SRP L}^{-1}$. Phosphorus diffusing periphytometers deployed in the P-stripped flumes confirmed that the reduced rates of algal growth were due to P-limitation, rather than a physical affect of FeSO_4 addition.

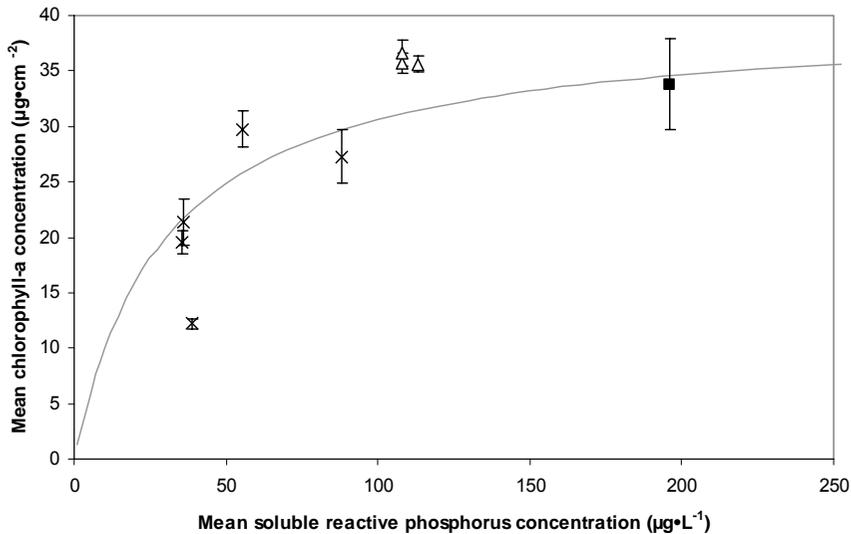


Figure 1. Mean chlorophyll-a concentrations on slate substrates at the end of Experiment 2. x = FeSO_4 addition. Δ = control. \blacksquare = phosphorus addition. Error bars = ± 2 standard deviations, derived from the three sub-samples analysed from each slate.

Conclusions

Epilithic algal growth in the River Frome was not phosphorus-limited throughout the summer of 2005. The P-stripping methodology showed that periphyton become P-limited at an SRP concentration of $\sim 90 \mu\text{g L}^{-1}$, and periphyton concentrations would be halved if the SRP concentration was reduced to $\sim 60 \mu\text{g L}^{-1}$, which would be equivalent to introducing tertiary water treatment at two of the catchment's largest sewage treatment works.

Determining the SRP concentration at which algae become growth-limited is vital for effective eutrophication management, as this provides the basis for nutrient target setting within a catchment. Previous studies have shown that this target SRP concentration varies greatly between study areas, ranging from $3 \mu\text{g L}^{-1}$ (Scrimgeour and Chambers, 1997) to $> 110 \mu\text{g L}^{-1}$ (Matlock et al., 1999). As the cost of introducing nutrient remediation measures to a catchment is closely dependent on the phosphorus target concentration that is set, it is vital that target setting is

'knowledge-based' for individual catchments. The new methodology presented in this paper could greatly assist catchment managers and policy makers in setting realistic phosphorus targets for individual nutrient-impacted rivers, allow them to identify what specific P load reduction is required before an environmental improvement will be observed, and should quantify reductions in algal biomass that would result from various levels of remediation.

References

- Kelly, M.G. & Wilson, S., 2004. Effect of phosphorus stripping on water chemistry and diatom ecology in an eastern lowland river. *Water Research* 38(6), 1559-1567.
- Matlock, M.D., Storm, D.E., Smolen, M.D. & Matlock, M.E., 1999. Determining the lotic ecosystem nutrient and trophic status of three streams in eastern Oklahoma over two seasons. *Aquatic Ecosystem Health and Management* 2, 115-127.
- Scrimgeour, G.J. & Chambers, P.A., 1997. Development and application of a nutrient-diffusing bioassay for large rivers. *Freshw. Biol.* 38(1), 221-231.

Defining phosphorus concentrations for maintenance of good ecological condition of agricultural streams

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Abstract

Inputs of nutrients (phosphorus, P, and nitrogen, N) to fresh waters can result in excessive aquatic plant growth, depletion of oxygen, and deleterious changes in abundance and diversity of aquatic invertebrates and fish. As part of a “National Agri-Environmental Standards Initiative”, the Government of Canada committed to development of non-regulatory environmental performance standards that establish desired environmental quality (for nutrients, sediments, pathogen and ecological flows) for agricultural streams. For P, efforts focus on identifying biological indicators and targets and, from these, P standards consistent with desired environmental condition. Our research into standards development employed an integrated approach consisting of analysis of existing data from forested and agricultural watersheds combined with experimental studies in networks of 10-15 streams. Preliminary standards for P to maintain good ecological condition of agricultural streams in southern Ontario are 0.025 mg/L total P (TP) which should maintain suspended algae below ~1.4 µg/L chlorophyll *a* (chl_a) and benthic algae below ~50 mg chl_a/m². Research is continuing on refining indicators for greater sensitivity and ease of measurement, and improving approaches for defining aquatic plant targets indicative of desired environmental quality.

Introduction

Addition of nutrients, in the form of phosphorus (P) and nitrogen (N), to aquatic ecosystems promotes excessive growth of algae and rooted aquatic plants, a condition known as eutrophication. The aim of this study was to develop and test approaches for setting standards for P and N, as agents of eutrophication, in order to protect and provide suitable conditions for a diverse community of aquatic organisms in Canadian agricultural streams. This study forms part of a “National Agri-Environmental Standards Initiative” (NAESI), a program established by the Government of Canada to develop non-regulatory environmental performance standards to protect waters draining agricultural land.

Using data from southern Ontario and Quebec, Canada, we developed and tested several approaches for setting standards for P:

1. an empirical approach in which standards are based on a given percentile or other statistical descriptor of the dataset,
2. identification of critical junctures in the relationship between agricultural land cover in the watershed and P concentrations, and
3. identification of critical thresholds in relationships between aquatic plant abundance and P.

These analyses were undertaken using long-term (1971-2005) monitoring data from forested and agricultural watersheds, and new data (2005-2007) from 15 instrumented watersheds in agricultural and forested landscapes.

Methods

Three data sets were assembled for the Mixedwood Plains ecozone of southern Ontario and Quebec, Canada:

1. P concentrations from 117 stations on 101 streams ($\leq 75 \text{ km}^2$ in watershed area and $< 10\%$ urban land cover), from the Ontario Provincial Water Quality Monitoring Network.
2. aquatic plant biomass (suspended and benthic algae, and rooted plants) and P concentrations for 55 stations on 43 streams and rivers in Ontario and Quebec, from provincial databases and published studies.
3. concurrent monthly measures of P and aquatic plant biomass (suspended, benthic and filamentous algae) from 15 streams in agricultural and forested landscapes.

Results and discussion

Concentrations of P in forested and agricultural watersheds in southern Ontario and Quebec, Canada ranged from 0.007-0.71 mg/L TP (station means) and 0.001-0.85 mg/L soluble reactive P (SRP). Aquatic plants were generally prolific, with summer maxima of 12 $\mu\text{g chl}a/\text{L}$ for suspended algae and 243 $\text{mg}/\text{m}^2 \text{ chl}a$ for benthic algae. Up to 60% of the streambed was covered with benthic algae, rooted plants, or filamentous algae.

Comparison of empirical approaches for setting standards showed that two methods based on a given percentile of the data yielded similar results (Table 1). In the case of TP, empirical approaches yielded values of 0.020-0.026 mg/L for streams in southern Ontario. Plant abundance standards derived based on an empirical approach were 1 $\mu\text{g chl}a/\text{L}$ for suspended algae and 59 $\text{mg chl}a/\text{m}^2$ for benthic algae. Although this approach makes good use of available data, standards may be inflated due to inclusion of a high number of impaired streams.

Relationships between P and land cover were also evaluated for relevance in standards development: (1) linear regression was used to relate P concentrations or plant abundance to percent cropland cover in the watershed and the y-intercept was defined as the condition if zero cropland was present (Dodds and Oakes 2004), and (2) tree regression was used to identify a split in the X axis (i.e., percent cropland cover) which most significantly separated P concentration or plant abundance into two groups and a median was then calculated of the data in the lower range of cropland cover. These analyses resulted in values of 0.01 and 0.036 mg/L TP (Table 1). Although use of land cover versus nutrient relationships allows assessment as to whether specific P standards can be attained, the resulting standards are not linked to aquatic ecosystem health.

Table 1. Comparison of potential P standards developed for agricultural streams in Ontario, Canada using two approaches.

Parameter	Units	Empirical Approach			Land Use Approach	
		US EPA ¹		Australasia ²	Y-intercept ³	Split value
		75 th percentile of data for reference streams	25 th percentile of data for all streams	20 th percentile of data for all streams for biologically active seasons		
TP	mg/L	0.026	0.020	0.023	0.013	0.036
SRP	mg/L	0.0055	0.0033	0.0044	0.0023	0.004
Suspended algae	µg chl a /L		1.2		0.87	1.7
Benthic algae	mg chl a /m ²		58.7		42.5	51.2

¹US EPA 2000c, ² ANZECC 2000a,b, ³Dodds and Oakes 2004

Finally, analysis of aquatic plant data showed that suspended algae exhibited a linear response to TP whereas benthic algal abundance peaked at 0.040–0.050 mg/L TP and showed no response to higher concentrations while rooted plant abundance was not correlated with TP. To maintain aquatic plant growth below aesthetically and ecologically desirable limits of ~1.4 µg chl a /L and 50 mg chl a /m² benthic algae, our data indicate that TP should not exceed 0.025 mg/L.

Research on approaches for setting nutrient standards is continuing and focuses on analyzing data for other regions of Canada, and testing the efficacy of indicators of aquatic plant abundance, composition or production as biological standards. These and other performance standards will be used in efforts to develop and promote the adoption of beneficial agricultural management systems and other practices that reduce environmental risks, and as benchmarks to measure progress toward identified goals.

References

- ANZECC & ARMCANZ, 2000a. Australian and New Zealand guidelines for fresh and marine water quality. Volume 1. Australian and New Zealand Environment and Conservation Council, Agriculture and Resource Management Council of Australia and New Zealand.
- ANZECC & ARMCANZ, 2000b. Australian and New Zealand guidelines for fresh and marine water quality. Volume 2. Australian and New Zealand Environment and Conservation Council, Agriculture and Resource Management Council of Australia and New Zealand.
- Dodds, W.K. & Oakes, R.M., 2004. A technique for establishing reference nutrient concentrations across watersheds affected by humans. *Limnol. Oceanogr. Meth.* 2, 333–341.
- USEPA (U.S. Environmental Protection Agency), 2000. Nutrient criteria technical guidance manual: Rivers and streams. Office of Water, Office of Science and Technology. Washington, D.C.

The impact of trophic interactions on the recovery of Loch Leven after reduction in phosphorus loads

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Introduction

Loch Leven is a shallow eutrophic loch situated in the lowlands of south-east Scotland, UK. It has a catchment area of 145 km². Of this, almost 80% is agricultural land and a further 11% is woodland. The resident population of the catchment is just over 10,000 people, 10 per cent of whom live in rural (unsewered) areas (Frost, 1996).

The loch has become increasingly eutrophic over the last 100 years, mainly due to agricultural, industrial and sewage related activities within the catchment. By the mid-1980s, this had led to a marked deterioration in water quality which was shown to be closely linked to an increasing phosphorus (P) load (LLCMP, 1999). Over the next 10-15 years, identifying, quantifying and controlling these sources became the major focus of efforts to improve water quality in the loch and aid its ecological recovery. This paper describes the results of these activities and highlights the need to consider trophic interactions when implementing or interpreting ecological water quality targets.

Reduction in P load

The first step in addressing water quality issues at Loch Leven was to set restoration targets (LLAMAG, 1993). These were based on achieving the maximum macrophyte growing depth that had been recorded in 1910, i.e. 4.5 m. Corresponding targets for Secchi depth transparency, and in-lake P and chlorophyll_a concentrations were calculated from this value using relationships given by Chambers and Kalff (1985) and OECD (1982). These restoration targets are summarised below:

- o Mean annual total phosphorus concentration: 40 mg m⁻³
- o Mean annual chlorophyll_a concentration: 15 mg m⁻³
- o Mean annual water clarity (Secchi depth): 2.5 m
- o Maximum macrophytes growing depth: 4.5 m

The highest external P load that would allow these water quality targets to be met was estimated to be about 11 t y⁻¹ (WQWG, 1997). This meant that the annual P load of 20.53 tonnes, which was recorded in 1985 by Bailey-Watts and Kirika (1987), had to be reduced by about 50 per cent. This reduction was achieved over the next

10 years by removing a significant industrial source and upgrading several waste water treatment plants (Table 1). It should be noted, however, that the apparent reduction in P-laden runoff recorded during 1995, simply reflects the lower rainfall in that year in comparison with 1985.

Table 1. Estimated annual P load to Loch Leven before (1985) and after (1995) the reduction in P loads from point sources.

Source	Estimated annual P load (tonnes)	
	1985	1995
Runoff	8.13	4.34
Woollen mill	6.29	0
Sewage	5.32	3.09
Rainfall	0.42	0.42
Wildfowl	0.37	0.37
Fishery	0	0.19
Total	20.53	8.41

Impact on lake water quality

Mean annual in-lake P concentrations remained high at first due to an increase in P release from the sediments. As this process became less important, from the mid 1990s onwards, open water P concentrations began to fall towards the target value. Surprisingly, however, there was no corresponding decrease in mean annual in-lake chlorophyll_a concentrations.

The most likely explanation for this was that stocking with rainbow trout, which began in 1993, had reduced the number of phytoplankton grazers (primarily *Daphnia*) through predation. So, while phytoplankton productivity was falling due to reduced P availability, fewer algae were being lost through grazing. The net effect of this was of little or no change in algal biomass (i.e. chlorophyll_a). Historical records support this hypothesis by showing marked changes in the relationship between chlorophyll_a and P concentrations within the loch when *Daphnia* appeared after a long absence (1973) and when their numbers were reduced by fish stocking from 1993 onwards (Figure 1).

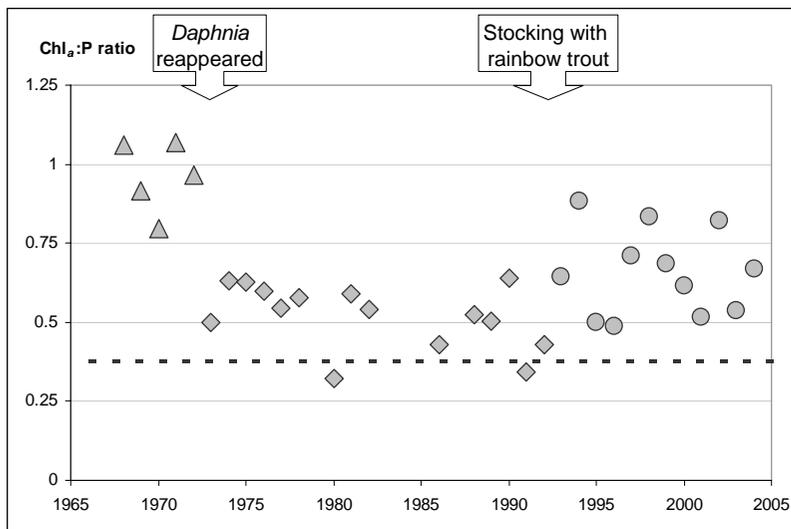


Figure 1. Impact of changes in *Daphnia* abundance on the chlorophylla:P ratio in Loch Leven; restoration target is shown by the horizontal dotted line.

Conclusions

The simple equations that were used to generate target chlorophylla and P concentrations for the loch did not take into account the effects of zooplankton grazing and fish predation on the relationship between these two parameters. The results of this study highlight the importance of taking a whole ecosystem approach to setting ecological water quality targets and looking for evidence of recovery.

References

- Bailey-Watts, A.E. & Kirika, A., 1987. A re-assessment of the phosphorus inputs to Loch Leven (Kinross, Scotland): rationale and an overview of results on instantaneous loadings with special reference to runoff. *Trans. R. Soc. Edinb., Earth Sci.* 78, 351-367.
- Chambers, P.A. & Kalff, J., 1985. Depth distribution and biomass of submersed aquatic macrophytes communities in relation to Secchi depth. *Canadian Journal of Fisheries and Aquatic Science* 42, 701-709.
- Frost, A., 1996. Loch Leven and diffuse pollution. In: Petchey, T., D'Arcy, B. and Frost, A. (eds.). *Diffuse Pollution and Agriculture. Proceedings of a conference held in Edinburgh, 14-14 April 1995.* Scottish Agricultural College, Edinburgh. Pp. 174-182.
- LLAMAG, 1993. Loch Leven: the Report of the Loch Leven Area Management Advisory Group. Forth River Purification Board, Edinburgh.
- LLCMP, 1999. Loch Leven Catchment Management Plan. 94 pp.
- OECD, 1982. *Eutrophication of Waters: monitoring, assessment and control.* Technical Report, Environmental Directorate, OECD, Paris.
- WQWG, 1997. Minutes of the Water Quality Working Group meeting, December 1997.

Annual variations in algal nutrient limitation at Lake Eucha, Oklahoma, 2003–2005

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Introduction

Lake Eucha in Oklahoma, USA has come into sharp legal, political, and environmental focus because of a past lawsuit between the municipal drinking water supply (City of Tulsa, Oklahoma – plaintiffs) and several poultry integrators and one municipal wastewater treatment plant (defendants) in Arkansas. The lawsuit was based on the claim that poultry industry via effluent discharge and nonpoint source P loading has elevated TP concentrations in Lake Eucha over last few decades, which coincided with increased drinking water costs related to elevated algal growth and the number and magnitude of taste and odor events in the finished drinking water. The lawsuit was settled out of court, resulting in the adoption of P management strategies to limit poultry litter application rates based on the Eucha–Spavinaw Phosphorus Index (ESPI) (Delaune et al., 2006). The purpose of this study was to evaluate the seasonal variation in algal nutrient limitation at Lake Eucha, and we focused on whether P and or N were limiting the growth of either reservoir phytoplankton and or periphytic algae.

Methods

We used floating enclosures to assess phytoplankton nutrient limitation and passive diffusion periphytometers to assess periphytic algal nutrient limitation at Lake Eucha from August 2003 through August 2005. The floating enclosures were filled with reservoir water, spiked with nutrient treatments (i.e., control, N, P and N+P) and left on site for less than one week, and the passive diffusion periphytometers with the same treatments were modified and constructed based on Matlock et al. (1998) and left floating near the air–water interface for 10 to 14 days during these experiments. These apparatuses were deployed 10 times and are shown below as deployed.



FAR LEFT: Deployed floating phytoplankton enclosure at the riverine zone at Lake Eucha.

LEFT: Deployed passive diffusion periphytometer [with an individual bottle pictured in the inset] near the dam at Lake Eucha.

Reservoir water samples were collected at least three times during each deployment, and water samples were analyzed for concentrations of soluble reactive P (SRP), nitrate-N ($\text{NO}_3\text{-N}$), ammonium-N ($\text{NH}_4\text{-N}$), total P (TP), total N (TN), and sestonic chlorophyll a (chl-a). Physico-chemical parameters (including pH, dissolved oxygen (DO), conductivity, and water temperature) were measured also. Water was collected from the enclosures at the end of the on-site incubation, then analyzed for chl-a. The growth substrate (i.e., glassfiber filter) from the periphytometer bottles was collected at the end of the deployment, and then analyzed for chl-a.

Results and conclusions

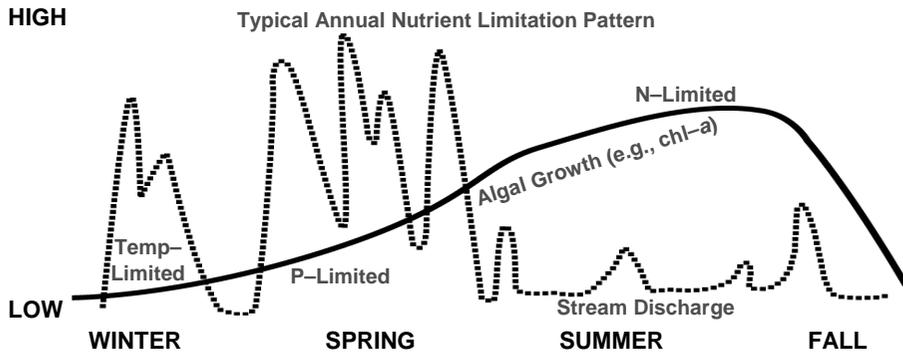
Concentrations of available nutrients varied widely, where SRP, $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ varied from <0.01 to 0.06 , <0.05 to 3.17 , and <0.02 to 0.42 mg L^{-1} . Nutrient availability was low (decreased concentrations) during Summer when the reservoir was stratified and nutrient availability was high (increased concentrations) during late Fall and Winter. Nutrient availability also varied spatially and was greater in the headwaters (i.e., riverine zone) compared to near the dam (i.e., lacustrine zone).

Sestonic algal growth (measured indirectly via chl-a) varied seasonally and spatially, as typically observed in other regional reservoirs; sestonic chl-a concentrations were greatest in late Summer and in the riverine zone. The control treatments within the phytoplankton enclosures typically had increased chl-a concentrations compared to that measured in the reservoir in close proximity to the enclosures. The enclosures maintain algal population near the water surface (i.e., within 1-m) and often slightly elevated temperatures which is the likely cause for increased algal growth in the enclosures.

The treatment results from the phytoplankton enclosures and periphytometers showed distinct temporal patterns in algal nutrient limitation and to a lesser degree some spatial differences between the riverine and lacustrine zone. The typical pattern would be phytoplankton are temperature limited during late Fall and Winter, P limited during Spring and early Summer and then shifting to N limitation in late Summer and early Fall. Thus, algal growth is likely limited by temperature, when nutrient availability is greatest following mixis, or turnover in this reservoir.

However, this typical annual pattern in algal nutrient limitation can be disrupted if and when large storm events occur during early to mid Summer. These storm events provide an advective supply of nutrients, particularly $\text{NO}_3\text{-N}$, to the epilimnion of a reservoir and increase nutrient availability to algae causing nutrient limitation pattern to shift. In fact, Lake Eucha experienced a rather large storm event one Summer which shifted the pattern to P limitation during Summer and to co-limitation during late Fall. Thus, seasonal fluctuation in nutrient transport from the catchment to the

reservoir has a profound impact of the individual nutrient limiting algal growth in water column.



Overall, phytoplankton and periphytic algae exhibited similar nutrient limitation characteristics over the study period. It was noted that periphytic algae were more often P or co-limited when phytoplankton were generally N limited during Summer.

Conclusions

- The algae at Lake Eucha switch from P to N limitation seasonally and in relation to large summer storm events which replenish the nutrient supply, particularly $\text{NO}_3\text{-N}$, in the epilimnion.
- Nutrient availability was greatest in the reservoir following mixis, but algal growth potential during this period was low because the water temperature was colder.

Acknowledgements

This project was a component of a larger project funded by the USDA Nutrient Science for Improved Watershed Management Program that developed the Nutrient Management Decision and Education Support System (NMD ESS) based on watershed modeling, reservoir hydrodynamic and water quality modeling, and stakeholder engagement, analysis and deliberation.

References

- Delaune, P.B., Haggard, B.E., Daniel, T.C., Chaubey, I. & Cochran, M.J., 2006. The Eucha / Spavinaw phosphorus index: A court mandated index for litter management. *Journal of Soil and Water Conservation* 61, 96–105.
- Matlock, M.D., Matlock, M.E., Storm, D.E., Smolen, M.D. & Henley, W.J., 1998. A quantitative passive diffusion periphytometer for lotic systems. *Journal of the American Water Resources Association* 34, 1141–1147.

Reduced nutrient losses to rivers from changes in Swedish agriculture

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Introduction

Significant proportions of the large quantities of nitrogen (N) and phosphorus (P) reaching watercourses, lakes and seas emanate from agricultural land. In Sweden, problems from this eutrophication are most evident in southern and agriculture-dominated regions. Changes to Swedish agriculture on EU entry involved more environmental and production subsidies and increased fallow. Fertilisation with mineral P, which has decreased since the 1970s, is currently at the same level as in the early 1900s. Swedish regulations on livestock density and manure spreading acreage have also become stricter recently.

This study determined changes in water quality in major Swedish rivers over time and investigated whether these could be related to the recent changes in Swedish agriculture.

Statistical methods

We obtained data from the official Swedish monitoring programme of the major rivers and studied the different main forms of N and P in the water. We also applied commonly used flow-normalisation and compensation for the distribution of the concentrations between months. However, if the relationship between concentration and flow changes over time, such flow-normalisation may be erratic. Therefore the period 1983-2004 was divided into two halves and cross-correlation was performed on each month separately. Flow correction was only regarded as valid with no significant cross-correlation term. Only monotonous trends in the same direction were accepted. Therefore more significant trends could be estimated in the shorter period 1993-2004 than in the longer period 1983-2004. The reason was frequent wet years in the 1980s.

Table 1. Basin number (No.), river and annual statistically significant and monotonous trends (mg L⁻¹ year⁻¹) in the period 1993-2004, calculated for flow-normalised concentrations in the water.

No.	River	Nitrogen		Phosphorus	
		Inorganic	Organic	Non-reactive	Reactive
1	Sagån	-	-	-	-
2	Örsundaån	-	-	-	-
3	Dalbergsån	-	-	-	-
4	Lidan	-	-0.034*	-	+0.002**
5	Nossan	-	-0,031**	-	+0.002**
6	Gothemsån	-	-	-0.002**	-
7	Smedjeån	+0.016*	-0.104***	-0.001*	-
8	Råån	-	-0.264***	-0.003**	-0.002*
9	Tolångaån	+0.014*	-	-0.001**	-
10	Klingavälsån	+0.012**	-	-0.001*	+0.001**
11	Skivarpsån	+0.021**	-0.057**	-	-

* Marginally significant trend (0.05<p≤0.10), ** Significant trend (0.01<p≤0.05), ***Highly significant trend (p≤0.01).

Fewer livestock units and more constructed buffer strips have had an effect

The concentration of organic N increased in some of the rivers (Table 1) but this increase was small. The concentration of inorganic N decreased in five rivers. This was correlated with reduced numbers of grazing and non-grazing cattle. The resulting reduced manure supply may also have had some effect on N mineralisation and N leaching from the soil.

The concentration of non-reactive P decreased in several of the rivers. This was weakly correlated with the length of the river-side buffer strips in agricultural areas. The buffers probably reduced surface erosion from the arable land and stabilised the banks of the rivers, thus reducing the P losses to the rivers. Concentrations of reactive P increased in three of the rivers. However, in the river Råån (basin 8) there was a reduction that was most obvious during low-flow periods. Improved treatment of wastewater from single household outlets was suggested as the reason.

The trends in inorganic N were compared with estimated leaching through the root zone from leaching coefficients. Differences may be expected, especially since the concentrations may be diluted by groundwater in the rivers. There was approximately the same pattern between the concentrations measured in the rivers and the estimated concentrations through the root zone, especially in the rivers north of the

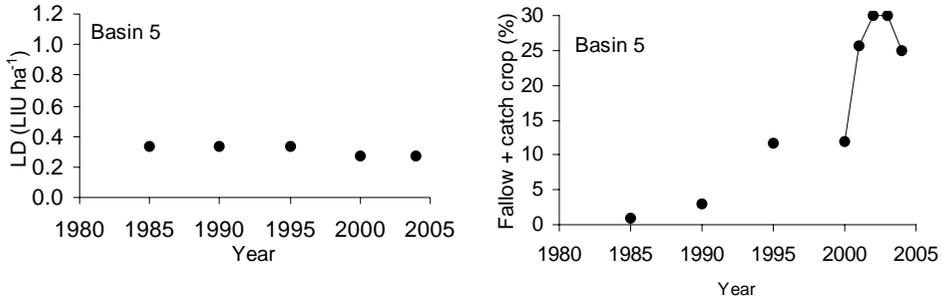


Figure 1. Number of livestock units and changes in the area (%) of fallow and catch crops in the basin of river no. 5 (Nossan), 1983-2004.

county of Scania (rivers 1-7). However, there was no clear tendency since areas with the highest monotonous trends in inorganic N also had the estimated fastest decrease in leaching based on increased use of catch crops. The latter change mainly took place in very recent years (2000-2004), with a non-linear pattern during the period 1993-2004 (Figure 1).

Best improvements in river Råån

The most obvious and overall best improvements took place in the river Råån in south-west Scania. The catchment area has 74% arable land and is intensively cultivated, with winter wheat, oats, sugar beet and winter rape. It is well-documented that single household wastewater treatment has improved in this area. Buffer strips were also installed along the river Råån long before subsidies for such measures became available. In addition, this river is well-known for its good co-operation between farmers, the fishing association and the municipality, parameters hard to quantify but proven to be prerequisites for good results in water quality management work (Ulén & Kalisky, 2005).

Conclusions

Significant and constant trends for declining concentrations of inorganic N were determined for several rivers for the period 1993-2004. Reduced concentrations of non-reactive P were also determined for some rivers. In southern Sweden, eutrophication has probably passed its maximum and is now in a declining phase. Agricultural measures to reduce eutrophication by decreasing N and P concentrations in surface waters should now focus on the region of central Sweden in order to fulfil the environmental objectives of the Swedish Government.

Reference

Ulén, B. & Kalisky, T., 2005. Water erosion and phosphorus problems in an agricultural catchment – lesson from implementation of the EU Water Framework Directive. *Environ. Sci. & Policy* 8, 485-492.

Impacts of agricultural land use on streamwater and sediment P concentrations: implications for P-cycling in lowland rivers

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Introduction

Research over the last decade has identified agricultural land-use practices of highest risk for P-losses at the field and farm scales, and the forms in which P is exported (Withers and Lord, 2002). However, the impacts on streamwater quality and ecology are less well characterised at the catchment scale (Jarvie *et al.*, 2006). In this study, we tested the hypothesis that differences in agricultural land use have a significant impact on streamwater and bed sediment P concentrations and the ability of bed sediments to act as sources/sinks of soluble reactive P (SRP). Three catchments systems were monitored in lowland England and Wales (Wye, Avon and Loddington), characteristic of selected livestock and farming systems with variable P-loss risk (Table 1). For each catchment system, one stream drained a very low intensity agriculture (providing a 'control'), while the other stream(s) drained high-risk agriculture. Streamwater nutrient chemistry was monitored on a weekly basis over two years. Surface fine (<2mm) bed sediments were sampled in spring and late summer and analysed for total P and total Fe concentrations and Equilibrium Phosphorus Concentrations (EPC₀) (Jarvie *et al.*, 2005). Diffusive Equilibrium in Thin films (DET) gel probes (Jarvie *et al.*, in press) were deployed at Loddington for *in-situ* assessment of sediment-water SRP diffusion gradients and fluxes.

Results and discussion

The streams cover a broad continuum in water and sediment P concentrations (Table 1), with median SRP concentrations ranging from 7 µg-P l⁻¹ at Digby Farm (low intensity grassland at Loddington) to 244 µg-P l⁻¹ at Kivernoll (intensive arable, poultry farming and a sewage treatment works in the Wye). Prior's Farm (intensive livestock farming on clay soils in the Avon) had the highest median concentrations of Dissolved Hydrolysable P (DHP) (38 µg-P l⁻¹) and Particulate P (PP) (109 µg-P l⁻¹). Belton Bridge (intensive arable at Loddington) and Prior's Farm had the highest median bed sediment total P concentrations (>2 g-P kg⁻¹) and suspended sediment P concentrations (>7 g-P kg⁻¹). Indeed, there was a strong positive correlation between median P concentrations in suspended sediments and bed sediments (p<0.005). Belton Bridge had the highest bed-sediment total Fe concentration (46 g-Fe kg⁻¹). Fe

oxyhydroxides have a high P-sorption capacity, accounting for the relatively high P enrichment of the Belton Bridge sediments, but low EPC_0 values and low streamwater SRP concentrations. The highest EPC_0 values were at Kivernoll ($70 \mu\text{g-P l}^{-1}$) and Prior's Farm ($42 \mu\text{g-P l}^{-1}$), which had the highest streamwater SRP concentrations, and there was a very strong positive correlation between median EPC_0 and median SRP concentration across the sites ($p < 0.001$). At all sites, SRP concentrations exceeded the EPC_0 concentrations, indicating potential for the surface bed sediments to act as a net sink for SRP, under steady-state low-flow conditions. The EPC_0 results were consistent with *in-situ* DET measurements of streamwater and sediment porewater profiles for Digby Farm and Belton Bridge at Loddington, which showed reductions in SRP concentrations from the streamwater into the surface bed sediment porewaters of up to 67% at Belton Bridge and 43% at Digby Farm. This indicated SRP diffusion gradients from the streamwater into the surface sediment layers. Diffusive fluxes into the surface sediments from the river water were up to $66 \mu\text{g-P m}^{-2} \text{d}^{-1}$ at Belton Bridge and up to $6 \mu\text{g-P m}^{-2} \text{d}^{-1}$ at Digby Farm.

Conclusions

For each catchment system, the agriculturally-impacted streams exhibited consistently higher streamwater P concentrations than the 'control' streams draining low intensity agriculture. Weekly water chemistry monitoring showed that the majority of this increased P in the agriculturally-impacted streams was soluble P for the Avon and Wye catchments, but particulate P at Loddington. Increased P concentrations in the impacted streams may be attributed to increased soil P status in the intensively managed agricultural catchments and thus higher P concentrations in runoff from the agricultural land. However, these catchments contain a diverse range of rural P sources, including farmyard and septic tanks runoff and village sewage treatment works, as well as runoff from agricultural land. There was a strong link between suspended-sediment P concentrations and bed sediment P concentrations, confirming that greater influxes of suspended sediment P from more intensive agriculture leads to greater total P concentrations in river bed sediments. However, the EPC_0 of sediments was not directly related to bed sediment total P concentrations, but instead to streamwater SRP concentrations. Sediment EPC_0 and *in-situ* analysis of sediment-water SRP exchange using DET probes indicated that the bed sediments had potential to act as a sink for streamwater SRP under low flows and times of ecological sensitivity, thus helping to mitigate SRP inputs from rural P sources.

References

- Jarvie, H.P. et al., 2005. Role of river bed sediments as sources and sinks of phosphorus across two major eutrophic UK river basins: the Hampshire Avon and Herefordshire Wye. *J. Hydrol.* 304, 51-74.

Table 1. Summary of sub-catchment characteristics and streamwater and sediment chemistry.

	Loddington			Wye			Avon	
	Digby Farm ('Control')	Belton Bridge (Impacted)	Whitchurch ('Control')	Dinedor (Impacted)	Kivernoll (Impacted)	Cools Cottage ('Control')	Prior's Farm (Impacted)	
Catchment area (km²)	0.4	1.5	6.5	8.7	9.9	1.6	4.7	
Land use	Low intensity permanent pasture (sheep)	Cereal based arable rotations	Low input grassland farming (beef and sheep).	Intensive cereal and potato	Intensive cereal and potato; poultry farming & sewage treatment works	Low intensity grassland (cattle in summer only).	Beef and dairy farming with forage maize; farmyard runoff and cattle crossings	
Soils	Heavy clay soils	Heavy clay soils	Dispersive silty soils	Dispersive silty soils	Dispersive silty soils	Heavy clay soils	Heavy clay soils	
Median water SRP (µg-P l⁻¹)	7	28	25	71	244	54	170	
Median water DHP (µg-P l⁻¹)	7	11	8	14	16	13	38	
Median water PP (µg-P l⁻¹)	5	49	18	21	29	42	109	
Median suspended sediment P (g-P kg⁻¹)	1	9.9	1.7	2.1	4	3.2	7.9	
Median bed sediment Total P (g-P kg⁻¹)	0.9	2.1	0.4	0.5	0.9	0.7	2.3	
Median bed sediment EPC₀ (µg-P l⁻¹)	2.5	4.3	12	15	70	11	42	
Median bed sediment Total Fe (g-Fe kg⁻¹)	22	46	8.5	11	18	12	22	

(SRP, soluble reactive P; DHP, Dissolved Hydrolysable P; PP, Particulate P; EPC₀, Equilibrium P concentration)

- Jarvie, H.P. et al., 2006. Sewage effluent phosphorus: a greater risk to river eutrophication than agricultural phosphorus? *Sci. Tot. Env.* 360, 246-253.
- Jarvie, H.P. et al., (in press). Measurement of Soluble Reactive Phosphorus concentration profiles and fluxes in river-bed sediments using DET gel probes. *J. Hydrol.*
- Withers, P.J.A. & Lord, E.I., 2002. Agricultural nutrient inputs to rivers and groundwaters in the UK: policy, environmental management and research needs. *Sci. Tot. Env.* 282, 9-24.

New sampling method for monitoring of N, P in surface water

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Introduction

Water quality monitoring of surface water is usually done by taking "grab samples" of water in the field, followed by subsequent analysis in the laboratory for e.g. nitrate, phosphate or pesticides. The results are dependent on the time of the sampling, because this sampling method provides a "snapshot" concentration value at a fixed point in time. We have developed a passive sampling method giving weighed average concentration values over longer time periods in the field. This paper describes the method and in-situ measurements of nitrate and phosphorus in six surface streams in Denmark.

Sampling principle

The sampler consists of porous cartridges (cells) containing an adsorbent with a high sorption affinity for the compounds to be measured (figure 1, 2). Further, the cells contain sparingly soluble tracer substances, with known solubility and initial mass m_0 . A porous frit at the entrance regulates the flow rate in the cartridge after exposure of the cells to flowing water, e.g. in a surface stream. The porous cells are in capillary contact with the surrounding water, so there is convective flow of water through the cartridge in response to a hydraulic gradient in the surrounding environment.

The cells are removed after a given installation time in the field, after which the mass of accumulative sorbed compounds is measured in the laboratory. Also, the mass of remaining tracer compound is quantitatively measured in the lab. The leached tracer mass ($m_0 - m_t$) is, through the aqueous solubility K , related to the water volume V that passed through the cartridge:

$$V = (m_0 - m_t) / K$$

After the chemical analyses, the mass of the solute M is known, as well as the volume V in which the solutes were dissolved. The average concentration of the solute is then simply derived from M/V .

We have identified a range of suitable adsorbents for different types of solutes, and a range of suitable salts (de Jonge & Rothenberg, 2005). The new method can be used for example in groundwater, waste water and surface water.

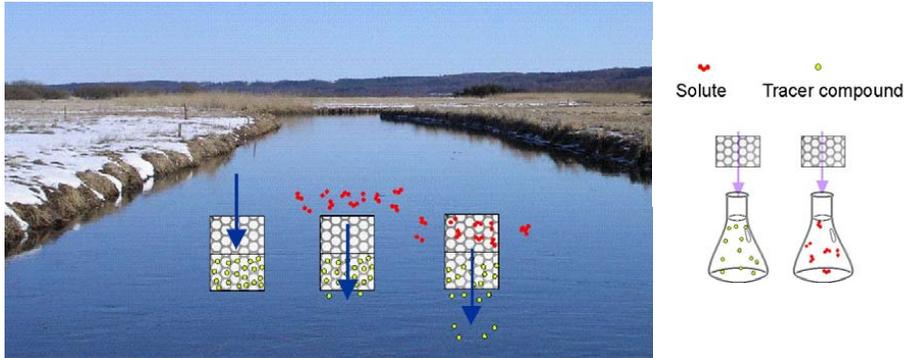


Figure 1. Working principle of samplers in streams. The picture shows one typical monitoring location.



Figure 2. Pictures showing sampling cells before and after installation.

Experimental design

The method was validated for measurements for nitrate and phosphorus in two independent experiments. Sampling cartridges (Figure 2) were installed in the different streams for a period of 10 to 16 days. The concentration of nitrate and phosphorus was calculated from the extracted amount on the resin and the leached amount of tracer salt as described above. Further, conventional water samples were taken at the start and end of the installation period. Water samples were measured for nitrate, total phosphorus, ortho-P after filtration over 0.45 μm and Molybdate-reactive P (MR-P). MR-P is measured the way as for the two other methods, however the filtration and destruction of the water samples is omitted.

Results and discussion

Figure 3 shows the correlation plot of traditional nitrate measurements from grab samples and the Sorbisense method ($R = 0.975$). The slope of the correlation is 0.996. The average deviation between the individual measurements was 0.30 mg/L. This covers uncertainties in both the grab samples and the sampling cells.

The average repeatability of three replicate sampling cells was 19% of the measurement value ($n=11$). The average difference between grab sample values from the same location was small, 11%. This indicates that the nitrate concentration during the given sampling periods was rather constant and therefore suitable for validation purposes.

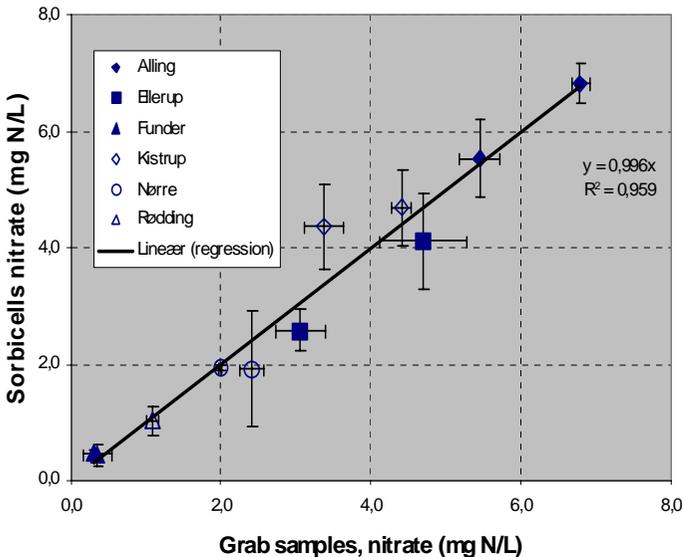


Figure 3. Nitrate measurements (average values \pm s.d.) in six surface streams in Denmark. The graph shows the correlation between results from grab samples (x-axis) and three replicate sampling cells (y-axis).

The dissolved phosphorus fraction usually represents only a small part of the total phosphorus load in surface streams. Phosphorus is often measured both as ortho-phosphate (after filtration through a $0.2 \mu\text{m}$ membrane filter) and as total-P after oxidative destruction. Molybdenum-reactive P (MR-P), where samples are neither filtered nor destroyed, represents the labile form of phosphorus that is analytically available without oxidative destruction.

Concentrations values derived from the sampling cells are at the same level as MR-P measurements that are in the interval between total-P and ortho-P (Figure 4). MR-P concentrations and concentration from sampling cells are linearly correlated ($R = 0.987$). The slope of the correlation plot is 1.05. Deviations between individual measurements from MR-P grab samples and sampling cells are on average 0.006 mg/L .

Porous entrance frits cause a certain filtration of the water samples, depending on the pore size (in this case $20 \mu\text{m}$). This means that colloids and small suspended

particles below this cut-off diameter enter the sampling cells, while larger particles are excluded. We observed the colloids as a coloured chromatographic band on the upstream part of the cells (see Figure 2).

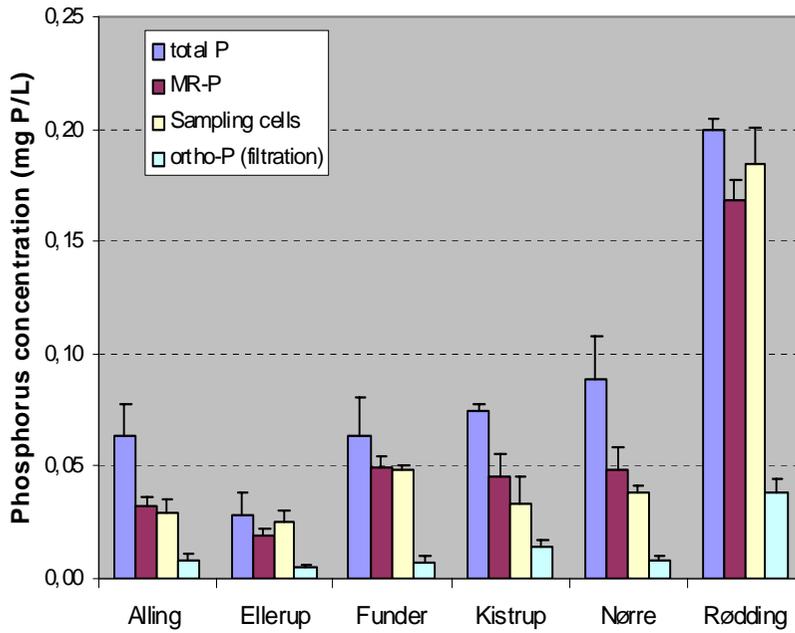


Figure 4. Phosphorus measurements in six surface streams in Denmark. The graph shows average results from grab samples analyzed for total-P, molybdate-reactive P (MR-P) and ortho-phosphate, together with results from sampling cells that were installed for a period of 13 days.

Conclusions and perspectives

For nitrate, nearly identical results were obtained with the Sorbisense method and the traditional grab samples.

Considering phosphorus, the sampling cells intercept labile phosphorus that is either free dissolved or complexed with colloids and suspended particles. Sorbisense results are comparable to MR-P levels in water samples. The Sorbisense method can be advantageous over grab samples, if the expected variation between sampling events is higher than 20%.

References

de Jonge, H. & Rothenberg, G., 2005. New device and method for flux-proportional sampling of mobile solutes in soil and groundwater. *Environ. Sci. Technol.* 39, 274-282.

Phosphorus balances in Europe and implications for diffuse pollution policy

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Anthropogenic sources involved in eutrophication account for about 83% but natural sources only 17% of the total global yearly P flux of >103 Mt. About 90% of these anthropogenic P sources are (in-)directly from the nutrition sector. Therefore there is a need for a lifecycle analysis (LCA) not only for C, N and S but also for P within the total food chain of agriculture with plant and animal nutrition, human nutrition and waste as well as waste water management to optimize sustainable use and management of P in respect to efficiency, consistency and sufficiency. Phosphorus surpluses in agricultures began to decrease dramatically during the 1970's and 80's. In Western Europe this was especially due to reduced use of mineral fertiliser P and P in feedstuffs as well as increased P offtake in crops yielding greater P use efficiency. In the Central Eastern European countries, as the result of economic collapse following the political changes in 1989, fewer farm animals have produced less animal manure P. Nevertheless, actual average farm-gate balances [$\text{kg P ha}^{-1} \text{yr}^{-1}$] still show a P surplus, which, especially in Western European countries, is too high. (>20-30: Netherlands; >10-20: Norway, Denmark, Ireland, Ukraine (PR China); 5-10: United Kingdom, Finland, Switzerland, Germany; >0-5: Luxembourg, Sweden, Poland, Slovenia; <0: Hungary, Czech Republic, Bulgaria). About 50-90% of the soils especially in Western Europe have optimum or high/excessive P status. About 90% of both P surplus and excessive P status are caused by high animal consumption and subsequent production. On average in most European countries there is no need for a P surplus in agriculture for the next few decades, also as a consequence of world mineral P supplies becoming exhausted within the next 100 years. The required reduction in animal consumption and production in EU-27 (ranging from 19% for Malta to 94% for Ireland) in respect to a healthy and even more cost effective human nutrition (Tab.1) will also solve the nutrition-related environmental problems of eutrophication, acidification, climate change by about 60-80%, fulfilling also the requirements of the hydrosphere/surface waters for a good nutritional status (Tab.2). - But too high an animal consumption and production with $>0.1 \text{ AU capita}^{-1}$ and livestock densities of $>1 \text{ AU ha}^{-1}$ are still encouraged by corruption, (il-)legal subsidies accompanied by unhealthy human nutrition, esp. with animal food. In respect of the present national and EU unsustainable and disintegrated legislation with regard to the total food chain, there is an urgent need for an integrated sustainable nutrient legislation and implementation with corresponding framework directives for the nutrients C, N, P and S with adequate thematic strategies for pedosphere, hydrosphere, atmosphere, lithosphere and biosphere.

Tab. 1: Necessary reduction of animal production and livestock of agriculture both in the countries of EU-25+2 and in the Federal lands of Germany on the basis of the actual capita-specific animal densities (AU·capita⁻¹) in comparison with a maximum tolerable animal density of 0.1 AU= 50 kg life weight·capita⁻¹ (Isermann 1995/2006) according to a healthy human nutrition with animal food, especially with meat [Net: max. 23,4 kg meat·capita⁻¹·year⁻¹ (DGE 2000/01) instead of actually i.e. in Germany (2002):60 kg·capita⁻¹·year⁻¹]
 [Actual animal stockings and densities according to EUROSTAT 2005]

Countries	Actual Animal densities (AU·capita ⁻¹)	Necessary Reduction Livestock (%)	Countries	Actual Animal densities (AU·capita ⁻¹)	Necessary Reduction Livestock (%)	Federal Lands of Germany	Actual Animal densities (AU·capita ⁻¹)	Necessary Reduction Livestock (%)
1.Ireland	1.606	-94	14. Hungary	0.263	-62	1. Schleswig-Holstein	0.466	-79
2.Denmark	0.846	-88	15. Bulgaria	0.254	-61	2. Niedersachsen	0.456	-78
3.France	0.390	-74	16. Estonia	0.241	-59	+Hamburg		
4.Belgium	0.382	-74	17. United kingdom	0.240	-58	+Bremen		
5.Netherlands	0.380	-74	18. Greece	0.238	-58	3. Mecklenburg-Vorp.	0.404	-75
6.Cyprus	0.359	-72	19. Finland	0.227	-56	4. Bayern	0.311	-68
7.Luxemburg	0.355	-72				5. Sachsen-Anhalt	0.252	-60
8.Spain	0.341	-71				6. Thüringen	0.232	-57
9.Lithuania	0.339	-71				Germany	0.226	-56
10. Austria	0.308	-67	20. Germany	0.226	-56	7. Sachsen	0.156	-36
11. Romania	0.304	-67	21. Portugal	0.224	-55	8. Nordrhein-Westf.	0.154	-35
			22. Czech. Republic	0.205	-51	9. Baden-Württemb.	0.140	-29
			23. Sweden	0.197	-49	10. Brandenburg	0.130	-23
EU-15	0.294	-66	24. Latvia	0.177	-44	+Berlin		
12.Slovenia	0.293	-66	25. Slovakia	0.174	-43	11. Hessen	0.106	-6
13. Poland	0.292	-66	26. Italy	0.174	-43	12. Rheinland-Pfalz + Saarland	0.094	+7
EU-25+2	0.290	-64	27. Malta	0.123	-19			
EU-10+2	0.275	-64						

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Tab. 2 Qualification of surface waters (Groundwater/Draft EU-Groundwater Directive 2006) in respect to the nutrient levels C, N, P, S in Germany (LAWA / UBA 2006) (basis value: 90-percentile)

[http://www.umweltbundesamt.de/wasser/themen/ow_s3_3.htm]

		Chemical water quality categories regarding the nutrient levels						
Nutrients	Unit	I	I-II	II ¹⁾	II-III	III	III-IV	IV
		Anthropogenic Levels						
		Not influenced Geogenic background dark-blue	very low ½ aim light-blue	moderate aim green	obvious up to 2x aim light-green	increased up to 4x aim yellow	high up to 8x aim orange	very high up > 8x aim red
1. TOC	mg l ⁻¹	<= 2	<= 3	<= 5	<= 10	<= 20	<= 40	> 40
2. total N(TN)	mg l ⁻¹	<= 1	<= 1,5	<= 3	<= 6	<= 12	<= 24	>24
...of it:								
2.1 Nitrate-N	mg l ⁻¹	<= 1	<= 1,5	<= 2,5 ²⁾ (drinking water:<11,3)	<= 5	<= 10 (drinking water:<11,3)	<= 20	>20
2.2 Nitrite-N	mg l ⁻¹	<= 0,01	<= 0,05	<= 0,1	<= 0,2	<= 0,4	<= 0,8	>0,8
2.3 Ammonium-N	mg l ⁻¹	<= 0,04	<= 0,1	<= 0,1 <= 0,3	<= 0,6	<= 1,2	<= 2,4	> 2,4
3. Total-P (TP)	mg l ⁻¹	<= 0,05	<= 0,08	<= 0,15	<= 0,3	<= 0,6	<= 1,2	> 1,2
...of it:								
Ortho- Phosphate-P	mg l ⁻¹	<= 0,02	<= 0,04	<= 0,1	<= 0,2	<= 0,4	<= 0,8	> 0,8
4. Sulfate	mg l ⁻¹	<= 25	<= 50	<= 100	<= 200 (drinking water: = 240)	<= 400	<= 800	> 800

¹⁾ Demand up to 2010: strict observance on all measuring locations of LAWA

²⁾ Maximum tolerable for total life of adults (70 kg life weight, 2 l drinking water · d⁻¹): 2,9 mg NO₃-N l⁻¹

Phosphorus balances in Swedish dairy farms

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Introduction

The parliament in Sweden has decided fifteen environmental objectives. One of them - zero eutrophication - has a great impact on agriculture in Sweden, especially in the south. Interim targets for this objective concern both waterborne anthropogenic nitrogen emissions into the Baltic Sea in the form of ammonia; and also phosphorus emissions (www.internat.naturvardsverket.se, access 2006-10-17). The focus has centred on decreasing nitrogen emissions, but there is a debate as to whether it is more important to decrease the phosphorus surplus, especially referring to the situation in the Baltic Sea. (Boesch *et al.*, 2005).

To fulfil the demands by society for environmental problems to be reduced, there is a need to measure the environmental output from a farm. One way of doing this is to use the farm-gate balance. With farm-gate balances it is possible to calculate, in principle, everything entering and leaving the farm, for example, energy, minerals or metals such as cadmium. The farm-gate balance is an example of an area-based indicator. According to Halberg *et al.* (2005) an area-based indicator should be chosen if the focus is on a specific region, as the indicator is more site-specific.

Since the middle of the 1990s there have been several publications on farm-gate balances, both at country or regional level (Nevens *et al.*, 2006; Nielsen & Kristensen, 2006; Swensson, 2003). However, few investigations have published results covering several years showing the trends in farm-gate balances.

This paper presented here is an analysis of six years of farm-gate balances concerning phosphorus from the same dairy farms in southern Sweden. The hypothesis is that the phosphorus surplus should decrease during the period of analysis.

Materials and methods

Since 1997, the dairy plant, 'Dairy Skane', situated in southern Sweden has supplied dairy farmers with calculations of their mineral balances. Included in these balances are N, P and K. Results from these farm-gate balances have been presented in Swensson (2002) and Swensson (2003).

In the year 2000 a new, state-financed campaign called “Focus on nutrients” was started among farmers in the south of Sweden. The aim of the campaign was to make the farmers more aware of environmental problems and to present measures to decrease the emissions of nitrogen and phosphorus. The campaign was implemented by advisers who visited the farmers several times. At the first farm visit a farm-gate balance was calculated to show the flow of nitrogen and phosphorus on the farm (www.greppa.nu, accessed 2007-07-30).

The same methodology was used as in Swensson (2003) referring to farm-gate balances from 1997, 1998 and 1999. Farm-gate balances from the years 2001, 2002 and 2003 were from the campaign, “Focus on nutrients.” There are no results from year 2000, because the campaign was planned during this year. These balances were calculated using a software called STANK. The information was from the same sources as in previous years and is described in Swensson (2003).

The whole-farm balance was defined as the difference between inputs to the farm and recovery in agricultural products. The nitrogen surplus per hectare was defined as the difference between input and output of nitrogen divided by the size of the farm in hectares. The farm size was defined as land on which manure could be spread; which includes all arable land on the farm but not natural pasture. The nitrogen efficiency was defined as the ratio between nitrogen output and nitrogen input (van der Hoek, 1998).

In summary, to fit in with the analysis the following criteria should be fulfilled:

- There should be farm-gate balances from six years: 1997, 1998, 1999, 2001, 2002 and 2003;
- The dairy farms should not export or import manure;
- The dairy farms should not have major pig or beef production. There was no chicken or egg production on the dairy farms;
- The dairy farms should not have a major export of nitrogen from crop production; Farms with an outflow of nitrogen from crops above fifty per cent were excluded.

Results

Table 1 presents results from the analyses. There is only a small decrease in P surplus during these years. In the same period the N surplus decreased by approximately 40 Kg N (not shown in table).

Table 1. Results of mineral balances measured between 1997-2003 (average values) from 51 dairy farms in southern Sweden.

Year	1997	1998	1999	2001	2002	2003
Delivered milk, kg milk/ha	7,007	6,710	7,169	7,531	7,180	7,087
Phosphorous surplus, kg P/ha	6	5	6	6	4	5

Standard deviations of P surplus were 8 kg P 1997 and 7 kg P 2003. P surplus varied between -4 - +44 kg P in 1997 and -8 - +25 kg P in 2003.

Discussion

Compared to similar investigations of P surplus on dairy farms in Denmark, the Netherlands and the United States, the P surplus was lower in our investigation (Kuipers & Mandersloot, 1999; Nielsen & Kristensen *et al.*, 2006; Hristov *et al.*, 2006). The low reduction in P surplus during the analysed years was a disappointment. One explanation could be, as mentioned in the introduction, that the focus has been on decreasing the N surplus during these years. The large variation in P surplus between farms shows that there is a potential to decrease the P surplus on dairy farms. One option is to minimise the P inflow from feed by precision-feed management (Ekelund, 2003; Ghebremichael *et al.*, 2007)

References

- Boesh, D., Hecky, R.O., Melia, C., Schindler, D. & Seitzinger, S., 2005. Expert evaluation of the eutrophication of the seas surrounding Sweden. Swedish Environmental Protection Agency, Mimeo, 54 pp.
- Ekelund, A., 2003. Phosphorus and the dairy cow. Doctoral dissertation, Dept. of Animal Nutrition and Management, SLU. Acta Universitatis Agriculturae Sueciae. Agraria vol. 412.
- Ghebremichael, L.T., Cerasaletti, P.E., Veith, T.L., Rotz, C.A., Hamlett, J.M. & Gburek, W.J., 2007. Economic and phosphorus-related effects of precision feeding and forage management at a farm scale. *J. Dairy. Sci.* 90, 3700-3715.
- Halberg, N., Van der Werf, H.M.G., Basset_Mens, C., Dalgaard, R. & de Boer, I.J.M., 2005. Environmental assessment tools for the evaluation and improvement of European livestock production systems. *Livest. Prod. Sci.* 96, 33-50.
- Hristov, A.N., Hazen, W. & Ellsworth, J.W., 2006. Efficiency of use of imported nitrogen, phosphorus and potential for reducing phosphorus imports on Idaho dairy farms. *J. Dairy. Sci.* 89, 3702-3712.
- Kuipers, A. & Mandersloot, F., 1999. Reducing nutrient losses on dairy farms in The Netherlands. *Livest. Prod. Sci.* 61, 139-144.
- Nevens, F., Verbruggen, I., Reheul, D. & Hoffman, G., 2006. Farm gate nitrogen surpluses and nitrogen efficiency of specialised dairy farms in Flanders: evolution and future goals. *Agric. Syst.* 88, 142-155.
- Nielsen, A.H. & Kristensen, I.S., 2006. Nitrogen and phosphorus on Danish dairy and pig farms in relation to farm characteristics. *Livest. Prod. Sci.* 96, 97-107.
- Swensson, C., 2002. Ammonia release and nitrogen balances on south Swedish dairy farms 1997-1999. Doctoral diss. Dept. of Rural Buildings, SLU. Acta Universitatis Agriculturae Sueciae. Agraria vol. 333.
- Swensson, C., 2003. Analyses of mineral element balances between 1997 and 1999 from dairy farms in the south of Sweden. *Eur. J. Agr.* 20, 63-69.
- van Der Hoek, K.W., 1998. Nitrogen efficiency in global animal production. *Environ. Poll.* 102, 127-132.

The contribution of inorganic feed phosphates to the European P-soil status

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Introduction

This is a summary of the report, “The contribution of Inorganic Feed Phosphates to the European P-soil status” compiled by IFP (Inorganic Feed Phosphates), a CEFIC sector group.

The objective of this document is to put into perspective the contribution that inorganic feed phosphates (IFP) make to the phosphorus (P) content in manure and to the total application of P to the soil by agriculture.

Feed usage and composition

Today the EU has become the world’s largest exporter and importer of agricultural produce, including animal products. Animal feeds are mostly composed of plant material, including roughage, grain and oilseed products totalling approximately 450 million tonnes of feedstuffs each year (FEFAC, 2005). All plant materials contain intrinsic levels of phosphorus, but this is usually present at low and variable levels. Moreover, the digestibility of this phosphorus is often too low to satisfy the animal’s need for phosphorus. In general, only 30% of the P in vegetal sources is available for monogastric animals.

Phosphorus requirements

Both crops and animals have a basic P requirement. Together with livestock manure, a number of P-containing fertilisers are used to supply the soil and thereby the plant with sufficient levels of P. The removal of P in crops at harvest is around 20 kg P per hectare (Sibbesen *et al.*, 1996). Like plants, animals also have a basic P-requirement. P-deficiency can impair animal health and welfare and also have significant economic consequences for the livestock producer. Under most circumstances, in order to meet the P requirements for production, feeds for monogastric animals have to be supplemented with P-rich materials such as IFPs, which contain high levels of digestible P (dP).

IFP sources in Europe today

A wide choice of inorganic feed phosphates is available to livestock producers and feed manufacturers in Europe today (EC, 1998). The most commonly used forms of feed phosphates are monocalcium (MCP), monodicalcium (MDCP), anhydrate and

dihydrate dicalcium phosphates (DCP.0H₂O/ DCP.2H₂O), and to a lesser extent magnesium phosphate (MgP), monoammonium phosphate (MAP) and monosodium phosphate (MSP). Also available, although not produced in the EU, is defluorinated phosphate (DFP). IFP sources differ in mineral content and chemical composition and consequently in total P-content and P-digestibility. In vivo trials consistently show that the bioavailability of inorganic feed phosphates (expressed as dP¹) for both pigs and poultry ranks as follows: MSP, MAP and MCP have the highest availability ranging between 75-92%, followed by DCP dihydrate (DCP.2H₂O) and MDCP (75-85%). DCP anhydrate (DCP.0H₂O) has a lower digestibility (55-73%) and DFP has by far the lowest value (55-60%) (Bleukx, W., 2005; CVB, 2005; Jongbloed *et al.*, 2002; Kemme *et al.* 2001; Van der Klis *et al.*, 1996).

P application to the soil

Phosphorus excretion is an inevitable consequence of livestock production. In total almost 1.4 million t of P per year is excreted in the manure within EU 15 and this manure is used as an agricultural fertiliser. To fulfil the mineral requirements of plants, mineral fertilisers are used to balance livestock manure. The total P application of both manure and mineral fertilisers per hectare is 26 kg of P on average, totalling more than 2.5 million tonnes of P (EFMA, 2005).

P contribution of IFP to the soil in perspective

IFPs are used to balance animal feeds due to their P-content. The annual total use of IFP is estimated to reach 1.4 million t for EU 25 or 0.25 million t expressed as total P (in DCP 18 equivalence) (CEH, 2004), which is almost 11% of the total P consumed by livestock in feed. Taking into account the IFP consumption per species and the livestock numbers within the EU15, the IFP contribution to the total P in manure can be estimated. Due to the high digestibility of IFP sources the total contribution of P in the manure from IFP is calculated to be about 57,000 t P per year, which is approximately 4 % of the total P in the manure. The balance, 96%, is P originating from other feed materials in the diets, including the use of phytase. For the total average P-application per hectare from livestock manure and mineral fertilisers (26 kg) IFP contributes only 2% to the total P-application, 0.6 kg/ha/year.

Possible scenarios to limit P-excretion in livestock manure

Although most European countries are already working to reduce agricultural phosphorus losses to the environment, some possibilities still exist for limiting P-excretion further. This includes 1) reduced use of IFP (by legislation) by setting maximum levels; 2) lower P norms for animal feeds, or 3) increase the dP level in the IFP. With the exception of the latter, all these measurements have only a small or even counter-productive effect, certainly when IFP is replaced by other feed

¹ Digestible P = the difference between intake of P and excretion of P via the faeces, in case of poultry including excretion via the urine.

materials with a lower P-digestibility. The risk exists of feeding the animals below requirement and increasing the P output in the environment.

There is, however, no common definition for highly digestible IFP sources in the EU and no standard evaluation method to differentiate between the digestibilities of IFPs. The development of a standardised test for calculating P-digestibility values for feed materials and IFP should be prioritised. Only IFP with a sufficiently high digestibility should be allowed in animal nutrition. It is therefore recommended that such a method is developed within the framework of one of the projects of DG-research. The setting of benchmark values is essential to further optimise the use of IFP.

Conclusion

Both plants and animals have a requirement for P and P-excretion is an inevitable result of animal production. Due to its high P digestibility, IFP only contributes a small part of the total P excretion in manure and thus to the P-application to soil. Replacing IFP sources by other P-rich sources with a lower digestibility could be counter-productive, as animals may be fed below requirement, increasing P output to the environment. It is advisable to standardise P-digestibility values of feed materials, including IFP. Only IFP with a high digestibility should be allowed in animal nutrition.

References

- Bleukx, W., 2005. Production et Qualité Nutritionnelle des Phosphates Alimentaires. INRA Prod. Anim. 18(3), 169-173.
- CEH, 2004. Marketing research report: Animal Feeds: Phosphate Supplements. Commission Directive 98/67/EC part B 11 (non exclusive list), 1998.
- CVB, 2005. Tabellenboek Veevoeding 2004. Centraal Veevoederbureau, Lelystad.
- EC, 1998. Commission Directive 98/67/EC, part B11. Amending Council Directive 96/25/EC. The European commission, Brussels.
- EFMA, 2004. Non-fertilizer nutrient sources in Europe. ECOPT report, Suffolk, UK.
- FEFAC, 2005. Feed and Food, Statistical Yearbook 2004. FEFAC, Brussels.
- Jongbloed, A.W., Kemme, P.A., De groote, G., Lippens, M. & Meschy, F., 2002. Bioavailability of Major and Trace Minerals. 112 pp. EMFEMA, Brussels.
- Kemme, P.A., van Diepen, J.T.M., Jongbloed, A.W. & Ferreira, P., 2001. Apparent digestibility of phosphorus in pigs of five feed phosphates from KK Animal Nutrition. Confidential report ID-Lelystad no. 2162. Kemira GrowHow - ID TNO Animal Nutrition, Lelystad, The Netherland.
- Sibbesen, E. & Runge-Metzger, A., 1996. In: Phosphorus Balance in European Agriculture – Status and Policy options; SCOPE 54, 1996.
- Van der Klis, J.D. & Versteegh, H.A.J., 1996. Phosphorus Nutrition of Poultry. In: Recent Advances in Animal Nutrition.

A fecal P test for evaluating P status of dairy cows

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Introduction

Phosphorus is essential for the health and production performance of dairy cows. However, for various reasons many farms feed animals with diets containing excessive amount of P, which is subsequently excreted in manure (Wu et al., 2000). Long-term manure applications to agricultural land have led to soil P buildup to levels exceeding what is typically required for satisfactory crop production (Sims, 2000). Such high-P soils are often associated with elevated P loss in runoff water or subsurface drainage, contributing to water quality decline in regions. Therefore, it is vital to develop management tools that help producers lower unnecessary P in diets without impairing animal performance. The objective of this study was to develop and evaluate a fecal P test procedure for assessing if P overfeeding is occurring in a dairy herd.

Experimental design

Feed and fecal samples were obtained from lactating cows (Holstein breed, fed total mixed rations) on commercial dairies in Northeast and Mid-Atlantic region, USA. Feed samples were obtained at the feed bunk at feed delivery, mixed by hand and randomly subsampled, and analyzed at a commercial laboratory for standard feed quality parameters using established procedures. Fecal samples were rectal grabs from randomly selected healthy cows (7 to 17 cows depending on the herd size) and were kept individually. Fecal samples were kept frozen at -20°C until analysis in a research laboratory. The analyses, after thawing frozen samples at room temperature, included (i) DM, (ii) acid digest total P (TP) and Ca and Mg, and (iii) extractable P in deionized water or various acid solutions including HCl (concentrations of 0.4, 0.1, 0.02, 0.008, 0.004% v/v), citric acid (2, 0.1, 0.01, and 0.005%), and acetic acid (4, 2, 1, 0.5, 0.1, 0.01, 0.005%). All extractions were performed with 2 g wet fecal sample in 98 mL extractant, shaken on a reciprocal shaker for 1 h, and filtered through Whatman 42 paper. After measuring pH, an aliquot of extract was analyzed for P_i by the phosphomolybdate blue method of Murphy and Riley (1962). Another aliquot (0.5 mL) was digested in concentrated nitric acid (1 mL) at 90°C overnight, then determined for extractable total P (P_e) and total Ca and Mg on ICP-MS (mass spectroscopy). Extractable organic P (P_o) was calculated as $P_i - P_e$.

Best suitable extractant

A previous study using research feeding trials has demonstrated a close linear relationship ($R^2=0.95$, $n=8$) between water extractable P concentrations in dairy feces and P concentrations in diets (Dou et al., 2002). However, the relationship is more complicated with farm-based samples, as reported by Chapuis-Lardy et al. (2004) that other parameters such as pH and Ca content of feces also affected the extractability of P in water. In the present study, a greater amount of P was released in acid extracts than in deionized water. Increasing the concentration of the acids released greater amount of Pi but the effect of the increment tended to diminish at higher acid concentrations. Our core interest was to overcome the impact of factors such as pH, Ca, etc. on P extractability while keeping the procedure simple and the acid extractant as dilute as possible. The final selection was 0.1% HCl as the best suitable extractant. Concentrations of Pi in 0.1% HCl extracts were no longer affected by pH, Ca, or sample handling (wet vs. dry based extraction), but closely reflect dietary P changes ($R^2=0.69$; see Dou et al., 2007 for details).

Relationship between fecal P and dietary P

More intensive testing using samples from a total of 92 farms resulted in trends similar to that reported by Dou et al. (2007), that is, 0.1% HCl extracted Pi closely reflects the changes in dietary P whereas extractable Po plus residual P (i.e. TP-Pi) was independent of dietary P changes (Fig. 1).

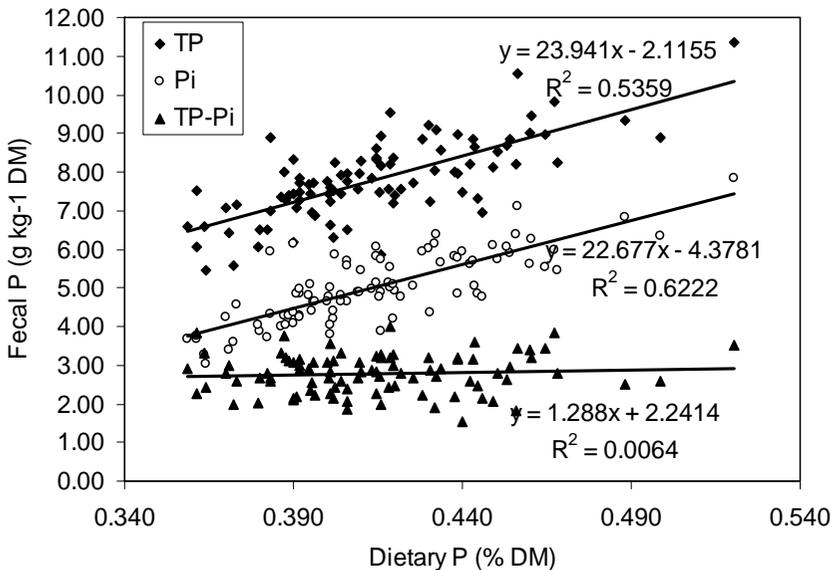


Figure 1. Relationship between dietary P and fecal P (n=92) as acid digest total P (TP), 0.1% HCl extractable Pi or the rest of fecal P (TP-Pi).

The common understanding is that P in dairy feces consists of P as undigested feed residues, microbial residues, sloughed tissues, and digested but unabsorbed P. The latter is most important concerning the common farm practice of P overfeeding: excessive dietary P is digested but not absorbed and subsequently excreted in feces. It is reasonable to assume that much of the digested but not absorbed P in feces is soluble in 0.1% HCl, whereas the remaining fecal P (in feed residues, microbial residues, and sloughed tissues) is organic with limited solubility. Thus, use of 0.1% HCl as an extractant enables us to measure the relative magnitude of the digested but not absorbed P. Its potential usefulness as a management tool is apparent: If a benchmark value is designated to represent adequate P status, we would be able to assess whether excess P intake is occurring in a herd by determining 0.1% HCl-extractable P_i in feces and comparing it with the benchmark.

A fecal P based assessment tool would be relatively simple to adopt in the field because sample collection is easy and non-invasive and the laboratory procedures are straightforward and inexpensive. From the environmental perspective, 0.1% HCl extracted P_i is mostly labile and includes much of the P fraction that is most vulnerable to potential losses after the manure is applied to field.

References

- Chapuis-Lardy, L., Fiorini, J., Toth, J.D., Ferguson, J.D. & Dou, Z., 2004. Phosphorus concentrations in dairy feces: variability and affecting factors. *J. Dairy Sci.* 87, 4334-4341.
- Dou, Z., Knowlton, K.F., Kohn, R.A., Satter, L.D., Wu, Z., Zhang, G., Toth, J.D. & Ferguson, J.D., 2002. Phosphorus characteristics in dairy feces affected by diets. *J. Environ. Qual.* 31, 2058-2065.
- Dou, Z., Ramberg, C.F.Jr., Chapuis-Lardy, L., Toth, J.D., Munson, R., Wu, Z., Chase, L., Kohn, R., Knowlton, K. & Ferguson, J., 2007. A novel test for measuring and managing potential phosphorus loss from dairy cattle feces. *Environ. Sci. Tech.* (in press).
- Murphy, J. & Riley, J.P.A., 1962. modified single-solution method for the determination of phosphate in natural waters. *Anal. Chim. Acta.* 27, 31-36.
- Sims, J.T., 2000. The role of soil testing in environmental risk assessment for phosphorus. In *Agriculture and phosphorus management*. Sharpley, A.N. Ed.; Lewis Publ., Boca Raton, FL. 2000. pp 57-81.
- Wu, Z., Satter, L.D. & Sojo, R., 2000. Milk production, reproductive performance, and fecal excretion of phosphorus by dairy cows fed three amounts of phosphorus. *J. Dairy Sci.* 83, 1028-1041.

Curtailing fertilizer P inputs on the P status of soils and P losses

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Introduction

A range of Olsen-P concentrations from 14 to 67 mg P L⁻¹ were established in the top 75 mm of grazed grassland swards in Northern Ireland by surface applying different rates of P fertilizer (0, 10, 20, 40 or 80 kg P ha⁻¹ yr⁻¹) over a period of five years (Mar 2000 – Feb 2005) (Watson *et al.*, 2007). The current study investigates how the Olsen-P status of these soils changes when no fertilizer P is applied and how P losses to surface runoff and land drainage water are affected.

Recent soil samples from intensive grassland farms in Northern Ireland showed that 55% of silage fields were above the agronomic optimum for Olsen-P (>25 mg P L⁻¹) (Jordan, 2007, personal communication). For Northern Ireland to comply with a range of European Union Directives (European Community, 1991; Directive 2000/60/EC) on the aquatic environment, it has declared a 'total territory' approach to control the agricultural contribution to nutrient losses. Part of the proposed action plan includes withholding fertilizer P application on soils of high Olsen P status. Curtailing fertilizer P applications at the current site should demonstrate how rapidly environmental improvements can be achieved under grazing management and provide information to support policy decisions.

Methods

Grassland plots (each 0.2 ha) received 0, 10, 20, 40 or 80 kg P ha⁻¹ yr⁻¹ applied as triple superphosphate (46% P₂O₅) in six equal applications from March 2000 to February 2005. Thereafter, no P fertilizer was applied to any of the plots. All plots received the same inputs of N (250 kg N ha⁻¹ yr⁻¹), potassium and sulphur. Plots were grazed by beef steers from April to October to maintain a constant sward height of 7 cms. Plots were hydrologically isolated and artificially drained to v-notch weirs with flow proportional monitoring of drainage water. Surface runoff collectors were installed across the width of the plots at the lowest point and were connected to portable water samplers (ISCO, Inc). Soluble reactive P (SRP) was determined on 0.45 µm membrane filtered water samples by the acidic molybdate-ascorbic acid method of Murphy and Riley (1962). Total soluble P and total P were determined on filtered and unfiltered samples, respectively, by digestion with potassium persulphate and sulphuric acid, followed by analysis of the digest as for SRP. For soil sampling,

each plot was divided into three sections (top, middle and bottom). On a weekly basis, thirty cores (15 mm diameter x 75 mm depth) were randomly taken from each section and bulked. Soil samples were air-dried at 30 °C, ground to pass a 2 mm sieve and analysed for bicarbonate-extractable inorganic P (Olsen-P), using a soil to Olsen reagent ratio of 1:20.

Results

Withholding P fertilizer for over five years from a sward at the agronomic optimum soil P status lowered Olsen-P by 0.75 mg L⁻¹ yr⁻¹, but had no adverse effect on herbage P concentrations.

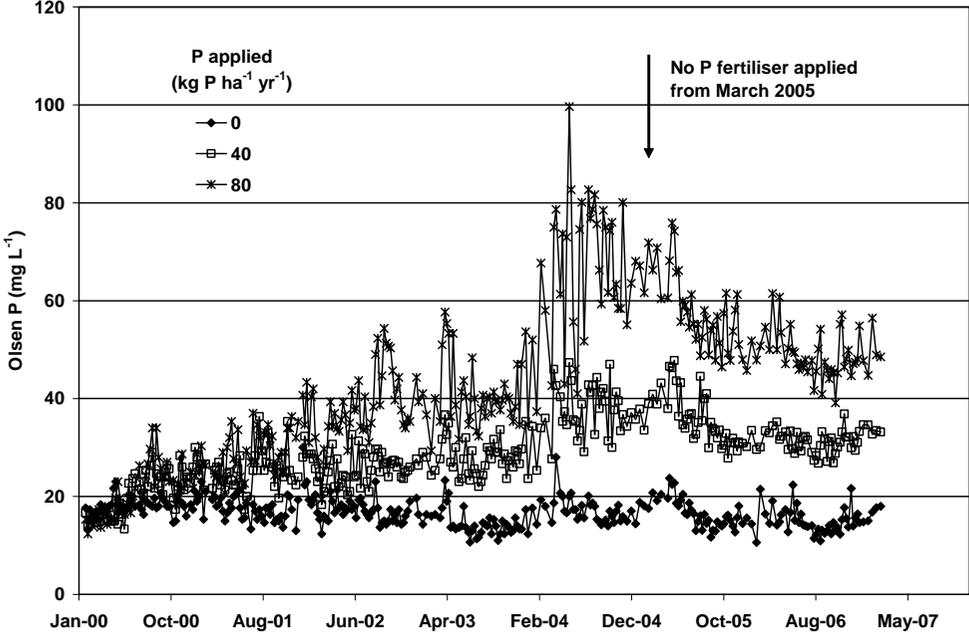


Figure 1. Change in weekly Olsen P (mg L⁻¹) status in the plots receiving 0, 40 and 80 kg P ha⁻¹ yr⁻¹ from March 2000 to February 2005. No P fertilizer was applied from March 2005 onwards.

The increase in Olsen P by applying 10, 20, 40 or 80 kg P ha⁻¹ yr⁻¹, from March 2000 to February 2005 was estimated to be 0.54, 1.83, 4.42 and 9.58 mg P L⁻¹ yr⁻¹, respectively. In Jan/Feb 2005, the Olsen-P status of the plots receiving 0, 10, 20, 40 and 80 kg P ha⁻¹ yr⁻¹ was 19, 24, 28, 38 and 67 mg P L⁻¹, respectively. Subsequently, withholding P fertilizer for two years from these plots, resulted in a significant decrease in the Olsen P status. In Jan/Feb 2007 Olsen-P was 16, 19, 23, 34 and 49 mg P L⁻¹, respectively. Fig. 1 shows the results of weekly soil sampling on the plots

receiving 0, 40 and 80 kg P ha⁻¹ yr⁻¹. The arrow indicates when a zero P fertilizer policy was introduced (March 2005).

SRP and TP concentrations in land drainage water and surface runoff from March 2005-February 2007 were generally within the range measured in the previous 5 year period, when P fertilizer was applied. However, interpretation is confounded by annual variation in hydrological events and flows as well as hydrological differences between plots.

Conclusions

The current study showed that the Olsen P status of grassland swards decreased in response to a zero P fertilizer policy. However, the time interval was not long enough to lower P losses to surface runoff or land drainage. Results to date suggest that improving water quality with respect to P may be a slow process under grazing management, unless P is re-distributed within the soil profile by, for example, ploughing for the reseeded of pastures.

References

- Murphy, J. & Riley, J.P., 1962. A modified single solution method for the determination of phosphorus in natural water. *Analytic. Chim.* 27, 31-36.
- Watson, C.J., Smith, R.V. & Matthews, D.I., 2007. Increase in phosphorus losses from grassland in response to Olsen P accumulation. *J. Environ. Qual.* (in press).

Effect of tillage and liming on the water-soluble phosphorus in the clay soil fields

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Introduction

In Finland, almost 60 % of phosphorus (P) load originates from agriculture (Vuorenmaa et al., 2002). No-tillage, a method adopted to reduce erosion and particulate P load from fields to watercourses, is one of the several recommended methods to diminish the agricultural P load. However, in the no-tilled fields, where seeds and fertilizers are input straight to the soil without any tillage, P tends to accumulate in the uppermost soil layer. This increases the risk of P leaching in dissolved form with surface runoff (Puustinen et al., 2005). The objective of the present study was to examine the effects of tillage practice on the extractability and leaching of P from clay soil limed recently or 22 years ago.

Material and methods

The experimental material was collected in autumn from two clay-rich fields both of which had two no-tilled (for five years) and two ploughed plots. One of the fields was unlimed (limed 22 years ago), while the other field had been limed six months before soil sampling by mixing 7,000 kg of $\text{CaCO}_3 \text{ ha}^{-1}$ into surface soil in connection with seedbed preparation. The unlimed field contained 46% clay (Eutric Cambisol) and the limed one 62% clay (Vertic Cambisol) in the soil layer of 0–20 cm.

For chemical analyses, we took the soil samples from the 0–2.5, 2.5–10 and 10–20 cm layers, four samples from each plot. The samples were air-dried, sieved through a 2 mm sieve and analyzed for dissolved reactive P (DRP) by shaking 1 g of soil in 50 ml of deionized water for 21 hours. The extracts were filtered through a 0.2 μm Nuclepore polycarbonate filter and the filtrate was analyzed for DRP by a molybdenum blue method (Murphy & Riley, 1962). Soil organic carbon (OC) was determined with a LECO CN–2000 analyzer, soil pH in water suspension (1:2.5), and the easily soluble calcium (Ca) by shaking 5 g of soil in 250 ml ammonium acetate (pH 4.65).

Furthermore, we took four undisturbed soil columns from each plot for leaching tests. A PVC-column (diameter 15 cm and height 20 cm) was hammered into the soil, plant residues were removed from the surfaces of the samples and the bottom surfaces of the soil columns were prepared with a knife and cleaned by vacuum to open the

pores. In the laboratory, the samples were saturated from the bottom upwards with deionized water for 1.5 hours and the percolation waters were collected over 0.5 h. This treatment was repeated three times. Thereafter the 0–5 cm layers were removed from the columns, saturated with water that was then allowed to leach on same way as the 20-cm columns. The percolation waters were analyzed for DRP as above.

Results

We found that no-tillage had increased organic C in the surface soil layer of both fields (Table 1). In the unlimed field it had also slightly lowered pH of the surface soil, whereas in the ploughed plots pH remained almost the same in all layers studied (Table 1). In the limed field, the effect of liming was distinct in the surface soil layer, where pH and the concentration of Ca were clearly elevated irrespective of cultivation method. The concentration of Ca was notably higher in the limed field compared to the unlimed field. Interestingly, in the no-tilled plots, DRP had accumulated in the surface layer of the unlimed soil, but not in the limed one (Table 1).

Table 1. Soil organic C (OC), pH_{water} , easily soluble calcium (Ca) and dissolved reactive P (DRP) in various soil layers of unlimed and limed plots of fields cultivated with different tillage practices.

	Layer (cm)	Unlimed field				Limed field			
		OC (%)	pH	Ca (mg kg^{-1})	DRP (mg kg^{-1})	OC (%)	pH	Ca (mg kg^{-1})	DRP (mg kg^{-1})
No-tillage	0–2.5	3.0	6.0	1900	35	3.4	6.8	5000	19
	2.5–10	2.5	6.1	2200	24	2.8	6.1	2600	21
	10–20	2.4	6.3	2400	23	2.7	6.2	2500	19
Ploughing	0–2.5	2.5	6.2	2200	22	2.7	7.0	3700	14
	2.5–10	2.5	6.0	2200	23	2.7	6.5	3200	14
	10–20	2.5	6.1	2200	23	2.6	6.2	2400	16

Regardless of the tillage method, the percolation waters from the soil columns taken from the limed field had on average less dissolved P than the waters from the columns taken from the unlimed field (Fig. 1). The same trend was found in the surface soil samples (0–5 cm) removed from the soil columns after the percolation test.

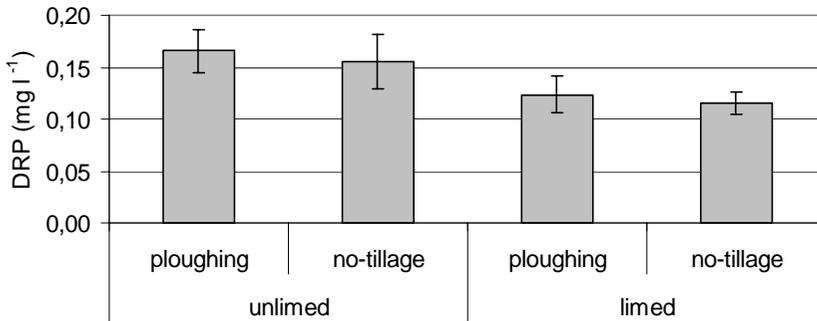


Figure 1. Dissolved reactive P (DRP) in the percolation waters from the undisturbed soil columns (height 20 cm). Each concentration is an average of three leaching events from eight soil profiles and the bars indicate the confidence interval of 90%.

Discussion

The switch from ploughing to no-tillage affects mainly the surface soil. When the soil is no longer ploughed, the residue of fertilizers and organic C easily accumulate in the uppermost soil layer, which can lower pH (Tarkalson et al., 2006, Muukkonen et al., 2007). Low pH further enhances the adsorption and saturation of P in the surface layer, and the risk of P leaching in surface runoff increases.

In this study, accumulation of P in the surface layer of the unlimed no-tilled plots was similar to the study of Muukkonen et al. (2007). However, the constant DRP concentration in the whole 0–20 cm layer of the limed no-tilled plots indicates that the liming-induced increase in soil pH had enhanced the solubility of P and, thus, promoted its uptake by plants and/or movement downwards in the soil profile.

On the other hand, the high concentration of Ca in the limed soil may have diminished DRP in the percolation waters, because divalent cations enhance the adsorption of P on the soil particles (Barrow & Shaw, 1979). We suggest that liming, combined with increasing the ionic strength of the soil solution is an interesting option for reducing the losses of dissolved P with surface runoff. However, we need more information about the liming-induced net changes in DRP, because an increase in pH is known to promote the solubility of P. In the future, we would be interested to know whether liming increases the lability of soil P even though the soluble P may be lowered through the increase in ionic strength of soil solution.

References

Barrow, N.J. & Shaw, T.C., 1979. Effects of ionic strength and nature of the cation on desorption of phosphate from soil. *Journal of Soil Science* 30, 53–65.

- Murphy, J. & Riley, J.P., 1962. A modified single solution method for the determination of phosphate in natural waters. *Analytica Chimica Acta* 27, 31–36.
- Muukkonen, P., Hartikainen, H., Lahti, K., Särkelä, A., Puustinen, M. & Alakukku, L., 2007. Influence of no-tillage on the distribution and lability of phosphorus in clay soils. *Agriculture, Ecosystems & Environment* 120, 299–306.
- Puustinen, M., Koskiaho, J. & Peltonen, K., 2005. Influence of cultivation methods on suspended solids and phosphorus concentrations in surface runoff on clayey sloped fields in boreal climate. *Agriculture, Ecosystems & Environment* 105, 565–579.
- Tarkalson, D.D., Hergert, G.W., Cassman, K.G., 2006. Long-term effects of tillage on soil chemical properties and grain yields of a dryland winter wheat–sorghum/corn–fallow rotation in the Great Plains. *Agronomy Journal* 98, 26–33.
- Vuorenmaa, J., Rekolainen, S., Lepistö, A., Kenttämies, K., Kauppila, P., 2002. Losses of nitrogen and phosphorus from agricultural and forested areas in Finland during the 1980s and 1990s. *Environmental Monitoring and Assessment* 76, 213–248.

