Sediment and Phosphorus

Erosion and delivery, transport and fate of sediments and sediment-associated nutrients in watersheds

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Preface

The papers in these proceedings were presented at a workshop on Erosion and Delivery, Transport and Fate of Sediments and Sediment-associated Nutrients in Watersheds partly sponsored by the Centre for Freshwater Environmental Research and the Centre for Root Zone Processes under the Danish Environmental Research Programme. The workshop was held at the National Environmental Research Institute, Silkeborg, Denmark in October 1995. The four-day workshop included 27 oral presentations and 10 poster contributions, plus one field excursion, and was attended by about 60 scientists and managers from 7 countries.

The aim of these proceedings is to provide a representative selection of new research topics and aspects of diffuse pollution which have become the main focus of surface and ground water pollution abatement efforts in many countries throughout the world. Abatement of diffuse pollution is difficult because of the complex nature of the processes that link the sources and the water bodies as well as the processes governing the transport and fate in watersheds. Diffuse pollution and the transport and fate of sediment and sediment-associated nutrients in watersheds is thus a broad topic which cannot be covered comprehensively in a single proceedings. The keynote papers and the paper contributions in these proceedings do, however, reflect important ongoing research topics and methods.

A secondary aim of the workshop was to provide a forum for discussions among researchers and managers from different disciplines working in the field of diffuse pollution. This is difficult to achieve but the volume contains contributions from scientists covering computing, agronomy, soil science, sedimentology, biology and geology disciplines.

The contributions to the proceedings are arranged in four sessions:

I Surface Runoff, Erosion and Delivery of Sediment and Sediment-associated Nutrients to Watercourses.

II Subsurface Runoff and Delivery of Sediment and Associated Nutrients to Watercourses.

III Transport of Sediment and Nutrients in Streams.

IV Fate of Sediment and Nutrients in Watersheds.

These section titles represent the division of the topic into broad areas corresponding to routing of sediment and nutrients within watersheds. Moreover, the proceeding includes a concluding report from each of the four technical sessions assembled by a rapporteur. The workshop included three key-note papers provided by Prof. D.E. Walling, UK, Prof. G. Govers, Belgium and Dr J.M. Dorioz, France.

Many people have contributed to the arrangement of the workshop and the production of this volume. Mrs Hanne Kjellerup Hansen and Mrs Winnie Meilstrup facilitated the smooth running of the workshop. Field excursions were organized by Drs L.M. Svendsen, B. Kronvang, R. Grant and C.C. Hoffmann (National Environmental Research Institute, Denmark). Finally, we thank Mrs Anne-Dorthe Matharu and Mrs Kathe Møgelvang, who did the layout of the proceedings report.

The editors

Brian Kronvang
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Erik Sibbesen

Concluding Reports

Report on Session I: Surface Runoff, Erosion and Delivery of Sediment and Sediment-associated Nutrients to Watercourses

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A first and important issue which was discussed during the workshop was the importance and relevance of model building for this type of environmental studies. It was felt that there is a need for models in environmental research: quantitative models are crucial in formalizing our knowledge of the system and in gaining further understanding of the way the system under study operates. They also allow us to evaluate the effect of future changes quantitatively so that possible measures can be objectively compared.

Ideally, a model should take into account all processes present in an as detailed and accurate way possible: it should be spatially distributed and dynamic and ten relevant data to calibrate, validate and operationalize the model should be available. If such model as well as the necessary data are available, the model user can be considered to be in the position of the demon of Pascal: prediction of the system's evolution should be very well possible, at least if the system is not affected by deterministic chaos. Research efforts over the last decades have led to the development of process based, distributed models in many areas of environmental research: however, there was a consensus that the implementation of such models still poses a host of problems, mainly because (i) their theoretical basis if often weak (e.g. the case of modelling erosion by overland flow), (ii) they are very data-intensive and (iii) difficult to validate. It is therefore an important task to develop model structures of a reduced complexity, which would allow to study the effects of the main controlling variables of the system over larger spatial units. Out ability to construct efficient reduced complexity models depends upon our understanding of the system, which is represented in and acquired through the construction of process-based, complex models. It was therefore concluded that there is no fundamental dichotomy between so-called physically-based and empirical models. On the contrary, various types of models have various functions: the type of model should be selected in function of the aim of the study.

With respect to the transport of sediments and associated nutrients to streams there are also important gaps in the available data on processes and process rates. It was clear from the presentations during the workshop that the available data on mechanisms of delivery of eroded sediments to streams is very scarce. Although a basic understanding exists of the factors and processes controlling surface erosion (e.g. on the role of topography, soil type and vegetation), there is virtually no knowledge available on the sediment delivery pathways and the mechanisms of sediment delivery from the slopes to the streams. Essential knowledge on the conditions of sedimentation, size-selectivity and nutrient enrichment during the sedimentation phase is lacking. It was also shown that there can be a substantial contribution to particulate P expert from agricultural land through subsurface drainage. Again, it was concluded that there is no knowledge available with respect to the source of this sediment and the pathways bringing the sediment to the drains.

With respect to the collection of field data, a 'system's approach' seems to be appropriate and should be used in research projects whenever possible: in many studies, the establishment of a sediment and/or nutrient budget allows the identification of important sources and/or sinks of sediments and/or nutrients which would go undetected otherwise, due to our limited understanding of environmental systems. New techniques (e.g. the use of tracers and fingerprinting to identify the sources of sediment) may be of considerable help in this type of research.