Diffuse phosphorus concentration in overland flow from grassland and potential for mitigation

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Introduction

A large proportion of farmland in many countries in northern Europe is grassland. In Ireland 90% of the 4.2 million ha of farmland is grassland. Phosphorus (P) deficiency limited grassland production in Ireland and this was corrected by chemical fertiliser use in the 1960s and 1970s. There was an input of about 3m tonnes of P in chemical fertiliser and purchased feed, which was over double the removals in milk and meat, over the past 50 years and this contributed to an ten-fold increase (about 0.8 to 8 mg L⁻¹ soil, Morgan's P) in mean soil test P (STP). There was also an increase in nitrogen and other nutrient inputs. This led to increased intensification of grassland (doubling of grass yield and of grazing animal numbers, from about 3m to over 6m livestock units) which contributed to increased P loss from grassland to water. There is little information on the relative contribution to P loss of increased chemical fertiliser use compared to increased grazing animal numbers. The main objective of this paper is to present results of the dissolved reactive P (DRP) concentrations in overland flow from six grassland (cut and grazed field plots with a range of STP levels) and the implications for mitigation of P loss to water.

Methods

The nutrient concentrations and nutrient loads from six grazed and cut field plots (0.24 to 1.54 ha each) were studied in this experiment which started in September 2000 and finished in March 2004, at Teagasc, Johnstown Castle, Wexford. The six plots, Warren 1 (Plot 1; grazed 2001 and 200, cut 2002), Warren 2 (Plot 2; cut 2001 and 2003, grazed 2002), Cowlands 1 (Plot 3: grazed 2001-2003), Cowlands 2 (Plot 4: cut 2001-2003), Dairy 1 (Plot 5: grazed 2001-2002, cut 2003) and Dairy 2 (Plot 6: cut 2001-2002, grazed 2003) had STP of 3.5, 4.8, 17.9, 16.7, 7.0 and 7.2 mg L⁻¹ soil respectively (Figure 1).

Flow proportional overland flow samples were collected and analysed for P and N fractions; in addition some samples were analysed for potassium, and suspended solids.

Results

There were significant variations in DRP concentrations over the seasons and between the six field plots (Figure 1 and Table 1). Concentrations of DRP in overland flow varied from under 0.005 mg L⁻¹ to over 3 mg L⁻¹. There was a significant (P<0.01) linear relationship between STP in the six plots and mean annual DRP concentrations in overland flow for 2002 and 2003. There was more than a ten-fold difference in mean DRP concentrations in overland flow between plots with the lowest (Plot 1, 3.5 mg P L⁻¹ soil) and highest (Plot 3, 17.9 mg P L⁻¹ soil) STP (0.05 versus 0.59 mg L⁻¹ DRP in 2002 and 0.03 versus 0.72 mg L⁻¹ DRP in 2003). This compared to a maximum of 66% increase (0.43 (Plot 5) to 0.72 (Plot 3) mg L⁻¹ DRP for cut and grazed, respectively in 2003) that could be attributable to the presence of grazing animals.

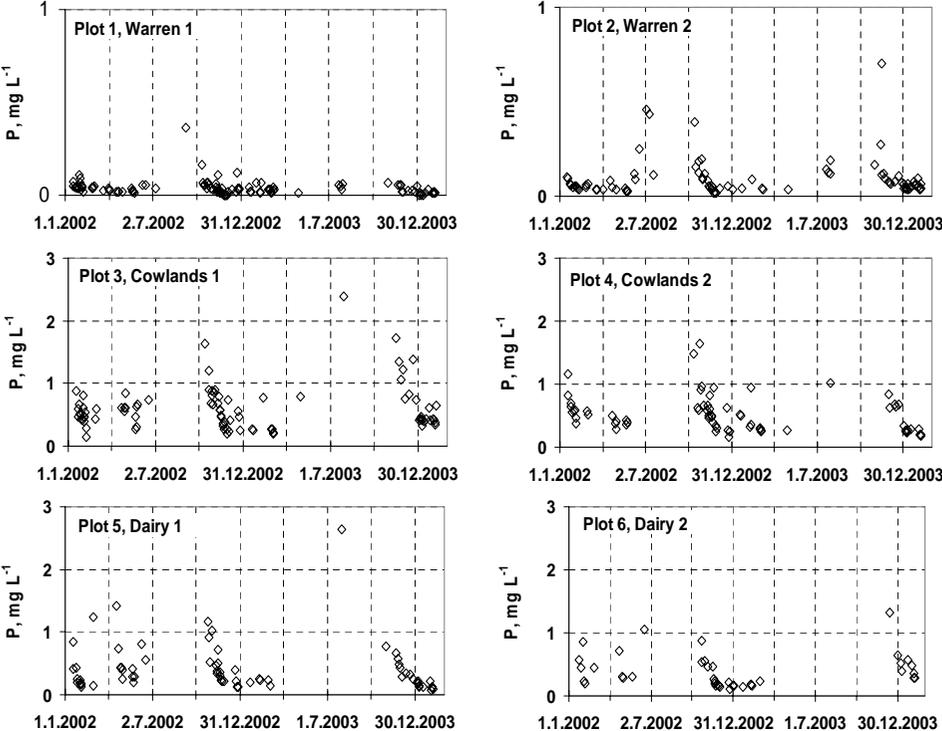


Figure 1. The mean daily DRP concentrations in overland flow, when overland flow occurred, from the six plots from January 2002 February 2004.

The estimated annual dissolved DRP loads in overland flow from the plots ranged from 0.1 to 1.2 kg ha⁻¹ year⁻¹. There was a significant correlation between the three P fractions measured, 86% of total P (TP) was total dissolved P (TDP), 77% of TP was DRP and 90% of TDP was DRP. Excluding a relatively small number of summer time

overland flows, the highest DRP concentrations and loads occurred in autumn when overland flow started (combining high flows and high P concentrations, Figure 1) after an extended summer dry period (autumn/winter wash-out effect).

The difference in mean DRP concentrations in overland flow between cut and grazed plots (most evident on the high STP plots) were generally highest at this time of year (autumn/winter). In contrast there was generally no difference between cut and grazed plots in January and February when concentrations were lowest and before the grazing season started.

Table 1. Seasonal mean concentrations of DRP (mg L^{-1}) in overland flow between paired grazed and cut plots at the Cowlands and Dairy sites, 2002 - 2004.

Site and treatment	Jan-Mar 2002	Apr-June 2002	Oct-Dec 2002	Jan-Mar 2003	Apr-Sep 2003	Oct-Dec 2003	Jan-Feb 2004
Cowlands							
Grazed (Plot 3)	0.48	0.64	0.567	0.250	1.593	1.010	0.433
Cut (Plot 4)	0.63	0.40	0.587	0.345	0.648	0.634	0.238
Significance	$p < 0.05$	$p < 0.001$,	NS,	$p < 0.05$	NS	$p < 0.05$	$p < 0.001$
Dairy (Plot 5, grazed in 2001-02, cut in 2003; Plot 6, cut in 2001-02, grazed in 2003)							
Grazed (Plot 5)	0.37	0.52	0.45	0.21		0.98	0.42
Cut (Plot 6)	0.33	0.54	0.31	0.17		0.83	0.15
Significance	NS	NS	NS	NS		NS	$p < 0.001$

Results of soil physical measurements in subplots in these field plots showed that grazed plots had significantly lower macroporosity (57-83%) and higher bulk density (8-17%) and resistance to penetration (27-50%) than cut plots. Rainfall simulation experiments on small plots (0.5 m^2) gave increased overland flow on grazed compared to cut areas.

Soil test P concentrations in soil under dung-pats were shown to increase 3 to 4 fold over the 90 day decomposition periods studied. Soil microbial biomass turnover of $51 \text{ kg P ha}^{-1} \text{ year}^{-1}$ was calculated for the Cowlands site (Plots 3 & 4).

The final reports of this work will be published in 2007 (www.epa.ie).

Conclusions

Three factors influencing DRP concentrations in overland flow were indicated in this work: a) the highest DRP concentrations were from plots with the highest STP and visa versa, b) a seasonal P cycle with high DRP concentrations in autumn/winter when overland flow commenced after the summer and decreasing over the following two months, c) relatively small difference in DRP concentrations between grazing and cutting treatments.

Overall the results indicate that high STP from surplus P inputs was a more important factor than the increased animal numbers in increasing P loss from grassland to water in Ireland. It is concluded that the P loss in overland flow from grassland can be increased by the presence of grazing animal. The main potential for P loss mitigation from grassland is to maintain soils at or near the lowest STP level compatible with good grassland production on soils subject to P loss in overland flow and match P inputs with outputs in milk and meat.

Mining soil phosphorus by zero P application: an effective method to reduce the risk of P loading to surface water

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Introduction

Leaching of phosphorus (P) from agricultural soils is an important factor in the ongoing deterioration of surface water quality in the Netherlands. To improve the quality of surface waters, the Dutch government has implemented a generic manure policy in the 90's. Despite this, the accumulation of P in soil has continued. At present more than 50 % of the Dutch agricultural soils are saturated with P (Schoumans, 2004). Recently, the Dutch manure policy has been adapted, and the application standard of P will be gradually decreased until equilibrium between P application and plant uptake (equilibrium fertilization) will be reached in 2015. Equilibrium fertilization will stop the ongoing enrichment of the soil with P and is a prerequisite to improve surface water quality. However, these reductions in P application rates are not sufficient to reach surface water quality standards of the European WFD in 2015 in all sensitive areas. Accordingly, additional measures have to be considered to further reduce P loading to surface water (e.g Chardon and Koopmans, 2005). Mining of soil P by crop harvesting without P application is a promising method to reduce the risk of P leaching to surface waters within a reasonable timeframe. Greenhouse experiments have shown a rapid reduction of the P concentrations in soil solution and readily available soil P with zero P application (Koopmans et al., 2004). Field data confirming these findings are scarce thus far. The aim of this study is to find field evidence for the effectiveness of P-mining to reduce the risk of P leaching to surface waters.

Experimental design

In 2002, a P-mining experiment started on four grassland sites on sand (two sites), peat and clay soils. The mining plots received no P and an annual N surplus of 300 kg ha⁻¹ yr⁻¹. Grass was removed by mowing five to seven times a year. At the same sites, effects of P surpluses varying between 0 and 40 kg P₂O₅ ha⁻¹ yr⁻¹ were studied on adjacent plots. This gives the opportunity to compare the effect of P-mining with regular manure policy. All terms of the soil P balance (i.e., P application, P plant uptake and P leaching) were measured. At each plot, soil samples were collected in late winter (before fertilization), early spring (after fertilization) and in late autumn at depths of 0-5, 5-10, 10-20 and 20-30 cm. Soil solution samples were obtained by centrifugation of field-moist samples. Soil solution was analyzed for total-P and molybdate reactive P (MRP) after centrifugation over a 0.2 µm filter. Concentrations

of molybdate unreactive P (MUP) were calculated as the difference between total-P and MRP. Soil samples were analyzed in autumn and incidentally in late winter for water-extractable P (P_w), P-AL, NH_4 -oxalate extractable P (P_{ox}), Al (Al_{ox}) and Fe (Fe_{ox}) and total-P (Table 1).

Table 1. P_w and P_{ox} at the start of the P-mining experiments in 2002.

Site	Soil type layer	P_w (mg $\text{P}_2\text{O}_5 \text{ l}^{-1}$)				P_{ox} (mg P kg^{-1})			
		0-5	5-10	10-20	20-30	0-5	5-10	10-20	20-30
		cm	cm	cm	cm	cm	cm	cm	cm
Heino	Sand	41	30	32	23	20	19	20	17
Cranendonck	Sand	38	27	26	21	20	19	19	16
Waiboerhoeve	Clay	51	20	10	9	19	12	9	8
Zegveld	Clay	36	20	9	5	45	41	32	21

Effect of zero P application

Soil solution concentrations fluctuated considerably during the four-year period making it difficult to show trends in P concentrations in relation to different P surpluses. Average P concentrations over the period 2002-2005, however, differed between the various treatments. In the upper soil layer (0-5 cm), the reduction in MRP concentrations at zero P application ranged from less than 40% in the sandy soil at Heino to more than 90 % in the clay soil (Figure 1). At 5-10 cm depth, the reduction in MRP concentrations ranged from 10-70 % but differences were not significant. In deeper layers, the reduction became even less prominent. MUP concentrations in the upper soil layer also declined at zero P application (Figure 1). However, the reduction in MUP was on average smaller than the reduction in MRP, and ranged from 30% at the sandy soil in Heino to 60-80% in the clay soil. At 5-10 cm depth, the decline in MUP was approximately 40%, but changes were only significant in the sandy soil at Heino.

Water-extractable P (P_w), which is the sum of P in soil solution and part of the reversibly bound P, showed a similar but slightly lower decrease at zero P application than the MRP concentrations in soil solution. The reduction in P_w amounted to 60-70% in the clay soil in the upper soil layer and to 20-30 % in the peat and sandy soils. In the deeper soil layers, the decline was smaller and generally not significant, except for the clay soil where P_w declined by 60%. P-AL, which contains both reversibly bound P and part of P bound inside metal oxides P, declined by 50-60% in the clay soil, by 30-50% in the peat soil and by 10-30% in the sandy soils. Again decline in P-AL was mainly restricted to the upper soil layer, whereas changes in the deeper soil layers were not significant yet. Significant changes in soil P pools which contain more strongly bound P forms (i.e., P_{ox} and total-P) were not found during the first four years of zero P application.

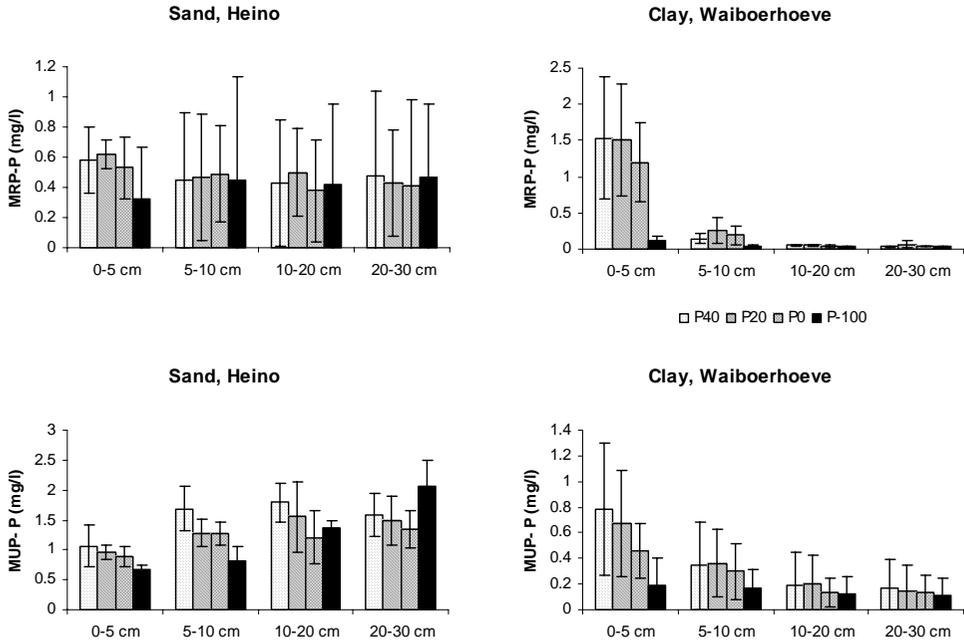


Figure 1. Average concentration and standard deviation of MRP and MUP in soil solution over the period 2002-2005 at P surpluses of 40, 20 and 0 kg P₂O₅ ha⁻¹ yr⁻¹ and zero P application (P-100). Note differences in scale on y-axis.

Conclusions

Mining soil phosphorus by zero P-application over a period of four years led to a strong (30-90%) reduction in both MRP and MUP concentrations in soil solution in the upper soil layer (0-5 cm). The reduction declined with depth and changes were generally not significant in the deeper soil layers. Mining also led to a decline in P pools in the soil solid phase. The largest decline was found in P_w and P-AL, whereas reductions in the more strongly bound P_{ox} and total-P were not significant yet. Since readily available P forms strongly decreased upon P-mining, the risk of P leaching was reduced already within a relatively short time frame of four years.

References

- Chardon, W.J. & Koopmans, G.F., 2005. Phosphorus Workshop. J. Environ. Qual. 34, 2091-2092.
- Koopmans, G.F., Chardon, W.J., Ehlert, P.A.I., Döfling, J., Suurs, R.A.A., Oenema, O. & van Riemsdijk, W.H., 2004. Phosphorus availability for plant uptake in a phosphorus-enriched noncalcareous sandy soil. J. Environ. Qual. 33, 965-975.
- Schoumans, O.F., 2004. Inventarisatie van de fosfaatverzadiging van landbouwgronden in Nederland. Rapport 730.4, Alterra, Wageningen.

Phosphorus forms and phosphorus release as affected by organic lowland geochemistry

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Introduction

A considerable effort is being directed towards reducing agricultural phosphorus losses to the aquatic environment. Lowland forms the boundaries to the aquatic environment, and management of lowlands may therefore be crucial for the water quality of adjacent waters. In this context focus has been directed towards estimating phosphorus losses from agricultural lowland. In addition, restoration of wetlands on reclaimed agricultural lowland has been recognized as a possible mitigation measure towards reducing nutrient losses to the aquatic environment. However, an increasing number of studies have called to the attention that wetland restoration on former agricultural soils may result in phosphorus release (e.g. Turner and Haygarth, 2001; Scalenghe et al., 2002). Clearly, there is a need for developing simple models based on easily accessible soil parameters for predicting potential phosphorus release from agricultural lowland and re-established wetlands. The objectives of this study were to enhance the knowledge on lowland geochemistry, phosphorus forms and phosphorus release in the main types of Danish organic lowland soils, and evaluate the possibility of using easily accessible soil parameters for phosphorus risk assessments.

Organic lowland types

The lowland area of the western part of Denmark (Jutland) covers roughly 6000 km², of which 65% is agricultural lowland. About 50% of the Jutland wetlands have been classified as potentially acid sulfate soils (Madsen et al., 1985). Four classes differing in acidification potential (ochre classes) have been defined. The soil survey showed that the potentially acid sulfate soils occur more frequently in the western part of Jutland where the old Saale moraine landscapes, outwash plains and marsh areas dominate, than in the northern and eastern part of Jutland where Weichsel moraine landscapes dominate (Madsen et al., 1985). In this study 30 lowland types were selected based on geological region, surface geology and ochre classes. From each site soil was sampled at 3-4 depths to 1 m. Soil was characterized with respect to bulk density, ash content, pH, total P (TP), oxalate extractable iron (Fe_{ox}), aluminium (Al_{ox}) and phosphorus (P_{ox}), bicarbonate-extractable iron (Fe_{BD}) and phosphorus (P_{BD}), and citrate-bicarbonate-extractable iron (Fe_{CBD}) and phosphorus (P_{CBD}). The composition of P-fractions was determined from a 5-step sequential fractionation as

described by Paludan and Jensen (1995). Additionally, phosphorus release during re-wetting was investigated in a 100-day batch release experiment. Suspensions of soil in demineralised water (soil:water ratio ~1:25) were incubated in the dark at 10°C in air-tight centrifugation bottles. At each sampling date, suspensions were centrifuged 20 min at 5000 g and the supernatant was used for analysis of pH, dissolved reactive P (DRP), total dissolved P (TDP) and total dissolved Fe.

Phosphorus contents and forms as affected by lowland geochemistry

The plough-layer content of TP was extremely variable from 450-10.500 mg kg⁻¹ (Figure 1) corresponding to values of 200-14.014 kg ha⁻¹. Compared to mineral agricultural soils, with TP from 300-900 mg kg⁻¹ (1125-3375 kg ha⁻¹), some organic lowland soils seemed to be rather enriched in phosphorus. The large variation in TP reflects large variations in phosphorus adsorption properties, as indicated by the strong correlation between the content of Fe_{CBD} (3000-135.000 mg kg⁻¹) and TP (Figure 1). The generally low content of Fe_{ox} and Al_{ox}, and the poor correlation between phosphorus content and oxalate-extractable sesquioxides, indicate that crystalline iron constitutes the major binding capacity of these soils.

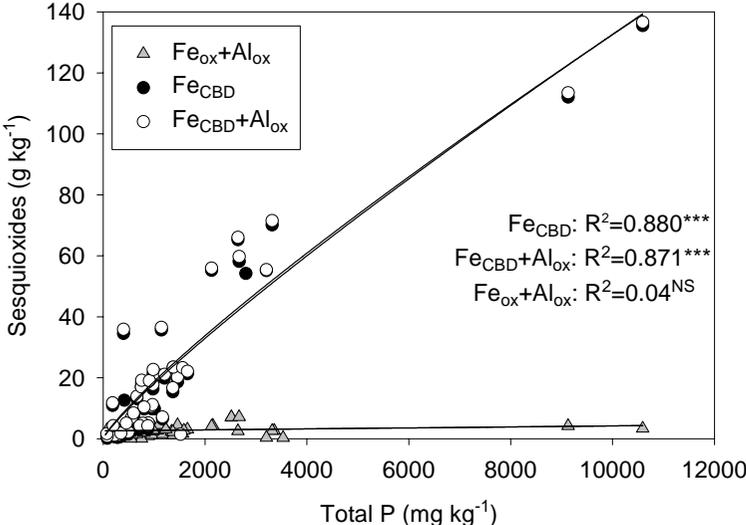


Figure 1. Correlations between fractions of sesquioxides (Fe_{CBD}, Fe_{ox}, Al_{ox}) and total phosphorus (TP).

Phosphorus release following lowland re-wetting

Dissolved reactive P (DRP) release rates were investigated during 100 days of anaerobic incubation. The rate of Fe(III)-reduction was well described by a Monod growth rate expression ($R^2=0.77^{***}$), where Fe_{BD} represents the initial substrate concentration. The soil Fe_{BD}:P_{BD}-molar ratio was the parameter that correlated best

with the DRP release rate, when the DRP-release was normalized with respect to concentration of Fe_{BD} (Figure 2).

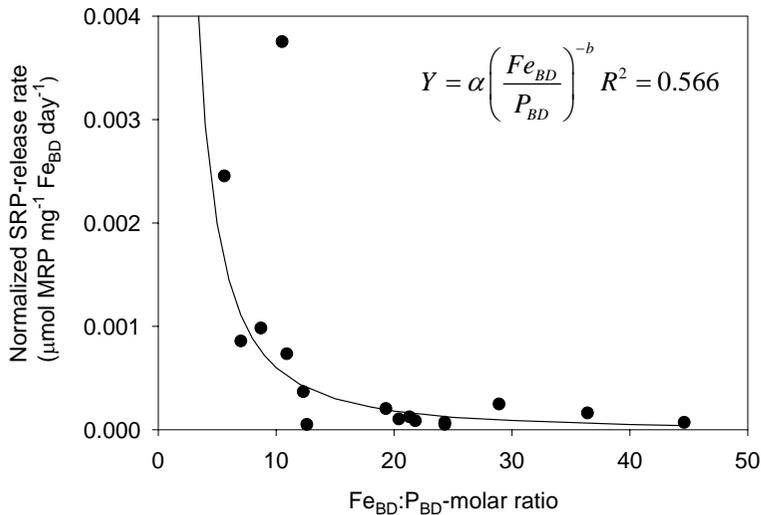


Figure 2. Phosphorus release rate (day 0-7) normalized to Fe_{BD} -content, as a function of the soil Fe_{BD} : P_{BD} -molar ratio.

Conclusions

Danish organic lowlands exhibit an extremely diverse geochemistry. Crystalline iron constitutes the major binding component, as indicated by the strong correlation between Fe_{CBD} and TP. Many lowlands contain very high amounts of iron-bound phosphorus, and may constitute a potential problem with respect to P-release during anaerobic conditions as well as wetland restoration. The soil Fe_{BD} : P_{BD} -molar ratio seems to be a valuable tool for predicting the potential risk of P-release at anaerobic conditions.

References

- Madsen, H.B., Jensen, N.H., Jakobsen, B.H. & Platou, S.W., 1985. A method for identification and mapping potentially acid sulfate soils in Jutland, Denmark. *Catena* 12, 363-371.
- Paludan, C. & Jensen, H.S., 1995. Sequential extraction of phosphorus in freshwater wetland and lake sediment: Significance of humic acids. *Wetlands* 15(4), 365-373.
- Scalenghe, R., Edwards, A.C., Marsan, F.A. & Barberis, E., 2002. The effect of reducing conditions on the solubility of phosphorus in a diverse range of European agricultural soils. *European J. Soil Sci.* 53, 439-447.
- Turner, B.L. & Haygarth, P., 2001. Phosphorus solubilization in rewetted soils. *Nature* 411, 258.

Hydrological pulsing and grass species effects on nutrient retention in soils of differing microbial community composition

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Introduction

Drying and rewetting (D/RW) events in soil and physiological stresses that impact on the soil microbial community, especially in the soil surface layer (Lundquist *et al.*, 1999). The aim of this study was to test the following hypotheses: (1) that differences in soil properties, including changes in soil microbial community composition, resulting from contrasting and long-term grassland management alter the ability of soils to retain phosphorus (P) and nitrogen (N) under D/RW stress, i.e., hydrological pulsing; and (2) plants are able to alter the soil microbial biomass, and hence the concentrations of P and N lost through leaching under D/RW stress. This paper will present the key findings related to P from this study. Overall, the results will contribute to our knowledge of the role of soil microbes in P loss to water, in relation to soil wetting and drying perturbations.

Methodology

To test these hypotheses soil was taken from two adjacent, sheep-grazed grasslands of different long-term management, located at Littledale, Lancashire, United Kingdom (54°3' N, 2°42' W). One had not received any fertilizer applications and was grazed at low stocking densities (1-2 ewes ha⁻¹) (unimproved soil), whereas the other had received regular applications of fertilizer (NPK) and farmyard manure, and was heavily grazed (10-15 ewes ha⁻¹) (improved soil). Previous studies show that these 'improved' and 'unimproved' soils favour bacterial and fungal microbial communities, respectively (Grayston *et al.*, 2001). A microcosm study was conducted with treatments consisting of combinations of the following factors: (1) grassland type (improved or unimproved); (2) grass species (*Agrostis capillaris* or *Deschampsia flexuosa*); (3) hydrological pulsing (5 x 20 d cycles): pulsing (maintained at 55% WHC for 10 d, then dried to and maintained at 30% WHC over 5 d), and non-pulsing (maintained at 55% WHC for 15 d per cycle), both treatments were then leached using double de-ionized water at 0, 1, 5, 12, 24, 48 and 96 hours, then returned to 55% WHC. (4) sampling day (destructive harvests after cycles 1, 3 and 5). The parameters measured during the destructive harvests (after leaching cycles 1, 3 and 5) included: microbial biomass C, N and P; and, phospholipid fatty acid (PLFA) analysis to measure changes in microbial community structure. The parameters

measured during the leaching phase of each hydrological pulsing cycle were leachate dissolved C and N (organic and inorganic (DOC, DON and DIN)), and total, reactive and unreactive P (TP, TRP and TUP, respectively).

Results

Our data show that hydrological pulsing significantly ($F_{1, 56} = 13.64$, $P < 0.001$) decreased microbial biomass P by 14% in the unimproved soil, but had no effect on this measure in the improved soil. Total PLFA ($F_{1, 72} = 8.45$, $P < 0.01$) increased in both soils due to hydrological pulsing, and this stress also caused the ratio of fungal-to-bacterial PLFA ($F_{1, 72} = 3.99$, $P < 0.05$) to decrease in the improved soil, but had no effect on this measure in the unimproved soil. However, hydrological pulsing had no effect on TP and TUP concentrations in soil leachates; the leachate cycle had the greatest effect on the concentration of leachate TP ($F_{4, 672} = 47.04$, $P < 0.001$) and TUP ($F_{4, 668} = 47.88$, $P < 0.001$) in both soils, with concentrations decreasing from cycle one to cycle four then increasing sharply during the fifth cycle. The grass species treatment had only minor effects on the parameters presented in this paper.

Discussion

We set out to test the hypothesis that differences in soil properties, including shifts in soil microbial community composition, resulting from long-term differences in grassland management, determine the ability of the soil microbial community to retain phosphorus and nitrogen under physical stress, caused by drying and rewetting. Consistent with other studies we found that the unimproved soil had a greater abundance of fungi relative to bacteria than the improved soil (Bardgett *et al.*, 2003, Grayston *et al.*, 2004). Hydrological pulsing had no effect on the ratio of fungal-to-bacterial PLFA in the unimproved soil; however, in the improved soil this stress decreased this ratio, it is possible that the pulsing nature of this drying/ rewetting stress lysed a more susceptible part of the microbial community (Turner and Haygarth, 2001; Turner *et al.*, 2003) but then encouraged the growth of a more stress resistant part of the bacterial biomass. Our data show that hydrological pulsing decreased microbial biomass P, but only in the unimproved soil, and had no effect on the concentrations of leachate TP and TUP from either soil. The decrease in microbial biomass P in the unimproved soil would therefore not appear to be related to changes in microbial community composition caused by hydrological stress. The decrease in microbial biomass P may have been caused by a combination of cell death/ lysis (Grierson *et al.*, 1998; Turner and Haygarth, 2001; Turner *et al.*, 2003; Wu and Brookes, 2005), and osmotic regulation (Halverson *et al.*, 2000; Fierer and Schimel, 2003). The P lost from the soil microbial biomass was not released in leachates as evidenced by the lack of effect of hydrological pulsing on leachate TP and TUP concentrations; it is possible that this P may have been retained within the soil by being adsorbed to soil colloids.

Conclusions

Overall, our findings suggest that management induced changes in soil microbial communities can significantly affect the retention of P after hydrological pulsing. Such differences in microbial nutrient retention under stress are likely to have far reaching implications for the ability of different grassland systems to retain nutrients and hence the loss of nutrients to groundwater and waterways.

References

- Bardgett, R.D., Streeter, T. & Bol, R., 2003. Soil microbes compete effectively with plants for organic nitrogen inputs to temperate grasslands. *Ecology* 84, 1277-1287.
- Fierer, N. & Schimel, J.P., 2003. A proposed mechanism for the pulse in carbon dioxide production commonly observed following the rapid rewetting of a dry soil. *Soil Science Society of America Journal* 67, 798-805.
- Grayston, S.J., Griffith, G.S., Mawdsley, J.L., Campbell, C.D. & Bardgett, R.D., 2001. Accounting for variability in soil microbial communities of temperate upland grassland ecosystems. *Soil Biology & Biochemistry* 33, 533-551.
- Grayston, S.J., Campbell, C.D., Bardgett, R.D., Mawdsley, J.L., Clegg, C.D., Ritz, K., Griffiths, B.S., Rodwell, J.S., Edwards, S.J., Davies, W.J., Elston, D.J. & Millard, P., 2004. Assessing shifts in microbial community structure across a range of grasslands of differing management intensity using CLPP, PLFA and community DNA techniques. *Applied Soil Ecology* 25, 63-84.
- Grierson, P.F., Comerford, N.B. & Jokela, E.J., 1998. Phosphorus mineralization kinetics and response of microbial phosphorus to drying and rewetting in a Florida Spodosol. *Soil Biology & Biochemistry* 30, 1323-1331.
- Halverson, L.J., Jones, T.M. & Firestone, M.K., 2000. Release of intracellular solutes by four soil bacteria exposed to dilution stress. *Soil Science Society of America Journal* 64, 1630-1637.
- Lundquist, E.J., Scow, K.M., Jackson, L.E., Uesugi, S.L. & Johnson, C.R., 1999. Rapid response of soil microbial communities from conventional, low input and organic farming systems to a wet/dry cycle. *Soil Biology and Biochemistry* 31, 1661-1675.
- Turner, B.L. & Haygarth, P.M., 2001. Phosphorus solubilization in rewetted soils. *Nature* 411, 258.
- Turner, B.L., Driessen, J.P., Haygarth, P.M. & McKelvie, I.D., 2003. Potential contribution of lysed bacterial cells to phosphorus solubilisation in two rewetted Australian pasture soils. *Soil Biology and Biochemistry* 35, 187-189.
- Wu, J. & Brookes, P.C., 2005. The proportional mineralisation of microbial biomass and organic matter caused by air-drying and rewetting of a grassland soil. *Soil Biology and Biochemistry* 37, 507-515.

The impacts of organic matter incorporation and hydrological stress on microbial biomass phosphorus dynamics

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Introduction

Agricultural soils often receive large inputs of substrate, e.g. straw and manure. With anticipated/recent changes to legislation, compost and sewage-sludge inputs will also increase. With increased inputs of these substrates to soil, the size of the microbial biomass will increase and will mineralise or immobilise phosphorus (P) depending on the substrate P concentration and the requirement of the biomass for P. When the substrate is exhausted the microbial biomass will decline rapidly, releasing previously immobilised P into the system. Hydrological stresses, e.g. waterlogging and air drying-rewetting, kill appreciable quantities of biomass and may increase the amounts of organic and inorganic P leached from the soil (Turner & Haygarth, 2001; Turner *et al.*, 2003).

The objective of this study was to determine the impacts of organic matter incorporation and hydrological stresses on microbial P pool sizes and the potential for loss of this P in soil drainage water.

Methods

Grassland soil was sieved <2 mm and pre-incubated for one week at 25°C and 40% water holding capacity (WHC). Dried and ground pasture (grass) and <2 mm sieved farm yard manure (FYM) were added on day 0 at a rate of 5000 µg C g⁻¹ soil and kept at 25°C and 40% WHC. On day 7 different water regimes were applied. One set of soils was kept at 40% WHC (40% WHC), another set was kept waterlogged under anaerobic conditions (waterlogged) and a third set went through cycles of air-drying and rewetting (AD/RW). Samples were taken over a period of 140 days and analysed for biomass C, biomass P, ATP and water-extractable P.

Results

Addition of substrates caused an increase in the biomass C and P concentrations in the soils. Biomass C and ATP (results not given) followed the same trend as biomass P (Figure 1). Following addition of substrates, biomass P increased from 92 mg kg⁻¹ to 109 mg kg⁻¹ (FYM) and 111 mg kg⁻¹ (grass). When kept at 40% WHC,

biomass P gradually decreased and by day 140 all samples contained 70 to 75 mg kg⁻¹ with no differences between substrates (Figure 1). Air drying and rewetting caused a rapid decline in biomass. Although this fluctuated initially, depending on the timing of sampling within the drying cycle, the biomass in soils given FYM were the same as in the control after 28 days, while the biomass in soils given grass generally remained higher for longer (Figure 1). By day 140 the biomass had decreased to 15 to 20 mg biomass P kg⁻¹ soil in all treatments. Waterlogging caused an immediate decrease in biomass. By day 16 there was no difference between substrate treatments and by day 28 biomass P had reached its minimum and remained at about the same level for the remainder of the experiment (Figure 1).

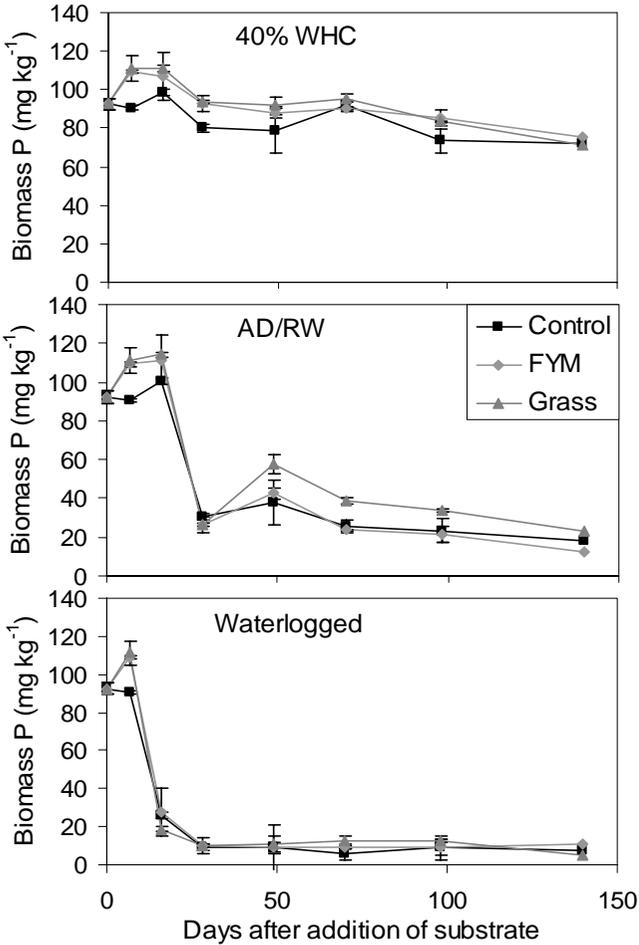


Figure 1. Changes in biomass P in soils with substrates added and different water regimes applied.

Biomass C followed the same trends as biomass P; however addition of FYM did not significantly increase the biomass C concentration, while grass did.

Cumulative water-extractable P was 3-4 times larger in FYM treated soils than grass or control soils for all water regimes (Table 1). Air drying and rewetting caused an increase in the amount of water-extractable P in the first 28 days and although this did decrease with further AD/RW cycles it was still 2-4 times higher than the other water regimes for all substrate treatments. Waterlogging caused an initial increase in water-extractable P (day 16) but this quickly decreased again and for the rest of the experiment it was significantly lower or not significantly different to the control.

Table 1. Water-extractable P concentrations ($\mu\text{g l}^{-1}$) at each sample time and total concentrations ($\mu\text{g l}^{-1}$) for different water regimes and substrates applied.

Day	40% WHC			AD/RW			WATERLOG		
	Control	FYM	Grass	Control	FYM	Grass	Control	FYM	Grass
0	10.1								
7	3.2	111.9	11.3						
16	15.4	85.0	12.7	11.1	87.0	13.0	57.3	86.7	27.4
28	5.6	46.1	13.6	45.8	150.1	75.7	8.3	24.2	10.4
49	4.3	38.4	11.7	23.9	71.8	21.4	14.2	14.4	14.4
70	4.5	22.3	6.5	20.6	47.3	28.6	6.8	13.9	9.1
98	5.1	29.9	8.2	14.6	42.2	23.2	5.0	6.3	13.8
140	17.3	29.7	16.1	23.4	38.6	12.0	13.9	10.2	4.8
Total	65.5	373.5	90.2	152.7	559.1	195.3	118.8	277.7	101.2

Conclusions

Hydrological stresses had a greater impact on biomass concentrations than substrate type but substrate type had a greater impact on water-extractable P. The FYM sample was obviously well degraded as demonstrated by the lack of response in biomass C, but contained large amounts of P which initially were very soluble (water-extractable P). Larger concentrations of water-extractable P were found when the biomass had decreased rapidly (i.e. day 16 when waterlogged and day 28 when AD/RW) which may be due to the dead biomass cells being lysed and releasing PO_4 .

References

- Turner, B.L. & Haygarth, P.M., 2001. Phosphorus solubilization in rewetted soils. *Nature* 411, 258.
- Turner, B.L., Driessen, J.P., Haygarth, P.M. & McKelvie, I.D., 2003. Potential contribution of lysed bacterial cells to phosphorus solubilisation in two rewetted Australian pasture soils. *Soil Biol. & Biochem.* 35, 187-189.

The influence of eco- and agro- practices on the fate and transport of phosphorus from altered wetland soils to waterways

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Introduction

The spatiotemporal changes in the hydrogeology of the Hula altered wetland may influence the water quality of Lake Kinneret, which provides up to 30% of the potable water for the state of Israel. Current management practices in this altered wetland call for rewetting of formally oxidized peat soils to prevent further soil deterioration. We hypothesized that the rewetting has generated an increase of P transport from the soils to waterways.

The main objectives of this work were (1) to characterize the groundwater flow pathways in this altered wetland (2) to assess the potential impact on water quality downstream and (3) to test the above hypothesis using large-scale ($> 1 \text{ km}^2$) field experiment in fractured soils by manipulating the water levels in the drainage canals.

Methods

We conducted variography of hydraulic heads, constructed decision-tree model of major ions, determined the hydraulic conductivity (K), and $\delta^2\text{H}/\delta^{18}\text{O}$ ratios to ascertain the spatial and vertical distribution of hydrogeological parameters. We performed large-scale field experiments ($\geq 1 \text{ km}^2$) and determined the connectivity between the waterways and the wetland's aquifer. Finally, we examined the difference between preferential flow and pore water (PW) to compute realistic water and P budgets.

Results

On the basis of the variography and decision-tree model we concluded that the wetland aquifer is fragmented by 3 parent-materials, namely deep peat, shallow peat/marl complex and marl sub-aquifers. The tree-based model, the isotopic ratios and K determinations suggest that the deep peat is composed of one homogeneous layer characterized with low K (0.001 m d^{-1}), thus even though it stores much water it does not function as an active aquifer. The other two sub-aquifers are consists of 3 hydrostratigraphic layers; (i) vadose zone, (ii) layer with well developed macropores at depth of 1.5 to 4m, and (iii) an aquitard layer at depth of 4 to 15 m (Figure 1).

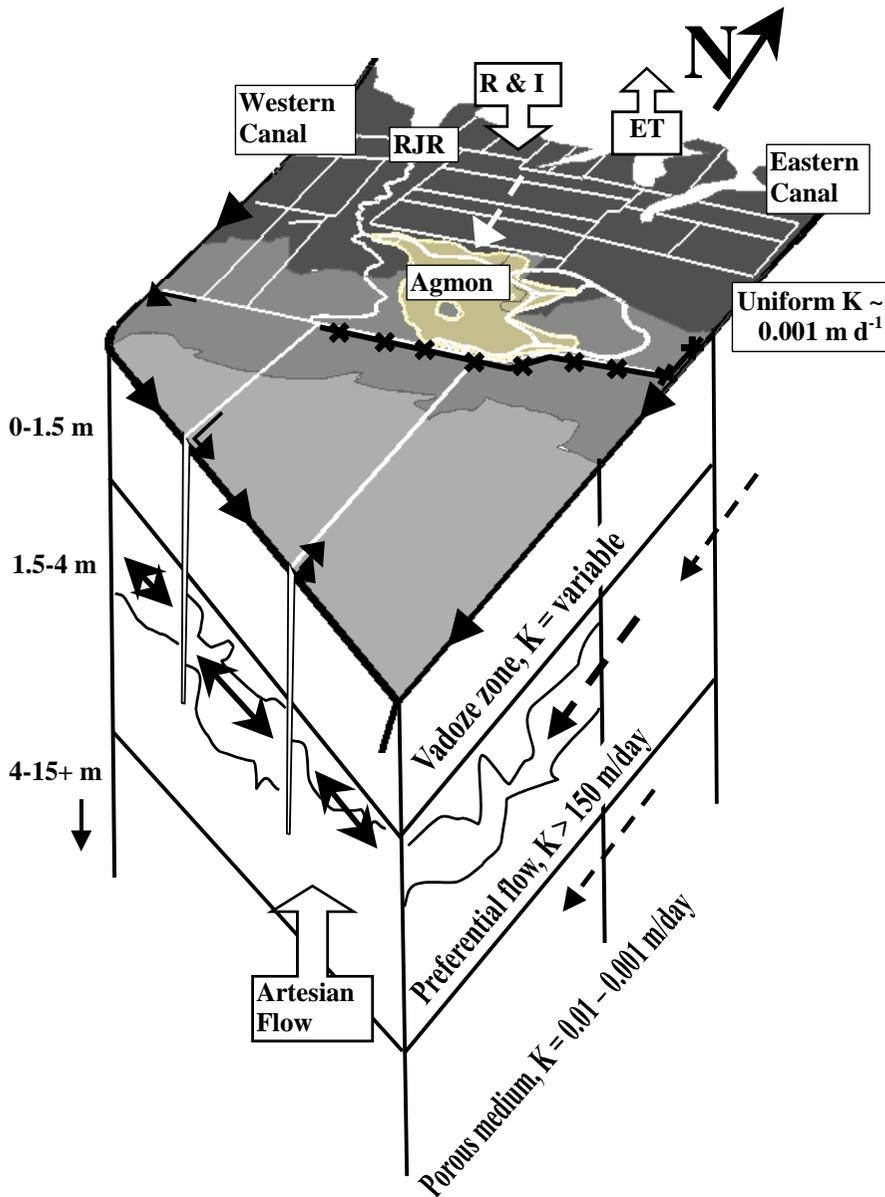


Figure 1. Conceptual model of the altered wetland Hula Valley, Israel.

The temporal head fluctuations, the high K values of the second layer ($> 150 \text{ m d}^{-1}$), and the large volume of water flowing to and out of the two sub-aquifers during large-scale field experiments, all attest to very rapid connectivity of the fields with the waterways. The upper layer of the shallow peat/marl complex was found to be very responsive to short term (several weeks) flooding. In such events, the redox fell

within few days from pe + pH of 14 to 3. The reduction of this soil released as much as $50 \text{ kg P ha}^{-1} \text{ yr}^{-1}$. The capacity of the underlying layer in its reduced state to retain the released P by rapid adsorption is low because the equilibrium P concentrations at zero adsorption (EPC_0) is high, thus allowing P to be transported through this layer.

During the first phase of the large-scale field experiment, water flowed from the drainage canal into the field and the electrical conductivity (EC) rose from $2500 \mu\text{S/cm}$ to $3300 \mu\text{S/cm}$ and from 1100 to $1400 \mu\text{S/cm}$ in the PW and preferential flow, respectively. The soluble reactive P (SRP) concentrations in the PW increased from 40 to $130 \mu\text{g/l}$ but decreased in the preferential flow from 40 to $30 \mu\text{g/l}$. During the second phase of the experiment, the water level in the drainage canals were lowered and water flowed from the field into these canals yielding an increase of EC and SRP in the preferential flow to $1800 \mu\text{S/cm}$ and $60 \mu\text{g/l}$, respectively. This experiment showed that fast flowing waters are important contributor of dissolved ions including SRP into the waterways and will impact water quality down stream.

On the basis of the above results, we adapted the concept of critical source area which claimed that most of P loss in a catchment derived from small areas in which efficient P release processes and low P retention capacity coincide with specific transport mechanisms. The high connectivity of this altered wetland coupled with the particular spatiotemporal flow attributes of the Jordan River and drainage canals provide this particular combination which partially explains the observed P load in the river.

Extraction tests in predicting potential phosphorus load from pasture soil

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Introduction

In pasture soils, phosphorus (P) has been found to enrich the surface soil especially in feeding and queuing areas (Jansson and Tuhkanen 2003). High P concentrations in the soil surface increase the risk of surface runoff loss of P (Sharpley and Withers 1994, Turtola and Yli-Halla 1999). Urination may further increase the P loading risk: increase in pH resulting from urea hydrolysis has been shown to cause contemporary increase in mobility of P and DOC in the soil (Hartikainen and Yli-Halla 1996, Shand et al. 2000). The objective of this work was to examine the effects of cattle urine and dung on the P chemistry in the pasture soil and to compare the sensitivity routine P-test methods (water extraction, P_w , and acid ammonium acetate extraction, P_{Ac} , used for soil testing in Finland) in assessing the P loading risk.

Material and methods

This study was done in a field by adding urine and dung in amounts corresponding to single excrement portions to the soil. Soil sampling in the urine and control patches was done 0, 1, 3, 5, 10, 21, 49, 77 and 120 days after the beginning of the experiment. The sampling of the dung patches started on day 3, and continued as above. Sampling depths were 0-2 cm and 2-10 cm. Soil samples were analysed for pH_{H_2O} , water extractable $P_{1:50}$ (PO_4-P_w) and acid ammonium acetate extractable P (P_{Ac}).

Results and discussion

In the uppermost soil layer (0-2 cm), the urine addition increased the soil pH immediately, but the effect vanished after 10 days. The concentration of PO_4-P_w followed closely the urine-induced increase in pH. This response was not observed in P_{Ac} where the low pH (4.65) of the extractant enhanced the resorption of P. In the top soil layer of the dung-treated plots, the pH increased towards the end of the trial. There was a concurrent increase in PO_4-P_w and P_{Ac} but the increase was a result of P added to soil in dung rather than a result from pH changes (Figure 1). The urine addition increased the pH also in the 2-10 cm soil layer and the effect vanished in 5 days (data not shown). However, PO_4-P_w or P_{Ac} concentrations were not increased. Dung addition increased the PO_4-P_w and P_{Ac} also in the 2-10 cm layer after 49 days, but the increase was smaller than in the 0-2 cm surface layer. Surface of the dung patch dried delaying the P movement into soil.

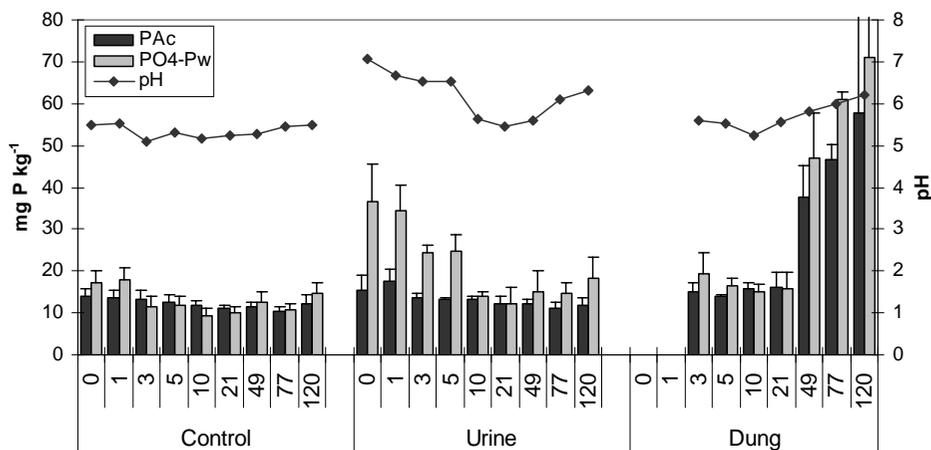


Figure 1. Soil pH, acid ammonium acetate extractable P (P_{Ac}) and water extractable P (PO_4-P_w) in the top soil (0-2 cm) in the control and after urine and dung additions. The error bars present the standard deviation of the mean.

Conclusions

In pasture soils, the interaction of urine addition and high P concentration in the top soil caused by repeated dung loading may lead to a high potential for P loss. According to these results, both water extraction and acid ammonium acetate extraction can be used to detect critical source areas in the pasture soil. However, the urine-induced increase in soil soluble P is not detected if acid ammonium acetate extraction is used. The impact of cattle excrement on P solubility should be carefully investigated before recommending grazing on buffer zones.

References

- Hartikainen, H. & Yli-Halla, M., 1996. Solubility of soil phosphorus as influenced by urea. *Zeitschrift für Pflanzenernährung und Bodenkunde* 159, 327-332.
- Jansson, H. & Tuhkanen, H.-R., 2003. Reducing environmental loading from dairy farming. In: Uusi-Kämpä, J., Yli-Halla, M. & Grek, K. (ed.). *Reducing nutrient load from dairy farm pastures*. Maa- ja elintarviketalous 25. Agrifood Research Finland p. 40-47.
- Shand, C.A., Williams, B.L., Smith, S. & Young, M.E., 2000. Temporal changes in C, P and N concentrations in soil solution following application of synthetic sheep urine to a soil under grass. *Plant and Soil* 222, 1-13.
- Sharpley, A.N. & Withers, P.J.A., 1994. The environmentally-sound management of agricultural phosphorus. *Fertilizer Research* 39, 133-146.
- Turtola, E. & Yli-Halla, M., 1999. Fate of phosphorus applied in slurry and mineral fertilizer: accumulation in soil and release into surface runoff water. *Nutrient cycling in agroecosystems* 55, 165-174.

Elevated phosphorus inputs to Loch Leven during storm events – implications for load estimation and catchment management

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Introduction

Loch Leven is a eutrophic lowland lake in Scotland with an area of 13 square km and average depth of 3.9 m. After much remediation work during the last several decades, diffuse sources now dominate estimated nutrient loads. Phosphorus loads from the catchment of Loch Leven have, historically, been calculated on the basis of water samples collected from inflow streams on an 8-day sampling interval (Bailey Watts and Kirika 1987), but it was not known whether this was sufficient to provide accurate load estimates. It has been suggested that up to 80% of the total pollutant input to surface waters can occur during high-flow events, which occur over just 3% of the time (Smith et al. 1991; Littlewood 1993), and these might not be well represented in a traditional sampling regime. During this study, an autosampler/ autoanalyser (AutoLab, made by Envirotech) instrument was used to sample and measure soluble reactive phosphorus (SRP) in one stream, the Green's Burn (56°11' N, 3°13' W), every two hours. SRP, in comparison with total phosphorus, has been seen to be more relevant to eutrophication issues, as this fraction of the total phosphorus pool is more available to aquatic primary producers (Jarvie 2006). Streamflow was also measured every fifteen minutes by a Scottish Environment Protection Agency gauging station.

Relationships between soluble reactive phosphorus concentration and streamflow

Several high-flow events and periods of low flow were observed in 2006 and three representative periods of 12 days each are illustrated in Figure 1. It was observed that, while SRP concentrations were quite often elevated at the same time as streamflow, this was not always the case. Three different scenarios, relating to streamflow and SRP concentrations, have been generalised; (i) after periods of low flow, an increase in streamflow was associated with an increase in SRP concentration (Figure 1a, 1c), (ii) during periods of sustained low flow, there were occasional dramatic increases in SRP concentration (from 100 to 400 $\mu\text{gP.L}^{-1}$ during one event in July), which then decreased back to baseline level within 12 hours, observed twice in 2006 (in July and October, Figure 1b), and (iii) during periods of sustained elevated streamflow, or elevated streamflow preceded by only a short interval of low flow, SRP concentrations did not appear to be related to flow (not illustrated).

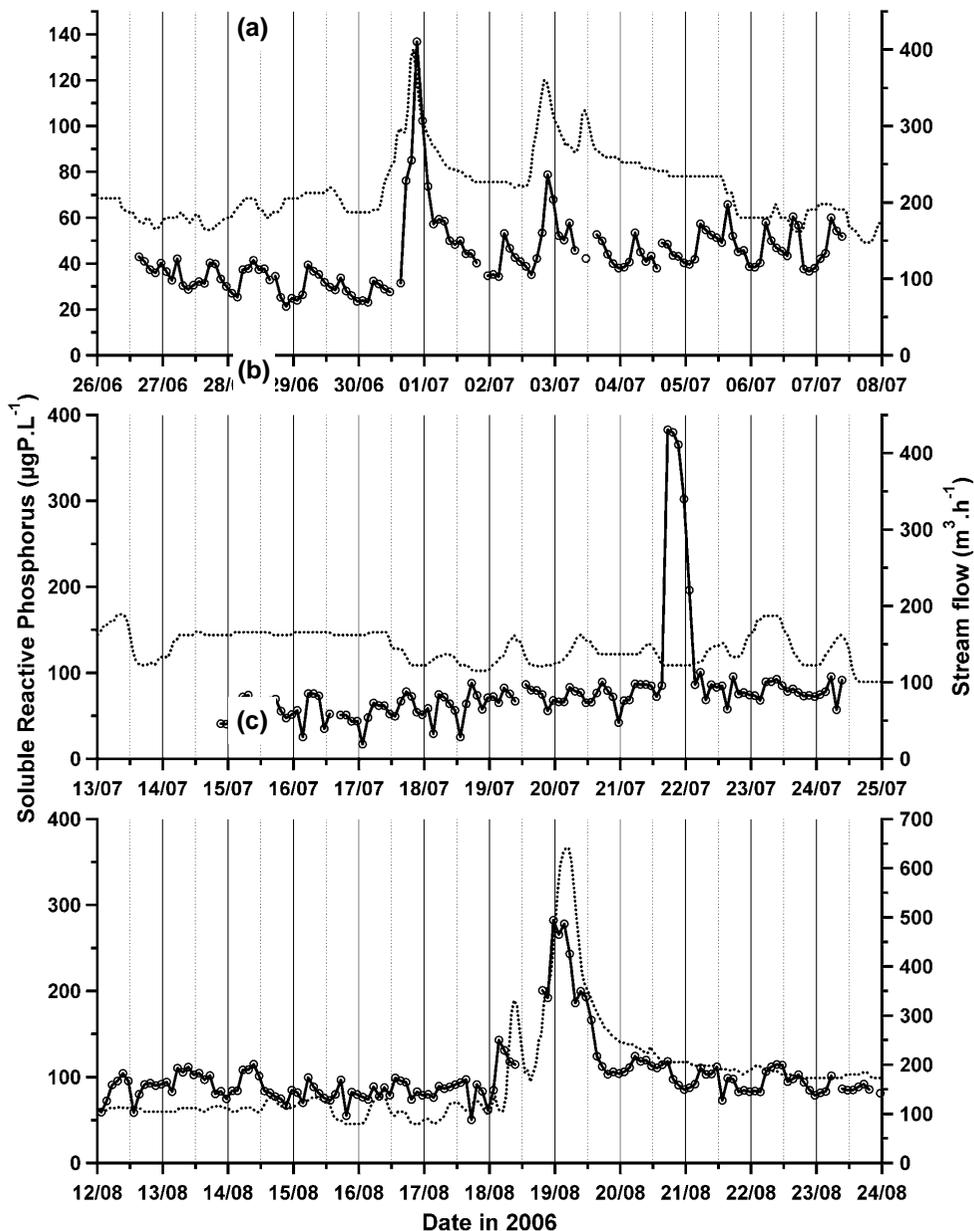


Figure 1. Streamflow (dotted line), measured every 15 minutes, and concentration of soluble reactive phosphorus (circles and solid line), measured every two hours, in Green's Burn, an inflow stream to Loch Leven, Scotland, during three different 12-day periods in 2006. Solid gridlines occur at midnight, dotted gridlines at midday. Note changes in scale between graphs in both y-axes.

The first of these scenarios could be explained by accumulation of SRP in soil and subsequent flushing during high-flow events, or by release of SRP from sediment associated with mobilisation of the sediment. The second scenario appears to be evidence of an intermittent point source of phosphorus. On one of these occasions, in November 2006, the stream water was observed to be distinctly green and had the odour of manure, suggesting a ruminant source. Both of these events occurred during the evening. It should be noted that, although total phosphorus was not measured in this study, a later similar study (unpublished) that did measure total phosphorus concentrations in parallel with SRP provided evidence of the same three scenarios presented here, and showed that, in this stream, the particulate fraction of phosphorus is generally larger than the soluble.

These results provide evidence that the traditional 8-day interval for sampling in this catchment will not be sufficient to estimate accurately loads of phosphorus to Loch Leven. In August, for example, the load of phosphorus as SRP in the stream between 18th and 20th August (2.8 kg) was nearly half of the total for the 12-day period (6 kg). The differences in total phosphorus loading later in the year were even more dramatic, with about 80% of total phosphorus in October delivered during short, elevated flow, periods, which would not have been sampled in the traditional regime. Estimates of the total phosphorus load for October 2006 were 47 kg when calculated on an 8-day sampling interval, and 249 kg when the two-hourly data was used.

Conclusions

The use of automated sampling and analysis equipment has allowed a more detailed examination of temporal patterns of phosphorus loading, which is not possible with standard, infrequent, sampling regimes. This has, in turn, provided evidence that previous studies have grossly under-estimated loads of both soluble and total phosphorus to Loch Leven, at least from this particular stream, but probably from others in the catchment as well. Relationships between flow and concentration data have provided evidence of soil and stream processes that need further study, if we are to effectively understand and manage nutrient loads to water bodies and fulfil our obligations under the EU Water Framework Directive.

References

- Bailey-Watts, A.E. & Kirika, A., 1987. A re-assessment of phosphorus inputs to Loch Leven (Kinross, Scotland): rationale and an overview of results on instantaneous loadings with special reference to runoff. *Transactions of the Royal Society of Edinburgh: Earth Sciences* 78, 351-367.
- Jarvie, H.P., Neal, C. & Withers, P.J.A., 2006. Sewage-effluent phosphorus: A greater risk to river eutrophication than agricultural phosphorus? *Science of The Total Environment* 360, 246-253.
- Littlewood, I.G., 1993. Estimating Contaminant Loads in Rivers: A Review. Institute of Hydrology, Report No. 117.

Smith, K.A., Jackson, D.R. & Withers, P.J.A., 1991. Nutrient losses by surface runoff following the application of organic manures to arable land. 2. Phosphorus. *Environmental Pollution* 12, 53-60.

Monitoring of nutrient export into the lake Vico, Central Italy

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Introduction

There is general agreement on considering agriculture the main nonpoint source of water pollution (Novotny and Chesters, 1989; Rekolainen *et al.*, 1999).

Nutrients are among the most common agricultural pollutants and nitrogen (N) and phosphorus (P) in particular. Nitrogen generally reaches groundwater dissolved in percolating water. Phosphorus, however, is mainly exported bound to eroded soil particles, thus representing a threat rather to surface waters.

Problems related to agricultural nonpoint pollution sources are receiving increasing attention as demonstrated by the introduction in Europe of the Water Framework Directive (Dir. 60/2000/EC).

The aim of this paper is to investigate the impact of P and N pollution due to agricultural land use in the lake Vico basin in Central Italy.

The study area and its environmental problems

The lake basin (Figure 1) is a perfect, natural laboratory to investigate agricultural diffuse pollution problems, because its water quality is almost exclusively threatened by these pollution sources. It also has a number of relevant, intrinsic characteristics including roads and forest paths that, when particularly intensive rainfall events occur, become themselves important water ways transporting nutrients into the lake. Further, the vulnerability of the lake is aggravated by its long hydraulic retention time (17 years).

In particular, within the basin waters, a considerable increase in P concentration has been observed during the last decades. This increase has been attributed to hazelnut tree cropping which between the late 1950s and early 1990s took the place of extensive cereal cropping.

In this paper the results of a monitoring campaign within the lake basin to investigate the contribution to nutrient export of the most common land use types are presented. In particular, runoff from a hazelnut tree field, from two forested sub-basins (one managed and the other one undisturbed), from a larger sub-basin with both

agricultural and forested land cover, representative of the whole lake basin, was monitored.

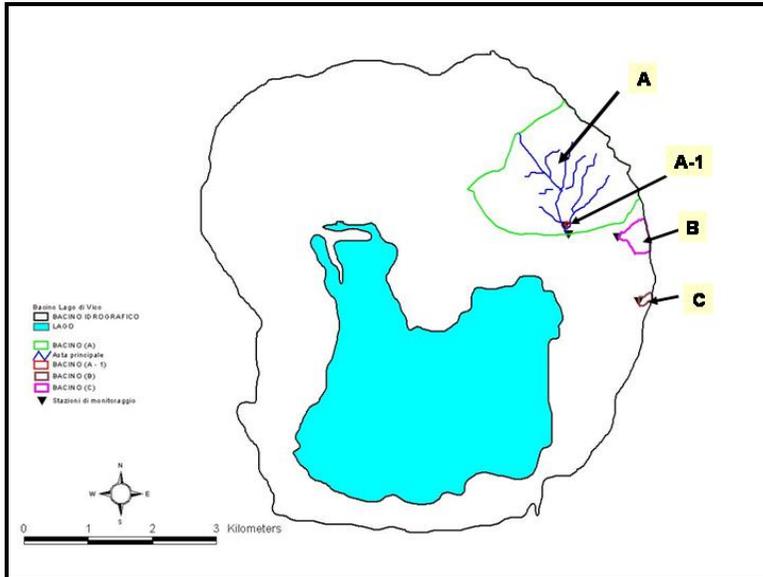


Figure 1. Lake Vico basin with the sub-basins and the sites considered for monitoring of surface runoff. (Sub-basin A, intensive hazelnut tree grove and managed forest. Sub-basin A-1, intensive hazelnut tree grove. Sub-basin B, unmanaged forest. Sub-basin C, managed forest).

Because the most up-to-date research in landuse planning to aid water pollution control recommends the integrated management of P and N (Heathwaite et al., 2000), and because the shift to intensive agriculture also creates the risk of increasing N pollution, some monitoring was also performed with respect to nitrate losses. To this end samples were taken following the hydrogeological pathways in an area of the northern part of the basin, where a number of sampling points were chosen, both from the deep aquifer and from the shallow aquifer close to springs.

Results and discussion

With reference to surface water monitoring, it has to be highlighted that the most interesting results derive from measurements taken during those few particularly intense rainfall events that were able to produce runoff.

Both the heavy drought of the last ten years and the soil characteristics of the area (coarse sandy soils) cause appreciable runoff to occur in particular hydrologic conditions only (at least 30 mm of total rain, high soil moisture from previous rain, at least one rainfall peak of 2mm/min). In particular, in the table below three of the most

recent runoff-generating rainfall events are considered. For each of them PO₄-P concentrations measured from each of the sub-basins shown in Figure 1 and representative of the most significant land uses of the area are shown.

Despite the scarcity of the data and the need for further monitoring, the data support the hypothesis that not only the hazelnut tree grove, but also the managed forest contribute significant loads of PO₄-P to surface water, while the unmanaged forest represents a low risk of P losses to surface waters.

Groundwater monitoring showed that the NO₃-N concentration from wells intercepting the deep aquifer is much lower than that from sampling points intercepting the shallow aquifer that, being much closer to the soil surface, is directly fed by leachate from agricultural soils.

From groundwater monitoring it was also possible to observe that as one moves from inland sites towards sites located closer to the lakeshore, NO₃-N quickly decreased from 60-80 mg/l to about 1mg/l. The evidence suggests that there is a clean deep aquifer eventually diluting high NO₃-N contents of waters reaching it from the surface.

Table 1. PO₄-P measured in surface water in correspondence of three rainfall events generating runoff.

Date	11/06/2005	11/29/2005	12/09/2005
Sub basin	PO ₄ -P mg/l		
A	1.12	-	-
A-1	53.90	-	0.24
B	0.73	0.27	0.41
C	-	42.10	46.9

Conclusions

Our data allows us to draw some interesting conclusions concerning the consequences of the most common land uses in terms of water pollution and on the particular hydrogeological characteristics of the basin. The main conclusion is that monitoring results, even though scarce and not from long time series, are extremely important for land use and water quality management because they allow specific environmental trends to be highlighted that simulation models alone would not have been able to point out.

References

Heathwaite, A.L., Sharpley, A.N. & Gburek, W., 2000. A conceptual approach for integrating phosphorus and nitrogen management at watershed scales. *J. Environ. Qual.* 29, 158-166.

- Novotny, V. & Chesters, G., 1989. Delivery sediments and pollutants from nonpoint sources: a water quality perspective. *J. Soil Water Conserv* 44, 568-576.
- Rekolainen, S., Grönroos, J., Bärlund, I., Nikander, A. & Laine, Y., 1999. Modelling the impacts of management practices on agricultural phosphorus losses to surface waters of Finland. *Water Sci. and Technol* 39, 265-272.

High resolution monitoring to characterise phosphorus transfers in complex catchments

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Introduction

At the catchment scale, diffuse phosphorus (P) losses from agricultural soils are combined with other catchment and in-stream transfers and transformations. Catchment scale monitoring is, however, likely to be the most cost-effective solution for assessing the efficacy of large scale policy driven measures designed to address nutrient loss from land to water. In the flashy catchments of the Irish border region, diffuse P transfers from soils during storm events are combined with those from rural point sources and include farmyard infrastructure and domestic septic systems.

Emerging technologies create opportunities to gather high-resolution data for an increasing number of environmental parameters. New data, therefore, provide new insights and pose new questions about certain environmental processes. Phosphorus (P) transfer rates in flowing waters is one emerging area where continuous monitoring has provided unsurpassed data coverage (Jordan et al. 2005; Jordan 2007) and avoids the need for interpolation and extrapolation using course statistical methods. These data offer new and unique insights into P dynamics at the catchment scale and provide a robust validation dataset for hydrochemical models.

Methods

In three small rural river catchments in the Irish border region (in Counties Tyrone, Armagh and Monaghan) with grassland agriculture and scattered dwelling houses, total phosphorus (TP) was monitored at a 10min resolution for up to 2 years using Dr Lange instrumentation and according to DIN EN 38405 D11. Concurrent and independent sensitive measurements of water level/discharge and other water quality parameters were also collected.

Results and conclusions

The extensive dataset (>90% coverage) showed that storms provided the largest TP transfer but that flow-concentration relationships were affected by 'flushing' and patterns of hysteresis. In addition, the dataset highlighted a persistent TP transfer during low flow periods that elevated baseflow concentration in two of the three

catchments during ecologically sensitive periods. Singular pollution episodes, unrelated to flow, were also monitored. This kind of monitoring enables assessment of mitigation measures to be properly targeted (soil/diffuse and point sources). Of real concern to catchment managers will be this ability to assess effects of P mitigation measures on multiple sources that may have impacts at different times, in different freshwater bodies (lakes and/or rivers) and at different scales. Analysis of an annual TP dataset, for example, indicates that a time-biased sampling strategy on a monthly or even weekly cycle (Fig. 1; similar to government agency standards) only manages to predominantly monitor the elevated P concentrations observed between storm events (and indicative of rural point source pollution and not storm driven diffuse pollution). In this type of scenario, mitigating the large soil P source may not register as a catchment scale P transfer change. Strategic use of high resolution nutrient monitoring may, therefore, provide a robust method to check multiple source (with multiple transfer mechanisms) mitigation measures at the catchment scale.

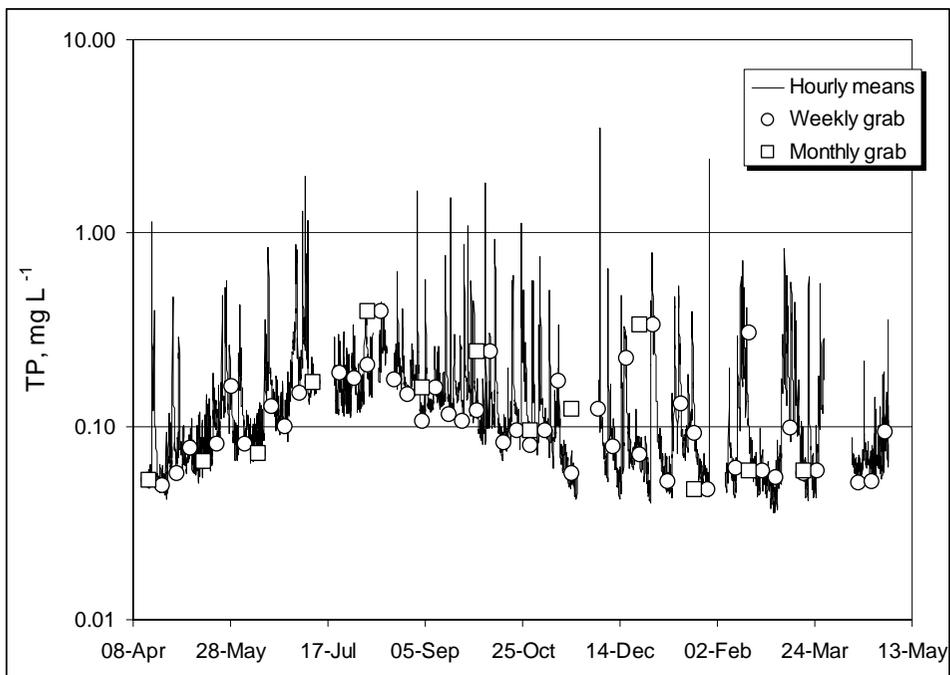


Figure 1. Comparison of hourly mean TP data (from measurements every 10mins) with monthly and weekly 'grab' samples. Very few storm events are captured with the coarse sampling regime and none of the peak concentrations ($1-2\text{mg L}^{-1}$).

References

Jordan, P., Arnscheidt, J., McGrogan, H. & McCormick, S., 2005. High resolution phosphorus transfers at the catchment scale; the hidden importance of non-storm transfers. *Hydrology and Earth Systems Science* 9, 685-691.

Jordan, P., Arnscheidt, J., McGrogan, H. & McCormick, S., 2007. Characterising phosphorus transfers in rural transfers using a continuous bank-side analyser. *Hydrology and Earth Systems Science* 11, 372-381.

Variation in phosphorus export resulted from urbanisation of former agricultural catchment (Southern River, Western Australia)

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Introduction

Greenfield urbanisation adversely affects the catchment water balance and often impacts on groundwater and stormwater quality (Choi, 2004). While most changes in the water balance are almost immediate, the water quality deterioration may occur over a considerable period of time. Effect of urbanisation on water quality depends on the degree of the pre-development water balance alteration and adopted water management practices. Prediction of the impact of urbanisation on the Southern River catchment under various water management scenarios is in a focus of current research within CSIRO National Flagship *Water for Healthy Country* Program “Swan Futures”.

Southern River catchment (190km²), previously cleared for agricultural use (predominately grazing), is currently being developed for new residential and light industrial purposes. The catchment is presently a major contributor of nutrients to the Swan-Canning estuary (Donohue *et al.*, 2001). Catchment urbanisation here is challenged by a high groundwater table, extensive seasonal waterlogging, and the existence of nutrient hotspots that result from intensive agricultural industries. Changes to the water balance arising from urbanisation have the potential to mobilise currently stored nutrient pools in addition to the new sources of nutrients introduced by new land use. The research objectives are to define the current water balance and P budget in the catchment, to develop predictive models allowing evaluating the impact of catchment water balance alteration on P loads and to introduce a monitoring network to investigate the impacts. Based on spatial analysis (ArcGIS), surface and groundwater monitoring data, surface and groundwater interaction modelling (MODHMS), an approach for evaluation of P export from the catchment has been developed to support the selection of water management options.

Effect of the catchment water regime on the P load in the Southern River

The surface water quality within the catchment is greatly influenced by the hydrological regime of the catchment. The latter is defined by catchment topography, geological setting, land use and climate. Three major hydrologic repose areas were identified: (a) hill sub-catchments, predominantly forested; (b) low-lying sub-catchments with predominantly clay-rich sandy soils (Guilford) (this area is partly urbanised and characterised by a fast response to the rainfall events due to an

extensive drainage network) and (c) low-lying sub-catchments underlined by predominately sand deposits (Bassendean) characterised by the high water storage in unsaturated zone and within multiple wetlands, and low overland runoff fluxes. The hydrological response in these areas varies throughout the year according to three major stages.

Stage 1 (Storage Recovery) is related to the period of the shallow groundwater recovery and wetland re-filling. At this stage the river recharge is limited stormwater generated during high intensity rainfall events within the hills and urbanised sub-catchments.

Stage 2 (Limited Storage) occurs following the recovery stage and usually marked by >400mm rainfall over the winter season. Surface inundation is most noticeable at this stage and catchment runoff significantly increases. During this stage the Bassendean sand sub-catchment significantly contributes to the Southern River discharge (50-80%).

Stage 3 (Storage Depletion) is predominantly the summer base flow attributed to the local groundwater discharge. Peak flow during this stage is related to the urban sub-catchments and to the hill sub-catchment

Phosphorus transfer from the catchment

Variation in P concentration, both total (TP) and soluble reactive (SRP), in surface water at the outflow of the catchment reflects the identified three stages in the catchment water regime (Table 1). P concentrations were the lowest and most variable during stage 1. The highest TP and SRP are related to the stage 2, when there is also the least variation between P concentrations in baseflow and peak flow.

Table 1. Phosphorus concentration (mg/L) at the outflow from the catchment.

		Base flow		Peak flow	
		Total P	SRP	Total P	SRP
Stage 1 <i>Storage Recovery</i>	average	0.131	0.069	0.159	0.073
	std dev	0.078	0.080	0.057	0.039
	range	0.023-0.5	0.013-0.48	0.083-0.34	0.007-0.21
	(count)	(31)	(31)	(43)	(42)
Stage 2 <i>Limited Storage</i>	average	0.211	0.128	0.237	0.131
	std dev	0.067	0.048	0.057	0.050
	range	0.052-0.39	0.022-0.25	0.15-0.37	0.06-0.26
	(count)	(31)	(31)	(25)	(25)
Stage 3 <i>Storage Depletion</i>	average	0.185	0.106	0.165	0.092
	std dev	0.060	0.037	0.046	0.037
	range	0.012-0.68	0.005-0.28	0.086-0.3	0.038-0.19
	(count)	(169)	(167)	(22)	(22)

The limited P retention capacity of the deep sandy soils and the disturbance of natural wetlands in the Bassendean sub-catchment manifests in high surface water TP and SRP. During stage 3 average TP and SRP concentrations are lower during peak flow events than concentrations observed during baseflow (Table 1). The dilution of P concentrations during the stage 3 peak flow events is likely to be associated with urban runoff. Whereas baseflow P concentrations are controlled by groundwater discharge. On average SRP contributes 50-55% to the TP, with the highest proportion of SRP during stage 2 (60-80%) in particular during the rising stage of the hydrograph event (up to 90%). Overall each stage of the catchment hydrological regime demonstrates a unique correlation between P load and river flow, which were used to reproduce the load in the river (Fig. 1). This data is used as a baseline for prediction of the urbanisation impact of P loads.

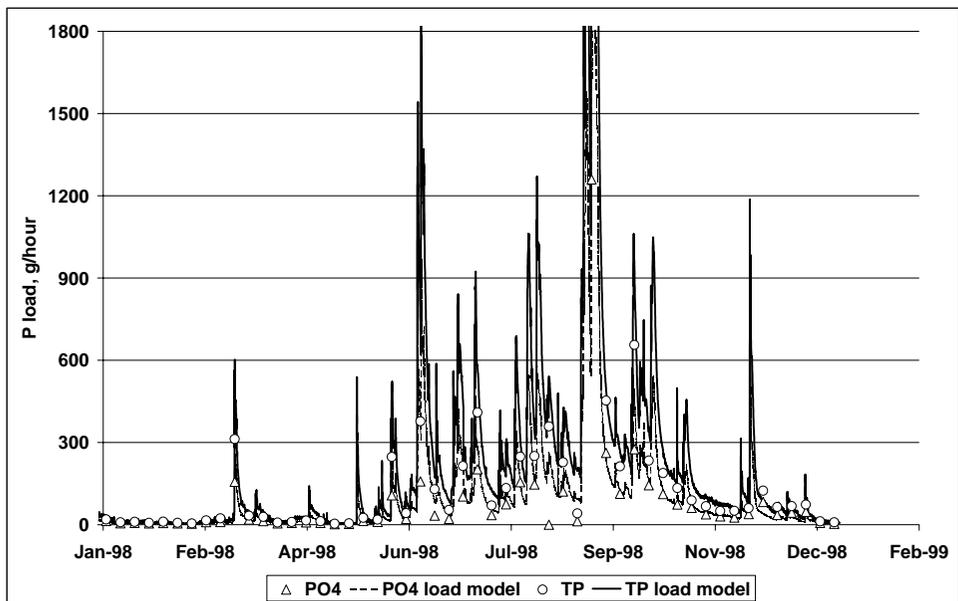


Figure 1. Observed and modelled P load for 1998 in the Southern River: (a) TP and (b) SRP load.

Conclusions

The hydrological characteristics of the catchment greatly influence P loads in the river and their seasonal variations. Correlation between the river flow and P load was defined for the various stages of the hydrological regime, and further applied to simulate the daily P loads in the river. The areas affected by water logging and wetlands provide the highest TP and particularly SRP contribution during the late winter period. Stormwater generated in urban areas on average provides a dilution effect for P concentration in the river. The analysis adopted in the project allows

evaluation of the urbanisation impact on the catchment water balance, which alongside with available data of P loads in the predominately urban environment allows for the prediction of overall P load during and under catchment urbanisation, which will be presented in the workshop paper.

References

- Donohue, R., Davidson, W.A., Peters, N.E., Nelson, S. & Jakowyna, B., 2001. Trends in total phosphorus and total nitrogen concentrations of tributaries to the Swan-Canning Estuary, 1987 to 1998. *Hydrol. Process.* 15, 2411-2434.
- Choi, W., 2004. Climate change, urbanisation and hydrological impacts. *Int. J. of Global Environ. Issues* 4, 267-286.

Phosphorus output from lowland agricultural watershed

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Introduction

One of the important topics of environmental protection in Poland is the problem of estimating nutrient exports from small, rural - usually not monitored - watersheds. Warsaw Agricultural University carries out investigations in the small lowland Zagożdżonka watershed in Central Poland. The results of changeability of dissolved reactive P (DRP) in the river investigated during 1992-1995 have been analyzed. The concentrations and loads from one measuring point (Czarna gauging station) were compared with those from a second investigation in 1999-2000. Both periods showed high average DRP concentrations of, respectively 0.44 mg PO₄ l⁻¹ and 0.41 mg PO₄ l⁻¹.

Watershed description and DRP analysis

The Zagożdżonka river catchment with the Czarna gauging station is located in central Poland (South-Mazovian Lowland) about 100 km south of Warsaw at the edge of Kozienski Landscape Park. The Zagożdżonka River is the left bank inflow of Vistula River (the inflow to Vistula is near Kozienski city). The Zagożdżonka is a typical lowland watershed. The absolute relief to Czarna gauging station is 26.5 m. The mean slopes of the main channels are in the range of 2.0-3.5 m per 1000 m. The exiting valleys are narrow and shallow. The dominant soils are loamy and about 9% are organic. Over 70% of the watershed is under cultivation with pasture dominating in stream valleys. The main crops are rye, wheat-rye and oats, but potato is also often grown. The forest cover is about 20% (Banasik 1983, 1994). The average discharge measured at the Czarna gauging station during first investigation period (hydrological years 1992-1995) was 0.055 m³ s⁻¹ and 0.091 m³ s⁻¹ in the second period (hydrological years 1999-2000). The hydrological year 1992 was the driest ever recorded and 1999 the second wettest ever recorded (Hejduk 2001). The average concentration of suspended sediment at the Czarna gauging station is 15 mg l⁻¹ and the maximum ever observed is 219 mg l⁻¹.

The dissolved reactive P was analyzed by reaction of stannous chloride and ammonium molybdate (Gajkowska- Stefańska *et.al.*,1990).

DRP concentrations and loads

The average consumption of phosphate fertilizers in terms of P was changeable during the investigation periods. Based on the Polish statistical yearbook the amount

of phosphates used in the province where the watershed is located has risen significantly during years 1992-1995 (from 6.6 kg ha⁻¹ of pure P in 1992 to 16.1 kg ha⁻¹ of pure P in 1995). It is difficult to find this phenomenon during the second period of investigation mainly due to non-homogeneity of data (after 1997 the statistical data are easy available only from much bigger areas and average phosphate consumption was about 18 kg ha⁻¹ in terms of P).

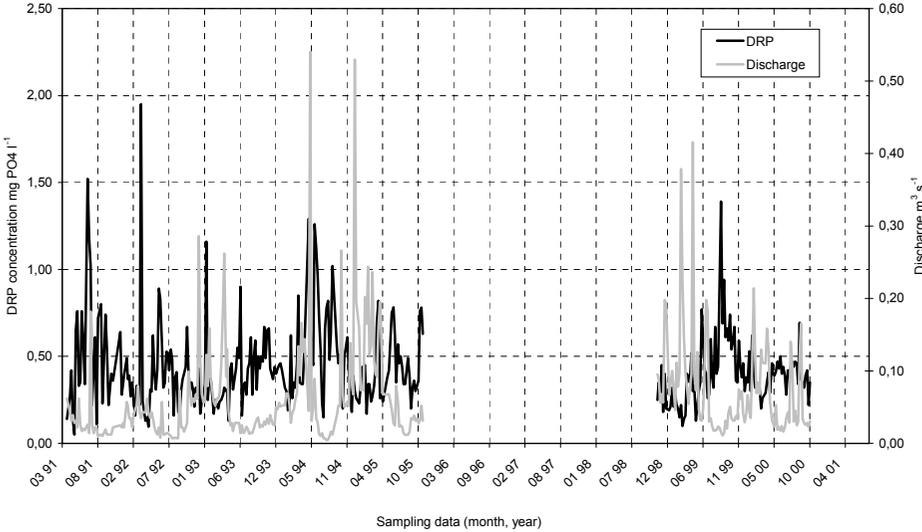


Figure 1. Changeability of PO₄ and discharge during investigation periods.

Table 1. The comparison of PO₄ concentration, discharge and loads during investigation periods.

Year	NS	Average mg PO ₄ l ⁻¹	MAX mg PO ₄ l ⁻¹	MIN mg PO ₄ l ⁻¹	Q m ³ s ⁻¹	VOL kg	LOAD kg ha ⁻¹ and year
1991	31	0.50	1.52	0.05	0.034	536	0.23
1992	52	0.39	1.95	0.13	0.033	406	0.17
1993	52	0.41	1.16	0.13	0.060	776	0.33
1994	49	0.50	1.29	0.15	0.060	946	0.40
1995	41	0.43	0.82	0.17	0.090	1220	0.52
Average		0.45			0.055	777	0.33
1999	52	0.43	1.39	0.1	0.107	1451	0.62
2000	43	0.39	0.69	0.2	0.074	910	0.39
Average		0.41			0.091	1181	0.50

NS number of samples; MAX maximum concentration; MIN minimum concentration; Q yearly average discharge; VOL volume of PO₄ from particular year.

In the Zagożdżonka watershed agriculture has the biggest impact on surface water quality probably dominated by non-point source pollution from fields. There is no significant trend of PO₄ concentration during the years. However, the average concentrations remain quite high (Table 1). The highest concentration was measured

during low flow (Figure 1) (usually 10 times lower than average flow) and only once during winter time. According to Polish limits of PO_4 concentrations for surface water the average PO_4 concentration is in the third class, which means that the quality (PO_4 only) is acceptable (UE A2 class).

References

- Banasik, K., 1983. Watershed land management as a factor for sediment transport reduction in river bed (in polish). Phd thesis Warsaw. Poland.
- Banasik, K., 1994. Sedimentgraph model of rainfall event in a small agricultural watershed. (in polish). Warsaw Agricultural University. Warsaw.
- Gajkowska-Stefańska, L., Guberski, S., Gutkowski, W., Mamak, Z. & Szperliński, Z., 1990. Laboratoryjne badania wody Ścieków i osadów Ściekowych (in polish) Warsaw Technical University.
- Hejduk, L., 2001. Suspended sediment and phosphorus transport during floods in Zagożdżonka catchment (in polish). Phd thesis. Warsaw Agricultural University. Warsaw.
- Statistical yearbooks of Poland (1985-2005). Central Statistical Office.

Relationships between available soil P and runoff P in the Sydney Drinking Water Catchment

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Introduction

Relationships between land use, available soil phosphorus concentration and phosphorus (P) in surface runoff are complex and understood only in general terms. A wide range of responses in the quantitative and qualitative mobilisation and transport of P by rainfall-runoff have been reported (Hart et al., 2004). Thus, few generalisations can be made with any confidence regarding this issue, which makes it difficult for decision-makers to target management interventions that reduce non-point source nutrient pollution. There is clearly a need for further research into the mechanisms of P export from land to water. This paper reports work in the Sydney drinking water catchment using a rainfall-simulator to gain a better understanding of P losses in surface runoff in relation to land use, soil type and concentrations of available soil P.

Methodology

An oscillating boom rainfall simulator was used in this study, consisting of three Veejet 80100 flat fan nozzles attached to a hollow boom, mounted on an aluminium A-frame, about 2.4m above the ground. The simulator allows 'rainfall' to be applied to two paired plots, 2.0 x 0.75 m in size. In this study, a rainfall intensity of about 60 mm hr⁻¹ was applied to the plots for a period of 15 minutes from the commencement of runoff. Sub-aliquots of the collected runoff were filtered in the field (0.45 µm) for dissolved reactive P (DRP) analysis; 500ml sub-samples of the runoff collected were put on ice and analysed for total P and total suspended solids on return to the laboratory. A total of 15 sites were included in this study, all under pastoral use varying from very low to very high intensiveness. Available (Colwell) soil P was measured using cores taken from 0-2cm and 0-10cm depth, within 24 hours of runoff generation.