Within the discussion it also became apparent that in environmental research other issues than the scientific importance/difficulty of a subject will also have an effect on our strategy. Several contributors insisted on the point that measures with respect to particulate P pollution are required immediately and should be effective. Therefore, the research community should also consider the prospect of practical solutions proposed when selecting research topics: it is clear that it is easier for a farmer to change his management practices that to change the soil type. Research on the effect of different managing systems may therefore contribute more quickly to feasible solutions. On the other hand, it has to be mentioned that our limited understanding of environmental systems as well as the long response times of such systems (due to the presence of buffers operating at various spatial and temporal scales) pose important problems and may limit our ability to give accurate answers to the questions asked. It should also not be forgotten that farmers and others who are responsible for land management practices take their decisions within a general socio-economic context where host of other factors than the effectiveness of a given management strategy are important (e.g. market forces). It may therefore well be worthwhile to search for collaboration with socio-economic scientists on such topics as the adaption (or refusal) of the measures proposed and their economic impact.

Report on session II: Subsurface runoff, erosion and delivery of sediment and sediment-associated nutrients to watercourses

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From the session may be concluded that the amount of sediment and phosphorus transported through the soil is important (Laubel et al., Oygaarden et al.) also in comparison with the total transport to the catchment (Grant et al.). Subsurface transport was also shown to sommetimes occur fast and episodic. Probable main source is the topsoil (Grant et al., Oygaarden et al.)

Most processes in the subsurface transport is poorly understood and within the topsoil different forms of sources may contribute. After mobilization, phosphorus either in particulate or in dissolved form fluxes through the soil to the drainage system. Phosphorus may not be in equilibrium with the passed soil by preferential flow. In a loamy soul, macropores were demonstrated to be an important path (Bergen Jensen et al.). Hydrology may have a large impact on the subsurface transport in other ways. By risen ground watertable lower parts of the catchment were shown to highly contribute to the transport as reduced conditions probably occurred (Grant et al.). Thus an important task is mapping such high risk areas of phosphorus leaching since in these areas special strategies may be necessary to make reduction of phosphorus export possible.

Precipitation may have an impact by detachment of particles (Laubel et al.). Weather may also have a direct influence of phosphorus mobilization as freezing and thawing may destruct soil aggregates, and facilitate detachment of particles and expose larger areas for desorption. However, this was not confirmed in some very preliminary results (Bergen Jensen et al.). Freezing and thawing is also known to destruct cellwalls of wintergreen crops and bring about mobilization of dissolved phosphorus. Dryness may induce dust particles susceptible for transport. Weather may also affect the very transport by shrinkage and swelling of the soil. Since the processes of macropore development and phosphorus transport by preferential flow are very poorly understood much more attention should be needed for this research field.

Besides hydrology, soil type was found to be a very important factor for the subsoil transport (Lundekvam, Ulén).

Except for geohydrological conditions, agriculture managements may highly influence subsurface transport thus make it possible to improve the situation. By reduced fertilization the source is reduced. The effect of this could take long time as indicated (Ulén). Better placement of the fertilizer (drilling of fertilizer tube spreader) could probably reduce the amount needed for growth. Agriculture management should improve aggregate stability since this is a significant factor (Lundekvam). Macropore development may be influenced by root channelisation and packing of the soil. Some evidence that tillage practices and grass vegetation affected the phosphorus losses were shown (Lundekvam, Ulén) but further investigations are highly needed in combination with other factors involved. Thus to reduce the phosphorus leaching, research of the impact of agricultural practices of subsurface transport is very important. Because of the high interaction with climate conditions, field measurements should be longtime lasting to reduce influence of one-sided weather conditions.

Report on session III: Transport of sediment and nutrients in streams

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The papers presented in Session III and the associated discussion highlighted four key areas requiring further attention and research. These were as follows:

i) Problems of flux estimation and design of sampling strategies;
ii) The role of channel storage and remobilisation;
iii) Source fingerprinting: problems and prospects;
iv) Timescales.

Each will be amplified in turn:

i) Problems of flux estimation and design of sampling strategies

Several papers presented in Session III served to underscore the problems of obtaining accurate estimates of sediment and nutrient fluxes based on infrequent sampling programmes (*eg* Kronvang, Laubel and Grant; Vagstad, Deelstra and Eggestad; Smith; and Wiggers. The paper by Wiggers, for example, demonstrated that in Danish catchments *ca* 75% of the P flux may occur in 10% of the time because of the importance of particulate P. Frequent sampling is required to represent these periods of increased P flux and in the case of two small agricultural catchments, the annual total P flux estimated on the basis of frequent sampling was 1.5 to 2.5 times greater than that estimated on the basis of infrequent samples (18-26 samples per year). Sampling strategies used for estimating nutrient fluxes clearly require careful planning in order to obtain an effective trade-off between cost and data reliability. In many cases, information on the *required accuracy* would provide a valuable input to such assessments. Furthermore, flux estimates cited in reports and in papers could usefully be qualified by an estimate of the likely level of accuracy. With the increasing use of flux estimates for establishing sediment and nutrient budgets, caution is required in using budget components estimated as the difference between two 'measured' values which themselves involve a degree of uncertainty.

Outstanding issues which may be identified as requiring further investigation and consideration include:

(a) The potential for developing improved control strategies for automatic sampling equipment. These could, for example, make use of turbidity records in addition to stage/discharge records.

(b) The need to take account of both the *accuracy* (or bias) and the *precision* of flux estimates. The latter is important in influencing reliability in that it provides a measure of the repeatability of flux estimates based on a particular sampling frequency and load calculation procedure and thus of the potential for applying correction factors to take account of bias (i.e. over- or under-estimation).

(c) The need to consider other potential sources of error associated with sediment and nutrient flux estimates and particularly the reliability of water discharge data. Effort in improving sample collection strategies could be wasted if the basic discharge data are themselves unreliable. Use of rated sections where weed growth may introduce shift in the rating relationship could provide discharge data with errors as high as 25%.

(d) The need to take account of the objectives of a catchment investigation when designing improved nutrient sampling strategies. Whereas pooling of samples may be a viable approach for obtaining accurate estimates of total flux, the loss of temporal resolution may be an important limitation for investigations of catchment behaviour *eg* of storm-period response of nutrient concentrations.

(e) Although much work has been undertaken in developing improved sampling strategies for estimating sediment and nutrient fluxes, such improvements are only applicable to current and future investigations. Past data involving infrequent sampling can provide a valuable basis for investigating longer-term trends and should not be discarded. It is important to establish the likely accuracy and precision of such flux estimates so that meaningful comparisons with contemporary data can be undertaken.

(f) Although the problems of obtaining accurate flux estimates are now generally recognised, equivalent attention has rarely been directed to measures of average concentration which are frequently cited in water quality assessments. These are again highly sensitive to sampling frequency and significant differences could exist between values calculated on a flow-weighted and a time-weighted basis. Accuracy and precision considerations are equally important for measures of average concentration.

(g) Whilst recent improvements in the design of sampling strategies have produced increased reliability in annual flux estimates, less is known about the inter-annual variability of such flux estimates. There is a need to investigate the length of record required to provide reliable estimates of long-term mean annual flux. Magnitude and frequency considerations are important here since in the case of sediment fluxes extreme events may cause major increases in sediment transport. Vagstad *et al.*, for example, cited erosion rates as high as 500 t ha^{-1} associated with the Spring 1995 floods in Norway. Such values are probably two orders of magnitude greater than those associated with more 'normal' conditions.

ii) **The role of channel storage and remobilisation**

The papers by Dorioz; Kronvang, Laubel and Grant; Svendsen, Kronvang and Laubel (Session IV) and others have clearly demonstrated the potential importance of channel storage and remobilisation in influencing both the magnitude and timing of suspended sediment and P fluxes. However, a number of uncertainties still exist in developing a full understanding of these processes. These uncertainties include:

(a) The relative importance of sedimentation of particulate-P versus adsorption of dissolved-P by sediment deposits in accounting for P accumulation on river beds.

(b) The extent to which the subsequent transport of remobilised sediment through the river system is continuous or discontinuous, i.e. is it likely to reach the basin outlet or is it likely to be redeposited and subject to further remobilisation, transport and deposition?

(c) The coupling of slope and channel systems and the extent to which seasonal discontinuities may exist in this coupling. Thus, for example, channel storage could represent temporary storage of material delivered from the slopes during the summer months which will be remobilised and transported to the basin outlet during the autumn and winter months when flows are higher.

(d) The need to distinguish remobilisation of temporary storage from channel erosion when evaluating the importance of different sediment sources. Material associated with the former could have originated from surface erosion of cultivated areas.

iii) **Source fingerprinting: problems and prospects**

Several papers presented in the Session have confirmed both the need for, and the potential of, sediment source fingerprinting procedures. However, this approach still requires further refinement if it is to provide meaningful and consistent results. Aspects requiring further investigation and attention include:

(a) The quest for additional fingerprint properties which are capable of providing improved source discrimination.

(b) The need for improved procedures for taking account of contrasts in grain size composition in source/sediment comparisons.

(c) Incorporation of other enrichment effects (*eg* organic fraction) into source/sediment comparisons.

(d) The role of temporary storage/buffers in complicating sediment pathways and source/ sediment linkages and in influencing sediment properties.

(e) Interpretation of channel sources in the light of channel storage and remobilisation. Fingerprinting may indicate that material remobilised from the channel represents sediment originating from 'topsoil' erosion rather than bank erosion. For example, Kronvang *et al.* used a sediment budget to estimate that *ca* 63% of the total P flux from the Gelbaek catchment was derived from 'internal stream bank and bed erosion'. However, the fingerprint of the transported sediment suggests that much of this material originated from topsoil sources. Although it is frequently assumed that rill erosion represents the most important component of surface/topsoil erosion, there is increasing evidence that more diffuse processes (sheet erosion) can represent an important source of surface/topsoil-derived sediment.

(f) The potential for combining sediment fingerprinting with runoff source investigations involving stable isotopes.

(iv) **Timescales**

When investigating the erosion, transport and delivery of sediment and sediment-associated nutrients in watersheds it is important to recognise the importance of the timescale involved. For example:

(a) Results presented in Session III have highlighted the important role of channel storage and remobilisation in attenuating sediment transport through the system. Short-term measurements which fail to take account of this attenuation could provide misleading results.

(b) There is a need to take account of likely system response times when considering potential control and management strategies. Remobilisation of stored sediment and nutrients could delay expected improvements. This is well-demonstrated by the classic work of Stanley Trimble in Coon Creek, Wisconsin. The Coon Creek basin was heavily impacted by accelerated soil loss due to poor agricultural practices in the late 19th and early 20th centuries. Soil conservation measures were subsequently introduced but the sediment yield of the basin remained essentially unchanged since large amounts of sediment were remobilised from the valley floors.

(c) With the lack of long-term records there is still uncertainty as to the role of high magnitude low frequency events in long-term sediment and nutrient fluxes from drainage basins.

(d) It is important to consider the long-term fate of sediment and nutrient sinks. Since it can be argued that all P forms are ultimately bioavailable, remobilisation of short- and medium-term sinks could cause long-term problems. The potential impact of climate change must also be considered when assessing the long-term fate of sediment and nutrient sinks. For example, the use of filter strips is seen as an effective means of trapping sediment and nutrients in channel marginal areas and reducing downstream fluxes. However, future changes in river behaviour consequent upon changes in hydrological regime could result in channel mobility or enlargement which would mobilise sediment stored in such channel margin sinks.