Results

Initially, there appeared to be little relationship between soil P and DRP concentrations in runoff. However a relationship was observed when the data from sites on basalt-derived soils were separated out from that of non-basalt soils (Figure 1). The relationship between soil P and DRP in runoff from the non-basalt soils was reasonably strong ($R^2 = 0.86$) across a wide range of soil types and soil P concentrations, but the most striking feature was that DRP did not rise above a low

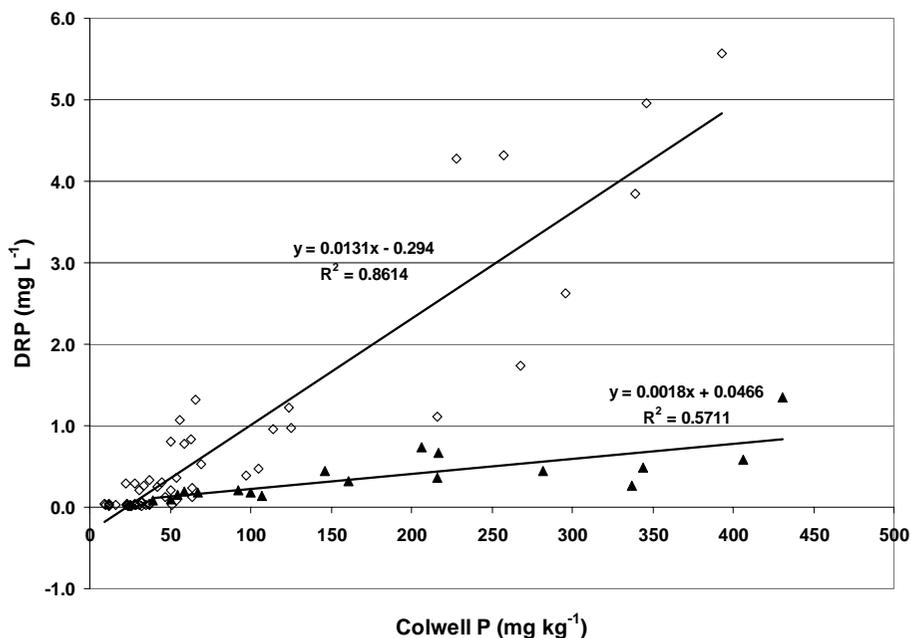


Figure 1. Runoff dissolved reactive P concentrations in relation to soil Colwell P (0-2cm depth) for non-basalt (\diamond) and basalt-derived (\blacktriangle) soils.

Table 1. Available soil P (0-2cm depth), total P and total suspended solids runoff concentrations, and approximate pasture cover at the study sites.

Site land use	Mean Colwell P (mg kg^{-1})	Mean runoff TP (mg L^{-1})	Mean TSS (mg L^{-1})	Approximate cover (%)
<i>Non-basalt soils</i>				
Rural residential	21.3	0.116	229.5	100
Beef cattle	24.7	0.165	262.3	100
Beef cattle	54.8	0.239	308.5	90
Beef cattle	28.7	0.264	105.3	90
Rural residential	48.7	0.285	129.4	100
Beef cattle	33.0	0.293	309.9	90
Beef cattle	63.7	1.307	647.5	50
Dairying	242.0	1.825	179.5	95
Rural residential	16.7	2.862	2713.3	50
Dairying	320.0	2.869	243.5	95
Sheep/beef	96.0	2.976	992.3	80
<i>Basalt soils</i>				
Beef cattle	62.5	0.155	214.0	95
Beef cattle	379.5	0.602	355.3	100
Beef cattle	230.3	0.721	306.5	80
Beef cattle	101.7	0.844	439.5	85

base level of about 0.1-0.2 mg L⁻¹ until soil P was greater than about 50 mg kg⁻¹ (0-2 cm depth), revealing a 'threshold' or 'break-point' response that is sometimes observed in such studies (e.g. McDowell and Sharpley, 2001; McDowell et al., 2003).

With the basalt soils, no break-point was observed. Concentrations of DRP in runoff remained relatively low, despite very high soil Colwell P concentrations, due to the high P sorption capacity of these soils. Correlations between soil and total P concentrations in runoff were weak, due to soil erosion from some plots which had low plant cover leading to large losses of total P, even from soils with very low Colwell P (Table 1). There was poor correlation between runoff total P and total suspended solids concentrations ($r^2 = 0.38$). However, if the two dairying sites data are excluded, the relationship improved significantly ($r^2 = 0.70$). This may be related to proportionally higher soil organic P contents at these sites, compared to the other soils, due to high manure/effluent inputs over time.

Conclusions

In most soil types investigated within the Sydney drinking water catchment, a reasonably good relationship between soil Colwell P and DRP in surface runoff was found, with relatively little DRP generated below about 50 mg kg⁻¹ Colwell P, a figure above the agronomic optimum for most crops. Basalt-derived soils, which constitute a significant class of agricultural soils in the catchment, exported much less DRP relative to soil P levels, and should be considered separately from other soils. P in agronomic (0-10 cm) and shallow (0-2 cm) depth soil cores was closely related ($R^2 = 0.94$, data not shown), suggesting that data from routine soil testing already undertaken by farmers could be used as a predictor of runoff DRP. The importance of plant cover on the mobilisation of total P via soil erosion was illustrated. Mitigation techniques for reducing P export from land to water in the catchment should therefore focus on controlling soil erosion before addressing dissolved P. Where low cover is related to low soil fertility, there may be a case for actually increasing plant-available P to promote plant growth, in order to decrease the risk of loss of P in surface runoff due to erosion.

References

- Hart, M.R., Quin, B.F. & Nguyen, M.L., 2004. Phosphorus runoff from agricultural land and direct fertilizer effects: A review. *J. Environ. Qual.* 33, 1954-1972.
- McDowell, R.W., Monaghan, R.M. & Morton, J., 2003. Soil phosphorus concentrations to minimise potential P loss to surface waters in Southland. *N.Z.J. Agric. Res.* 46, 239-253.
- McDowell, R.W. & Sharpley, A.N., 2001. Approximating phosphorus release from soils to surface runoff and subsurface drainage. *J. Environ. Qual.* 30, 508-520.

Influence of hydrodynamically rough grassed waterways on the runoff load with dissolved reactive phosphorus

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Introduction

A modified type of grassed waterway (GWW) with large hydrodynamic roughness has proven its ability to reduce sediment load and runoff under conditions where best management practices on the delivering fields prevent harmful sediment inputs, which would otherwise damage the grass cover (Fiener and Auerswald, 2003a; 2003b). It is unknown, however, how such a GWW affects the dissolved reactive phosphorus (DRP) load in runoff. The large amount of living and dead biomass needed to maintain the hydrodynamic roughness may be an additional source for DRP and thus partly counteract beneficial effects on streams with respect to nutrient input and eutrophication.

Material and methods

The effect on DRP was tested in a landscape-scale study, where DRP in runoff of two paired watersheds with and without GWW was continuously measured over five years (Fig. 1). The paired watersheds were largely identical in land use, soil, topography and predicted DRP concentration in runoff, when using either CREAMS (Knisel, 1980) or AGNPS (Young *et al.*, 1987) (Table 1). Details of pairing the watersheds and of runoff measurement are given by Fiener and Auerswald (2003b).

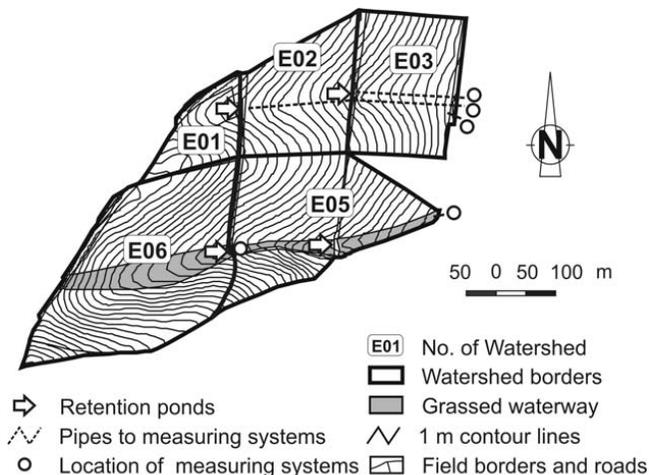


Figure 1. Topography of the sub-watersheds with and without grassed waterway; location of measuring system (flow direction from West to East).

In addition to the watershed monitoring, DRP was measured over one year in precipitation and in the throughfall under growing grass and crops within the watersheds (Weissroth 2000). The influence of straw cover on the fields between the crop growth periods was examined in laboratory rainfall simulations with and without straw cover (0.48 m² plots, rainfall intensity 26 mm h⁻¹, nine simulations during ~½-year, total rainfall 156 mm).

Table 1. Land use, soil, topography, predicted dissolved reactive phosphorus DRP concentration, mean measured DRP concentration and mean annual particulate phosphorus loss for paired watersheds with and without grassed waterway (Fig. 1).

		Upper sub-watersheds		Lower sub-watersheds	
		E01/02	E06	E02/03	E05
		without GWW	with GWW	without GWW	with GWW
Arable land	[%]	75	79	94	85
Set-aside areas	[%]	23	21	4	13
field borders structures	[%]	8	3	4	3
at the divide	[%]	14	4	0	0
along the thalwegs (grassed waterway)	[%]	0	13	0	10
Field roads	[%]	2.0	0.7	1.3	2.1
No. of fields		2	2	2	3
Crop rotation ¹⁾		ww-m-ww-p	ww-m-ww-p	ww-m-ww-p	ww-m-ww-p
Soil texture		silty loam	silty loam	silty loam	silty loam
Mean slope	[%]	7.1	9.3	7.3	9.0
DRP-CREAMS ²⁾	[mg L ⁻¹]	0.31	0.31	0.30	0.31
DRP-AGNPS	[mg L ⁻¹]	0.22	0.17	0.21	0.16
DRP ± SD ³⁾	[mg L ⁻¹]	0.61 ± 0.21	0.48 ± 0.32	0.35 ± 0.22	0.41 ± 0.24
Particulate P loss ± SD ⁴⁾	[mg m ⁻²]	13.0 ± 24.9	0.5 ± 0.8	17.1 ± 30.8	2.0 ± 2.1

1) ww = winter wheat, m = maize, p = potatoes;

2) Mean DRP predicted with CREAMS (Knisel, 1980) and AGNPS (Young *et al.*, 1987) with a spatial resolution of 12.5 x 12.5 m;

3) Mean measured DRP and standard deviation (total n = 674);

4) Mean annual particulate phosphorus loss from 1993 to 1998 with standard deviation between years.

Results

Measured mean DPR concentration in rain gauge precipitation was 0.06 mg L⁻¹ (SD 0.13). Mean DRP concentration of wet-only sampler precipitation was 0.02 mg L⁻¹ (SD 0.03) with summer values about twice as high as winter values.

The DRP in the throughfall for all tested grasses and crops was highly variable and highly enriched in P as compared to rain. High concentrations (up to 5 mg L⁻¹) occurred especially during the flowering of the respective crop and after frost events.

There was no significant difference, however, between the throughfall under the grass cover (mean 1.2, SD 1.3, n = 32) and under the crop cover (mean 0.8, SD 1.0, n = 35). DRP concentration in runoff from soil surfaces with and without straw cover was also in a similar range, mostly between 0.6 and 0.8 mg L⁻¹.

Considering the similar throughfall concentrations in GWW compared with crops and the lack of significant effects of straw cover, the GWWs should not alter the DRP concentration of the runoff. This was proved for the tested watersheds (Table 1), although runoff was much less in the GWW watersheds. Such GWWs will thus reduce the DRP load analogous to the reduction in total runoff.

Conclusions

The DRP in surface runoff of complex watersheds is composed of plant cover throughfall, runoff from bare surfaces and runoff from soil surfaces covered with plant residues. The DRP concentrations did not vary largely among these components. Hence, moderate differences in the contribution of the different components to total runoff have little impact on overall DRP concentration. Consequently, hydrodynamically rough GWWs, which provide a dense vegetation cover throughout the year but cover only a small area along the path of concentrated flow, are unlikely to alter the DRP concentration. This was confirmed in a long-term field-scale study with paired watersheds. Such GWWs will thus reduce the DRP load analogous to the reduction in total runoff. The extent of runoff reduction and its drivers have been elaborated already in many studies (e.g. Fiener and Auerswald 2003b). They also allow qualifying the effects of GWWs on DRP retention.

Furthermore, GWWs also lower the sediment routing and hence particulate losses were also lower in the GWW watersheds (Table 1). The effect of GWWs on sediment routing is larger than their effect on runoff reduction. In consequence, particulate losses declined more than DRP losses. GWWs are thus a highly effective measure to reduce dissolved and particulate phosphorous load downstream in addition to their numerous other beneficial effects (Fiener and Auerswald, 2003a).

References

- Fiener, P. & Auerswald, K., 2003a. Concept and effects of a multi-purpose grassed waterway. *Soil Use Manag.* 19, 65-72.
- Fiener, P. & Auerswald, K., 2003b. Effectiveness of grassed waterways in reducing runoff and sediment delivery from agricultural watersheds. *J. Environ. Qual.* 32, 927-936.
- Knisel, W.G., 1980. CREAMS: A field-scale model for Chemicals, Runoff, and Erosion from Agricultural Management Systems. USDA-ARS Cons. Res. Rep. 26.
- Weissroth, M., 2000. Ökosystemanalyse zur Befruchtung des Oberflächenabflusses mit gelöstem Phosphat aus landwirtschaftlich genutzten Flächen anhand langjähriger Freilandmessungen. Shaker, Aachen.

Young, R.A., Onstad, C.A., Bosch, D.D. & Anderson, W.P., 1987. AGNPS, Agricultural non-point-source pollution model - A watershed analysis tool. USDA-ARS Cons. Res. Rep. 35.

Effects of freezing and thawing on DRP losses from buffer zones

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Introduction

In Finland, agriculture is estimated to be the main cause of eutrophication and blue-green algal blooms in shallow fresh waters and coastal waters. Although only 7% of the total land area in Finland is in agricultural use, the contribution of agricultural P to waters is estimated to be 2,600 t yr⁻¹ representing 63% of the anthropogenic P loading (Finnish Environmental Institute, 2004). One way to mitigate P losses is to plant uncultivated buffer zones (BZ) between fields and surface-water bodies. The BZs have been found to fairly well retain eroded material, total P (TP), and particulate P (PP) from the surface runoff, while the retention of dissolved Molybdate-reactive P (DRP) with BZs has been under less consideration. This paper presents factors affecting DRP losses in surface runoff from a clay soil with BZs under spring cereals (1991–2001), grazing (2002–2005), and direct sowing (2006).

Experimental design

A six-plot experimental field was established on Typic Cryaquept/Vertic Cambisol soil at Jokioinen in SW Finland (60°48' N and 23°28' E) in 1989 (Uusi-Kämpä, 2005). Four 10-m-wide BZs (two grassed buffer zones = GBZ, and two vegetated buffer zones = VBZ) were planted below the 60-m-long cropland source area in 1991 while two 70-m-long field plots were cultivated without buffer zones (NBZ). The field area was fairly even, whereas the buffers were on a steep slope, varying between 12% and 18%. On the GBZ the buffer area was harvested annually, whereas on the VBZ the buffer area with scrub plants and herbs was not harvested. Results from the plots with the GBZ or VBZ were compared with the NBZ plots.

The effects of freezing and thawing on P losses from BZs were studied in two laboratory experiments. In the first study, a maximum potential for P release from frozen and thawed plant biomass was estimated from plant leachates. Plant samples were harvested from the GBZ, VBZ and NBZ in October 2003. Plant residues were leached with deionized water and after that freeze-thaw-cycles (-18°C; +4°C) were repeated four times. In the second study, simulated rain was applied to plant-soil samples (10 cm depth) both before and after freezing and thawing in autumn 2006. Rain simulations were conducted at room temperature (about +20°C) using deionized water (DRP < 0.02 mg L⁻¹, T = +22°C) and with a rainfall intensity of 20 mm h⁻¹. A modified Murphy and Riley (1962) method was used for TP and DRP (filter pore size 0.2 µm) determinations from surface runoff and plant leachates. The PP concentration was calculated as the difference between TP and DRP.

DRP in surface runoff

In the study with spring sowing and autumn ploughing, DRP surface runoff was highest from the VBZ plots in spring (Figure 1). On the VBZ buffer area, the concentration of Olsen-P was also high (60 mg L⁻¹) in the 2-cm-deep surface soil layer vs. 33 mg L⁻¹ on the GBZ (Uusi-Kämpmä, 2005). The high loss of DRP from VBZ was most likely due to P leaching from the soil surface and decaying grass residues on the VBZ in spring. Although the PP losses (<0.5 kg ha⁻¹) from the pasture were moderate, the DRP losses were larger than from the cereal field. In spring 2003, exceptionally high DRP concentrations (>1.0 mg L⁻¹) were measured in surface runoff water from the grassed field. One reason for the high DRP concentrations and losses may have been the sudden onset of winter with snowfall in autumn 2002, when the grass was still green, after a dry and warm summer. Some P was probably leached from plant tissues broken by frost. Also after destroying the grass in August 2005, the DRP load was high in the following spring. The mean DRP losses were 0.2–0.3 kg ha⁻¹ from the direct sown field between October 2006 and March 2007.

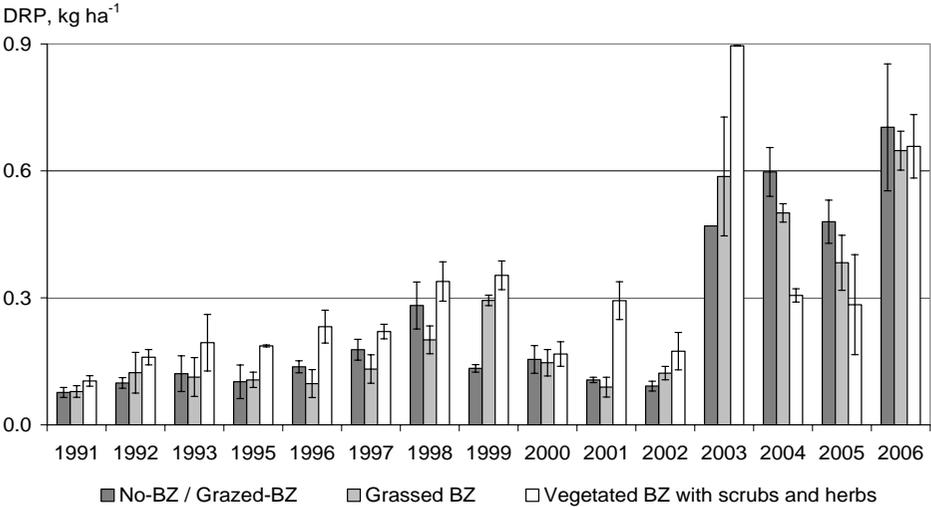


Figure 1. Losses of DRP in surface runoff from a spring cereal field (1991–2002), pasture (2003–2005), and spring barley and winter wheat (2006). Mean of two replicates with error bars showing maximum and minimum values.

According to the results of the plant leachates, potential P losses from the GBZ, VBZ and NBZ were also high (1.6, 3.1 and 1.7 kg ha⁻¹, respectively). Overall, between 60 and 80% of the biomass P was leached during freeze-thaw cycles and over 90% of the leached P was in the form of DRP. Most P was extracted after the first freeze-thaw cycle, with levels three times those before freezing. These findings explain the

high DRP losses from the grass field in spring. On the harvested GBZ, the grass was mostly green, whereas on the VBZ much of the aboveground biomass had matured. Sturite *et al.* (2007) suggested that most probably DRP is leached from the frost-injured plants that were green when the winter arrived.

Before freezing of the plant-soil-systems, the DRP concentrations in surface runoff were 0.65, 0.27 and 0.35 mg L⁻¹ from the NBZ, GBZ and VBZ, respectively (Table 1). After the freezing and thawing, there was a three-fold rise in DRP concentrations on the NBZ and VBZ, and up to seven-fold on the GBZ. Bechmann *et al.* (2005) reported an approximately 100-fold increase in runoff DRP concentrations from a frozen soil with catch-crops.

Table 1. Mean concentrations of DRP (mg L⁻¹) in surface runoff from the plant-soil-systems under the rainfall simulation before freezing (BF) and after freezing (AF).

	NBZ	GBZ	VBZ
BF, runoff (0–20 mm)	0.646	0.271	0.347
AF, runoff (0–10 mm)	1.743	1.990	1.076
AF, runoff (20–40 mm)	0.859	1.351	1.091
AF, runoff (40–60 mm)	0.684	0.951	0.884
AF, runoff (60–80 mm)	0.566	0.955	0.794
AF, runoff (80–100 mm)	0.485	0.744	0.714
AF, runoff (100–120 mm)	0.466	0.656	0.756

Conclusions

While buffer zones effectively retain TP and PP losses, the retention for DRP may be poor, and sometimes BZs may be sources of DRP. Freezing and thawing was found to increase the DRP losses. The high DRP concentrations and losses in spring runoff were most likely due to P leaching from the soil surface and frost-broken plant tissues. Therefore, the harvesting of buffer zones annually is recommended. More research is needed on how to increase DRP retention in the buffer zones.

References

- Bechmann, M.E., Kleinman, P.J.A., Sharpley, A.N. & Saporito, L.S., 2005. Freeze-thaw effects on phosphorus loss in runoff from manured and catch-cropped soils. *J. Environ. Qual.* 34, 2301–2309.
- Finnish Environment Institute, 2004. Vesistöjen ravinnekuormitus ja luonnon huuhtouma. <http://www.ymparisto.fi/print.asp?contentid=101882&clan=fi>
- Murphy, J. & Riley, J.P., 1962. A modified single solution method for the determination of phosphate in natural waters. *Anal. Chim. Acta* 27, 31–36.
- Sturite, I., Henriksen, T.M. & Breland, T.A., 2007. Winter losses of nitrogen and phosphorus from Italian ryegrass, meadow fescue and white clover in a northern temperate climate. *Agric. Ecosyst. Environ.* 120, 280–290.

Uusi-Kämpä, J., 2005. Phosphorus purification in buffer zones in cold climates. *Ecol. Eng.* 24, 491–502.

Colloid-facilitated phosphorus loss from diffuse agricultural sources via sub-surface pathways

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Introduction

Diffuse losses of phosphorus (P) from agricultural land may lead to eutrophication of surface waters and impact ecosystem health. Expanding our understanding of the mechanisms by which P is lost from agricultural soils is critical.

Colloid-facilitated transport is known to be important in P loss from agricultural soils (Heckrath, *et al.*, 1995; Stamm, *et al.*, 1998; Heathwaite *et al.*, 2005) but our understanding of this mechanism as a means of P loss through subsurface pathways is limited. The objectives of this study were to assess the contribution of colloids to Total P (TP) and Molybdate Reactive P (MRP) in leachate from an arable soil, to explore the possibility of using fluorescent polystyrene microspheres as a tracer for natural colloids, and to investigate the transport characteristics of different sizes of tracers through the soil.

Methodology

Aqueous suspensions of fluorescent microsphere tracers of four different sizes (0.2, 0.4, 0.8 and 1.2 μm) were applied to intact cores (30cm \varnothing x 60cm depth) of clay loam soil with a history of biosolids application, and to three undisturbed soil blocks, (each a 50cm cube) of a similar soil. A series of simulated rainfall events (15 min at 30 mm h^{-1}) was then applied to each. The cores allowed gross analysis of P fractions in the leachate, while the blocks were used to ascertain the spatial distribution, both of flow and tracer transport using a 10 x 10 flow cell array to split the flow of leachate.

Because P is thought to associate with different particle size fractions (Heathwaite *et al.*, 2005), TP, MRP and fluorescent microsphere in leachate from the cores was analysed for the following fractions: <0.2 μm , 0.2-0.45 μm , 0.45-1.0 μm and >1.0 μm . This allowed investigation into the role of colloid- and particle-facilitated transport in subsurface P loss together with the usefulness of fluorescent microspheres in tracing the movement of natural colloids and particles through soil.

Leachate from each flow cell in each block was analysed for the four sizes of microsphere. Comparison of the transport characteristics of each size of

microsphere, along with measurements of microsphere retention within the soil column provided further insight into the mechanism of colloidal transport.

Colloidal P in soil leachate

In this study, significant percentages of total TP and MRP in the leachate from the intact cores were found to be associated with both colloidal and particulate size fractions (Table 1).

Table 1. Average P size distributions in leachate from four intact soil columns.

	Core 1	Core 2	Core 3	Core 4	Mean
TP (mg P l ⁻¹ Leachate)	107.3	143.6	107.8	545.7	226.1
MRP (mg P l ⁻¹ Leachate)	46.8	34.2	29.1	130.9	62.4
% TP >1.0 µm	53.4	71.5	36.4	32.1	48.4
% MRP >1.0 µm	18.9	30.8	30.1	30.9	27.8
% TP >0.45<1.0 µm	3.4	4.8	6.6	25.3	10.0
% MRP >0.45<1.0 µm	7.8	11.0	3.7	34.7	14.9
% TP >0.2<0.45 µm	3.9	2.7	3.5	6.0	4.0
% MRP >0.2<0.45 µm	7.1	4.6	6.2	59.9	7.6
% TP <0.2 µm	39.3	20.9	53.5	36.5	37.6
% MRP <0.2 µm	66.2	53.6	11.8	22.5	49.7

Whether the MRP concentrations are a true measure of bio-available P in colloidal form or are overestimates due to the release of P from calcium phosphates by acidification during analysis (Hudson, *et al.*, 2000), these findings may have considerable implications for our reliance on MRP <0.45µm as a measure of bioavailable P in the aquatic environment.

Colloidal tracers and TP

Comparison of the breakthrough characteristics of fluorescent microspheres and TP showed some clear relationships. The 1.2µm, 0.8µm, 0.4µm and 0.2µm microspheres had significant linear correlations to the >1.0µm, 0.45-1.0µm, 0.2-0.45µm and <0.2µm TP fractions respectively, showing that the microspheres are a suitable proxy for tracing the movement of natural colloids through soil.

Colloids and preferential flow

In all three soil blocks, more than 60% of leachate was collected from less than 20% of the block, with more than 65% of each size of microsphere being detected in the leachate from those same flow cells. This confirms the importance of preferential flow in colloid transport. It was also found that the greatest concentrations of microsphere were detected in the flow cells where the highest peak flow rates occurred (Figure 1), rather than those showing the greatest flow volume. This suggests that flow rate may be the dominant factor in the mobilisation and transport of colloids and particles

within the soil column.

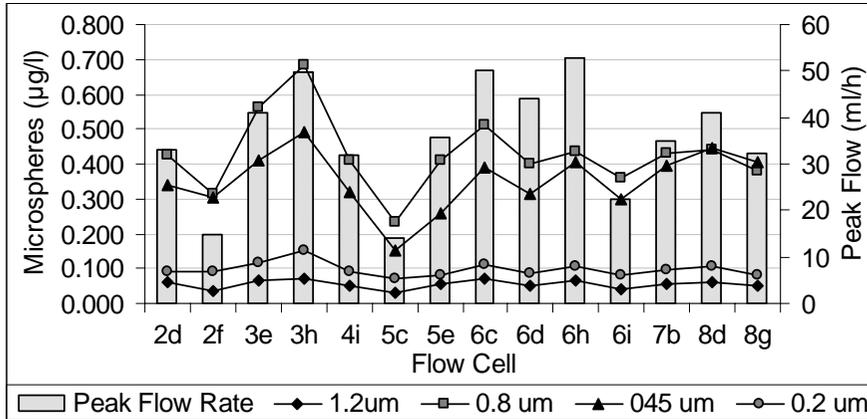


Figure 1. Microspheres and peak flow rate in active flow cells during a single simulated rainfall event on an intact soil block.

The relatively low concentrations of 1.2µm and 0.2µm microspheres in the leachate, combined with relatively high concentrations of 1.2µm microspheres in the top 5 cm of the soil block and lower, much more evenly spread concentrations of 0.2µm microspheres through out the soil profile, indicate that particle size is an important factor in subsurface transport.

Conclusions

Significant proportions of both TP and MRP were found to occur in association with colloids and particles in soil leachate, with more than 50% of MRP and 60% of TP on average being transported in association with material of radius greater than 0.2µm. Tracer investigations have indicated that this material may be easily transported through agricultural soils via preferential flow pathways, especially during periods of elevated flow rates, when transport occurs in greater concentrations.

References

- Heckrath, G., Brookes, P.C., Poulton, P.R. & Goulding, K.W.T., 1995. Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk experiment. *J. Env. Qual.* 24, 904-910.
- Hudson, J.J., Taylor, W.D. & Schindler, D.W., 2000. Phosphate concentrations in lakes. *Nature* 406, 54-56.
- Stamm, C., Fluhler, H., Gachter, R., Leuenberger, J. & Wunderli, H., 1998. Preferential transport of phosphorus in drained grassland soils. *J. Env. Qual.* 27, 515-522.
- Heathwaite, A.L., Haygarth, P.M., Matthews, R., Preedy, N. & Butler, P., 2005. Evaluating colloidal phosphorus delivery to surface waters from diffuse agricultural sources. *J. Env. Qual.* 34, 287-298.

Spatial distribution of P mobilisation in agriculture headwater catchments

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Introduction

The DESPRAL soil P dispersion test was used to estimate the SMP within 17 headwater (first-order) catchments across several geoclimatic regions in England and Wales. These samples were taken from several locations using a consistent sampling strategy, within each catchment and mapped digitally in order to gain a better understanding of P mobilisation in relation to other catchment features such as hillslope, soil surface wetness and manure applications. Relationships between base flow index, and soil clay content are discussed as means of understanding the role of simple variables in controlling P mobilisation. The DESPRAL index values are also discussed in relation to the catchment flow weighted mean concentration of total phosphorus at the catchment outlet, for which high resolution time series data are presented. These data provide insight into the processes by which P delivery is controlled, and may be used to refine and develop models currently used to predict phosphorus movement in the UK.

Methods

Catchments were selected on the basis that they were headwaters, and therefore we could minimise the impacts of in-stream processes on the estimates of load. Soil samples from across 11 catchments were sampled for Olsen P, DESPRAL index (filtered and unfiltered), water soluble phosphorus, suspended solids and total phosphorus. The despral samples also contained a subset of either 30 s or 280 s samples, as defined by the DESPRAL methodology (MAFF, 1982). Flow weighted mean concentrations (FWM) for the data rich catchments (i.e. “rich” time series) were estimated by linear interpolation of total phosphorus measurements. For the data poor catchments (<10 storm events), FWM estimations were made using the 2-strata approach (Kronvang, 1996). The resultant datasets of DESPRAL index values, and FWMs were then be used to obtain delivery coefficients for each of the catchments. Delivery coefficients are calculated as in Equation 1. Soil parameters were also extracted from UK national datasets e.g HOST (CEH, UK) for investigation of correlative data that could explain any potential variations in P mobility. The delivery of phosphorus is calculated as:

$$\text{Delivery} = \frac{\text{Annual Flow weighted Mean (mg l}^{-1}\text{)}}{\text{DESPRAL Index Value (mg l}^{-1}\text{)}}$$

Results

DESPRAL values were recorded for all catchments, and each value was associated with a spatially digitised so that the data could be associated with other datasets. The DESPRAL index values were highly variable and ranged from 0.10 mg/l to 2.63 mg/l across the 17 catchments for total phosphorus, and 0.03 mg/l to 0.23 mg/l for dissolved phosphorus. The values show good relationships with soil texture class (Figure 1b) and altitude. For example, at Colworth catchment, where a large number of field soil samples were collected (20) DESPRAL values increased towards to lower end of the catchment (Figure 1a).

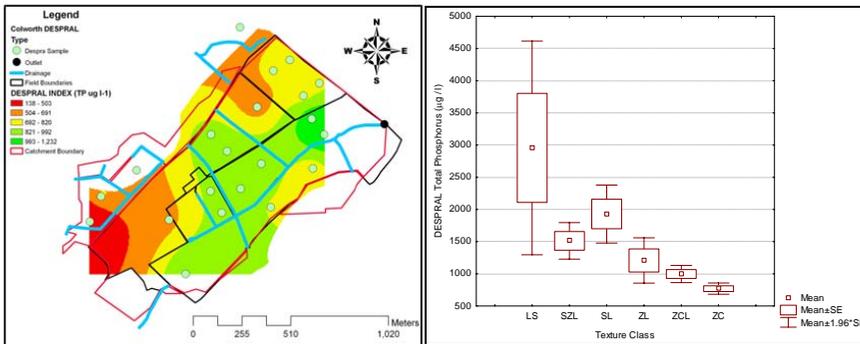


Figure 1a. Distribution of DESPRAL index values across the Colworth catchment and 1b.

Table 1. Data collected during the PEDAL project showing the flow weighted mean concentrations, the average DESPRAL values for each catchment and the calculated delivery coefficients.

Catchment Name	Average Flow weighted mean concentration (mg/l)	DESPRAL Mean TP (mg l ⁻¹)	Delivery Coefficient
Redesdale	0.25	0.18	1.39
Cools Cottage	0.10	0.44	0.23
Colworth	0.30	0.56	0.54
Cliftonthorpe	0.72	0.28	2.58
Childs Ercall	0.33	0.95	0.35
Waveney	0.66	0.73	0.90
Sydling St Nicholas	0.26	0.60	0.43
Weaver	4.10	0.39	10.41
Den Brook	0.64	0.94	0.68
Drewston	0.09	0.67	0.13

Conclusions

Flow weighted mean concentrations and DESPRAL index values were determined for a range of headwater catchments from different geoclimatic locale across England. These values were determined through both intensive and sparse data monitoring techniques. The delivery coefficients were highly variable. DESPRAL values correlated well with soil textures classes, but also with altitude and slope. This may mean that in areas where sediment is trapped, P mobilisation is enhanced, and may provide a way to map P mobility.

This work will form the basis of a toolkit and fuzzy modelling toolbox (e.g Scharer, 2006) which may be used to study other catchments and therefore plug the gaps in our existing knowledge of delivery, and will subsequently help to identify and constrain uncertainties in our knowledge of phosphorus transport. Prior to this study, very little data was available to describe the delivery of phosphorus to the catchment outlet. Now that the dataset has been collated, further analysis of the control parameters may be undertaken. This will yield better understanding of the controls that determine delivery in agricultural landscapes.

References

- Beven, K.J., Heathwaite, A.L., Haygarth, P., Walling, D.E., Brazier, R. & Withers, P., 2005. On the concept of delivery of sediment and nutrients to stream channels. *Hydrological Processes* 19, 551-556.
- Kronvang, B. & Bruhn, A.J., 1996. Choice of sampling Strategy and estimation method for calculating nitrogen and phosphorus transport in small lowland streams. *Hydrological Processes* 10, 1483-1501.
- MAFF, 1982. The determination of the dispersion ration is soil. In: *Techniques for Measuring Soil Physical Parameters*. Reference Book 441, Ministry of Agriculture, Fisheries and Food, Her Majesty's Stationary Office, London. pp. 100-101.
- Scharer, M., Page, T. & Beven, K., 2006. A fuzzy decision tree to predict phosphorus export at the catchment scale. *Journal of Hydrology* 331, 484– 494.

Phosphorus transport from row crop agriculture in the Midwestern U.S.: problems with scaling up from small plot to watersheds

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Introduction

The environmental basis for making nutrient management decisions is largely based on data from rainfall simulations performed at the plot scale (0.2 to 10 m²; DeLaune et al., 2004; Kleinman et al., 2004; Elliot et al., 2006). However, agricultural producers typically manage fields at the 1 to 100 ha scale, and eutrophication is the result of phosphorus (P) transport at the catchment scale (300 to 1,000,000 ha). Few studies have been able to study P transport at the plot, field and catchment scales. The objective of this research was to develop a better understanding of up-scaling of P transport from the plot to catchment.

Experimental design

Rainfall simulations occurred on three sizes of plots (2, 5 and 10 m²) that were nested within two fields (2.2 and 2.7 ha), which were in turn nested within a series of three catchments on the same drainage ditch, draining approximately 300, 2230, and 4300 ha. The north field and set of plots had been in a continuous no-tillage corn-soybean rotation for approximately 15 years. The south field and set of plots had also been in a continuous no-tillage corn-soybean rotation for the same period, but were converted to a rotational tillage (i.e. soil tilled before corn) in the year in which this project began (2004). Both fields were cropped to corn during this study period. Rainfall simulation protocol was 50 mm hr⁻¹ for 50 min, and 75 mm hr⁻¹ for 15 min. Runoff and nutrient data from the fields and catchments were collected daily during the growing season (April to November) and from natural storm events.

Results of scaling from plot to watershed scale

The runoff coefficients of the fields were lower than the plots (Table 1). Sharpley and Kleinman (2003), using the variable source area hypothesis of overland flow generation, observed a greater percentage of smaller plots contributing to runoff than larger plots. The same relationship is likely to be true for scaling from plot to field, as observed with the current data. Increasing scale from field to catchment resulted in increased runoff coefficients, since the catchments capture discharge from surface and subsurface processes in this tile-drained landscape. This observation was true for data collected for multiple years within this and adjacent catchments (Figure 1). Similar runoff coefficients have been observed in other watersheds ranging from 2 to 48,000 ha (Gentry et al., 2007).

Table 1. Runoff coefficient, soluble P (SP) and total phosphorus (TP) loads at the plot, field and catchment scales.

Plots	Runoff Coefficient Unitless	Seasonal P Loads [†]		Mean P Loads for Events Only [‡]		P Loads Normalized on Precipitation [¶]	
		SP	TP	SP	TP	SP	TP
		g/ha		g/ha/event		g/ha/mm precip	
N2	0.49			29.3	185	0.687	4.46
S2	0.31			1.04	147	0.021	3.36
N5	0.48			17.7	119	0.364	2.26
S5	0.40			2.05	85.9	0.046	2.01
N10	0.38			7.06	58.3	0.153	1.33
S10	0.39			1.06	231	0.023	5.54
Fields							
North	0.06	79.0	430	6.08	35.9	0.147	0.798
South	0.04	11.9	777	0.99	70.7	0.022	1.44
Catchments							
AME	0.21	40.5	180	1.11	4.79	0.079	0.352
ALG	0.24	235	1050	9.25	41.8	0.448	1.99
AXL	0.67	527	1420	13.1	46.5	1.00	2.71

[†]Seasonal loads calculated from daily samples collected April 1, 2004 to November 15, 2004.

[‡]Mean event loads calculated as the sum of loads occurring after runoff producing storms, divided by the number of storms.

[¶]Normalized P loads calculated by dividing the seasonal P load by the seasonal precipitation observed within the selected catchment.

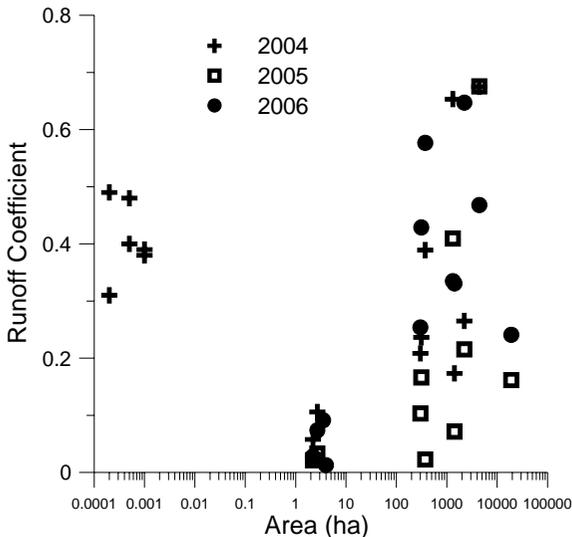


Figure 1. Runoff coefficients from plot, field and catchment scale data collected during the 2004, 2005 and 2006 monitoring years.

The mean event soluble phosphorus (SP) and total phosphorus (TP) loads were calculated for all runoff events during the study period (Table 1). Mean event SP loads were greater in the north field than the south field, which was reflected in the plot data. SP loads increased with increasing drainage area in the catchments. Plot data tended to overestimate TP loads from fields and catchments, likely due to greater erosion rates from the plots than fields. TP loads increased an order of magnitude between AME and ALG. Flow velocities at the AME site are nearly always slower than at ALG or AXL. Sediments from ALG and AXL have a greater particle size distribution and lower organic matter content than the AME site (Smith et al., 2005). Tile outlets with very high sediment and nutrient concentrations flow into this ditch between AME and ALG, contributing to greater TP loads. Soluble P and TP loads were normalized based on precipitation from rainfall simulators for plots or from all the natural storms for the field and catchment scales (Table 1). Normalized SP loads were greater from the plots in the north field than the south field, which is what was observed at the field scale. Normalized SP loads increased, from 0.08 to 1.0 g ha⁻¹ mm⁻¹ precipitation, with increasing catchment size for the ditch sites. This was likely due to inputs from tile outlets, and greater flow velocities for the ALG and AXL sites than the AME site.

Conclusions

Rainfall simulation experiments on small plots may produce similar magnitude of normalized P loads observed at the catchment scale, but this interpretation needs to be cautioned due to different processes involved in P transport at these scales. Rainfall simulations should only be used to study surface processes, whereas data from the catchment scale integrate both surface and subsurface transport processes. Modeling efforts are vital for identifying practices that will improve water quality, but these efforts must be ground-truthed so that the personnel involved with these efforts understand all the processes occurring in the catchment, and to better understand potential problems (i.e. problematic tile outlets) in the catchment.

References

- DeLaune, P.B., Moore, P.A., Jr., Carman, D.K., Sharpley, A.N., Haggard, B.E., & Daniel, T.C., 2004. Development of a phosphorus index for pastures fertilized with poultry litter – factors affecting phosphorus runoff. *J. Environ. Qual.* 33, 2183 – 2191.
- Elliot, H.A., Brandt, R.C., Kleinman, P.J.A., Sharpley, A.N., & Beegle, D.B., 2006. Estimating source coefficients for phosphorus site indices. *J. Environ. Qual.* 35, 2195 – 2201.
- Gentry, L.E., David, M.B., Royer, T.V., Mitchell, C.A., & Starks, K.M., 2007. Phosphorus transport pathways to streams in tile-drained agricultural watersheds. *J. Environ. Qual.* 36, 408-415.
- Kleinman, P.J.A., Sharpley, A.N., Veith, T.L., Maguire, R.O. & Vadas, P.A., 2004. Evaluation of phosphorus transport in surface runoff from packed soil boxes. *J. Environ. Qual.* 33, 1413 – 1423.

- Sharpley, A. & Kleinman, P., 2003. Effect of rainfall simulator and plot scale on overland flow and phosphorus transport. *J. Environ. Qual.* 32, 2172-2179.
- Smith, D., Haggard, B., Warnemuende, E. & Huang, C., 2005. Sediment phosphorus dynamics for three tile fed drainage ditches in Northeast Indiana. *Agric. Water Manage.* 71, 19-32.

Scale of measurement effects on phosphorus in runoff from cropland

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Introduction

Phosphorus (P) losses in runoff from cropland can contribute to water quality degradation in freshwater lakes and streams (Correll, 1998). Phosphorus-based nutrient management techniques have been widely adopted to minimize environmental damage from P in runoff and frequently include use of P loss risk assessment tools. Phosphorus loss risk assessment tools such as P indices are often developed largely from small plot-scale data showing the relationships between various site and management variables and runoff P losses. Little information is available on how small plot runoff composition compares with field or subwatershed scale measurements.

This study was conducted to compare runoff composition measurements at the subwatershed scale with those from natural runoff at the small plot scale, and to investigate the validity of using small plot data to develop field-scale P loss risk assessment tools.

Experimental procedures

Sediment (TS), soluble P (TDP), and total P (TP) in natural runoff from small plots (1 m²) located in two subwatersheds (7.2 and 12 ha) instrumented to measure and sample runoff events were compared with similar measurements from the subwatersheds over an 18-month period. Small plots were replicated four times in each subwatershed. The subwatersheds, cropped with either corn [*Zea mays* (L.)] or alfalfa [*Medicago sativa* (L.)], were located at the University of Wisconsin-Platteville Pioneer Farm (42°42' N, 90°22' W). Initial soil test P (Bray-1) values at the 0-15-cm depth were 37 and 19 mg kg⁻¹ in the corn subwatershed and small plots, respectively, and 114 and 136 mg kg⁻¹ in the alfalfa subwatershed and small plots, respectively.

Scale effects on runoff characteristics

Phosphorus and sediment concentrations and runoff volumes from the corn and alfalfa subwatersheds and small plots are summarized in Table 1. Runoff P concentrations were generally similar in small plots and subwatersheds, but varied by crop and season. For the 18-month period, there were 12 runoff events in both the corn subwatershed and small plots and 10 runoff events in both the alfalfa

subwatershed and small plots. To separate winter and summer runoff events, January through March was designated as winter and the rest of the year as summer. In both subwatersheds, most of the runoff volume occurred in the winter. The small plots had greater runoff volumes per unit area in all cases compared to the subwatersheds.

Table 1. Phosphorus and sediment concentrations in runoff and runoff volumes from corn and alfalfa subwatersheds and small plots from June 2004 to December 2005 at the Pioneer Farm, Platteville, Wisconsin, USA.

Field	Season	n ^{††}	Scale	TDP [‡]	TP [§]	TS [¶]	Runoff
				-----mg L ⁻¹ -----			L m ⁻²
Corn	Total	12	Subwatershed	2.45	4.19	2509	44.4
			Small plot [†]	1.83	2.96	1190	219
	Winter	4	Subwatershed	3.18	3.80	330	32.4
			Small plot	2.13	2.93	370	184
	Summer	8	Subwatershed	0.45	5.24	8425	11.9
			Small plot	0.26	3.14	5460	35.3
Alfalfa	Total	10	Subwatershed	1.24	1.53	130	64.7
			Small plot	1.78	2.39	220	181
	Winter	6	Subwatershed	1.23	1.54	126	57.5
			Small plot	1.75	2.33	180	163
	Summer	4	Subwatershed	1.26	1.52	169	7.20
			Small plot	2.10	2.99	550	17.6

[†]Small plot values are the average of four replications in each subwatershed

[‡]Total dissolved phosphorus

[§]Total phosphorus

[¶]Total solids (sediment)

^{††}Number of runoff events

Over the entire measurement period, the corn subwatershed had higher P and sediment concentrations compared to the alfalfa subwatershed. The corn subwatershed P concentrations were higher than in the small plots. This difference is consistent with differences between corn subwatershed and small plot soil test P values. The TDP concentrations were higher in the winter than the summer for both scale sizes. The fact that this seasonal change in TDP concentration was seen at both scales of measurement adds validity to use of small plot data for constructing P indices. Total P and total solids (sediment) concentrations were greater in corn than in alfalfa at both measurement scales.

Phosphorus runoff from small plots and subwatersheds were compared on an event by event basis using regression analysis. Results showed that runoff volumes at the

two scales were related ($R^2 = 0.73$) for both the corn and alfalfa subwatersheds. For individual runoff events, TDP concentrations at the two scales of measurement were strongly correlated in the corn subwatershed ($R^2=0.90$), but not in the alfalfa subwatershed ($R^2=0.10$). Statistical analysis (using repeated measures) of mean TDP values for the entire measurement period showed no significant differences between small plot and subwatershed scales of measurement.

Conclusions

Runoff P concentrations were similar in small plots and subwatersheds, but varied by crop and season. The seasonal and crop effects on runoff P and sediment concentrations were reflected at both scales of measurement. The agreement of small plot and subwatershed runoff dissolved P and sediment P concentrations supports use of small plot data in constructing P loss risk assessment tools.

References

Correll, D.L., 1998. The role of phosphorus in the eutrophication of receiving waters: A review. *Journal of Environmental Quality* 27, 261-266.

Quantifying agricultural phosphorus transfers at hillslope to catchment scales

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Introduction

The pathways of phosphorus (P) transfer in agricultural catchments are relatively well understood, but quantification of the transfer of P from agricultural land is necessary to improve the development and performance of models used for prediction and management of water quality, and to allow assessment of the role of agricultural phosphorus in diffuse pollution and stream biogeochemistry (Heathwaite *et al.*, 2005). Traditionally, studies have focussed on the hillslope plot and catchment as observation scales. Limited data and process understanding therefore exist for intermediate scales, and as a result, we currently have a poor understanding of how and why data and processes change with spatial scale (Brazier *et al.*, 2005). The objectives of this study were: (i) to undertake a programme of field monitoring to provide data to quantify event-based P transfers at multiple spatial scales within a small agricultural catchment, and (ii) to compare the P transfer characteristics of different scales of observation to determine differences and linkages between scales.

Field monitoring design

A number of spatial scales (circa 0.002 – 30.6 ha) were monitored over two hydrological years in a small mixed agricultural headwater catchment with silty clay loam soils, at ADAS Rosemaund, Herefordshire, UK. The observed scales included nested hillslope patches of 37, 74 and 111 m hillslope lengths, a 1.9 ha hillslope, two field areas of 2.5 and 3.7 ha, and a 30.6 ha catchment outlet. Continuous stage and turbidity monitoring of runoff through a hillslope surface flume, three field drains and the stream catchment outlet were integrated with event-based monitoring of runoff at the hillslope patch scales, and event-based sampling for total P (TP) and total dissolved P ($TP_{<0.45\mu\text{m}}$).

Results

Transfers of P within the catchment were monitored at multiple spatial scales in six events over the 2004-2005 and 2005-2006 hydrological years. Differences in TP and $TP_{<0.45\mu\text{m}}$ behaviour were observed both between scales and between events, and the relative importance of each of the observed pathways in catchment P transfer

also varied between events. In two of the monitored events, no surface runoff was observed within the catchment.

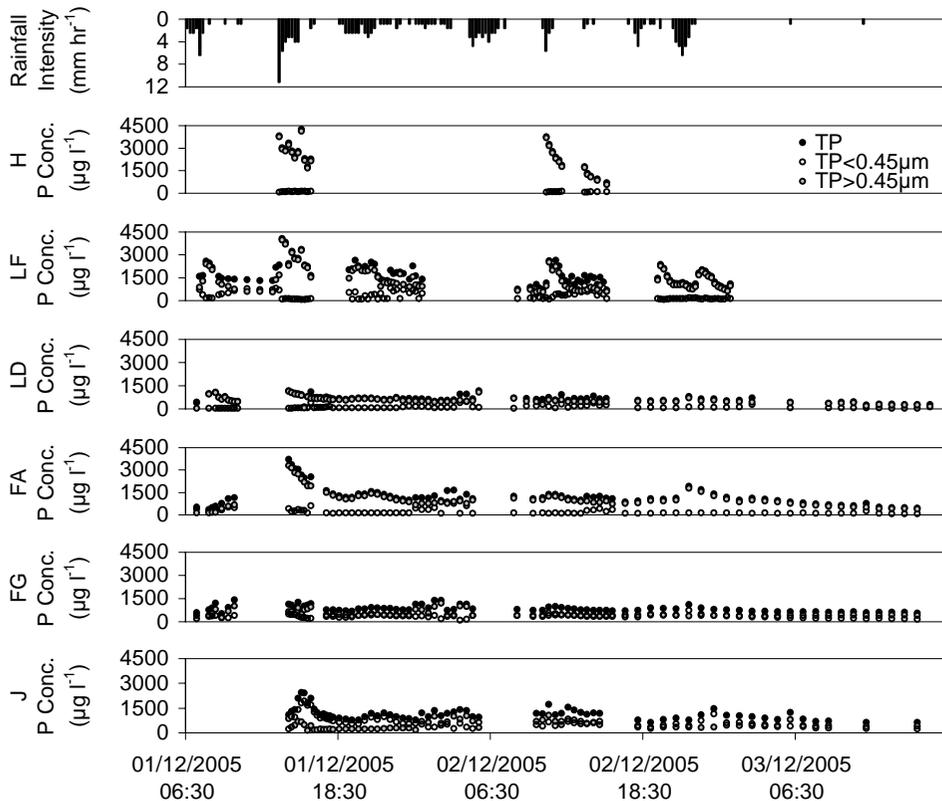


Figure 1. P transfer data for different spatial scales within the catchment, 1st-4th December 2005. H = hillslope patches, LF = hillslope surface runoff (1.9 ha), LD = hillslope drainflow (1.9 ha), FA = arable field drainflow (2.5 ha), FG = grass field drainflow (3.7 ha), J = catchment outlet streamflow (30.6 ha).

Figure 1 shows rainfall and P data for a storm event monitored on 1st-4th December 2005. Data for the hillslope patch scales (H) were found to be strongly correlated to the data for the hillslope scale (LF), but the linkages are less clear between the other scales. Analysis of event summary data shows that although surface runoff had the highest recorded peak TP concentrations, lower runoff volumes for this pathway meant that peak TP fluxes were slightly lower in surface runoff than in drainflow (LD) from the hillslope (Table 1). Surface runoff contributed lower TP loads to streamflow at the catchment outlet (J) than drainflow, and had lower TP yields than all the observed P transfer pathways. The highest TP loads were from the arable field drain (FA), and TP yields were also highest in drainflow from this scale. The TP_{<0.45µm} fraction was much more important in drainflow from grassland than drainflow from

arable land, with the highest peak $TP_{<0.45\mu m}$ concentrations and $TP_{<0.45\mu m}$ loads measured in the grass field drain (FG), although the arable field drain still had higher $TP_{<0.45\mu m}$ yields.

Table 1. Table of P characteristics for an event in the catchment on 1st-4th December 2005, for hillslope to catchment scales of observation.

Scale	TP				$TP_{<0.45\mu m}$			
	Peak TP ($mg\ l^{-1}$)	Peak Flux ($mg\ s^{-1}$)	Load (kg)	Yield ($kg\ ha^{-1}$)	Peak $TP_{<0.45\mu m}$ ($mg\ l^{-1}$)	Peak Flux ($mg\ s^{-1}$)	Load (kg)	Yield ($kg\ ha^{-1}$)
LF	4.1	4.1	0.12	0.06	1.0	0.4	0.01	0.01
LD	1.2	4.2	0.55	0.29	0.5	1.1	0.10	0.05
FA	3.7	21.4	2.23	0.89	0.8	2.3	0.51	0.20
FG	1.4	12.7	1.12	0.30	1.2	4.4	0.62	0.17
J	2.4	115.6	13.0	0.43	1.1	42.8	5.63	0.18

No significant differences were found in point data or event P transfer characteristics for different hillslope patch scales, suggesting that hillslope P transfer in this catchment is not always scale dependent. However, significant differences were found in TP characteristics for sub-catchment and catchment scales. Where significant differences were observed, some characteristics between scales were found to be strongly correlated, suggesting that transfer of event data between scales may be possible for this type of small agricultural catchment.

Conclusions

Phosphorus transfers were monitored during six events at multiple spatial scales within a small agricultural catchment. The resulting data have allowed the pathways of P transfer to be quantified, and have indicated the variability in P transfer characteristics between events and between scales. In the largest event monitored, P transfer was dominated by drainflow, with surface runoff being a relatively unimportant transfer pathway. Significant differences were observed in event data for hillslope to catchment scales, but strong relationships suggest that linkages exist between scales.

References

- Brazier, R.E., Heathwaite, A.L. & Liu, S., 2005. Scaling issues relating to phosphorus transfer from land to water in agricultural catchments. *Journal of Hydrology* 304, 330-342.
- Heathwaite, A.L., Dils, R.M., Liu, S., Carvalho, L., Brazier, R.E., Pope, L., Hughes, M., Phillips, G. & May, L., 2005. A tiered risk-based approach for predicting diffuse and point source phosphorus losses in agricultural areas. *Science of the Total Environment* 344, 225-239.

Grasslands, sediment, colloids and phosphorus: an interdisciplinary team approach with the 'GrasP' project

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Introduction

This paper describes an overview of the '**Grassland sediment and colloid Phosphorus (P) (GrasP)**' project that is using an inter-disciplinary collaborative team approach, involving soil scientists, geomorphologists, analytical chemists, hydrologists and mathematical modellers. The team is working at the plot, field and catchment scales in South-west England to help the UK government (Defra) better understand how intensive grassland management influences sediment and colloidal P losses to surface waters. Despite the topical nature of phosphorus, sediment and colloid transfers, surprisingly there are some critical deficiencies in our existing knowledge on aspects of their fate and transport, particularly in relation to intensive grassland systems, exemplified by the dairy pastures common in North-west Europe (Haygarth et al., 1998). There are four innovative research areas that the GrasP project is addressing: 1) the inadequacy of current soil erosion inventories for intensively managed grasslands, 2) the limitations of current operational definitions of soluble and particulate fractions and the importance of colloidal P loss from intensively managed grasslands, 3) the role that organic matter – particularly agricultural amendments - plays in sediment and colloidal P loss, and 4) the need for improved integration between field observations and modelling to better understand the uncertainties and complexity of processes controlling sediment and colloidal P loss at the plot, field and catchment scales.

Inadequate soil erosion inventories for intensively managed grasslands

There has been little research conducted on erosion from intensively managed temperate pastures. This is surprising considering: 1) the high return frequencies of potentially erosive maritime rainfall on steep sloping grasslands, 2) the high nutrient turnover in intensive grassland systems, and 3) the wide extent of intensive grassland systems across the world. The potential for high sediment and colloid

yields from intensive grasslands has been overlooked because grasslands appear to have high rates of vegetation cover that we associate with low erosion rates. It has been demonstrated by Quinton (2001) that particle selectivity can account for higher than expected phosphorus loss from an arable site, we suggest that this is also true for intensive grassland systems. Grassland erosion is more likely to be caused by sheet and interrill erosion than distinct rilling or gullyng and has hence attracted little attention from farmers and researchers. Poached and overgrazed areas, which are sources of colloidal P and sediment during winter months, quickly recover during spring and summer months. Current analytical often omit the detection of fine colloidal material and the organic fraction of sediments, which we hypothesise are important components of eroded material from intensive grasslands (Bilotta et al., 2007). In addition, inputs of excreta and recycled animal manures in combination with the high organic matter content of grassland soils are overlooked as potential sediment sources in intensive grasslands.

Limitations of current operational definitions of soluble and particulate fractions and importance of colloids

Sediment is defined as material that is transported from land to surface and ground waters and can include clay (<2 µm), silt (2-60 µm), sand (60 µm – 2 mm), pebbles (2-60 mm), cobbles (60-256 mm) and even larger material. Whereas, colloidal material is defined as particles in the size range 1 nm – 1 µm. The operational definition of 'dissolved' and 'particulate' phases is based on separation using a 0.45 µm threshold. We believe that current field and analytical techniques used to assess sediment and colloid budgets result in errors because of confusion over terminology, inadequate sampling, use of inappropriate membranes (Haygarth and Sharpley, 2000) and inaccurate means of separating and quantifying particles (Gimbert et al., 2005). The widespread use of the 0.45 µm membrane to separate 'sediment and particles' from 'solute and dissolved' fails to determine the important range of material between 0.1 and 1 µm. We believe this range is an important component of the material lost from intensive grasslands that is not characterised by conventional techniques and plays a key role in contaminant mobilisation from intensive grasslands. Sediment fractions and colloidal material comprise a continuous range of particle sizes and analytical techniques use operationally defined thresholds that need to be considered in the interpretation of their results. In this study we will present results from flow field-flow fractionation that has enabled improved characterisation of colloidal material.

Role of organic matter in sediment and colloidal losses

Intensive grassland soils contain high levels of organic matter (Bellamy et al., 2005), due in part to deposits of excreta and recycled animal manure. The fate of these sources of organic material in intensive grassland systems is poorly understood. We believe that these sources contribute to the overall colloidal and sediment budget of

intensive grassland systems. It is therefore, important that we improve our understanding of the sources of these materials, the relative importance of soil processes and agricultural land management activities and the timing and response of different transport pathways (surface and subsurface). One way of increasing our understanding of the relative contributions of deposited organic material, decaying organic matter and soil is through the use of tracers (Granger et al., 2007). In this project we have used a suite of novel tracing techniques that will improve our knowledge of the importance of agricultural amendments – specifically the role of slurry within grassland systems. These techniques include: natural abundance of (^{13}C) carbon tracer which allows carbon and associated phosphorus to be followed as well as natural fluorescence of protein-like substances characteristic of animal wastes and manures, and fluorescent-labelled particles matched to slurry characteristics by size and density.

Improved integration between field observations and modelling

Computer based mathematical models add value to the scientific process when they are combined with field based observations, enabling the evaluation of our system understanding. Improved dialogue between field based scientists and modellers is required to translate the uncertainties in the observations we make and that exist in our perceptual models and mathematical model inputs and parameter sets (Krueger et al., 2007). The temporal and spatial variability of potentially dominant processes in intensive grasslands requires careful and time consuming observations. There is a need to balance the processes included in models with the need to account for process variability and model uncertainty to enable development of parsimonious model structures based on the data available. In this project we have learned from the observations that we have made in the field, resulting in strong interactions between model development and data collection. This integration will improve our perceptual understanding and ability to predict colloidal P and sediment transfers from intensive grasslands.

References

- Bellamy, P.H., Loveland, P.J., Bradley, R.I., Lark, R.M. & Kirk, G.J.D., 2005. Carbon losses from all soils across England and Wales 1978-2003. *Nature* 437, 245-248.
- Bilotta, G.S., Brazier, R.E. & Haygarth, P.M., 2007. Processes affecting transfer of sediment and colloids, with associated phosphorus, from intensively farmed grasslands: erosion. *Hydrological Processes* 21, 135-139.
- Gimbert, L.J., Haygarth, P.M., Beckett, R. & Worsfold, P.J., 2005. Comparison of centrifugation and filtration techniques for the size fractionation of colloidal material in soil suspensions using sedimentation field-flow fractionation. *Environmental Science & Technology* 39, 1731-1735.
- Granger, S.J., Bol, R., Butler, P., Haygarth, P.M., Naden, P., Old, G., Owens, P.N. & Smith, B.P.G., 2007. Processes affecting transfer of sediment and colloids, with associated

- phosphorus, from intensively farmed grasslands: tracing sediment and organic matter. *Hydrological Processes* 21, 417-422.
- Haygarth, P.M., Chapman, P.J., Jarvis, S.C. & Smith, R.V., 1998. Phosphorus budgets for two contrasting grassland farming systems in the UK. *Soil Use and Management* 14, 160-167.
- Haygarth, P.M. & Sharpley, A.N., 2000. Terminology for phosphorus transfer. *Journal of Environmental Quality* 29, 10-15.
- Krueger, T., Freer, J., Quinton, J.N. & Macleod, C.J.A., 2007. Processes affecting transfer of sediment and colloids, with associated phosphorus, from intensively farmed grasslands: a critical note on modelling of phosphorus transfers. *Hydrological Processes* 21, 557-562.
- Quinton, J.N., 2001. Phosphorus detachment: processes and interactions. In "Proceedings of the International Phosphorus Transfer Workshop, 28th August to 1st September 2001, Plymouth, Devon, England" (P.M. Haygarth, L.M. Condron, P.J. Butler and J.S. Chisholm, eds.). Institute of Grassland and Environmental Research, Okehampton and Plymouth.

Phosphorus and sediment export from drained and undrained intensively managed grasslands

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Introduction

Grasslands cover a large proportion of the temperate land-mass including significant areas of Europe, North America, New Zealand and Australia. In the UK, for example, grasslands occupy approximately 65% of the agricultural area (Peeters, 2004). Much of this grassland is intensively managed and has been 'improved' in some way. Improvement of grassland can involve a whole range of modifications, from regular additions of fertiliser and weed-control treatments, to tillage and re-seeding, to installing subsurface drainage. These improvements are designed to enhance plant growth and herbage yield and increase the productivity of livestock and dairy farming. In temperate climates such as the UK, a major factor limiting the growth and utilisation of grass in grassland systems is excess water due to poor drainage (Armstrong and Garwood, 1991). Waterlogging of soils can lead to the development of anoxic conditions (Dils and Heathwaite, 1999), can decrease the rate of nutrient mineralization (Scholefield *et al.*, 1993), and can slow the warming of the soil during the spring (Farr and Henderson, 1986). This in turn can retard the growth of plant roots and inhibit their function, reducing the overall plant growth and herbage yield (Farr and Henderson, 1986). Furthermore, wet and saturated soils are more susceptible to damage (such as compaction, pugging and poaching) by grazing animals which can also reduce pasture herbage yield (Bilotta *et al.*, 2007). Consequently, subsurface drainage has been widely used as a strategy to alleviate these problems (Robinson and Armstrong, 1987). During the last decade, however, there has been increasing concern over the environmental impacts of subsurface drainage. One particular area of concern which remains a controversial and poorly understood issue relates to the influence that subsurface drainage has on the transport of contaminants, such as sediment and phosphorus (P), from intensively managed grasslands to surface waters. The objectives of this study were to determine the effects of subsurface drainage on grassland hydrology and the export of sediment and phosphorus to adjacent water courses.

Experimental design

The work presented here is based on results from a field experiment designed to investigate the influence of subsurface drainage on sediment and P export from intensively managed grasslands. The experimental site, run as part of the Rowden Experimental Research Platform at the Institute of Grassland and Environmental Research, in Devon (UK), comprises four one-hectare paired lysimeter plots (two drained and two undrained). The lysimeter plots were monitored for overland flow, subsurface throughflow and/or subsurface drainflow, as well as sediment flux and total P flux, over the 2005-2006 hydrological season. Plots were equipped to monitor surface flow and drained flow in isolation, so that the effect of land drainage, common to intensively grazed land on heavy soils, could be assessed not only in terms of the threat posed to water quality from individual hydrological pathways, but also in terms of the overall effect of subsurface drainage (i.e. drained versus undrained land) on the sediment and P export budget.

Results

Data from individual storm events (e.g. see table 1) suggest that subsurface drainage causes a reduction, by as much as 50%, in the total quantity of suspended sediment and TP transferred from the 1 ha grassland plots.

Table 1. An example of a budget for hydrological, sediment and phosphorus fluxes from drained and undrained 1 ha grasslands during an individual rainfall event on 07 March 2006. Minimum and maximum values are calculated using discharge uncertainty intervals developed by Krueger *et al.* (2007). Estimated values are calculated using discharge from a classical non-linear stage-discharge relationship.

07/03/06 Storm Event Budget	Undrained Plot (13)			Drained Plot (12)		
Rainfall Depth (mm)	11.43			11.43		
Total Event Discharge (000 L)	Min	Estimated	Max	Min	Estimated	Max
	90	103	120	42	52	60
% of Total Event discharge via the Interflow Pathway	100	100	100	51	47	49
% of Total Event Discharge via the Drainflow Pathway	0	0	0	49	53	51
Total Suspended Solids Exported (kg)	5.5	6.2	7.2	2.6	3.0	3.6
Total Phosphorus Exported (g)	17	19	22	8	9	11
% of Total Suspended Solids Transferred via the Interflow Pathway	100	100	100	74	71	73
% of Total Phosphorus Transferred via the Interflow Pathway	100	100	100	71	68	70

Conclusions

These results challenge existing environmental concepts which often suggest that subsurface drainage can act as a preferential pathway which enhances sediment and P transport from land to surface waters (e.g. Chapman *et al.*, 2001; Dils and Heathwaite, 1999). These findings will have important implications for those involved in land-use and mitigation of effects on water-quality.

References

- Armstrong, A.C. & Garwood, E.A., 1991. Hydrologic consequences of artificial drainage of grassland. *Hydrological Processes* 5, 157-174.
- Bilotta, G.S., Brazier, R.E. & Haygarth, P.M., 2007. The impacts of grazing animals on the quality of soils, vegetation and surface waters in intensively managed grasslands. *Advances In Agronomy* 94.
- Chapman, A.S., Foster, I.D.L., Lees, J.A., Hodgkinson, R.A. & Jackson, R.H., 2001. Particulate phosphorus transport by subsurface drainage from agricultural land in the UK. Environmental significance at the catchment and national scale. *The Science Of The Total Environment* 266, 95-102.
- Dils, R.M. & Heathwaite, A.L., 1999. The controversial role of tile drainage in phosphorus export from agricultural land. *Water Science and Technology* 39(12), 55-61.
- Farr, E. & Henderson, W.C., 1986. *Land Drainage*. Longman Handbooks in Agriculture. Longman, Harlow, 251 pp.
- Peeters, A., 2004. Wild and sown grasses. Profiles of a temperate species selection; Ecology, biodiversity and use. Blackwell, London.
- Robinson, M. & Armstrong, A.C., 1987. The extent of agricultural field drainage in England and Wales 1971-1980. *Trans. Insti. Bri. Geogr.* 13, 19-28.
- Scholefield, D. et al., 1993. Nitrate leaching from grazed grassland lysimeters - effects of fertilizer input, field drainage, age of sward and patterns of weather. *Journal of Soil Science* 44, 601-613.

Understanding the pathways and dynamics of agricultural diffuse pollution from intensively farmed grassland: the application of natural and artificial tracing techniques

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Introduction

The impacts of diffuse agricultural pollution on river systems arising from the application of slurry and inorganic fertilisers are widely acknowledged. Particulate, colloidal and dissolved phases of slurry may be important sources of organic material and vectors of phosphorus. However, the mobilisation, transport and dynamics of each of these phases have not been quantified. Knowledge of these processes is essential to assess the effectiveness of catchment sensitive farming procedures for slurry application and to support their future refinement (Water Code, 1998). The objectives of this study are to: apply tracers to elucidate the transport of particulate, dissolved and colloidal slurry phases through experimental grassland plots and to consider their significance for phosphorus transport.

Experimental design

Slurry (grass- and maize-derived with artificial fluorescent tracer) was applied, at a rate of 21 m³ ha⁻¹, to four of the 1-ha Rowden experimental platforms (Devon, SW England) in April 2006. Surface and subsurface drainage waters from both drained and undrained grassland plots were sampled prior to slurry application and during subsequent rainfall events. Three novel applications of tracing techniques were used to explore the movement of the various phases of slurry: 1) natural fluorescence (dissolved phase), 2) fluorescent-labelled particles (matched to slurry particulates), and 3) carbon isotopes (dissolved and particulate). The carbon isotopes thus provide an independent assessment of both the natural and artificial fluorescence methods.

Natural fluorescence may be used to detect the presence of animal waste in water (Baker, 2002). Technological advances enable fluorescence to be analysed rapidly on small water samples with extreme sensitivity. During analysis fluorescent organic

molecules are excited with UV light (wavelengths 200-400nm). Fluorescent centres in optical space are produced by the molecules emitting energy as they return to their ground state. Distinct fluorophores are produced by aromatic proteins (tryptophan and tyrosine) and high molecular weight organic molecules (humic-like and fulvic-like substances). The relative intensities of these fluorophores will reflect the source of organic matter whereas their absolute intensities will reflect their concentrations (Baker, 2002). Emission-excitation matrices of farm wastes have strong tryptophan-like and tyrosine-like fluorophores indicating the usefulness of the technique for detecting farm waste in water (Baker, 2002). Fluorescence intensity is also influenced by the absorbance of the water samples produced by dissolved organic matter. Absorbance was measured and used to correct the fluorescence data.

Artificial fluorescent particles were used to trace the movement of particulate slurry. The particle size distribution of the slurry was determined and a commonly occurring size thought to be important for phosphorus transport was identified. Fluorescent particles of this size (with a density equivalent to organic matter) were produced. Electrical and chemical properties of the slurry are more difficult to mimic hence the importance of any corroborative evidence from the carbon isotope method.

Naturally occurring carbon isotope (^{13}C) tracer may be used to trace organic matter in the fluvial environment. As organic forms of phosphorus are linked to carbon (Bol *et al.*, 2006) slurry derived organic phosphorus may be traced by tracing slurry derived carbon. By applying C_4 plant (maize) derived slurry (enriched in ^{13}C) to the grassland plots enrichment of ^{13}C in drainage waters should indicate the presence of the applied slurry (see Granger *et al.*, 2006 for details).

Preliminary results

Results are shown which illustrate the breakthrough of the different phases of slurry over selected hydrological events. A signal consistent with a contribution of the dissolved component of slurry to the drainage waters is illustrated for a small hydrological event in Figure 1. This event was monitored in the subsurface drainage waters of Plot 4 in May 2006. During the first 12 hours of this event both tryptophan-like and tyrosine-like fluorescence increase in intensity over the rising limb of the hydrograph and then flatten off. An important observation is that total dissolved phosphorus (TDP) also increases in concentration for the first 3 hours of this period. This is consistent with the natural fluorescence signal detecting the contribution of the phosphorus rich dissolved phase of slurry. Total phosphorus (TP) tends to increase at a faster rate over the first 12 hours of the event. As suspended sediment concentration (SSC) peaks early in the event the increasing TP may reflect high concentrations of TDP.

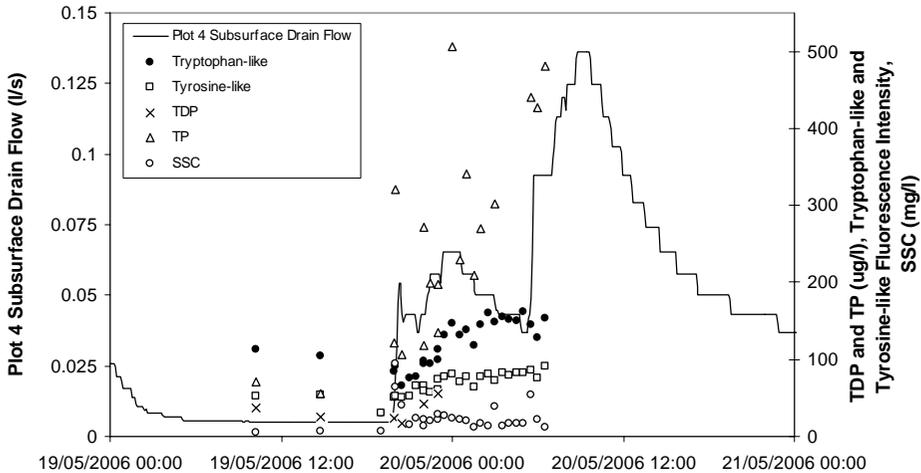


Figure 1. Natural fluorescence, TDP, TP and SSC in subsurface drainage water of Plot 4 during a small flow event, May 2006 (illustrating dissolved phase slurry tracing).

Results from an event simultaneously monitored in subsurface drainage waters of Plot 9 (slurry applied) and Plot 12 (no slurry applied) also illustrates the natural fluorescence signal of a contribution of dissolved phase slurry. Tryptophan-like and Tyrosine-like fluorescence intensities and TDP remain relatively constant preceding and during the event on Plot 12. However, on Plot 9 there is a strong increase in fluorescence and TDP during the event, confirming the results from Plot 4.

The movement of the particulate phase of the slurry was successfully traced using fluorescent-labelled particles. Data collected from the subsurface drainage waters of Plot 9 during a large event in May 2006 are presented in Figure 2. During this flow peak clear responses were observed in fluorescent particle counts and the concentrations of total and volatile suspended solids. It is also significant that TP also shows a similar pattern. Preliminary carbon isotope data also show an enrichment of ^{13}C during this event which is consistent with a contribution of slurry particles. Furthermore, the corresponding peak in TP during the event suggests that phosphorus-rich slurry particles are being traced. Interpreting data from tracing slurry in surface/interflow drainage waters (0-30 cm) has been more difficult owing to fewer samples of this more episodic and, at low flows, less spatially connected hydrological pathway.

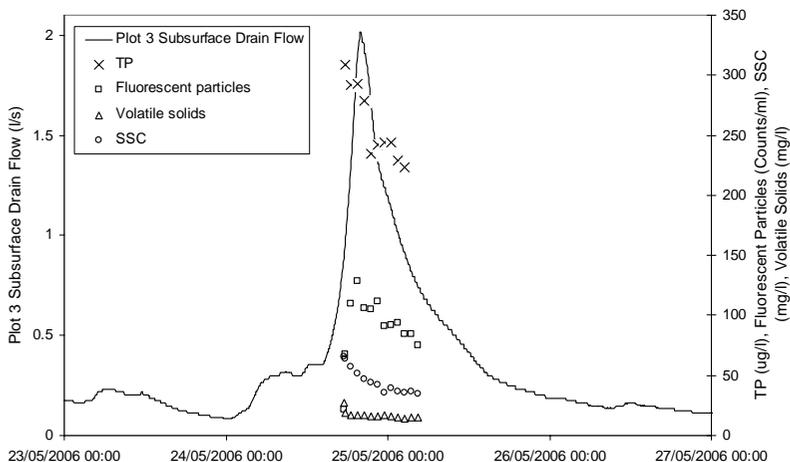


Figure 2. Artificial fluorescent particles, TP, Volatile solids and SSC in subsurface drainage waters of Plot 9 during a large event, May 2006 (illustrating particulate slurry phase tracing).

Conclusions

The novel tracing applications presented in this paper suggest that dissolved phases of slurry may be successfully traced using natural fluorescence while particulate phases may be traced using artificially labelled fluorescent particles and naturally occurring carbon isotopes. Preliminary carbon isotope data corroborate the results from the artificial fluorescent particle tracing. Isotopic analysis of the dissolved carbon is currently being carried out and will be used to corroborate the signals from natural fluorescence. The close correspondence between TDP and natural fluorescence and TP and the artificial fluorescent particles is consistent with the tracing of phosphorus-rich dissolved and particulate slurry phases. Extension of the tracing methods to provide a quantitative assessment of the contribution of slurry to the quality of drainage waters is currently being investigated.

References

- Baker, A., 2002. Fluorescence properties of some farm wastes: implications for water quality monitoring. *Water Research* 36, 189-195.
- Bol, R., Amelung, W. & Haumaier, L., 2006. Phosphorus-31 nuclear magnetic resonance spectroscopy to trace organic dung phosphorus incorporation in a temperate grassland soil. *Journal of Plant Nutrition and Soil Science* 169, 69-75.
- Granger, S.J., Bol, R., Butler, P.J., Haygarth, P.M., Naden, P., Old, G.H., Owens, P.N. & Smith, B.P.G., 2007. Processes affecting transfer of sediment and colloids, with associated phosphorus, from intensively farmed grasslands: tracing sediment and organic matter. *Hydrological Processes* 21, 417-422.
- Water Code, 1998. The Water Code, DEFRA UK, Available from www.defra.gov.uk/farm/environment/cogap/index.htm

Inferring processes of sediment and phosphorus transfer from replicated, intensive grassland plots

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Introduction

After decades of environmental modelling, it is still debatable how robust process descriptions can be inferred from incomplete and imperfect field observations. For the phosphorus (P) transfer problem, we have recently proposed an efficient model development strategy to address this issue (Krueger et al., 2007). Essentially, we argue that the conceptualisation of P transfer processes requires a 'top-down' strategy aiming at the dominant modes of system behaviour, where these can be supported by field observations. At the same time, models should be evaluated by including uncertainties in model structure, parameters and data.

In this study, we applied our model development strategy to data collected from intensively managed grassland plots at the Rowden Experimental Platform (SW England). Discharge, rainfall, suspended solids, total P, and total dissolved (<0.45 μ m) P were measured for two drained and two un-drained hydrologically isolated 1ha plots during the 2006 water year. As one of few grazed grassland field lysimeters in existence, the experimental setup provides a unique opportunity to study storm dynamics of sediment and P transfer at the hillslope scale. This scale of measurement could be considered one sub-unit of the catchment scale where water quality management decisions will be required by international legislation such as the EU Water Framework Directive (2000/60/EC). Discharge and rainfall was measured at 1min resolution, water quality samples were taken at 30-60min intervals during seven storms of 0.5-1.5d duration. At the drained plots, both the drain pathway (85cm) and the surface pathway (including sub-surface flow down to 30cm) were measured. The un-drained plots have only the surface pathway (down to 30cm).

Identification of model processes

The modelling problem of P transfer from the experimental plots can be described in a simplified form of three successive sets of processes: hydrology, sediment dynamics and P dynamics. In the following, we explore to which extent these three compartments could be inferred from the available data, both independently and sequentially.

Hydrologically, the information content in the 1min flow monitoring data for the plots was sufficient to identify a simple bucket model. This model was able to simulate the observed hydrographs for both the drained and un-drained pathways. To account for model input uncertainty, the time series of four rain gauges were used to define multiple rainfall scenarios. Data from a detailed experiment to assess the relationship between stage and discharge which spanned the whole range of observations was used to quantify discharge uncertainty intervals. These were utilised to assess the performance of model simulations.

As an initial hypothesis for the sediment dynamics of the plots, suspended solids concentration could potentially be modelled as a function of discharge, wetness and the slope of the hydrograph. However, when suspended solids concentration was plotted against the above factors, no consistent relationships were found, especially not between events (see Figure 1). The sediment dynamics during rising hydrographs showed erratic behaviour but appeared more consistent during recessions. Therefore, alternative hypotheses of the controls of sediment dynamics during the critical high flow periods have to be developed. Hypotheses could include soil dispersion and re-suspension in the drains and gravel filled ditches that surround the plots. Importantly, these are hypotheses which could not be rigorously tested using the available field observations. In general, measuring these processes at the plot scale remains difficult.

For the P dynamics of the plots, a simple process formulation could be identified from the data. Total P was modelled as a function of suspended solids, a reasonable approximation given that dissolved P accounted for no more than 20% of total P in the samples taken and no relationship between dissolved and total P was apparent.

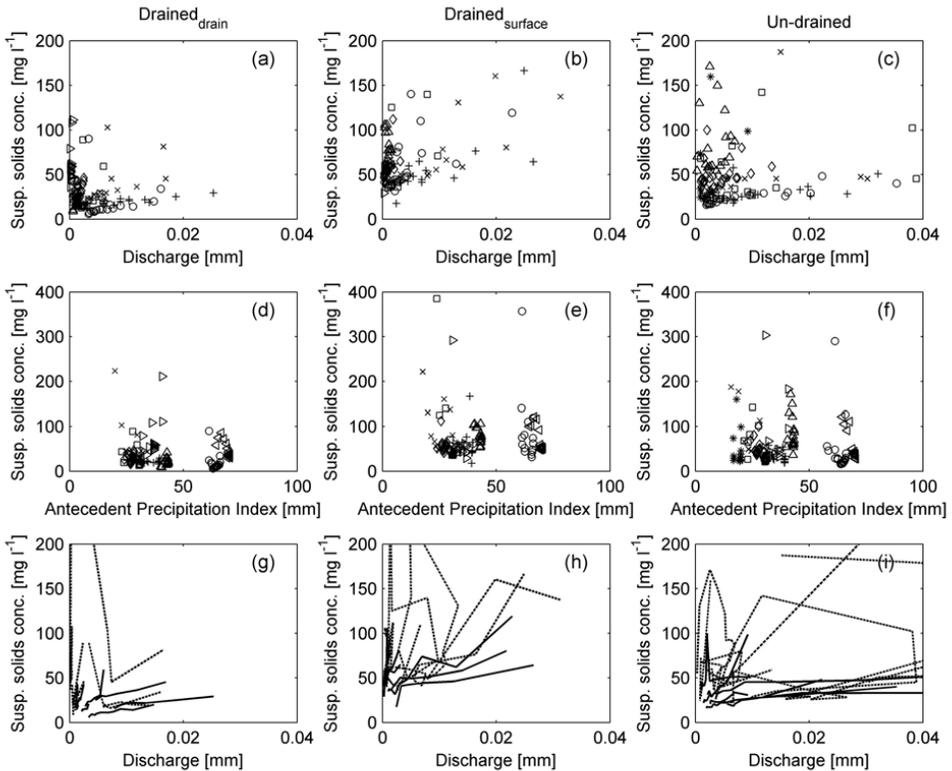


Figure 1. (a) – (c) Suspended solids concentration plotted against discharge for an example of one drained (dual pathway) and one un-drained plot; seven different events are identified by different markers. (d) – (f) Suspended solids concentration plotted against ‘wetness’ as expressed by an Antecedent Precipitation Index ($API_i = k \cdot API_{i-1} + P_i$ with index value API_i and rainfall P_i at the i th timestep and a recession factor k of 0.9 d^{-1}) for the example plots and events. (g) – (i) Suspended solids concentration plotted against discharge for the example plots; data during hydrograph recessions are drawn as solid lines, data during rising hydrographs as dashed lines.

Conclusions

With detailed storm event observations at the well defined 1ha plot scale we approached the limits of understanding the dynamics of sediment and P transfer. The data, however, was sufficient to describe the hydrological behaviour of the plots and quantify the data uncertainties. Between sediment and P dynamics robust relationships could be found, although these did not take data uncertainty into account since our current measurements (e.g. repeated samples) were sparse. The most important ‘missing’ part in the P transfer problem appeared to be the link

between hydrology and sediment dynamics. Employing our model development strategy (Krueger et al., 2007), we presently lack observations to reject competing model process representations. In this case, all feasible hypotheses must be retained and evaluated through model ensembles with potentially very uncertain results. As a concluding remark, the difficulties in robust model identification from typically available data are likely to increase significantly when we scale up from defined plots to heterogeneous catchments where multiple and dynamically varying sources/pathways (often poorly defined) of sediment and P will be common.

References

Krueger, T., Freer, J., Quinton, J.N. & Macleod, C.J.A., 2007. Processes affecting transfer of sediment and colloids, with associated phosphorus, from intensively farmed grasslands: A critical note on modelling of phosphorus transfers. *Hydrological Processes* 21(4), 557-562.

Approaches to estimate phosphorus (P) losses to surface waters at different scales in The Netherlands

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Introduction

In 1990 a methodology was developed to estimate the phosphorus (P) losses to groundwater in the Netherlands, based on the so-called Phosphate Saturation Degree protocol (PSD; Van der Zee 1990 a & b), which subsequently has been adopted in many other countries as an indicator for P loss. The phosphate saturation degree is based on the phosphate accumulation above the highest groundwater level in relation to the phosphate sorption capacity of the same layer. In fact, this measure is an indicator for the *potential* P loss to groundwater, but does not give any information on the *actual* P loss from agricultural land to surface waters. P leaching from the soil matrix is an important source of P loads to surface waters in flat areas with shallow groundwater or pipe drains. To locate those agricultural areas that are the prime contributors of diffuse P losses to the surface waters additional information is needed (Heathwaite et al., 2003). In order to predict the actual P loss to surface waters different approaches have been developed and compared in the Netherlands, namely (a) a comprehensive process orientated approach (b) a metamodel regression tree approach and (c) a simple approach at field scale. Each of the methodologies to estimate surface and subsurface losses will be discussed in more detail.

Methodologies

National approach: STONE

For the evaluation of the impact of Nitrate Action Plans, Environmental studies, Water Framework Directive and OSPAR reporting activities on the nutrient losses to groundwater and surface water the process oriented model STONE (Wolf et al., 2003) has been developed in the Netherlands. The STONE model includes modules for hydrology (SWAP/NAGROM; Kroes and Van Dam, 2003; Van Bakel et al., 2007), fertilizer application rates (CLEAN; Van Tol et al., 2002) and leaching ANIMO (Groenendijk et al., 2005) which comprises descriptions of the carbon, nitrogen and phosphorus cycle. Based on a number of data input sources, the Netherlands is schematized into 6405 unique calculation units. The division into plots is mainly based on a overlay of agricultural districts, meteorological districts, soil type, crop type, and drainage conditions. The STONE model is a nutrient emission model. The output of the model comprises the daily inorganic and organic phosphorus losses to surface waters caused by subsurface leaching and surface runoff (Fig. 1).

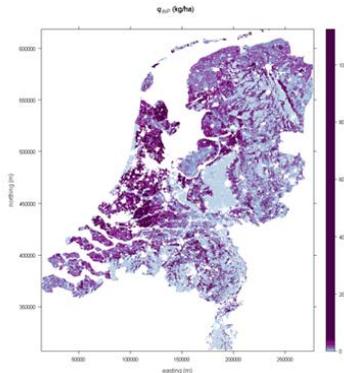


Figure 1. Diffuse P losses from rural areas at national scale caused by leaching (Willems, et al. 2006).

Catchment modelling: NL-CAT

In a number of studies on catchment scale some modules of the STONE (Wolf et al, 2003) model are used to calculate the diffuse losses from rural areas. E.g. in the EC-EUROHARP project ANIMO (Groenendijk et al., 2005) and SWAP were assembled in the NL-CAT tool (Nutrient Losses at CATchment scale; Schoumans et al., 2007) jointly with a surface water quantity (SWQN; Smit & Siderius, 2007) and surface water quality (SWQL; Siderius et al, 2007) tool in order to assess the contribution of agriculture to the total load at the outlet of the catchment. This type of modelling yields detailed results but requires high skills and costs and many efforts for data acquisition. However, where the tool has been set up, a better understanding of the catchment is possible and it is much easier to do simple to rather complex scenario analyses.