Report on session IV :
Fate of sediment and nutrients in watershed

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Background

It is necessary to develop strategies for the reduction of input of phosphorus (P) and suspended sediment (SS) to prevent the degradation of the quality of many stream ecosystems. Effective strategies require appropriate management efforts that are based on understanding the origins of P and SS in the watershed, how the P and SS are transferred through the watershed, and their impact on the receiving body of water.

Eutrophication resulting from excessive inputs of P into freshwater lake and marine ecosystem are well documented. The intensity of eutrophication depends not only on the magnitude of the inputs but also on the properties of these inputs, such as the bioavailability of P and the capacity of SS to fix P. Additional parameters, such as the distribution of the P inputs over the annual cycle, may also be important determinants of eutrophication.

The origins of P in watersheds are somewhat ambiguously classified as being either a point sources (PS) or a non-point sources (NPS) /1/. Point sources are traditionaly defined as everything that is discharged into the hydrologic network at an identifiable location, whereas non-point sources are considered to be all the remaining discharge of P. Thus, this traditional definition of non-point source includes not only diffuse sources of P from land runoff but also unidentified discharges of wastewater that are very small or intermittent. P and SS reduction strategies based on implementatin of land management and agricultural practices are effective only for controlling diffuse sources of P and sediment. Reduction of P and SS from point sources and the unidentified discharges is accomplished by treatment in appropriate treatment plants.

In many river systems diffuse sources of P (and other NPS, as well) must be managed to acheive water quality goals. The impacts of the NPS input of P and SS on surface water quality in watershed depends on a variety of transport, storage, cycling and export processes that operate in the watershed over a patchwork of different land uses and landscape components. Thus, there is great spatial and temporal variability to contend with in research studies conducted at the watershed scale that are designed to elucidate the dynamics and fate of P and SS in stream systems directly associated with extensive riparian areas, wetlands and floodplains.

Brief review of recent achievements and related new questions

The approaches are based on three main thrusts. Extensive mass balance analysis are used to evaluate the role played by major components of the stream system in the transport, storage and export of P during specific hydrologic conditions. Radioactive tracers work and other specific experiments are used to give insights into particular processes suspected to operate in the stream. Lastly, dynamic simulation modeling is used to describe the dynamics of transport, storage and export over an annual cycle.

Many papers in this workshop confirm that all the P and SS input into the stream network is not immediately transported to the watershed outlet. Often a significant amount of P and SS is stored in the river system and, for P, this storage results in transformation or change in speciation. When considered at an annual scale, a net retention is seen in some specialized landscape features such as wetlands, and in areas that are periodically flooded within the floodplain. Outbank flooding in some stream systems seems to be a very important mechanism for the storage and transformation of P. Thus, these areas in which P is deposited and cycled must be considered as critical land areas to manage in P and SS reduction strategies.

In the stream systems itself storage of P and SS is due to uptake by biota and sediment, deposition during periods of low stream flow, and possible precipitation of excess ortho-P from the water. This in-stream storage is much more temporary and reversible than the outbank processes because of periodic high stream flows scouring the stream channel

The various stream ecosystem compartments, that is, suspended solids and bottom sediments, flowing water, interstitial water, periphyton, and macrophytes are well studied, but the rates of exchange among these compartments and the chemical transformations during this storage are not well understood. The transport of P during high flows of the P accumulated in the river system during low flow periods may play an important role in periodically renewing the storage capacity of the river system. In the stream we studied, autumn stormflows often resulted in major export of P accumulated during the summer period. To summarize, storage in the river (1) delays P and SS transport to the downstream portions of the ecosystem, (2) modifies P speciation, and (3) increases the probability of eventual retention in outbank flooded areas.

Thus, it appears clear that rivers are more than mere conduits through which P and SS pass on their way to the watershed outlet. Although it might assumed that there is an inherent environmental benefit in the in-stream storage and transformation mechanisms, their quantitative effects are largely unknown. Such lack of knowledge represents a major gap in the information needed by managers to construct P and SS reduction strategies.

Conclusions, perspectives

A major accomplishment of this workshop has been to high-light the lack of understanding of the processes in river ecosystems relative to P cycling (P mobility in sediments, exchange rates among biota, water and sediments...). Most of these processes are a function of P and SS concentration and all are controlled by hydrologic conditions (low flow periods, episodic stormflows...) and by the physical properties of the river channels and areas flooded.

Basic research on these processes has begun and interactions between hydrodynamic, biological, and chemical processes in river ecosystems has resulted in the presentation of a dynamic model synthesized from basic principles of growth, adsorption, etc. to assess P residence times, P storage and transport, P export, and P concentrations in flowing water /2/. Such types of model, in order to be of management use, need to include, additionally, changes in P speciation and impact on receiving waters.

Our present level of knowledge and lack of understanding points to long-term goals for the scientists to pursue in order to provide the information needed by managers of river ecosystems and watersheds. However, since managed watershed cannot wait, the scientists are confronted by another more immediate challenge, that is, how, with our available knowledge and lack of understanding of critical processes, can we now provide managers with the tools necessary for effective environmental protection?