Regional approach: Metamodelling; TEMPLE

For screening purposes, a simplified relationship between the main controlling factors and P-leaching can be derived by relating the calculated phosphorus losses from the dynamic process-oriented model, like STONE, and the main input characteristics of that model (SIMPLE; Schoumans et al., 2002). This is called the metamodel approach. Recently, more sophisticated statistical approaches have become available to derive such relationships based on ensemble modelling with regression trees (Breiman, 1996). Figure 2 shows the relationship between the process-oriented model and the regression tree approach called the metamodel TEMPLE (**T**ree-based **E**nsemble **M**odeling of **P**hosphorus **L**eaching to the **E**nvironment'; Walvoort et al., 2007). Based on this statistical procedure it was shown that the most important factors that determine the P losses by subsurface leaching are: actual P status of the top soil (0-0.5 m) and the subsoil (0.5 – 1.2 m), the phosphate sorption capacity of the top and the subsoil, mean highest groundwater level and the amount of upward

seepage. In order to assess also the overland flow (runoff), also land use, P application rate and net precipitation excess are important (Walvoort et al., 2007).

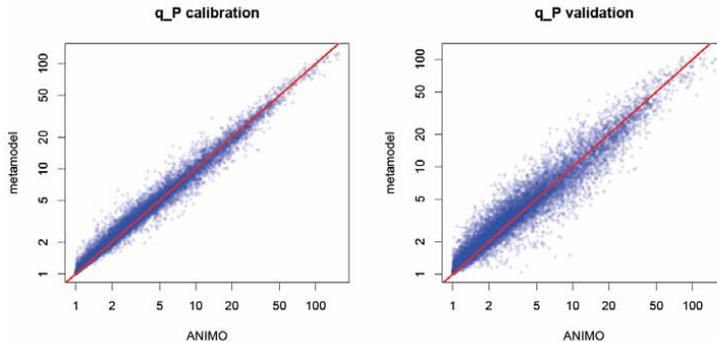


Figure 2. The relationship between the derived metamodel (called TEMPLE) and the process-oriented soil nutrient model and hydrological model (ANIMO/SWAP).

Local field approach: PLEASE

Besides predictions of phosphorus losses on a national scale (Action Plans) or regional / catchment scale (management scenarios in relation to e.g. Water Framework Directive), there is also an increasing interest in the assessment of P losses at local field scale. The main reason is that measures to reduce P losses to the surface water have to be implemented on this scale. Therefore, a procedure (PLEASE; P losses by LEAching to groundwater and P losses by Soil Emissions to surface waters) was developed based on measurements at field scale (Schoumans et al., 2007). The main parameters are the P soil conditions (P source), hydrological conditions (water transport) and location of the field in relation to the location of the surface water (connectivity). With respect to the P source of the soil the soil P test values of the plough layer (0-20 cm) and the layer beneath the plough layer (20-50 cm) of the field are used in relation to the parameters that determine the phosphate sorption capacity of the soil (oxalate extractable aluminium and iron hydroxides: Al_{ox} and Fe_{ox}). For the hydrological conditions the fluctuation in groundwater level is used in combination with the depths of the trenches and ditches (drainage conditions). For the influence of the connectivity a relationship is used between the penetration depth of phosphorus in the soil and the distances of the edges of the field to the surface waters (ditches, brooks, rivers). For a study area in the sandy district of the Netherlands the relevant data were available to estimate the P losses from fields (Schoumans et al., 2007).

Results and discussion: Inter-comparison of the approaches on a regional scale

The different approaches (STONE, TEMPLE and PLEASE) have been compared on a regional scale for the Schuitembeek catchment, a sandy area in the middle of the

Netherlands. Figure 3a shows the results of the national approach (STONE) for the catchment (part of The Netherlands enlarged; Willems et al., 2006). Figure 3b shows the results of the metamodelling approach (TEMPLE; Walvoort et al., 2007). Figure 3c shows the clustering of the results of the field approach (average of combinations of soil type, hydrological conditions and land use based on PLEASE results; Schoumans et al. 2007).

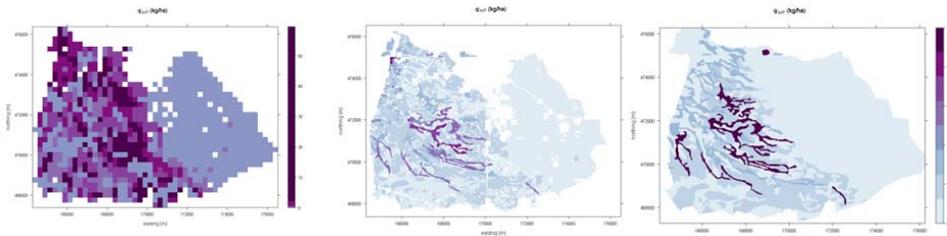


Figure 3. The diffuse P loads from on regional scale (Schuitembeek catchment) based on different approaches (a: national scale, b: regional scale and c: field scale).

It is obvious that an enlarged part of the Netherlands based on the national approach (STONE) does not give sufficiently detailed information on a regional scale because general national input data were used and the discretisation of the area was insufficient. With the metamodelling (TEMPLE) approach and regional input data a more detailed figure of the problem areas is shown. Clustering the results of P losses calculations based on the field approach (PLEASE) shows more or less a similar detailed view of the problem areas, although some additional problem areas are also shown compared to the metamodel approach. A validation of measurements of the P losses within the catchment is necessary in order to address which procedure is best with respect to the regional scale. Furthermore, the results of the NL-CAT approach should also be compared. This part of Dutch research in the near future.

Conclusions

In the Netherlands different methodologies and tools have been developed in order to predict the P losses from agricultural land to surface waters on different scales.

- With respect to P emissions from rural areas a process-orientated dynamic approach has been developed (STONE), which is used to evaluate the impact of Action Plans at national scale. This approach is also used for reporting purposes (OSPAR and national inland Emissions Registrations).
- For regional or catchment scale different approaches are available in the Netherlands. A process-orientated dynamic approach for the interaction between land and surface water is available for catchments (NL-CAT) and a screening metamodel approach called TEMPLE
- A simple local field approach, called PLEASE, was developed to predicted the diffuse losses from a field by subsurface losses to surface waters.

- A comparison of the different approaches was made on regional scale for the Schuitembeek catchment in the Netherlands. Enlargement (blow up) of the results of a national approach (STONE) shows a very rough distribution of the P losses. The results of the regional (TEMPLE) approach and the clustered results of the field approach (PLEASE) give more or less similar results, although there seem to be some differences for specific subcatchments. For that reason, detailed measurements are needed in order to determine which of these simple approaches will give the best results on the regional scale. This is part of future Dutch research.

References

- Bakel, P.J.T van, Massop, H.Th.L., Kroes, J.G., Hoogewoud, J., Pastoors, R. & Kroon, T., 2007. Actualisatie Hydrologie voor STONE 2.3. Aanpassing randvoorwaarden en parameters, koppeling tussen NAGROM en SWAP, en plausibiliteitstoets. Alterra report (in press).
- Breiman, L., 1996. Bagging Predictors. *Machine Learning* 24,123-140.
- Heathwaite, L., Sharpley, A.N., Bechmann, M. & Rekolainen, S., 2005. Assessing the risk and magnitude of agricultural nonpoint source phosphorus pollution. Chapter VI-3. In Sims, J.T., Sharpley, A.N, editors. *Phosphorus: Agriculture and the Environment*. Madison, WI: American Society of Agronomy. p. 981-1020.
- Groenendijk, P., Renaud, L.V. & Roelsma, J., 2005. Prediction of Nitrogen and Phosphorus leaching to groundwater and surface waters; Process descriptions of the animo 4.0 model. Alterra report 983.
- Kroes, J.G. & van Dam, J.C. (eds), 2003. Reference Manual SWAP version 3.0.3. Alterra-report 773, 211 pp, Alterra, Research Institute, Wageningen, The Netherlands.
- Siderius, C., Groenendijk, P., Jeuken, M.H.J.L. & Smit, A.A.M.F.R., 2007. Process description of NUSWALITE.A simplified model for the fate of nutrients in surface waters. Alterra-Report 1226.2.
- Smit, A.A.M.F.R. & Siderius, C., 2007. Process description of SURFACEWATER. A simplified hydraulic model. Alterra-Report 1226.1
- Schoumans, O.F., Groenendijk, P. & Siderius, C., 2007. NL-CAT application to six European catchments. Report 1205, Alterra, Wageningen, NL.
- Schoumans, O.F., Mol-Dijkstra, J., Akkermans, L.M.W. & Roest, C.W.J., 2002. SIMPLE: Assessment of non-point phosphorus pollution from agricultural land to surface waters by means of a new methodology. *Water Science & Technology* 45(9), 177-182.
- Schoumans, O.F., Groenendijk, P., van der Salm, C. & Pleijter, M., 2007. Methodiek voor het aanwijzen van fosfaatlekkende gronden. PLEASE. Alterra rapport, Wageningen, NL.
- Van Tol, S., van den Born, G.J., van Egmond, P.M., van der Hoek, K.W., Hoogervorst, N.J.P. & Knol, O.M., 2002. CLEAN2.0, model voor de berekening van stikstof- en fosfaatemissies uit de landbouw. Modelbeschrijving. RIVM rapport 773004010/2002.
- Van der Zee, S.E.A.T.M., van Riemsdijk, W.H. & de Haan, F.A.M., 1990a. Het protocol fosfaatverzadigde gronden. Deel I: Toelichting. Vakgroep Bodemkunde en Plantevoeding. Landbouwwuniversiteit Wageningen.

- Van der Zee, S.E.A.T.M., van Riemsdijk, W.H. & de Haan, F.A.M., 1990b. Het protocol fosfaatverzadigde gronden. Deel II: Technische Uitwerking. Vakgroep Bodemkunde en Plantevoeding. Landbouwwuniversiteit Wageningen.
- Walvoort et al., 2007. Walvoort, D.J.J., van der Salm, C. & Schoumans, O.F. (in press). Delineating areas vulnerable to P leaching by statistical metamodelling. Alterra, Wageningen, the Netherlands.
- Willems, W.J., Beusen, A.H.W., Renaud, L.V., Luesink, H.H., Conijn, J.G., Oosterom, H.P., v.d. Born, G.J., Kroes, J.G., Groenendijk, P. & Schoumans, O.F., 2006. Nutriëntenbelasting van bodem en water: evaluatie van de gevolgen van het nieuwe mestbeleid. Bilthoven: Milieu- en Natuurplanbureau & Wageningen UR, 2006. Rapport 50031003. 111 pp.
- Wolf, J., Beusen, A.H.W., Groenendijk, P., Kroon, T., Rötter, R. & van Zeijts, H., 2003. The integrated modeling system STONE for calculating nutrient emissions from agriculture in the Netherlands. *Environmental Modelling & Software* 18, 597—617.

Large-scale phosphorus transport model

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Introduction

The Watershed Simulation and Forecasting System (SYKE-WSFS) of the Finnish Environment Institute (SYKE) simulates the hydrological cycle for the whole land area of Finland (Vehviläinen & Huttunen 2002). The phosphorus transport model has been developed as a part of SYKE-WSFS. The SYKE-WSFS phosphorus transport and load model can be classified as a conceptual nutrient transport model lying between the physically based nutrient transport models and simple source apportionment assessment tools such as VEPS (Tattari and Linjama, 2004). The phosphorus transport is based on the simplified conceptual method and calibration is used to estimate the parameters describing the phosphorus transport.

Description of the model

The phosphorus transport modelling is incorporated directly into the SYKE-WSFS as separate subroutines. The phosphorus model consists of three parts: phosphorus transport from land areas, phosphorus transport and retention in rivers, and phosphorus balance in lakes. The model unit for phosphorus transport from land areas is a 3rd level subcatchment. There has been an attempt to go to the smaller modelling unit – to divide the 3rd level subcatchment into smaller subcatchments depending on the number of the lakes in the 3rd level subcatchment. There are four types of load simulated within each subcatchment - load from agricultural fields, load from other areas (forests and bogs), load from scattered settlement and point load.

Phosphorus transport modelling is based on the assumption that the main dynamic variable governing the phosphorus transport is runoff (Eq. 1). The total concentration from the subcatchment is then calculated as the weighted average depending on the percentage of the different land cover types.

$$c_x = b \times x_{SCA} \times x_{plough} \times \frac{r_{sim}^d}{r_{max,sim}} \times c_{max} + c_{min} \quad (1)$$

c_x – daily total phosphorus concentration ($\mu\text{g/l}$), b , x_{SCA} , x_{plough} , c_{max} , c_{min} , d , – calibrated parameters, r_{sim} – simulated daily runoff, mm, $r_{max,sim}$ – simulated mean maximum runoff of 1990-2006.

Phosphorus transport and retention through the river is simulated similar to the runoff routing through the river. For each 3rd level subcatchment the river length and width are estimated. Rivers are divided into about 1 km long river stretches. Each river

stretch is then simulated as a reservoir, including the inflow, volume and outflow of the each river stretch. The phosphorus concentration is then simulated taking into account the simulated volumes of the river reaches.

Lake hydrology in SYKE-WSFS is simulated by a water balance model. Water balance components, e.g. daily inflow, lake evaporation, lake precipitation and the daily volume of the lake are simulated, current water level is estimated depending on the lake's volume, the outflow from the lake is calculated according to the functional relationship between outflow and water level in the lake. The lake phosphorus balance is simulated according to the mass balance equation (Eq. 2):

$$\frac{dm}{dt} = I(t) - Q_{out} - \partial_{sed} Vc + rA \quad (2)$$

The mass balance differential equation is solved by using the Euler's method with the following variables: m is the mass of total phosphorus (M), t the time (T), $I(t)$ the inflow loading to the lake ($M T^{-1}$), Q_{out} the outflow load from the lake ($M T^{-1}$), $\partial_{sed} Vc$ the sedimentation load ($M T^{-1}$), rA the internal load ($M T^{-1}$), ∂_{sed} the sedimentation rate (T^{-1}), V the volume of the lake (L^3), c the phosphorus concentration ($M L^{-3}$), A the surface area of the lake (L^2) and r the release rate of phosphorus from the sediment ($M L^{-2} T^{-1}$).

The statistical equation of Canfield and Bachmann (1981) relating the yearly sedimentation coefficient and external loading and the lake's volume has been tested to estimate the yearly sedimentation rate for lakes. It has been successfully used to estimate yearly sedimentation rates for various types of lake, e.g. by Frisk et al. (2001). Daily sedimentation rate σ_{day} (d^{-1}) is estimated by dividing σ into 365. Suggested Canfield and Bachmann (1981) equation cannot be directly applied, because there is a very wide range of simulated lakes in terms of both catchment areas and volumes of the lakes. At the moment the daily sedimentation rate is a calibrated parameter and it can be calibrated separately for each lake (limits for σ_{day} is from 0.002 to 0.003 which corresponds to yearly sedimentation rates of 0.73 to 1.09). There are seasonal coefficients applied to simulate changing sedimentation rates depending on the season. Internal load component was added to the mass balance equation, because we had difficulties in simulating the concentration pattern within the year in eutrophic lakes where internal load is an important source of load during the summer months.

Results

The SYKE-WSFS phosphorus transport model has now been applied to three catchments both without and with lakes – Aurajoki river catchment (874 km²),

lisalmen watercourse catchment (5583 km²) and Karvianjoki river catchment (3438 km²). The modelling system is built up for the whole of Finland, but the calibration of the model is for the present done for the three above-mentioned catchments. The SYKE-WSFS phosphorus transport model allows simulations of a very wide range of phosphorus output variables - daily simulated concentration in rivers and lakes, daily simulated load from the river catchment to the river and load to the estuary of the river, daily simulated loading to the lakes, sedimentation and outflowing load from the lakes, annual/monthly/weekly phosphorus loads to the estuary of the river, annual/monthly phosphorus balances for each lake and for various sizes of river catchments including a large number of lakes. Calibration against all available observation data gives us the possibility to simulate the heterogeneity of the phosphorus load from the different 3rd level subcatchments.

Discussion and conclusions

The development of the phosphorus transport model is still ongoing, but at this stage we can conclude that a simple conceptual method of phosphorus transport simulation can be used within large-scale hydrological models. It works well in river catchments without lakes where concentrations in the streams represent the phosphorus leaching dynamics from the catchment land area. In this case the observed phosphorus concentrations in the river can be used to calibrate parameters for land transport equation. The simulated system gets more complex when there are lakes in the catchment. Then observed concentrations in the streams do not solely represent the phosphorus leaching dynamics from the land areas and the calibration procedure is less efficient. We are working on both further development of the calibration system and description of the lake processes by more physically based equations, to reduce the number of calibrated processes.

References

- Canfield, D.E., Jr. & Bachmann, R.W., 1981. Prediction of total phosphorus concentrations, chlorophyll a and Secchi depths in natural and artificial lakes. *Can. J. Fish. Aquat. Sci.* 38, 414-423.
- Frisk, T., Pjankovskis, V., Bilaletdin, Ä., Paananen, A., Kaipainen, H. & Bruveris, J., 2001. The Possibilities of Water Protection and Restoration of Lake Sila. *The Finnish Environment* 484, Pirkanmaa Regional Environment Centre. 51 pp.
- Tattari, S. & Linjama, J., 2004. Vesistöalueen kuormituksen arviointi. *Vesitalous* 3/2004, 26-30 (in Finnish).
- Vehviläinen, B. & Huttunen, M., 2002. The Finnish watershed simulation and forecasting system (WSFS). XXIst Conference of the Danubian countries On the hydrological forecasting and hydrological bases of water management. Bucharest-Romania. 2-6 September 2002.

Predicting phosphorus transfers within agricultural catchments across England and Wales using the PSYCHIC model

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Introduction

The PSYCHIC (Phosphorus and Sediment Yield Characterisation In Catchments) model has been developed as a decision support tool (DST) to enable catchment stakeholders to identify areas within catchments at elevated risk of phosphorus transfer to surface waters. The model was developed with field scale data, but a key strength is its applicability at catchment scale using national datasets such as those within the MAGPIE system (Lord and Anthony, 2000). Like all models with an empirical element it will require calibration for use in a particular catchment, and there is a risk that calibration will be used incorrectly to compensate for failings in model design by altering uncertain input data such as the point source inventory or processes such as channel retention. There is thus a need to demonstrate general validation of the model across a range of environmental and landscape conditions, by considering a large number of catchments such that the uncertainty in input data is controlled, and we can test whether the model does properly respond to variation in environmental risk factors such as soil type, climate and land use.

Methodology and results

The PSYCHIC model was applied to calculate diffuse total phosphorus (TP) inputs from agricultural land to 7,800 river catchments across England and Wales. The national total loss was 5.7 kt P equivalent to 0.38 kg ha^{-1} TP from all land uses, including urban areas. Inputs of total phosphorus from point sources were estimated using an inventory of consented discharges from sewage treatment works and compliance sample results for soluble reactive phosphorus (SRP). The national total loss was 18.1 kt P contributing 75% of the total load input to rivers annually. The modelled loads were adjusted for retention using the model of Behrendt and Opitz (2000) and compared against regional measurements of total phosphorus exported to the sea areas around England and Wales under the Harmonised Monitoring Scheme. Measured and modelled loads were in the range 0.4 to 4.3 kg ha^{-1} TP (Figure 1). The ratio of measured and modelled loads demonstrated a slight over-estimate (0.91 ± 0.04) but an overall high predictive power ($r^2=95\%$).

The load comparison was regarded only as a validation of the point source inventory due to the dominance of point source contributions at this scale. Reliable observed loads of phosphorus are available for very few rural catchments in England and Wales, but measurements of SRP concentrations in rivers are more widespread. We

therefore developed a statistical methodology that predicted average SRP concentrations as a weighted sum of modelled concentrations due to the point and diffuse sources inputs alone. The model was calibrated using 5 years of monitoring data for 1850 sites. A non-linear solver was used to calibrate the model weights W_P and W_D (for point and diffuse sources, respectively) using log-transformed measured and predicted average concentration data. The point source weight W_P was estimated as 1.15 ± 0.037 and the diffuse source weight W_D was estimated as 0.68 ± 0.020 . The calibrated model explained 78% of the variance in the measured concentration data. The model fit was consistent for both point and diffuse source dominated catchments, therefore supporting the PSYCHIC model design (Figure 2).

Discussion

Overall, the PSYCHIC model was able to provide unbiased regional estimates of loads and to reproduce the spatial pattern of observed SRP concentrations in both point and diffuse source dominated catchments. Some model design issues were identified. For example, the erodibility of clay soils and the mobilisation of soil and associated phosphorus by rainfall kinetic energy under permanent grass are believed to be generally under-estimated. Issues of model bias due to input data were also identified in areas where the presence or absence of assisted soil drainage was incorrectly inferred from national datasets. A greater proportion of the phosphorus input from sewage effluent is bio-available compared with the phosphorus in agricultural drainage (Ekholm and Krogerus, 2003). The comparison of the modelled diffuse loads and point source inventory indicate that sewage effluent discharges continue to be the dominant source of bio-available phosphorus input to rivers in England and Wales despite major investment in phosphorus stripping. Model validation is planned to continue using both high quality field scale experimental data, and at basin scale utilising storm event sampling supported by sediment fingerprinting studies to provide a source apportionment of observed sediment inputs.

Acknowledgements

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References

- Behrendt, H. & Opitz, D., 2000. *Hydrobiologia* 410, 111-122.
- Lord, E. & Anthony, S., 2000. *Soil Use and Management* 16, 167-174.
- Ekholm, P. & Krogerus, K., 2003. *Hydrobiologia* 492, 29-42.

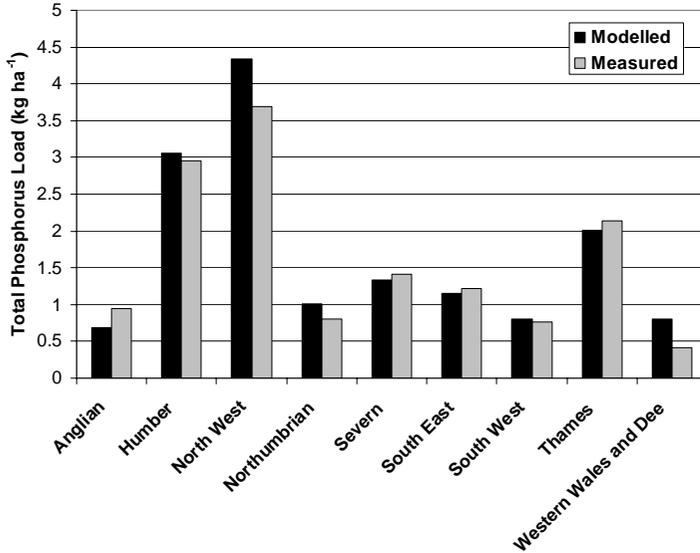


Figure 1. Measured and modelled annual average total phosphorus loads exported from river basins of England and Wales. Measured data are from Harmonised Monitoring Scheme sites (OSPAR, 1999 to 2003).

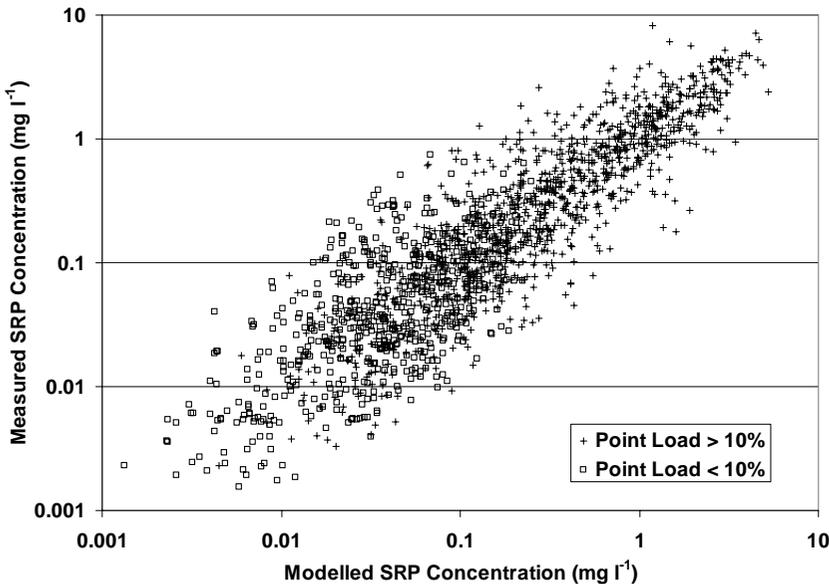


Figure 2. Measured and modelled annual average SRP concentrations at river monitoring sites, for point and diffuse source dominated catchments. Measured data are from Environment Agency monitoring sites (EA, 1996 to 2000).

The integrated catchment model of phosphorus dynamics (INCA-P), a new structure to simulate particulate and soluble phosphorus transport in European catchments

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Introduction

The Integrated Catchment Model of Phosphorus dynamics (INCA-P) provides a process-based representation of phosphorus storage and transport in the terrestrial and in-stream components of river catchments while attempting to minimise data requirements and model structural complexity (Wade et al., 2002). INCA-P is designed to investigate:

- the key factors and processes controlling phosphorus storage and transport in the catchment terrestrial component at a daily time-step and over the longer term;
- the in-stream dynamics of particulate and soluble phosphorus;
- aquatic plant and epiphyte growth as a function of available phosphorus, flow (residence time), solar radiation and water temperature.

INCA-P builds on the established Integrated Catchment Model of Nitrogen dynamics (INCA-N) and the 'Kennet' model which simulates in-stream phosphorus and macrophyte and epiphyte dynamics (Wade et al., 2001). INCA-P has been applied to the River Lugg, a large river system in the UK to determine the relative contribution from diffuse and point sources, and to investigate the likely impact of phosphorus removal from effluent on the in-stream phosphorus concentrations (Wade et al., in press). As part of the Euro-limpacs project, INCA-P is currently being developed and tested in the UK and Scandinavia. The talk will review the applications of INCA-P to date, the new model structure, and will consider some future directions in modelling phosphorus dynamics at the catchment scale.

Results from an initial application

The River Lugg in the UK was chosen as a study area because it is known to have very high (approximately 1 mg P l^{-1}) stream water phosphorus concentrations (measured as total phosphorus, TP) in its lower reaches. The Lugg is designated a 'eutrophic sensitive' area under the Urban Wastewater Treatment Directive.

INCA-P was able to simulate the seasonal variations and inter-annual variations in the in-stream phosphorus concentrations. This result suggests that at larger spatial (> 50 km²) and temporal (seasonal and inter-annual) scales, readily-available land use and hydrological data can be used with a medium complexity model to simulate in-stream P loads. However, there were difficulties in simulating the daily variations in the in-stream phosphorus concentrations due to uncertainty in the model structure and parameters, and the limitations of the input data.

The upper reaches of the Lugg are dominated by grassland and woodland, and the patterns in the streamwater total phosphorus concentrations are typical of diffuse source inputs. The modelled estimate of the relative contribution to the in-stream phosphorus load from diffuse and point sources was 9:1. In the lower reaches, which are more intensively cultivated and more urban, the streamwater total phosphorus concentration dynamics were predominantly influenced by point sources and the simulated relative diffuse to point source contribution to the in-stream phosphorus load was 1:1. Given the predominance of grassland in the catchment, this is the main diffuse source in terms of total contribution to the in-stream P load, though cereal is important in terms of annual P export per hectare.

It was difficult to identify P source areas given the uncertainty in the Agricultural Census data used; these data are modified under the Non-disclosure Act to prevent the identification of individual holdings. Better spatial accuracy of land use and livestock statistics, and more frequent measurements of flow and TP and soluble reactive phosphorus (SRP) concentrations in the final effluent from Sewage Treatment Works, are required to improve the phosphorus mass balance at a catchment scale.

Preliminary scenarios suggest that phosphorus removal to a total phosphorus concentration of 1 mg P l⁻¹ at works discharging directly to the main channel of the Lugg may reduce the mean stream water SRP concentration to approximately 60 µg P l⁻¹ during the summer. Further work is needed to verify this result and to understand the potential ecological implications and likely costs of phosphorus removal from effluent.

Model revisions

The following model revisions are being tested as part of the Euro-limpacs project:

- additional landscape units that describe variations in soil type, slope and vulnerability to sediment erosion;
- a physically-based method of calculating sediment delivery and in-stream routing based on the INCA-Sed model (Jarritt and Lawrence, in press);

- separation of particulate and soluble forms of phosphorus in both the terrestrial and in-stream components of the model, with a mechanism to describe phosphorus sorption to, and release from, soil and stream sediment based on the concepts of phosphorus release kinetics and an equilibrium concentration (House and Dennison, 2000);
- initial values for soil phosphorus that relate to readily available measures of the phosphorus attached to the soil (mg P kg⁻¹), such as soil test P (Yli-Halla et al. 2005).

An update will be given on the success of these revisions.

Acknowledgements

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References

- House, W.A. & Denison, F.H., 2000. Factors influencing the measurement of equilibrium phosphate concentrations in river sediments. *Water Research* 34(4), 1187-1200.
- Jarritt, N.P. & Lawrence, D.S.L., Fine sediment delivery and transfer in lowland catchments: Modelling suspended sediment concentrations in response to hydrologic forcing. *Hydrological Processes*. In press.
- Wade, A.J., Hornberger, G.M., Whitehead, P.G., Jarvie, H.P. & Flynn, N., 2001. On modelling the mechanisms that control in-stream phosphorus, macrophyte and epiphyte dynamics: An assessment of a new model using general sensitivity analysis. *Water Resources Research* 37, 2777–2792.
- Wade, A.J., Whitehead, P.G. & Butterfield, D., 2002. The Integrated Catchments model of Phosphorus dynamics (INCA- P), a new approach for multiple source assessment in heterogeneous river systems: model structure and equations. *Hydrol. Earth Syst. Sci.* 6, 583–606.
- Wade, A.J., Butterfield, D., Griffiths, T., Whitehead, P.G., Eutrophication control in river-systems: an application of INCA-P to the River Lugg. *Hydrol. Earth Sys. Sci.* In press.
- Yli-Halla, M., Tattari, S., Bärlund, I., Tuhkanen, H.-R., Posch, M., Siimes, K., Rekolainen, S., 2005. Simulating processes of soil phosphorus in geologically young acidic soils in Finland. *Transactions of the ASAE* 48(1), 101-108.

Risk assessment of P-losses and uncertainties in soil and surface water systems at catchment scale

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Introduction

The Water Framework Directive demands the implementation of measures in order to reach the defined targets for water bodies. Risk assessment tools can be very helpful in identifying the sources of nutrient pollution and the magnitude of relevant pathways. Risk can be defined as the product of the probability of an event occurring and the consequences of that event. The assessment of risk is an important part of the characterisation process of catchments and in making judgements of the likelihood that water bodies will fail to meet defined environmental objectives. The spatial and temporal variability of P-export rates from agricultural land is high, since many factors influence the processes of P- mobilization and the P-loss pathways to surface water bodies. There is a key role for modelling risk in order to relate pressures to impacts.

Risk assessment methods

Estimation of diffuse pollution has commonly been conducted by: (1) simple models and expert systems that have the advantage of operating on the basis of the limited data available, and (2) deterministic, physically-based process models mostly used by research communities to get a more detailed understanding of P loads. Heathwaite *et al.* (2005) provide a detailed evaluation of the key features of risk tools and models used for risk assessment.

An example of a simple risk assessment tool method is the Irish REALTA risk mapping method (Hughes *et al.*, 2005), applied in the EuroHarp-project (EVK1-CT-2001-00096) for a number of European catchments, based on a number of factors that affect the loss of P from agricultural systems and their subsequent transport to surface water. A ranking scheme was developed whereby each of the five P loss indicators was subdivided into zones of relative risk, each of which was given a numerical value for scoring purposes. The relative importance between factors was represented by a 'weighting'. The resulting composite map establishes the range of potential agricultural risk areas across the catchment. The results of an Irish water quality monitoring programme confirmed a strong correlation between the areas identified as being of high or very high potential risk and poor water quality.

The NL-Cat modelling system is an example of a complex mechanistic process model operating at a catchment scale and from which some process modules are identical to the modules of the nation-wide STONE model for the Netherlands (Wolf et al., 2003). The modelling system comprises specialised modules for soil water flow, soil nutrient transformations and leaching, surface water flow and nutrient concentrations.

Uncertainty analysis of the NL-Cat model for the Vechte-catchment

Information about uncertainty is generally assumed to improve the quality of decision making; it adds a useful dimension by revealing the reliability of the knowledge produced. One of the objectives of the EC-FP5 HarmoniRiB project (EVK1-CT2002-00109) was to establish a practical methodology for assessing and describing uncertainty originating from data and models used in decision making processes (Refsgaard et al., 2005). An uncertainty analysis was conducted for the NL-Cat modelling system (Schoumans et al., 2005) using data of a sub-catchment of the Vecht basin (Bijlsma et al., 2006). The objective of the case study was to predict surface water concentrations and its associated uncertainty due to variability in parameters and input data. A number of uncertain information sources were selected on the basis of prior sensitivity analysis of the leaching module (Groenenberg et al., 1999) and expert judgement (Table 1). The quantitative assessment of fertilizer application rates covered both organic and mineral fertilizer and was restricted to the dominant land use of grass and maize cultivation, and the dominant soil type of sand. A uniform distribution for fertilizer application rates was defined (Table 1). Soil air oxygen diffusion related to aeration and denitrification is described by a simple power function of soil air content for which uncertainties of the multiplier and the exponent were assessed jointly on the basis of experimental data (Bakker et al., 1987) by performing bootstrapping techniques (Efron and Tibshirani, 1993). An uncertainty distribution for the phosphorus background concentration in groundwater was estimated by aggregating observed concentration data, which resulted in a normal distribution. The uncertainty in the iron and aluminium content of the upper soil (1-120 cm) was assessed by a geostatistical analysis at the plot scale using conditional sequential Gaussian (block) simulation (Goovaerts, 1997). Vertical segmentation was reflected by the original deterministic ratios.

Table 1. The median and 90% confidence interval (90%-CI) for the input of selected uncertainty sources, represented by the median, 0.05- and 0.95-quantile for the total catchment average input.

Uncertainty source	Median	5%-quantile	95%-quantile	90% CI of median
Fertilizer application load (kg N/ha effective N)	527	442	643	38%
Phosphorus background concentration (PBC) in groundwater (mg/l)	0.16	0.15	0.18	19%
Multiplicator (first) and exponent (second) of soil air oxygen diffusion relation (-)	0.65; 2.46	0.34; 1.95	1.59; 2.94	92; 40%
Iron and aluminum content of the upper soil (mmol/kg)	56.7	54.4	62.4	14%

Results and conclusions of uncertainty analysis

Summer averaged phosphorus loads from point sources and diffuse sources (including uncertainty bandwidth) and the relative contribution of the uncertainty of different factors to the total uncertainty for 1999 have been depicted in Fig. 1.

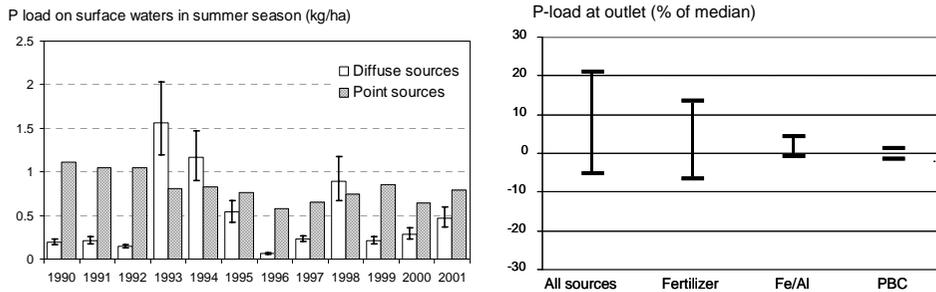


Figure 1. P-load on surface water from point sources and diffuse sources, incl. 5-95% range, (left) and the 90% CI of the uncertainty bandwidth for P-loads at the Regge-outlet in 1999 for the total assumed variation of all sources and the contribution of the individual uncertainty sources (right). The uncertainty bandwidth is characterized by the output of the simulation runs with 0.05- and 0.95 quantile input.

Uncertainty in fertilizer application rates contributed most to the output uncertainty. From long-term scenario simulations it appeared that the uncertainty associated with phosphorus concentrations increased in time despite the reduction of P-surpluses. The results showed smaller uncertainty bandwidths for all variables than expected. The different spatial or temporal scale of the output chosen largely influenced the median and uncertainty bandwidth of the results. For larger spatial and temporal aggregation levels the uncertainty bandwidths in P-concentrations and loads in surface waters generally decreased due to averaging out of extreme values.

References

- Bakker, J.W., Boone, F.R. & Boekel, P., 1987. Diffusion of gasses in soil and oxygen diffusion coefficients in Dutch arable fields. Rep. 20. Inst. for Land and Water management, Wageningen, the Netherlands (in Dutch).
- Bijlsma, R., Groenendijk, P., Boers, P. & Blind, M., 2006. Uncertainty assessment on the nutrient concentration in the Regge catchment, Vecht River basin, the Netherlands. HarmoniRiB report D7.4 www.harmonirib.com.
- Efron, B. & Tibshirani, R.J., 1993. An Introduction to the Bootstrap. Chapman & Hall, New York, USA.
- Goovaerts, P., 1997. Geostatistics for Natural Resources Evaluation. Oxford University Press, UK.
- Groenberg, J.E., Van der Salm, C., Westein, E. & Groenendijk, P., 1999. Gevoeligheidsanalyse en beperkte onzekerheidsanalyse van het model ANIMO. Rep. 446, DLO-Staring Centre, Wageningen, the Netherlands.
- Heathwaite, L., Sharpley, A.N., Bechmann, M. & Rekolainen, S., 2005. Assessing the risk and magnitude of agricultural nonpoint source phosphorus pollution. Chapter VI-3. In Sims, J.T., Sharpley, A.N. eds. Phosphorus: Agriculture and the Environment. Madison, WI: American Society of Agronomy. p. 981-1020.
- Hughes, K.J., Magette, W.L. & Kurz, I., 2005. Identifying critical source areas for phosphorus loss in Ireland using field and catchment scale ranking schemes. *Journal of Hydrology* 304, 430-445.
- Refsgaard, J.C., Nilsson, B., Brown, J., Klauer, B., Moore, R., Bech, T., Vurro, M., Blind, M., Castilla, G., Tsanis, I. & Biza, P., 2005. Harmonised techniques and representative river basin data for assessment and use of uncertainty information in integrated water management (HarmoniRiB). *Environmental Science & Policy* 8, 267-277.
- Schoumans, O.F., Groenendijk, P. & Siderius, C., 2005. NL-CAT application to six European catchments. Report 1205, Alterra, Wageningen, the Netherlands.
- Wolf, J., Beusen, A.H.W., Groenendijk, P., Kroon, T., Rötter, R. & Van Zeijts, H., 2003. The integrated modeling system STONE for calculating nutrient emissions from agriculture in the Netherlands. *Environmental Modelling & Software* 18, 597-617.

Application of the ICECREAMDB model to quantify phosphorus losses from Sweden

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Introduction

Because of the adverse effects of eutrophication, reduction of phosphorus losses to the Baltic Sea is considered necessary. To achieve this in a cost-effective way, methods for estimation of the contribution from different sources is required. Since agriculture is the single largest source in many areas, it is especially critical to quantify the phosphorus losses from arable land. For this purpose, a new method to calculate P-losses from arable land in Sweden based on the dynamic simulation model ICECREAMDB has been developed. Here, we will briefly describe the application of this method for the Pollution Load Compilation (PLC-5) to HELCOM and present some results including comparisons with measured data.

The model

ICECREAMDB is a shell program around the simulation model ICECREAM (Larsson et al., 2007) developed to fit into the NLeCCs-system (Nutrient Leaching Coefficient Calculations) already applied for the Swedish national calculations of N, containing file structures for model input data at this scale. The NLeCCs-system also produces crop-sequences to support calculation of annual climate-normalized loss coefficients. ICECREAM is a dynamic model for quantification of P losses from arable land, taking all major P transformation processes and flow pathways into account. It is based on the CREAMS model developed in USA, but it has been adopted for Nordic agro-environmental conditions by adding descriptions of snow pack and soil freezing, macropore flow and impact of low rain intensities in RUSLE (Posch and Rekolainen, 1993; Rekolainen and Posch, 1993).

Model application

Simulations with ICECREAMDB were performed for Sweden, which has been divided into 22 climate-production regions (CPR) for this purpose. The factors characterizing a climate-production area is the climate, the accumulated annual discharge, yield and amount of applied fertilizer and manure for each crop, dates for crop management operations such as sowing, harvesting, fertilizer application, soil tillage etc., and dates for beginning and end of the growing season. For each of the 22 regions, climate-normalized phosphorus loss concentrations for the year 2005 were calculated based on a matrix with separate data for: 15 crop classes; 10 soil texture

classes; three soil P classes and three slope classes, resulting in a matrix of 29700 specific P-loss concentrations.

Model evaluation

The model simulations were compared with measured concentrations from the 13 agricultural ‘observation fields’ included in the national programme for monitoring of nutrient loads from arable land (Johansson and Gustafson, 2006). These fields are tile-drained and integrate losses by surface runoff and via leaching through the soil profile and they cover a wide range of Swedish agro-environmental conditions (e.g. soils, P-amounts in soil, slopes, crops and climates).

Results

Figure 1 shows simulated total P losses (kg ha^{-1}) for the 22 regions in Sweden. The losses are weighted in relation to the distribution of different crops, soils, slopes and P-amounts in soil in each region. Under the agro-environmental conditions prevailing in Sweden, the model was by far most sensitive to soil texture followed by CPR, P-HCl, crop and slope, and the highest losses could consequently be found in regions dominated by clay soils (e.g. Mälardalen and the plain of Östergötland, and the East-Swedish valley area).

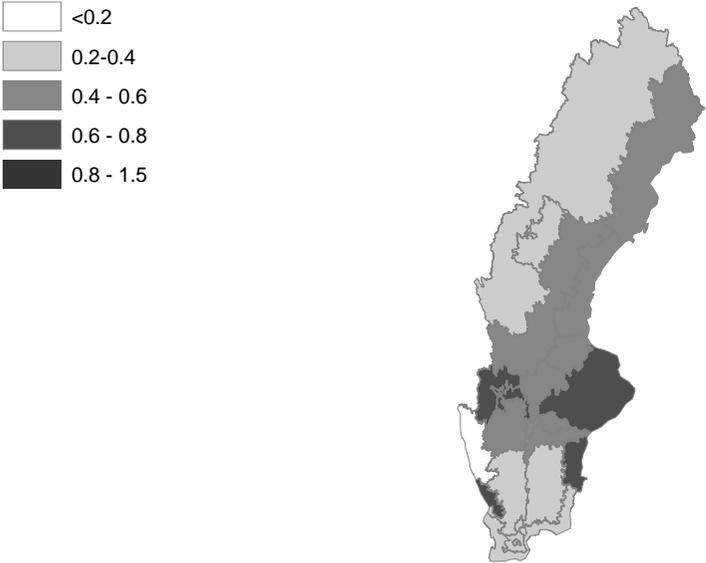


Figure 1. Simulated losses of tot-P (kg ha^{-1}) from the 22 climate-production regions in Sweden.

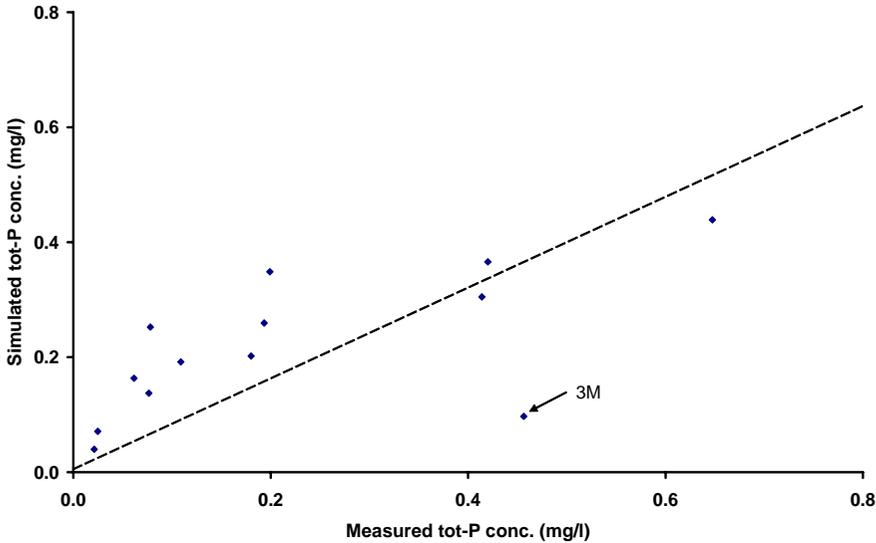


Figure 2. Correlation between measured and simulated tot-P.

The correlation between simulations and measurements was quite poor with an r^2 -value of 0.44. However, if the field denoted 3M was excluded, r^2 improved to 0.75 (Fig. 2.). This field has received large amounts of manure and is not representative for the calculated P-loss coefficients representing average management practices for each region.

Conclusions and recommendations

To conclude, ICECREAMDB is a comprehensive and complex dynamic simulation model and yet relatively easy to use at this scale. The relatively good match between measured and simulated concentrations indicates that the model can be a useful tool for analysis of P losses at regional and national scales. To improve the calculations, further testing and improvement of the parameterization is desirable. However this is partly constrained by the lack of high-quality data sets.

References

- Larsson, M.H., Persson, K., Ulén, B., Lindsjö, A. & Jarvis, N.J., 2007. A dual porosity model to quantify phosphorus losses from macroporous soils. Accepted for publication in *Ecol. Model.*
- Posch, M. & Rekolainen, S., 1993. Erosivity factor in the Universal Soil Loss Equation estimated from Finnish rainfall data. *Agric. Sci. Finland* 2, 271-279.
- Rekolainen, S. & Posch, M., 1993. Adapting the CREAMS model for Finnish conditions. *Nordic Hydrol.* 24, 309- 322.

Parameter variability affecting simulated field scale phosphorus losses

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Introduction

The ICECREAM model has been used in field scale studies to detect variables affecting phosphorus (P) processes and losses at this scale in Finland. Singular field scale results have been extrapolated to describe effects of changed management practices on agricultural land in a catchment. This approach is based on typical soil-crop-slope combinations. Geological soil maps, agricultural statistics and digital elevation data can be utilised to derive information on these factors but only on a very general level, i.e. we might know that the most frequent soil is heavy clay. The most important question for model use remains: how to parameterise this soil. Since not one correct parameter set exists, sensitivity studies are needed to find out the range of variability of key parameters and their effect on simulated output variables. The ICECREAM model was run for a range of soil physical parameters. The output variables studied were hydrological variables (surface runoff, root zone percolation and evapotranspiration) as well as P losses: sediment bound P and soluble P in surface runoff, soluble P in root zone percolation as well as P taken up by the crop.

Material and methods

A comprehensive study on physical properties of Finnish agricultural soils (KUTI, Puustinen *et al.*, 1994) was carried out in the early 1990s. In this study fractions of clay, silt and sand were analysed from the sampled fields (n=1065) and statistically analysed within the soil type groups utilised in Finland. Thus, e.g., the clay content in soil samples in the KUTI database classified as clay varies between 65% and 92%. This variation, that is largest for the fine soil types and smaller for sandy soils, was fed into a Raws & Brakensiek pedotransfer function (ptf, Schäfer, 1999) to calculate soil moisture contents at wilting point and field capacity, as well as the saturated hydraulic conductivity (Ks). In a GLEAMS/CREAMS based model like ICECREAM (Tattari *et al.*, 2001; Yli-Halla *et al.*, 2005) the variability in soil physical properties additionally affects empirically derived parameters like the K factor in the USLE equation and the curve number CN2 used to calculate surface runoff. In the second step expert judgement was utilised to derive typical soil profile parameterisations with differentiated plough layer and subsoil from these homogeneous soil profiles. This was based on minimum, mean and maximum clay content for six soil types. The simulations were performed for a standard spring barley cultivation practice and 16-

year climate data from Jokioinen, south-west Finland. The field slope was 3% which is a typical field slope in several sub-catchments of the studied Aurajoki river basin.

Results

The results show that the variability in the output differs with the output variable studied and the soil type. One reason is the variability in input parameters derived from soil texture like the difference in soil moisture content at field capacity and at wilting point that is larger for the clay (AS) than for the sandy (KHt) topsoils (Table 1). A relative change was calculated for the maximum clay content (max) and the minimum clay content (min) with respect to the mean for all soil types.

Table 1. Difference in soil moisture content at field capacity and wilting point for two soil types and both parameterisation methods, min refers to minimum and max to the maximum clay content within each soil type.

Soil type	Soil type	Source	Difference $\Theta_{fc}-\Theta_{wp}$ [%]		
			min	mean	max
Finnish	American	KUTI + ptf	17.4	17.3	12.7
		KUTI + expert judgement	23.1	14.5	10.5
AS	Clay	KUTI + ptf	13.8	13.5	15.2
		KUTI + expert judgement	22.0	17.2	15.4
KHt	Sand	KUTI + ptf			
		KUTI + expert judgement			

The smallest relative effect was found for crop uptake of P, where also the effect of soil type was found to be negligible, varying between 0 and 11% for both methods. The largest relative changes were found for soluble P in root zone percolation for the sandy clay soil (HtS) for the KUTI+ptf method (65% and 61% for min-mean and max-mean, respectively) and for particulate P in surface runoff for the silt loam soil (KHs) (44% and 39% for min-mean and max-mean, respectively) using the KUTI + expert judgement based profiles. The variability in the sum of P in runoff seems to be large, especially for the loam and sandy loam soils (Table 2). For KHs the P loss difference in runoff is 0.27 kg ha⁻¹ yr⁻¹ utilising the KUTI + ptf parameterisation approach. For the KUTI + expert judgement parameterisation approach the P loss difference in runoff is largest for sandy loam (Hht): 0.91 kg ha⁻¹ yr⁻¹.

Conclusions and outlook

In this study two sources of variability were detected. The principle variability has its origin in the multitude of texture combinations that make one particular soil type. In the theoretical first parameterisation step the largest differences in fraction combinations were sought for and the pF and Ks values were systematically derived from these using a pedotransfer function. This approach leads, however, to soil profiles that do not exist as such on Finnish agricultural land. Thus, in the second step expert judgement was introduced to adjust the texture fraction combinations and to link measured pF and Ks values to each of the profiles. The second, even larger

variability thus arises from the method to derive other parameters than the measured texture fractions. On a field slope of 3% choices affecting particulate P transport in surface runoff are particularly important. This study highlights the effect of parameter variability on simulated P transport, in the second step based on “real soil profiles within different soil groups” which separates the approach from an ordinary sensitivity analysis that allows also unrealistic parameter combinations. In the next step we need to analyse how many of the agricultural soils in the Aurajoki river basin fall into the soil classes where variability is especially high, since it seems that at catchment scale this variability has to be taken into account when a model like ICECREAM is used for management action assessment.

Table. 2. The simulated sum of P in runoff (dissolved and particulate P in surface runoff and dissolved P in root zone percolation) for six soil types and both parameterisation methods, min refers to minimum and max to the maximum clay content within each soil type.

Soil type Finnish	Soil type American	Sum of P in runoff [$\text{kg ha}^{-1} \text{yr}^{-1}$]					
		KUTI + ptf			KUTI + expert judgement		
		min	mean	max	min	mean	max
AS	Clay	0.84	0.76	0.73	0.47	0.44	0.41
HsS	Silty clay	0.94	0.87	0.99	0.83	0.56	0.55
HeS	Clay loam	0.78	0.77	0.96	0.79	0.61	0.64
KHs	Silt loam	0.64	0.84	0.91	0.81	1.18	1.39
HHt	Sandy loam	0.42	0.41	0.54	0.60	1.09	1.51
KHt	Sand	0.31	0.34	0.33	0.42	0.44	0.66

Acknowledgements

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References

- Puustinen, M., Merilä, E., Palko, J., & Seuna, P., 1994. Kuivatustila, viljelykäytäntö ja vesistökuormitukseen vaikuttavat ominaisuudet Suomen pelloilla. National Board of Waters and Environment, Research Report A 198. (Summary in English: Drainage level, cultivation practices and factors affecting load on waterways in Finnish farmland).
- Schäfer, D., 1999. Bodenhydraulische Eigenschaften eines Kleineinzugsgebietes - Vergleich und Bewertung unterschiedlicher Verfahren. PhD Thesis, Dissertationsreihe am Institut für Hydromechanik der Universität Karlsruhe, Heft 1999/1.
- Tattari, S., Bärlund, I., Rekolainen, S., Posch, M., Siimes, K., Tuhkanen, H.-R. & Yli-Halla, M., 2001. Modeling sediment yield and phosphorus transport in Finnish clayey soils. Transactions of the ASAE 44(2), 297-307.
- Yli-Halla, M., Tattari, S., Bärlund, I., Tuhkanen, H.-R., Posch, M., Siimes, K. & Rekolainen, S., 2005. Simulating processes of soil phosphorus in geologically young acidic soils in Finland. Transactions of the ASAE 48(1), 101-108.

Elicitation of expert opinion regarding the primary sources of uncertainty associated with predicting the risk to surface water bodies from phosphorus

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Introduction

The initial stage of the UK Environment Agency’s River Basin Characterisation for the Water Framework Directive (WFD) highlighted a high degree of uncertainty with the current phosphorus (P) risk assessments (e.g Carvalho et al., 2003). This project aimed to identify the relative magnitude of uncertainties associated with the principal factors determining the risk to surface water bodies from phosphorus. Identification of uncertainties was achieved by elicitation of expert opinion, in both quantitative and qualitative form. This aided the identification of academic agreement/disagreement regarding conceptual models of the risk from both diffuse and point source P and subsequently provided consensus regarding an appropriate conceptual model for WFD risk assessment. Additionally, expert opinion was sought on the consequences of uncertainty on policies for phosphorus management, the Environment Agency’s WFD monitoring strategy and priorities for future P research.

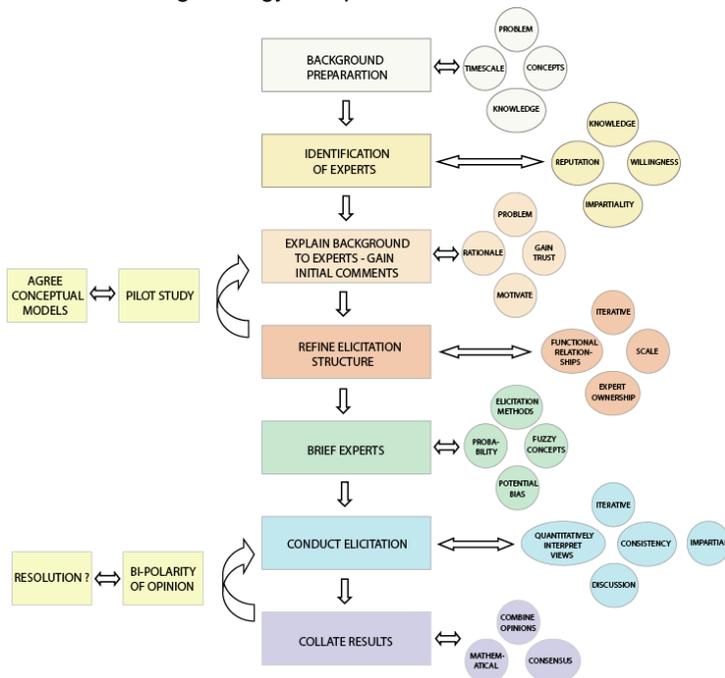


Figure 1. Structure of the Elicitation Procedure.

Structure of elicitation

The elicitation centred around a conceptual model for the area of interest (See Figure 1 for a schematic layout). The conceptual models were developed by the chosen experts in a flexible and iterative manner and played a crucial role in the elicitation procedure. Experts chose model components which they believe to be dominant at the spatial and temporal scale(s) of interest. The chosen components were then the focus for questioning during the structured interview stage. There were number of core questions relating to the relative importance of each chosen model component, the relative level of associated uncertainty and the principal sources of this uncertainty. Responses to these core questions were obtained in a quantitative (relative rather than absolute) manner as fuzzy distributions of their opinion. Fuzzy distributions were collected to prevent the loss of information when point estimates are elicited. Figure 2 shows an example fuzzy distributions of model component importance where importance in some cases was specific to a given typology (typologies were predefined by the experts – e.g. groundwater-dominated catchments or surface water-dominated catchments). Qualitative questions were also elicited as was other qualitative information from conversations during the entire interview. The elicitation structure was based upon many concepts including: Meyer et al. (2002); von Krauss et al. (2004); Funtowicz & Ravetz,(1990) & NUSAP, (2006).

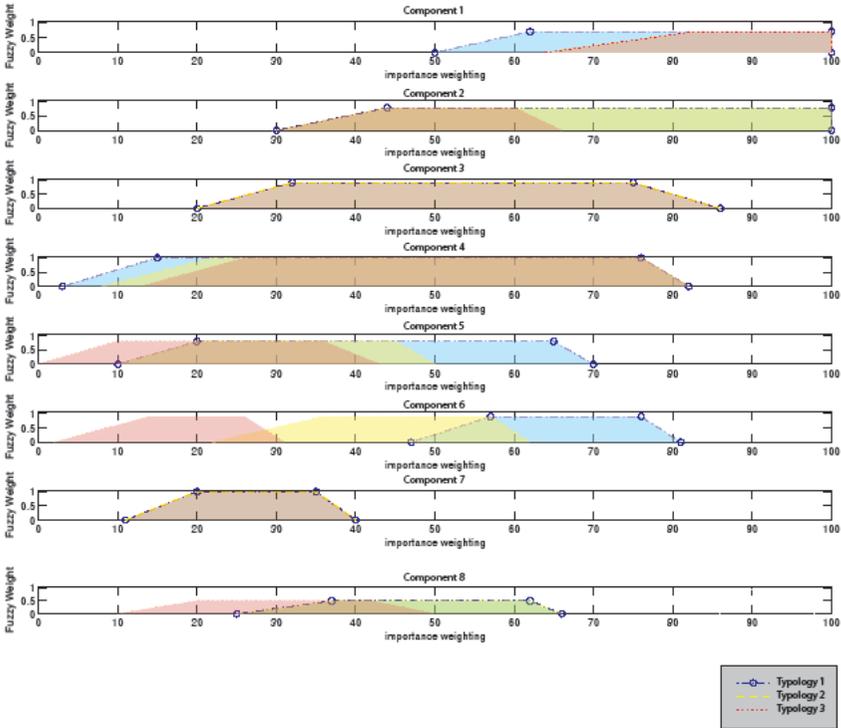


Figure 2. An example of fuzzy distributions of expert opinion collected for 8 components of a conceptual model (note the differing results for typologies).

Discussion and conclusion

The example given in Figure 2 shows opinion for one expert from one sub-area of the project which covers conceptual models for: agronomic practices, catchment processes, in-stream processes, hyporheic zone processes, river ecology, lake ecology and point sources of P. The full set of data, however, covers opinion from approximately 3 experts per sub-area and results highlight areas of consensus and disagreement. Areas of disagreement were dealt with in a number of ways, including the mathematical combination of fuzzy distributions to form a composite distribution (i.e. under the rationale that disagreement between experts is a valid result) or iteration between experts to try and achieve consensus and to determine that they were commenting upon the same phenomenon. During the presentation of this work multiple examples of expert opinion will be discussed in particular relation to the combination of fuzzy measures for importance and uncertainty for given components: i.e. those areas which will impose a particularly strong control on the uncertainty associated with WFD risk assessments.

References

- Carvalho et al., 2003. Risk Assessment Methodology for Determining Nutrient Impacts in Surface Freshwater Bodies (NUPHAR PROJECT): EA Science Report SC020029/SR.
- Meyer et al., 2002. Guidelines for Eliciting Expert Judgment as Probabilities or Fuzzy Logic: in Ross, Booker & Parkinson (Eds.).
- von Krauss et al., 2004. Elicitation of Expert Judgments of Uncertainty in the Risk Assessment of Herbicide-Tolerant Oilseed Crops: Risk Analysis 24(6), 1515.
- Funtowicz & Ravetz, 1990. Uncertainty and Quality in Science for Policy. Kluwer Academic Publishers.
- NUSAP, 2006. <http://www.nusap.net/>.

Phosphorus fate and transport modelling in a catchment of Western Greece and identification of critical source areas

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Introduction

In accordance with the European Water Framework Directive, considerable efforts are needed to achieve the required improvements in the ecological status of surface waters across Greece, primarily focusing on those areas likely to present the greatest risk on catchment scale. The eutrophication of the aquatic environment, mainly attributed to high phosphorus (P) concentrations, is acknowledged as a major environmental threat, especially in countries where there is a great concern about the diffuse P losses associated with particulates involved in soil erosion (Edwards and Withers, 1998). Towards this end, the GIS-based SWAT (Soil and Water Assessment Tool) model (Arnold *et al.*, 1998) was implemented in study areas across Europe (Panagopoulos *et al.*, 2006), in order to facilitate the understanding of P fate and transport to surface waters. The objectives of the study were to implement the SWAT model for the Greek Kalamas river basin in order to estimate the contribution of different sources in P losses to surface waters, to clarify the main pathways by which these losses were caused as well as to produce maps for highlighting areas that potentially pose a high risk to surface waters due to diffuse phosphorus losses.

Model implementation

The SWAT model has the ability to predict the impact of land management practices on water, sediment and agricultural chemical yields in large complex watersheds. The model incorporates the complete phosphorus natural cycle (Neitsch *et al.*, 2001). Kalamas river basin is located in Western Greece covering an area of almost 1800 km² and flows to the Ionian Sea. The mean annual precipitation depth approaches 1500 mm that are non-uniformly distributed over time, appearing as intense rainfall episodes during winter months. Subsequently, significant runoff, sediment and nutrient transport occur. Forest, arable land and pastureland are the predominant land-cover types in the catchment. Arable land (510 km²), which mainly includes corn and wheat cultivations is partly irrigated and fertilized. Grazing constitutes the main P pressure in pastureland (210 km²). Two main groups of soil types in terms of permeability are encountered in the catchment. Karstic limestones and alluvial deposits, which occupy approximately 70% of the total area, have significant infiltration capabilities and contribute with baseflow to the main channel preserving river flow throughout the year, while flysch deposits are almost impermeable.

As for P loadings, 293 tonnes of TP were deposited directly to the river network on an annual basis from point sources consisting of six major urban and farm waste water treatment installations in the catchment. Additionally, 16.2 kg of TP per hectare of total agricultural land (arable + pasture) were deposited annually with high spatial variability, as some fields received more than 40 kg of TP/ha and others were not pressured at all. The model ran for the period 1985-2004 on a monthly basis and was calibrated in several nested basins and at the total outlet for flows and TP yields. Mean monthly TP river concentrations varied greatly between 0.02 and 0.35 mg/l. However in some periods much higher concentrations of TP (up to 1.12 mg/l) had been measured, but the mean annual TP concentrations never exceeded 0.22 mg/l.

Results

SWAT estimated different sources contributing to TP loss in surface waters (Table 1).

Table 1. TP mean annual budget of the Kalamas river (1985-2004).

Catchment Area: 1809 km ² , Agricultural land area (arable + pasture): 720 km ²			
		tn P/a	flow (m ³ /s)
Inflows	Point sources loads	293	10
	Loss from woodland and other non-agricultural land	16	36
	Loss from agricultural land (arable + pasture)	136	23
	Total Inflows	445	
	- <i>Surface runoff (agricultural land)</i>	<i>101</i>	<i>3,1</i>
Reactions	Retention (surface waters: rivers, lakes etc)	53	
Outflows	Total outlet losses = Total Inflows - Retention	392	71

Point sources (urban + livestock farms) and fertilizer and manure deposition on agricultural land were responsible for the annual 445 t of TP losses to the river (gross loads). Diffuse sources caused the 30% of TP losses (136+16 t/yr). The mean annual flow of 71 m³/s at the watershed outlet caused the annual transport of 392 t TP. The model calculated these losses by subtracting the annual P retention (53 t/yr) in rivers from gross loads. In Table 1 the significant role of surface runoff processes in diffuse P losses is also demonstrated. As it is indicated, 101 t P out of 136 that are released in total from agricultural activities were transported to the river by surface runoff. Actually, P consists of organic P transported with sediments, mineral P sorbed to sediments and soluble mineral forms of P transported directly with surface runoff. As phosphorus is a stable element that is mainly attached to sediments and due to the closely-related surface runoff and sediments transport mechanisms, it can be concluded that surface runoff is the pathway that directly or indirectly causes the transport of all the aforementioned P types to the river. Finally, Figure 1 represents a map with the catchment classification into areas of high to low sensitivity in P losses. According to the results, the mean annual diffuse P losses to surface waters vary between 0.01 and 2.05 kg P/ha with the highest values representing fields with inten-

sive agriculture while the lowest refer to forested areas. P loss from agricultural land was estimated at 1.8 kg/ha/yr, much more than that from bare areas (0.15 kg/ha/yr). As a mean of the total area, TP losses from diffuse sources were 0.83 kg per ha of the catchment.

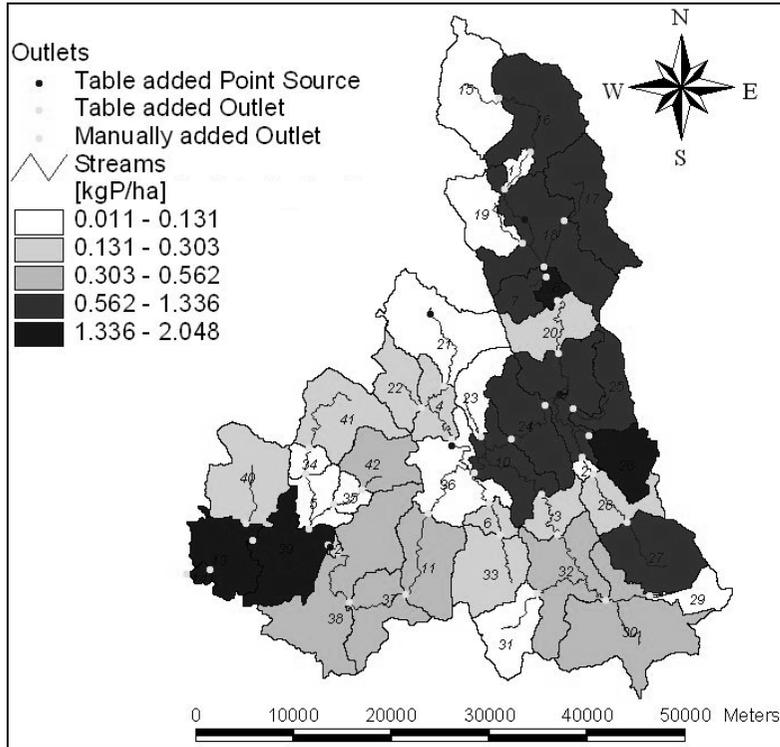


Figure 1. Identification of critical source areas of P losses to surface waters.

Conclusions

The modeling revealed that P from livestock and urban wastes must be further eliminated in treatment plants, as point sources constitute the main source of P in the Kalamas catchment. Additionally, P losses from diffuse sources were significant in areas with intensive agriculture and were mainly transported to the river by surface runoff and sediments. This is strongly believed to be due to the particular meteorological conditions in Greece with rainfall concentrated in winter months, sometimes in storms that subsequently cause severe floods and soil erosion. As a consequence, even in the Kalamas river basin which is characterized by well-developed soil-karstic systems that can smooth the annual river flow, soil loss and transport with surface runoff constitutes the main pathway for the particulate and dissolved emissions of P to surface waters. Thus, mitigation of water pollution in Greece must be specified not only for agricultural practices but also with erosion restriction measures.

References

- Arnold, J.G., Srinivasan, R., Muttiah, R.S. & Williams, J.R., 1998. Large area hydrologic modelling and assessment part I: model development. *J Ame Wat Res Ass.* 34(1), 73-89.
- Edwards, A.C. & Withers, P.J.A., 1998. Soil phosphorus management and water quality: a UK perspective. *Soil Use and Manag.* 14, 124-130.
- Neitsch, J.G., Arnold, J.G., Kiniry, J.R. & Williams, J.R., 2001. Soil and Water Assessment Tool, Theoretical Documentation - Version 2000. Blackland Research Center – Texas.
- Panagopoulos, I., Mimikou, M. & Kapetanaki, M., 2006. Estimation of nutrient losses to surface water and groundwater through the implementation of the SWAT model for Norwegian soils. *J of Soils and Sed* (submitted for publication).

Spatial predictions of P losses from soil and manure and monitoring data in a small agricultural catchment point to soil P as the main source

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Introduction

Avoidance of P surpluses on critical source areas and minimizing incidental P losses by a proper management of manure are important measures to reduce diffuse P losses to surface waters. While the spatial distribution of the critical source areas for soil-derived P losses may be expected to be fairly stable in time, the contributing areas for incidental losses are probably more variable. This is not only due to the intermittent character of this P source but also due to the temporary effects of manure increasing surface runoff and preferential flow after manure applications to grassland soils (Burkhardt et al., 2005, Stamm et al., 2002). Hence, a quantitative understanding regarding the importance of soil-derived versus incidental P losses is important for an appropriate spatial focus of mitigation measures to reduce P losses.

This paper presents results from monitoring and modelling work that aimed at the identification of critical source areas and at the assessment of the relative contribution of incidental losses. The work focused on the Lake Sempach region (Switzerland), which is characterized by intensive animal production.

Study area

The study region was the watershed of Lake Sempach in the central part of the Swiss Plateau. Grassland is the dominant land use. Due to the intensive animal production the region experienced severe eutrophication problems caused by event-driven P losses into the small creeks (Lazzarotto et al., 2005). For modelling the hydrology, we investigated four small subcatchments of Lake Sempach (Lippenrütibach (3.3 km²), Rotbach (6.2 km²), Meienbach (1.2 km²), Greuelbach (2.6 km²)). These subcatchments markedly differ in their soil properties: poorly drained soils cover 16–40% of the respective subcatchments. Average annual precipitation amounts to about 1150 mm yr⁻¹ and highest monthly rainfall occurs normally in May and June. In subcatchments dominated by poorly drained soils, roughly 30% of the agricultural fields are artificially drained, mostly with tiles. P losses were modelled for the Lippenrütibach area (Lazzarotto et al., 2005) where good input and monitoring data were available for P including data on manure applications.

Modelling

For delineating the critical source areas, we developed a parsimonious semi-distributed hydrological model (Lazzarotto et al., 2006). It distinguishes two soil types (well- and poorly-drained according to the conventional soil map) and assumes that areas of the same soil type and the same topindex (Kirkby, 1975) have the same hydrological response in all four subcatchments. It is further assumed that both soil types produce a fast and a slow discharge component. The volume of fast discharge was modelled depending on rain intensity and on soil moisture. The model parameters were obtained by simultaneously calibrating the model to all four subcatchments. Thereby, we retained all parameter combinations yielding a Nash-Sutcliffe criteria > 0.6 .

Losses of P were simulated to originate from three sources: background losses from the subsoil with low P concentrations, soil-derived P losses with fast flow and incidental P losses with the fast flow component. P concentrations in the fast flow component were derived from data on soil contents of all fields and plot experiments on the relationship between soil P content and DRP concentrations in runoff (Schärer et al., accepted, Withers et al., 2003)

Field data, monitoring

The data on soil P status, manure application, and P balances in the Lippenrütibach subcatchment were obtained in the framework of an evaluation project for agro-ecological measures (Prasuhn and Lazzarotto, 2005). The P losses were measured at high temporal resolution at the outlet of the subcatchment using a flow-injection analyzer (Lazzarotto et al., 2005).

Results

The model results show a strong dominance of P losses originating from the P-enriched topsoil compared to the incidental losses due to manure applications. The model simulations attribute 75 to 95% of the total DRP load to soil-derived P for an entire growing season (1999). For single events, the values may be slightly lower and incidental losses may account for up to 30% of the DRP loads. The high percentage of P losses originating from the soil can be explained by the exceptionally high P contents in many of the soils in the subcatchment. Due to a long history of over-fertilization, about 50% of the agricultural area have water-extractable P values (CO₂-saturated water) $> 16 \text{ mg P kg}^{-1}$ soil reaching maximum values of 94 mg P kg^{-1} soil (topsoil 4 cm depth).

The model results are supported by the temporal development of the P balance of the subcatchment and the P losses. Due to implemented measures including the use of low-P feedstuff for pigs, the P balance changed from a P surplus in the 1990s to a deficit in 2003 (Table 1, (Prasuhn and Lazzarotto, 2005)). Under the hypothesis that

the P losses originated mainly from freshly applied manure (incidental P losses), the reduced P input to the soils should be directly reflected by a decline of P losses of about 25%. This was obviously not the case as can be seen from Table 1.

Table 1. P balance of and P losses from the subcatchment Lippenrütibach. P input and balance are normalized to the agricultural area.

Year	P input to soils (kg P ha ⁻¹)	P balance (kg P ha ⁻¹)	P losses (normalized to discharge) (kg DRP 10 ⁻⁶ m ⁻³)
1992	49	17	161
2003	35	-2	176

Conclusions

The model as well as the field-data indicate a dominant role of the soil as the main reservoir of P for its mobilization into water bodies. Hence, reduced P input into the soils affects losses to streams only at the time-scale of years or even decades. Mitigation options to enhance this process like ploughing seem to be of limited success (Schärer et al., accepted). In order to reduce P losses from soils enriched strongly with P, it is important to reduce the P input immediately. However, farmers as well as legislators must take into account that positive effects will take a considerable time to become evident. Accordingly, such mitigation programmes have to be planned for an appropriate time-span.

References

- Burkhardt, M., Stamm, C., Waul, C., Singer, H. & Müller, S., 2005. Surface runoff and transport of sulfonamide antibiotics and tracers on manured grassland. *J. Environ. Qual.* 34, 1363-1371.
- Kirkby, M., 1975. Hydrograph modelling strategies. IN Peel, R., Chisholm, M. & Haggett, P. (Eds.) *Processes in Physical and Human Geography*. London, Heinemann.
- Lazarotto, P., Prasuhn, V., Butscher, E., Crespi, C., Flühler, H. & Stamm, C., 2005. Phosphorus export dynamics from two Swiss grassland catchments. *J. Hydrol.* 304, 139-150.
- Lazarotto, P., Stamm, C., Prasuhn, V. & Flühler, H., 2006. A parsimonious soil-type based rainfall-runoff model simultaneously tested in four small agricultural catchments. *J. Hydrol.* 321, 21-38.
- Prasuhn, V. & Lazarotto, P., 2005. Abschwemmung von Phosphor aus Grasland im Einzugsgebiet des Sempachersees. *Schriftenreihe FAL* 57, 95-107.
- Schärer, M., Stamm, C., Vollmer, T., Frossard, E., Oberson, A., Flühler, H. & Sinaj, S., (accepted) Reducing phosphorus losses from over-fertilized grassland soils proves difficult in the short term. *Soil Use and Management*.
- Stamm, C., Sermet, R., Leuenberger, J., Wunderli, H., Wydler, H., Flühler, H. & Gehre, M., 2002. Multiple tracing of fast transport in a drained grassland soil. *Geoderma* 109, 245-268.

Withers, P.J.A., Ulén, B., Stamm, C. & Bechmann, M., 2003. Incidental phosphorus losses - are they significant and can they be predicted? *J. Plant Nutr. Soil Sci.* 166, 459-468.

Evaluation of a P Index for NE Germany for a large cattle production operation

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Introduction

In NE Germany (Mecklenburg-Vorpommern, MV), diffuse P losses from agricultural fields are comparatively low for most regions. However, there are a few hot spots with elevated P concentrations and large inputs of P as manure, mostly in the vicinity of cattle feeding operations and dairy farms. For the surroundings of such hot spots, a tool for the assessment of the risk of diffuse P loss would be useful. In the USA and several European countries, the "Phosphorus Index" (PI) as a semi-quantitative risk assessment tool for diffuse P losses from agricultural fields has been used for about 15 years. We adapted this approach to take into account the specific conditions of P loss in NE Germany. Here, we present this PI for NE Germany ("PI-MV") along with results from its application to the area of a large cattle feeding operation.

PI for NE Germany (PI-MV)

Diffuse P loss in MV is characterized by several special factors:

1. Wind erosion accounts for 1/3 of total eroded sediment mass in MV. Wind erosion risk has been mapped area-wide for MV (LUNG 2002).
2. Tile drainage: In MV, 34 % of the agricultural area is artificially drained.
3. Peat bogs cover 12.6 % of the area of MV. Peat soils have a low P sorption capacity which leads to a great risk of P loss via soil leaching (Daly *et al.*, 2001). Moreover, many bog areas have been re-wetted during the last years. This leads to reducing conditions resulting in enhanced chemical mobility of P (Rupp *et al.*, 2004; Tiemeyer *et al.*, 2005).
4. Besides peat soils, a large proportion of soils in MV are hydric/aquic with shallow groundwater tables.

The structure and components of the PI-MV are based largely on the P indices such as those of Pennsylvania, Maryland and Denmark. In addition, a factor for P loss by wind erosion is introduced. The PI-MV value is calculated as:

$$\text{PI-MV} = (\text{STP factor} + \text{Fertilizer factor} + \text{Manure factor}) \\ \times \left(\frac{[(\text{Water erosion} + \text{Wind erosion} + \text{Distance}) \times \text{Buffer factor} + \text{Drainage} + \text{Leaching factor} + \text{GW factor}]}{22} \right)$$

The first term on the right side represents the source term whereas the second is the transport term. The factors of this equation are explained in Table 1.

Table 1. Factors of the PI-MV.

Factor	Explanation
STP factor	Data on STP (CAL or double-lactate extraction method) are largely available as nutrient content classes, A – E, corresponding to STP values (ppm P ₂ O ₅ in air-dry soil): A: < 70; B: 70 – 120; C: 130 – 180; D: 190 – 270; E: > 270. For calculating the PI-MV, the STP classes are assigned the values A = 10, B = 20, C = 30, D = 40, E = 50
Fertilizer factor	application rate (kg P ₂ O ₅ /ha) x method (0.2: Placed or injected 5 cm or more deep; 0.4: Incorporated in less than 1 week; 0.6: Incorporated after 1 week or not incorporated, April to October; 0.8: Incorporated after 1 week or not incorporated, November to March; 1.0: Surface applied during frozen or snow-covered conditions)
Manure factor	application rate (kg P ₂ O ₅ /ha) x method x availability (values ranging between 0.2 and 1)
Water erosion	Risk ratings are categorized into 5 classes (none => 0; low => 2; medium => 3; high => 4; very high => 5)
Wind erosion	Risk ratings are categorized into 5 classes, similar as water erosion
Distance	Distance to the nearest surface water: > 50 m => 0; < 50 m => 5
Buffer factor	Width of riparian buffer: < 2 m => 1.1; 2 – 6 m => 1; 6 – 20 m => 0.7; > 20 m => 0.5
Drainage	Subsurface drainage: no => 0; yes => 2
Leaching factor	Soil type: Loam=> 0; Sand/Loam => 1; Sand => 2; Peat/clay=> 3
GW (groundwater) factor	Deep GW table => 0; medium GW table => 1; shallow GW table => 2

The PI-MV has been tested at the area surrounding a huge cattle feeding operation in the eastern part of MV. In this area, P applications in the form of manure exceeded 200 kg P₂O₅ / ha for some fields (in 2006).

Results

Calculated values of the PI-MV are presented in Figure 1. Only a very small fraction of the area exhibits PI values above 85, which are considered to represent a high risk of P export (Figure 1a). Correlation between PI values and STP classes reveals only a weak correlation ($R^2 = 0.1581$; Figure 1b), whereas the correlation between PI values and applied manure-P is strong ($R^2 = 0.6872$; Figure 1c). On the other hand, the correlation between PI values and the transport term is low ($R^2 = 0.2776$; Figure 1d).

Discussion and conclusions

The application of the PI-MV revealed only a small area fraction with high P loss risk. These are fields that received P₂O₅ applications as manure in excess of 100 kg / ha in 2006. Whereas the manure factor has in this case a large influence on PI values, the influence of transport factors is somewhat lower, but significant ($p < 0.05$). For

this example, the area-averaged risk of P loss (at the watershed scale) is low, in line with measured P concentrations at the watershed outlet, which are in general below $50 \mu\text{g TP} / \text{l}$ (data not shown here). P loss risk for the few fields with excessive PI values could be easily diminished by a more homogeneous distribution of manure applications. The high PI values of these single fields have until now not been compared with P concentrations in the outflow of single fields. Such measurements are focused on in our current and future work.

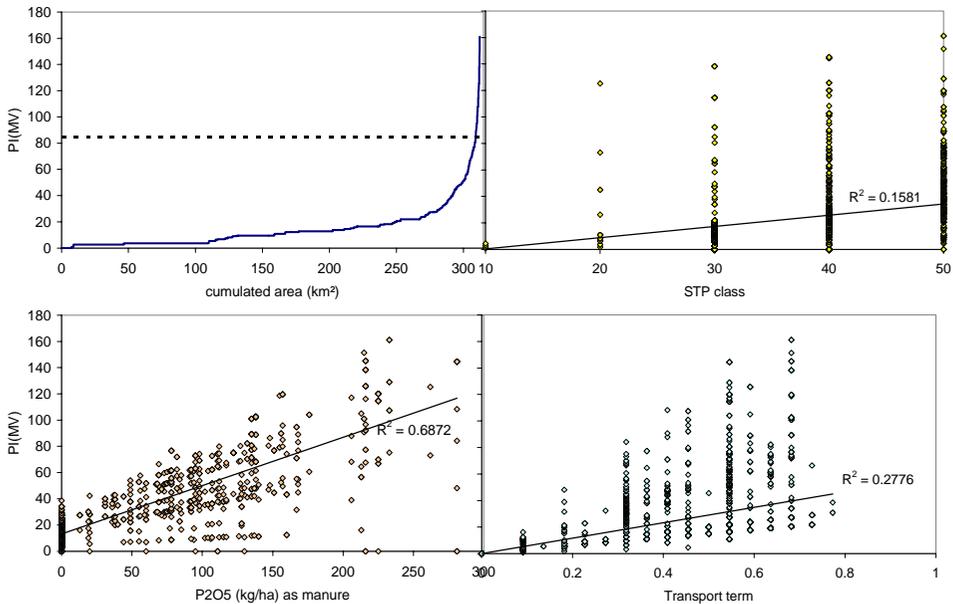


Figure 1. PI values for the area around a cattle feeding operation in MV for 2006. Top left (a): Cumulated area with PI values lower than PI value on vertical axis; top right (b): PI values vs. STP class; bottom left (c): PI values vs. applied P₂O₅ as manure; bottom right (d): PI values vs. transport term.

References

- Daly, K., Jeffrey, D. & Tunney, H., 2001. The effect of soil type on phosphorus sorption capacity and desorption dynamics in Irish grassland soils. *Soil Use Manage.* 17, 12-20.
- Landesamt für Umwelt, Naturschutz und Geologie Mecklenburg-Vorpommern (LUNG) (eds.), 2002. *Bodenerosion*. 2. edition. Beiträge zum Bodenschutz in Mecklenburg-Vorpommern, 83 pp.
- Rupp, H., Meissner, R. & Leinweber, P., 2004. Effects of extensive land use and re-wetting on diffuse phosphorus pollution in fen areas - Results from a case study in the Drömling catchment, Germany. *J. Plant Nutr. Soil Sci.* 167, 408-416.
- Tiemeyer, B., Lennartz, B., Schlichting, A. & Vegelin, K., 2005. Risk assessment of the phosphorus export from a re-wetted peatland. *Phys. Chem. Earth* 30, 550-560.

A phosphorus index approach for Denmark

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Introduction

Water quality of Danish lakes and estuaries has continued to deteriorate due to excess P loadings (Kronvang et al., 2002). Today diffuse P losses from agricultural areas are the main P source to rivers and lakes. Hence, compliance with the EU Water Framework Directive requires substantial reductions in diffuse P losses. Identifying areas in agricultural landscapes that pose the greatest risk of P loss to surface waters is critical for cost-effective mitigation planning. To this end Lemunyon and Gilbert (1993) originally proposed the P Index in the USA, which integrates both transport and source factors to identify areas vulnerable to P export. The P Index is well suited for screening large areas, since it uses readily available data and effectively ranks fields by their potential for P loss (e.g. Andersen and Kronvang, 2006). A suit of different P Indices has evolved (Sharpley et al, 2003), as the P Index ought to reflect regional characteristics and practices. Here we present the outline of a Danish P Index.

Danish premises

Direct losses of applied mineral fertilizer or animal manure P are considered of less importance in Denmark due to current manure management practices. Such losses occur with only few transport processes and require differentiated treatment of P applications in the P Index. Due to the topography, low-intensity rainfall and nature of Danish soils, soil erosion and surface runoff are not thought dominant processes of P loss in Denmark. Andersen et al. (2005) have, however, shown the importance of P loss through artificial drains. Hence, a Danish P Index must account for the risk of P loss to tile drains both by matric and macropore flow. The complexity of a P Index handling several processes of P loss poses the problem of determining the relative importance of the individual processes, if the risk is to be described by a single score. At $<0.5 \text{ kg P ha}^{-1}$, average diffuse P losses are relatively low in Denmark. A challenge for P Index design is to accurately capture such small loads.

Approach to constructing the Danish P Index

We have deconstructed the traditional P Index. Instead of one single index value per field we calculate sub-indices for individual P loss processes. The risk of P loss is

assessed according to the P transfer continuum (Haygarth et al., 2005). For each P loss process we describe and parameterize the factors: source, mobilisation, and transport. All factor values are multiplied and the result subsequently standardized by dividing with anticipated maximum conditions in Denmark to yield a value between 0 and 100. Although we acknowledge some relation between P Index score and load estimate, due to the complexity of the P transfer continuum the P Index aims solely at ranking fields according to relative risk. The more open structure for a P Index has several advantages: (i) it gives more realistic combinations of source and transport factors than a traditional P Index where the same source factor is used for all transport processes, (ii) it is easy to incorporate new knowledge, (iii) it directly shows the relative risk for each process of P loss compared to the anticipated maximum, increasing user-friendliness and facilitating mitigation planning, (iv) effects of mitigation measures can easily be incorporated into the index by adjusting the factor value, which is affected by the measure. At present the Danish P Index includes parameterized descriptions for soil erosion, surface runoff, and leaching by both matric and macropore flow. Concepts are being developed for incorporating two further processes into the P Index, i.e. P loss from lowland soils (Kjaergaard et al., 2007) and P loss associated with stream bank erosion.

Soil erosion. The risk of P loss is estimated as a function of soil mobilization, soil P status, fertilization and distance to a recipient. The amount of soil eroded is determined using a Danish empirical model for rill erosion (Djurhuus et al., 2007). The extent and location of eroding areas within fields is determined with WaTEM (<http://www.kuleuven.ac.be/geography/frg/index.htm>), a distributed grid-based erosion model. The P load mobilised is approximated as a function of soil test P (STP) and surface applied fertilizer (incl. manure). The risk of eroded P reaching surface water depends on the flow distance between the eroding area and nearest recipient, assuming that the probability for deposition increases with distance.

Surface runoff. The risk of P loss is a function of runoff volume, soil P status, fertilization and distance to recipient. As a proxy for runoff volume we use the product of a topographically derived discharge area at the point where runoff leaves the field and cumulated winter precipitation. The amount of P that is mobilized by runoff depends on STP and surface applied fertilizers. The risk of surface runoff reaching surface waters declines with the flow distance from the edge of the field.

Leaching by matric flow. The risk of P loss to tile drains is a function of precipitation, soil P status, fertilization, and P retention in subsoils. The corresponding source factor accounts for STP and both surface applied and incorporated fertilizer P. Transport is assumed to be proportional to hydrological effective precipitation. The P load depends on the P binding capacity in subsoils, which in turn is determined by the contents of amorphous iron and aluminium (Greve et al., 2007).

Leaching by macropore flow. Macropores may transfer P in dissolved and particulate forms. The risk of P loss to tile drains is a function of precipitation, soil P status, fertilization, macropore connectivity in subsoils, and likelihood of macropore flow. The corresponding source factor accounts for STP and both surface applied and incorporated fertilizer P. Subsoil macropores are assumed to exist at clay contents larger than 10%. Macropores are active when rainfall exceeds the matric hydraulic conductivity. The likelihood of macropore flow has been estimated by a hydrologic model and mapped based on the distribution of precipitation and combinations of hydraulic parameters for the A, B and C horizon (Iversen et al., 2007).

Phosphorus leaching in undrained soils is considered relevant only in a zone of 100 m from a recipient. In the present version of the Danish P Index sub-indices are not combined into a single P Index score. The extensive data necessary for weighting the relative importance of different processes of P loss are not available, yet.

Outlook

The Danish P Index will be implemented in a web-based tool and made available to environmental planners and agricultural advisors in 2008. It draws on national databases and has interactive facilities to cater for local conditions. Administrators, landusers and researchers agree on the purely advisory role of the P Index.

References

- Andersen, H.E., Kronvang, B. & Larsen, S.E., 2005. Development, validation and application of Danish empirical phosphorus models. *J. Hydrol.* 304, 355-365.
- Andersen, H.E. & Kronvang, B., 2006. Modifying and evaluating a P Index for Denmark. *Water, Air and Soil Pollution* 174, 341-353.
- Djurhuus, J., Højsgaard, S., Heckrath, G. & Olsen, P., 2007. An expert system for predicting rill erosion in Denmark. This issue.
- Greve, M.H., Rubæk, G.H., Djurhuus, J. & Heckrath, G., 2007. Mapping phosphorus sorption capacity in soils. This issue.
- Haygarth, P., Condron, L.M., Heathwaite, A.L., Turner, B.L. & Harris, G.P., 2005. The phosphorus transfer continuum: Linking source to impact with an interdisciplinary and multi-scaled approach. *Science of the Total Environment* 344, 5-14.
- Iversen, B.V., Børgesen, C.D., Lægdsmand, M., Greve, M.H. & Heckrath, G., 2007. Mapping the risk of P loss through soil macropores. This issue.
- Kjaergaard, C., Hoffmann, C.C. & Greve, M.H., 2007. Phosphorus forms and phosphorus release as affected by organic lowland geochemistry. This issue.
- Kronvang, B., Grant, R., Laubel, A.R. & Pedersen, M.L., 2002. Quantifying Sediment and Nutrient Pathways within Danish Agricultural Catchments. In: *Agriculture, Hydrology and Water Quality*. Haygarth, P.M. & Jarvis, S.C. (eds.) CABI Publishing, 282-301.
- Lemunyon, J.L. & Gilbert, R.G., 1993. The concept and need for a phosphorus assessment tool. *J. Prod. Agric.* 6, 483-496.

Sharpley, A.N., Weld, J.L., Beegle, D.B., Kleinman, P.J.A., Gburek, W.J., Moore, P.A. & Mullins, G., 2003. Development of phosphorus indices for nutrient management planning strategies in the United States. *J. Soil Water Conserv.* 58(3), 137-152.

Possibilities to reduce diffuse phosphorus load from managed forest areas by buffer zones

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Introduction

Phosphorus (P) losses from boreal forested areas may increase after forestry operations such as timber harvesting, drainage and fertilization, especially from peatland dominated catchments (e.g. Ahtiainen and Huttunen, 1998). To prevent P leaching to recipient waters, it is recommended in Finland to leave unmanaged buffer zone areas (BZAs) along forest brooks, streams and lakes. However, previous studies have indicated considerable variation in the efficiency of BZAs to reduce P leaching (e.g. Kubin *et al.*, 2000; Nieminen *et al.*, 2005; Väänänen *et al.*, 2006). To improve the design, dimensioning and construction of BZAs, information is needed on the factors that control P retention in different types of BZAs. Especially the role of chemical P sorption in soil needs to be examined as, in the regions with distinct winter period, approximately half of the annual load occurs during snow-melt period in spring (Kortelainen and Saukkonen, 1998) when the conditions are unfavourable for biological P retention.

Methods

P retention in wetland buffer zones was studied by conducting a field experiment at six BZAs in south-central Finland. The area of the three smallest BZAs varied from 0.12 to 0.20 ha (0.09-0.23% of the total catchment area) and the three largest BZAs from 1.00 to 1.03 ha (1.12-4.88%). The BZAs were either natural wetlands or restored and rewetted former drainage sites where the surface soil was peat. To study the capacity of the areas to reduce P concentration from the water flowing through the areas, 10 kg of P (as PO₄-P solution) was added to the water flowing into BZAs and the P concentration in inflow and outflow water was followed for 2-4 years. The total P retention was calculated for the three smallest BZAs using measured runoff and P concentrations in outflow water during the five-day adding period. The allocation of the retained P was studied in one of the BZAs by adding radiotracer ³²PO₄-P along with the fertilizer solution and measuring ³²P recovery in vegetation and soil matrix five days after the adding period.

Laboratory analysis was used to determine P desorption-adsorption behaviour of soils typical for buffer zones and operated areas. Soils included humus layer of forested buffer zones and clear-cut areas from three small watersheds, surface and

subsurface peat from five peatland buffer zone areas and three podzol profiles. Maximum desorption, maximum sorption and threshold concentration where net desorption changes to net adsorption were used as reference values to describe P retention behaviour.

Results and discussion

During the adding period, the lower P concentration in outflow than inflow water indicated that all BZAs were able to retain P. Total retention during the adding period in the three smallest BZAs was 66-93% of the added P; however, the overall retention is lower, as outflow of P continued at an elevated level a few months after the adding had ended. P outflow from the two largest BZA remained at the background level throughout the study period indicating 100% retention, whereas elevated outflow P in one of the large areas suggested similar partial retention as was measured from the three smallest BZAs.

The radiotracer study in one BZA showed a total retention of 16% of the added ^{32}P . Total retention was low most likely because the water flow did not slow down sufficiently and penetrate deeply enough to enable a close contact between P in the runoff and the soil matrix. The major part, 92%, of the recovered ^{32}P was in the surface peat layer, suggesting that adsorption by soil was the most important sink for P. The biological accumulation of P, 8% of the total retention, was low, probably because the vegetation biomass was low when the study was carried out in early spring. The short study period (10 days) in our radiotracer study may also explain the low rate of biological accumulation as long-term studies generally indicate significantly higher accumulation rates (Silvan *et al.*, 2004).

The P retention by BZAs appeared to depend on both hydrological conditions during the adding period and soil P retention properties. There was no clear difference in peat P retention properties between the three smallest BZAs, but the total P retained by BZAs in the addition experiment was negatively correlated with the hydraulic load. High hydraulic load increases the formation of flow channels which reduce the effective BZA area and the contact time between P in through-flow water and the processes that retain P (Koskiaho *et al.*, 2003). Peat in the two largest BZAs had higher P retention capacity than peat in the three smallest BZAs and their larger size in proportion to the catchment can have levelled off high hydraulic peaks enabling a closer contact between P in through flow water and the processes that retain P. Both higher P retention capacity in peat and the larger buffer size may explain why the two largest BZAs retained all the added P.

P sorption properties in the humus layer varied considerably. The humus had a noticeable P adsorption capacity in the forested buffer zones in the first autumn after tree harvesting, whereas in the clear-cut sites retention varied from moderate

adsorption to significant desorption. Three-four years after the harvesting the P adsorption capacity had decreased noticeably and the humus layer only desorbed P, except for one forested buffer zone. Therefore, the role of the forest humus layer in P retention in clear-cut areas and buffer zones may be insignificant. P retention properties of the podzol E horizon resembled that of the humus layer with relatively large maximum desorption and low maximum adsorption, whereas the B horizon adsorbed all P within the range of P loading generally occurring in nature. Consequently, if water percolates through the B horizon the risk of P leaching is minor.

P was more efficiently adsorbed in the surface peat layer than in the subsurface peat. Fertilizer P load added to the BZAs significantly decreased maximum adsorption and increased maximum desorption in surface peat. This suggests that peat had a considerable P sorption capacity, but the retention was partly due to the increase in the labile P pool in peat which may be a short-term sink of P.

Conclusions

Correctly designed and dimensioned BZAs are efficient in reducing P load from managed forest areas. The reduction is strongly affected by how much the water flow is slowed down in the buffer area to allow a close contact between P in through-flow water and the processes that retain P. Therefore, to efficiently control the transport of P from forested catchments, small and narrow buffer areas may not function properly and wider areas are needed.

References

- Ahtiainen, M. & Huttunen, P., 1999. Long-term effects of forestry managements on water quality and loading in brooks. *Boreal Environ. Res.* 4, 101-114.
- Kortelainen, P. & Saukkonen, S., 1998. Leaching of nutrients, organic carbon and iron from Finnish forestry land. *Water Air Soil Poll.* 105, 239-250.
- Koskiaho, J., Ekholm, P., Rätty, M., Riihimäki, J. & Puustinen, M., 2003. Retaining agricultural nutrients in constructed wetland – experiences under boreal conditions. *Ecol. Eng.* 20, 89-103.
- Kubin, E., Ylitolonen, A., Välitälo, J. & Eskelinen, J., 2000. Prevention of nutrient leaching from a forest regeneration area using overland flow fields. In: M. Haigh and J. Křeček (eds.). *Environmental Reconstruction in Headwater Areas*. Kluwer Academic Publishers, Dordrecht, Netherlands. Pp. 161-169.
- Nieminen, M., 2003. Effects of clear-cutting and site preparation on water quality from a drained Scots pine mire in southern Finland. *Boreal Env. Res.* 8, 53-59.
- Silvan, N., Vasander, H. & Laine, J., 2004. Vegetation is the main factor in nutrient retention in a constructed wetland buffer. *Plant Soil* 258, 179-187.
- Väänänen, R., Nieminen, M., Vuollekoski, M. & Ilvesniemi, H., 2006. Retention of phosphorus in soil and vegetation of a buffer zone area during snowmelt peak flow in southern Finland. *Water Air Soil Poll.* 177, 103-118.

Effect of hydromorphological interventions on nutrient concentrations in surface waters

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Introduction

In the Netherlands implementation of the Water Framework Directive (WFD) requires definition of a Good Ecological Condition (GEC) as a standard for natural water bodies. This is expressed in biological quality elements as well as general physical and chemical conditions. In (heavily) modified water bodies the effects of irreversible hydro-morphological interventions may be taken into account in the level of standards (Good Ecological Potential, GEP).

This study aims at describing and, if possible, quantifying the effects of hydromorphological interventions on nutrient concentrations in surface waters.

Methods

For this study the separate interventions are aggregated to clusters with more or less the same range of effect. The clusters and their location in the water systems are presented in figure 1.

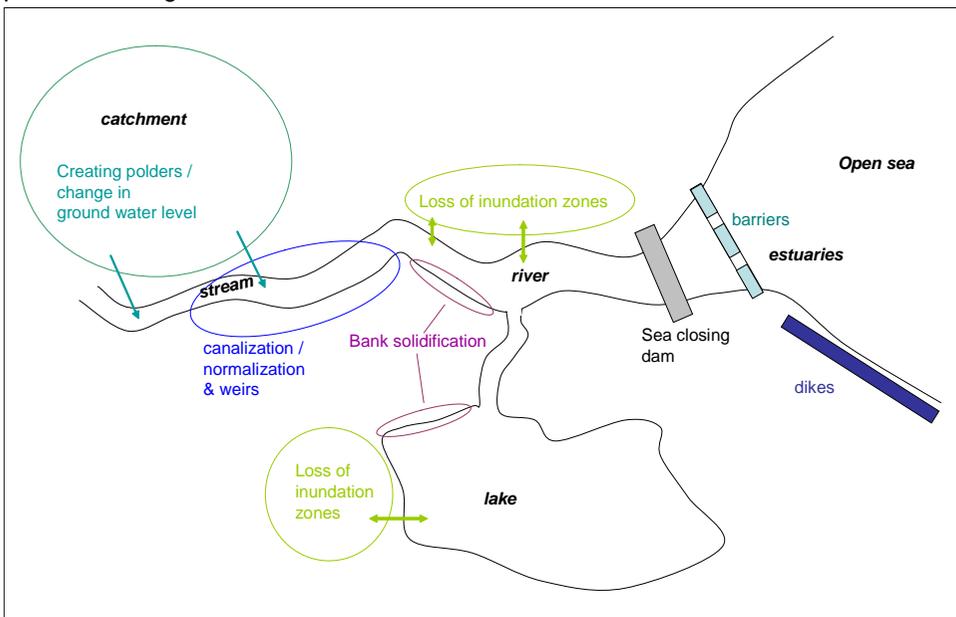


Figure 1. General positioning of the interventions in the surface water systems.

All the clusters are evaluated for their effects on nutrient loads and concentrations, based on existing research (literature and expert judgement).

For all clusters, a fact sheet is set up containing a description of the interventions, existing and lacking knowledge, intervention-effect relations and possible mitigation options. To keep uniformity in the fact sheets, the evaluation of the interventions is divided into three parts:

- effect on diffuse emissions and loads from upstream
- effect on residence time in the surface water system
- effect on turn-over processes

The combination of these effects finally determines the concentrations in the surface waters.

Results

The results of the separate fact sheets are summarized in Table 1. Obviously, effects can be reversed. The final judgement of the relevance and the direction of influence is based on the predominant effect or a range is presented.

Table 1. Summarized effects of the intervention clusters on nutrient concentrations.

Intervention Cluster	Effect on nutrients
canalization / normalization	++
Weirs and sluices	+
Bank and bottom solidification	+
Loss of inundation zones	++
Groundwater level change / polders *	++/--
Dikes	+
Sea closing dams	+
Barriers on the transition of sea and estuaries	- / +

Legend:

-- clear decrease; - moderate decrease o no effect; + moderate increase, ++ clear increase

Conclusions

Most of the hydromorphological interventions lead to higher nutrient concentrations in the surface waters. On one hand, this is the result of increased loads to the surface waters. Moreover, these interventions generally result in decreased residence times, so the effect of turnover processes will diminish and retention decreases.

We conclude that canalization/normalization, loss of inundation zones and decrease in groundwater level have strong effects on nutrient concentrations. The effects of these three clusters should play a role in determining the references and targets for the Maximum and Good Ecological Potential.

The effect of the other clusters is less clear and difficult to quantify.

In a follow up research the major effects will be quantified. In addition, the process of defining MEP/GEP will be explored in two case studies.

Risk and mitigation of P losses following organic manure applications

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Introduction

Agriculture is a significant source of phosphorus (P) loss to water causing eutrophication and ensuing ecological imbalance in aquatic ecosystems. The availability of P for transfer in runoff from agricultural land largely arises either from a build up of P in the soil, or from freshly applied P in manure or fertiliser. Manures contain less water-soluble P than inorganic fertilizers but are a particular concern because repeat applications are made and manure is often left on the surface during winter when the risk of runoff is greatest. Large quantities of solid and liquid manures (ca. 90 million tonnes/annum) are recycled to land each year in the UK. Of this total figure an estimated 47 million tonnes (ca. 50%) is livestock slurry supplying c.210,000 tonnes of nitrogen (N) and c.50,000 tonnes of phosphorus (Williams *et al.*, 2000). Slurries have been found to represent a greater risk of P transfer than solid manures due to the higher proportion of water-soluble P found in slurries (Hodgkinson *et al.*, 2002) and associated hydraulic loading.

Land cropped to maize, and drained clay soils (in particular grassland) pose the greatest risk in terms of P loss after manure application. Maize crops receive regular dressings of manure and due to lack of soil cover over winter are highly susceptible to erosion. Grassland receives surface applications with no opportunity for soil incorporation, which, coupled with the fact that grass occurs predominantly in wetter areas, gives rise to a high risk of transfer. The underdrainage systems normally associated with agricultural production on clay soils encourage rapid connectivity and transfer between the soil surface and receiving water.

Materials and methods

Hydrologically isolated plots were established to investigate P losses in overland flow at a maize site in Devon; overland flow and drainflow at a drained clay grassland site in Herefordshire (ADAS Rosemaund) and in drainflow from an arable site (drained clay soil) in Cambridgeshire (ADAS Boxworth).

Maize site - Five treatments, each replicated 3 times, were applied to 15 plots. A single surface application ($50\text{m}^3 \text{ha}^{-1}$) of dairy slurry was applied to maize stubble, which was either left uncultivated, tine cultivated, or ploughed to 20cm. The same

amount of slurry was also applied to uncultivated stubble, but split into 3 equal monthly dressings. An uncultivated plot receiving no slurry acted as the control.

Grassland site (drained clay soil) - Different applications of cattle slurry were applied to 12 plots on a silty clay loam soil. The study examined the effects of solids loading (0.1 to 6 t ha⁻¹) in the first two years, and the timing of applications (October, November, January, February) in a third year.

Arable site (drained clay soil) - This study compared spring and autumn cultivations. Pig slurry was applied to stubble in autumn or spring and ploughed down, surface applied to cereal stubble in autumn or winter and top-dressed to the growing crop in winter or spring. Pig slurry was also applied to stubble in autumn and incorporated with discs. Treatments were compared against a control receiving no P. Autumn applications were in September, winter applications in December or January, and spring applications in March or May, supplying typically 60-70 kg P ha⁻¹ each year.

Results and discussion

Maize site - Surface application of slurry (50 m³ ha⁻¹, 30 kg P ha⁻¹) to maize stubble produced the greatest P loss (up to 23% of the P applied), with mean flow-weighted concentrations over three years being 3.7 mg L⁻¹ c.f. 1.5 mg L⁻¹ from the control. The soluble P mobilised from the manure accounted for 60% of the total P loss. The most extreme losses occurred when intense rain fell soon after manure application, and initial runoff P concentrations were in proportion to the amount of P applied. In one year, splitting the manure application (3 x 10 kg P ha⁻¹) reduced total P export by 25% compared to a single surface application. In two years, incorporation of the slurry either by ploughing or tine cultivation reduced the amount of overland flow by 50% and P export by 60% compared to surface-applied treatments.

Grassland site (drained clay soil) - Losses of P were up to 3 kg ha⁻¹ and were greatest when cattle slurry was applied to wet soils and least when the soils had a moisture deficit. Loss increased linearly with P application rate, with flow-weighted P concentrations reaching up to 30 mg L⁻¹ in surface runoff and 6 mg L⁻¹ in drainflow. The relative differences in P losses between surface runoff and drainflow were dominated by differences in flow and whether there was a soil moisture deficit when the slurry was applied. The presence of the drains reduced the flow-weighted concentrations in all years, but not always losses of P because most flow occurred via the drains. The data indicates that the timing of manure applications is more important than the application rate. There was evidence of a significant 'memory effect' lasting up to 2-3 months in drainflow losses which was absent in surface runoff losses.

Arable site (drained clay soil) - Results are presented for two contrasting years. In the first year (1999/2000), 130mm of drainage was measured from the autumn-sown plots and 180 mm from the spring-sown whereas the second year (2000/2001) was much wetter with 299 mm being measured. Total P losses over the two years ranged up to 2.8 kg ha⁻¹ (Table 1) with significantly elevated losses ($P<0.05$) arising where pig slurry was applied to stubble and not incorporated (connectivity maintained) or where incorporation was with discs (incorporation incomplete due to wet soils). Background losses from the control were considerably higher in year 2 with flow weighted concentrations from the control of 0.3 mg L⁻¹ c.f. 0.1 mg L⁻¹ in year 1; this shows that greater mobilization occurred in year 2.

Table 1. Total P losses in drainage water (kg ha⁻¹ yr⁻¹).

Autumn Cultivations	Control	Autumn slurry, ploughed in	Autumn slurry, disced in	Winter slurry to growing crop	Spring slurry to growing crop
Year 1	0.15	0.26	0.82*	0.52	0.60
Year 2	0.99	1.60	2.80*	0.74	1.10
Spring Cultivations	Control	Autumn slurry to stubble	Winter slurry to stubble	Spring slurry, ploughed in	
Year 1	0.24	1.35*	1.51*	0.57	
Year 2	1.13	2.27*	1.91*	0.80	

* significant increase over control ($P<0.05$)

Total dissolved phosphorus (TDP) losses were significantly increased ($P<0.05$) in both years by autumn or winter applications of slurry to uncultivated stubbles and in year 2 only by autumn applied slurry incorporated by discs. In year 1, when drainage continued until the end of May, significant ($P<0.05$) increases were also measured when spring applied slurry was either top dressed to the growing crop or applied to stubble and ploughed down.

Implications for mitigation

Applications to wet soils and directly before rainfall, should be avoided, especially on grassland soils and uncultivated cereal/maize stubbles where opportunities for incorporation are limited. Manures should be incorporated as soon as possible after application to reduce the risk of surface runoff losses and optimise contact time for P equilibration and adsorption by the soil. Opportunities for splitting manure applications should be explored to spread the risk from wet weather after application.

References

- Hodgkinson, R.A., Chambers, B.J., Withers, P.J.A. & Cross, R., 2002. Phosphorus losses to surface waters following organic manure applications to a drained clay soil. *Agricultural Water Management* 57, 155-173.
- Williams, J.R., Chambers, B.J., Smith, K.A. & Ellis, S., 2000. Farm manure land application strategies to conserve nitrogen within farming systems. *Proceedings of the SAC/SEPA Conference Agriculture and Waste Management for a Sustainable Future*, 167-179.

The impact of slurry management practices to reduce nitrate leaching on phosphorus losses from a drained clay soil

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Introduction

An estimated 47 million tonnes of livestock slurries supplying c.210,000 tonnes of nitrogen (N) and c.50,000 tonnes of phosphorus (P) are applied to agricultural land in the UK each year. On drained land, which accounts for an estimated 6.4 million ha of England and Wales (Withers *et al.* 2000), the rapid transfer of water from the soil surface to field drains, via soil macropores (commonly called 'by-pass' flow), has the potential to lead to high nutrient concentrations and losses in drainage waters following slurry application. Also, it is likely that the pattern of nutrient losses from arable land will be different to that from grassland, because factors that affect water movement (e.g. ground cover, cultivations, surface compaction, pore size distribution and continuity) differ between the two land use types. The objective of this study was to quantify the effects of different timings of cattle slurry application on nitrogen and phosphorus losses in water from a drained clay soil under arable and grassland management.

Materials and methods

The experiment was carried out on a heavy clay (54%) textured soil of the Denchworth Association, at the Brimstone Farm experimental facility, near Faringdon (Oxfordshire, UK). The site consists of 18 hydrologically isolated plots (40m x 48m), which had been in arable production for over 20 years until autumn 2001, when grassland was established on 9 plots. The soils are drained with pipe drains at 1 m depth and 48 m spacing, and have gravel backfill to within 30 cm of the surface, with secondary mole drains at 2 m spacing and 50 cm depth at right angles to the pipe drains. Cattle slurry (c.40 m³/ha, 120 kg/ha total N; 20 kg/ha P) was applied to the arable and grassland plots in autumn, winter and spring in 2003/04 and 2004/05, using an 11 m³ Joskin tanker fitted with a 12 m trailing hose boom. There were three replicates of each application timing. Inorganic fertiliser nitrogen was applied to all plots at the end of March 2004 and late April 2005 at standard recommended rates, to ensure that subsequent crop growth was representative of commercial practice. Drainage and surface runoff volumes were measured continuously using V-notch weirs. Drainage water samples were collected on a flow-proportional basis using automatic water samplers and analysed for nitrate-N and molybdate reactive phosphorus (MRP).

Results and discussion

In both years, nitrate concentrations in drainage waters from the arable plots (Figure 1 a) were greatest ($P<0.05$) following the autumn slurry applications and peaked at 70-130 mg/l N in the first 5-10 mm of drainage. The winter slurry applications had no effect ($P>0.05$) on nitrate concentrations in drainage waters, probably as a result of the cold and wet soil conditions delaying the nitrification of slurry ammonium-N to nitrate-N. Nitrate concentrations declined on all the treatments to less than 50 mg/l $\text{NO}_3\text{-N}$ after c. 25mm of drainage. In contrast, on the arable reversion grassland plots nitrate concentrations in drainage waters (Figure 1 b) were low and generally below the EC limit of 11.3 mg/l $\text{NO}_3\text{-N}$ throughout the drainage season. On these plots, there was no effect ($P>0.05$) of slurry application timing on drainage water nitrate concentrations in either year.

Mean nitrate-N losses (until fertiliser N was applied) were greatest ($P<0.05$) from the arable plots at 37 kg/ha N in 2003/04 and 15 kg/ha N in 2004/05, compared with 3.7 kg/ha N and 1.1 kg/ha N from the arable reversion grassland plots, respectively. On the arable plots, nitrate-N losses following the autumn and winter slurry applications (up to the time slurry was applied in the spring) were equivalent to 11% and 5% of the total slurry N applied in 2003/04, and 10% and 6% of total slurry N applied in 2004/05, respectively. On the arable reversion grassland plots, slurry application timing had no effect ($P>0.05$) on nitrate leaching losses, with losses following autumn / winter applications less than 4% of the total N applied in both years. The low nitrate leaching losses from the arable reversion grassland system were probably a reflection of the recently established grass sward accumulating N within organic reserves and greater crop uptake of slurry N in the period between application and the start of drainage.

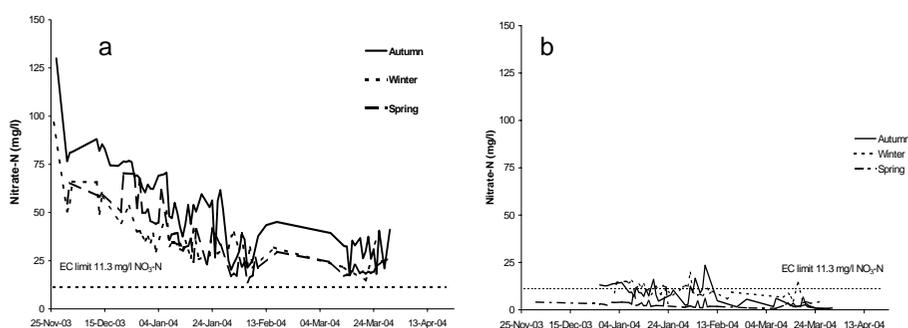


Figure 1. Mean nitrate-N concentrations in drainage waters at Brimstone Farm (2003/04) : (a) arable plots (b) grassland plots.

In both years, the autumn slurry application had no effect ($P>0.05$) on MRP losses in drainage waters from either the arable or grass plots. In 2003/04, c.20 mm of rain fell

10 days after the spring application that led to peak MRP concentrations of 0.9 mg/l P from the arable plots and 1.1 mg/l from the grassland plots. Similarly, in 2004/05, c. 30mm of rain fell in the 7 days following both the winter and spring slurry applications, and resulted in peak MRP concentrations of 1.3 and 0.4 mg/l from the arable plots, and 4.1 and 4.7 mg/l from the grassland plots respectively. These measurements show that rainfall on to 'wet' soils soon after slurry application caused phosphorus to move rapidly through the soil profile, via macropores ('by-pass' flow), in to drainage water.

In 2003/04, there were no differences in MRP losses between arable land and grassland with mean losses from both land uses less than 0.1 kg/ha P. In contrast, in 2004/05 mean grassland MRP losses at 0.68 kg/ha P were greater ($P < 0.05$) than from arable land at 0.11 kg/ha P (Figure 2). Also, MRP losses following the winter application to grassland were greater ($P < 0.05$) than from autumn application timing. The higher losses from grassland probably reflected the greater connectivity between the soil surface and field drains, as a result of 'by-pass' flow in cracks/mole channels, than on the cultivated arable plots.

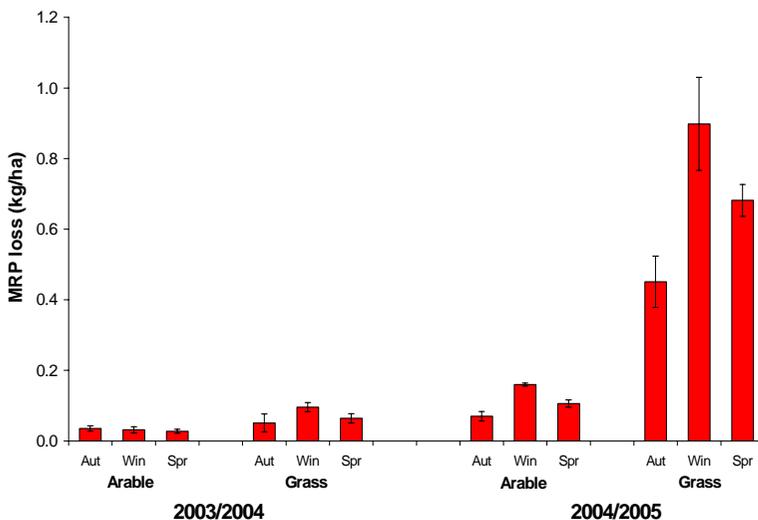


Figure 2. MRP losses in drainage waters at Brimstone Farm (2003/04 and 2004/05).

Conclusions

Moving the timing of the slurry application on arable land from autumn to winter and spring reduced nitrate leaching losses. However, applying slurry to 'wet' drained soils (e.g. in winter / spring), where rainfall followed soon afterwards, resulted in elevated drainflow MRP concentrations (so called 'pollution swapping'). This is particularly likely where there is good connectivity between the soil surface and field drainage system, via cracks/mole channels.

References

Withers, P.J.A., Davidson, I.H. & Foy, R.H., 2000. Prospects for controlling non-point phosphorus losses to water: a UK perspective. *Journal of Environmental Quality* 29, 167-175.

Mitigation options for reducing phosphorus runoff from biosolids¹

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Introduction

Biosolids, referred to as sewage sludge in the past, are normally applied based on their nitrogen (N) content, at least in cases where heavy metals are below the guidelines identified by federal law. Approximately 8 million dry tons of biosolids are produced in the U.S. each year (Bob Bastion, personal communication). About half of these biosolids are currently being land-applied, however, an increasing percentage are being placed in landfills or incinerated. In many cases the biosolids are not being land-applied because of concerns of P runoff and are currently being placed in landfills. Placing biosolids in landfills not only represents a waste of nutrients, it is taking landfill space needed for other materials. Incineration of biosolids wastes all of the N in this resource, while increasing CO₂ emissions into the atmosphere, which may affect global warming. Landfilling and incineration are more costly than land application. It costs about \$20-25/wet ton to landfill biosolids, not including transportation.

One way to reduce P runoff from biosolids would be to add aluminum (Al) or iron (Fe) compounds to them which would react with the P to form insoluble metal phosphate minerals. Typical products that might be used for this are alum or ferrous sulfate. However, these compounds are somewhat expensive, which may make the practice cost prohibitive. An alternative to refined Al and Fe chemicals would be water treatment residuals (WTRs). Water treatment residuals are formed when chemical flocculants, like alum or ferrous sulfate, are added to surface water taken from rivers or lakes for removal of sediments, algae, and other impurities. The WTRs formed

¹Mention of a trade name, proprietary product, or specific equipment does not constitute a guarantee or warranty by the USDA and does not imply its approval to the exclusion of other products that may be suitable.

from alum-treatment of drinking water were referred to as “alum sludge” in the past. This is the predominate WTR in the U.S., since alum is used to purify drinking water more often than other chemicals. The exact quantity of WTRs produced in the U.S. annually is unknown. However, we know that 68% of all public water systems use surface water. If Beaver Water plant, our local water provider, produces a typical amount of WTRs per consumer, then roughly 1.2 million dry tons of WTRs are produced in the U.S. annually.

The objective of this study was to measure P runoff from land fertilized with biosolids and to determine the effect of various chemical amendments, including WTRs, on P runoff.

Methods and materials

Rainfall simulations were conducted on 72 small plots cropped to tall fescue on a Captina silt soil. Prior to the simulations, soils were analyzed for water soluble P and Mehlich 3 extractable P. There were a total of 24 fertilizer treatments, with 3 reps/treatment in a randomized block design. Simulations were conducted twice prior to fertilizer applications to determine the effect of soil test P on P runoff. Simulations were also conducted on the day of fertilizer application, then at 1, 2, 4 and 6 weeks after fertilizer was applied.

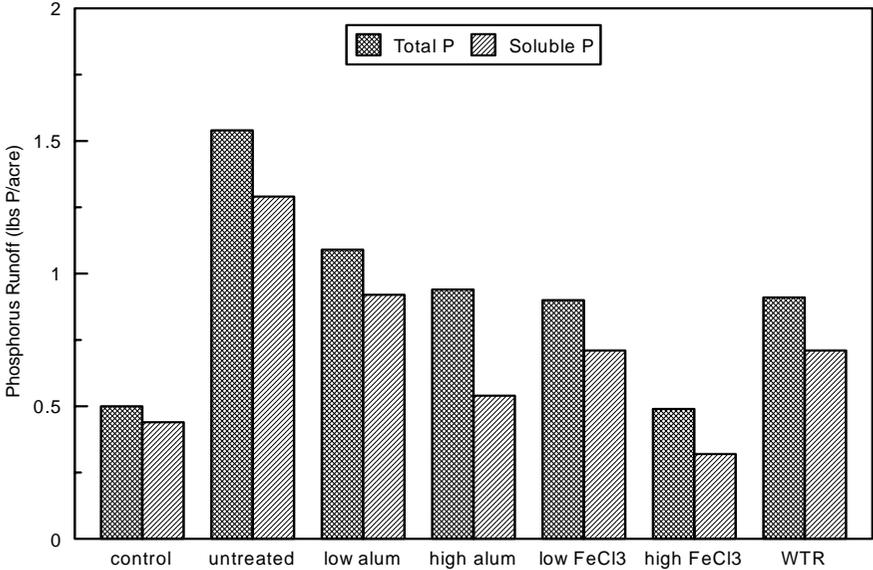


Figure 1. Effect of Al and Fe treatments on P runoff from biosolids.

Results and discussion

Phosphorus runoff from plots fertilized with untreated biosolids with roughly three times higher than from unfertilized control plots (Figure 1). Additions of alum and ferric chloride to biosolids resulted in a reduction in P runoff. However, these WTRs worked nearly as well as alum and ferric chloride.

Conclusions

- When WTRs were mixed with biosolids at a rate of 20% (wet basis) it reduced SP runoff by 45% and TP runoff by 41%. This could save cities a lot of money, since both biosolids and WTRs are being landfilled in many parts of the U.S. If the money saved on landfill expense were used to buy water treatment chemicals, then point source P discharges from WWTPs could be greatly reduced.
- Assuming our estimates of WTRs are accurate (1.2 million dry tons/year), then there may be enough WTRs to treat the entire amount of biosolids produced in the U.S. each year (8 million dry tons/year).

Ten years of progress in improving agricultural phosphorus management: a case study of the State of Delaware, USA

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Introduction

Nonpoint phosphorus (P) pollution of Delaware's fresh and coastal surface waters has been a recognized ecological problem for more than 30 years. Concerns reached a peak in 1997 when algal blooms (e.g., *Pfiesteria piscicida*) in bays and rivers were linked to major fish kills, human health problems, and agricultural nonpoint nutrient pollution. Land application of manures generated by the state's large and geographically concentrated poultry industry was regularly cited as a major contributing factor to water quality degradation. State and federal agencies responded by bringing together scientists, farmers, poultry integrating companies, environmental groups, and many others to devise a strategy that would mitigate the impacts of nutrients on surface and shallow ground waters. Following heated debates and often contentious discussions on the scientific principles of nutrient cycling, transport, and ecological impact, the Delaware Nutrient Management Act was passed in 1999. This law established a statewide nutrient management commission (DNMC) to develop science-based approaches to manage nutrients in a manner that could sustain agricultural productivity and improve water quality. Ten years after the original incidents that sparked the passage of this law, and similar laws in other U.S. states, considerable progress has been made in agricultural nutrient management, particularly with respect to P. In this presentation, we review and critically analyze the impacts of the process used in Delaware for the past decade to develop mitigation options for nonpoint P pollution. We also look to the future and assess new opportunities to further improve P management and restore the ecological quality of Delaware's waters.

Advances in nutrient management in Delaware from 1997 to 2007

The establishment of the DNMC effectively institutionalized the process of statewide nutrient management planning and has led to the systematic development of numerous programs that are improving agricultural P management. One of the key benefits of creating the DNMC was the provision of a formal structure for ongoing interactions between University of Delaware (UD) scientists and extension specialists, federal and state technical agencies, environmental regulators, and the agricultural community. This continuity in cooperative planning and action has been central to the progress made in nutrient management Delaware in the past 10 years.

The composition of the DNMC¹, which combines farmers, environmentalists, and technical experts in nutrient management, has also greatly facilitated cooperative discussions about causes and solutions to Delaware's nutrient management and water quality problems. At the same time, scientists and technical experts in the USDA Natural Resources Conservation Service (NRCS) and local conservation districts worked hand-in-hand with the DNMC to develop cost-effective solutions that are leading to more efficient "P-based" nutrient management by agriculture. Some of the more important actions that have occurred in the past 10 years include:

Nutrient management education and certification: To date, ~2000 farmers and other nutrient users have participated in formal nutrient management training and received certification as "nutrient generators", "private nutrient handlers", "commercial nutrient handlers", or "nutrient consultants". Continuing education is also required and in 2006 alone, Delaware Cooperative Extension specialists offered 60 training programs for > 2000 individuals.

Nutrient management planning: Any operation that applies nutrients to > 4 ha of land or manages > 8 animal equivalent units must become certified in nutrient management, develop and implement an approved nutrient management plan and submit an annual report to the DNMC. Nutrient management plans are now in place for ~185,000 ha, or about 99% of Delaware's agricultural crop land.

Manure nutrient relocation programs: A DNMC supported manure relocation program was established in 2001 to help farmers with manure surpluses transport excess manure to farms in need of manure nutrients or to alternative uses. About 400,000 tons of poultry manure has been relocated to date; key outlets include a plant that produces pelletized manure-based fertilizer and regional mushroom farms.

Risk assessment protocols for nonpoint P pollution: Research and on-farm field assessments conducted by University of Delaware scientists led to the development, and adoption by the DNMC, of (i) soil test criteria to identify "high P" soils where fertilizers and manures must be applied at crop P removal rates; and (ii) a Phosphorus Site Index (PSI) used to characterize the risk of P loss to water based on site characteristics, hydrology, and fertilizer/manure P management practices. Farmers have the option of using the soil test P threshold or a PSI to guide P management of manures and fertilizers on their farms.

¹ The DNMC is composed of 8 members of the agricultural community (representatives of grain crop producers, animal producers, vegetable growers, commercial nurseries, and golf/lawn care industry), 2 representatives from environmental advocacy groups, a representative of commercial nutrient applicators, a nutrient consultant, a public citizen, the director of the state Soil & Water Conservation division. Ex-officio members include the Secretary of the Department of Natural Resources & Environmental Control and the Secretary of the Department of Agriculture. See <http://dda.delaware.gov/nutrients/index.shtml> for more details and publications on the DNMC.

DNMC memoranda of understanding with poultry industry: All poultry integrating companies signed a formal agreement in 2001 to foster cooperative efforts to improve nutrient management, leading to a number of successful actions: (i) the improved management of P in poultry feed, which, by use of phytase enzymes and reductions in mineral P, has reduced manure P content by ~30%; (ii) environmental stewardship programs for poultry farmers; (iii) certification and education programs completed by poultry industry employees and poultry farmers; and (v) ongoing, industry-wide evaluations of alternative uses for excess poultry litter.

Environmental permitting for certain animal operations: The DNMC has worked to improve nutrient management on Concentrated Animal Feeding Operations (CAFOs) that are required by the U.S. Clean Water Act (1972) to have permits designed to protect surface water quality. An agreement between the DNMC and the US Environmental Protection Agency followed numerous public meetings where policy decisions were debated and discussed. The agreement authorizes the DNMC to implement regulations and handle initial enforcement efforts for CAFOs in Delaware.

Research and demonstration projects support best management practices (BMPs): Years of research conducted by UD scientists and colleagues led to a wide range of cost-effective nutrient management BMPs. The DNMC identified and published criteria for 56 BMPs designed to minimize nonpoint nutrient pollution (“Nutrient Best Management Practices: Today’s Agriculture: A Responsible Legacy”). The DNMC also funds research and demonstration projects each year to aid in the development of science-based environmental policies.

Nutrient surpluses have been reduced: Excess nutrients are common in areas dominated by animal-based agriculture and often lead to nonpoint nutrient pollution of ground and surface waters. Recent analyses of trends in statewide nutrient balances have shown that the actions taken by the DNMC, the University of Delaware, NRCS, and Delaware’s agricultural community have significantly reduced nutrient surpluses. For example, estimated statewide P surpluses decreased by 42%, from 25 kg P ha⁻¹ to 14 kg P ha⁻¹, between 1997 and 2005. Nitrogen surpluses were also reduced, from 62 to 35 kg N ha⁻¹. Major contributing factors were reductions in fertilizer use, likely due to increased nutrient management planning, and improved P feeding practices by Delaware’s poultry industry.

In conclusion, the concerted efforts of a diverse group of individuals, agencies, and organizations committed to sustaining agriculture and protecting water quality, have markedly improved agricultural nutrient management in Delaware in a relatively short period of time. Significant challenges do remain, but the partnerships built in the past 10 years between scientists, policy-makers, technical and regulatory agencies, and the agricultural community provides a solid foundation for continued progress.

Effects of destruction and burial dates of cover crops on runoff, erosion and phosphorus losses in a maize cropping system

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Introduction

In Wallonia (Belgium), fields planted with spring row crops (sugarbeet, potato, maize, chicory) are particularly prone to runoff and soil erosion by water (Bielders *et al.*, 2003) that results in surface water eutrophication, in particular by labile phosphorus. To reduce these problems, planting cover crops is an admitted efficient solution. Indeed, it ensures a soil cover during the winter. It may possibly also improve the soil structure after its burial. However, little information exists about the optimal cover crop destruction and burial dates with respect to erosion control, as these dates influence both the duration of soil cover and the amount of biomass added to the soil.

The objectives of the study are to measure the impact of the date of winter cover crop destruction and burial on runoff, erosion and phosphorus losses in a maize cropping system.

Experimental design

Runoff, soil and P losses were measured in a fully randomised, continuous maize cropping experiment with 2 factors repeated 3 times and 3 control plots was carried out during 2 years (2004-2005 and 2005-2006) on a loamy soil with an average 8 % slope (Nodebais) and on a sandy loam soil with an average 12 % slope (Bonlez). The two factors were (1) the combination of destruction date of the cover crop and its date of burial (table 1) and (2) the cover crop species (rye and ryegrass).

Table 1. Cover crop destruction dates and burial dates in 2005 and 2006.

Destruction date of the cover crop		Burial date	
Mid - march	D1	40 days later	D1B1
Mid - april	D2	40 days later	D2B2

Runoff, soil and P losses were measured by means of 90 m² runoff plots on control and rye and ryegrass plots during the winter period and on plots previously covered by rye, ryegrass and on the control plots during the maize season. Labile phosphorus was determined in the eroded sediments for selected erosive events (method ammonium acetate-EDTA). Dissolved phosphorus (method ammonium molybdate) was also measured in the runoff water of these events. We call bio-

available P (P_{bio}) the sum of labile, sediment-bound P and dissolved P. Total phosphorus (TP) was determined (Olsen & Sommers method) for a limited number of samples. Total above and belowground biomass of the cover crop was measured before destruction. In case of absence of normality of the data when performing ANOVA ($\alpha = 0.05$), the criteria of normality and homogeneity of the variances of the residuals were selected to impose a logarithmic transformation. Treatment averages were compared with the Tukey test.

Results

During the 2004-2005 intercropping period (1/11/04-17/05/05, total rainfall on the sandy loam site = 298 mm, total rainfall on the loamy site = 354 mm), cover crops developed well and a large amount of dry biomass was achieved: 4.75 t/ha on average for the 2 destructions date, resulting in a significant reduction by more than 95 % of runoff ({sandy loam site : control plots = 62 mm}, {loamy site control plots = 55.5 mm}), soil loss ({sandy loam site : control plots = 40.1 t/ha}, {loamy site : control plots = 18.5 t/ha}) and P_{bio} loss ({sandy loam site : figure 1}, {loamy site : control plots = 3.3 kg /ha}). Moreover, we found a linear relationship between labile sediment-bound P and TP contents: labile sediment-bound P represents on average 9.5 % of TP. In 2005-2006 (1/11/05-16/05/06, total rainfall on the sandy loam site = 279 mm, total rainfall on the loamy site = 161 mm), cover crops only reached 1.5 t/ha on average on the 2 destruction dates but runoff (sandy loam site: control plots = 11.6 mm) erosion (sandy loam site: control plots) and P_{bio} losses were still reduced by 80 % (sandy loam site: figure 1). Due to low intensity rainfall, the loamy site did not experience any runoff, erosion and P_{bio} losses during this period. There was no effect of cover crop management and species on runoff, soil and P_{bio} losses.

During the maize cropping season 2005 (17/05/05-1/10/05, total rainfall on the sandy loam site = 200 mm, total rainfall on the loamy site = 220 mm), previously covered plots of the loamy site showed up to 90 % reduction in runoff (control plots = 11.6 mm), soil loss (control plots = 5.5 t/ha) and bio-available P loss (figure 1) due to large amounts of buried biomass. Among the previously covered plots of the loamy site, the previously D2B2 plots showed a significant decrease of 50 % in runoff, erosion and P_{bio} losses compared to the previously D1B1 plots, due to larger amount of buried biomass (total dry biomass at the time of the first destruction = 3 t/ha, total dry biomass at the time of the second destruction = 6.5 t/ha). This residual cover effect was not observed on the sandy loam site. However, this absence may be due to the greater textural heterogeneity at this site as well as less erosive rainfall during that season.

During the next maize season (16/05/05-1/10/06), none residual cover effect was observed at any site.

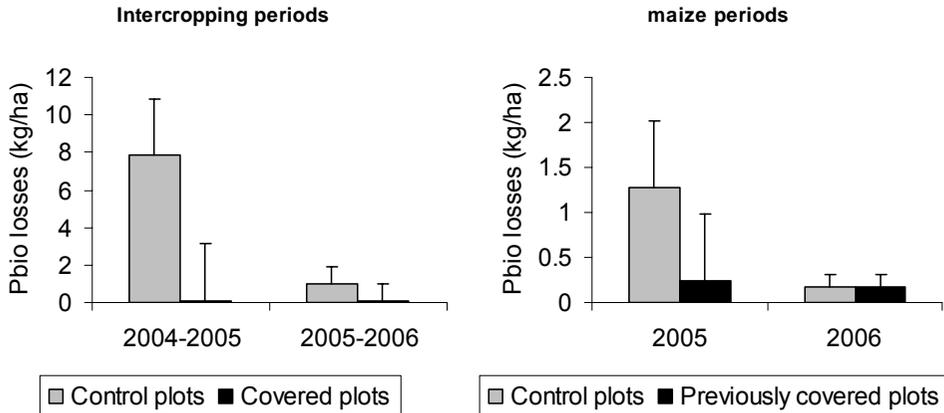


Figure 1. Pbio losses during the 2004-2005 and 2005-2006 intercropping periods on the sandy loam site and Pbio losses during the maize periods 2005 and 2006 on the loamy site.

Conclusions

During the intercropping period, cover crops are very effective: erosion and phosphorus losses were already reduced by 80 % with poor cover development (an average of 1.5 t/ha of dry biomass). When larger amounts of cover biomass were reached (an average of 4.75 t/ha of dry biomass) these reduction reached up to 95%. During the first maize cropping season, we measured up to 90 % reduction in erosion and phosphorus losses, most likely as a result of improved soil aggregate stability and reduced crusting due to the high quantities of buried biomass in 2005. Later cover destruction reduces erosion and P losses during the maize cropping season. On the contrary, the low quantities of buried biomass in 2006 did not allow the expression of a residual cover effect during the subsequent maize season.

References

Bielders, C., Ramelot, C. & Persoons, E., 2003. Farmer perception of runoff and erosion and extent of flooding in the silt-loam belt of the Belgian Walloon Region. *Environmental Science and Policy* 6, 85-93.

Can tramline management be an effective tool for mitigating phosphorus and sediment loss?

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Introduction

In England, the Environmental Stewardship scheme awards points for specific environmentally beneficial activities ranging from maintaining hedgerows to the use of vegetated headlands around field margins. The scheme initially focused on improving biodiversity and limiting the ecological impacts of land management. However, recent attention has turned to how this agri-environment scheme might be modified to incorporate practical and cost-effective management techniques for controlling diffuse pollution to water.

Tramline wheelings are the unseeded bout lines left bare in autumn and used to facilitate spraying operations in combinable cropping systems. Recent research has identified that tramlines can represent an important pathway in the loss of sediment and phosphorus from a silty clay loam soil under arable management on a 7 degree slope in western England (Silgram, 2004; Silgram *et al.*, submitted). The project reported here extends this work to confirm (i) whether tramline wheelings are important across a broader range of soil textures and on shallower slope angles, and (ii) whether relatively simple tramline management can function as an effective mitigation technique in agri-environment schemes aimed at reducing erosion and loss of sediment and phosphorus from land to surface water systems.

Methodology

The Mitigation of Phosphorus and Sediment (MOPS) project includes three sites: only the two ADAS sites are discussed here. On a uniform 4 degree slope on a sandy soil at Hattons in Staffordshire, England, unbounded long hillslope segments each 3 m wide were established to quantify surface runoff. Similar unbounded hillslope segments were established at a second site at Rosemaund in Herefordshire, England, in a slightly wetter area 100 km southwest of the first site on a uniform 5 degree slope and contrasting silty clay loam soil. Hillslope segments were 3.5 m wide at both sites, and were 270 m long at Hattons and >100 m long at Rosemaund.

Replicated treatments were imposed on the hillslope segments: (i) vegetated areas between tramlines (“no tramline”); (ii) areas with conventional tramlines (“tramline”);

(iii) areas where a cultivator fitted with a ducksfoot tine were used to disrupt (to 6cm depth) the compacted surface of the tramline wheeling (“tramline disrupted”); and in year two only, (iv) areas where the sprayer was run over the emerging crop during the autumn (“tramline offset”), rather than running on the unseeded tramline area.

Surface runoff from each hillslope segment was channelled via guttering and plastic pipes through novel sampling devices which divert a user-defined proportion of the total runoff into collection tanks. Event-based samples were analysed for total phosphorus (TP), total dissolved phosphorus, particulate phosphorus, total nitrogen (TN), total dissolved nitrogen, organic nitrogen, and suspended sediment.

Results

In the first winter of monitoring at Rosemaund (Table 1), results showed surface runoff on undisrupted tramlines was 5-17% of rainfall, but was <0.6% of runoff on “no tramline” and “disrupted tramline” areas. Results illustrate the highly significant (P<0.001) dominant effect of tramlines as surface pathways for the transport of surface runoff, sediment, and phosphorus loads to the edge-of-field area, and supports earlier results (Silgram, 2004). Across all monitored events and treatments, 77-85% of the TP in measured runoff was in particulate form. For TN, 71% was present as organic N on disrupted tramlines but only 54-57% of TN was present as organic N where there were no tramlines or where tramlines had been disrupted. Similar results were observed at the second, sandy site at Hattons (not shown).

Table 1. Summary results from runoff events at Rosemaund, winter 2005/6.

		Runoff mm	Runoff litres	% rainfall as runoff	Sediment kg ha ⁻¹	TP kg ha ⁻¹
No tramline		0.3	63	0.5	3	0.01
Tramline	Not disrupted	5.8	1807	10.9	357	1.32
Tramline	Disrupted	0.3	105	0.6	6	0.02

The dominant role of tramlines as surface loss pathways at both sites reflects the combined effects of (i) compaction caused by the sprayer traversing the unseeded area in the autumn, and (ii) the lack of vegetation in the unseeded tramline area. The first effect reduces the infiltration rate, while the second effect means there is no emerging vegetation canopy to limit the erosive energy in incident rainfall. Deducing the relative importance of these effects could help develop targeted mitigation options, and led to treatment (iv) being introduced in winter 2006/7 at both sites.

Preliminary results for winter 2006/7 are shown in Figure 1 for an example event at the Hattons site which totalled 18 mm rainfall with a peak intensity of 2.2 mm h⁻¹. The percentage of rainfall lost as runoff was 3.0% (no tramline), 14.4% (tramline), 3.9% (tramline disrupted), and 11.7% (tramline offset). Corresponding sediment

loads in runoff were 7.0 kg ha^{-1} (no tramline), 61.3 kg ha^{-1} (tramline), 12.7 kg ha^{-1} (tramline disrupted), and 33.4 kg ha^{-1} (tramline offset). Results from winter 2006/7 (Figure 1) verify the effects reported for the previous winter (Table 1). Initial data from offset tramlines suggest the majority of the enhanced losses from tramline areas are related to surface compaction rather than to a lack of vegetation cover (Figure 1).

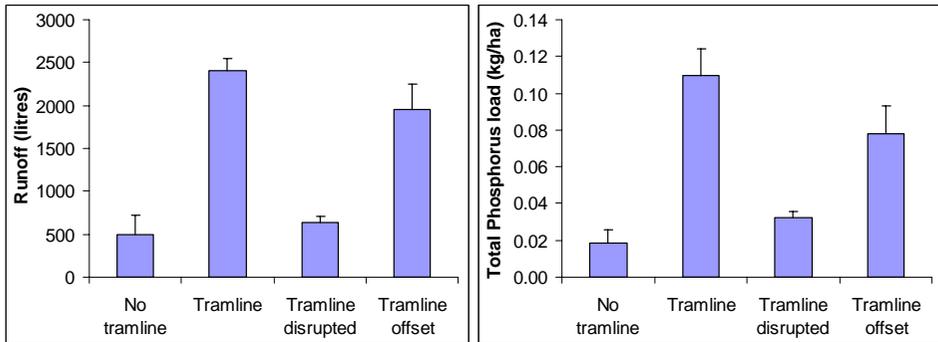


Figure 1. Runoff and TP loads from Hattons in event 3, winter 2006/7. Standard error bars are shown (four replicates).

Conclusions

Research on both sandy and silty clay loam soils across two winters has shown that tramline wheelings can represent the dominant mechanism for surface runoff and transport of sediment, phosphorus and nitrogen from arable fields with shallow to moderate slopes. Shallow disruption of tramline wheelings using a simple ducksfoot tine device consistently and dramatically reduced ($P < 0.001$) surface runoff and loads of sediment, TN and TP to levels close to those measured in vegetated areas between tramlines. Ongoing work is needed (i) to explore the effectiveness of different tine types and depths on contrasting soils, and (ii) deduce the most practical method of integrating disruption techniques into field operations for mitigating against diffuse pollution in nationally-coordinated agri-environment schemes.

Acknowledgements

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References

- Silgram, M., 2004. Towards understanding factors controlling transfer of phosphorus within and from agricultural fields. Final report to Defra for project PE0111. 18pp.
- Silgram, M., Jackson, B. & Hutchins, M., (submitted). Surface runoff, sediment and phosphorus loss from an arable clay soil: effects of slope length and tramline wheelings. Submitted to Journal of Environmental Quality.

High risk areas of phosphorus losses from agriculture - three different production systems

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Introduction

High risk areas of phosphorus (P) losses comprise areas of high natural risk of P losses and areas of high risk caused by agricultural management. For example, the risk of P losses is naturally high on areas with steep slopes and highly erodible soils. Other natural factors that influence the risk include the distance from field to stream and the geological P content of the soil. The natural risk of P losses may be reduced by management practices, for example by no till or reduced P application rate. Agricultural production systems differ widely in their risk of P losses, among others with regard to soil erosion risk and soil P status.

In some production systems mitigation methods for reduced P losses are not easy to implement and risks of P losses are high even with best management practices implemented. Production of potato- and some vegetables include high P application rates and for some of these crops the soil management by harvesting the crop contributes to a high erosion risk. Dairy production may cause high P application rate by manure application, but the soil erosion risk is often low on grassland. In contrast, in cereal production, P application may be relatively low, whereas the erosion risk may be high. To avoid high risk areas within a catchment, high risk production systems should be located in areas with a naturally low risk of P losses.

The objectives of this study were to use a P index to identify the agricultural production systems with the highest risk of P losses and to evaluate the importance of these areas for the total P contributions from a catchment.

Table 1. Characteristics of three catchments.

Catchment	Size	Agricultural area	Precipitation	TP conc.	SS conc.
	Ha	%	mm year ⁻¹	mg L ⁻¹	mg L ⁻¹
Cereal	449	61	785	0.27	153
Grassland and dairy	91	94	1189	0.15	11
Potato/vegetable and livestock	65	62	1230	0.46	109

Study sites

Three catchments within The National Agricultural Environmental Monitoring Programme (JOVA) were chosen to illustrate three different agricultural production systems, 1) cereal production with low livestock density (Skuterud), 2) grassland with high livestock density (Time) and 3) potato and vegetable production with high livestock density (Vasshaglona). A characterization of the three catchments is included in Table 1.

Monitoring of the three sites include discharge measurements, water quality sampling and registration of agricultural practice. Information from farmers regarding their agricultural management practices were collected each year and comprise among others information on crop, yield, nutrient application (time, amount and method) and soil tillage. Values for soil P status (mg P-AL/100g) and erosion risk as well as other factors needed to calculate the P index for fields within the catchments were collected from farmers or statistical sources according to Bechmann (2005).

Risk factors for P losses in three production systems

The risk of erosion and the soil P status of the three agricultural production systems differed widely. The soil P status was lowest for the cereal production areas and highest for the potato/vegetable areas with high livestock density (Table 2). The natural soil erosion risk was highest for the cereal production areas, whereas the other areas had relatively low erosion risk - that is low slope and erodibility of soil. The soil management contributed most to erosion in the potato/vegetable area, whereas the soil management factor was low in the grassland and dairy catchment.

Table 2. Risk factors for three catchments in the JOVA-programme.

Catchment	Soil P status mg P-AL/100g		Erosion risk* Tonne/ha		Soil management factor
	Mean	Std.dev. (min.-max.)	Mean	Std.dev. (min.-max.)	Mean C- factor**
Cereal	8	3 (4-17.5)	1.1	0.5 (0.4-2.5)	0.4
Grassland and dairy	21	10 (1-45)	0.4	0.1 (0.05 - 1.0)	0.1
Potato/vegetable and livestock	26	10 (13-53)	0.7	0.2 (0.4-1.1)	0.9

*Erosion risk at autumn ploughing

**Soil management factor in the USLE

Reductions in P losses may be obtained by reductions in soil P status caused by reduced P applications. The most important mitigation strategy, however, in Norway has been the reduction of erosion risk. The soil erosion risk of cereal areas may be

efficiently reduced with no-till management. For potato and some vegetable areas, however, efficient reduction of erosion risk may not be obtained by this practice.

Importance of production systems

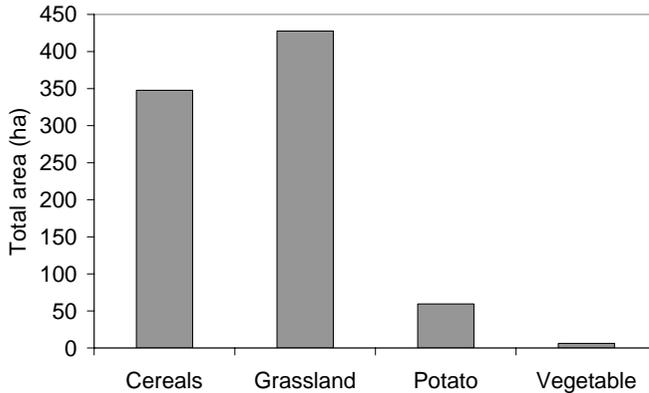


Figure 1. Importance of different production systems in Norwegian agriculture.

Figure 1 illustrates the importance of cereal and grassland in Norwegian agriculture. Potato and vegetable areas constitute only a small area. On potato and vegetable areas, however, serious difficulties may arise when trying to reduce P losses and at the same time keeping the production at maximum level. Reduced erosion by no till will reduce the risk of P losses on cereal areas. Correspondingly, reduced P application will reduce the risk of P losses on grasslands. When these measures have been implemented, the potato and vegetable areas may be left as the main high risk areas with significant contributions even though they are only covering a small area.

During this presentation the evaluation of risk of P losses for the different production systems will be based on P index calculations and also the effects of mitigation strategies for reduced P losses will be based on the P index approach.

References

Bechmann, M.E., Stålnacke, P. & Kværnø, S.H., 2007. Testing the Norwegian Phosphorus Index at the field scale. *Agriculture, ecosystems and environment* 120, 107-128.

Mitigation options for phosphorus and sediment (MOPS): tillage treatments and the use of vegetative barriers

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Introduction

Diffuse phosphorus (P) pollution presents a serious problem in the UK, contributing to the eutrophication of surface waters. Losses of P from agriculture are of particular concern, as agricultural systems traditionally have high inputs of P applied in fertilisers and manures to enhance productivity. In the UK, the agricultural P surplus has been estimated to average around 16 kg ha⁻¹ per year (Withers *et al.*, 2001). Although there has been extensive research into effective treatments for reducing soil erosion from arable land (e.g. Quinton & Catt, 2004), less is known about the effectiveness of mitigation options for reducing P losses. To address this research gap, the Defra funded MOPS (Mitigation Options for Phosphorus and Sediment) project is investigating a range of tillage treatments with potential for mitigating P losses from arable land associated with combinable crops.

Experimental design

Field monitoring is being carried out at three field sites in the UK (Herefordshire, Staffordshire, Leicestershire) with contrasting soils (silty clay loam, sandy loam, clay). At each site, trial treatments have been selected which are appropriate for each soil type. The treatments investigated include the use of crop residues at the sandy soil site, and minimum tillage, contour cultivation, and a beetle bank as a vegetative barrier half way down the slope, at the clay soil site. Fifty-two unbounded hillslope length plots are being monitored across three sites, which allow replication of different treatments and combinations of treatments. From each hillslope length, surface runoff is intercepted at the base of the slope by a 3 m trough, from which runoff is piped into a flow splitter and collection tank for sampling. Samples are collected from each tank on an event basis, and analysed for suspended sediment (SS), total P (TP) and total dissolved P (TDP).

Results

The results show that P losses at all three sites are principally particulate (>76 %), and that control of erosion is therefore important in mitigating losses of P from arable land. Results from the first year of monitoring showed that the use of crop residues significantly reduced runoff, SS and P losses from arable land (Table 1).

Table 1. Results from crop residue trials on sandy soils in Staffordshire, UK. Data are cumulative loads from eight events monitored in 2005-2006. TP = total phosphorus, SS = suspended sediment, B = straw baled and removed, C = straw chopped and spread, NT = no tramline on hillslope, T = tramline on hillslope.

Treatment		Runoff (l)	Rainfall:Runoff (%)	SS (kg ha ⁻¹)	TP (kg ha ⁻¹)
B	NT	312	0.2	21	0.06
C	NT	172	0.1	12	0.03
B	T	6802	5.2	499	1.52
C	T	5056	4.0	298	0.99

Chopping and spreading straw, instead of baling and removing it, consistently reduced TP losses per unit area (typically by 30-60 %) in eight events, both for hillslope lengths containing tramlines, and for hillslope lengths without tramlines. No significant differences at $p < 0.05$ were found between the other treatments in 2005-2006, and results from the second year of monitoring also showed no significant differences in P losses between treatments, due to the variability in P losses between events.

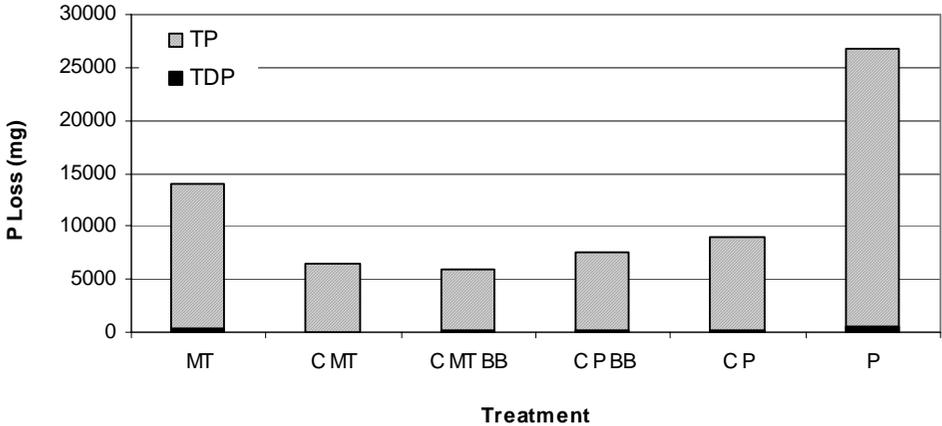


Figure 1. Cumulative phosphorus losses for eight events monitored in 2006-2007, for hillslope lengths under different treatments on clay soils in Leicestershire, UK. TP is total phosphorus, TDP is total dissolved phosphorus. MT = minimum tillage, C = contour cultivation, BB = beetle bank, P = plough.

Cumulative losses of P from eight events in 2006-2007 for the field site in Leicestershire (clay soils) are presented in Figure 1. The results show that both minimum tillage and cultivation on the contour reduced P losses compared to conventional tillage and up and down slope cultivation at this site. On average, minimum tillage reduced TP losses by 12.7 g from the up and down cultivation, and by 2.5 g for the contour cultivation. Contour cultivation reduced TP losses by 17.7 g

from ploughed land, and by 7.5 g from the land under minimum tillage. The beetle bank is also effective, although this may be because it promotes contour cultivation.

Conclusions

Phosphorus losses are being monitored at three experimental sites in order to determine effective treatments for mitigation of P losses from arable land. The results from the first two years of monitoring indicate that as P losses at the three sites are principally particulate, treatments which reduce erosion have potential for reducing P losses. The use of crop residues, cultivation on the contour and minimum tillage are all effective treatments, although variability between events means that the reductions in P losses are not significant for the clay site. Results from the 2007-2008 field season may provide further evidence to support the effectiveness of minimum tillage and contour cultivation in reducing P losses from arable land.

References

- Quinton, J.N. & Catt, J.A., 2004. The effects of minimal tillage and contour cultivation on surface runoff, soil loss and crop yield in the long-term Woburn Erosion Reference Experiment on sandy soil at Woburn, England. *Soil Use and Management* 20, 343-349.
- Withers, P.J.A., Edwards, A.C. & Foy, R.H., 2001. Phosphorus cycling in UK agriculture and implications for phosphorus loss from soil. *Soil Use and Management* 17, 139-149.

Site specific measures to mitigate P-loads in the Dutch Province of Limburg

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Introduction

Reaching surface water quality goals in rural areas of the Netherlands, especially for P, requires site-specific measures (SSM) in addition to the generic manure policy already in force. For the efficient introduction of SSM, critical fields need to be located and the cost-effectiveness of SSM needs to be known. Non-source measures need particular attention, because the generic manure policy has focused on source measures only. We developed a fast and simple method to locate critical fields and to select SSM, and calculated cost-effectiveness with models.

Method

A P-pilot was set up in the Province of Limburg by the government service for land and water management in a rural sandy region of approximately 1500 ha with high past manure rates and extreme P-concentrations in main streams (up to 2 mg.L⁻¹ P). The water board, Province and farmers' organizations are involved in the project; Alterra is doing the research.

To select high-risk fields, a semi-quantitative regional diagnosis was developed based on P-transport risk and P-source risk. After mapping the pilot area for P-transport risks (soil and hydrology GIS-data), the P-source risk of the fields with highest total P-transport risk was assessed by interviewing the corresponding farmers on cropping and manure history to calculate accumulated P. Only the soil of fields with the highest total weighted risk was sampled to confirm the P-source risk.

To select SSM, four categories were distinguished, based on the depth of the P-transport routes and the depth of the P-rich topsoil (table 1). In this way we were able to match the right measure with the right field. After selecting measures and fields, we projected a plan to execute and monitor the measures for the next three years (Noij et al., 2006).

Our preliminary modelling, a separate project for the Ministry of Agriculture, Nature Conservation and Food Security, considers the cost-effectiveness of SSM. Farm models were used to calculate costs and fertilizer rates, and the hydrological model SWAP and the nutrient model ANIMO were used to calculate the average effects of the measures on P-load (over a 15 year period, 6 years after implementation).

Regional diagnosis

We produced risk maps, based on the hydrological transport routes for runoff, trenches and shallow ditches, pipe drains, deeper ditches, and two versions of the total transport risk. The first, risk per entire field, reveals a higher risk for larger fields. The second, risk per ha, indicates hot spots. These total transport risk maps were integrated with a source risk map, derived from interviews and subsequent soil sampling (figure 1).



Figure 1. Total integrated risk of P-loss to surface water in the pilot area. (Only fields with high total transport risk, as source risk, and therefore total risk, is not available for fields with low transport risks!).

Matching measures with fields

Blocking surface runoff can be applied to fields with superficial transport routes (table 1). P-mining matches fields with deeper transport routes and shallow P-profile. P-mining may also be suggested for fields with a deeper P-profile, although this will not reduce P-load in the short term. It would need to be combined with CDDS. CDDS is also suggested for fields with superficial transport routes and a deep P-profile; blocking surface runoff would be insufficient here, as this would result in transport through deeper layers that still contain too much P. The constructed wetland is not specific for any category, but can be considered as an end-of-pipe measure for a group of fields, an entire farm or a subcatchment.

Table 1. Matching measures with corresponding risks. Measures in **bold** were selected for monitoring.

Transport risk		Source risk	
		Depth of P-profile	
Depth	Specified routes	Shallow	Deep
Superficial	Runoff Shallow trenches Deep trenches Shallow ditches	Blocking surface runoff Leveling soil surface Installing buffer strip Filling shallow trenches Filling shallow ditches	
		Conventional pipe drains or CDDS¹	CDDS¹
Deep	Pipe drains Deep ditches	P-mining²	CDDS¹ washing machine ³
Total	All routes	Constructed wetland with horizontal flow ⁴Stream diversion Source measures	

¹Controlled Deep Drainage System (CDDS): prevent runoff and interflow, and force water through deeper soil layers, that are able to retain P. ²Extract P from the topsoil by crops.

³Leach P from the subsoil by alternating high and low groundwater levels; ⁴Divert water with bad quality to protect nature area. See also van Os (2007).

Modelling

SSM can be ranked in order of decreasing effectiveness [constructed wetland > conventional drainage (-90 cm) > P-mining > CDDS (controlled level at -60 cm) > blocking surface runoff], in order of increasing costs [P-mining (arable) <, blocking runoff < P-mining (dairy) < conventional drainage < constructed wetland < CDDS] and in order of cost-effectiveness [P-mining (arable farm) > conventional drainage > P-mining (dairy farm) > blocking surface runoff > constructed wetland >> CDDS]. P-mining costs are low only if the original rotation can be maintained on arable farms and grazing remains possible on the dairy farm grassland that is not mined for P. Note, however, that only the costs of P-mining are restricted to a period of 15-30 years, depending on the original P status. The unfavourable results for CDDS are caused by the high, controlled groundwater level.

Conclusion

This regional diagnosis can be used to locate critical fields quickly, allowing fast implementation of SSM to prevent P-loss to surface water. It may also be used to suggest the best measure for each critical field, something we hope to confirm with our monitoring plan, which will collect data on our experimental area over the next three years. Meanwhile, preliminary modelling results can be used for the cost-effectiveness of the measures being studied. P-mining and drainage are the most

interesting measures, but drainage may hinder regional groundwater management goals. None of the measures reduces P-loads by more than 35%, except for an intensively maintained constructed wetland under average load conditions (85%). Therefore, we recommend P-mining until the P-status of the soil and the P-loads are reduced to moderate values that suit a cost-effective, constructed wetland. Blocking surface runoff may be helpful and can easily be introduced by the farmers themselves.

References

- Noij, I.G.A.M., van Bakel, P.J.T., Smidt, R.A., Massop en, H.T.L. & Chardon, W.J., 2006. Fosfaatpilot Noord- en Midden-Limburg. Plan van Aanpak en Monitoring. Alterra-rapport 1255, Wageningen the Netherlands.
- Os, E.A. van, Noij, I.G., van Bakel, P.J., Groenwold, J.G. & van der Bolt, F.J., 2007. Effect of source and hydrological measures on reducing the load of N and P to surface water. Poster presentation to IPW5-conference, Silkeborg, Denmark.

Modelling cost-minimising strategies for improving the aquatic environment of the Baltic sea

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Key words:

Nutrient reductions; Abatement costs; Programming model; Cost-Effective; Water Framework Directive.

Background and aim

This paper and the model development described here are parts of the research programme MARE (MARine Research of Eutrophication), which is an eight-year (1999—2006) research programme funded by the Swedish Foundation for Strategic Environmental Research, MISTRA. The objective is to develop a user-friendly decision support system termed NEST. NEST demonstrates what could be a cost-effective distribution of the measures needed in order to achieve a desired environmental quality in the Baltic Sea, and NEST aims to enable users to evaluate possible reductions in nitrate and phosphate loadings with respect to costs, effects on the Baltic Sea as well as the timescale(s) for the effects to take place. (See: www.mare.su.se).

Inputs of nutrients, such as nitrogen and phosphorus, to the sea are natural prerequisites for life, and nutrient emissions only become problematic when the inputs increase to such extents that the original properties or functions of the ecosystem are affected, and the sea becomes too eutrophicated (see www.mare.su.se). Intense algal blooms, turbid water with reduced transparency, oxygen deficiency and reductions in the amount and presence of sediment-living animals may be the severe results of too much eutrophication.

The economic values of eutrophication of coastal ecosystems, and reductions in the eutrophication levels, are not investigated very intensively, but studies performed in the Baltic region indicate that the values can be significantly large. The enclosed brackish-water Baltic Sea, with its slow water exchange and natural barriers, is particularly sensitive to eutrophication. However, within this sea region different effects will dominate in separate parts of the sea basins due to large differences in natural conditions between the basins, as well as different loads of nutrients to the basins.

In all there are nine countries surrounding the Baltic Sea, and in the cost-minimisation model 24 drainage basins within these countries form basic units for the estimation of costs and loads for the different policy measures relevant for reducing nutrient loads to the Baltic Sea.¹ The drainage basins can be seen from the map (Figure 1).

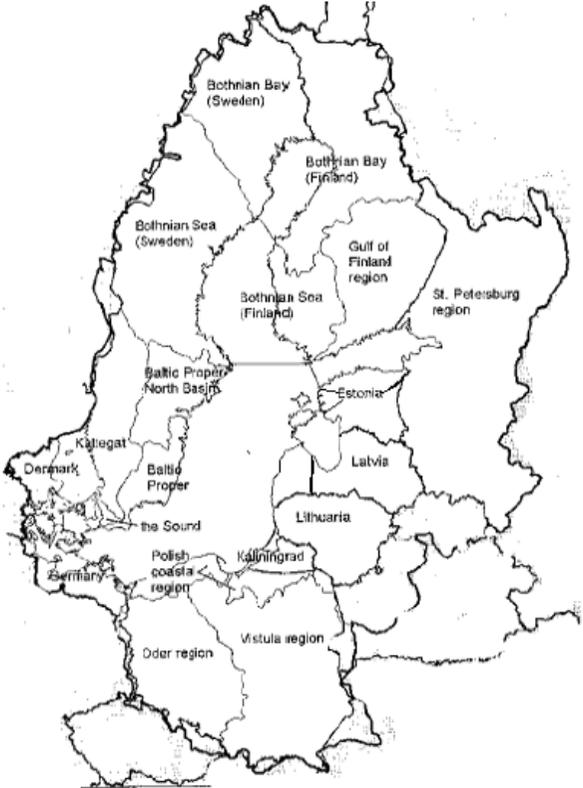


Figure 1. The Baltic Sea, sea regions and the drainage basins.

Because of the large scale of the Baltic Sea, MARE divides the entire Baltic into seven sea regions (basins) enabling the set-up of regional environmental standards for each of these sea regions. The division into sea regions reflects the fact that the emissions from each country contribute differently to the loads of the different sea regions because of retention, dilution and denitrification during transport to the sea regions, but also between the water bodies in the sea regions. This is because the cost-effectiveness of the measure should be related to the loads and not to the emissions. Much is already known about the effects of eutrophication in the Baltic Sea ecosystem and about the costs of measures to reduce the nutrient loads.

¹ The former version of the model consisted of 82 drainage basins in 9 countries. The drainage basins were formerly termed emitting regions.

The background for the development of this cost-effectiveness assessment tool is to build on this knowledge, and the aim of the modelling is to provide results indicating the optimal mix of measures and where to implement them, as no single measure or solution is valid for the entire Baltic Sea. The purpose of the cost minimisation model is therefore to establish a framework for prescribing cost-efficient scenarios of reduced nutrient loads to the Baltic Sea – including both nitrogen and phosphorus.

The cost-minimisation model

Nutrient loads derive from both airborne and waterborne emissions, but the model work has only focused on emissions from countries with coastlines adjacent to the Baltic Sea, or with water drainage and transport to this sea region. The main elements of the model comprise cost - and load functions. The present model is based on a previously developed model for the Baltic (cf. Söderquist 2002; Wulff et al 2003), and both model version are set to minimise the costs in the Baltic Sea to fulfil different environmental targets in the sea region. But the model itself has changed significantly compared to the former version. All data inputs of the model are updated, the cost functions applied are non-linear and the structure of the model has been changed in order to make the modelling more transparent and to provide facilities for easy updating. This also enables a transparent implementation of the estimated load functions based on the results from the drainage basin model.

The cost-minimisation problem is formulated as a choice of the cost-minimising mix of policy measures within drainage basins. The cost-minimising mix of measures consists of an optimal mix of measures and an optimal localisation of the measures, so that a stated goal for reductions of loads to one or more sea regions is obtained at the lowest total abatement cost. The minimisation problem is thus constrained by the limited potential of each policy measure. These limits on the potential emission reductions of the measures can be explained by the fact that each measure, e.g. wetlands, has a limited feasibility range within each drainage basin. Secondly, the implementation range should reflect that the estimates of costs and emissions reductions are coherent with the assumptions of prices, technology, etc.

The cost-minimization problem is a non-dynamic (static) problem described by:

$$\begin{aligned} \min_{x_{i,d}} \sum_{i=1}^n \sum_{k=1}^m TAC_i^k(x_{i,p}) \\ \text{st.} \\ \sum_{k=1}^m g_j^k(x_{i,p}) = T_j \\ h_i^k(x_{i,p}) \leq h_i^k \max \end{aligned} \quad \text{a)}$$

Where $x_{i,p}$ is the reduction of nutrient load, p is nutrient load (nitrogen, phosphorus) in each drainage basin, g_j is a function describing the share of reduced emissions emitted from drainage basin i reaching sea region j , and h_i is a function describing the extent of policy measure k implemented when reducing the emissions in drainage basin i . T_j is the target load reduction for sea region j , and $h_i^k \max$ is the maximum emissions reduction of policy measure k implemented in drainage basin i . A more detailed description of the cost-minimising model can be found in Schou et al (2006).

The measures incorporated in the revised model comprise wetland restoration, reduced fertiliser use, introduction of catch crops in agriculture, livestock reduction in agriculture, improved treatment of sewage and NO_x reduction. Results can be provided both for separate modelling of nitrogen reductions as well as phosphorus reductions, or by reductions in both nutrients.

The model is now implemented in the NEST model system.

References

- Söderqvist, T., 2002. Constructed wetlands and nitrogen sinks in Southern Sweden: An empirical analysis of cost determinants. *Ecological Engineering* 19, 161-173.
- Schou, J.S., Neye, S.T., Lundhede, T., Martinsen, L. & Hasler, B., 2006. Modelling Cost-Efficient Reductions of Nutrient Loads to the Baltic Sea. Model Specification, Data and Cost-Functions. National Environmental Research Institute. - NERI Technical report 592: 69 pp. (electronic). Available at: <http://www2.dmu.dk/Pub/FR592.pdf>.
- Wulff, F., Bonsdorff, E., Gren, I.-M., Johansson, S. & Stigebrandt, A., 2003. Giving advice on cost effective measures for a cleaner Baltic Sea: a challenge to science. *Ambio* 30(4-5), 245-259.

Implementation of measures to reduce nonpoint source loading of phosphorus at the catchment level

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Extended abstract

Eutrophication caused by an overabundance of nutrients (nitrogen and phosphorus) in bodies of water is one of the leading water quality issues in developed countries. Achievement of mandated water quality standards has increasingly focused on the role of nonpoint pollution source (NPS) discharges, in particular, runoff of nutrients from agricultural activities. Numerous abatement measures have been identified which could reduce nutrient losses coming from NPS. A market for tradable discharge permits (TDP) has the potential of achieving water quality targets in a cost-effective manner. Unfortunately, the attempts to start up permit markets that are able to exploit abatement cost differences between sources have not met with the success expected (EPA, 2001). Two of the reasons for the lack of success have been the problem of transaction costs and in the case of nonpoint sources, undefined property rights (Collentine, 2002).

The composite market design is a proposal for a TDP system that specifically includes agricultural NPS dischargers and addresses both property rights and transaction cost problems (Collentine, 2006). An integral component of the composite market model is the use of natural science models for calculating nutrient losses. The simulated quantification of losses allows these modelled values to be used for assigning limited property rights to NPS discharges. The structure of the composite market allows this system to be phased in over time with existing institutions and limited demands on financing.

This paper describes a study based on the application of a water quality trading program. The first part of the study describes how a measure to reduce phosphorus losses from agricultural practices (buffer zones along watercourses) can be used in the composite market model to calculate a selling price for tradable discharge permits in a sub-catchment area in southern Sweden. The second part describes how demand for permits in the same sub-catchment can be created through enforcement of current standards for phosphorus losses from private septic systems. Finally, the paper uses the modelled permit supply from part one and combines this information with the modelled demand from part two to illustrate how the composite market trading model can lead to the cost efficient reduction of phosphorus loads in a subcatchment.

References

- Collentine, D., 2002. Search for the Northwest Passage: the assignation of NSP (non-point source pollution) rights in nutrient trading programs. *Water Science and Technology* 45(9), 227-234.
- Collentine, D., 2006. Composite market design for a Transferable Discharge Permit (TDP) system, *Journal of Environmental Planning & Management* 49(6), 929-946.
- EPA (United States Environmental Protection Agency), 2001. *The United States Experience with Economic Incentives for Protecting the Environment*. EPA-240-R-00-001, EPA/ Office of Policy, Economics and Innovation, Washington, DC.

Mitigation of phosphorus and sediment: is there a cost-effective solution?

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Introduction

The EU Water Framework Directive introduced across Europe in 2003 requires governments to set water quality objectives based on good ecological status and includes specific requirements to control diffuse pollution. Given the pivotal role that phosphorus (P) and sediment play in influencing water quality, and that agriculture is thought to be responsible for 50% of the inputs to surface waters, controlling the movement of these diffuse inputs from land to water bodies will become increasingly important.

There are already a wide range of options for reducing soil erosion and the subject has received considerable research effort, however, there is significant potential to evaluate, modify, combine and improve the mitigation effectiveness of existing practices, particularly focused on reducing P losses using within field mitigation measures.

The Mitigation Options for Phosphorus and Sediment (MOPS) project (2005 to 2008) is investigating the cost effectiveness of specific control measures, representing different levels of farmer intervention, in terms of mitigating sediment and P loss from combinable crops.

Project outline

Three contrasting case study farms in England covering vulnerable soil types and slope forms are involved in the project to discover which preventative techniques are the most efficient. The three field sites are the Allerton Trust farm in Loddington, Leicestershire which is on clay soils, ADAS Rosemaund in Herefordshire which is on silt soils, and Severn Trent Water's farm at Old Hattons near Wolverhampton which is on sandy soils.

The mitigation options are focused on within field measures and include different cultivation techniques, vegetative barriers, tramline management and crop residue management. In the first year of the project six treatments were investigated at Loddington: ploughing up and down the slope, across the slope, and across the

slope with the establishment, within field, of a beetle bank along the contour; and minimum tillage up and down the slope, across the slope, and across the slope again with a contour beetle bank. At Rosemaund plots were established to examine losses within and between tramlines and specifically tramline wheeling disruption using a cultivator fitted with a ducksfoot tine to disrupt the compacted surface of the wheeling after its establishment in the late autumn. At Old Hattons plots were established to examine the management of post harvest cereal straw residues which had either been baled and removed or chopped and incorporated into the soil.

Cost effectiveness analysis

To determine the cost effectiveness of the different mitigation options data for each of the case study farms are being collected for each treatment in each year. This focuses on (i) field records on crop establishment, fertiliser and spray applications and harvesting and (ii) the additional costs associated with the mitigation options.

The analysis involves the construction of a simple spreadsheet model to examine impacts on individual cereal crop margins and thence the overall arable rotation. In the first instance, three farm level versions of the model have been developed to represent each of the three case study farms. The model includes both gross margin calculations and an 'operating' margin based upon labour and machinery costs which can be directly allocated to each crop enterprise. The operating margin goes beyond an enterprise gross margin as it includes some fixed costs, however, it is not a true net margin as certain building, land and general overhead costs are excluded.

The resultant gross and operating margins reflect the impacts of the different mitigation options on the costs of crop establishment and fertiliser and agro-chemical applications. Once impacts on individual crop margins have been calculated, the impact on the overall arable rotation can be determined. To do this, each crop margin is multiplied by the percentage area that is grown on the farm taken from the 2006 harvest year farm records. The resultant rotational farm operating margin for each of the mitigation options, and any additional capital costs associated with these, then need to be compared with data on the runoff, sediment and P loss to determine how effective and hence cost effective the options are. Data for this has been collected over the winter period, October through to March, when erosion risk and hence soil and diffuse P loss is at its highest.

Results

Table 1 shows the impact of the introduction of the various mitigation options on the operating margin. At Loddington, minimum tillage improves the operating margin, however, there are additional costs and implications from the introduction of the in-field vegetative strip. At Rosemaund and Old Hattons the additional tramline disruption process and straw incorporation activities reduce the operating margin.

Table 1. Mitigation options: additional costs and impact on margin.

Site	Mitigation option	Additional cost	Resultant operating margin
Loddington	Plough (control)	n/a	£215 per ha
	Contour plough	n/a	£215 per ha
	Contour plough with in-field vegetative strip	Year 1: £163/ha Each yr: £21/ha	£213 per ha
	Minimum tillage	n/a	£263 per ha
	Contour minimum tillage	n/a	£263 per ha
	Contour minimum tillage with in-field vegetative strip	Year 1: £163/ha Each yr: £21/ha	£261 per ha
	Rosemaund	Plough (control)	n/a
Tramline disruption		n/a	£186 per ha
Old Hattons	Plough (control)	n/a	£243 per ha
	Straw bale and removal	n/a	£242 per ha
	Straw chop and incorporate	n/a	£224 per ha

Initial results from the winter of 2005/06, which was quite dry compared to long term averages, indicate that at all case study farms tramlines are responsible for the majority of run-off, sediment and P lost, and that measures focused on this area as opposed to other within field measures may help in mitigating P losses. The results from Rosemaund show that, significantly, tramline disruption consistently and dramatically reduced run off and P fluxes to levels comparable to no-tramline areas. At the Loddington, the results also indicated that the use of beetle banks combined with contour cultivation could reduce runoff, soil and nutrient losses although this effect is not as clear as the difference between tramline and no-tramline areas. At Old Hattons, the results indicated that the treatments receiving 2.5t/ha straw chopped and incorporated consistently and substantially reduced surface run-off and total P loss per unit area compared with those where straw had been baled and removed.

The first year results present some potentially interesting solutions for the mitigation of P and sediment loss from arable cropping, however, and at this early stage, no concrete conclusions can yet be drawn. Further work is ongoing on tramline disruption, and contour and minimum tillage combined with in field vegetative. In terms of cost-effectiveness, the extrapolation of the case study data to generic farm typologies and from a farm level to regional basis is also being undertaken.

Assessment of water quality concerning nutrients in agricultural runoff

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Abstract

The paper deals with assessment of the surface water quality and includes recommendations for the water quality classification system based on nutrient concentrations. For EU Member States the overall aim of the Water Framework Directive is to achieve “good ecological status” and “good surface water chemical status” in all water bodies by 2015. Human activities including the use of the fertilizer and manure in agriculture have caused widespread increases in nitrate in shallow ground water and concentrations of nitrogen and phosphorus in streams (Clercq P.De. *et al.*, 2002; Baltic Marine Environment Protection Commission, 2005). Therefore agricultural runoff is main subject of the research.

Introduction

A water quality classification system is not well established in Latvian legislation. During the project ‘Transposition and Implementation of the EU Water Framework Directive in Latvia’ consultant company Carl Bro made recommendations for surface water quality classification, where small streams and drainage field outlets are not included.

Nutrient concentration data were collected at two agricultural monitoring sites – Berze and Mellupite – with measurement structures and sampling equipment. The Berze catchment is an intensive farming area, but Mellupite catchment could be considered typical in terms of nutrient application. Share of agricultural land in the Berze catchment is 98%, and in Mellupite it is 69% of the total area. The runoff measurements and water sampling were carried out in small streams and drainage field outlets.

Materials and methods

It may be concluded that both Berze and Mellupite monitoring sites represent typical agricultural areas in Latvia, therefore water quality data may be included in one data set. At the above-mentioned monitoring sites data were collected during the years 1994-2006. At field level 224 and at catchment level 259 water samples were collected and analyzed.