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Technical Session I

Surface Runoff, Erosion and Delivery of Sediment and Sediment-associated Nutrients to Watercourses

Illuminating drainage basin sediment delivery systems

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Introduction

Recent years have evidenced a growing awareness of the off-site impact of the increasing rates of soil erosion occurring in many agricultural areas. In addition to the essentially physical effects of increased suspended sediment loads in river systems, fine sediment may also play an important role in the transport of nutrients and contaminants. Against this background, there is an important need for an improved understanding of the processes associated with the mobilisation and transfer of sediment within agricultural drainage basins. There have been important advances in the understanding and modelling of on-site soil loss /1,2/, but attempts to develop meaningful models linking on-site soil loss with downstream sediment yield face many uncertainties. The complex suite of pathways and processes interposed between the mobilisation of sediment on the slopes of a drainage basin and its final exit from the basin is frequently referred to as the sediment delivery system, and the sediment delivery ratio concept has been widely used in the past as a simple and convenient means of representing this system /3/. This lumped and essentially black box approach is, however, of limited value when attempting to predict both output fluxes and the fate of sediment moving through the system. Although there has undoubtedly been significant progress in the development of distributed sediment yield models /4,5/, their success has been hampered by a limited understanding of the delivery system and by a lack of field data for model verification and testing. In this context, there is evidence of an increasing gap between the work of modellers and of empirical field scientists. The latter could argue that model development lacks a sound empirical base and rigorous verification, but the former could equally contend that current measurement programmes have been unable to supply the information required for model development and testing. Although laboratory experiments are able to provide important and valuable information on process dynamics for model formulation, there is undoubtedly a need for 'real world' data for model development, verification and testing. This in turn requires the development of new approaches to field measurement capable of generating both an improved representation and understanding of the sediment delivery system and data that meet the needs of model development. Several examples of recent advances in field measurement techniques which are capable of meeting this need and illuminating drainage basin sediment delivery systems are presented below. These include the use of multicomponent fingerprinting techniques to identify suspended sediment sources, the use of fallout radionuclides to trace sediment movement from source to sink or basin outlet and methods for determining the **in situ** grain size characteristics of sediment transported through the system.

Use of multicomponent fingerprinting procedures to identify suspended sediment sources

If used correctly, existing measurement techniques are capable of generating reliable information on the suspended sediment output from a drainage basin /6/. However, such spatially-lumped measurements of sediment output provide no indication of the nature and relative importance of the **sources** involved. In some situations it will be important to identify the **spatial location** of the major sediment sources, whereas in others, information on the relative importance of different source **types** may be required. Such information can also afford a valuable basis for verifying simulations provided by distributed sediment yield models, since successful reproduction of the record of sediment flux is not in itself confirmation that the actual processes of sediment mobilisation and delivery have been successfully simulated. Information concerning the source of the suspended sediment transported by a river is, however, notoriously difficult to assemble using traditional monitoring techniques. The 'fingerprinting' technique offers an alternative approach to documenting both the spatial location and the nature of the major sediment sources operating within a drainage basin /7,8/. In essence, the fingerprinting technique makes use of chemical and physical properties of the suspended sediment to trace its source. It involves, firstly, selection of physical or chemical properties that clearly differentiate potential source materials and, secondly, comparison of measurements of the same properties obtained from suspended sediment with the equivalent values for the potential sources to establish the likely source of the sediment. Initial attempts at using the fingerprinting approach to elucidate the major sediment sources in a drainage basin commonly involved the use of a single tracer property. More recent work has, however, demonstrated that it is preferable to use several sediment properties, in order to provide a **composite fingerprint** which is capable of successfully discriminating a range of potential sources /9/. By using composite fingerprints, it is possible to establish the relative importance of a series of sediment sources defined in terms of both location and source type.

Table 1. Mean contributions from individual source types to the suspended sediment load of the River Culm

Source Type	Mean Contribution (%)	Estimated mean annual suspended sediment yield (t km^{-2} year^{-1})
Cretaceous/Eocene		
Pasture	9.2	-
Cultivated	10.6	-
Total	19.8	24.2
Triassic		
Pasture	12.3	-
Cultivated	29.5	-
Total	41.8	51.2
Permian		
Pasture	6.7	-
Cultivated	19.7	-
Total	26.4	17.3
Channel Banks	12.0	3.8

Based on Walling and Woodward (1995) /10/

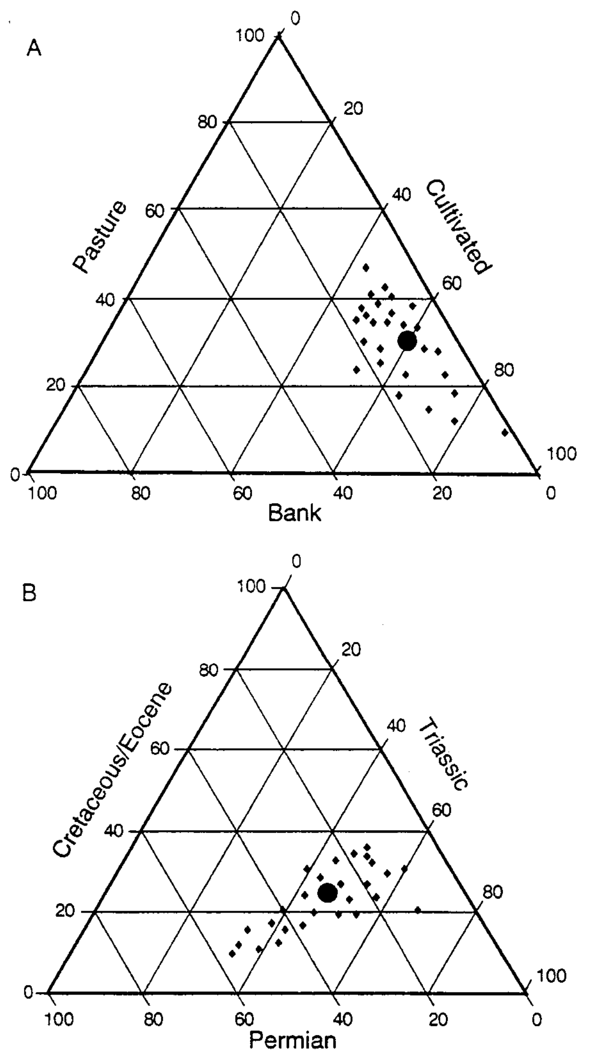


Fig. 1. Ternary plots of the estimates of the relative importance of contributions from cultivated and pasture topsoil and channel erosion (A) and from surface erosion of areas underlain by different rock types (B), to suspended sediment samples collected from the River Culm at the outlet of its drainage basin.