All available total nitrogen and total phosphorus concentration data were analyzed using normal distribution curves. According to Stergess formula, class intervals (c) depending on number of observations (n) can be calculated, $n < 100 \quad c = (X_{max} - X_{min}) * (1 + 3,32 \lg(n))^{-1}$ where X_{max} un X_{min} is minimal and maximal values (Arhipova I., Bāliņa S., 1999).

Percentile selections of data plotted as frequency distribution are used to establish water quality class boundaries: 10% high status, 10% - 25% good, 25% - 75% moderate, 75% - 90% poor, >90% bad status (Figure 1) (Buck S., Denton G. *et al.*, 2000; Cardoso A.C., Duchemin J. *et al.*, 2001; CIS Working group 2.3, 2003). The approach described above will produce criteria of greater scientific validity.

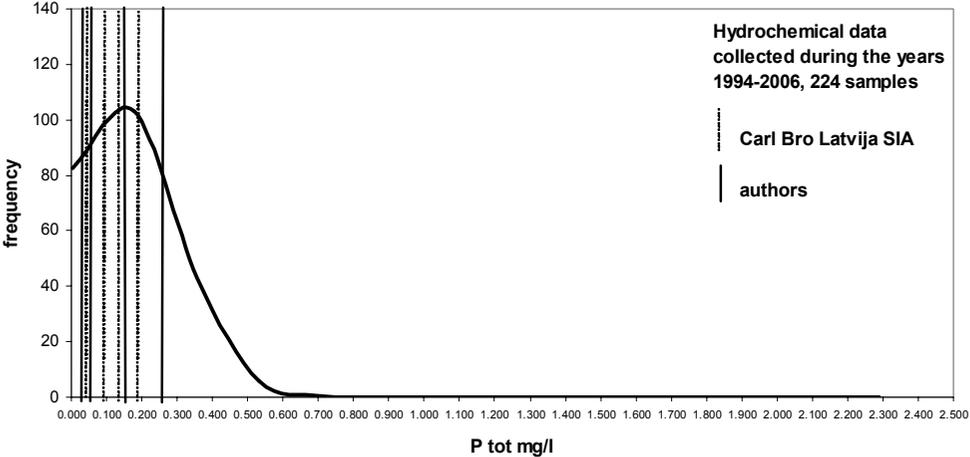


Figure 1. Evaluation of the small stream runoff.

Results

The consultants Carl Bro made recommendations for surface water quality classification based on professional judgments. It is obvious that Carl Bro recommendations cannot be used for evaluation of the runoff water quality from agricultural land. This conclusion is supported by results of agricultural monitoring data measured during year 1994 - 2006. The most important findings of the research are that water quality requirements for drainage water as well as for water in small streams in agricultural area should be less stringent (Table 1). Otherwise, it will not be possible to fulfil the objectives of the Water Framework Directive.

Table 1. Range of N and P values for water quality classification.

Quality class	P _{tot} (mg/l)		
	Agricultural area		Carl Bro
	Drainage water	Small streams	Small river
High	<0.015	<0.025	<0.045
Good	0.015-0.020	0.025-0.050	0.045-0.090
Moderate	0.020-0.075	0.050-0.150	0.090-0.135
Poor	0.075-0.135	0.150-0.250	0.135-0.180
Bad	>0.135	>0.250	>0.180

References

- Arhipova, I. & Bāliņa, S., 1999. Statistika ar Microsoft Excel 97 ikvienam. 1. daļa / Mācību līdžeklis. Rīga: Datorzinību Centrs, 22-25.
- Baltic Marine Environment Protection Commission, 2005. Nutrient Pollution to the Baltic Sea in 2000, Baltic Sea Environment Proceedings No.100, 3–4.
- Buck, S., Denton, G. et. al., 2000. Nutrient Criteria Technical Guidance Manual: Rivers and Streams, EPA, A27–A33.
- Cardoso, A.C., Duchemin, J., Magoarou, P. & Premazzi, G., 2001. Criteria for the identification of freshwaters subject to eutrophication. European Commission, Joint Research Centre, EUR 19810 EN, Luxembourg, 40-41.
- Clercq, P. De., Gertsis, A.C., Hofman, G., Jarvis, S.C., Neeteson, J.J. & Sinabell, F., 2002. Nutrient Management Legislation in European Countries, Wageningen, The Netherlands, 347 pp.
- Guidance on establishing reference conditions and ecological status class boundaries for inland surface waters, 2003. CIS Working group 2.3. (Reference conditions for inland surface waters), 42–45.

Critical phosphorus load of a stratified lake calculated by an ecological model

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Introduction

According to the Water Framework Directive (WFD), the lakes in EU must achieve as a minimum a “good ecological status” by the year 2015. The main biological indicators in lakes are abundance and diversity of fish, phytoplankton, macrophytes and invertebrates. Phosphorus is the main environmental stressor in Danish lakes and the primary determining factor for numerous biological variables. In an EU-Life project called Agwaplan (www.agwaplan.dk) we have determined the critical phosphorus load of lake Ravn in Jutland, DK. The objectives of the study were to calculate the effect of various scenarios of phosphorus load from the catchment area on biological and chemical indicators in the lake using an ecological model, and set up an action plan for reduction of phosphorus loss from agricultural areas in cooperation with the Danish Faculty of Agricultural Science, local farmers and agricultural advisers.

Study site

Lake Ravn (area 182 ha) is a deep (maximum depth 33 meters) lake stratifying in the summer time. It has been included in the Danish national monitoring programme since 1989 with annual sampling of plankton, macrophytes and water chemistry parameters in the lake and inlet streams. Mean (summer) concentration of phosphorus, chlorophyll and secchidepth is 25 µg P/l (total-P), 10 µg/l and 3-3,5 m, respectively. In warm periods the lake is often subject to blooms of very poisonous bluegreen algae. The main source of phosphorus load is agricultural activity and scattered dwellings without waste water purification.

Ecological model

According to Søndergaard et al. (2005) the lake cannot achieve a good ecological status, because the chlorophyll concentration is higher than 6.5 µg chl/a/l and maximum depth of submersed macrophytes is often less than 5 m. A hydrodynamic and ecological model was used (Danmarks Miljøundersøgelser og Århus Amt, 2007). Calibration and validation of the model showed a satisfactory description of physical and biological parameters. The model was able to describe the seasonal dynamics and therefore suitable for predictions of ecological status of the lake in various scenarios of external nutrient load. Data on phytoplankton biomass (carbon weight) were used to model a chlorophyll concentration reflecting the true phytoplankton

biomass, since chlorophyll is one of the present EU-intercalibrated indicators of ecological status in lakes.

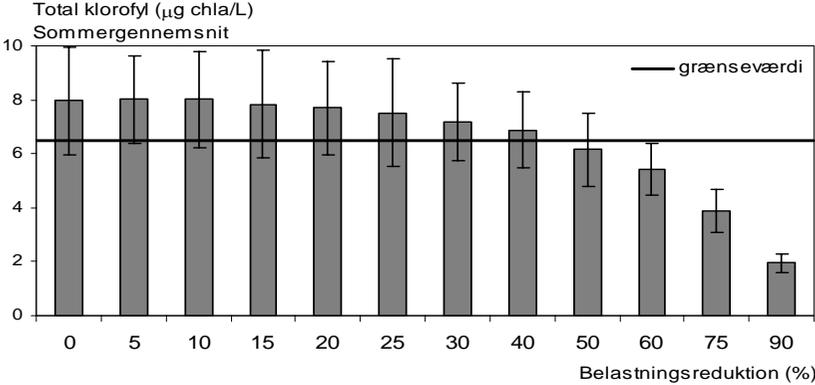


Figure 1. Calculated chlorophyll concentration (summer mean) after 0, 5, 10....90% reduction of external phosphorus load. The line represents good ecological status (6,5 µg chla/l).

Data from the calibration period (1992-2002) indicated a need for a 50% reduction of the average phosphorus load at that time in order to achieve 6,5 µg chla/l (figure 1). In this scenario the phosphorus concentration in the lake could be reduced to approx.15 µg P/l and the secchi depth increased to approx. 4 m enhancing the chance of a higher distribution of submerged macrophytes.

The present phosphorus load was calculated using data from 2000-2005 (figure 2).

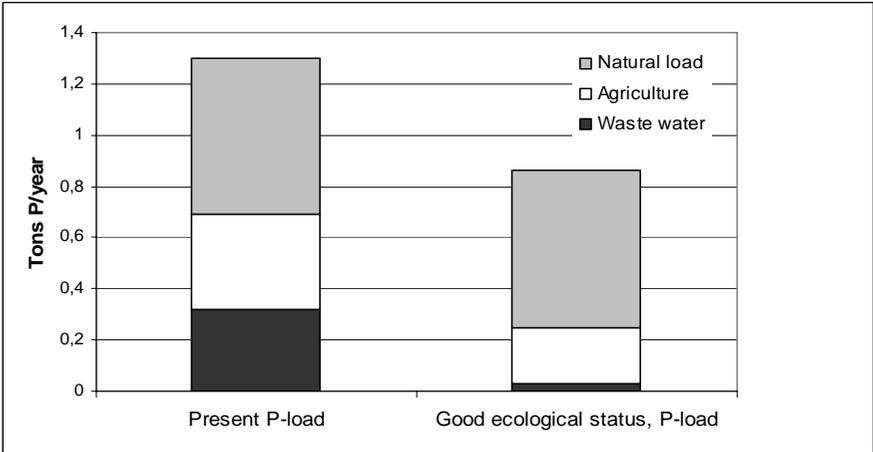


Figure 2. Present P-load and a calculated P-load resulting in a good ecological status. The colours of the columns indicate main P-sources.

The main P-source now is natural washout and erosion of phosphorus in the catchment area (reference condition), wastewater and diffusive washout and erosion from agricultural areas along the streams. The reduction target of phosphorus load in order to achieve a good ecological status in the lake is approx. 450 kg P/year, which may be achieved by improved wastewater purification and by focusing on fields in agricultural areas with a high phosphorus concentration in the soil and high risks of erosion. These fields have been mapped and analyzed (texture, drainage conditions, slope, etc.) and meetings with local farmers using GIS data and a manual of good agricultural practice was held in order to minimize the future manuring of the vulnerable spots.

Conclusions

Chemical and biological indicators in a lake can be satisfactorily described using a hydrodynamic and ecological model. It is an important tool for calculations of the required reduction of nutrient load in order to obtain a good ecological status in 2015 according to the WFD. The limiting factor may be input data for the model. Monitoring data from several years, both from the lake and inlet streams area, are necessary due to natural variations of precipitation, temperature, etc. The present P-load should be based on data from more than one year, thus avoiding unusual situations with a high or low transport of phosphorus in the streams. Alternatively, the measured load of phosphorus could be adjusted to a standard year with an average precipitation. The sources of phosphorus are determined using measured or empiric data on wastewater treatment plants, scattered dwellings and reference concentrations in streams (40 µg P/l in this case). The contribution from agriculture cannot be measured but is calculated by subtracting the other sources from the total load. This approach gives an element of uncertainty, and deviations of the true size of natural load and phosphorus loss from scattered dwellings may lead to inexact conclusions on the diffuse contribution from agriculture.

The final indicator boundaries between good ecological status and moderate ecological status in various types of lakes will be identified in the EU- intercalibration network and published at the beginning of 2008.

References

- Danmarks Miljøundersøgelser og Århus Amt: Modellering af scenarier for næringsstofftilførselens påvirkning af den økologiske kvalitet i Ravn Sø. Dennis Trolle. (In Danish).
- Søndergaard, M., Jeppesen, E., Jensen, J.P. & Amsinck, S.I., 2005. Water Framework Directive: ecological classification of Danish lakes. *Journal of Applied Ecology* 42.

River sediments as a source of soluble reactive phosphorus in a mixed land use river system

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Aims - to investigate (i) whether river sediments were potential sources or sinks of soluble reactive phosphorus (SRP) to freshwaters, (ii) how sediment water interactions control river SRP concentrations over a range of flow conditions during summer, a critical period for P effects on ecosystems, and (iii) to compare sediment and soil P sorption properties.

Sampling and methods

The River Dee (NE Scotland) has montane, forestry and agricultural land use, acidic glacial soils, low population density and is an important aquatic habitat (Natura 2000). Water quality in certain tributaries and receiving waters is at risk from agricultural nutrient inputs. We sampled water and river bed sediments from the main stem and tributaries covering a range of agricultural to semi-natural land use (Table 1) over contrasting flow conditions during two summers: (i) May-04 (peak of a storm event), (ii) Aug-04 (extended baseflow period), and (iii) May-05 (flow event recession after cultivation period). Agricultural soils were also sampled (catchments 1, 4, 5).

For bed sediments and soils sieved to <250 μm we determined geochemical and physical properties describing P dynamics. We used extractions (acid ammonium oxalate extractable Fe_{ox} , Al_{ox} , P_{ox} , total Ca, FeO paper strip test desorbable P) and batch isotherms (0-20 μM $\text{PO}_4(\text{P})$ in 0.01M CaCl_2 , at 5°C) and calculated the following to simply model surface - solution P exchange:

(i) Equilibrium P concentration (EPC_0 , the solution concentration at which no net adsorption/desorption occurs) derived from isotherm data according to $\Delta n_a = K_d c - n_i$, where K_d is the distribution coefficient, n_a and n_i the adsorbed and original (FeO paper test) amount of P per mass of sediment/soil, c is the equilibrium concentration of SRP, then $\text{EPC}_0 = c$ when $\Delta n_a = 0$ (House & Denison, 2002).

(ii) The molar ratio, $Z = \text{P}_{\text{ox}} / 0.5(\text{Fe}_{\text{ox}} + \text{Al}_{\text{ox}})$, the P saturation of poorly-crystalline complexes comprising exchangeable P in acid soils (van der Zee et al., 1988).

(iii) $\Delta\text{P} = \text{sediment } \text{EPC}_0 - \text{river } [\text{SRP}]$ (+ve values denote P release from sediments).

Results

Spatio-temporal variability

Sediment properties of n_i , EPC_0 , K_d , Z , organic C, total Ca and pH were small for low intensity land use tributaries and upstream sites on the main stem (8-11) but increased downstream on the main stem (12, 13) and were greatest for lowland agricultural (sites 1-7) (Table 1, Fig. 1). Extreme concentrations of n_i , EPC and lowest K_d values were found for catchment 7 which contains a sewage treatment works (1200 population). Greater values of n_i , EPC_0 and sediment organic contents occurred during a period of prolonged baseflow (Aug-04) and may indicate biologically-associated P. Temporal mean Z correlated with logit transformed area of intensive grazing ($r=0.81$, $p<0.001$) and mean stream pH ($r=0.88$, $p<0.001$).

Table 1. (i) Catchment attributes, temporal mean water and bed sediment physico-chemical properties, (ii) properties of representative agricultural soils.

Site	Catchment area (km ²)	% area of catchment land use		River pH	River SRP ($\mu\text{g l}^{-1}$)	Soil / sediment pH _{water}	Al _{ox} + Fe _{ox}	Total Ca	Org C	Z	% < fine silt
		Pasture + arable	Moor + montane				(mg kg ⁻¹)				
(i) Catchments and temporal mean characteristics of river waters and bed sediments											
1	4	46	33	7.49	10	6.48	11.5	16	15	0.18	32
2	31	67	15	7.51	32	6.40	8.9	20	20	0.17	35
3	35	32	9	7.16	5	6.21	8.8	11	28	0.14	37
4	37	63	15	7.53	14	6.42	10.6	22	13	0.16	20
5	51	60	18	7.49	11	6.54	6.3	31	5	0.22	19
6	58	63	12	7.46	15	6.57	5.4	27	5	0.19	14
7	71	56	17	7.48	41	6.35	6.1	11	13	0.20	23
8	94	2	77	6.50	2	5.93	6.6	11	7	0.08	24
9	212	10	75	6.70	4	5.63	9.9	6	5	0.07	40
10	1005	3	84	6.84	2	6.03	4.6	13	8	0.10	16
11	1348	9	73	6.97	2	6.23	2.5	9	6	0.08	25
12	1775	13	67	7.05	3	5.97	4.7	18	6	0.17	15
13	2005	19	60	7.13	3	6.04	13.7	11	42	0.12	30
(ii) Spatial mean characteristics of soils											
						5.03	12.5	26	41	0.14	38
						5.88	15.4	26	34	0.24	51
						5.76	16.1	28	20	0.11	34

Are sediments sources or sinks of P to waters?

In 5, 10 and 18 cases ΔP values were respectively categorized as <-5 , -5 to 5 and >5 $\mu\text{g P l}^{-1}$. Hence, sediments were dominantly classed as sources of SRP to waters. Three main stem sites downstream of agricultural tributaries showed large positive ΔP values due to strongly desorbing sediments and small river SRP concentrations. During baseflow (Aug-04) bed sediments were potentially strong sinks for P at sites 2, 4 and 6, where diffuse pollution pressures are greatest (arable + intensive grazing $>63\%$).

Comparing properties of sediments and soils

Smaller K_d values for bed sediments than soils suggested that SRP was less tightly sorbed to sediments. This may be attributed to loss of finer, reactive soil particles from bed sediments, or P desorbed in the change from acid soil conditions to higher

pH river water. Fig. 1e shows how the potential for P release to solution (EPC_0) varies with surface P saturation ($>Z$). In Aug-04 a sharp increase in EPC_0 for $Z>0.12$ is similar to the 'change point concept' applied to soils. The three outliers to this relationship were explained by including sediment total Ca concentrations ($EPC_0 = -30.7 + 718 Z - 0.003 Ca$; $R^2=0.87$, $p<0.001$) which may indicate stronger P adsorption related to clay contents of more basic soils. During highflow (May-04) sediment EPC_0 were smaller and Z values larger than at other times and sediment properties became similar to arable soils.

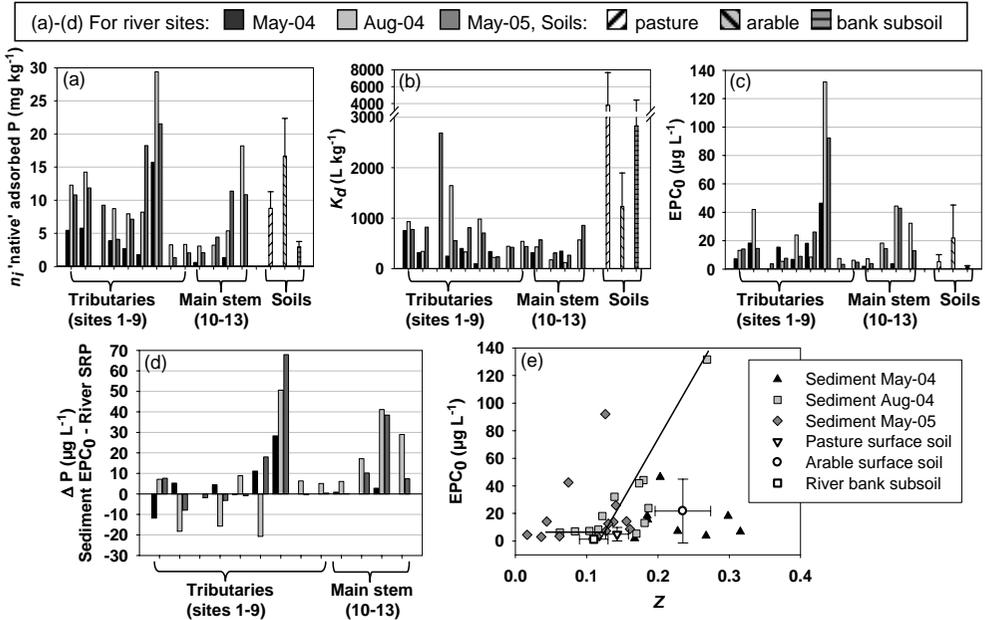


Figure 1. Sorption properties of (a) n_i , (b) K_d , (c) EPC_0 for river bed sediments and soils, (d) ΔP values for sediments (sites as in Table 1), and (e) EPC_0 against Z .

Conclusions

The sediment native adsorbed P (n_i) and potential to desorb P (EPC_0) was enhanced in tributaries with diffuse pollution pressures and to a greater degree in the presence of a point source. At the majority of sites sediments were potential P sources to the water column. Main river sites downstream of polluted tributaries had accumulated releasable inorganic P in bed sediments. With the exception of much larger K_d values, bed sediments showed similar properties to agricultural topsoil, especially during high flows. Increased potential P desorption and native P reserves for bed sediments during prolonged baseflow suggested in-situ accumulation of P, likely by biotic as well as geochemical mechanisms. These simplistic methods of modelling P desorption potential may be further refined by incorporation of the effects of biotic processes, redox and changing water chemistry in P modelling approaches.

References

- House, W.A. & Denison, F.H., 2002. Exchange of inorganic phosphate between river waters and bed-sediments. *Environ. Sci. Technol.* 36, 4295-4301.
- Van der Zee, S.E.A.T.M., Nederlof, M.M., van Riemsdijk, W.H. & de Haan, F.A.M., 1988. Spatial variability of phosphorus adsorption parameters. *J. Environ. Qual.* 17, 682-688.

Impacts of agricultural land-use practices upon in-stream ecological structure and processes

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Introduction

Phosphorus loss in run-off and drainage from agricultural land has been linked to eutrophication problems in a number of freshwaters. Many farming systems and land management practices have been indicated as those most likely to lead to high phosphorus losses at field and farm scale. However, it is unclear exactly what level of phosphorus loss is necessary to cause a significant eutrophication response in receiving streams, or the speed with which the response occurs.

The objective of this research is to quantify the influence of phosphorus concentration on the ecology of the lowland streams by measuring changes in structure of macroinvertebrate and diatom communities and by measuring changes in ecological processes. The stream waters of three paired headwater catchments in lowland England and Wales, which are characteristic of selected livestock and farming systems with variable P-loss risk, provide a gradient of phosphorus concentrations from limits of detection to highly eutrophic (median concentrations for all data collected ranging from 10 to 240 $\mu\text{g-SRP/L}$ and from 22 to 334 $\mu\text{g-TP/L}$). Ecological structure and processes were measured in comparable meso-habitats (functional habitats) at each site to minimise physical differences. Invertebrate community composition and diatom species diversity with chlorophyll 'a' biomass were the structural measurements; photosynthesis, respiration, phosphatase activity and leaf decomposition the functional measurements.

Results

Structure

The main structural change in macroinvertebrates was the decrease in number of genera as P concentration increased (28 genera $\leq 7 \mu\text{g-TP/L}$ to 14 genera at 421 $\mu\text{g-TP/L}$; mean P values are hereafter shown from the sampling period of the ecological data). This trend was clear in the spring and less visible in summer and autumn but still could be measured. Diptera were the dominant taxonomic group in all sampling sites where TP concentration was above 120 $\mu\text{g/L}$ and the highest P sites always had high numbers of individuals of the family Chironomidae (e.g. a median density 2.10 ind/cm² at 181 $\mu\text{g-TP/L}$, compared with 00.4 ind/cm² at 55.1 $\mu\text{g-TP/L}$). Other

changes in taxonomic composition were a higher proportion of Crustacea in streams as P concentration reduced; high numbers of Ephemeroptera in streams with TP below 130 µg/L and Plecoptera only in sites with TP below 70 µg/L. The number of Coleoptera decreased in concentrations above 170 µg/L.

Numerical analyses (Average Linkage and Euclidean Distance) grouped the sites into 4 categories: 1 TP < 55.1 µg/L, 2 TP 55.2 to 93.2 µg/L, 3 TP 93.3 to 184.2 µg/L and 4 TP 184.3 to 421.5 µg/L.

The main change in diatoms was decrease in diversity with elevated phosphorus concentrations. Once concentrations exceeded 100 µg/L, the number of species dropped to 10 per site. With higher P (250 µg/L), pollution tolerant species especially *Navicula lanceolata* and *Amphora pediculus* approached 90% of total count (%PTV). In such cases the Trophic Diatom Index showed high values (above 50) indicating higher levels of nutrients and the GDI (Generic Diatom Index) showed low values around 6-7 indicating heavy pollution. Measurements of chlorophyll 'a' were higher with elevated P. Chlorophyll 'a' biomass showed no relationship with TP below 80 µg/L but after this biomass increased rapidly from 1.5 to 5 µg/cm² at 250 µg-TP/L.

Processes

There was a clear increase in community respiration with increasing P concentration. Respiration ranged from 0.36 mg O₂/L at Digby Farm; the control site for the Welland catchment with average 25.8 µg-TP/L to 2.54 mg O₂/L at Priors Farm; the impacted site in river Avon catchment with 421 µg-TP/L. Net photosynthesis also displayed a clear increase with P, from 0,14 mg O₂/L in Lewston Mill – the control site of the Avon catchment, to 5,69 mg O₂/L in Priors Farm the impact site.

Phosphatase activity showed negative relationships with increasing phosphorus concentration at the three study catchments. There was always low phosphatase activity in high-phosphorus sites (>100 µg-TP/L), never exceeding 80 mmol/cm²/1/2h. Leaf breakdown was fastest in the control sites, and observations suggest that this is due to the lower quantity of fine sediment, which blankets the leaves.

Discussion

Results shows that P concentration in land runoff, caused by low, medium and high intensity farming systems and other rural P sources have direct impact on the ecology of streams.

Agriculturally-impacted catchments maintained higher P concentrations in streams than catchments with a low agricultural intensity, which was reflected in clear differences in the magnitude and diversity of in-stream plant and animal communities as well as in changes in ecological processes along the P gradient.

Some of the results suggest that differences in biodiversity and/or ecological processes may be caused by various sediment concentrations in the water column.

Results suggest that significant and clearly noticeable changes in stream ecology occur in P concentrations above 100 µg/L TP with 80 µg/L being an alarm point where the first symptoms can be observed as:

- Decrease in diversity of macroinvertebrates and benthic algae taxonomic groups (disappearance of highly-valued groups like Plecoptera)
- Increase in number of Diptera with Chironomidae as the dominant family
- Increase in algal biomass
- Increase in ecosystem Respiration
- Increase in Photosynthesis
- Decrease of Phosphatase activity

Quantitative eutrophication risk assessment model of phosphorus from different anthropogenic sources

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Introduction

Environmental Risk Assessments tools (ERA) are designed to estimate the magnitude and likelihood for adverse effects resulting from a human activity. They involve parallel assessments of hazards (identification of adverse consequences and setting the expected effects) and environmental exposure. Then both assessments are combined in the risk characterization to assess the probability of each hazard to occur. Currently regulatory ERA's are mostly based on low-tier deterministic approaches focusing on the exceedance of certain triggers (ECB, 2003). However the scientific developments now permit high-tier probabilistic approaches with more realistic quantifications of the expected effects. A probabilistic risk assessment approach has been developed to quantify the potential eutrophication risk of Total Phosphorus concentrations (TP) in $\mu\text{g l}^{-1}$, and the contribution associated with different anthropogenic sources (see De Madariaga & Tarazona, 2007).

Exposure estimation

A simplified river basin exposure model was developed to estimate environmental TP based on inputs of population density, river flow, landuse areas, and export coefficients. It is possible to develop exposure scenarios with generic values to estimate regional predictions of TP, or develop more local estimations. Comparisons of predicted *versus* observed TP showed good results. However, for very large catchments the results were poorer, mainly at the river's mouth stations, because the model does not take into account in-lake retention processes that reduce TP in the river waters. In order to achieve better exposure estimations, river specific models and/or monitoring data can be used instead to predict the risk outcome.

Effect assessment

The effect assessment part was covered through a probabilistic approach based on a meta-analysis of peer-reviewed field data of inland lentic waterbodies affected or not by anthropogenic eutrophication. The criteria to assess eutrophication impacts developed under the Common Implementation Process of the Water Framework Directive (2000/60/CEE) were the leading conceptual considerations to determine, whether a waterbody was affected or not by eutrophication in one-year period (EC, 2006). For that purpose a database of 300 different situations was developed collecting relevant information on biological aspects of different European inland freshwater

ecosystems and the annual average TP. A classification of these waterbodies was done, resulting in two datasets of TP in both kind of possibilities, affected or not by eutrophication, i.e. fulfilling or not the good quality status criteria set by the WFD. Both datasets were implemented into two conditional probabilistic distributions of TP with a Monte Carlo Analysis. These conditional distributions, $p(\text{TP}|\text{G}+)$ and $p(\text{TP}|\text{G}-)$, represent the distribution of probabilities of TP in water bodies with good (G+) or less-than-good status (G-). The datasets provided the basis for a generic European risk characterisation. The datasets were also broken down into Atlantic & Central Europe ecosystems and Mediterranean ecosystems, and into shallow and deep water bodies allowing to obtain different distributions that can be used for different risk characterisations. This methodology can also be used to obtain distributions based on a single country or ecological region or a single water body data. Risk predictions at finer scales may be more realistic.

Risk characterization

The eutrophication risk associated with each estimated exposure TP is defined as the likelihood of a sensitive site (vulnerable to be affected by eutrophication) to be less-than-good status. Mathematically, it is defined as $p(\text{TP} \cap \text{G-})/(p(\text{G-})\text{max})$, i.e., the joint probability for being in less-than-good status and having a particular TP corrected by the percentage of sensitive sites in the ecological region. The risk outcome is finally presented as a range of percentage values, the upper bound will be estimated from the $p(\text{TP}|\text{G}+)$ distribution; the lower bound, from $p(\text{TP}|\text{G}-)$. By the implementation of the exposure estimations with a probabilistic approach (input variables are distributions not deterministic values) the risk is finally estimated as joint probability distributions that determine a range of probabilities of risk for a particular management scenario. To compare the risk outcomes of different management scenarios it is necessary to know the specific contribution to the TP pool of the different P sources.

Conclusions

The methodology offers quantitative comparisons for the eutrophication risk associated to different P sources, providing information for the cost/benefit analysis when selecting the best measures to control eutrophication symptoms in freshwater ecosystems at catchment scale. We present an example of the use of this model for risk assessment focusing on the major anthropogenic P sources. Based on probabilistic distributions for the consumption of P-based detergents, human metabolic releases, export coefficients, P-removal at the sewage treatment plant and catchment area *versus* river flow relationships; and with the assumption that in-lake sedimentation processes are negligible (worst-case situation) a quantitative estimation was conducted using the Mediterranean effects assessment scenario. Figure 1 shows several management options each plotted by a single colour. Black curves represent the total eutrophication risk, with the total contribution of diffuse and point sources includ-

ing P-based detergents. Red curves represent the risk without detergents, i.e. diffuse plus point sources, without detergents contribution. Green curves represent the total risk in a scenario of 20% P-reduction of diffuse releases (e.g. improving agricultural practices). Blue curves represent the eutrophication risk in a scenario of 70% P-reduction in point sources contribution through WWTP (e.g. implementing P-removal technologies in urban areas). The results suggest that the implementation of P-removal technologies at the sewage treatment plans is required regardless the use of P-based or P-free detergents.

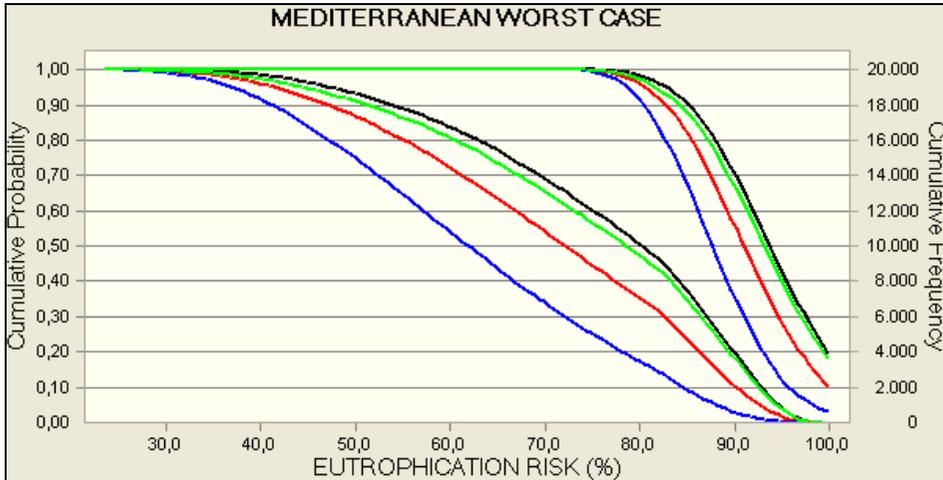


Figure 1. Joint probability curves of different management options in a Mediterranean worst-case scenario.

References

- De Madariaga, B.M. & Tarazona, J.V., 2007. Development of an European quantitative risk assessment of polyphosphates in detergents. Model implementation and quantification of the eutrophication risk associated to the use of polyphosphates in detergents. INIA (Spanish National Institute for Agriculture & Food Research & Technology)/Green Planet Research Report GPR-CEEP-07-1.
- European Chemical Bureau (ECB), 2003. Technical Guidance Document in support of Commission Directive 93/67/EEC on Risk Assessment for new notified substances, Commission Regulation (EC) No 1488/94 on Risk Assessment for existing substances and Directive 98/8/EC of the European Parliament and of the Council concerning the placing of biocidal products on the market. European Chemical Bureau, Ispra, Italy.
- European Commission (EC), 2006. Towards A Guidance Document on Eutrophication Assessment in the Context of European Water Policies. Common Implementation Strategy for the Water Framework Directive (2000/60/EC).

Characterisation of colloidal material in soil suspensions and agricultural runoff waters

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Introduction

Colloidal material (0.001-1 μm) in soil leachate and agricultural runoff waters is an important route for the transport of contaminants from land to catchments. Traditionally environmental samples have been filtered using the water industry 'standard' threshold of <0.45 μm to separate the dissolved and particulate material; however this has led to information regarding colloidal material that spans this threshold being 'lost' (Gimbert *et al.*, 2005). To be able to characterise the colloidal material in the 0.1-1 μm range a relatively mild separation technique called flow field-flow fractionation (FIFFF) can be used to determine particle size distributions (PSDs).

FIFFF separates molecules or particles in complex matrices using a crossflow field and can be used to obtain information on particle size distributions in complex environmental matrices over the entire colloidal size range. The output from the instrument is a fractogram of UV absorbance (254 nm) against elution time and this is converted to a plot of relative mass against particle size. The aim of this study was to characterise the particle size distribution of soil suspensions and agricultural waters using FIFFF and laser sizing.

Experimental

Soil was sampled from one drained plot (plot 9) and one undrained plot (plot 10) at different depths (0-2, 10-12 and 30-32 cm) along a transect from the Rowden Experimental Platform (RERP). This was to determine whether there was any change in the size profiles between drained and undrained plots, and also at depth. The soil was prepared in suspension form using an optimised sample preparation method (Gimbert *et al.*, 2006). Briefly, the soil was dried and sieved to <2 mm then <63 μm before 1 % m/v soil suspensions were prepared and shaken overnight. The suspensions were then settled for 1 h and the top 20 mL pipetted out as this layer contained the <1 μm particles. The samples were then injected (20 μL) into the FIFFF. As a complementary particle sizing method, the same soil samples (sieved <2mm) were analysed using laser sizing. This was to give added information about the samples as FIFFF determines the size profile for the 0.06-1 μm particles, whereas laser sizing can determine size information on particles 1-2000 μm therefore

generating more detailed information for a wide range of particle sizes (Gimbert *et al.*, 2007).

FIFFF was also used to determine the particle size distribution of colloidal material in runoff waters during storm events. Samples were collected and transported back to the lab within 12-24 h of collection, shaken for 10 min, settled and the <1 μm fraction extracted after 1 h. This was to determine whether the colloidal profile changes during a high energy event which has implications for phosphorus which is associated with the colloidal material.

Results and discussion

The results showed similar PSDs for surface samples (0-2 cm) along the transect but significant heterogeneity for sub-surface samples (30-32 cm), resulting in bimodal distributions with a predominance of particles in the size range 0.4-0.6 μm . There was also more variability in the drained plot samples than the undrained plot samples with significant variability in the particle size distribution over short distances (20 m). This was possibly due to the presence of mole drains at 30 cm altering the soil characteristics (Figure 1). The laser sizing data showed less variability with a mean particle size of about 8 μm .

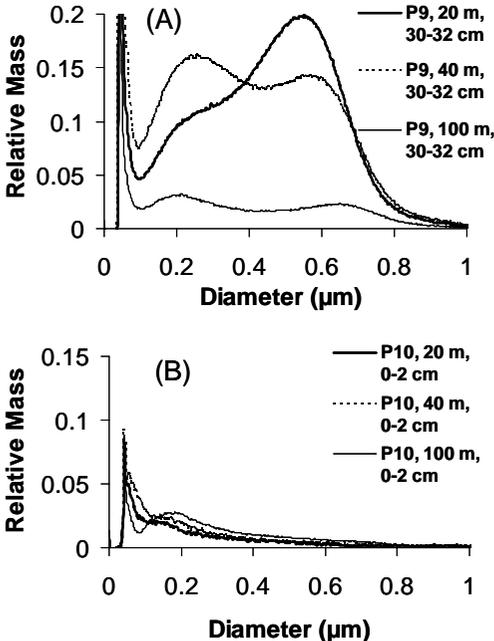


Figure 1. Comparison of particle size distributions between: (A) Drained plot 9 where soil was sampled at different distances across the transect at 30-32 cm depth; (B) Undrained plot 10 where soil was sampled at different distances across the transect at 0-2 cm depth.

FIFFF was also used to obtain the PSDs of agricultural runoff waters from Denbrook collected during storm events. The colloidal profile determined using FIFFF shows how the profile changes during a storm event with a peak maximum at 0.2 μm (Figure 2).

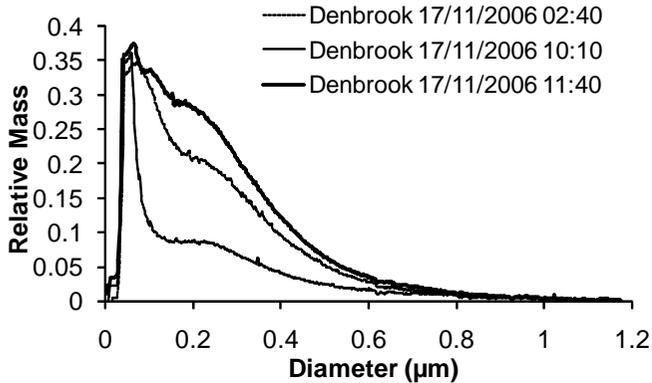


Figure 2. FIFFF particle size distributions for agricultural runoff waters sampled during a storm event.

Conclusions

FIFFF has been shown to be a suitable technique for characterising the colloidal material in soil suspensions and agricultural runoff waters. It has the potential to be combined with sensitive spectrophotometric detection to determine the distribution of molybdate reactive phosphorus in the colloidal material. This will improve our understanding of the association of phosphorus with the colloidal material as a potentially significant pathway for its transport from agricultural land to catchments.

References

- Gimbert, L.J., Haygarth, P.M., Beckett, R. & Worsfold, P.J., 2005. Comparison of centrifugation and filtration techniques for the size fractionation of colloidal material in soil suspensions using sedimentation field-flow fractionation. *Environ. Sci. Technol.* 39, 1731-1735.
- Gimbert, L.J., Haygarth, P.M., Beckett, R. & Worsfold, P.J., 2006. Partitioning and stability of engineered ZnO nanoparticles in soil suspensions using flow field-flow fractionation. *Environ. Chem.* 3, 184-191.
- Gimbert, L.J., Worsfold, P.J. & Haygarth, P.M., 2007. Processes affecting transfer of sediment and colloids, with associated phosphorus, from intensively farmed grasslands: colloid and sediment characterization methods. *Hydrol. Proc.* 21, 275-279.

Effect of flow-pathways on leaching of dissolved and particulate phosphorus from the plough-layer

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Introduction

Particle- or colloid-facilitated transport is generally recognized as an important mechanism for phosphorus leaching from agricultural soils (e.g. Stamm et al., 1998; Laubel et al., 1999; Schelde et al., 2006). Several investigations have demonstrated that macropores may provide preferential pathways for surface-applied solutes and suspended colloids. However, less is known about the actual role of preferential flow pathways on the *in situ* mobilization of colloids and phosphorus. Mobilization and leaching of *in situ* phosphorus depends mainly on the degree of phosphorus saturation and colloid dispersibility, as well as the spatial distribution of the soil-bound phosphorus relative to the soil specific flow patterns. Preferential flow may increase the leaching of phosphorus located at the preferential flow paths, while leaching of phosphorus located within the soil matrix may decrease during preferential flow conditions. This study was conducted to determine the influence of flow pathways resulting from changing soil-water potential and precipitation intensity on the leaching of total dissolved P (TDP) and particulate P (PP).

Leaching experiment

Intact soil cores (10 cm in diameter and 8 cm length) were collected at two sites (12% and 18% clay) along a naturally occurring clay gradient of an arable field at Lerbjerg, Denmark. Both sites have high contents of total P (1116 and 854 mg kg⁻¹) and very high degrees of phosphorus saturation (DPS ~ 61% and 51%) for 12 and 18% clay, respectively. Further details on soil characteristics and sampling strategy are described in Kjaergaard et al. (2004a). Leaching of colloids, TDP and PP was investigated from intact soil cores at different soil-matric potentials (ψ) (0, -5, -10 hPa) and different irrigation intensities IR (1 and 10 mm hour⁻¹). The leaching experiments were conducted in an experimental system developed for colloid leaching experiments at unsaturated and saturated conditions (Kjaergaard et al. 2004a,b). A constant suction of 0, 5 or 10 hPa was applied to the lower boundary of the soil columns. Soil columns were irrigated with an electrolyte solution, having a chemical composition similar to natural rainwater. A ³H₂O-tracer experiment was performed on all columns at steady outflow. Effluent samples were analyzed for turbidity, pH, electric conductivity, ³H₂O, total P (TP) and TDP. TDP was defined as

the P present in the supernatant after centrifuging, yielding a lower cut-off particle diameter of 0.1 μm . All experiments were performed in triplicate.

Effect of flow-pathways on phosphorus leaching

The flow pattern was strongly affected by soil type and flow conditions. At 12% clay and low irrigation intensity ($IR = 1\text{ mm hour}^{-1}$), tracer breakthrough curves (BTC) were characterized by matrix-dominated flow behaviour at unsaturated conditions ($\psi = -10\text{ hPa}$). Saturation ($\psi = 0\text{ hPa}$) only resulted in slightly asymmetric BTCs, while increasing the IR markedly increased the degree of preferential flow. The flow pattern significantly affected the mobilization and leaching of colloids and PP at 12% clay (Figure 1a,b,c). Saturation increased colloid and PP leaching (Figure 1b) compared with unsaturated conditions (Figure 1c) as a result of a larger active flow volume and minimum filtering in the larger pores. Increasing the IR at saturation, however, decreased colloid and PP leaching (Figure 1a), probably as a result of a smaller active flow volume and a reduced diffusive exchange between immobile and mobile flow domains. Phosphorus leaching was dominated by TDP, and there was no clear response of flow patterns on the leaching of TDP, which was probably a result of the very high DPS enhancing equilibrium concentrations of dissolved P.

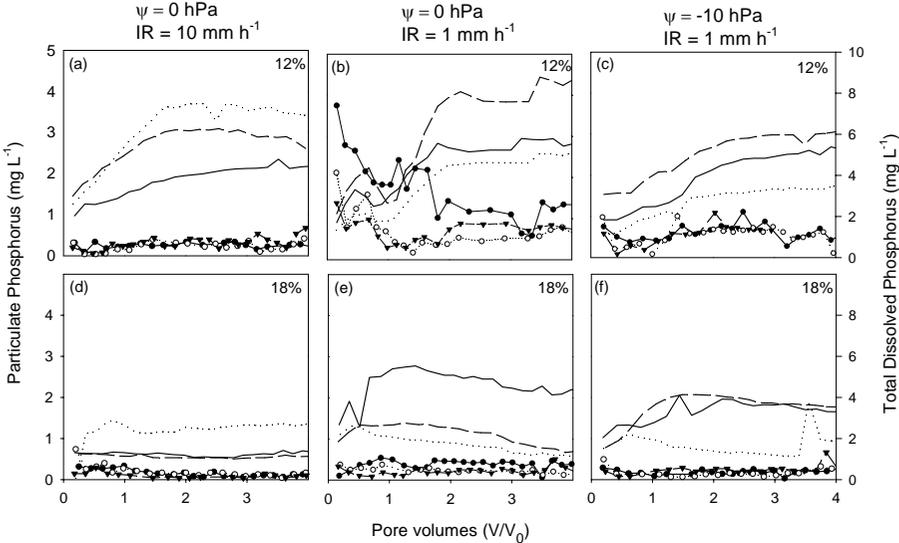


Figure 1. Effluent concentrations of particulate P (lines with markers) and total dissolved P (lines) against number of eluted pore volumes (V/V_0) as a function of matric potential (ψ) and irrigation intensity (IR) for 12 and 18% clay. Each line and marker represents a replicate column.

At 18% clay, flow was always preferential with multiple peaks reflecting at least two mobile flow domains. Increasing tension resulted only in minor changes in tracer-

BTCs, while increasing IR resulted in faster breakthrough of tracer. These flow patterns were additionally reflected in the mobilization and leaching of colloids and PP, with no effect of increased tension, while increasing IR at saturation resulted in decreased colloid and PP leaching (Figure 1,d,e,f). Again, leaching was dominated by TDP, and there was no clear response of flow-patterns on the leaching of TDP, which is attributed to the very high DPS. The larger mobilization and leaching of colloids and PP at 12% compared to 18% clay, was probably a result of a higher inherent colloid dispersibility, and a larger active flow volume increasing *in situ* colloid mobilization (Kjaergaard et al., 2004b). The larger leaching of TDP from 12% compared to 18% clay is attributed both to the higher DPS, and the larger active flow volume at 12% clay.

Conclusions

The results demonstrated the importance of the active flow volume and diffusive exchange of high-ionic strength intra-aggregate water with low-ionic strength infiltration water facilitating colloid dispersion, and additionally the diffusion of colloid-bound phosphorus from immobile to mobile flow domains. The leaching of phosphorus in the present study was significantly dominated by TDP, which was not affected by the prevailing flow conditions. This was, however, attributed to the very high DPS, and it must be expected that leaching studies on soils with low DPS would yield different results.

References

- Kjaergaard, C., Poulsen, T.G., Moldrup, P. & de Jonge, L.W., 2004a. Colloid mobilization and transport in undisturbed soil columns. I. Pore structure characterization and tritium transport. *VZJ* 3, 413-423.
- Kjaergaard, C., Moldrup, P., de Jonge, L.W. & Jacobsen, O.H., 2004b. Colloid mobilization and transport in undisturbed soil columns. I. The role of colloid dispersibility and preferential flow. *VZJ* 3, 424-433.
- Laubel, A., Jacobsen, O.H., Kronvang, B., Grant, R. & Andersen, H.E., 1999. Subsurface drainage loss of particles and phosphorus from field plot experiments and a tile-drained catchment. *J. Environ. Qual.* 28, 576-584.
- Schelde, K., de Jonge, L.W., Kjaergaard, C., Laegdsmand, M. & Rubæk, G.H., 2006. Effects of manure application and plowing on transport of colloids and phosphorus to tile drains. *VZJ* 5, 445-458.
- Stamm, C., Flühler, H., Gächter, R., Leuenberger, J. & Wunderli, H., 1998. Preferential transport of phosphorus in drained grassland soils. *J. Environ. Qual.* 27, 515-522.

Phosphorus mobilization by water-dispersible colloids from agricultural soils

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Introduction

Colloids are agents of rapid transport of phosphorus (P) from P-enriched agricultural topsoils to drains and further on to surface waters (de Jonge et al., 2004). If the stringent quality targets for surface waters are to be achieved, diffuse P losses from agricultural land have to be reduced in Denmark. Targeting critical source areas of P loss offers a cost-effective way of mitigation planning. However, so far a simple test is lacking that permits the identification of areas with a high potential of colloidal P mobilization. Therefore, the objectives of our study were to suggest a suitable methodology and to explore relationships between conservative soil properties and colloidal P mobilization. Such relationships may be integrated in field-based risk assessment of P loss in the future.

Materials and methods

Intact 0.1 dm³ soil cores were collected from 47 of 590 agricultural sites of the Danish KVADRATNET grid-sampling scheme in autumn 2005. The 7 km-grid covers the whole country and is used for environmental monitoring. Soil properties in the plough layer had been characterized previously based on bulked samples from a 50 by 50 m area at each grid node (Table 1). Samples were taken in triplicate at 0.05-0.1 m depth covering a range of total P and clay contents. The soil cores were kept field moist at 5°C prior to use. One subset of soil cores was used for soil moisture determination.

Table 1. Selected soil properties at 0-0.25 m depth of 47 sites included in the study.

	pH _{CaCl2}	clay	silt	OM	total P	Olsen P	P _w	Al _{ox}	Fe _{ox}	P _{ox}
		%			mg kg ⁻¹			mmol kg ⁻¹		
Mean	6.0	7.4	8.0	3.6	566	41	14.4	41.7	40.9	13.4
STD	0.5	3.8	3.9	1.6	188	18	7.0	18.7	18.4	5.3
Minimum	5.0	2.1	2.2	1.6	194	14	4.1	12.6	8.4	3.0
Maximum	7.0	15.0	15.3	7.5	918	81	29.0	99.2	88.9	22.8

Colloid dispersion was done according to the low-energy water dispersible colloids (LE-WDC) method of Kjaergaard et al. (2004). Briefly, samples were slowly saturated by capillary wetting with a low ionic strength electrolyte (EC = 0.025 mS cm⁻¹) and then drained to $\Psi = -5$ hPa. After drainage samples were equilibrated in moisture-saturated atmosphere for 7 days at 10°C. Samples then were dispersed in the weak

electrolyte in two subsets at soil:solution ratios of 1:11 (SS1) and 1:8 (SS2). Colloid suspensions $<2 \mu\text{m}$ ($C<2.0$) were produced by one-step gravity sedimentation for SS2 and colloid suspensions $<0.45 \mu\text{m}$ ($C<.45$) by centrifugation for both SS1 and SS2. All colloid suspensions were ultrafiltered with Amicon® Ultra-4 devices (UF1 for SS1; UF2 for SS2) with a nominal molecular weight cut-off of 10 kDa. Fifteen ml of the original SS1 suspensions were filtered through Advantec® 0.45 μm cellulose acetate membrane filters, discarding the first few drops of filtrate ($F<.45$). Phosphorus was determined colourimetrically by the molybdate-blue method (MRP). Total P was determined after perchloric acid digestion. Additionally, dissolved ortho-P was determined in $F<.45$ membrane filtrates by ion chromatography (IC-P). Colloidal P (CP) was the difference between total P and UF-MRP. To describe CP as a function of general soil properties, regression models were fitted to log-transformed CP by the least square method. The model choice was based on a 5% significance level for model terms and the Akaike Information Criterion. Data from SS1 and SS2 were included in the model for $\log(CP<.45)$, while $\log(CP<2.0)$ was based on SS2 alone.

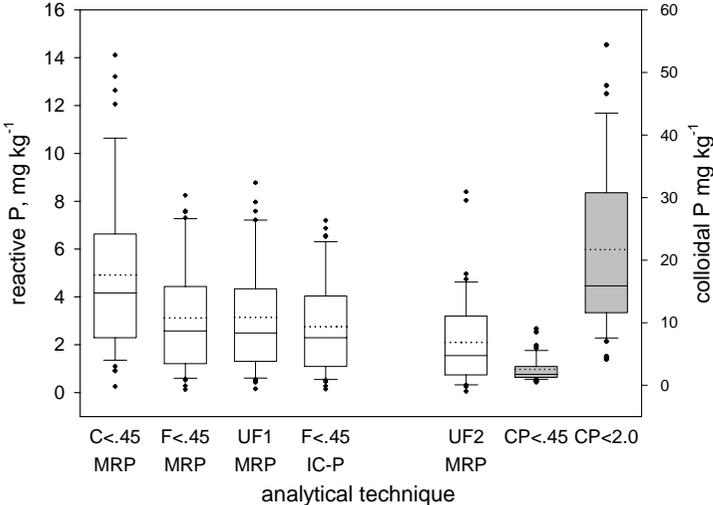


Figure 1. Boxplot of different analytical P forms in colloid suspensions produced by the LE-WDC method. Whiskers show 10th or 90th quantiles of P contents; the means are dotted lines and outliers points. Open boxes represent reactive P for different pretreatments. Filled boxes show colloidal P. The left and right group of boxes represent respectively SS1 and SS2. Ultrafiltrates were produced from $<0.45 \mu\text{m}$ suspensions. For explanations see text.

Results

LE-WDC provides a physically realistic estimate of the potential, in situ colloid mobilization by infiltrating rain in the upper soil horizon (e.g. Kjaergaard et al., 2004). Therefore, we applied this test to determine colloid P mobilization. Analytically,

colloidal P relies on the sound measurement of actual dissolved P. The difference between MRP in $<0.45 \mu\text{m}$ suspensions ($C_{<0.45}$) and IC-P in membrane filtrates ($F_{<0.45}$) indicated that a substantial part of colloidal P was easily hydrolyzed during colourimetric P determination (Fig. 1) overestimating dissolved P, when an appropriate pretreatment was lacking. The similarity between IC-P and MRP in membrane filtrates (Fig. 1) showed that membrane filtration removed colloids much more efficiently than the nominal cut-off implied. Due to these uncertainties and the cost of IC we chose Amicon® Ultra-4 filtration as an alternative, simple technique for removing colloids from aqueous solutions before MRP analysis. Less than 5% of total P in ultrafiltrates was molybdate unreactive. UF-MRP in $<0.45 \mu\text{m}$ and $<2.0 \mu\text{m}$ suspensions was virtually identical (data not shown).

The amounts of different P forms mobilized varied widely across the soils (Fig. 1) in line with variations of soil P and clay contents. Colloidal P was much more important exceeding dissolved reactive P on average two and 19 times in $<0.45 \mu\text{m}$ and $<2.0 \mu\text{m}$ suspensions, respectively. Ten times more colloidal P was associated with the <2.0 than the $<0.45 \mu\text{m}$ fraction (Fig. 1) and $CP_{<2.0}$ were in the same order of magnitude as Olsen P. Sediment mass and colloidal P were strongly linearly correlated. Highly significant regression models for $\log(CP_{<0.45})$ and $\log(CP_{<2.0})$ explained 70% and 60% of the variance. Olsen P and OM were included in both models. Clay was third explanatory variable in the former and Fe_{ox} in the latter model (Table 2).

Table 2. Parameter estimates of regression models for colloidal P mobilization in LE-WDC test. 'Class' is a class variable (0, 1) for two soil:solution ratios in SS1 and SS2.

Response variable	DF total	intercept	class	clay %	Olsen P Mg kg^{-1}	OM %	Fe_{ox} mmol kg^{-1}	R^2	Pr > F
$\log(CP_{<0.45})$	92	0.32	-0.51	0.1	0.016	-0.14		0.70	<0.0001
$\log(CP_{<2.0})$	45	2.42			0.013	-0.21	0.02	0.60	<0.0001

Conclusion

Potentially mobile colloidal P can satisfactorily be described as simple function of few conservative soil properties across a wide range of soils. If current research confirms our findings, these relationships may substantially improve the assessment of colloidal P loss by macropore transport in tools like the P Index.

References

- de Jonge, L.W., Moldrup, P., Rubæk, G.H., Schelde, K. & Djurhuus, J., 2004. Particle leaching and particle-facilitated transport of phosphorus at field scale. *Vadose Zone J.* 3, 462-470.
- Kjaergaard, C., de Jonge, L.W., Moldrup, P. & Schjønning, P., 2004. Water-dispersible colloids: Effects of measurement method, clay content, initial soil matric potential and wetting rate. *Vadose Zone J.* 3, 403-412.

Distribution of extractable phosphorus in soil profiles on heavily fertilized clay and sandy soils in South Eastern Norway

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Introduction

Extensive tile draining is common in Norway. The recommended distance between the tiles has been 6-8 m on clay soil and 8-10 m on sandy soils. This practice results in a significant risk for phosphorus (P) loss through the tiles, especially if the P content in the soil close to the tile drains is high. To increase our knowledge about the risk of diffuse P losses through heavily fertilized soil, nine profiles in a clay soil and nine profiles in a sandy soil in South Eastern Norway were investigated. In order to compare the results from the cultivated clay soil with uncultivated soils, three profiles on uncultivated clay soil adjacent to the cultivated soil profiles were included in the study. The aim was to study if the topsoil P influenced the P concentration further down in the profile. In this presentation the main focus will be on the clay soil profiles.

Materials and methods

The sites for the profile sampling were chosen according to the topsoil values of ammonium acetate lactate extractable P (P-AL) (Egnér et al. 1960), such that a range in topsoil P-AL values was obtained. The clay soil profiles were located on a pasture, which has not been ploughed for several years, whereas the sandy soil profiles were located on a field which is regularly ploughed. For each profile, soil samples every 10 cm from 0-100 cm were analysed for total P, P-AL and P extracted with 0.0025 M CaCl_2 (P- CaCl_2). To characterise the soil pH, clay content and AL extractable Ca were also analysed.

Results and discussion

The distribution of P-AL and P- CaCl_2 in three selected profiles on the clay soil is shown in Figure 1. In the profile on uncultivated soil P-AL is rather low down to 40 cm depth. Further down in the profile P-AL increased with depth. The profiles from the cultivated soil showed the same pattern, except that the upper soil layers had higher P-AL values due to P fertilisation. This pattern is in accordance with results from a study on a Swedish clay soil (Djordjic et al. 1999). However, an acceptable understanding is that the content of plant-available P is highest in the topsoil and decreases to a low soil P level in the subsoil (Mozaffari and Sims, 1994). Average values for the nine profiles on cultivated clay soil showed that the P-AL values were

generally higher in the 70-100 cm layer than in the topsoil (averages of 13.4 and 9.1 mg P 100 g⁻¹, respectively), whereas the average for the 30-40 cm layer was low (2 mg P 100 g⁻¹). In fertilisation planning, P-AL value of 0-2 is characterised as low, 3-6 as intermediate, 7-15 as high and >15 as very high. There was a tendency for increased P-AL in the 70-100 cm layer with increased P-AL in the topsoil (0-20 cm). For P-CaCl₂ there was no such relationship. The P-CaCl₂ concentration in the subsoil was zero or close to zero for all the profiles. In the sandy soil there were no relationships between topsoil values and subsoil values for neither of the P extractions. However, also here the P-AL values increased from the 40-50 layer towards the bottom of the profiles, but the increase was smaller than for the clay soil.

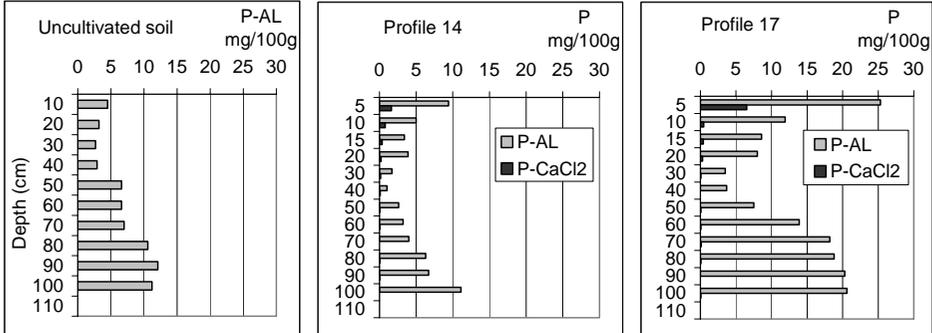


Figure 1. Distribution of P-AL and P-CaCl₂ in three selected profiles on a clay soil.

The total P concentration was higher in the topsoil than in the subsoil due to accumulation of excess fertiliser P in the topsoil (Table 1). In the subsoil total P generally increased with depth together with pH.

Table 1. Some soil characteristics at three depths of the three soil profiles presented in Figure 1. U = uncultivated soil.

Depth (cm)	Clay %		pH			Total P (mg kg ⁻¹)			Ca-AL (mg 100 g ⁻¹)		
	P17	P14	P17	P14	U	P17	P14	U	P17	P14	U
0-10	26	23	5.8	6.5	4.4	1423	1247	614	151	149	41
50-60	36	26	6.9	6.8	6.9	764	770	770	150	109	159
90-100	35	32	7.4	7.5	7.3	966	881	933	125	152	175

In the clay soil, P-AL as a fraction of total P was higher in the subsoil than in the topsoil at comparable P-AL levels, showing that the total P in subsoil had a slightly higher availability than total P in topsoil. This could be explained by a higher pH in the subsoil, and thereby a higher content of Ca-phosphates, which may be dissolved in the acid AL extract (pH 3.8). However, the concentration of Ca-AL is not higher in the subsoil than in the topsoil, and is therefore probably not an explanation for the high P-AL values in the deepest layers of the investigated profiles.

Figure 1 shows that the P-AL values are rather high in the deeper layers also in the uncultivated soil, such that P-AL values of 10-12 mg P 100 g⁻¹ at the depth of the tile drainage is of natural origin. The higher P-AL values observed (15-20 mg P 100 g⁻¹) may be a result of transport of P-rich particles from the topsoil through macropores.

Independent of the origin of the high level of P-AL at the depth of the draining tiles, the question is to what extent the P found here contributes to eutrophication of the watercourses. Therefore, biotests with soil samples from the deep layers will be carried out to estimate the algae availability and thereby the potential eutrophication risk if particles from the deep soil layers are transported to the watercourses via the drain pipes.

References

- Djodjic, F., Bergström, L., Ulén B. & Shirmohammadi, A., 1999. Mode of transport of surface-applied phosphorus-33 through a clay and sandy soil. *J. Environ. Qual.* 28, 1273-1282.
- Egnér, H., Riehm, H. & Domingo, W.R., 1960. Untersuchungen über die chemische Boden-Analyse als Grundlage für die Beurteilung des Nährstoffzustandes der Boden. *Kungliga Lantbrukshögskolans Annaler* 26, 199-215.
- Mozaffari, P.M. & Sims, J.T., 1994. Phosphorus availability and sorption in an Atlantic Coastal Plain watershed dominated by animal-based agriculture. *Soil Sci.* 157, 97-107.

Changes in soil phosphorus fractions caused by air-drying

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Introduction

Air-drying has been reported to increase water-soluble and bicarbonate-extractable phosphorus (P) in soil. The main mechanism explaining this response of labile P to drying is probably the enhanced mineralization of organic matter in re-wetted soil. Turner and Haygarth (2001) reported that increase in water-soluble P as a result of drying and re-wetting took place mainly in organic form and was positively related to microbial P.

The objective of this study was to characterize drying-induced changes in various chemical P fractions. We hypothesized that drying and re-wetting maintains the cycling of labile P. The study was done using the Hedley fractionation procedure, based on the biological availability of P in soil.

Material and methods

Soil samples were collected from the topsoil (10 cm thick layers) of a young clayey soil of southwest Finland. Sampling sites had dissimilar management histories: a forest soil and fields under conventional farming or organic farming, and grassland on set-aside land. After sieving (5 mm), part of each soil sample was air-dried and another part was stored in the sampling moisture at +5°C. Samples were stored for 3.5 months prior to the analyses. Characteristics of the soil samples were: clay 55-65%, Oxalate-extractable Fe_{ox} 5130-5790 mg kg⁻¹ and Al_{ox} 2650-3400 mg kg⁻¹, soil organic carbon 3.6-5.2% (Leco[®] CNS-1000) and $\text{pH}_{\text{H}_2\text{O}1:2.5}$ 5.2-6.0.

Soil phosphorus was fractionated according to a modified Hedley's procedure (Hedley et al. 1982). One gram soil was successively extracted with 60 ml of deionised water (4 hours), 60 ml of deionised water (16 h), 60 ml of 0.5 M NaHCO_3 (pH 8.5)(16 h), 60 ml of 0.1 M NaOH (16 h) and 60 ml of 1.0 M HCl (16 h). The extracts were centrifuged for 15 minutes (3000g, r=18, 3900 rpm) and each supernatant was divided into two size-fractions by filtering through a 0.2 µm membrane filter (Whatman Nuclepore). The centrifugation time and rpm were adjusted so that according to Stokes' equation soil particles having a diameter larger than 0.2 µm would settle during centrifugation. All supernatants were analysed for total phosphorus (TP) and the filtrates were analysed for molybdate reactive phosphorus (MRP) and TP. Total P was analysed using persulfate (50 g $\text{K}_2\text{S}_2\text{O}_8$ / 1 l 0.4 M H_2SO_4) digestion (autoclaving for 30 min at 120°C and 1 bar pressure). Phosphorus concentrations were measured spectrophotometrically with the ascorbic

acid -method (Perkin-Elmer Lambda 15 UV/VIS). The difference between TP and MRP is referred as molybdate unreactive phosphorus (MUP). The MUP concentrations in filtrated and unfiltrated samples are referred to as MUP<0.2 (small-sized MUP) and MUP>0.2 (large-sized MUP), respectively. Statistical differences in the extractable P between the dry and moist samples were tested with paired samples t-test using SAS.

Results and discussion

Total P extracted with Hedley's procedure was not affected by drying being 772 mg kg⁻¹ in moist and 779 mg kg⁻¹ in dried samples (standard error of the difference between the treatment means was 11 mg kg⁻¹). A general trend was that air-drying increased MRP and small-sized MUP and diminished the large-sized MUP. This response was distinct in the first water extraction. In the second water extraction MRP and small-sized MUP<0.2 were lower than in the first water extraction but the large-sized fraction was higher. The higher extractability of larger organic molecules was probably attributable to a decrease in hydrophobicity of organic matter caused by the re-wetting in the first water extraction. According to these results, originally wet sample soil would release substantially more water-extractable large-sized MUP than small-sized MUP and MRP.

Table 1. Concentrations of phosphorus in moist and dry soil samples in different phosphorus fractions.

	MRP		MUP<0.2		MUP>0.2		TP					
	moist	dry	moist	dry	moist	dry	moist	dry				
	mg P kg ⁻¹											
4h H₂O	0.9	4.0	***	0.4	3.1	***	15	9.9	***	16	17	
16 H₂O	2.2	2.5		0.4	1.3	**	20	19		23	23	
NaHCO₃	26	29		44	52	***	1.9	0.9		71	82	***
NaOH	120	127		59	67	**	285	257	***	462	451	
HCl	199	205										

* Significant at the 0.05 probability level

** Significant at the 0.01 probability level

*** Significant at the 0.001 probability level

As for the NaHCO₃ fraction, statistically significant changes caused by drying were found in MUP<0.2 and TP. The concentrations of large-sized MUP were small. Compared to water, the NaHCO₃ solution has a larger salt concentration, which can cause the larger molecules to collapse and settle during the centrifugation. If the increase in MUP<0.2 in dried samples results from the break-up of large-sized P molecules, the exclusion of these large organic molecules in centrifugation leads to the drying increasing TP. The settling of large organic molecules in NaHCO₃ transfers them to be extracted with the subsequent fraction.

The changes in NaOH extractable P were similar to those in the first two fractions; the amount of MRP and MUP<0.2 increased whereas large-sized MUP decreased as a result of drying. Also TP extracted with NaOH seemed to decrease because of drying. Although this change was not statistically significant, it was similar to the drying-induced increase in TP in the NaHCO₃ fraction. This trend supports the assumption that the larger molecules were settled from NaHCO₃ extracts as a result of salt-evoked exclusion during centrifugation and were transferred to the NaOH fraction. Changes in H₂SO₄ fraction were not statistically significant.

Conclusions

In this study, drying did not affect the total fractionable P but changed the amount of P in different pools. As for water-extractable P, it resulted in the breakage of large-sized organic molecules to smaller ones. When estimating the drying-evoked changes in water-extractable P, the extraction procedure should be carefully considered. These results emphasize the importance of TP analysis of unfiltered samples as the effect of drying on P fractions can be overestimated if samples are centrifuged and filtrated through 0.2 µm membranes and only the filtrate is analyzed for P. Drying may play an important role in nutrient turnover, breaking down bigger organic molecules, thus rendering them more available to microbial degradation. Chepkwony et al. (2001) found that subjecting soil to drying and rewetting cycles increased the extractability of P and further increased plant P uptake and dry matter yields. According to these results the use of dried soil samples could be justified when analyzing plant-available P. The higher concentrations of labile P extracted in dried soil samples might give information on the potential of soil to release plant-available P.

References

- Chepkwony, C.K., Haynes, R.J., Swift, R.S. & Harrison, R., 2001. Mineralization of soil organic P induced by drying and rewetting as a source of plant-available P in limed and unlimed samples of an acid soil. *Plant Soil* 234, 83-90.
- Hedley, M.J., Stewart, J.W.B. & Chauhan, B.S., 1982. Changes in inorganic and organic soil phosphorus fractions induced by cultivation practices and by laboratory incubations. *Soil Science Society of America Journal* 46, 970-976.
- Turner, B.L. & Haygarth, P.M., 2001. Phosphorus solubilization in rewetted soils. *Nature*, London 411, 258.

Variation throughout the year in different parameters associated with the risk of P loss in soils

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Introduction

Agricultural activities are one of the main sources of diffuse contamination of water as many of the nutrients supplied in fertilizers can pass to water bodies (Sharpley *et al.*, 1992). To mitigate this problem, we must know at what times of year the levels of nutrients are highest in soils and when the greatest risk of erosion or leaching of the nutrients occurs (Heathwaite, 2002). The aim of the present study was to find out the annual variation in different soil parameters in relation to P dynamics in order to establish the best times of year to apply fertilizers to soils to minimize the risk of contamination of water bodies.

Material and methods

The study was carried out in the surroundings of the Fervenza reservoir (Galicia NW Spain). The soils in the area are developed on acid rocks and are classified as Leptosols and Umbrisols. Twenty sampling sites were selected: seven forest soils (Forest), three natural grassland soils with a low level of management and fertilization (NF grasslands) and 10 highly fertilized grassland soils (F grasslands). The upper A horizon (0-10 cm depth) was sampled monthly between November 2000 and November 2001. Available P forms (total, P_t ; inorganic, P_i , and organic, P_o) were extracted by the Olsen procedure (Olsen *et al.*, 1954). P_t was determined by the method of Davidescu and Davidescu (1982), P_i was determined after flocculating the extracted organic matter with H_2SO_4 , and P_o was estimated as the difference between P_t and P_i . Phosphorus desorption was estimated at soil:water ratios of 1:20 and 1:100, with shaking for 30 minutes. The total P (TP) was determined by digestion (Rowland and Haygarth, 1997), and the molybdate-reactive phosphorus (MRP) was determined as the inorganic P present in the extract after filtration through a membrane of pore size 0.45 μm (Haygarth and Sharpley, 2000). The difference between TP and MRP represented the amount of particulate phosphorus (PP). In all cases the P in the extracts was determined by the method of Murphy and Riley (1962). All analyses were carried out in triplicate.

Results and discussion

In the forest soil the contents of Pt, Pi and Po extracted by the Olsen method were almost constant throughout the year, in the non-fertilized pasture they peaked in summer, and in the fertilized pasture they peaked at several different times. In the latter type of soil, Pt and Po were out of phase each month, as maximum levels of Pt were reached before maximum levels of Pi, which suggests that Pi is basically derived from organic forms of P, possibly provided by fertilizers (data not shown).

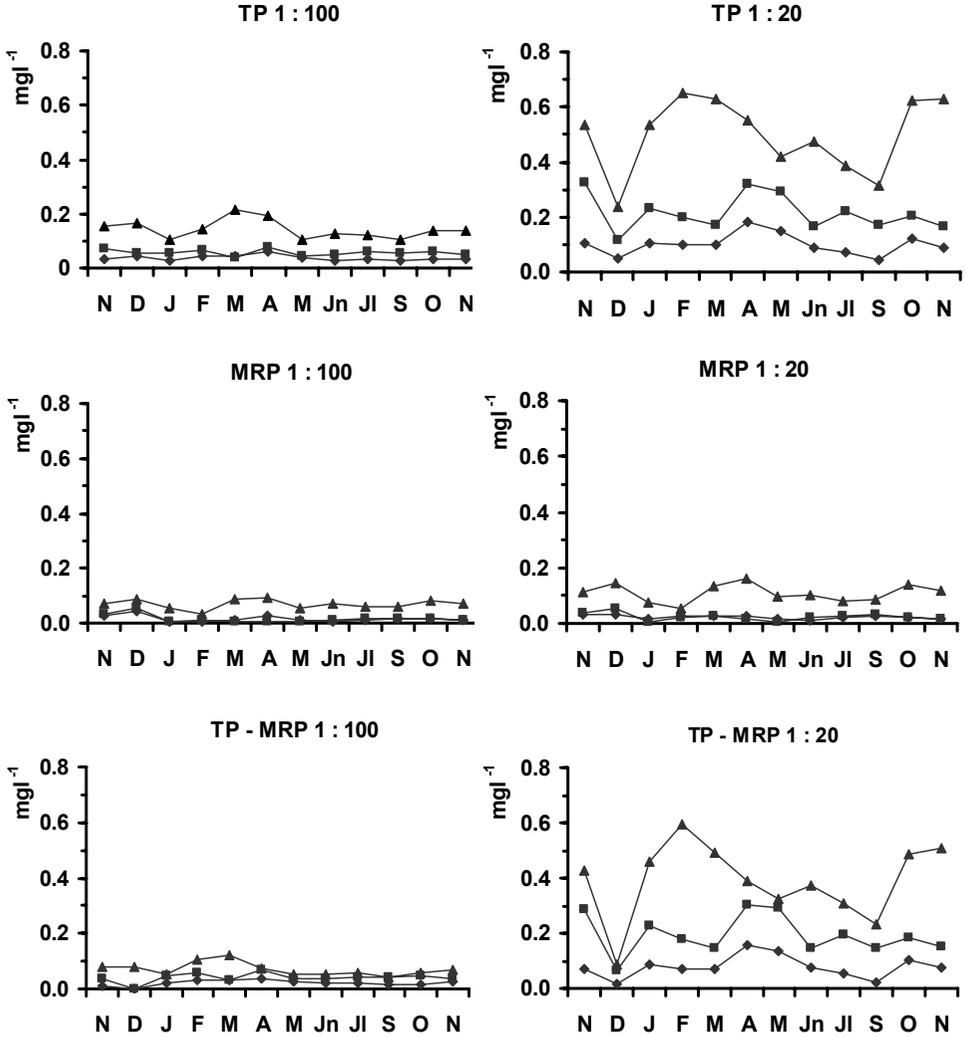


Figure 1. Evolution throughout the year of several fractions of desorbed phosphorus.

The dynamics of water-soluble P were independent of those followed by Olsen P. The phosphorus was scarce in the 1:100 soil:water extract and was dominated in all soils by PP, except in the fertilized pasture in which PP and MRP were present in similar proportions (Figure 1). Desorption peaks were observed in the 1:20 soil:water extract of forest and natural pasture soils, and were caused by greater desorption of PP in spring and at the beginning of autumn. In the fertilized soils, there was an intense increase in desorbable P from February onwards, with a decrease in spring, a new increase at the beginning of summer, a further decrease and new increase in autumn. These dynamics were again dominated by PP, and the MRP showed less intense peaks that were displaced with respect to the PP maxima. On the other hand, the dynamics followed by the water-soluble P were totally independent of those followed by the Olsen P. This suggests that estimation of the risk of P loss based on the variations in Olsen P (i.e. the usual method of estimating the available P in soils) would not be accurate in these soils.

Conclusions

The results indicate that measurement of soil-available P by agronomic techniques, such as Olsen extraction, does not provide accurate information about the risk of P loss by soils. In the most heavily fertilized soils, the highest risk of loss was produced almost two months after addition of fertilizer and tended to coincide with periods of heavy rain (early spring and mid autumn), which increases the risk of contamination of waters because erosion is favoured at these times and because most of the water-soluble P is concentrated in the PP fraction. To minimize the risk, farmers should space out the timing of fertilizer application on the same piece of land, as well as on different fields close to each other.

References

- Davidescu, D. & Davidescu, V., 1982. Evaluation of fertility by plant and soil analysis. Ed. Abacus Press, London, UK.
- Haygarth, P.M. & Sharpley, A.N., 2000. Terminology for phosphorus transfer. *J. Environ. Qual.* 26, 10-15.
- Heathwaite, L., 2002. Empirical phosphorus export models. In: Kronvang, B. (ed.). Diffuse phosphorus losses at catchment scale. COST Action 832, Alterra, Wageningen. Pp. 35-40.
- Murphy, J. & Riley, J.P., 1962. A modified single solution method for the determination of phosphate in natural waters. *Anal. Chim. Acta* 27, 31-36.
- Rowland, A.P. & Haygarth, P.M., 1997. Determination of total dissolved phosphorus in soil solutions. *J. Environ. Qual.* 26, 410-515.
- Sharpley, A.N., Smith, S.J., Jones, O.R., Berg, W.A. & Coleman, G.A., 1992. The transport of bioavailable phosphorus in agricultural runoff. *J. Environ. Qual.* 21, 30-35.

Meat and bone meal and fox manure as P sources for plants - a field experiment

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Introduction

Byproducts of agriculture, meat and bone meal (MBM) in particular, contain considerable amounts of phosphorus (P). Since apatite reserves are limited and overuse of fertilizer P has negative effects on the quality of surface waters, P in the byproducts needs to be efficiently recycled to avoid unnecessary nutrient inputs to the agro-ecosystem. In Finland, as 1.5×10^6 kg P flows annually into MBM. About 70% of this P, representing 0.5 kg P per each hectare of agricultural land, is available for recycling either directly as fertilizer or through feeding fur animals with it, the latter representing presently the main user of MBM. With the P content of about 1% (MBM 5%), fur animal manure comprises about 1.0 kg P ha^{-1} . However, due to the origin, most of P in fur animal manure and MBM is in the form of sparsely soluble calcium phosphates (Ylivainio et al. 2007). The present study aims to promote efficient recycling of these P sources in agriculture by determining the bioavailability of MBM-derived P in plant production.

Material and methods

A four-year field experiment (2004-2007) was established on a slightly acid (pH 6) sandy soil with very low P status ($P_{AAAC} \sim 1 \text{ mg P l}^{-1}$) and high organic C content (20%). The P sources were MBM, composted fox manure (cFM), pelletized cFM (pcFM), dairy manure and superphosphate (SP) as reference. At the beginning of the experiment single P doses for the whole four-year experimental period were applied at 40 or 100 kg P ha^{-1} , except for the dairy manure treatment, which received 15 or 30 kg P ha^{-1} . There were additional plots receiving annually 6, 10, 30 and 75 kg P ha^{-1} as SP. Two crop rotations were established: barley – oats – barley – oats, and barley – barley with grass undersown – grass – grass. The yields of barley, oats and grass were converted to feed units (fu) using coefficients of 1.13, 0.98 and 0.94, respectively.

In the first year of the experiment, nitrogen (N) requirement of the crops were fulfilled by taking into account the amounts of soluble N in the different P sources and supplementing with commercial N fertilizer to reach the target level of 70 kg ha^{-1} . In the later years commercial fertilizer alone was used to achieve 70 kg ha^{-1} and 100 kg ha^{-1} of N for cereals and for both cuts of the grass, respectively. Potassium (K as

KCl) fertilizer was applied annually to reach 70 kg K ha⁻¹ for the cereals and for both cuts of the grass.

Results

Results represent the first three years (2004-2006) of the four-year experiment. The highest single application of one of the sparsely soluble P sources, pcFM (100 kg P ha⁻¹), increased the grain yields as much as a similar amount of P applied as SP, whereas the least efficient P source for barley/oats was MBM (Table 1). However, after more than two years from the P application, MBM (100 kg P ha⁻¹) increased the 2nd grass yield most, even more than the annual P application of 75 kg ha⁻¹ as SP (Table 1). Although the experimental soil was P-deficient according to the Finnish P status classification, the yield increment (control yield subtracted) dropped sharply along with the rising P application levels, except in the MBM treatment (Fig 1). Dairy manure was as efficient a P source as SP (Fig 1).

Table 1. Yields of barley and oats (2004-2006) and grass (2006), fu ha⁻¹, fertilized with different P sources at the beginning of the experiment (single application) or annually with superphosphate. Abbreviations in the text.

Treatment, kg P ha ⁻¹	Cereals	Grass, 1 st cut	Grass, 2 nd cut
Control (no P applied)	10211	3620	4749
SP, 6 (annual application)	12156*	4177	5166
SP, 10 (ann. app.)	13287	4379	5230
SP, 30 (ann. app.)	14808	4715	5296
SP, 75 (ann. app.)	14803	4885	5631
SP, 40 (single application)	13577	4325	5012
SP, 100 (s. app.)	14859	4771	5272
Dairy manure, 15 (s. app.)	12446	4296	4933
Dairy manure, 30 (s. app.)	13543	4404	5057
cFM, 40 (s. app.)	12108	4646	4892
cFM, 100 (s. app.)	13288	4625	5077
pcFM, 40 (s. app.)	12226	4621	5373
pcFM, 100 (s. app.)	14835	4699	4900
MBM, 40 (s. app.)	10690	4123	4917
MBM, 100 (s. app.)	11902	4563	5716

*SP was applied in two (2004-2005) out of the three years.

Discussion

According to the results so far, cFM and pcFM appear suitable for both cereals and grass as long-term P suppliers, while MBM seems to be best for crops with a long growth period, such as perennial grasses. The incorporation of MBM in connection with sowing of the grass would reduce the need for surface application of P in the

later years, mitigate P enrichment on the thin surface layer of the grasslands and decrease P losses in surface runoff (Turtola and Yli-Halla 1999).

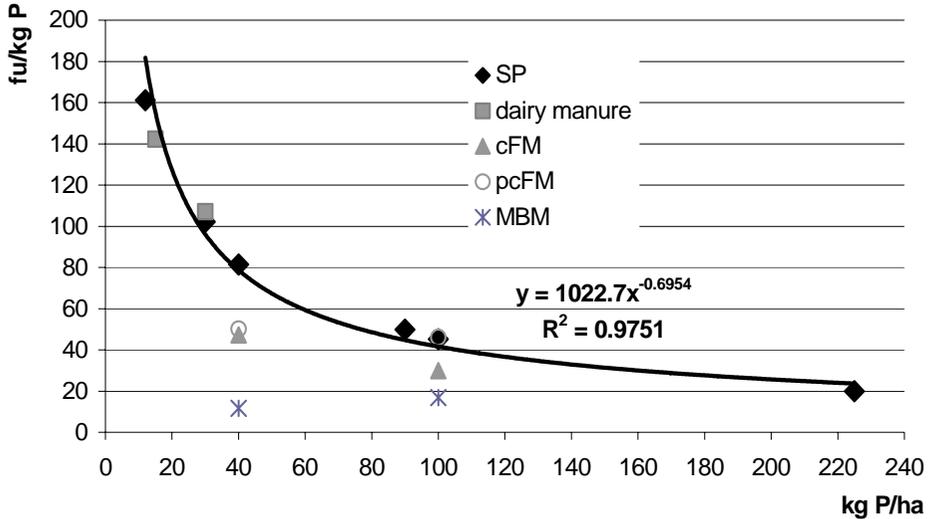


Figure 1. Total yield increment of barley and oats in three years (2004-2006) with different P sources. Abbreviations in the text.

The high availability of P in dairy manure was in contradiction with the present conversion coefficient of 85% (75% until 2006) set for P availability in manure. The validity of the respective coefficient for fur animal manure and MBM (40%) will be evaluated on the basis of this field experiment, a parallel field experiment on a clayey soil, pot experiments with ryegrass (Ylivainio et al. 2007) and data on the forms of P in soils receiving large repeated doses of fur animal manure (Uusitalo et al. 2007).

Conclusions

The results suggest that the P fertilization recommendations need to be critically evaluated, especially for the different manures, agricultural byproducts, and for soils with higher P status.

References

- Turtola, E. & Yli-Halla, M., 1999. Fate of phosphorus applied in slurry and mineral fertilizer: accumulation in soil and release into surface runoff water. *Nutr. Cycl. Agroecosyst.* 55, 165-174.
- Uusitalo, R., Ylivainio, K., Turtola, E. & Kangas, A., 2007. Phosphorus forms in soils amended with either readily or sparsely soluble phosphorus sources. Submitted.
- Ylivainio, K., Uusitalo, R. & Turtola, E., 2007. Meat bone meal and fox manure as P sources for ryegrass (*Lolium multiflorum*) - a three-year pot experiment. Submitted.

Developing new guidelines on biosolid applications of phosphorus to agricultural soils in the UK

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Introduction

Biosolids produced from municipal wastewater treatment centres are a useful source of nutrients and organic matter for soils, and their recycling to agricultural land is a practical and environmentally sustainable disposal option. In many regions, land application is restricted on high-P soils because of the increased risk of phosphorus (P) transfer in runoff to watercourses leading to eutrophication. Biosolid applications increase soil test P (STP) more rapidly than other P amendments at equivalent nitrogen rates, and numerous studies have shown linear or non-linear increases in runoff P as STP increases. However, biosolids contain P binding elements (Fe, Al and Ca) may inhibit P release to water despite increasing STP (Maguire *et al.* 2000). To test this hypothesis further we investigated the impact of variably-treated biosolids on the soil P sorption properties of five different soils and the potential transfer of P in land runoff to watercourses assessed by changes in soil water-extractable P.

Experimental details

Triplesuperphosphate (TSP) fertiliser and seven biosolid types were applied to each of five different soil types in a laboratory incubation study. The TSP and biosolids were mixed with the soils at three rates, supplying 200, 1000 and 2000 kg ha⁻¹ P, to provide a large range in potential percentage soil P saturation. The biosolids had been treated to varying degrees by anaerobic digestion, dewatering to produce a cake, lime stabilisation, Fe-dosing, and/or thermal drying (Table 1). The soils ranged in texture from sandy to clayey and one soil was calcareous.

Table 1. Biosolid characteristics.

Biosolid source	Biosolid type [§]	pH	DS (%)	Ca (g kg ⁻¹)	Fe (g kg ⁻¹)	P (g kg ⁻¹)
Cambridge	DSC	7.1	50	68	7	22
Bedford	DSC + dosed Fe	6.9	58	46	70	38
Davyhulme	DSC with high Fe	7.2	76	50	68	21
King's Lynn	LSC	10.8	52	138	13	12
Broadholme	LSC + dosed Fe	11.5	43	138	28	17
Blackburn	LSC with high Fe	7.9	49	111	44	16
Grimsby	TDS	7.3	94	45	21	24

[§]DSC = digested sludge cake; LSC = lime stabilised cake; TDS = thermally-dried sludge

Soils and biosolids were slowly air-dried, passed through a 6 mm sieve, mixed, and incubated in polyethylene containers in the dark for 90 days at 20°C, prior to analysis for STP (Olsen-P method) and water-extractable P (WEP). Sorption isotherms describing the relationship between the amount of P adsorbed ($Q \text{ mg kg}^{-1}$) and the final solution concentration ($c, \text{mg L}^{-1}$) were fitted by a double Langmuir function.

Results

Biosolids gave less than 50% of the increase in STP that was obtained after TSP fertiliser application, with lowest increases obtained when biosolids contained Fe. Addition of biosolids often had no effect on WEP concentrations and Fe-rich biosolids tended to reduce them. Biosolids with total Fe:total P ratios above about 2 gave no increase in WEP (Fig. 1a) even though they increased STP levels.

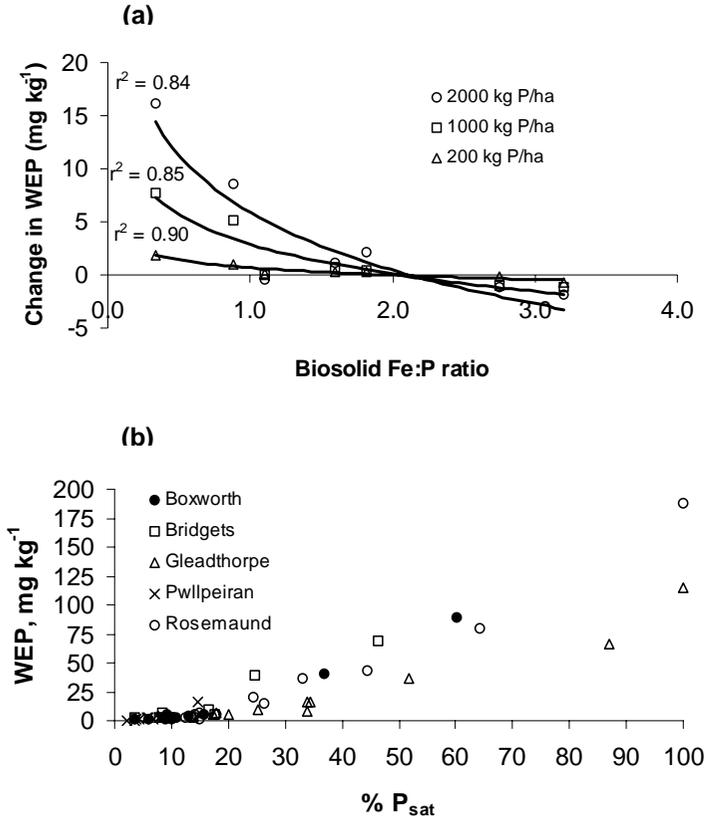


Figure 1. Biosolid effects are evident in (a) the relationship between biosolid Fe:P ratio and soil water-extractable P at each rate of P added but largely explained by (b) differences in the percentage saturation of the soil P sorption capacity (P_{sat}).

Application of lime-stabilized biosolids consistently produced the largest increase in soil P sorption capacity, while Fe-treated biosolids increased P sorption capacity on soils low in Fe. However, Fe-treated biosolids reduced P binding energy (k) more than lime-stabilized biosolids. Hence, the impact of biosolid application on potential P release to runoff was best predicted by differences in % P saturation, with increased risk of P loss to runoff above ca. 15-20% P saturation as calculated from the P sorption isotherm (Fig. 1b).

The results of the incubation study were incorporated into a risk-based management approach to biosolid use based on hydrological connectivity between the field and the watercourse, and site runoff and erosion risk. It was proposed that biosolids can be more liberally applied to fields with no direct hydrological connectivity by rapid flow pathways to the watercourse, whilst adoption of more sensitive biosolid and land management measures to reduce runoff and erosion risk are required for fields with medium and high connectivity risk. More sensitive land management may include adjusting the amount of biosolid applied, the method of application or the way the soil is cultivated, or by establishing protection zones (e.g. buffer strips). Improved management of land may lower the P loss risk class sufficiently to allow increased biosolid application rates on some soils without detriment to water quality, depending on biosolid type.

Conclusions

A better understanding of the fate of biosolid P in soils and subsequent risk of P loss in land runoff is required in order to safeguard land application as a beneficial disposal route. We concluded that STP alone is not a good indicator of P loss risk on biosolid-amended soils. Our incubation study suggested that even when the equivalent of 20 years of typical sludge P supply is incorporated into the soil, there may be no increase in runoff P, depending on the degree of P saturation, soil and biosolid type. Our work provides a means of quantifying the amounts and type of biosolid that can be sustainably applied to agricultural land without detriment to surrounding watercourses.

Acknowledgements

This work was funded by United Kingdom Water Industry Research (UKWIR), the Department of the Environment, Food and Rural Affairs (Defra) and the Environment Agency (EA) in England and Wales.

References

Maguire, R.O., Sims, J.T. & Coale, F.J., 2000. Phosphorus solubility in biosolids-amended farm soils in the Mid-Atlantic region of the USA. *J. Environ. Qual.* 29, 1225-1233.

The effect of soil phosphorus on the phosphorus sorption properties of suspended sediment in runoff

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Introduction

The portion of the phosphorus (P) in soil solutions, land runoff, and waterbodies that is associated with suspended sediments (SS) $>0.45\mu\text{m}$ in size is often termed the particulate P (PP) fraction. This PP fraction is dominant in runoff from cultivated agricultural land, and sometimes grassland, due to the detachment of soil particles during storm events. These eroded particles are composed of clay and silt-sized aggregates which are often enriched with P compared to the soil from which they were derived (Sharpley, 1980). The P sorption properties of these fine-textured, P-enriched particles in land runoff exert an important influence on dissolved P concentrations both in the runoff and on entering a waterbody.

Whilst there is much information on the relationship between soil P and dissolved ($<0.45\mu\text{m}$) P concentrations in agricultural runoff, there is less information on the impact of soil P level on the PP fraction in runoff, or on the P sorption properties of the SS. We investigated the variation in P sorption characteristics of SS in surface runoff from selected EU soils and in a plot experiment with different levels of soil P.

Experimental details

As part of the DESPRAL project (Withers *et al.* 2007), surface runoff was collected from seven EU field sites with variable slopes under outdoor simulated rainfall (60 mm hr^{-1} , 2.7m height, 30 mins). At one additional site (Rosemaund), surface runoff was collected under natural rainfall from fifteen adjacent and unreplicated field plots ($15\text{m} \times 2\text{m}$) with Olsen soil test P values ranging from 19 to 194 mg kg^{-1} .

The SS in the collected runoff was determined as dry residue and bulk sediment samples air-dried for determination of P sorption parameters (maximum P sorption capacity (Q_{max}), % P saturation (P_{sat} defined as Q_0/Q_{max} where Q_0 is native P) and the equilibrium P concentration at net zero P sorption (EPC_0)) calculated using either single (EU soils) or double (Rosemaund) Langmuir functions. These P sorption parameters were compared to those obtained for representative bulk soil samples from each site/plot prior to runoff collection and related to basic soil characteristics. Where there was enough sediment, Olsen-P and total P were also determined.

Results

Outdoor rainfall simulation

Some basic physical and chemical characteristics of each of the EU soils together with the differences in P sorption parameters between the bulk soil and the runoff SS collected in the outdoor rainfall simulation study are shown in Table 1. All sediments had greater Q_{\max} values (average +57%) compared to soils, except for Ritzlhof 5 soil where there was very little difference. However, sediment EPC_0 concentrations tended to increase in soils with low/medium P fertility (Naghorvati and Rottenhaus 5) and tended to decrease in soils with high P fertility (Tetto Frati, Riva, Ritzlhof 4 and Ritzlhof 5). Whether EPC_0 in the SS increased or decreased relative to the soil appeared to coincide with changes in the degree of P saturation (P_{sat}) between the sediments and the soils, although this was not always the case (e.g. Ritzlhof 5). The P impoverished Somogybabod soil showed relatively little change in all parameters.

Table 1. Selected characteristics and P sorption parameters of soils and the suspended sediments (SS) derived from them in runoff under outdoor rainfall simulation.

Site	Soil or SS	pH	Clay (%)	TP (mg kg ⁻¹)	Olsen-P (mg kg ⁻¹)	Q_{\max} (mg kg ⁻¹)	P_{sat} (%)	EPC_0 (mg L ⁻¹)
Somogybabod	Soil	7.7	17	605	3	52	4	0.01
	SS					80	1	0.06
Naghorvati	Soil	6.8	20	682	12	46	26	0.14
	SS					146	27	0.51
Rottenhaus 5	Soil	6.4	39	1075	13	76	36	0.14
	SS					176	71	0.96
Riva	Soil	6.3	20	767	39	119	45	1.86
	SS					237	40	0.51
Tetto Frati	Soil	7.7	12	1062	41	87	51	1.34
	SS					177	28	0.4
Ritzlhof 4	Soil	7.3	23	1314	57	152	38	1.4
	SS					829	13	1.09
Ritzlhof 5	Soil	7.3	23	1613	95	395	19	10.3
	SS					381	28	2.8

Rosemaund

The soil at Rosemaund is a predominantly silty soil (pH 6.8, 30% clay) that readily disperses during storm events. For all plots regardless of soil P status, Q_{\max} in the runoff SS increased by an average 56% (range 37-83%) compared to the soil (441 v 283 mg kg⁻¹). As for the EU soils, EPC_0 concentrations both decreased and increased depending on the change in P_{sat} between the SS and the soil; data for all sites/plots are shown in Fig.1. Compared to the soils, the values of P_{sat} (15 - 42%), and EPC_0 (0.7 – 5.5 mg L⁻¹) in sediments fell within narrower ranges and were unrelated to the initial soil P concentrations. However, sediment EPC_0 concentrations were lower than those in the soil when soil Olsen-P exceeded ca. 110 mg kg⁻¹. For

Rosemaund soils, sediment P enrichment ratios were always highest on the low P soils (up to 6 for Olsen-P and 3 for total P) and decreased with increasing soil P.

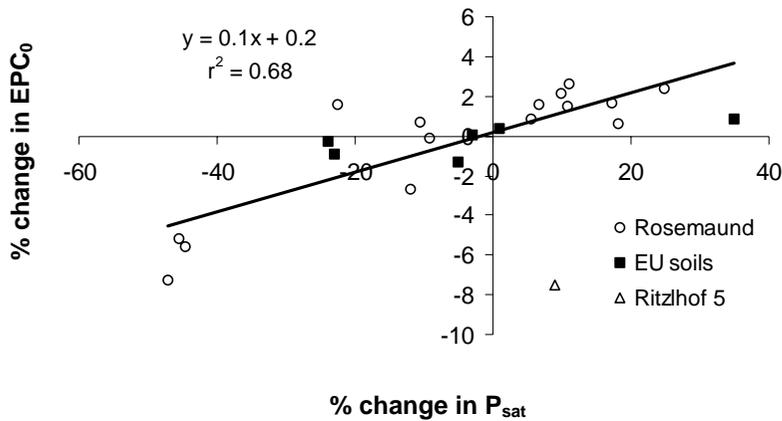


Figure 1. The percentage change in the equilibrium P concentrations (EPC₀) (suspended sediments minus soil) as a function of the percentage change in the degree of P saturation (P_{sat}) for the EU and Rosemaund soils.

Discussion

Suspended sediments collected in runoff from soils had a higher P sorption capacity than the soils from which they were derived because of the selective detachment and transport of fine particles and aggregates. These particles were also enriched in P to various degrees depending on the history of P fertilisation and clay enrichment ratio in the runoff (probably a function of the amount of SS transported, Sharpley, 1980) producing variable source/sink effects once in the water phase. Where soil P was relatively low, detached sediment was more P-saturated than the soil (e.g. for Rosemaund: 15-34% v 8-15%) due to a high *relative* P enrichment ratio. Where soil P was high, detached sediment was less P saturated than the soil (e.g. for Rosemaund: 25-42% v 55-76%) probably due to a combination of P sorption onto coarser soil particles, precipitation of P in soil and some release of P to runoff water.

Conclusions

Our data suggest that suspended sediments in runoff from both low and high P soils have the potential capacity to release P on entering streams depending on the stream dissolved P concentration and the EPC₀ of the sediment.

References

Sharpley, A.N., 1980. The enrichment of soil phosphorus in runoff sediments. J. Environ. Qual. 24, 920-926.

Withers, P.J.A., Hodgkinson, R.A., Barberis, E., Presta, M., Hartikainen, H., Quinton, J., Miller, N., Sisak, I., Strauss, P. & Mentler, A., 2007. An environmental soil test to estimate the intrinsic risk of sediment and phosphorus mobilisation from European soils. *Soil Use and Management* (In press).

Effects of ditch dredging on P transfers in a coastal plain setting

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Introduction

Accelerated eutrophication is the most common surface water impairment in the United States (U.S. Geological Survey, 1999), and it can be greatly accelerated by the influx of phosphorus (P) in surface flow from agricultural land. In the Atlantic Coastal Plain region, large concentration of intensive poultry operations coincide with extensively ditched fields, resulting in high potential for nutrient loading by surface and sub surface flow to ditches (Mozaffari and Sims, 1994). Ditches represent a direct conduit to surface water, but their role in mediating nutrient transport has been not fully understood. Dredging is a common practice to remove sediments from ditches, streams, rivers, or estuaries, and is generally performed to optimize the flow of water and ensure adequate capacity for drainage and represents a severe ecological disturbance and may function to either increase or decrease nutrient loads. The sediment exposed by dredging may not have the same nutrient buffer capacity as the original sediment. However, dredging may also function to remove enriched sediments from the flow system and may expose non-enriched subsurface sediments that could act as a new nutrient sink (Needelman et al, 2006). Several processes — such as sediment sorption and desorption — occur that may influence P transport through aquatic systems (Froelich 1988). We sought to elucidate the role of fluvial processes in P transfers and their role in modifying downstream water quality impacts when sediments are physically removed by dredging.

Material and methods

Sediments were collected in October 2006 from dredged and undredged ditches located on the UMES Research Farm on the campus of the University of Maryland Eastern Shore, Princess Anne, MD, USA. The vegetative cover was removed before sediment collection in the undredged ditch. Sediments were stored wet at 278 K until placed in a purpose built fluvium described by McDowell and Sharpley (2003). Two replicates of each sediment were placed into separate flumes to a depth of approximately 5 cm. Each flume was filled with 200 L of tap water (P less than 0.005 mg L⁻¹ detection limit) and flow pumped over the sediment at 0.17 L s⁻¹ (which reflects the base flow of the field sites) for 48 hours. After this initial equilibration period, water in the flumes was replaced with 200 L of simulated runoff water, enriched P (2.5 mg L⁻¹) from poultry manure. This water was recirculated for 48 hrs to

evaluate the adsorption of P by the sediments (hereafter referred to as adsorption experiment). At the end of the adsorption experiment, the water was again replaced, this time with 200 L of deionized water so that P desorption from the could be examined (hereafter referred as the desorption experiment). Water samples were collected during all phases (equilibrium, sorption and desorption) using automated water samplers (Model 3700; Isco, Lincoln, NE) at varying intervals over 48 hours. Particle size distribution and Mehlich-3 P of sediments were determined. Water samples were filtered and analysed for dissolved reactive phosphorus (DRP).

Results and discussion

Sediments from the undredged ditch were finer in texture than those exposed in the dredged ditch, and had a lower concentration of Mehlich-3 P, confirming the removal of enriched sediments by dredging (Table 1).

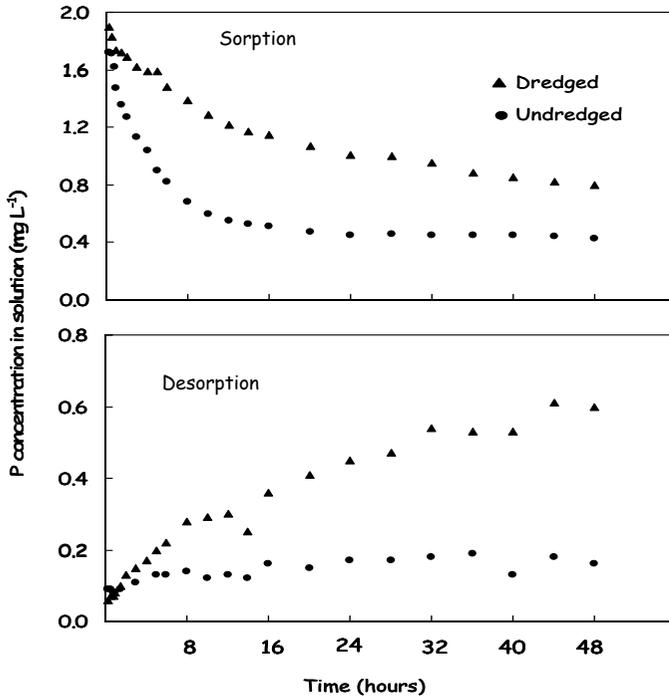
Table 1. Particle size distribution and M3-P content for dredged and undredged sediments.

Sediments	Sand	Silt	Clay	M3-P*
	----- % -----			mg kg ⁻¹
Dredged	68	17	15	53
Undredged	19	47	34	117

* Mehlich-3 extractable P.

The greatest declines in P concentrations in the water column were observed during the first 8 hours of the soption experiment for both dredged and undredged sediments (Fig. 1). After 48 hours of sorption, P concentrations in solution for undredged sediments were significant less than P concentrations in dredged sediments ($P<0.01$). These results show the higher capacity of the undredged sediments in removing P from the system, despite their higher Mehlich-3 P, pointing to a greater overall P sorption capacity, consistent with greater clay content.

More P desorption was observed from the dredged sediments, and P desorption from these sediments was continuous across the 48 h. In contrast, equilibrium P desorption from the undredged sediments was > 3 times lower than that observed from the dredged sediments, stabilizing after approximately 4 h. These results indicate that the dredging of ditches removes sediments that are better at buffering dissolved P in flow and, by forming relatively stable complexes, support lower equilibrium P concentrations. Similar results were observed by Smith et al. (2006) when comparing P sorption and desorption from exposed sediments before and after dredging.



Conclusions

Our research sheds light on the effects of ditch dredging on P transport in a coastal plain setting. As the regolith was underlain by coarse marine materials, dredging resulted in the exposure of materials with relatively low capacity to sorb P from simulated runoff water. When base flow conditions were simulated, P desorption from the coarse sediments was much higher than that observed from the finer textured sediments representing conditions before dredging. Results highlight the potential for well managed ditches to buffer nutrient transfers from agricultural soils to downstream water bodies.

References

- McDowell, R.W., Sharpley, A.N., 2003. Uptake and release of phosphorus from overland flow in a stream environmental. *J. Environ. Qual.* 32, 937-948.
- Smith, D.R., Warnemuende, E.A., Haggard, B.E. & Huang, C., 2006. Dredging of drainage ditches increases short-term transport of soluble phosphorus. *J. Environ. Qual.* 35, 611-616.
- Mozaffari, M. & Sims, J.T., 1994. Phosphorus availability and sorption in an Atlantic coastal plain watershed dominated by animal-based agriculture. *Soil Sci.* 157, 97-106.
- Needelman, B., Kleinman, P. & Allen, A., 2006. The ditch project. In: Needelman, B., S. Will. Improved management of agricultural drainage ditches for water quality protection. Field tour guide. The Ditch Project's Field Tour and Symposium. Princess Anne and College Park, MD. p. 1-45.

Phosphorus loss in a reclaimed marsh soil as affected by irrigation and Ca amendments

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Introduction

A significant portion of the Guadalquivir river marshes in SW Spain has been reclaimed for agricultural use (40000 ha from a total surface of 140000 ha), accounting for one of the largest reclaimed marsh areas in southern Europe. Before reclamation, these soils were highly saline and had a shallow, extremely saline, water table (Moreno et al. 1981). Reclamation practices involved artificial draining (tile-drains), leaching and soil amendment to reduce Na saturation. Phosphogypsum (PG), a by-product of the P fertilizer industry, has been effective in reducing sodium saturation in marsh soils from SW Spain (Domínguez et al. 2001). The main objective of this work was to study P losses in a representative soil of this area, and how losses are affected by irrigation management and Ca amendment.

Materials and methods

The experiment was conducted on a commercial farm in the Marisma of Lebrija, in the estuarine region of the river Guadalquivir, SW Spain (36°56'N, 6°7'W). Main soil properties in the experimental site were: (i) from 0 to 30 cm depth, 70 % clay content; 24 % carbonate content; 19 mg kg⁻¹ Olsen P; (ii) from 30 to 90 cm depth, 47 % clay content; 35 % carbonate content; 12 mg kg⁻¹ Olsen P. The drainage system was installed in 1977 and phosphogypsum (PG) amendment started (25 Mg ha⁻¹ every two years).

Two complete randomized blocks experiments were performed: one between 1999 and 2000, and the other between 2003 and 2004, with a typical biannual crop rotation (sugar beet-cotton). The experiments involved two (1999-2000) or three replications (2003-2004), and one factor (Ca amendment: control without application and PG at the usual rate, 25 Mg ha⁻¹). Phosphogypsum was applied before seeding of each crop, except in sugar beet grown in 2004, the amount of P applied with PG being 100 kg ha⁻¹ (most water soluble). Cotton was grown under furrow irrigation and sugar beet under sprinkler irrigation. Applied P fertilizer was 70 kg P ha⁻¹ in all the cases, and irrigation + rain was 1003 mm in 1999, 900 mm in 2000, 1116 in 2003, and 890 in 2004.

Results and discussion

Tile drain installation enhanced P losses related to preferential flow, particularly in cracking soils such as those of the area (Delgado et al., 2006). The fraction of applied water lost through drainage during the growing season was higher with cotton under furrow irrigation (35 % in 2000 and 10 % in 2004, on average) than with sugar beet under sprinkler irrigation (3 % in 1999-2000, and 3.5 % in 2004). The difference can be ascribed to an increased flow through cracks under furrow irrigation.

Table 1. Drainage and concentration of P forms in drainage water in different crops grown in different agronomic years.

Crop and year	Drainage		DRP		DTP		TP	
	Control	PG	Control	PG	Control	PG	Control	PG
	mm		mg L ⁻¹					
Sugar beet 1999	25	29	0.068 ± 0.008	0.073 ± 0.008	0.116 ± 0.026	0.136 ± 0.009	0.123 ± 0.066	0.152 ± 0.110
Cotton 2000	321	299	0.039 ± 0.024	0.043 ± 0.023	0.053 ± 0.024	0.057 ± 0.025	0.077 ± 0.048	0.079 ± 0.052
Cotton 2003	82	101	0.046 ± 0.026	0.043 ± 0.023	0.054 ± 0.027	0.049 ± 0.023	0.080 ± 0.070	0.102 ± 0.142
Sugar beet 2004	13	15	0.041 ± 0.029	0.030 ± 0.023	0.054 ± 0.038	0.050 ± 0.040	0.109 ± 0.180	0.076 ± 0.083

DRP, dissolved molybdate reactive P; DTP, dissolved total P; TP, total P
Means and standard deviations

Table 2. Phosphorus loss in drainage water in different crops grown in different agronomic years.

Crop and year	DRP		DTP		TP	
	Control	PG	Control	PG	Control	PG
	kg ha ⁻¹					
Sugar beet 1999	13c	18c	19c	28bc	19c	28c
Cotton 2000	113a	109a	154a	153a	211a	200a
Cotton 2003	35b	40b	39b	43b	51b	61b
Sugar beet 2004	5c	6c	7c	7c	9c	12c

DRP, dissolved molybdate reactive P; DTP, dissolved total P; TP, total P

Means followed by the same letter in the same column are not significantly different according to the LSD test (P < 0.05)

Non-significant differences were observed between treatments and years in the concentration of different P forms (Table 1). Although phosphogypsum amendment accounted for a significant P supply to soil, non-significant effects were observed on P losses through drainage due to the application of this amendment. Significant differences in P lost were observed between years (Table 2): year 2000 and 2003 showed greater P losses than the others. This can be ascribed to a higher drainage in these two years, mainly related to the use of furrow irrigation. This irrigation system promoted a greater loss of applied water by drainage. Thus, it can be concluded that water efficiency use in irrigation (mainly affected by irrigation system and cracking) was a key factor explaining P losses, more than the amount of P applied as fertilizers or amendments (PG treatments received twice the soluble P than that of Control), in agreement with previous results by Delgado et al. (2006).

References

- Domínguez, R., del Campillo, M.C., Peña, F. & Delgado, A., 2001. Effect of soil properties and reclamation practices on phosphorus dynamics in reclaimed calcareous marsh soils from the Guadalquivir Valley, SW Spain. *Arid Land Res. Manage.* 15, 203–221.
- Moreno, F., Martín, J. & Mudarra, J.L., 1981. A soil sequence in the natural and reclaimed marshes of the Guadalquivir river, Seville (Spain). *Catena* 8, 201–211.
- Delgado, A., Hurtado, M.D. & Andreu, L., 2006. Phosphorus loss in tile drains from a reclaimed marsh soil amended with manure and phosphogypsum: *Nutr. Cycl. Agroecosys.* 74, 191-292.

Reclaimed wetlands and the uncertainties of the European policy: environmental risk related to rewetting of reclaimed marshes

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Introduction

Wetland reclamation has resulted in significant physical and chemical changes in soils and also in the enrichment in nutrients as a result of fertilization. In northern Europe, peatland reclamation has been a typical example of wetland reclamation during the last decades (million of hectares have been reclaimed in temperate European regions). In southern Europe, important reclamation projects have been developed on marsh areas, where, besides a shallow watertable, salinity and sodicity constrain their potential agricultural use.

Changes in agricultural and environmental policies have promoted rewetting programmes in reclaimed peatlands resulting in an increased risk of nutrient loss, particularly phosphorus. On reclaimed marsh soils from Southern Europe, rewetting has not been an issue, yet. However, also here the potential environmental risk due to P release increases significantly under rewetting. The main objective of this work was to do a comparative study of P forms in reclaimed marsh soils and representative agricultural soils from Southern Europe with respect to estimating the P release potential of those soils, particularly under reducing conditions.

Materials and methods

Seventeen soils from the Guadalquivir Valley, South Spain (Palma del Río, 37°43'N, 5°13'W) were selected in such a way as to include those most typical for Mediterranean areas as per the Soil Taxonomy (Soil Survey Staff, 1998). Also, 12 different marsh soils located in the Marisma of Lebrija, in the estuarine region of the river Guadalquivir, SW Spain (36°56'N, 6°7'W), were studied. After reclamation, the soils studied can be classified as Aeric Endoaquepts (Soil Survey Staff, 1998).

A sequential extraction of P was performed according to Ruiz et al (1997). The process involves eight extractions using

- (i) 0.1 M NaOH + 1 M NaCl (NaOH-P), which releases adsorbed P, P precipitated as Fe- and Al-phosphates, and P bound by Fe and Al organic complexes;
- (ii) 0.27 M Na citrate + 0.11 M NaHCO₃ (CB-P), which extracts adsorbed P and highly soluble Ca-phosphates, partly precipitated or adsorbed on calcite after the NaOH extraction in calcareous soils;

- (iii) 0.25 M Na citrate at pH 6;
- (iv) 0.2 M Na citrate pH 6 (C-P), which releases pedogenic Ca phosphates not dissolved by CB;
- (v) 0.2 M Na citrate + 0.05 M ascorbate at pH 6 (CA-P, “mild reductant soluble P”), which releases mostly P occluded in poorly crystalline Fe oxides;
- (vi) 0.27 M Na citrate + 0.11 M NaHCO₃ + 2 % Na dithionite (CBD-P, “reductant soluble P”), which releases P occluded in crystalline Fe oxides;
- (vii) 1 M NaOAc buffered at pH 4 (OAc-P), which releases residual pedogenic Ca phosphates previously not dissolved by citrate; and
- (viii) 1 M HCl, which dissolves most lithogenic apatite.

Iron in CA extracts can be ascribed to poorly crystalline Fe oxides and that in CBD extracts to crystalline Fe oxides.

Results and discussion

In all the studied soils, Ca-related P accounts for the dominant P form in soil. Phosphorus occluded in Fe oxides accounted, on average, for 17 % of inorganic P in marsh soils and for 22 % of inorganic P in the other soils. However, in marsh soils 69

Table 1. Estimated amounts of different P forms in different representative soils from South Spain and in reclaimed marsh soils from the Guadalquivir Valley.

Representative soils	Adsorbed P + soluble Ca phosphates	Low soluble Ca phosphates	Lithogenic Fluorapatite	Occluded in poorly crystalline Fe oxides	Occluded in crystalline Fe oxides	Organic P	Total P
	----- mg kg ⁻¹ -----						
Aquic Alfisols n = 7	95 ± 50	73 ± 72	12 ± 6	12 ± 10	45 ± 20	248 ± 97	485 ± 163
Calcic Alfisols n = 6	98 ± 43	98 ± 66	13 ± 4	8 ± 3	48 ± 19	269 ± 146	534 ± 249
Vertic soils n = 4	126 ± 51	114 ± 17	19 ± 15	15 ± 5	40 ± 12	239 ± 42	554 ± 51
Reclaimed marsh soils n = 12	133 ± 43	402 ± 85	60 ± 5	82 ± 11	37 ± 10	124 ± 43	838 ± 104

Means and SD

Inorganic P forms estimated according to the P fractions determined according to the sequential fractionation scheme proposed by Ruiz et al. (1997)

Adsorbed P + P in soluble Ca phosphates estimated as the sum of NaOH extractable P and citrate-bicarbonate extractable P

P in low soluble Ca phosphates estimated as the amount of P extracted in two consecutive extractions with citrate and one extraction with acetate

P in lithogenic fluorapatite estimated as the amount of P extracted with HCl

P occluded in poorly crystalline Fe oxides estimated as the amount of P extracted with citrate-ascorbate

P occluded in crystalline Fe oxides estimated as the amount of P extracted with citrate-bicarbonate-dithionite

% of occluded P was related to poorly crystalline Fe oxides (CA-P), while in the other soils only 20 % was occluded in these Fe oxides (Table 1), which are supposed to be more readily reducible under water-saturated conditions.

The P/Fe mole fraction in CA extracts (poorly crystalline Fe oxides) was greater in marsh soils, thus indicating a higher occlusion rate, probably due to a higher rate of precipitation of Fe oxides after land reclamation (oxidizing conditions, Figure 1). The negative relationship between the P/Fe mole ratio in CA extracts and the ratio of CA-extractable Fe to total extractable Fe probably suggested that P occlusion was determined by the formation of poorly crystalline Fe oxides (Figure 1).

If adsorbed P, soluble Ca phosphates, and CA-P are taken into account, the potentially releasable P accounts for 35 % of inorganic P in marsh soils. Although a sizeable portion of the initially Fe-related P can precipitate as Ca phosphates, the P release potential is going to be significantly increased under reducing conditions. This risk should be considered under a perspective of set-aside from agricultural use promoted by the EU agricultural policy or the rewetting of these lands with the objective of recovering their original ecological diversity.

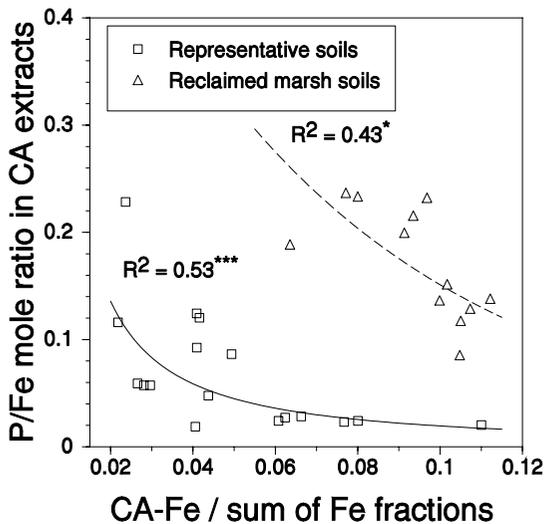


Figure 1. P/Fe mole ratio in CA extracts as a function of the ratio of CA extractable Fe to the sum of Fe fractions.

References

- Ruiz, J.M., Delgado, A. & Torrent, J., 1997. Iron-related phosphorus in overfertilized European soils. *J. Environ. Qual.* 26, 1548-1554.
- Soil Survey Staff, 1998. *Keys to soil taxonomy*, 8th edn. US Government Printing Office, Washington.

Restored floodplains as P buffers

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Introduction

The phosphorus (P) concentration and load in rivers, lakes and estuaries in Denmark is often too high to reach a good ecological quality as required by the EU Water Framework Directive. A reduction of the diffuse P input to the surface waters will in many cases be difficult because of the continuous enrichment of soils with agricultural P that has taken place during the last 60 years. Overbank storage of sediment-associated P may, therefore, be an important buffer for an excess P transport in river systems. The goal of this study is to assess the deposition of sediment and P on a restored floodplain in the Odense River and to estimate the storage efficiency compared to the amount of P transported by the river. The studied floodplain of the Odense River, between Brobyværk and Hillerslev Bro, was restored four years ago by remeandering the river (Figure 1). The restoration project involved 68 ha of riparian areas being in many cases transformed from intensive arable land to extensive grassland. Since the restoration, the floodplain has been inundated during part of the winter period.

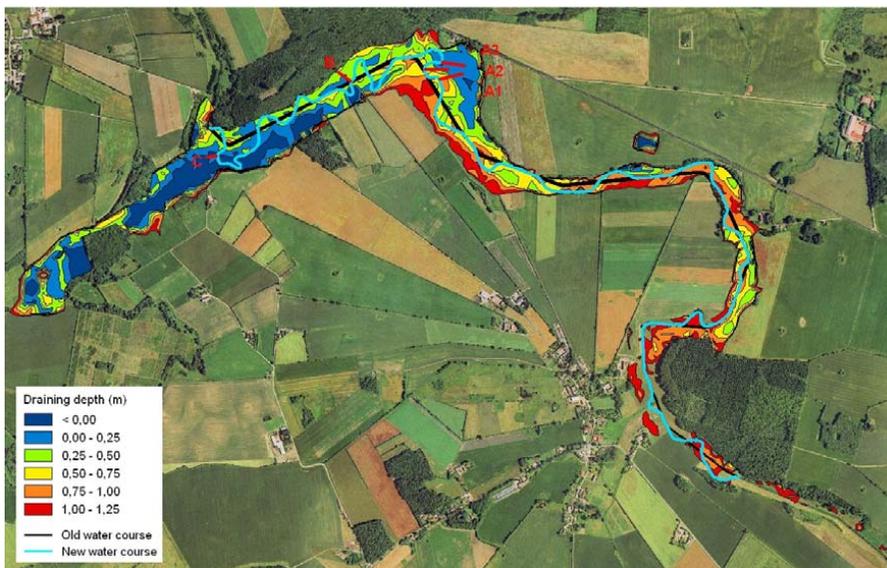


Figure 1. Study area with the new and the old water course. In 2004–2005, the deposition rate has been measured in five locations (Transects A1, A2, A3, B and C).

Materials and methods

The deposition of sediment and P was measured on artificial grass mats (15 × 15 cm) installed on the floodplain for two separate years (the winter of 2003–2004 and 2004–2005). The grass mats were installed in five transects on the floodplain perpendicular to the river channel for the entire winter period (October to April). Several flooding events took place during the two winter periods as measured at a stage monitoring station (Figure 2). After removal of the grass mats, the amount of deposited sediment on the grass mats was measured and the total P concentration was determined in the laboratory.

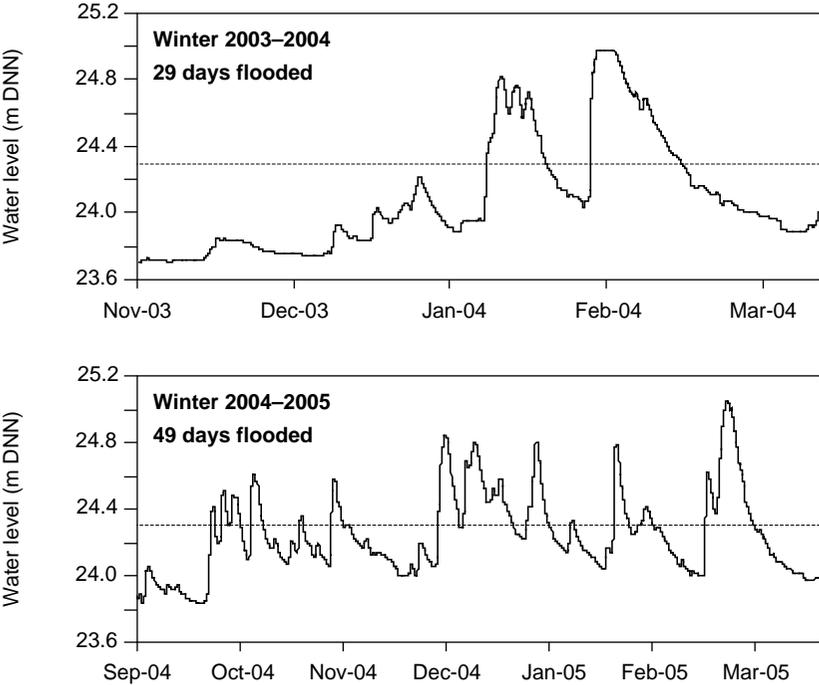


Figure 2. Water level during the two studied winter periods of 2003–2004 and 2004–2005. A water level of more than 24.3 m indicates that inundation of the floodplain starts.

Flood periods and deposition results

The total number of days with flooding was lower in the winter of 2003/2004 than in winter 2004/2005 (Figure 2). The average flooding depth was, however, higher during the winter of 2003/2004 than the winter of 2004/2005 (Figure 2). A sediment and particulate P deposition of up to 180 kg DW m⁻² and 41 g P m⁻² was measured for a single grass mat placed near the river channel. The average deposition rates of sediment and particulate P per transect were similar for both winters (Table 1).

Table 1. Average deposition rates of sediment and phosphorus per studied transect in the winters of 2003-2004 and 2004-2005.

	Sediment (g DW m ⁻²)		Phosphorus (g P m ⁻²)	
	2003-2004	2003-2004	2003-2004	2003-2004
A1	n.m.	0	n.m.	0.00
A2	n.m.	398	n.m.	0.80
A3	12609	11603	7.80	8.83
B	0	1784	0	1.53
C	1923	2174	4.66	5.65

n.m.=not measured.

Modelling sediment and P deposition

Estimation of the total sediment and particulate P deposition on a floodplain can only be reliably assessed using some kind of deposition model. Four different empirical deposition models were established based on the results from the winter of 2004/2005. The deposition of sediment and particulate P was based on explanatory variables such as distance to the river, flooding depth and coarse material deposition within the first 15 m of outer bends of meanders. Using the four different models, a sediment deposition of 450–1100 t and a particulate P deposition of 250–1100 kg were estimated for the entire floodplain.

Conclusions

This study shows that the annual suspended sediment and total P transport in the Odense river decreases with, respectively, 4.7-7.9 % and 0.4–1.4 % due to the deposition of sediment and particulate P on the inundated and restored 68 ha floodplain. This implies that natural and restored floodplains can buffer the transport of sediment and P in the rivers.

Liberation of phosphorus from sediment deposited after flooding

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Introduction

The amount of sediment and sediment-associated nutrients deposited during inundation of floodplains is generally high and may account for a substantial amount of the total annual sediment and nutrient load. Kronvang et al. (2007) reported that 28-47 % of the total suspended sediment export from four Danish rivers was entrapped on the floodplains during overbank flood events, while the storage efficiency for phosphorus varied from 4 to 7 %. Tockner et al. (1999) found that retention of suspended solids on the Danube River floodplain, downstream of Vienna, amounted to 25 kg m⁻² year⁻¹ or 50% of the load. Walling (1999) found for the River Ouse and its primary tributaries that floodplain storage accounted for 39.5% of the total amount of suspended sediment delivered to the main channel.

Studies of soil cores sampled from inundated floodplains seem to show that the retention of particle-bound phosphorus to some extent is permanent, although French studies have shown that liberation/mobilisation of dissolved phosphorus may take place during periods following low water, e.g. first autumn flood (Brunet & Astin, 1998, 2000).

In this study we have focused on the fate of phosphorus following sedimentation on the floodplain. Dried sediment samples were incubated with artificial rainwater and desorption of phosphate to the water phase was followed.

Experimental design

The laboratory experiment was carried out with deposited sediment sampled on the floodplain at a restored reach of Odense River (Funen, Denmark). Sediment was trapped on 15 x 15 cm artificial grass mats (AstroTurf®) placed along transects laid out at right angles and at increasing distances to the river (1 m, 8 m, 10.7 m, 16.5 m, 23.8 m, 31.1 and 40.4 m).

The sediment was removed from the grass mats and oven dried. Triplet samples of approximately 1.0 g dried sediment were weighed out and put into 1000 ml beakers. Due to a high content of sand at a distance of 1 m, the sample weight was increased to 5 g, and due to scarcity of sediment, only one sample from a distance of 40 m was included. The beakers were filled with artificial rainwater (1000 g). The laboratory procedure was as follows: Every day the beakers were stirred, and subsequently the

sediment was allowed to settle before sampling of water from the beakers took place. At day 6 and 13 the beakers were emptied, and the sediment dried at 105 °C, before 1000 g of artificial rainwater again was added to the beakers. Water samples were analysed for phosphate-P according to Danish Standards Association (DS 1189). Total-P in sediment was determined as phosphate-P after combustion in a muffle furnace at 550 °C followed by boiling in mild hydrochloric acid (Andersen, 1976).

Results

Phosphate is liberated from the sediment to the water phase gradually both during the first period (day 1 – day 5), the second period (day 8 – day 12) and the third period (day 15 – day 26) as shown in figure 1. The measurements from day 19 and day 23 show that accumulated phosphate liberation decreased and phosphate was resorbed to the sediment. This decrease might be a result of the laboratory procedure with stirring before sampling, as there was a four-day break in water sampling.

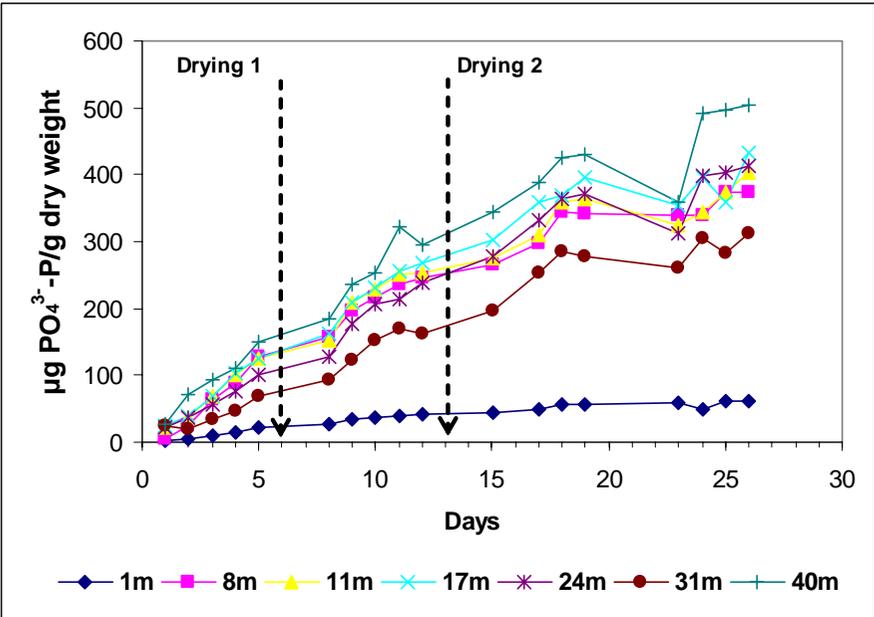


Figure 1. Average accumulated amount of dissolved inorganic-P desorbed from the incubated sediment to the water phase during a 26-day experimental period. At day 6 and 13 the beakers were emptied for water and the sediment dried. The overbank deposited sediment used in the experiment was sampled at different distances from the river as shown in the legend below the graph. For all sample points n=3, except at distance 40 m with only one sample due to scarcity of sediment.

The amount of phosphate desorbed from the sandy sediment at 1 m distance from the river is much smaller than the amount desorbed from all other sample points at greater distances from the river (Table 1). The amount of desorbed dissolved phosphate-P as compared to the total amount of phosphorus in the sediment before incubation varies between 11.4 % and 24.7 %, the three highest liberations taking place at 8, 11, and 17 m distances from the river.

Table 1. Total amount of PO_4^{3-} -P desorbed from the sediment, TP in sediment after and before incubation, percentage P dissolved as compared to TP in sediment.

Distance from river meter	Dissolved-P $\mu\text{g PO}_4^{3-}\text{-P g DW}^{-1}$	Incubated P in sediment $\mu\text{g P g DW}^{-1}$	P in sediment + dissolved-P $\mu\text{g P g DW}^{-1}$	Not incubated sediment $\mu\text{g P g DW}^{-1}$	% P dissolved	% P recovered
1	60.6	149.4	210.0	411.6	14.7	51,0
8	373.7	788.1	1161.8	1515.8	24.7	76,6
11	403.0	1585.2	1988.2	1817.4	22.2	109,4
17	433.1	1328.9	1762.0	2240.3	19.3	78,7
24	413.6	1624.2	2037.8	2796.2	14.8	72,9
31	312.7	2815.1	3127.8	2754.6	11.4	113,5
40	503.2	3479.7	3982.9	3599.5	14.0	110,7

Conclusion

Our 'worst case' batch experiments show that newly deposited sediment released 11 – 25 % of the total amount of P trapped on the artificial grass mats. Alternating wetting and drying of deposited sediment may be of importance for the desorption of phosphate from sediment as indicated by the two drying treatments.

References

- Andersen, J.M., 1976. An ignition method for determination of total phosphorus in lake sediments. *Water Research* 10, 329-331.
- Brunet, R.C. & Astin, K., 1998. Variation in phosphorus flux during a hydrological season: the river Adour. *Wat. Res.* 32(3), 547-558. *Regulated Rivers: Research & Management* 16, 267-277.
- Brunet, R.C. & Astin, K., 2000. A 12-month sediment and nutrient budget in a floodplain reach of the river Adour, southwest France.
- D.S., 1997. *Water Quality – Determination of phosphorus – Ammonium molybdate spectrometric method (identical to European Standard EN 1189:1996)*. Danish Standards Association, Copenhagen.
- Kronvang, K., Andersen, I.K., Hoffmann, C.C., Pedersen, M.L., Ovesen, N.B. & Andersen, H.E., 2007. Water exchange and deposition of sediment and phosphorus during inundation of natural and restored lowland floodplains. *Water Air Soil Pollut.* 181, 115-121.
- Tockner, K., Pennetzdorfer, D., Reiner, N., Schiemer, F. & Ward, J.V., 1999. Hydrological connectivity, and the exchange of organic matter and nutrients in a dynamic river-floodplain system (Danube, Austria). *Freshwater Biology* 41, 521-535.

Walling, D.E., 1999. Linking land use, erosion and sediment yields in river basins. *Hydrobiologia* 410, 223-240.

Phosphorus losses in surface runoff under different grazing pressures on a volcanic soil from Chile

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Introduction

The Lake Region in southern Chile (39 to 44 S and 71 to 74 W) has suitable conditions for cattle production. Thus, more than 45% of the national cattle herd is concentrated in this Region (INE, 1997), grazed on natural and improved pastures.

Volcanic soils are widespread in southern Chile accounting for 60% of the total arable land. These soils have high phosphorus (P) fixation capacity (85-90% of the P applied as fertilizer), high organic matter (OM) content and low water pH (Escudey *et al.*, 2001). In fact, the P fixation capacity of Chilean volcanic soils has created the perception of low P transfer to water, so that no Chilean studies have been conducted on the transference of P from grazed pastures to surface water.

The aim of this study was to evaluate the quantity of P lost in surface runoff in beef production systems as affected by the grazing animals.

Materials and methods

The experiment was carried out at the National Institute for Agricultural Research (INIA), Remehue Research Centre (40°35' S, 73°12' W), during 2004 to 2006. Soil at the site is an Andisol of the Osorno soil series (Typic Hapludands; CIREN, 2003), which has >1 m depth, 20 mg kg⁻¹ of Olsen P (0-20 cm) and 18% of organic matter content.

Reactive phosphorus (RP) and organic phosphorus (OP) losses from grazed paddocks with immediate stocking rates of 63 and 191 steers ha⁻¹ day⁻¹ (c. 220 kg initial live weight) were determined on a volcanic ash soil (4% field slope), with the use of surface lysimeters (5x5 m), as described by Alfaro & Salazar (2007). Paddocks were grazed by Holstein Friesian steers (3.5 steers ha⁻¹) and animals were managed under rotational grazing on a 20-year-old permanent pasture that had always been used for grazing. The main species in the pasture were *Lolium perenne*, *Dactylis glomerata* and *Holcus lanatus*. Treatments were fertilized in autumn with 45 kg N ha⁻¹ (urea (46% N) in 2004 and sodium nitrate (16% N) in 2005 and 2006) and in spring (all years) with 22.5 kg N ha⁻¹ (sodium nitrate, 16% N) and 30 kg P ha⁻¹ (triple superphosphate, TSP, 46% P₂O₅).

Surface runoff samples were collected three times per week from the surface lysimeters. All individual samples were stored at 4°C until analysis. Reactive P (RP) was measured with the ascorbic acid method (Clesceri *et al.*, 1998) and it might include particulate P. Total P (TP) was determined by digestion with acid persulphate (method 8190 @Hach, 2000). Organic P (OP) was estimated as the difference between TP and RP for each sample. Total P losses were calculated as the product of runoff and P concentration in the respective samples.

Analysis of variance (ANOVA) was used to compare P concentrations and P losses between treatments over the three year period, using Genstat 7.1 as statistical package.

Results and discussion

Total drainage was 634, 941 and 809 mm in 2004, 2005 and 2006, respectively. Surface runoff amounted to only 1% or less of total drainage, in agreement with Alfaro & Salazar (2007). This was related to the low tendency of volcanic ash soils to generate surface runoff, in agreement with Dorel *et al.* (2000).

The high peak (25-200 mg L⁻¹) of RP concentrations after TSF addition was probably because of the direct transport of fertilizers granules in runoff after the application. The average P concentrations (Table 1) were greater than those reported by McColl *et al.*, (1977) for grasslands on volcanic soils. Peaks of OP concentration were measured during spring and they were probably related to the flush of organic matter mineralization produced at that time of the year, in agreement with Turner & Haygarth (2000).

Table 1. Average P concentration in surface runoff samples (mg L⁻¹) and total P losses (g P ha⁻¹ yr⁻¹) in the treatments for the period 2004-2006 (± standard error of the mean).

Treatment	63 steers ha ⁻¹ day ⁻¹	191 steers ha ⁻¹ day ⁻¹
Average P concentration in surface runoff samples (mg L ⁻¹)		
Reactive P	2 ± 0.4 a	1 ± 0.2 a
Organic P	0.2 ± 0.04 a	0.2 ± 0.03 a
Total P losses (g ha ⁻¹ yr ⁻¹)		
Reactive P	4 ± 3.8 a	4 ± 0.8 a
Organic P	2 ± 2.4 a	1 ± 0.2 a
Total	6 a	5 a

Different letters in columns indicate significant differences (P≤0.05)

Phosphorus losses measured ranged between 1 and 9 g P ha⁻¹ yr⁻¹. The overall average P loss was not significantly different between treatments (P>0.05; Table 1). Overall P losses were low when compared with results for grazed land in Europe

(Haygarth & Jarvis 1997) and New Zealand (McColl *et al.*, 1977). This can be associated mainly with the low amount of surface runoff produced in both treatments and the high P fixation capacity of the soil. In both treatments, total losses were affected by incidental P loss associated with spring P addition, in agreement with Haygarth & Jarvis (1997). Total losses were mainly as RP (67 to 80% of the total loss), in agreement with Sharpley & Rekolainen (1997).

Conclusions

The increase in the immediate stocking rate did not increase total P losses, because the treatments did not change P inputs as recycling. Total losses were low, ranging between 1-9 g P ha⁻¹ yr⁻¹ because of the low generation of surface runoff in the volcanic soil. Phosphorus was mainly lost as RP (73% as overall average).

References

- Alfaro, M. & Salazar, F., 2007. Phosphorus losses in surface runoff from grazed permanent pastures on a volcanic soil from Chile. *Soil Use Manage* (in press).
- CIREN, 2003. Descripciones de suelos, materiales y símbolos, estudio agrológico X Región Vol. II, Centro de Información de Recursos Naturales Santiago Chile.
- Clesceri, L.S., Greenberg, A.E. & Eaton, A.D., 1998. Standard methods for the examination of water and waste water. American Public Health Association, American Water Works Association, Water Environment Federation. United Book Press Inc. Baltimore USA.
- Dorel, M., Roger-Estrade, J., Manichon, H. & Delvaux, B., 2000. Porosity and soil water properties of Caribbean volcanic ash soils. *Soil Use Manage* 16, 133-140.
- Escudey, M., Galindo, G., Förster, J., Briceño, M., Diaz, P. & Chang, A., 2001. Chemical forms of phosphorus of volcanic ash-derived soils in Chile. *Comm. Soil Sci. Plant Anal.* 32, 601-616.
- Hach, 2000. Procedures Manual Colorimeter DR/890 for total P determination. Hach Company Baltimore USA.
- Haygarth, P.M. & Jarvis, S.C., 1997. Soil derived phosphorus from grazed grassland lysimeters. *Water Res.* 31, 140-148.
- INE, 1997. Resultados preliminares VI Censo Agropecuario. Instituto Nacional de Estadísticas, Impresos Universitarios SA Santiago Chile.
- McColl, R.H.S., White, E. & Gibson, A.R., 1977. Phosphorus and nitrate runoff in hill pasture and forest catchments, Taita, New Zealand. *N. Z. J. Marine Fresh Water Res.* 11, 729-744.
- Sharpley, A.N. & Rekolainen, S., 1997. Phosphorus in agriculture and its environmental implications. In: Tunney, H., Carton, O., Brookes, P., & A. Johnston (eds.). *Phosphorus loss from soil to water*. CAB International, Wallingford. Pp. 1-53.
- Turner, B.L. & Haygarth, P.M., 2000. Phosphorus forms and concentrations in leachates under four grassland soil types. *Soil Sci. Soc. Am. J.* 64, 1090-1099.

Forms of particulate P in urban and agricultural runoff to Lake Nordborg, Denmark

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Introduction

Many Danish lakes have received or are still receiving high amounts of nutrients (N and P) from external sources. A majority of the lakes became eutrophicated before a major reduction in P loading from sewage treatment plants was effectuated in the 1980s (e.g. Jeppesen et al. 1999). The lakes often remain in a eutrophic state due to internal P loading and continued high P loading from diffuse runoff such as agricultural drainage water and storm water from separate sewer systems. Both external and internal loading influence eutrophic Lake Nordborg, Denmark, where 25% of the watershed is drained by separate sewer systems and most of the remaining watershed is agricultural soils.

To establish a restoration plan for the lake a careful evaluation of the external loading was needed and the aim of this study was therefore to characterize and quantify the phosphorus loading to Lake Nordborg from urban runoff and agriculture. Since there are only few data in the literature on the bio-availability of particulate and dissolved P in urban runoff, a special aim of this study was to provide a measure of the potential bio-availability of P entering the lake.

Design of the investigation

Fourteen stations (six with urban runoff and eight in the agricultural watershed) were sampled four times during a winter season where 80% of the runoff occurs. Samples were analysed for dissolved (inorganic and organic) and particulate P and filtered particles were analysed with a sequential extraction procedure (Psenner, 1984; Jensen & Thamdrup, 1993). At one occasion the settling velocity of particles from five selected inlets with different catchments were determined. In 2006/2007 newly established precipitation-filtration ponds in two inlets have been monitored.

Phosphorus loading to Lake Nordborg

The concentration of TP was significantly higher in the inlets that drained agricultural areas (0.174 ± 0.032 mg P L⁻¹) than in the inlets from urban runoff (0.082 ± 0.019 mg P L⁻¹). Similarly, the concentration of PP (Table 1) was higher in agricultural drainage water (0.063 ± 0.015 mg P L⁻¹) than from urban runoff (0.03 ± 0.0131 mg P L⁻¹). Some 40 % of the P-loading occurred as PP, independent of catchment type. For urban runoff, 62 % of PP was in the form of surface adsorbed P, iron-bound P and extractable organic P (Table 1), which may be considered bio-available since these

forms are mobilized from lake sediments. The corresponding value for agricultural runoff was 75 %. Concerning TP, 72 % was bio-available in urban runoff and 77 % in agricultural runoff.

Table 1. PP in mg P/L⁻¹ and percentage distribution of P-fractions in particles from urban runoff and agriculture. The sum of fraction 1-3 is considered bio-available.

Catchment	Total PP	% of total	% of total	% of total	% of total	% of total	% of total
		1	2	3	4	5	6
Agriculture	0.063 (0.015)	10.2 (0.5)	43.4 (2.7)	21.6 (2.2)	7.8 (0.7)	7.4 (0.7)	9.5 (1.0)
Urban run-off	0,03 (0,0131)	7.8 (0.6)	35.2 (3.3)	19.7 (2.8)	13.1 (1.7)	9.9 (1.1)	14.4 (1.5)

The P-fractions are: 1: loosely adsorbed phosphate, 2: iron-bound P, 3: extractable organic P, 4: P bound to aluminium and clay, 5: calcium-bound P and 6: refractory organic P. Values in parenthesis represent standard error of mean.

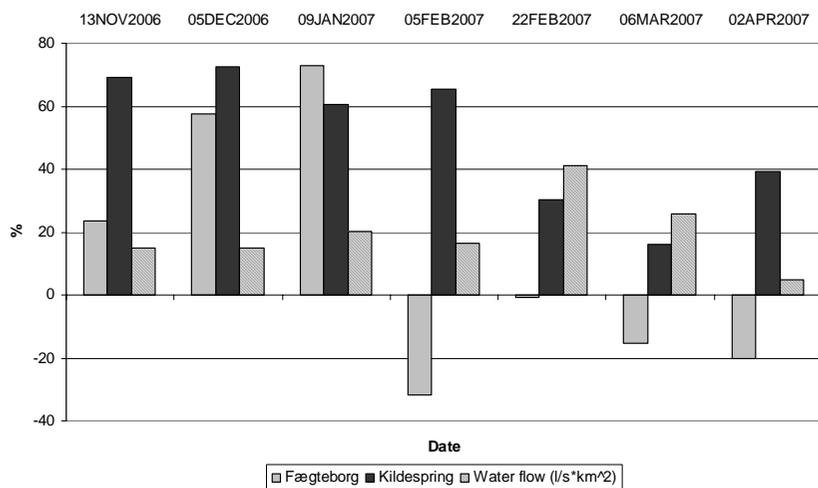


Figure 1. Particulate P removal efficiency in two wet ponds with sand filter. A positive value indicates a removal of P in the pond whereas a negative value means that the outlet concentration was higher than the inlet concentration. The relative average water flow in l/s*km² is indicated by the striped columns.

The diffuse loading of bio-available P to Lake Nordborg could potentially be reduced by ~30 % if particles were retained in the inlets, however, settling experiments have shown that the average settling velocity for particulate P was less than 1 cm/hour for

50 % of the particles which can make it difficult to reduce P-transport in retention ponds.

In autumn 2006 wet ponds with integrated sand filter were established in two of the inlets to Lake Nordborg. The preliminary results from the ponds indicate removal efficiency for Kildespring between 16 and 73 %. The picture is more negative for Fægteborg with removal between 23 and 73 % in Nov-Jan, whereas the last 4 measurements show P-release from the pond.

Conclusions

Seventy two percent of the total P in tributaries with urban runoff sampled through a winter was bio-available. Most was in the form of dissolved inorganic P but around one third was attached to particles in the form of surface adsorbed P, iron-bound P, or extractable organic P. Similarly, 77% of TP from tributaries draining agricultural soils was bio-available with a slightly higher contribution from particulate P.

A spring measurement in five tributaries shows that 30 to 70% of the particulate P had settling velocities less than 1 cm h^{-1} . Thus, detention ponds for storm water or drainage water either need to have very long water retention time or they should be combined with other treatment facilities like for example sand filters or wetlands to retain fine particles in order to significantly reduce the transport of bio-available P to downstream systems.

Data from two wet ponds with sand filter established in two of the inlets show particulate P removal around 20-70 % in autumn, but in the early spring the removal rates are lower or the ponds are releasing P, which to some extent can be explained by a higher water flow in spring.

References

- Jensen, H.S. & Thamdrup, B., 1993. Iron-bound phosphorus in marine sediments as measured by bicarbonate-dithionite extraction. *Hydrobiologia* 253, 47-59.
- Jeppesen, E., Søndergaard, M., Kronvang, B., Jensen, J.P., Svendsen, L.M. & Lauridsen, T.L., 1999. Lake and catchment management in Denmark. *Hydrobiologia* 395/396, 419-432.
- Psenner, R., Pucsko, R. & Sager, M., 1984. Die fraktionierung organischer und anorganischer phosphorverbindungen von sedimenten, *Archiv für hydrobiologie. Supplementband, Monographische Beiträge* 70, 111-155. (German)

Effect of soil use on the composition of circulating waters: the Fonte Espiño river basin (Galicia, NW Spain)

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Introduction

High concentrations of nutrients in surface waters can have serious effects on ecosystems, and the need to improve the quality of such waters has been recognised (EU Water Framework Directive, 2000). One important source of nutrients is the diffuse contamination from agricultural land. There is therefore a need to establish the relative importance of these inputs, which can be done by comparing sources of water that cross land where there is little agricultural activity, with those that cross intensively managed land. The composition of the water in two small rivers in the same basin was compared: one stream crosses forest land and the other agricultural land.

Material and methods

The rivers studied were the *Rego de Abellas* and *Fonte Espiño* (tributaries of the river Xallas, Galicia, NW Spain). The river *Rego de Abellas* stretches for 4 km before converging with the river *Fonte Espiño* (1.5 Km). The basin that drains the rivers occupies an area of 7.5 km². The topography of the basin is hilly (altitudes of between 300 and 400 m). The soils are Lithosols and Umbrisols developed on gneisses and granulitic rocks. Forest land use dominates in the *Rego de Abellas* sub-basin (85%) (*climax* forests of *Q. robur* L., reforestations of *P. pinaster* Aiton and *E. globulus* L.), whereas in the *Fonte Espiño* sub-basin, 90% of the land is dedicated to pasture and agricultural use (permanent grasslands with intensive pasture and organic fertilization, with some corn-grassland rotations). In the *Rego de Abella* system, the sampling points were at 0.60, 1.50, 2.25 and 3.50 km from the source of the river (points B1, B2, B 3 and B4 drain 1.40 km², 2.18 km², 2.60 km² and 3.30 km² of the sub-basin respectively). In the *Fonte Espiño* system, the sampling sites were at the source (0.00 km), and at 0.50 and 1.25 km from the source (points A1, A2, A3, drain 0.17 km², 0.31 km² and 1.72 km² of the sub-basin, respectively). Water samples were collected weekly throughout one year. The analytical determinations were carried out according to official techniques for analysis of water samples in Spain (Taboada *et al.*, 2002). The concentrations of the ions are expressed in mg l⁻¹.

Results

All of the parameters studied showed clear annual variation (Table 1). In both rivers, particularly the *Fonte Espiño* there was a progressive increase in the concentration of most of the parameters studied with increasing distance from the source. The greatest increase occurred in Ca, total C, sulphates and chlorides, whereas the lowest increases corresponded to total P and ammonium (Table 1). In general, there was a linear relationship between the concentrations and the area drained. The gradients (concentration/km²) obtained for each of the parameters were different in each of the rivers and were generally higher in the river *Fonte Espiño* than in the *Rego de Abellas* (Figure 1).

Table 1. Mean values (\pm s.d.) of the parameters measured throughout the period of study (mg l⁻¹).

	Fonte Espiño			Rego de Abellas			
	A1	A2	A3	B1	B2	B3	B4
pH	5.7 \pm 0.2	6.4 \pm 0.2	6.7 \pm 0.2	6.5 \pm 0.3	6.5 \pm 0.3	6.6 \pm 0.3	6.8 \pm 0.3
Total C	4.7 \pm 1.4	5.8 \pm 2.2	7.1 \pm 2.6	6.1 \pm 2.0	6.3 \pm 1.7	5.9 \pm 1.7	6.1 \pm 1.7
Sol. susp.*	1.2 \pm 2.5	3.6 \pm 5.1	2.4 \pm 2.9	1.1 \pm 1.5	2.0 \pm 4.6	1.8 \pm 2.5	2.0 \pm 2.9
Na ⁺	7.1 \pm 1.2	7.2 \pm 1.1	7.5 \pm 1.6	6.1 \pm 1.0	6.1 \pm 0.9	6.1 \pm 1.0	6.3 \pm 1.0
K ⁺	0.8 \pm 0.2	0.9 \pm 0.4	1.0 \pm 0.4	0.4 \pm 0.3	0.5 \pm 0.3	0.5 \pm 0.3	0.5 \pm 0.2
Ca ⁺²	1.5 \pm 0.8	2.7 \pm 2.0	4.1 \pm 3.1	1.8 \pm 1.2	2.0 \pm 1.2	2.0 \pm 1.5	2.3 \pm 1.7
Mg ⁺²	1.1 \pm 0.6	1.3 \pm 0.8	1.7 \pm 1.0	0.8 \pm 0.5	0.8 \pm 0.6	0.8 \pm 0.5	0.9 \pm 0.6
N-NH ₄ ⁺	0.14 \pm 0.09	0.18 \pm 0.10	0.21 \pm 0.14	0.17 \pm 0.11	0.17 \pm 0.12	0.18 \pm 0.15	0.20 \pm 0.12
N-NO ₃ ⁻	6.1 \pm 1.1	6.4 \pm 1.7	6.8 \pm 1.5	2.8 \pm 1.6	2.5 \pm 1.0	2.7 \pm 1.0	3.0 \pm 1.0
Total P	0.02 \pm 0.01	0.03 \pm 0.02	0.03 \pm 0.02	0.02 \pm 0.01	0.02 \pm 0.01	0.02 \pm 0.01	0.02 \pm 0.01
MRP	0.01 \pm 0.01	0.02 \pm 0.01	0.01 \pm 0.01	0.01 \pm 0.00	0.01 \pm 0.01	0.01 \pm 0.01	0.01 \pm 0.01
CO ₃ H ⁻	9.8 \pm 1.7	11.0 \pm 2.2	16.1 \pm 3.0	10.0 \pm 3.6	10.8 \pm 2.9	11.4 \pm 3.1	12.0 \pm 3.1
SO ₄ ⁼	2.9 \pm 1.4	3.9 \pm 1.4	4.5 \pm 0.9	3.3 \pm 1.0	3.4 \pm 1.0	3.6 \pm 1.2	3.4 \pm 0.8
Cl ⁻	7.7 \pm 5.9	8.1 \pm 5.6	8.1 \pm 5.7	6.2 \pm 4.2	6.3 \pm 4.2	6.3 \pm 4.0	6.5 \pm 4.4

* Solids in suspension

Discussion and conclusions

The increase per unit of drained area was greater in the agricultural sub-basin than in the forest sub-basin, which indicates the important effect of diffuse contamination on water quality. There are various factors that favour this increase. On one hand, the large amounts of manure used to fertilise the land dedicated to pasture, and on the other the basal fertilization carried out at the end of the grassland cycle, which accounts for the leaching of Ca, Mg, sulphate and bicarbonate, as well as the increase in pH. The intense increase in sulphate suggests that in the soils under study the processes whereby this anion is retained are not very important or that the previously fixed sulphate ions have been displaced by phosphate ions. There was very little leaching of P, with the increases in relation to the drained area being similar

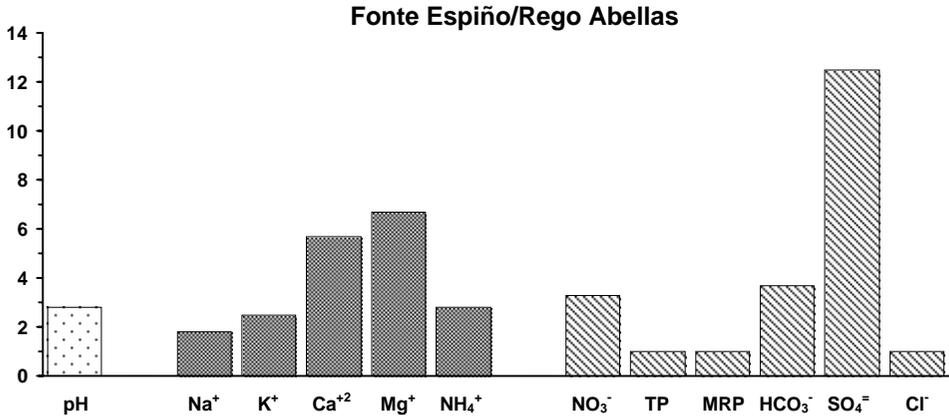


Figure 1. Relation between the gradients (concentration/km²) obtained for the agricultural area and the forest area for the different parameters studied.

in the forest and agricultural sub-basins, which suggests the presence of strongly bound P in the soils. In addition, in the agricultural sub-basin, the crops (both grass and other) reach the riverbanks where natural vegetation, which would act as a filter, is scarce in these areas. By contrast, the riverside vegetation is abundant in the forest sub-basin, covering both edges along almost the entire length of the river and preventing leached elements and eroded particles reaching the water. Despite the relatively high levels of nitrates in the agricultural sub-basin, the water is not severely contaminated.

References

- EU Water Framework Directive, 2000. Directive 2000/60 of the European Parliament and of the Council. Official Journal of the European Communities 2000, L327/1-72.
- Taboada, M.M., Diéguez, A. & Taboada, M.T., 2002. Effect of soil use and agricultural practices on heavy metal levels in surface waters. *Commun. Soil Sci. Plant* 35, 2833-2849.

Interactions between the phosphorus content of animal manure and losses of phosphorus to water

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Introduction

The feeding of imported concentrates in Irish dairy systems results in phosphorus (P) surpluses that contribute elevated levels of soil P as well as increased amounts of P in manures. However manipulation of concentrate composition can lower P content of diets without compromising animal performance (Ferris et al., 2005) and is therefore a mitigation option for P-sensitive catchments. Rainfall simulation studies conducted over the summer and winter seasons of 2006 investigated how varying the P content of cattle manure applied to grassland impacted on P concentrations in surface runoff.

Materials and methods

The study was based on manures from Holstein dairy cows fed four levels of dietary P from a mixture of grass silage and concentrates, this produced slurry P contents expressed as % dry matter (DM) of 1.3, 1.0, 0.9 and 0.5 % P. Twenty five runoff plots (1 x 0.5 m) were laid out in a Latin Square design on a hillslope site (slope 5%) under permanent pasture with five replications per treatment plus a control. Following surface applications of manure at a rate of 50 m³ Ha⁻¹ and 6.0% DM, rainfall simulations using a portable rainfall simulator of the Amsterdam design (Bowyer-Bower & Burt, 1989) were delivered using deionised water at a rate of 20 mm hr⁻¹ for a 30 minute period on days 2, 9 and 31 after application to examine the persistence of the P signal over time. Composite runoff samples were analysed for Total P (TP) and dissolved reactive P (DRP) using standard methods.

Results and discussion

Compared to the summer experiments the application of manure in the winter experiment was followed by substantially larger concentrations of TP in runoff. A reason for this may be the differing antecedent soil moisture prior to rainfall simulation. In the summer the soil was drier giving a greater opportunity for the manure P applied to percolate into the soil profile. Overall this had the effect that the period between initiation of rainfall and commencement of runoff was longer in the summer, although runoff was generated at 9 minutes on average for the first rain event in the summer with little difference in the winter (11 minutes) due to natural

precipitation during the days when the first simulated events were performed in summer.

Despite this difference, patterns of TP loss were similar for both seasons, with concentrations declining markedly with time. As a result and in comparison to the control, the signal of elevated concentrations in runoff following application was of short duration, with significant differences in runoff TP concentrations in runoff only on the simulations on days 2 and 9 and by day 31 the differences with the control were negligible (Figure 1). However in the summer period there was a significant increase between day 2 and day 31 from 3.2 to 6.6 mg P L⁻¹ in the TP concentration from the runoff of the control treatment suggesting substantial natural variation in runoff and a reflection of soil P mineralization. This effect was relatively large as concentration of TP in runoff from the plots receiving manure were 2.5-4.9 mg P L⁻¹ greater than the control at 2.5 mg P L⁻¹ measured on day 9.

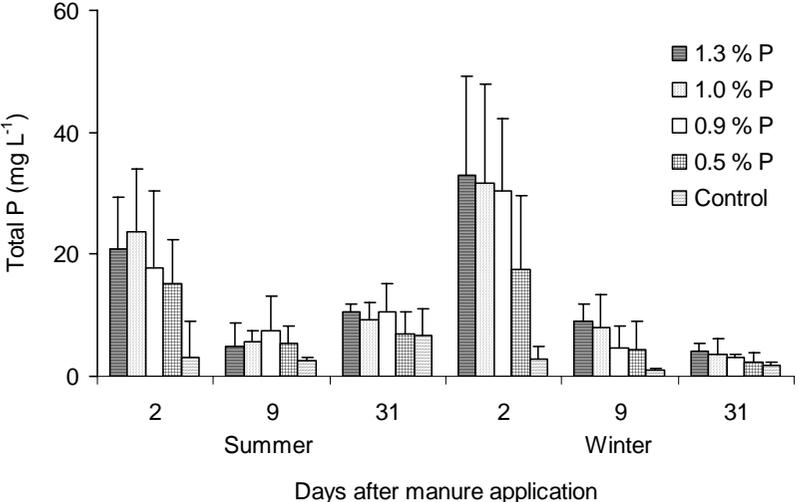


Figure 1. Mean TP concentration over 3 simulated events in summer and winter.

The largest impacts of varying manure P composition were most evident in runoff TP concentrations on day 2 of both seasons although there was not a significant difference in the runoff TP concentrations from the two highest manure P treatments (Table 1). Reductions of P in runoff were not proportional to the reductions of P in manure. A total reduction of 61% between the highest and lowest manure P treatments attained a 38% reduction in runoff TP in summer and 55% in winter.

While the ranking of DRP concentration followed that of the P content of the applied manure only under the lowest manure P treatment was DRP concentration in runoff significantly lower than from other treatments (Table 1) in summer. In all slurry

treatments DRP accounted for the majority of TP collected in runoff. The percentage of DRP was greatest at the highest P treatment (80% of TP) in summer while in winter DRP P contributed less (only 60% of TP).

Table 1. Concentrations of TP and DRP in runoff two days following manure application in summer and winter.

Slurry P content (%P)	Mean TP (mg P L ⁻¹) Summer	Mean DRP (mg P L ⁻¹) Summer	Mean TP (mg P L ⁻¹) Winter	Mean DRP (mg P L ⁻¹) Winter
1.3	9.82 ^a	7.83 ^a	13.77 ^a	8.28 ^a
1.0	8.32 ^a	6.07 ^a	12.56 ^a	7.86 ^a
0.9	6.21 ^b	6.69 ^a	7.52 ^b	3.78 ^b
0.5	5.94 ^c	3.26 ^b	6.17 ^c	2.18 ^c
Control	0.43 ^d	0.20 ^c	1.42 ^d	0.85 ^d
Probability	0.004	<0.001	<0.001	<0.001
SEM	1.34	0.732	1.05	0.572

Conclusions

Reduced P diets show a clear potential to reduce P concentrations in manure impacted runoff but this effect was relatively short-lived. The large drop in TP concentrations between days 2 and 9 suggests that strategies which lower the interaction between applied manure and runoff could have a greater impact on P losses compared to varying dietary P.

Acknowledgements

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References

- Bowyer-Bower, T.A.S. & Burt, T.P., 1989. Rainfall simulators for investigating soil response to rainfall. *Soil Technology* 2, 1-16.
- Ferris, C.P., McCann, M.E.E. & Patterson, D.C., 2005. Improving the efficiency of nitrogen and phosphorus use in livestock systems through dietary modification. *Agricultural Research Institute of Northern Ireland, 78th Annual Report*, 35-53.

P loss from land under tillage

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Introduction

Nutrient losses from grassland have received considerable attention in Irish agricultural research. It has been shown that where overland flow occurs on grassland, considerable quantities of P can be lost (Tunney et al, 1998, Kurz, 2000). The rate of P loss increases rapidly with increasing soil P level.

In Irish agriculture tillage is a minority activity. It represents approximately 10% of the farmed area. This tillage trial is the first of its kind in Ireland. The objective is to monitor P losses from a tillage rotation and to relate these to P loss in water. The period of the trial is three years, two of which are reported here.

Materials and method

The trial was set up on high yielding tillage land under winter wheat in the Southeast of Ireland. This soil was free-draining on grey brown podzolic soils underlain by glacial sands and gravels. Three plots of almost 1 ha each were laid out on gently sloping ground (3 to 5 %). Soil test P was the treatment and P fertilizer was applied at rates designed to give Soil P indices of 2, 3 and 4 on plots 1, 2 and 3 respectively.

Overland flow was collected using lined collection drains constructed along the base of each plot. Flow from the drains passed through V-notch flow-meters equipped with proportional samplers. The samplers recorded flow-rate and collected samples at a rate proportional to flow. Samples were tested for total P, to take account of P attached to particles. Sediment accumulated in the drains and in the water passing through the flow meters. It was collected from 10 strips in each drain. Each strip was 0.5 m long and covered the full width of the drain (approx 0.3 m). Only part of an overland flow sample was required for P analysis. The remainder was tested for sediment concentration by settlement followed by drying in ambient air and weighing of the residue.

Macro-pore samplers were constructed according to the design of Simmons and Baker (1993). This instrument consists of a 1.2 m long pipe with a water inlet close to the lower end. It is placed in a hole at an angle of 45° to the horizontal. This arrangement ensures the inlet is placed at the usual depth of a land-drain, 0.6 m below the soil surface. A mesh at the inlet is pressed by springs against the upper wall of the hole to capture water trickling from macro-pores. Bentonite placed around

the top of the hole, excludes overland flow. Eighteen samplers are distributed throughout the site on a grid.

Results

The attempt to establish P index values of 2, 3 and 4 on the three sites was only partly successful. Soil test P (STP) values in Table 1 indicate that plots 1 and 2 reached their target in one year at least. Plot 3 failed to reach P index 4 despite application of fertilizer at rates up to double that recommended.

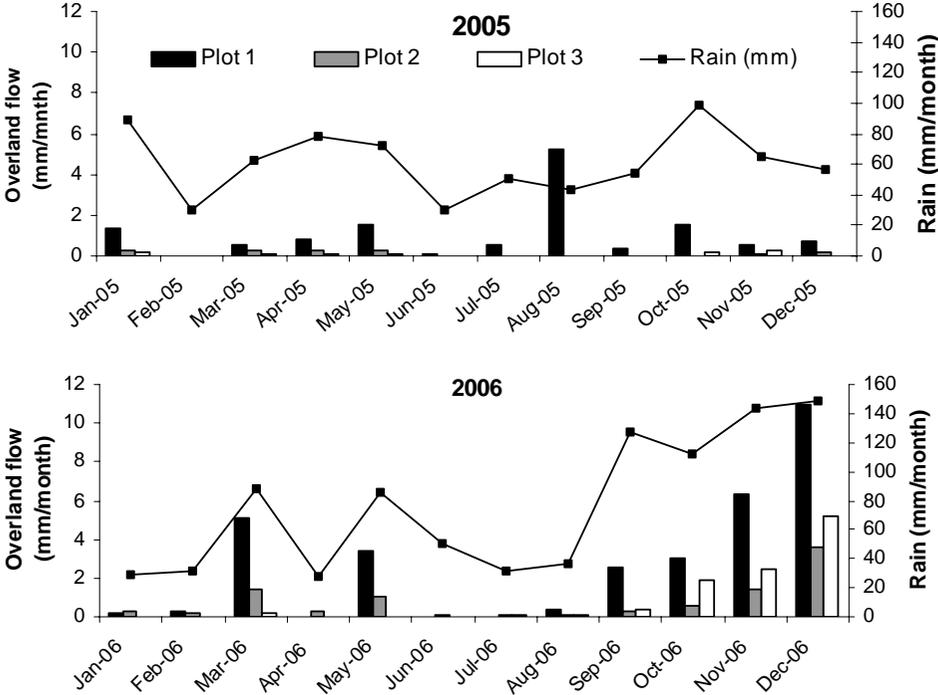


Figure 1. Overland flow and rainfall at a tillage site in Southeast Ireland.

Rainfall in 2005 and 2006 was 730mm and 910 mm respectively. The quantity of overland flow was small in 2005 for plots 1, 2 and 3 but it rose substantially in 2006 amounting to over three times the total flow in the previous year (Fig. 1). The rise in overland flow was large compared with the increase in annual rainfall of only 25%. Most flow in the second year occurred during frequent heavy rain in the last four months. A significant portion of flow was recorded in August/September of both years, when rivers were low. This amounted to 34% of total flow in 2005 and 7 % in 2006.

Table 1. Losses of P and sediment in overland flow.

Year	Plot	STP (mg/l) (P index)	Overland Flow (mm)	P conc. in water (mg/l)	P Export (kg/ha)	Sediment loss (kg/ha)
2005	1	4.2 (2)	13.3	1.24	0.165	146
	2	6.7 (3)	1.4	1.11	0.015	42
	3	6.9 (3)	1.2	5.08	0.8	59
2006	1	3.1 (2)	32.2	0.24	0.078	146
	2	5.4 (2)	9.3	0.42	0.039	54
	3	7.6 (3)	10.4	1.05	0.109	91

The influence of STP on P concentration in water is evident in Table 1. Plot 3 had the highest soil P values and the highest P concentrations in water for both years. The relativity was maintained in P export values but the effect of higher flows on plot 1 also boosted P export from that plot.

Sediment loss was low on all three plots in both years. Samples taken from the drain were sandy in nature but material settled from water samples ranged from sand to clay with sediment in some water samples taking many days to settle.

Phosphorus concentration in macro-pore samples in 2005 averaged 3.7, 6.6 and 3.2 mg/l for plots 1, 2 and 3, respectively. Concentrations rose slightly the following year to 7.97, 5.46 and 4.67 mg/l. Sampling was erratic with only 25% of sampling events producing samples. The volume of sample tended to increase with time but this could be due to erosion around the sampler inlet or increased rainfall at the end of 2006.

Discussion

This is the first detailed investigation concerning P loss from tillage land in Ireland. While the project has another year to run, measurements to date suggest that P loss from tillage should not be ignored especially where high STP occurs.

Variation in overland flow from one year to the next reflects rainfall. The sequence and intensity of showers as well as the overall amount are all likely to affect the amount of overland flow. Likely pathways for overland flow include surface crust in late summer and perched water table during winter. Further analysis is needed to explore these issues.

The calculations shown here are predominantly based on average values and are summary in nature. More detailed calculations are planned to show details in individual overland flow events. This is unlikely to affect the main results but will offer insight into what influenced them.

Conclusion

Significant quantities of P can be exported from tillage fields to surface water. The primary source of variation in overland flow over time is rainfall. Other sources will be explored in future analysis. Sediment loss was small and similar in both years. It showed little response to increased rainfall.

References

- Kurz, I., 2000. Phosphorus exports from agricultural grassland with overland flow and drainage. In: Quantification of phosphorus loss from soil to water. Environmental Protection Agency, PO Box 3000, Johnstown Castle, Co. Wexford, pp 9–56.
- Simmons, K.E. & Baker, D.E., 1993. A zero tension sampler for the collection of water in macropore systems. *J. of environmental quality* 22, 207–212.
- Tunney, H., Carton, O., O'Donnell, T. & Fanning, A., 1998. Phosphorus loss from soil to water. End of project report Armis 4022, ISBN No. 1 84170 012 X, Teagasc, Oak Park, Carlow, Ireland, 10 pp.

Influence of P-status and hydrology on phosphorus losses to surface waters on dairy farms in the Netherlands

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Introduction

The contribution of agriculture to the contamination of Dutch surface waters has increased from 43 % in 1985 to 57 % in 2002. Dairy farming is the largest producer of animal manure in the Netherlands (circa 75% of the total annual manure production). Information on nutrient budgets and leaching of nutrients from dairy farms was, however, limited. To mitigate this problem, monitoring programmes were set up at three dairy farms, namely one in the sandy area of the Netherlands, one in the peat district and one on a river-clay soil. The three farms together represent the environmental conditions encountered in the Dutch dairy region. This study focuses on the losses of phosphorus from grazed grasslands on a clay soil, a sandy soil and a peat soil, with different soil P-status and different hydrological pathways.

Experimental design

Phosphorus losses to ground- and surface waters were measured for a period of two to three years on a site with a heavy clay soil, a peat soil and a sandy soil. All sites were almost level and were draining on a dead-end ditch. At the end of each ditch a weir was placed with a flow meter connected to a sampling device for flow-proportional sampling. The sandy site was well drained and drain pipes were absent. About 3.5 m below the surface, a confining loam layer prevents exchange of water and solutes with deeper groundwater. The clay site was drained by tile-drains and trenches. The subsurface drains were located at a depth of 80 cm below the surface. The ditches were shallow (50 cm depth) and located at intervals of 46 m. Due to the low permeability of the heavy clay there was no seepage and groundwater recharge. The peat site has a man-made topsoil (0-40 cm) of organic matter-rich sandy clay, the subsoil consisted of woody peat. At 3 m depth a dense clay layer prevented extensive groundwater recharge. At the clay site the losses to surface water were determined by flow-proportional measurement of the discharge of trenches and drains. For sand and peat soil, the soil solution in the unsaturated zone was sampled using porous suction cups. Phosphate leaching fluxes at these sites were based on measured soil solution concentrations and simulated water fluxes. Surface runoff was assessed using catchment plates and simple balance models for the peat soil (Van Beek et al., 2003) and was modelled for the sand soil (Torenbeek and Voskamp,

2003). At the clay site surface runoff was collected by the trenches, direct runoff from the field to the ditch was negligible (van der Salm et al., 2006, 2007).

Hydrological pathways, phosphate saturation of the soils and phosphate leaching losses

The three sites differed considerably with respect to the main hydrological pathways. The clay site had a very low hydraulic conductivity and large part of the discharge took place by runoff or interflow through the upper soil layers to the trenches (Table 1). The peat and sand sites were better drained and most of the water was conducted through the soil matrix.

Table 1. Distribution of discharge to surface water and phosphate sorption capacity, DPS and phosphorus sorption characteristics of the sites (0-40 cm) and phosphate balance.

		Sand	Clay	Peat
Discharge to surface water (%) ¹	Q _{Surface}	12	67	8
	Q _{shallow}	4	32	53
	Q _{deep}	84	-	29
Alox (mmol/kg)		65	59	144
Fe _{ox} (mmol/kg)		8	172	209
Langmuir adsorption constant (K)		0.37	0.11	0.18
Maximum amount of P bound to Al and Fe (β) (-)		0.21	0.09	0.19
DPS (%) ²		37	7	15
Phosphorus surplus (kg P/ha/yr)		20	21	14
Phosphorus losses to surface water (kg P/ha/yr)		2	3	5

¹⁾ Q_{surface} = Run-off, matrix flow (0-10 cm), trenches; Q_{shallow} = Matrix flow (10-70 cm) and drains, Q_{deep} = matrix flow (> 70 cm). Due to differences in methodologies Q_{sum} ≠ 100

²⁾ DPS = P_{ox} / (0.5 * (Al + Fe_{ox})) for 0-40 cm depth

Phosphate binding capacity increased from sand to clay, resulting in a high degree of phosphate saturation in the sandy soil and a low phosphate saturation in the clay soil. Despite the higher DPS, losses of P from the sandy site and the clay site were comparable. Highest P losses were found at the peat site, which had an intermediate DPS. These differences can be explained by differences in sorption characteristics, differences in hydrological pathways and differences in the distribution of P within the soil profile. The sandy site was quite deeply drained and 84% of the water discharge is through deeper soil layers where the DPS was relatively low (Table 3). At the clay site 67% of the discharge was by means of drainage through trenches. This route leads to 75% of the P losses to surface water at this site and also a large part of the total P losses (60-80%) to surface water occurred as a response to incidental heavy rainfall following manure application in early spring (Van der Salm et al., 2006). The peat site had an intermediate position with a somewhat higher DPS and drainage

through deeper soil layers. This intermediate position contributes to the high P losses observed at the peat soil. Another reason for the high losses from the peat soil is the release of P from eutrophic layers in the peat, contributing to about 50% of the total discharge.

A rough estimate of the impact of a reduction of manure application on leaching was made using the Langmuir isotherms and the sorption characteristics of the sites (van Beek et al., *subm*). Changes in P surpluses had the largest impact on P leaching on the sandy soil, followed by the peat soil and finally the clay soil. This sequence is strongly influenced by the fraction of reversibly bound P, which decreased from sand to clay.

Conclusions

P losses were strongly determined by the hydrological pathways in combination with the DPS and adsorption characteristics. Relatively high losses were found at the clay site, despite the low DPS, due to the important role of trenches and the resulting shallow drainage of the site. The highest losses were found at the peat site due to a combination of DPS, an intermediate sorption constant, shallow drainage and presence of eutrophic peat layers. Reducing P inputs as a measure to reduce N and P leaching to surface water is likely to be most effective on sandy soils, but for clay and peat soil hydrological conditions and release of P from eutrophic peat layers hamper the effectiveness of such measures. For these soils additional measures should be considered.

References

- Van Beek, C.L., Eertwegh, van den G.A.P.H., Schaik, van F.H. & Toorn, van den A., 2003. Surface runoff from intensively managed grassland on peat soils; a diffuse source of nitrogen and phosphorus in surface waters. Diffuse input of chemicals into soil and groundwater, Workshop 26-28 Februari. Dresden pp. 9-17.
- Van Beek, C.L., Van der Salm, C., Plette, A.C.C. & van der Weerd, H., (*subm.*). Nutrient loss pathways from grazed grasslands; experimental results for three soil types. *Subm. to Nutr. AgroEcosys.*
- Van der Salm, C., Dolfing, D., Van Groenigen, J.W., Heinen, M., Koopmans, G., Oenema, J., Pleijter, M. & Van den Toorn, A., 2006. Diffuse belasting van het oppervlaktewater met nutriënten vanuit grasland op een zware kleigrond. Alterra report 1266.
- Van der Salm, C., Dolfing, J., Heinen, M. & Velthof, G.L., 2007. Estimation of nitrogen losses via denitrification from a heavy clay soil under grass. *Agriculture, Ecosystems and Environment* 119, 311-319.
- Torenbeek, R. & Voskamp, T., 2003. Nieuwe inzichten over uit-en afspoeling van meststoffen. *H2O* 18, 19-21.