Work undertaken by the author and his co-workers in the 276 km^2 basin of the River Culm in Devon UK /10/, provides a useful example of the potential of this approach. In this case composite fingerprints comprising nine sediment properties were used in combination with a multivariate mixing model to establish the relative importance of seven potential sediment sources. These represented surface erosion from areas of pasture and cultivation within each of three zones of the catchment underlain by different rock types, and channel erosion (cf. Table 1).Using these results it was possible to establish the relative contributions of surface erosion from pasture and cultivated areas and channel erosion to individual suspended sediment samples collected at the basin outlet and the relative importance of the three sub-areas of the catchment as a source of the sediment represented by the same samples (Fig. 1).

Values of the mean contribution from these sources for 27 suspended sediment samples covering a representative range of hydrometeorological conditions were used to provide an indication of the overall importance of the seven potential sources and to estimate the mean suspended sediment yield from the three sub-areas (Table 1). The same general approach could be used to reconstruct changes in sediment sources through time, for example in response to land use change, where lake or floodplain sediment deposits provide a record of the fingerprint of past sediment.

Use of fallout radionuclides to trace sediment mobilisation and storage within a drainage basin

Measurements of sediment output from a drainage basin provide no information on the proportion of the eroded sediment that is transported to the basin outlet, the spatial distribution of sediment mobilisation rates within the basin, or the location and significance of sediment sinks. Such information is essential for establishing a detailed sediment budget for a drainage basin, but it is again difficult to assemble using traditional monitoring techniques. The fallout radionuclides caesium-137 and unsupported lead-210 have, however, been shown to possess considerable potential for tracing sediment movement and storage in drainage basins over periods of 25-100 years. Both of these fallout radionuclides are rapidly and strongly fixed by the surface soil and their subsequent movement is directly related to the erosion, transport and deposition of soil and sediment particles. Measurements of the spatial variation of the inventories of these fallout radionuclides in the soils and sediments of a drainage basin can thus be used to provide information on rates and patterns of erosion and deposition. Caesium-137 is an artificial fallout radionuclide with a half-life of 30.17 years, which was introduced into the environment by the testing of nuclear weapons during the late 1950s and 1960s. In this case, measurements of the contemporary distribution of radiocaesium inventories in the drainage basin provide information on rates of erosion and sediment redistribution over the past 30-40 years. In some areas of the world, however, interpretation of contemporary radiocaesium inventories will be complicated by further inputs of caesium-137 associated with the Chernobyl accident in 1986. Unsupported lead-210 which has a half-life of 22.26 years is, in contrast, a natural fallout radionuclide associated with the uranium-238 decay series. It is derived from the decay of gaseous radon-222, the daughter of radium-226, which occurs naturally in soils and rock. Diffusion of a small proportion of the radon-222 from the soil introduces lead-210 into the atmosphere and its subsequent fallout provides an input of this radionuclide to surface soils and sediments which is not in equilibrium with its parent radium-226. This component is termed **unsupported** since it cannot be accounted for (or supported) by decay of the **in situ** parent. Due to its natural source, inputs of unsupported lead-210 fallout to the land surface may be viewed as being essentially constant through time and it therefore offers potential for documenting rates of erosion over longer time periods than caesium-137 (ie. 50-150 years). Both caesium-137 and unsupported lead-210 have been successfully used to document rates and patterns of soil redistribution on agricultural land /11/ and to investigate rates of deposition on floodplains /12/ and their potential in sediment budget investigations can be usefully demonstrated by introducing results from two studies undertaken by the author and his co-workers. The first relates to an investigation of rates of erosion and soil redistribution within a small (7.5 ha) cultivated field at Butsford Barton near Colebrooke, Devon, UK, /13/. In this case more than 200 bulk soil cores for caesium-137 analysis were collected from the field at the intersections of a 20 m grid. The pattern of erosion and soil redistribution within this field estimated using caesium-137 measurements is shown in Figure 2 and key parameters of its sediment budget are listed in Table 2.

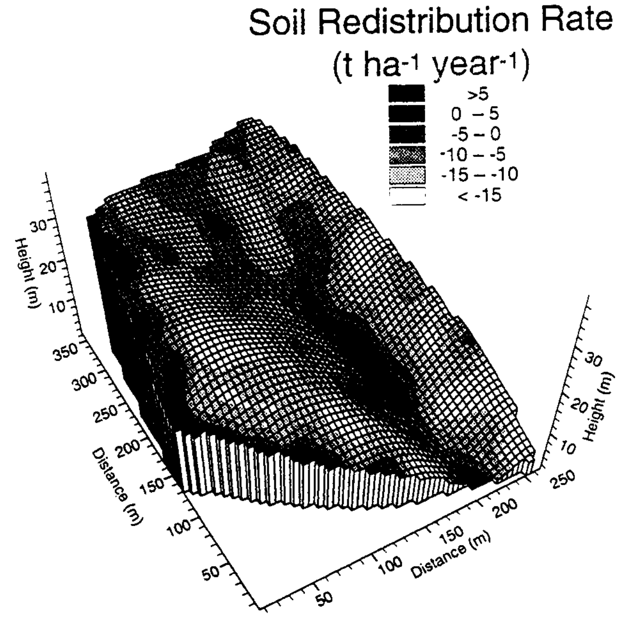


Fig. 2. The distribution of soil redistribution rates within the study field at Butsford Barton estimated using caesium-137 measurements. Positive values denote deposition and negative values indicate erosion.

Table 2. Soil redistribution rate estimates for the study field at Butsford Barton

Parameter	Estimate
Percentage area eroding (%)	79
Percentage area depositing (%)	20
Mean erosion rate for the eroding area (t ha^{-1} $year^{-1}$)	10
Mean deposition rate for the deposition zones (t ha^{-1} $year^{-1}$)	7.5
Net Erosion rate for the field (t ha^{-1} $year^{-1}$)	6.5
Sediment delivery ratio (%)	81

The data contained in Figure 2 and Table 2 would be effectively impossible to obtain using any other approach and there is clearly potential to use the spatially distributed data obtained from the caesium-137 measurements as a basis for verifying estimates of erosion and deposition rates **within** a drainage basin generated by distributed erosion and sediment yield models. Furthermore, such distributed data are clearly well-suited to coupling with digital terrain models and GIS techniques for development of improved models of the sediment delivery system.

Figure 3 provides an example of the potential for using fallout radionuclide measurements to investigate rates and patterns of floodplain sedimentation /14/ Floodplains can represent important sinks for sediment moving through the delivery system and information on the rates and patterns of deposition involved is required both to establish the magnitude of conveyance losses and to identify areas with enhanced deposition rates where contaminants might accumulate. This example refers to a small portion of the floodplain of the River Culm in Devon, UK, which represents the core or inner area of a meander bend (Fig. 3A). The floodplain area inundated during flood events extends over the entire valley floor at this location. A total of 53 cores were retrieved from the site at the intersections of a 7m x 7m grid for determination of unsupported lead-210 inventories. These values were used in conjunction with information on the grain size distribution of the deposited sediment to derive the map of mean annual sediment accumulation rates for the past ca. 100 years depicted in Figure 3B.

At this location, sediment deposition rates average ca. 0.3 g cm^{-2} $year^{-1}$ and the detailed pattern shows evidence of maximum sedimentation rates in areas close to the channel and relatively low rates in the depressions within the meander bend. Again, such data would be difficult to assemble using any other approach and they could provide a valuable basis for validating distributed floodplain sedimentation models /15/.

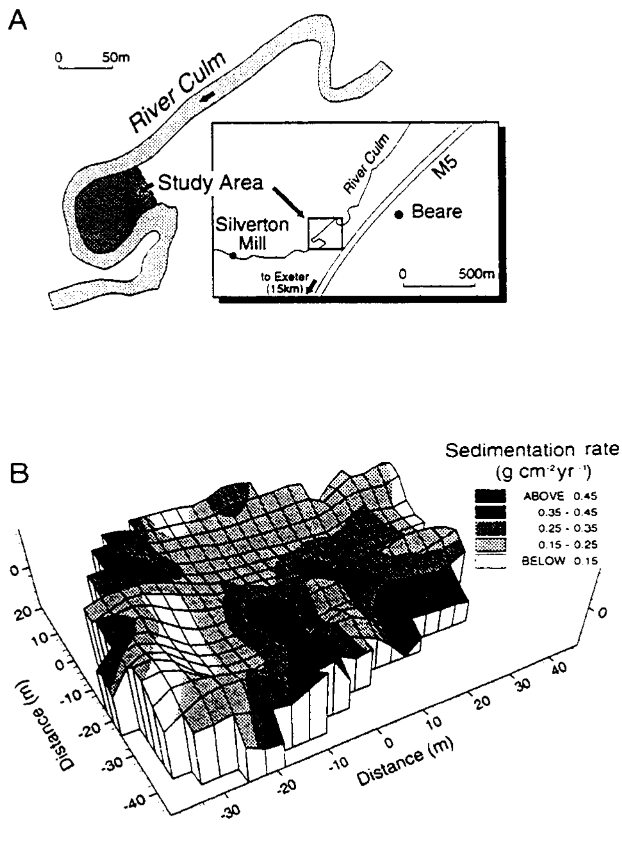


Fig. 3. The study reach of the River Culm at Silverton Mill, Devon, UK, (A), and the distribution of mean annual deposition rates over the past ca. 100 years estimated using unsupported lead-210 measurements, (B).

Determining the *in-situ* grain size characteristics of fluvial suspended sediment

A major limitation of many existing field and modelling investigations of suspended sediment transport and delivery is their failure to take account of the fact that a substantial proportion of the sediment may be transported in the form of aggregates or composite particles and to recognise the need to distinguish traditional laboratory measurements of **ultimate** grain size composition, based on chemically dispersed mineral sediment, from the **in-situ** or **effective** size distribution actually existing in the river.

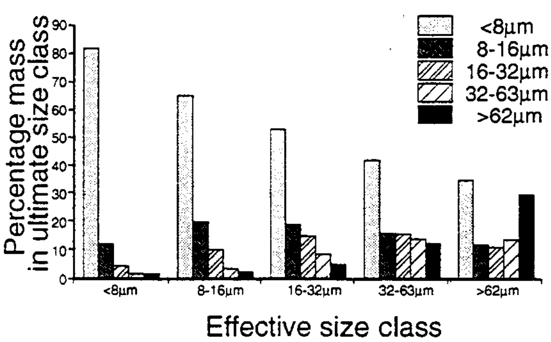
The existence of such aggregates will clearly exert an important influence on depositional processes and conveyance losses in terms of both settling velocities and the grain size composition of deposited sediment. There is a need to develop new procedures for establishing the in-situ or effective grain size characteristics of fluvial suspended sediment and the author and his co-workers have made use of both a portable laser backscatter probe /16/ and a field-portable water elutriation system /17/ for this purpose. Figure 4A provides an example of the results obtained using a water elutriation system to investigate the grain size composition of suspended sediment transported by the River Culm in Devon, UK. In this case the ultimate grain size composition of each of the effective size fractions recovered from the water elutriation system has also been determined using laser diffraction equipment and, although these data are not entirely comparable with the water elutriation data, it is clear that substantial amounts of fine sediment are contained within larger composite particles. To emphasise further the importance of incorporating information on the effective grain size of transported sediment into models of sediment transport and deposition, Figure 4B compares model predictions of the ultimate grain size composition of a near channel overbank floodplain deposit in the lower reaches of the River Culm, obtained using both discrete (ultimate) and composite (effective) particle size data to characterize the transported sediment /18/.