Factors influencing diffuse loss of dissolved inorganic phosphorus to streams

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Introduction

The diffuse P load to streams partly comes from dissolved inorganic P (DIP). In sandy Danish streams catchments DIP is the dominant form typically contributing with 50% - 75% of the total diffuse P load and with concentrations typically at 40 - 80 µg P/l. In clayey catchments the contribution is lower, typically 30% - 50% of the total P load and with concentrations ranging between 20 - 60 µg P/l (Wiggers & Nehmdahl, 2006).

The correlation between the stream concentrations and catchment characteristics can be used in estimating P load from unmeasured catchments. The correlations could also indicate casual relations which could be used in relation to action plans for lowering the P load.

Experimental design

The average DIP concentration in 45 Danish streams has been estimated as a discharge weighted concentration calculated for a period of four years. The streams investigated are without pollution from sewage or industry. In order to analyse the impact of catchment characteristics such as degree of agricultural land use, livestock density, phosphorus surplus at field level, soil texture, potential degree of tile drainage, phosphorus adsorption capacity in the soil, peat soil and distance between fields and streams, these factors have been estimated by GIS analyses for the catchments (ConTerra, 2006).

The correlations between DIP and the different catchment characteristics have been analysed in simple and multiple regression analyses (Wiggers & Nehmdahl, 2006)

Results

The concentration of DIP is most strongly correlated to the agricultural surplus of phosphorus ($r^2 = 0.43$, $p = 0.05$). Including distance between the fields and the streams in the multiple regression increase the correlation coefficient to $r^2 = 0.50$ (Table 1).

The concentration of DIP is also significantly correlated to soil texture with the maximum concentration in streams on sandy soil, but because of intercorrelation

between surplus and texture, only phosphorus surplus is included in the multiple regression analysis. The correlation between surplus and texture is caused by a higher phosphorus surplus in cattle farming dominating on sandy soil and generally to a higher livestock density on sandy soil.

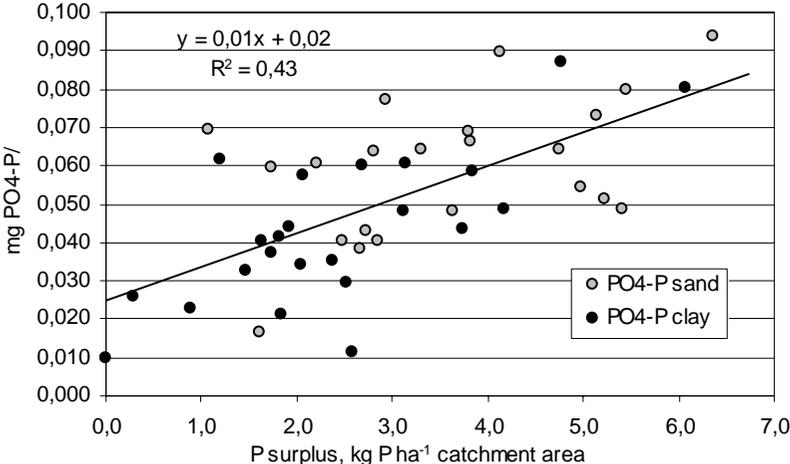


Figure 1. Concentration of DIP in the streams (discharge weighted average concentration) in relation to P surplus at catchment level.

In clayey catchments the transport pathway between field and stream is expected to be more direct due to macropores and drainage. When data from the 28 clayey catchments are analysed, P surplus is still the most important factor followed by distance between fields and streams. The correlation coefficient for the multiple regression including these two factors is $r^2 = 0.63$. For these catchments P adsorption capacity is also included as a significant factor in the multiple regression analysis, but only resulting in a small increase in the correlation coefficient to $r^2 = 0.65$ (Table 1).

Conclusion

Data from the 45 Danish streams indicate that the concentration of DIP is strongest correlated to the P surplus in the catchments although distance between the fields and the streams appears also to be a significant but less important factor. The correlation between DIP and the agricultural surplus could be partly indirectly reflect that catchments with a high P surplus probably also have a high P status in the soil. The P status of the soil in the catchments is unknown in this investigation, but probably catchments with a high P surplus at present have had an accumulation of P during decades resulting in a high P status. The correlation between P surplus and P status has been documented in Nielsen et al. (2006) and Grant et al. (2005).

Although P status in soil is supposed to be of importance in relation to the concentration of DIP in streams, the stream concentration can probably also be more directly related to P surplus, especially in catchments with good hydraulic contact between the field and the stream.

The results emphasize the importance of including P surplus as a factor in coming action plans to reduce the diffuse P load.

Table 1. Results from the multiple stepwise regression analysis.

All catchments				
n = 45				
Multiple stepwise regressions accumulated r-values for significant parameters				
Cut off value: 0.017				
	r	r ²	beta	signif. beta
P surplus, kg P ha ⁻¹ catchment area	0.656	0.43	0.62	0.000001
Distance between fields and stream	0.70	0.50	0.29	0.024

Clayey catchments				
n = 28				
Multiple stepwise regressions accumulated r-values for significant parameters				
Cut off value: 0.042				
	r	r ²	beta	signif. beta
P surplus, kg P ha ⁻¹ catchment area	0.75	0.57	0.88	0.000004
Distance between fields and stream	0.80	0.63	0.27	0.034
P adsorption capacity	0.81	0.65	-0.67	0.047

References

- ConTerra, 2006. CTtools – en landbrugs- og rodzonedatabase. <http://www.cttools.dk>.
- Grant, R., Blicher-Mathiesen, G., Jensen, P.G. & Rasmussen, P., 2005. Landovervågningsoplande 2004. NOVANA. Danmarks Miljøundersøgelser, 140 s. Faglig rapport fra DMU nr. 552. <http://faglige-rapporter.dmu.dk>
- Nielsen, K. et al., 2006. Begrænsning af fosfortab fra husdyrbrug. Metoder til fremtidige miljøgodkendelser. 42 s. Faglig rapport fra DMU nr. 566. <http://faglige-rapporter.dmu.dk>.
- Wiggers, L. & Nehmdahl, H., 2006. Fosfat i vandløb – betydning af oplandsfaktorer. Vand & Jord 13(3), 114-119.

The existence of bypass flow conduits in Northern Ireland soils

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Introduction

Infiltration of precipitation and the subsequent downward percolation is a complicated process dependent on soil texture, structure, wetness and heterogeneity. The types of flow identified are piston flow, bypass flow and finger flow. In piston flow water from recent precipitation events forces the older soil water to flow downward infiltrating through the pores in a wetting front, residing in the soil matrix for a time and ending up in groundwater. In fingered flow, wetting front instability results in greater flow through fingers rather than uniformly through a porous medium. In bypass flow, there is preferential fast flow through macropores which occur naturally as worm holes, decayed root channels, cracks and as part of soil structure. (eg Van Ommen et al.1989; Komor and Emerson, 1994).

An increased understanding of the mechanisms involved in the transport of nutrients from land to water is essential if the problem of accelerated eutrophication resulting from diffuse agricultural pollution is to be addressed. Different flow types result in different isotopic profiles particularly when rainfall events are isotopically distinct. In bypass flow, recent precipitation can pass older water that remains in the soil matrix. Piston flow after an isotopically distinct rainfall event results in an abrupt isotopic front in the soil. Therefore by characterising the isotopic composition of precipitation and soil drainage water it is possible to identify the mechanisms of flow dominating in a soil and mixing of water from different precipitation events.

We propose to use the stable oxygen isotopic composition of precipitation and soil water as determined by the H₂O-CO₂ equilibration method to investigate the mechanisms of soil water movement in Northern Ireland soils.

Methods

A grassland plot (0.2 ha) was hydrologically isolated and artificially drained to a v-notch weir with flow proportional monitoring of drainage water (Watson et al., 2000). The plot received 20 kg P ha⁻¹ yr⁻¹ applied as triple superphosphate (46% P₂O₅) in six equal applications per year from March 2000 to February 2005. Thereafter, no P fertilizer was applied to the plot. The plot received input of N (250 kg N ha⁻¹ yr⁻¹) and potassium and sulphur according to soil analysis and standard recommendations for grazing management. Beef steers grazed the plot from April to October of each year to maintain a constant sward height of 7 cm. Samples of soil drainage water and

precipitation were taken every day from 11th April 2005 to 18th April 2005. During this time precipitation occurred from April 11th to 16th amounting to 10 mm of rain. On the 17th April samples of soil drainage water were taken every hour until the 18th April. For rainfall and soil water samples the oxygen isotopic composition was determined using a CO₂ equilibration technique which was optimised in preliminary experimentation. After equilibration each 12 mL sample was analysed for ¹⁸O isotopic composition by isotope-ratio mass spectrometry.

Results

The isotopic composition of soil drainage water is presented in Fig. 1. The ¹⁸O isotopic enrichment of soil drainage water prior to the measurement period was 2059 ppm. From the 11th April to 16th April the isotopic composition of rainfall averaged 2064 ppm. The 95% confidence limit for measurement of ¹⁸O is 1.1 ppm. As a result there was, a significant increase in the ¹⁸O enrichment of soil drainage water which reached a peak of 2062.6 at 1500 h on 17th April and then decreased to a value < 2060 ppm which was not significantly enriched with ¹⁸O.

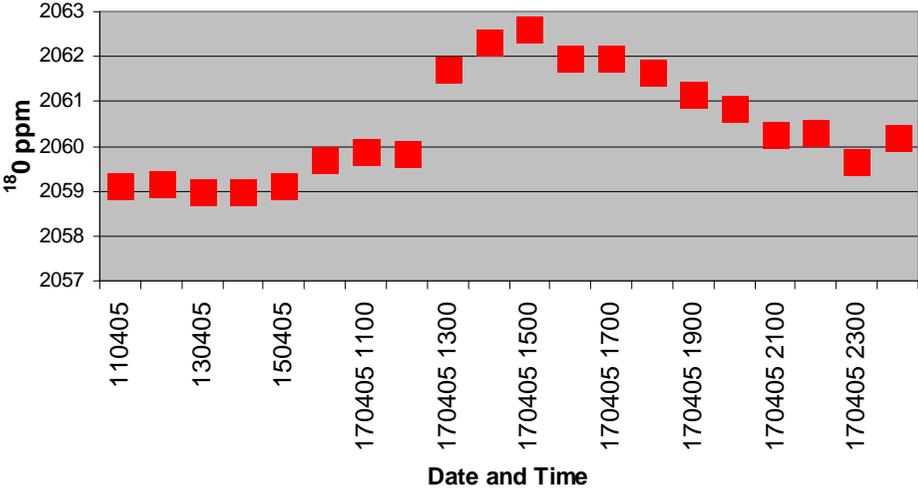


Figure 1. The ¹⁸O isotopic composition of soil drainage water.

Comparison of the ¹⁸O isotopic composition of soil water with the isotopic composition of precipitation prior to sampling can indicate the type of water flow through soil. For example, the maximum value of ¹⁸O in the soil drainage water was approximately 2063 ppm which was similar to the average value of precipitation prior to sampling (2064 ppm). This would suggest that water movement through this soil was due to bypass flow. If the flow had been piston flow there would have been mixing of soil water at an isotopic enrichment of 2059 ppm with precipitation at 2064 ppm and therefore a lower value of ¹⁸O in the drainage water.

Further evidence for bypass flow being the dominant flow mechanism can be determined from the measurement of P and NH_4^+ concentrations in soil drainage water prior to and after the measurement period. The average value of soluble reactive P for a week prior to measurement was 0.016 ppm. The value peaked abruptly on 17th April to 0.234 ppm and returned to a value similar to that before the measurement period, by 18th April. The $\text{NH}_4\text{-N}$ concentration followed the same pattern as P i. e. low levels prior to the measurement period, an abrupt peak on 17th and a fast return to low levels.

Conclusions

This study suggests that bypass flow could be the dominant flow mechanism in temperate grassland soils in Northern Ireland. No P fertiliser had been applied to this plot since August 2004 but still there were high concentrations of P in the soil drainage water whenever there was a significant rainfall event. This suggests that P loss to drainage water is via bypass flow and is of particular environmental concern as this plot has not received P fertilizer since August 2004.

References

- Komor, S.C. & Emerson, D.G., 1994. Movement of water, solutes, and stable isotopes in the unsaturated zones of two sand plains in the upper Midwest. *Water Resour. Res.* 30, 253-267.
- Van Ommen, H.C., Van Genuchten, M.Th., Van der Molen, W.H., Dijkema, R. & Hulshof, J., 1989. Experimental and theoretical analysis of solute transport from a diffuse source of pollution. *J. Hydrol.* 105, 225-251.
- Watson, C.J., Jordan, C., Lennox, S.D., Smith, R.V. & Steen, R.W., 2000. Inorganic nitrogen in drainage water from grazed grassland in Northern Ireland. *J. Environ. Qual.* 29, 225-223.

Spatial variation of available soil phosphorus in microplot rainfall simulation studies

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Introduction

A popular technique for studying phosphorus runoff is the use of small, portable rainfall simulators, but these are not without their problems. For example, in experiments aiming to relate runoff P to soil P levels, plots are disturbed by removal of soil cores for analysis. Thus, sequential studies examining nutrient losses over time are compromised, since the physical structure of the plots is not constant. Also, it has been shown that available soil P can vary significantly at sub-meter scales (e.g. Raun *et al.*, 1998; Wright, 1998). The advantage of their portability has the concomitant disadvantage that these machines must necessarily only cover a very small area, with runoff plots usually 2m² or less in size. The placement of the simulators within a field or other given area, and the number of treatment replications therefore becomes very significant in light of this very small scale variability in soil P. In this study, the sub-meter spatial distribution of surface soil P within and beyond plot boundaries was measured, to determine whether sampling outside runoff plots is a viable strategy for ongoing runoff plot experiments. The plot scale variability is compared with the spatial variation of soil P at paddock/hill slope scale at the study sites.

Methodology

Following a simulated rainfall runoff event, individual soil cores from 0-2cm depth were taken at intervals of 40cm along transects 25cm apart, from paired 2 x 0.75m microplots, beginning and ending 20cm outside the plot boundaries, with a further transect 25cm from the outer edge of each plot. This was done in triplicate at three pastoral sites, under low (unimproved pasture), moderate (beef cattle) and high (dairy) farm management intensity, and the soil samples analysed for bicarbonate-extractable (Colwell) P. After drying and sieving, 3.0g was taken from each individual core sample and mixed together thoroughly, to create composite samples. Soil cores (5, taken over an area ca. 1.4m² and composited) from 0-2 cm depth were taken from the paddock in which the runoff events were conducted, except at the Moderate site, which was from the adjacent field, on a 30x30m to 35x35m grid (depending on the size of the paddock), and similarly measured for Colwell P. Soil P data was spatially analysed using the Vesper software package (Minasny *et al.*, 2005). Block kriging was conducted using global variograms with lag = 20, with an exponential model weighted by the number of pairs.

Results

Viewed as complete sets, there was no significant difference ($p=0.05$) between the Colwell P concentration of the soil inside the runoff plots compared to that just outside them at any of the sites (Table 1), although there were three individual runoff plots where a significant difference was found (two at the Low site, one at the Moderate; data not shown). With one exception, measuring soil P using composite samples gave slightly lower values than the arithmetic mean of the individual cores, and had higher errors, although the latter may largely be influenced by the sample size ($n=60$ or 66 , versus 6). Differences in Colwell P increased between individual cores from a given runoff plot with increasing land use intensity, with the Low site ranging from $8-40$, the Moderate site from $41-191$, and the High site from $160-717$. However, the concurrent increase in variance in the data sets meant that these differences were not statistically significant (within plots, when comparing inside v. outside). These ranges in soil P at the runoff plot scale were smaller than those found for the paddocks as a whole, with the exception of the High site (Colwell P = $18-65$, $\bar{x} = 34$; $29-274$, $\bar{x} = 101$; and $138-497$, $\bar{x} = 339$, for the Low, Moderate and High sites, respectively).

Kriging analysis showed that there was some order in the soil P distribution at the paddock scale, induced by changes in soil type, landscape and other factors, at each of the sites (e.g. Figure 1), but the variograms for the runoff plots were flat. The exception for this was the High site, where some structure was found at the runoff plot scale as well.

Table 1. Colwell P (mg kg^{-1} , 0-2cm depth) (mean, s.e.) from individual cores sampled from inside or outside runoff plot boundaries, from sites under low, moderate and high land use intensity, compared to data from composite samples. Average of 3 replicates per site.

Soil core location	Site (land use intensity)					
	Low		Moderate		High	
	Individual cores	Composite soil	Individual cores	Composite soil	Individual cores	Composite soil
Inside	23.5, 0.8	21.7, 2.9	107.5, 5.6	104.2, 18.5	326, 11.1	309.8, 27.4
Outside	23.4, 0.7	24.5, 1.6	114, 5.0	112.2, 15.9	347.2, 13.1	317.5, 19.5

Conclusions

For these sites, it appears that soil Colwell P from just outside runoff plot boundaries may be used as a predictor of the Soil P inside the runoff plots, and hence one could do multiple runoff events on the same plots without disturbing them by soil sampling. However, as land use intensity increases, so does the risk of hitting a “hot spot” of soil P that is not representative of the runoff plot P. The runoff plot mean soil P was

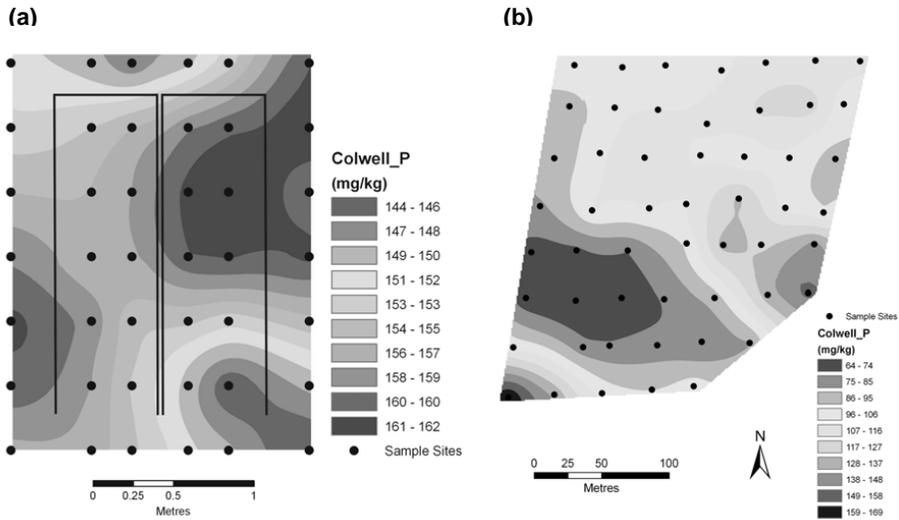


Figure 1. Soil coring layout (black circles) for runoff plot (a), and paddock (b) for the moderate intensity land use site, and kriged Colwell P values (0-2cm depth).

similar to that of the paddocks as a whole, but the grid sampling demonstrated that even in very low-input system, the soil P can vary substantially at paddock scale. Obvious sites where P may accumulate, such as around gates or camping sites, can be easily avoided, but caution should still be applied when considering how representative the data from microplot runoff studies is of the field in which they are sited, especially in high-input systems.

Acknowledgements

Spatial and some statistical data analyses were carried out by Dr Brett Whelan, Australian Centre for Precision Agriculture, University of Sydney.

References

- Minasny, B., McBratney, A.B. & Whelan, B.M., 2005. VESPER version 1.62. Australian Centre for Precision Agriculture, McMillan Building A05, The University of Sydney, NSW 2006. (<http://www.usyd.edu.au/su/agric/acpa>).
- Raun, W.R., Solie, J.B., Johnson, G.V., Stone, M.L., Whitney, R.W., Lees, H.L., Sembiring, H. & Phillips, S.B., 1998. Microvariability in soil test, plant nutrient, and yield parameters in bermudagrass. *Soil Sci. Soc. Am. J.* 62, 683–690.
- Wright, N.A., 1998. Soil fertility variograms from "True Point Sampling"™ on 20.0, 0.9, and 0.1 meter grids in two fields. *Commun. Soil Sci. Plant Anal.* 29, 1649-1666.

COST Action 869 - Mitigation options for nutrient reduction in surface water and groundwaters

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Overview

COST 869 (European **CO**operation in the field of **Sc**ientific and **T**echnical Research) will focus on the steps that need to be taken within the EU Water Framework Directive in order to effectively reduce the nutrient losses from point and diffuse sources to surface waters and groundwater. The Action will be undertaken in the context of balancing measures to reduce phosphorus (P) losses with those necessary to reduce other nutrient losses such as nitrogen (N). Such measures are often conflictory, and need to be considered as part of an integrated program of measures. Since P is the element that is limiting in most European inland waters, the focus of COST 869 is on P losses. However, positive or negative influences of mitigation options on the loss of fine sediment, nitrogen and pesticides to either surface water or other environmental compartments such as groundwater will be discussed during the Action. The outcomes of the discussions within the COST Action will be reported to the new board of the WFD dealing with the interaction between agriculture and water quality.

History

Early 2005, a pre-proposal for a new COST action was launched. The full proposal was submitted mid June 2005, under the program Agriculture, Biotechnology and Food Sciences (AgriBioFood). The proposal was approved by the Technical Committee in October 2005, and by the Committee of Senior Officials in March 2006. It started officially in November 2006, and will end in 2011.

Signatures

At this moment (June 2007), the following 27 countries participate in the Action: Austria, Belgium, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Israel, Italy, Latvia, Lithuania, Luxembourg, Netherlands, Norway, Poland, Portugal, Romania, Slovakia, Slovenia, Spain, Sweden, Switzerland, and United Kingdom

Management Committee

The COST Action has 4 working groups (WG, see below) and a Management Committee (MC), that consists of one or two representatives from each participating country. The MC meets once a year, and has the tasks: (i) to assess and report on the progress made by the WGs towards their respective objectives within the overall framework of the COST Action; (ii) to coordinate the Action meetings, Short-Term Scientific Missions [STSM, see below] and publications; (iii) to promote research cooperation and data exchange between the WGs; (iv) to prepare the Annual Reports of the program for the COST Office; and (v) to help organize contacts and common workshops with other appropriate ongoing COST Actions and relevant technology or scientific platforms, and to address problems of common interest (e.g. erosion abatement).

Working groups

Within COST 869, 4 working groups are active. Their goals are, respectively, to:

- develop methodologies to localize critical source areas and transport routes in catchments,
- study the influence of nutrients on ecological processes in surface waters,
- evaluate different types of mitigation options,
- evaluate projects in example areas.

The following table shows the names of coordinators of the working groups.

Table 1. Working groups within COST 869 with coordinators.

		<i>chair</i>	<i>vice-chair</i>
Action		Wim Chardon (NL)	Louise Heathwaite (UK)
WG1	Critical source areas	Louise Heathwaite (UK)	Iggy Litaor (IL)
WG2	Ecological processes	Seppo Rekolainen (FI)	Nicolaos Skoulikidis (GR)
WG3	Mitigation options	Oscar Schoumans (NL)	Antonio Lo Porto (IT)
WG4	Evaluation of projects	Brian Kronvang (DK)	Peter Strauss (AT)

Short Term Scientific Missions

Within the COST Action there is a budget available for short visits of scientists to an institute in another country that participates in the Action. See the website for more information and for an application form that has to be filled in.

Past meetings

The first Management Committee (MC) meeting was on 7 and 8 November 2006 in Brussels; a small organizing meeting of coordinators was in December 2006 in Amsterdam; an organizing meeting of WG2 in May 2007 in Oslo, and a full workshop of WG1 in May 2007 in Hamar.

Future meetings

The first Management Committee (MC) meeting is planned on 7 and 8 November 2006 in Brussels, during which a chair, vice chair, and working group coordinators will be elected. During this meeting also the program of the Action for the next years will be discussed, and the results of an inventory among participating countries.

Website

More information can be found at the website: www.COST869.alterra.nl, e.g. the full text of the proposal, the Memorandum of Understanding, a publication list and a list of experts that showed interest. Also, it contains reports of meetings that have taken place, in some cases including presentation that were given, and announcements of future meetings.

Interested?

If you want to join the Action and want to be informed about future meetings, please mail: (i) your name; (ii) name, place, and country of institution, (iii) telephone number, (iv) e-mail address, and (v) the Working Group(s) in which you are interested, to: wim.chardon@wur.nl.

Vegetation-induced temporal changes in phosphorus cycles in differently managed vegetation of buffer zones

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Introduction

Buffer zones (BZ) have been used successfully in removing suspended solids and particle-bound nutrients, especially phosphorus (P) from agricultural runoff also in the Nordic countries (Syversen and Borch 2005, Uusi-Kämpä 2005). However, in terms of readily available dissolved P, there was a highly variable and even an opposite trend, probably caused by P leaching from the soil surface and plant material (Uusi-Kämpä 2005). Although the plant/root system has an essential role in minimizing soil erosion, plant uptake tends to accumulate nutrients from the rooting depth to the soil surface (Jobbágy and Jackson 2001).

The objective of this study was to evaluate the role of vegetation in P dynamics in differently managed BZs. It was hypothesized that the removal of plant material reduces considerably P accumulation onto the soil surface and thus, delays the need for recurrence of BZ.

Materials and methods

The study site was located in Jokioinen in south-western Finland. The soil was classified as Vertic Cambisol (clay content over 50%, $\text{pH}_{\text{CaCl}_2}$ at 0–5 cm 4.9– 5.7). The study was performed in six different BZs:

- the old BZ (15 yr.) with natural vegetation, scrub and hardwood trees (e.g. bent grass; *Agrostis capillaris*, goat willow; *Salix caprea*),
- the old BZ (15 yr.) with grass species which are annually harvested and removed (mainly timothy; *Phleum pratense* and meadow fescue; *Festuca pratensis*),
- the young BZ (4 yr.) with grass species which are grazed by cattle with a mean stocking rate of 3.5 cows ha^{-1} ,
- the old BZ with grass species which are grazed by cattle,
- the young BZ with grass species which are annually harvested and removed,
- the old BZ with natural vegetation, mainly grass species.

The first three plots were located in an area of a long-term BZ experimental field with widths up to 10 m at an average slope of 16% (Uusi-Kämpä and Ylärinta 1992).

Leaving out woody plants, aboveground biomass was sampled in 2005 on May 02, June 28, August 09, October 10 and November 23, and in 2006 on April 26 by clipping the vegetation to a height of 2 cm in five areas of 0.25 m² per BZ. Plant materials were oven-dried at 60°C, ground and analyzed for total P using the procedure of high-temperature thermal oxidation at 500°C (Jones 2001). Plant P in extracts was determined colorimetrically using the procedure by Murphy and Riley (1962). Soil samples were collected at two different depths (0–2.5, 2.5–5 cm) for the determination of the agronomic soil test P using a 0.5 M ammonium acetate 0.5 M acetic acid solution (pH 4.65) (Vuorinen and Mäkitie 1955).

Results and discussion

The dry matter yields were largest in August varying from 2120 kg ha⁻¹ for the old grazed BZ to 6270 kg ha⁻¹ for the young harvested grass BZ. Grazing substantially reduced the total coverage of aboveground vegetation. The P concentration in plant material was 0.8–2.7 mg g⁻¹. In August, total P in biomass varied from the minimum of 3.1±1.6 kg ha⁻¹ for the old grazed BZ to the maximum of 8.7±0.8 kg ha⁻¹ for the old (woody) natural BZ (Figure 1).

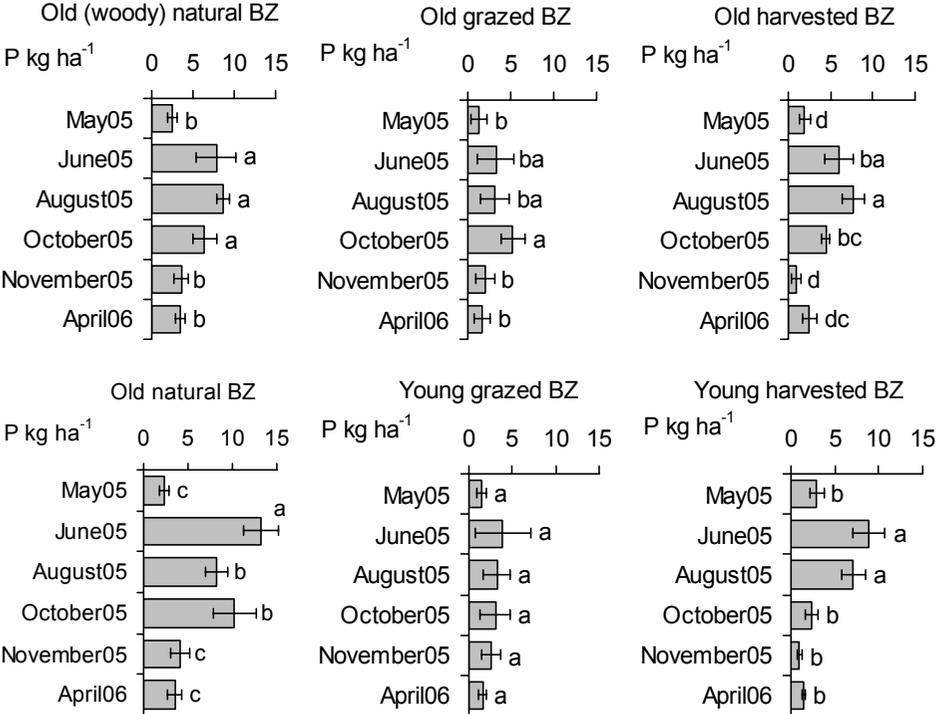


Figure 1. Total contents of P (kg ha⁻¹) in the aboveground vegetation at different times (mean ± STDEV, TUKEY; *p* < 0.05).

After first frosts between October and November biomass P contents decreased noticeably between 0.5 and 6.1 kg ha⁻¹ across all variants. This was in accordance with Bechmann et al. (2005) who showed that freezing and thawing cycles of plant material converted all the P in the catch crop to a water-soluble form and thus probably partly accounts for the commonly observed increase in dissolved P in surface runoff water. Part of the decrease in P contents of the aboveground vegetation may be attributed to the transport of P to the belowground parts of plants.

The ammonium acetate solution extracted 5.6–22 mg P l⁻¹ from the upper 2.5 cm of soil and 4.0–10 mg P l⁻¹ from the 2.5–5 cm soil layer. The concentrations were substantially higher in the old BZs with natural vegetation than in the other BZs. This indicates a strong tendency towards P accumulation to the soil surface over years as a result of the non-harvesting practice. According to Turtola and Yli-Halla (1999) the concentration of dissolved P in surface runoff water increased with the increasing values of P status at the soil surface. Thus, the vegetation-induced accumulation of P in the surface soil increases the potential of BZ to act as a source of dissolved P in runoff and drainage water. However, BZs will probably result in a net decrease of bioavailable P loading because they substantially reduce erosion and the load of particle-bound P.

Acknowledgements

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References

- Bechmann, M., Kleinman, P.J.A., Sharpley, A.N. & Saporito, L.S., 2005. Freeze-thaw effects on phosphorus loss in runoff from manured and catch-cropped soils. *J. Environ. Qual.* 34, 2301–2309.
- Jobbágy, E.G. & Jackson, R.B., 2001. The distribution of soil nutrients with depth: Global patterns and the imprint of plants. *Biogeochemistry* 53, 51–77.
- Jones, J.B., Jr., 2001. Laboratory guide for conducting soil test and plant analysis. CRC Press, USA. 363 pp.
- Murphy, J. & Riley, J.P., 1962. A modified single solution method for the determination of phosphate in natural waters. *Anal. Chim. Acta* 27, 31–36.
- Syversen, N. & Borch, H., 2005. Retention of soil particle fractions and phosphorus in cold-climate buffer zones. *Ecol. Eng.* 25, 382–394.
- Turtola, E. & Yli-Halla, M., 1999. Fate of phosphorus applied in slurry and mineral fertilizer: accumulation in soil and release into surface runoff water. *Nutrient Cycling in Agroecosyst.* 55, 165–174.
- Uusi-Kämpö, J., 2005. Phosphorus purification in buffer zones in cold climates. *Ecol. Eng.* 24, 491–502.
- Uusi-Kämpö, J. & Ylänta, T., 1992. Reduction of sediment, phosphorus and nitrogen transport on vegetated buffer strips. *Agric. Sci. Finl.* 1, 569–575.

Vuorinen, J. & Mäkitie, O., 1955. The method of soil testing in use in Finland. *Agrogeol. Publ.* 63, 1-44.

Effect of source and hydrological measures on reducing the load of N and P to surface water

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Introduction

The aim of the European Water Framework Directive (EWFd) is to ensure that the quality of the surface water and groundwater in Europe reaches a good ecological status (Water Framework Directive, 2000). To meet the 2015 deadline, a programme of measures has to be taken by river basin authorities by 2009. In an explorative study (van der Bolt, 2003) it became clear that a systematic approach for the choice of goals and measures is of great importance. At this moment it is still unclear which measures have the greatest effect on reducing the load of nitrate, phosphate, heavy metals and crop protection chemicals to surface and groundwater.

This project has the goal of providing an overview of measures in the rural area in terms of effects and costs. Available knowledge, measurements and model results are systematically collected, described, stored and interpreted in order to determine the effects of individual and combined measures to reduce the loads mentioned. To realise this goal a Knowledge Information System is set up. At first the effects of source and hydrological measures on P and N losses to the surface water are built in.

Selection criteria for a knowledge information system

(Water) authorities (and for the final realisation also land and water users like farmers) want to know which measures influence the water quality positively. They want to know if the measure influences groundwater levels and/or surface water levels or discharges, but they also want to know if the measure reduces the leaching of nitrate, phosphate, heavy metals or pesticides. It has to be realised that the effect of the measure may depend on environmental circumstances like soil type, land use, groundwater table, the presence of pipe drainage or irrigation.

It became clear in this study that measures are not well defined. Some use a strict definition, for others a measure is a broader range of several actions to decrease the emission to surface or groundwater.

In the knowledge system designed the following specific approach has been chosen:

- Measures relevant in relatively flat river deltas such as The Netherlands are selected;

- Effects are primarily focused on groundwater levels and the diffuse emission of nitrate and phosphate to ground- and surface water. The emission of heavy metals and/or crop protection chemicals can be added at a later stage;
- The specified measure can be realised at field level, which means that the description of the measure has to be made very concrete and, consequently, effects and costs can be determined more specifically and more efficiently;
- Each measure will be described in a fact sheet which encompasses a definition, a description how the measure works, the expected result of a measure, its additional effects on, for example, agriculture, environment or landscape, a qualification if the measure is still experimental or applicable for practice and if there are sources for references via expert judgement, modelling or applied research;
- Effects of measures will be given in terms of the relative reduction of the load of nitrate and phosphate compared to a certain starting point;
- For each measure the reduction in load will be estimated for possible transport routes like direct spill, surface runoff, interflow, pipe drainage, (shallow) ditches, up or downward seepage (Figure 1);
- A geographical tool will be added to select the most promising measures at specific sites;
- The final result will be maps of The Netherlands and regions or river basins showing at different scales where a specific measure can be taken most efficiently (reduction emission and costs). It may also be a list of measures in order of effectiveness for a specific spatial unit (field, river basin) with specific conditions (land use, soil type, groundwater table).

Results

A first version of the Knowledge Information System (KIS) is available for some specific measures for which accurate information was easily available. This input for the KIS is being structured and analyzed for implementation. Data collection about the costs of measures still takes place, while the effects of measures as described in literature have already been incorporated in the fact sheet per measure. Algorithms for specific effects of a measure for certain transport routes at several scales are being composed or are already incorporated.

Future

KIS data about specific measures can be used for different kinds of goals. For example to generate a list of measures that reduce the emission of phosphate from agricultural fields to surface water, or to make a specific map for a water authority that wishes to reduce the emission to surface water including a list of measures and costs. KIS can be helpful to set up measures for the EC or to evaluate river basin management plans or even to optimise specific plans for spatial development. Last but not least, KIS informs about the gaps in knowledge about specific measures and,

consequently, can be helpful to solve knowledge gaps. During the coming years the KIS will be continuously updated by new knowledge coming from applied research or from model studies to increase the efficacy of measures to be taken by water authorities and farmers to realize the good ecological status required by the EU Water Framework Directive.

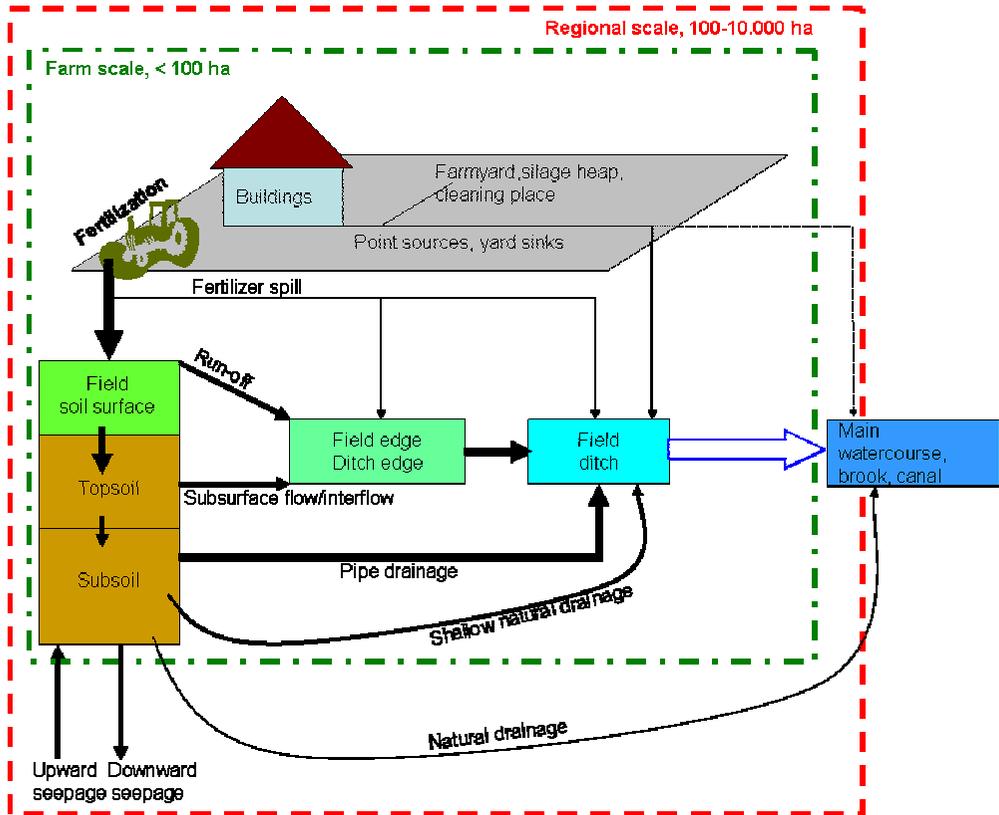


Figure 1. Emission routes of nitrate and phosphate.

References

- Van der Bolt, F.J.E., van den Bosch, H., Brock, Th.C.M., Hellegers, P.J.G.J., Kwakernaak, C., Leenders, T.P., Schoumans, O.F. & Verdonschot, P.F.M., 2003. Aquarein, gevolgen van de Europese Kaderrichtlijn Water voor landbouw, natuur, recreatie en visserij. Wageningen, Alterra rapport 835, 152 p. (in Dutch).
- Van der Bolt, F.J.E. & van Os, E.A., 2006. KIS maatregelen in het landelijk gebied. In: Leenders, T.P. en C. Kwakernaak, 2006. 20 Puzzelstukjes voor de KRW. Een bloemlezing uit het onderzoek van Wageningen UR voor de Europese Kaderrichtlijn Water. Alterra rapport 1403, 68p. (in Dutch).
- Water Framework Directive, 2000. EU, Brussels, 72p.

Mitigation options for reducing diffuse P losses: which to choose where?

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Introduction

Today the greatest contributor of phosphorus to the aquatic environment is non-point source pollution from agricultural areas. Focus therefore is both on input reduction in the agricultural production sector and on preventing phosphorus losses from phosphorus-enriched agricultural land.

The twofold perspective calls for different measures; both nationally and locally targeted. General economic measures such as taxes and tradable quotas aim to reduce the input of phosphorus in the agricultural production sector in general and locally targeted measures aim to prevent phosphorus losses to the aquatic environment from critical source areas. Two essential questions arise: which types of measures will be most efficient in the regulation of phosphorus in the short and long run? And how should these measures be implemented in order to obtain a cost-efficient solution? This paper is concerned with the question of implementation and how to combine the short and long-term goals.

The paper is the first outline of an analysis using different data (e.g. agricultural production data, hydrological data, the P-index and different spatial GIS-based data) linked to economic analyses of the farmers' behavior adjusted to different measures and implementation instruments. A lake in the Odense River Basin is used as an example.

Short and long-term perspectives

The long-term perspective is to reduce the input of phosphorus in the agricultural production sector. The aim is to reduce the use of phosphorus such as to obtain phosphorus balance in the soil. Input reduction can be obtained through different economic incentive measures such as taxes or quotas. Taxes on phosphorus in fodder or on fertilizer according to the content of phosphorus are two possibilities. Also a deposit-refund system, such as suggested by Hansen (1999) for nitrogen, is a possibility.

The short-term perspective is to prevent immediate losses of phosphorus from agricultural areas to the aquatic environment. Where the long-run perspective includes a national goal, the short-term perspective has a local focus. Only where

there is a substantial quantity of potentially mobile phosphorus in soils and at the same time an effective transport route to the aquatic environment is the area critical with respect to phosphorus losses. The local regulation thereby addresses specific areas in Denmark. Using a phosphorus index (PI), it is possible to identify these areas with some error margin (Andersen & Kronvang 2006; Sharpley et al. 2001). Another local perspective, which from an economic point of view has to be considered, is the importance of upstream and downstream catchments. That is, not all critical areas are of equal importance. Initiatives carried out upstream may have an effect on all downstream recipients, whereas initiatives carried out downstream only have effect on the same recipient. This means that cost and benefits from the same measure vary according to where it is implemented. Phosphorus reductions therefore might induce less economic effect in different recipients.

Suggestion for an implementation mechanism

Economic incentive measures require numerous agents in order to achieve full competition and thereby obtain the socially economic level of pollution (Hanley et al. 1997). Because of the local aspect of the phosphorus problem, this requirement may in most cases not be fulfilled. How many agents are required to ensure trade between farmers? Another dilemma is the local targeted goals set for each recipient. The overall objective from general economic incentive measures is to reach some reduction target at lowest cost. This is fulfilled by introducing e.g. subsidies for phosphorus reduction measures. However, we do not know which agents choose to implement some reduction measure and which do not. This knowledge is of greatest importance in the phosphorus regulation (Huhtala 2005). Otherwise the local goal fulfilment may not be obtained.

Choosing between different local implementation mechanisms to reduce phosphorus losses entails identifying the agents we wish to regulate (e.g. Birr & Mulla 2001). Using the phosphorous index it is anticipated that critical areas of phosphorus losses and hence individual farms can be identified. According to the Water Framework Directive, targets are set for each recipient in the EU member states. These targets are going to be evaluated at latest in 2012 providing guidance for loss reduction targets. How do we obtain these reductions without expropriation of fields and without command-and-control tools?

One possibility is to pool farmers into different groups according to the category of target recipient, e.g. a lake and an inlet in some target area. Pooling farmers into groups aiming to reduce the phosphorus discharge to the same recipient gives the possibility of setting up different implementation systems for the different groups. One possible way to implement cost-efficient measures such as wetland restoration, border zones and crop change for the pooled groups of farmers is to set up a contract system, where the contracts are to be sold at auctions. Here farmers are the

suppliers of environmental services while the regulator is the demander. The farmer is asked how many units of abatement he would offer at a given price (subsidy). The farmer would be free to choose the abatement measure or farm management practice at the lowest cost that results in a reduction in the risk of phosphorus losses. Contract systems have been suggested in the literature and several designs recommended, see Latacz-Lohmann & van der Hamsvoort (1998), Cason et al. 2003 and Levy & Vukina (2002).

In the present study, several auction/contract designs are to be evaluated with the aim of reducing phosphorus losses from non-point agricultural areas. One design is to present two different forms of contract. A basis contract where the farmer chooses which measure(s) to implement based on some compulsory reduction target and an extended voluntary contract. "The more environment obtained, the better a price offered". By use of e.g. simulation modelling and empirical Danish data a case study is carried out.

References

- Andersen, H.E. & Kronvang, B., 2006. Modifying and evaluating a P index for Denmark. *Water, Air and soil Pollution* 174, 341-353.
- Andersen, H.E., Heckrath, G. & Kronvang, B., 2006. Dansk P-index til bestemmelse af fosfortab. *Vand & Jord* 13(1), 13-17.
- Birr, A.S. & Mulla, D.J., 2001. Evaluation of the phosphorus index in watersheds at the regional scale. *Journal of Environmental Quality* 30, 2018-2025.
- Cason, T.N., Gangadharan, L. & Duke, C., 2003. A laboratory study of auctions for reducing non-point source pollution. *Journal of Environmental Economics and Management* 46, 446-471.
- Hanley, N., Shogren, J.F. & White, B., 1997. *Environmental Economics. In theory and practice*. Macmillan Press Ltd, London.
- Hansen, L.G., 1999. A Deposit-Refund System Applied to Non-Point Nitrogen Emissions from Agriculture. *Environmental Economics and Policy Studies* 1999. 2(44), 23-247.
- Huhtala, A., 2005. Incorporating economics into risk assessment. In: Heckrath, G., Bechmann, M., Ekholm, P., Djodjic, F., Ulén, B., Andersen, H.E. & Olsen, P. (Eds.), 2005. *Tools for assessing phosphorus loss from Nordic agriculture*. TemaNord 2005, 583. 139 pp.
- Latacz-Lohmann, U. & van der Hamsvoort, C.P.C.M., 1998. Auctions as a means of creating a market for public goods from agriculture. *Journal of Agricultural Economics* 49(3), 334-345.
- Levy, A. & Vukina, T., 2002. Optimal linear contracts with heterogeneous agents. *European Review of Agricultural Economics* 29(2), 205-217.
- Sharpley, A.N., McDowell, R.W., Weld, J.L. & Kleinman, P.J.A., 2001. Assessing site vulnerability to phosphorus loss in an agricultural watershed. *Journal of Environmental Quality* 30(6), 2026-2036.

Cost benefit analysis of nutrient reductions to Danish aquatic environments

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Background

Over the next couple of decades, the development of the aquatic environment is mainly influenced by The Danish Action Plan for the Aquatic Environment¹ and EU's Water Framework Directive (WFD²). The overall objectives of these programmes are decided at the national or the EU level. However, realization of the objectives typically leaves room for choices between different local and national implementation measures. The WFD allows modification and exemptions to the environmental objectives if compliance costs are disproportionate to benefits. The existence of multiple benefits, as well as multiple implementation measures (and costs), place great demands on decision-making. This in turn will increase the need to secure efficiency of the decision making process required to identify the relevant policy measures.

Objectives

The joint implementation of the Danish Action Plan for Aquatic Environment and WFD entails some very interesting policy issues and research questions, and the overall objectives of this research project are to provide answers to the following questions:

- Which combinations of general and local measures and instruments to reduce nutrient loads within different catchments are most cost-effective and which combinations of nitrogen and phosphorus abatement strategies are most cost-efficient? How do these strategies interact? How do they influence the regions, and the employment?
- Do benefit considerations change the efficiency of abatement strategies compared to choices based on only cost considerations? Is it possible to transfer benefit and cost estimates between different water basins, and to generalize results?

¹ Abatement strategies have been implemented through a number of national aquatic action plans for the aquatic environment, as well as by implementation of EU Directives on nitrate, drinking water, groundwater, bathing water and the water bodies in general. The Aquatic Action Plan III aims at a general reduction in nutrient loading by reduced fertiliser quotas, phosphorus fertiliser tax, requirements for utilisation of animal manure, etc.

² The Water Framework Directive (WFD) has been implemented in Danish legislation (Ministry of Environment 2003).

Project content and methods

The project comprises a part where scenarios are outlined, followed by work packages of benefit assessments, cost assessments, benefit transfer tests and a full cost benefit assessment.

Scenarios

The most relevant scenarios to obtain the environmental objectives of the WFD and the Danish Aquatic Action plan III are intensive implementation of wetland restoration, fallowing, border zones, forestation and general fertiliser reductions (quotas and tax). A comprehensive description of measures and possibilities can be found in the work presented by the so-called Godtfredsen Committee where measures to obtain the objectives in WFD are investigated and described (www.dmu.dk). The experiences from the WFD pilot case area Odense river basin are also comprehensive.

The assessment of benefits from the scenarios

Primary valuation surveys will be conducted in the river basin, and the study focuses on valuation of different characteristics in different parts of the water body. Examples of attributes are surface water quality in lakes, waterways, rivers and the fjord, as well as costs; and/or indicators for flora and fauna connected to the aquatic environment. The valuation is based on the choice experiment method, as this method has proved very suitable for valuation of multi-attributed environmental effects, as well as for benefit transfer (Carlsson et al 2003). The project builds on the experiences using this method in a recently finalised project on groundwater valuation (Hasler et al 2005). Hasler et al (2005) valued improvements of both surface water quality and drinking water quality. In a choice experiment study, respondents are requested to choose between pre-defined alternatives, expressing environmental effects from scenarios, which are each described by different costs, indicators for water quality and other environmental impacts. Hereby the respondents are provided with an explicit basis for assessing benefits and costs in relation to the relevant effects and, therefore, the method is recommended in situations where an environmental change or project can be ascribed to several attributes.

Cost assessment at farm, regional and macroeconomic levels

Two different analytical tools are used for assessing the costs of different local and general nutrient abatement scenarios. Compliance and adjustment costs at farm and catchment levels are assessed using mathematical optimisation models, whereas costs at regional and national level are assessed using an Applied General Equilibrium model. The two types of model are carefully developed and used to ensure that results at different levels for the same scenarios are consistent.

At farm and catchment levels, a number of farm models will be developed and used as building stones for river basin models. The aim is to estimate the distribution and level of the production of crops, husbandry, fertilisation application, spraying, etc. in the catchments to the water bodies within the river basins. For each catchment, the costs to farmers and the public sector of different policy measures are quantified. Finally, costs for different catchments are compared. These farm level models are non-linear mathematical models, based on profit maximisation, under certain binding restrictions such as crop rotation, climate, soil type, regulatory measures, etc. The models are built on data on acreage, livestock production, crops, fertiliser use, soil types, etc., by use of data from the general registers of crops and husbandry (GAR/CHR), as well as the fertiliser accounts. The farm type models will be spatially distributed and hereby aggregation to catchment levels and river basins are facilitated. The models will be developed upon the experience from previous research (cf. Hansen 2004, Hasler 1998, Vatn et al 2001, Brady 2003). To facilitate assessment of the reductions in nitrogen and phosphorus emissions, nitrogen leaching functions and phosphorus index will be linked to the models. At the macro and regional level, the Agricultural Applied General Equilibrium model of the Danish economy (AAGE) is used. The AAGE model is originally based on the Australian static ORANI model with a detailed description of Danish agriculture. The AAGE model has been developed into a fully dynamic model (Adams 2000). The model allows for regional assessment of policy instruments, focusing on the behaviour in Danish agriculture, and also for regional and economy wide assessment of using different policy measures in different regions. The overall outcome of this work package is an integrated and holistic model framework that allows the consequence at national, regional, catchments level, and farm level to be addressed. The results include cost-effectiveness analyses and assessments of local specific and general abatement measures towards both nitrogen and phosphorus, including assessments of the final impacts on the aquatic environment, if the environmental knowledge to assess this suffices. The results also include changes in national welfare, GDP, import, export, sectorial impact on production and employment, regional impact for the agricultural sector's relation to production, employment, and land use.

Benefit transfer

Valuation studies are often expensive, and benefit transfer can be a cost-effective alternative as compared to primary valuation studies of the aquatic environment. Benefit transfer reduces the need for new studies in areas being interesting for policy purposes, but where original valuation studies have not been performed.

Cost-benefit assessments

In this final part the costs and benefits of the different scenarios are compared and benefit-cost ratios are measured. An important element – and uncertainty - of such a cost-benefit analysis is the connections between the changed emissions of nitrogen

and phosphorus from the root zone and the field, the resulting loads to the water bodies and the changed environmental conditions in these recipients. Analysis of this is not part of this project, but uncertainties will be discussed. However, the main focus in this part of the project will be on demonstrating how cost-benefit ratios can be measured, as a basis for further innovation and use when the environmental models are developed further. The conclusions will be disseminated to relevant stakeholders, and discussed.

Expected results and benefits

The results can directly aid the implementation of the Aquatic Action Plan III, as well as the Danish implementation of the WFD, and used by national and local authorities. Both costs and benefits are assessed and compared within a welfare economic framework at local level and national level, and the development of the aquatic environment is related to the general structural development in the agricultural production.

References

- Adams, P.D., 2000. Dynamic-AAGE: A Dynamic Applied General Equilibrium Model of The Danish Economy Based on AAGE and MONASH models. Report number 115. Available from the Danish Research Institute of Food Economics.
- Carlsson, F., Frykblom, P. & Liljenstolpe, P., 2003. Valuing wetland attributes: an application of choice experiments. *Ecological Economics* 47, 95-103.
- The Danish Economic Council (DORS), 2004. Danish Economy, Autumn 2004.
- Dixon, P.B., Parmenter, B.R., Sutton, J.M. & Vincent, D.P., 1982. ORANI: A Multisectoral Model of the Australian Economy, Amsterdam, North-Holland.
- Hansen, L.G., 2004. Nitrogen Fertilizer Demand from Danish Crop Farms. *Canadian Journal of Agricultural Economics* 52(3), 313-332.
- Hasler, B., Lundhede, T., Martinsen, L., Neye, S. & Schou, J.S., 2005. Valuation of Danish groundwater protection with the Choice experiment and Contingent valuation methods. NERI scientific report. Forthcoming.
- Jacobsen, L.-B. & Frandsen, S.E., 1999. Analyse af de sektor- og samfundsøkonomiske konsekvenser af en reduktion i forbruget af pesticider i dansk landbrug (Report on the Sectoral and Economy-Wide Effects of a Ban on Pesticides in Danish Agriculture, summary in english), Danish Institute of Agricultural and Fisheries Economics, report no. 104.
- Schou, J.S. & Ørum, J.E., 2004. Omkostninger ved reduktion af landbrugets næringsstoffab til vandmiljøet - Forarbejde til Vandmiljøplan III = Costs of Reducing Nutrient Losses from Agriculture - Analyses prior to the Danish Aquatic Programme III. Fødevareøkonomisk Institut. - Rapport 167. 299 pp. Available at: <http://www.foi.dk/Publikationer/Rapporter/rap167.pdf>
- Turner, R.K., Bateman, I.J., Georgiou, S., Jones, A., Langford, I.H., Matias, N.G.N. & Subramanian, L., 2004. An ecological economics approach to the management of a multi-purpose coastal wetland. *Regional Environmental Change* 4(2-3), 86-99.

Mapping phosphorus sorption capacity in soils

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Introduction

Many Danish lakes and some estuaries suffer from eutrophication due to diffuse phosphorus (P) losses from agriculture. Much of Denmark has for decades been intensively farmed and received large surplus P additions. Soil P status, therefore, is generally high or excessive. The capacity of a soil to retain added P much influences its vulnerability to P loss. In native soils, total P content and degree of P saturation (DPS) are directly related to geology and soil formation. In agricultural soils DPS are mainly the result of surplus P additions. The P sorption capacity (PSC) of soils, however, depends on inherent soil characteristics. The PSC of mineral soils may, therefore, be mapped using geo-referenced soil databases and geological maps. In Denmark, soil depth, texture, pH and content of various iron and aluminium oxides are considered the most important determinants for PSC in the root zone.

Pedotransfer functions for PSC based on oxalate-extractable aluminium (Al_{ox}) and iron (Fe_{ox}) (Borggaard et al., 2004; Van der Zee, 1988) are currently being evaluated and combined with the extensive Danish soil database to create national maps of PSC at three soil depths (<http://www.djfgeodata.dk/>). This information will be incorporated in a Danish P Index modifying the potential for P retention in subsoils. Here we present the approach used and the first results of mapping Al_{ox} and Fe_{ox} in soils.

Materials and methods

The soil data used here derive from a nationwide 7-km-grid sampling scheme. Composite samples of 16 soil cores were collected at fixed depth intervals of 0.25 m to a depth of 1.0 m from a 50 by 50 m area at each grid node. Soil profiles were described at all grid nodes. Data from a total of 386 grid nodes were included in this investigation. Al_{ox} and Fe_{ox} were determined according to Borggaard et al. (2004).

These data were extrapolated to all of Denmark by assuming that certain geographic areas can be assigned typical Al_{ox} and Fe_{ox} contents. For this purpose the country was subdivided into landscape typological units (LTUs). The LTUs were established on the basis of a geological soil map and geo-regionalization. The geological soil map at the scale of 1:50.000 (Hermansen and Jakobsen, 1999) distinguished 11 quaternary soil classes according to the soils' geo-genetic origin and their texture

(gravel, sand, silt, clay). The geo-regionalization yielded five geo-regions based on differences in the glacial and post-glacial history of the landscape.

Table 1. Average contents of oxalate-extractable Al, Fe, P, clay (<2 µm) and fine sand (63-200 µm) in the B horizon (0.5-0.75 m) for each LTU. N is the number of observations.

Geo-region	Geological soil class [#]	N	Al	Fe	P	Clay	Fine sand
			mmol kg ⁻¹			%	
North Jutland	FS	1	28.5	8.1	1.9	n.d.	n.d.
	HS	2	12.5 (2.8)	17.8 (11.7)	2.7 (0.7)	3.6	87.8
	MS	6	59.3 (23.5)	43.6 (36.1)	12.0 (13.3)	8.9	64.2
	YS	6	52.5 (18.9)	35.4 (16.8)	6.8 (3.9)	5.8	69.9
THY	DS	5	39.7 (17.6)	39.4 (17.8)	9.0 (5.5)	12.4	41.7
	ES	1	4.4	6.1	0.6	n.d.	n.d.
	HS	1	5.8	5.9	2.3	2.7	89.5
	ML	13	34.8 (9.6)	43.6 (12.7)	6.3 (3.7)	15.7	34.7
	MS	1	42.9	36.1	4.0	13.2	48.7
	TS	1	59.4	16.0	1.7	n.d.	n.d.
West Jutland	DS	28	47.5 (14.5)	26.8 (15.5)	3.6 (2.5)	7.7	31.4
	ES	3	50.7 (13.6)	35.4 (11.1)	3.2 (1.3)	11.3	49.4
	FS	9	43.6 (22.3)	22.3 (12.2)	3.3 (3.3)	3.9	26.4
	HS	1	12.2	26.9	5.2	3.7	72.3
	ML	15	49.4 (20.9)	28.0 (12.3)	2.8 (1.8)	12.5	39.5
	TS	25	49.7 (23.9)	21.4 (11.8)	4.3 (2.5)	5.4	22.8
Central Jutland	DS	24	52.7 (16.7)	27.5 (9.9)	5.7 (2.7)	8.0	36.8
	ES	1	71.0	17.2	5.1	6.2	39.6
	FS	6	35.2 (15.7)	23.1 (12.8)	6.0 (3.7)	7.8	31.0
	HS	2	14.8 (2.9)	37.0 (38.4)	3.6 (2.9)	5.1	19.6
	ML	48	37.2 (15.2)	34.8 (13.7)	4.7 (3.2)	13.6	31.6
	MS	4	52.8 (17.5)	27.0 (2.6)	5.8 (1.3)	10.1	50.2
	TS	5	39.5 (11.1)	16.5 (10.0)	3.5 (1.8)	3.8	30.4
East Denmark	ZK	1	13.9	8.4	4.2	7.6	46.5
	DL	1	30.2	30.9	10.2	21.4	35.1
	DS	15	35.2 (12.7)	32.0 (10.9)	6.4 (2.7)	9.5	33.7
	FS	16	24.7 (6.5)	30.5 (9.1)	4.5 (1.4)	14.6	36.4
	HS	3	14.8 (8.4)	45.7 (49.7)	6.4 (3.5)	15.0	36.8
	ML	95	31.3 (7.3)	35.2 (10.1)	5.2 (2.4)	16.4	35.7
	MS	1	26.1	33.5	6.6	14.0	48.2
TS	1	32.7	39.0	5.6	14.7	30.0	

[#] **DL**: glacial deposit in lakes, alluvial clay; **DS**: glacial deposit, alluvial sand; **ES**: postglacial aeolian sand; **FS**: postglacial alluvial sand; **HS**: postglacial marine sand; **ML**: glacial till >12% clay; **MS**: glacial till <12% clay; **TS**: late-glacial alluvial sand; **YS**: late-glacial marine sand; **ZK**: clay deposit in glacial lakes, kames; **n.d.**: not determined; (Standard deviation in brackets).

Results

Combining geo-regions with relevant geological soil classes resulted in 31 distinct LTUs (Table 1) mapped at a 500-m grid resolution. For each sampling point, the LTU

was derived from the soil profile database. Al_{ox} and Fe_{ox} contents were averaged within each LTU (Table 1) and assigned to the national LTU map correspondingly (Fig. 1). Lowland soils are omitted, as the origin and the pedological transformations of Al and Fe oxides in these soils are very variable and not closely associated with the LTUs. An alternative classification for lowland soils and suitable pedotransfer functions for PSC are under development (Kjaergaard et al., 2007).

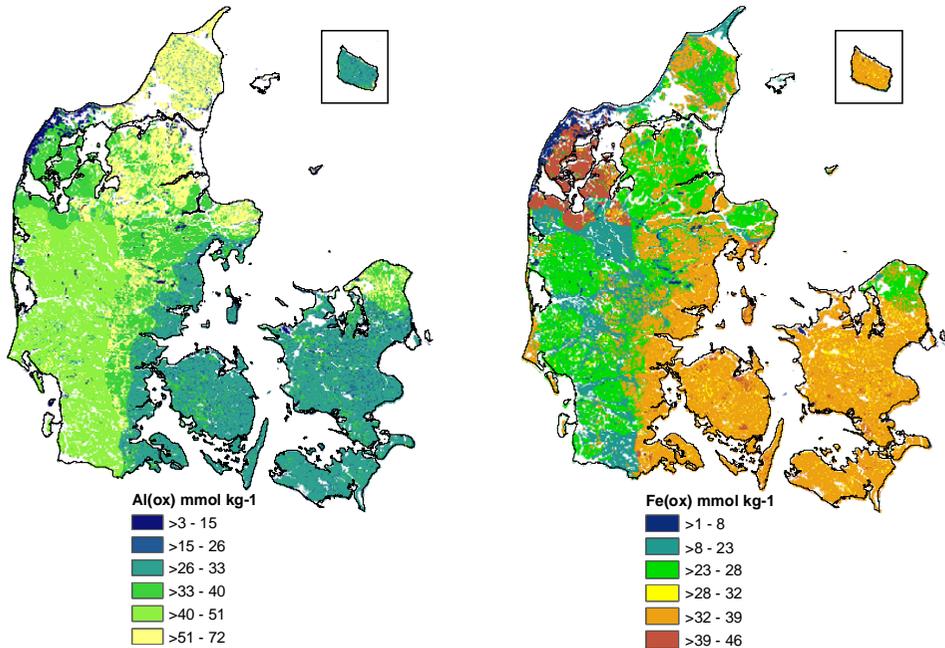


Figure 1. Contents of oxalate-extractable Al (left) and Fe (right) in the lower B horizon (0.5-0.75 m) of mineral soils in Denmark. Lowland soils are blanked.

The resulting median Al_{ox} and Fe_{ox} contents of the area mapped were 37 and 32 $mmol\ kg^{-1}$, respectively. Assuming equal importance for P retention, Al_{ox} and Fe_{ox} contents may be added to yield an expression for PSC. The respective 25th, 50th and 75th percentiles were 66, 71 and 77 $mmol\ (Al+Fe)_{ox}\ kg^{-1}$. Contents <60 or >100 $(Al+Fe)_{ox}\ mmol\ kg^{-1}$ are considered, respectively, low or high in PSC (H.E. Andersen, pers. comm.). Hence many subsoils are mapped as having a low to moderate PSC.

References

Borggaard, O.K., Szilas, K., Gimsing, A. & Rasmussen, L., 2004. Estimation of soil phosphate adsorption capacity by means of a pedotransfer function. *Geoderma* 118, 55-61.

- Hermansen, B. & Jakobsen, P.R., 1999. Danmarks digitale jordartskort 1:25 000. CD-ROM version 1.0. Danmarks og Grønlands Geologiske Undersøgelse Rapport 1998/79. GEUS, København.
- Kjaergaard, C., Hoffmann, C.C. & Greve, M.H., 2007. Phosphorus forms and phosphorus release as affected by organic lowland geochemistry. This issue.
- Van der Zee, S.E.A.T.M., 1988. Transport of reactive contaminants in heterogeneous soil systems. Ph.D. thesis. Agricultural University, Wageningen, The Netherlands.

Mapping the risk of P loss through soil macropores

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Introduction

In Denmark an interactive web-based tool for mapping agricultural areas at risk of phosphorus (P) losses is presently being developed. This tool is essentially based on the P Index concept (Lemunyon and Gilbert, 1993; Sharpley et al., 2003). P loss to tile drains mainly by colloid-facilitated transport in macropores is an important process in Denmark and hence ought to be accounted for in the tool. The initiation of water transport in soils with macropores is to a large degree controlled by the hydraulic conductivity in the soil matrix. The objectives of the study were firstly to develop a point pedotransfer function (PTF) predicting the near-saturated hydraulic conductivity (here defined as the conductivity at a soil water potential of -0.5 kPa) using simple soil parameters as predictors. Secondly, to use this information in connection with a newly developed raster-based soil property map of Denmark to identify areas with a high risk of P macropore flow.

Materials and methods

The dataset was based on measurements on almost 500 large soil columns (6280 cm³) sampled at 68 different localities in Denmark covering a variety of different soil types. In the laboratory, the near-saturated hydraulic conductivity was measured on a drip infiltrometer (van den Elsen et al., 1999). Saturated hydraulic conductivity was measured using the constant head method. In addition, soil water characteristics were measured on 100 cm³ soil samples. To determine the near-saturated hydraulic conductivity, hydraulic measurements were optimized using the van Genuchten (1980) and Mualem (1976) models combined with the power function of Børgesen et al. (2005). The developed PTFs were based on neural networks and the Bootstrap method using different sets of predictors. The Bootstrap method is a non-parametric technique to estimate model uncertainty by simulating alternative (replica) data sets out of a single, and presumably representative, data set. Sixty replica data sets were generated each of which was used to calibrate the neural network model. Different combinations of five texture classes, amount of organic matter, soil horizon, and bulk density were used as predictors in the neural network.

The results from the neural networks together with newly developed raster based soil property map of Denmark (Greve et al., 2006) were used to construct a map showing the near-saturated hydraulic conductivity in the A-, B-, and C horizon down to a depth

of one meter. As a tool to point out areas having a high risk of leaching through macropores, water flow was simulated using the one-dimensional hydrological model, HYDRUS-1D (Simunek et al., 2005). In the A- and B horizon, the near-saturated hydraulic conductivity was divided into five equally spaced classes. In the C horizon, the conductivity was divided into four equally spaced classes. As upper boundary condition for HYDRUS-1D, representative weather data for a ten year period were used at hourly timesteps. Winter wheat was the model crop for the whole simulation period. The lower boundary condition was a groundwater table at a constant depth of 2 m. To represent the soil water retention and hydraulic conductivity functions in the model, parameters of the van Genuchten (1980) and Mualem (1976) models were used as input. The near-saturated hydraulic conductivity developed from the PTF was defined as the saturated hydraulic conductivity in the soil matrix. The remaining van Genuchten-Mualem parameters were estimated from the HYPRES database (Wösten et al., 1999). Even though HYDRUS-1D is able to simulate water transport in the soil matrix only, it is believed that the model is able to give a qualitative estimate of the risk of macropore flow. When the infiltrability in the soil matrix is exceeded, water is supposed to flow out into the macropores. Therefore, there will be a high risk of macropore flow in areas where the soil matrix frequently is saturated.

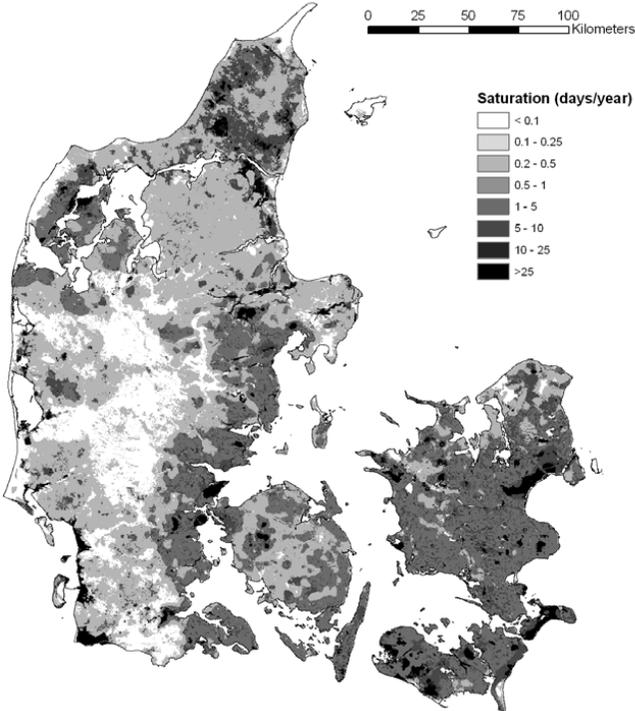


Figure 1. Map of Denmark showing areas with varying risk of water saturation in the soil matrix (saturation at the soil surface). Grid size is 250 m.

Results and discussion

Using four texture classes as input to the neural network gave a PTF with a root square mean error (RMSR) of 0.63 cm d^{-1} . Using four texture classes, amount of organic matter, horizon, and bulk density the RMSR decreased to 0.58. The results therefore showed that the neural network was able to develop reasonably accurate PTFs predicting the near-saturated hydraulic conductivity. The modelling results were converted to a map of Denmark showing varying degrees of areas with risk of saturation in the soil matrix (Fig. 1). Qualitatively, the map can be used to indicate areas with a high risk of macropore flow.

References

- Børgesen, C.D., Jacobsen, O.H., Hansen, S. & Schaap, M.G., 2006. Soil hydraulic properties near saturation, an improved conductivity model. *J. Hydrol.* 324(1-4), 40-50.
- Greve, M.H., Greve, M.B., Bøcher, P.K., Balstrøm, T. & Madsen, H.B., 2006. Generation of a raster based nationwide topsoil property map of Denmark combining choropleth maps and point information. *geoEnv 2006*, 6th International conference on Geostatistics for Environmental Applications, Rhodos, Greece.
- Lemunyon, J.L. & Gilber, R.G., 1993. The concept and need for a phosphorus assessment tool. *J. Prod. Agric.* 6, 483-496.
- Mualem, Y., 1976. A new model for predicting the hydraulic conductivity of unsaturated porous media. *Water Resour. Res.* 12, 513-522.
- Sharpley, A.N., Weld, J.L., Beegle, D.B., Kleinman, P.J.A., Gburek, W.J., Moore, P.A. & Mullins, G., 2003. Development of phosphorus indices for nutrient management planning strategies in the United States. *J. Soil Water Conserv.* 58(3), 137-152.
- Simunek, J., van Genuchten, M.Th. & Sejna, M., 2005. The HYDRUS-1D software package for simulating the movement of water, heat, and multiple solutes in variably saturated media, Version 3.0, HYDRUS Software Series 1, Department of Environmental Sciences, University of California Riverside, Riverside, California, USA, 270 pp.
- van den Elsen, E., Stolte, J. & Veerman, G., 1999. Three automated laboratory systems for determining the hydraulic properties of soils. p. 329-340. In van Genuchten, M.T., Leij, F.J., and Wu, L. (ed.) *Characterization and measurement of the hydraulic properties of unsaturated porous media, Part 1*. U.S. Salinity Laboratory, USDA-ARS. Department of Environmental Sciences, University of California, Riverside, CA.
- van Genuchten, M.T., 1980. A closed-form equation for predicting the hydraulic conductivity of unsaturated soils. *Soil Sci. Soc. Am. J.* 44, 892-898.
- Wösten, J.H.M., Lilly, A., Nemes, A. & Le Bas, C., 1999. Development and use of a database of hydraulic properties of European soils. *Geoderma* 90, 69-185.

An expert system for predicting rill erosion in Denmark

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Introduction

In Denmark efforts to abate eutrophication have raised the concern for water erosion as a process transporting sediment and phosphorus from agricultural land to waters. As the relative importance of diffuse agricultural P loss for eutrophication has been increasing since the early 1990s, monitoring programmes have confirmed the impact of water erosion (Kronvang et al., 2001). To facilitate the planning of appropriate mitigation, there has been a demand for a practical tool predicting the risk and the magnitude of water erosion on agricultural land. Previous attempts with the USLE (Universal Soil Loss Equation) have proved unsatisfactory in Denmark, where prevalent topographic, climatic and soil conditions vary from those the USLE was developed for. In Denmark water erosion frequently is associated with snowmelt or rainfall on frozen soil, the topography typically is gently rolling with short slopes, and rainfall intensity generally is low. Overall, water erosion is a relatively infrequent and highly variable event under Danish conditions suggesting that the probability of erosion occurring ought to be explicitly considered in modelling.

Here we present a summary of the results from a project that *i*) quantified erosion rates by means of rill erosion surveys on a wide range of agricultural land in Denmark and *ii*) developed an expert system for predicting water erosion.

Materials and methods

Twenty study areas represented different soil and climatic conditions. Each study area included up to 21 hillslopes. An experimental unit (slope unit) was either a field with permanent borders comprising a hydrological catchment or a subsection of a field that was hydrologically isolated. In total 189 slope units were chosen. Sandy loams and loamy sands were the dominant soil types. Based on topographic surveys, different terrain attributes were calculated including a two-dimensional LS factor (Desmet and Govers, 1996). At each slope unit soil texture, organic matter and the presence of an impermeable layer in the upper soil profile was determined. In two seasons water infiltration and soil surface roughness were measured and the latter used for calculating an upslope depression index (MUD) (Hansen et al., 1999). Cropping and cultivation practices were classified into six categories: 'cereal stubble', 'ploughed', 'harrowed stubble', 'winter crops', 'grass' and 'spruce plantation'. Daily weather data were taken from the nearest weather stations. At each slope unit

erosion was surveyed and rill volume estimated in late autumn and in spring for a period of five years. Each visit comprised an observation.

In total there were 1041 observations, of which only 213 had erosion. The non-zero erosion rates were highly right-skewed, with a 75% quantile of $1.49 \text{ m}^3 \text{ ha}^{-1}$ and a maximum of $28.4 \text{ m}^3 \text{ ha}^{-1}$. Erosion was never observed with ‘grass’ and ‘cereal stubble’. The collected data consist of two types of explanatory variables: type I are variables, which typically are easy to measure while type II are those difficult to obtain (MUD). Data for type II variables were not collected in about 60% of the cases. The statistical model underlying the expert system was established in two steps. First, based on the subset of the complete data, the structure of the prediction model was determined by regression analysis, and parameter estimates were obtained. Second, the incomplete data were used to improve parameter estimates by means of a Bayesian network using Markov Chain Monte Carlo methods (Gilks et al., 1996).

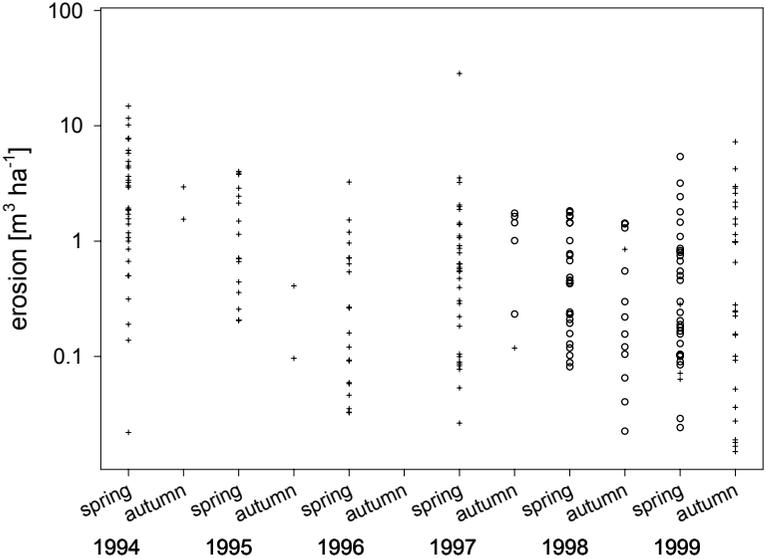


Figure 1. Measured rill erosion rates in spring (S) and autumn (A). Mean Upslope Depression (MUD) and infiltration determined (●) or not determined (x).

Results

The expert system consists of three parts, a logistic model predicting the probability of erosion, a model predicting a conditional erosion rate only in cases where erosion occurs, and a third model predicting the soil surface roughness (MUD). MUD in turn is a variable in the probability model. The actual erosion estimate then is obtained by combining the probability of erosion with the conditional erosion rate. The expert systems’ different components are explained in Figure 2. As erosion was never

recorded for 'grass' or 'cereal stubble', they were not included in the model, i.e. the best estimate of erosion for these categories is 'no erosion'.

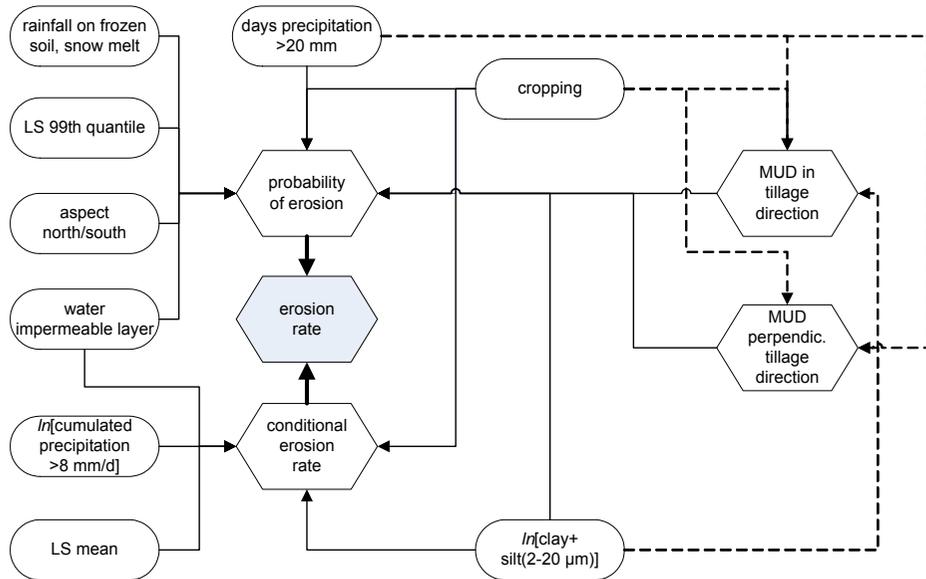


Figure 2. Expert system for predicting rill erosion in Denmark. Hexagons represent response variables. Arrows indicate the variables included in the different models. Dashed arrows indicate an effect on the type II variables. *Days precipitation >20 mm*: number of days with precipitation above 20 mm; *rainfall on frozen soil, snow melt*: sum of precipitation and snowmelt on frozen soil; *ln(accumulated precipitation for days >8 mm/d)*: natural logarithm of cumulated precipitation for days with >8 mm/day; *ln(clay+silt(2-20 μm))*: natural logarithm of percentage clay+ silt(2-20 μm); *LS 99th quantile* and *LS mean*: calculated terrain attribute; *water impermeable layer*: class; *aspect north/south*: class variable for slope unit orientation; *MUD*: experimentally determined mean upslope depression; *cropping*: one of six cropping categories.

Conclusion

The large spatial and temporal variability of water erosion means there exist limited high-risk areas where under certain conditions erosion becomes an environmental problem. Therefore, targeted and cost-effective erosion control requires identification of such high-risk areas. The aim of this study was to develop a practical tool for the prediction of erosion risk that accounted for the specific conditions in Denmark. To this end we built a stochastic, empirical model approach incorporating meaningful physical parameters in the model that are readily accessible for future model users.

References

- Desmet, P.J.J. & Govers, G., 1996. A GIS-procedure for automatically calculating the USLE LS-factor on topographically complex landscape units. *J. Soil Wat. Cons.* 51(5), 427-433.
- Gilks, W., Richardson, S. & Spiegelhalter, D., 1996. *Markov Chain Monte Carlo in Practice*. Chapman & Hall, London.
- Hansen, B., Schjønning, P. & Sibbesen, E., 1999. Roughness indices for estimation of depression storage capacity of tilled soil surfaces. *Soil & Tillage Research* 52, 103-111.
- Kronvang, B., Iversen, H.L., Jørgensen, J.O., et al., 2001. Fosfor i jord og vand. Udvikling, status og perspektiver. 88 s. Faglig rapport fra DMU 380.

Options for reducing the phosphorus surplus in the agricultural economy of Northern Ireland

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Introduction

In regions or countries where the agricultural economy is based on intensive animal production there is often a large imbalance between phosphorus (P) exported in agricultural products such as meat and milk and the much larger inputs of P contained in imported animal feedstuffs. Over time, the accumulation of surplus P causes soil P concentrations to increase with associated higher risk of higher P losses. One such area is Northern Ireland, where production from grass-based agricultural systems is supplemented by imported animal feeds that sustain almost all pig and poultry production but also, to varying degrees, milk, beef and sheep-meat production. Two consequences of intensive agricultural production in Northern Ireland have been: a) the majority of lowland lakes are eutrophic and b) long term increases in dissolved reactive P (DRP) exports from major rivers. These increases have occurred as point sources of P have declined but correlate with increasing levels of soil P. Current eutrophication control measures adopted under the European Nitrates Directive, seek to minimise nitrogen (N) and P losses associated with the spreading of animal manures by enforcing a closed period when runoff rates are highest, setting of safe distances for manure applications near water courses and limiting maximum application rates of manures to crop requirements (albeit for N). These measures will impact on P losses from manures, but do not address increasing soil P, which in turn reflects the imbalance between inputs and outputs of P in most farming systems in Northern Ireland.

This paper presents a summary of a desk-based exercise on the capacity of agriculture in Northern Ireland to reduce the regional agricultural P balance without lowering the productive capacity of agriculture in terms of milk, meat and crop outputs. Three separate strategies are examined:

- 1) Lower P fertiliser use as most soil P levels in lowland soils are above the optimum for maximum crop outputs with zero demand for further P and, for soils at the optimum P index for crop production, crop requirements can be met from P in animal manures. Currently, a regulation adopted in 2007 under the European Waste Directive mandated that chemical P fertiliser used on farms in Northern Ireland cannot exceed crop specific demand for P determined by reference to official fertiliser recommendations, a valid soil P test and the availability of P in manures.

- 2) Reducing the P content of animal diets as research has shown that P in imported concentrate could be lowered from current levels (Ferris et al., 2006).
- 3) The removal of manure P from the agricultural system. The practicality of this option centres on the use of poultry litter as a fuel for combined heat and power units.

Methodology

Data presented are from the annual agricultural census conducted by the Department of Agriculture and Rural Development in Northern Ireland and are used to produce a P budget (Foy et al., 2002). Inputs of P are purchases of chemical fertiliser and imported feedstuffs and outputs of P are sales of meat, milk, eggs and crops. Year 2003 is taken to be the base year for comparison and strategy 1 is a lowering of chemical P fertiliser inputs by 90%. For strategy 2, the P contents of concentrates, expressed as % P on a fresh-weight basis, are assumed to be lowered as follows: daily cattle 0.55% to 0.38% P; beef cattle 0.45% to 0.38% P; sheep 0.55% to 0.38% P; poultry and pig diets 0.65% to 0.55% P. Based on the area of crops and grass but excluding upland rough grazing in Northern Ireland, the regional P surplus is the difference between inputs less outputs and is expressed as kg P/ha

Results and discussion

During 2003 inputs of P from all concentrates exceeded inputs of fertiliser P (10267 vs 8728 tonnes P). The largest single category of feed P input was for milk production, which accounted for 28% of P in concentrates and 15% of total P inputs (Table 1). Each category of animal production (milk, beef, sheep-meat, pigs, poultry-meat and eggs) was in surplus in 2003, in that the sector input of P in concentrate feeds exceeded sector outputs of P in product. These surpluses highlight the potential to utilise P in manures to meet the P demands for grass and crop production. The 2003 fertiliser P input was equivalent to an application rate of 9.6 kg P/ha, compared to the overall surplus of 14.0 kg P/ha. Strategy 1, lowering P fertiliser use by 90% or 8.7 kg P/ha therefore has a proportionally large impact on lowering the P surplus to 5.3 kg P/ha.

The P balance for strategies 1 and 2 shows the combined impact of diet reduction and lowering fertiliser P inputs (Table 1). Compared to the 2003 budget, the sector balance for beef production becomes slightly negative after diet reduction as the concentrate P input of 1768 is exceeded by the sector P output of 1866 tonnes P. However the dairy sector balance remains in surplus after diet reduction with an input of 1966 tonnes P exceeding the output of 1697 tonnes P in milk. The diet reduction strategy lowers the P surplus by 2.5 kg P/ha or 18% of the 2003 surplus. In combination, diet and fertiliser P reduction lowers the surplus to 2.8 kg P/ha.

The final strategy was for manure from poultry production to be exported as a fuel for heat and power generation. In 2003 broiler production was the second largest category of P concentrate input but the industry is efficient in converting feed and P inputs into product so that the poultry meat surplus of 1039 tonnes P in 2003 was equivalent to only 1.1 kg P/ha. With diet reduction this P surplus is further reduced to 703 tonnes P or 0.8 kg P/ha. Including strategy 3 in combination with strategies 1 and 2 therefore only lowers the P surplus from 2.8 kg P/ha to 2.0 kg P/ha.

Table 1. Summary of P outputs and inputs by agricultural sector. Base year = 2003. Strategies 1&2 = P inputs after fertiliser P reduction of 90% and adoption of low P diets.

Outputs		Inputs	
Sector	Tonnes P	Year 2003	Strategies 1 & 2
		Tonnes P	
Arable	784	Fertiliser	8728
		Animal feeds	873
Milk	1697	Dairy	2889
Beef	1866	Beef	2140
Sheep	210	Sheep	275
Total grass based	3773	Total grass based	5304
Pigs	511	Pigs	1922
Poultry meat	1148	Poultry meat	2187
Eggs	119	Eggs	854
Pigs & poultry total	1777	Pigs & poultry total	4963
Total outputs	6335	Total inputs	18995
		Balance	12661
		Balance as kg P/ha	14.0
			2.8

Conclusion

In theory the P budget can be almost balanced while maintaining existing sector outputs of P but diet reduction may incur higher costs. Its impact on the P balance is modest but it has the added benefit of lowering the P content of manures. The benefit to the P balance of using broiler litter as an energy source is small but P surpluses are highest on poultry farms as production takes place on only 0.2% of active farms. On these farms export of manure is often the only option within the limits on stocking rates imposed by the Nitrates Directive.

References

- Ferris, C.P., McCann, M.E.E. & Patterson, D.C., 2006. Improving the efficiency of nitrogen and phosphorus use in livestock systems through dietary modification. Agricultural Research Institute of Northern Ireland 78th Annual Report 2004-2005, pp. 35-53.
- Foy, R.H., Bailey, J.S. & Lennox, S.D., 2002. Mineral balances for the use of phosphorus and other nutrients by agriculture in Northern Ireland from 1925 to 2000 – methodology, trends and impacts of losses to water. Irish Journal of Agricultural and Food Research 41, 247-263.

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The central role of diffuse phosphorus (P) losses in eutrophication of surface waters has long been recognized. Eutrophication impairs ecological quality and biodiversity of aquatic ecosystems, restricting the use of surface waters for drinking water abstraction and recreation.

The 5th International Phosphorus Workshop in Silkeborg, Denmark, 3-7 September 2007 has been jointly organized by the National Environmental Research Institute and the Faculty of Agricultural Sciences, both from the University of Aarhus. The scope of the workshop is holistic and comprises P cycling and P loss from agriculture, tools for predicting and mapping the risk of P loss, effectiveness of different mitigation options, and the impact of P on the aquatic environment.

This report contains the extended summaries of the 112 presentations made at the workshop by international experts following up on the latest developments and focusing on strategies for abating P losses to the aquatic environment.

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