Use of the discrete grain size data causes the model to predict deposition of only limited amounts of fine sediment (ie. 15.3 % < 16 μm), whereas use of the composite grain size data increases the proportion of fine sediment (< 16 μm) to 59 %. The latter value is in close agreement with direct measurements of the ultimate grain size composition of sediment retrieved from sediment traps at an adjacent site. The total mass of deposited sediment predicted by the model based on the composite grain size data was also 30% greater than that predicted using discrete grain size data.

Perspective

The examples presented above have attempted to demonstrate how new approaches to field investigations of sediment delivery systems can provide an improved understanding of the functioning of the system and provide a basis for the development of more realistic models and for verifying and testing these models. There is clearly a need for closer integration of field measurement and modelling activities if we are to make significant progress in understanding and predicting the behaviour of sediment delivery systems.

A



B

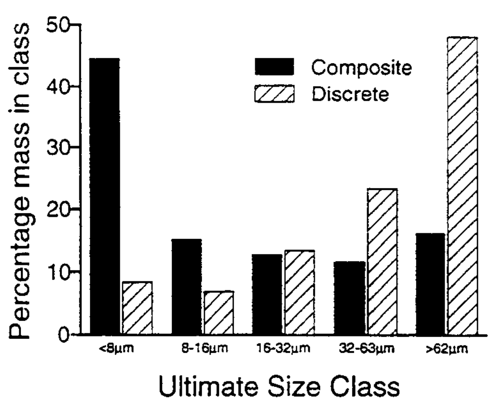


Fig. 4. **Ultimate** size distributions for the sediment in each of five **effective** size classes recovered from the water elutriation apparatus (A) and a comparison of the **ultimate** grain size distribution of sediment deposited on the floodplain predicted by the numerical model using the mean measured **effective** size distribution and the **ultimate** size distribution of transported sediment (B). (Based on Nicholas and Walling, 1996 /18/).

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The role of experiments in soil erosion research: some problems and prospects

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Introduction

Erosion modelling and prediction was for a long time dominated by the USLE (Universal Soil Loss Equation), the development of which started in the USA in the 1930s. The empirical, statistically based USLE is undoubtedly the most successful model in soil erosion research and prediction. Its appeal is mainly based on its simple structure which allows its application in situations where data are relatively scarce and its relatively good performance. Refinement of the USLE still continues /1/.

Despite this success, some shortcomings of this type of erosion model were already noted in 1958 by Wischmeier and Smith /2/ who suggested that there were interactions between the various multiplication factors involved in the equation. Other disadvantages of the USLE are: /3/ (1) the impossibility of predicting deposition (2) the impossibility of making accurate predictions on an event basis (3) the impossibility of predicting size selectivity during the erosion and transport process, (4) the impossibility of evaluating effects on soil erosion on surface water quality and (5) its limited adaptability to complex slope forms and two-dimensional landscapes. Meyer and Wischmeier /4/ and Foster and Meyer /5/ were among the first to propose that new erosion models should be based on a physically-based description of the processes operating. Such models can be made dynamic and spatially distributed so that the disadvantages of empirical, statistical models can be dealt with.

The work by Foster and Meyer and others caused a major shift in research orientation: if process-based models were to be developed into application tools, more information about the mechanics of erosion processes was necessary. As a consequence, the number of experimental studies of soil erosion processes in the field and in the laboratory, both by agronomists and geomorphologists, increased strongly. The process-oriented "paradigm" is still dominant and the belief that, ultimately, process-based models will prove to be a major step forward for erosion prediction, still holds. In 1990, G.R. Foster wrote: "No major improvements in prediction technology are likely to come from the USLE or similar empirically based technology. Major improvements are much more likely to originate from erosion prediction technology based on fundamental hydrologic and erosion processes" /6/.

In this paper an attempt is made to evaluate to what extent the goal set by this 'paradigm' has been fulfilled. The examples presented relate mostly to the author's own research on rill initiation and development. However, similar results could also be presented for other erosion processes. Also, the references cited are not an exhaustive overview, but merely examples. The major question to be asked is: has the experimental study of erosion processes led to process-based soil erosion models which allow improved predictions of erosion and/or deposition or will it do so in the foreseeable future?

Rill initiation: some experimental results of the last decade

Most present-day soil erosion models make a fundamental distinction between interrill and rill erosion. The question where a rill starts has also a more general geomorphic significance. A rill can be considered as an embryonic channel: Therefore, if one wants to understand how landscapes become incised, it must also be understood how rills form. Horton /7/ already developed the first model for the prediction of rill initiation. He postulated that rills will form when a critical shear stress or critical shear velocity is exceeded and used the Manning formula to relate the critical slope length and gradient for incision to the critical shear stress.

Rill erosion may then be modelled using an excess shear expression /8/

$$D_r = K_r (\tau - \tau_{cr}) \qquad (1)$$

where

D_r = the detachment capacity at the rill bed (mass length^{-2} time^{-1})

K_r = the soil's detachability

although other model formulations are certainly possible.

If the Horton model is accepted, the basic problem is how the value of the critical shear stress or shear velocity which initiates rilling can be determined. It can then be hypothesised that rill erosion can only start when (1) the transporting capacity of the flow is sufficiently high and (2) the erosion and transport by the flow is non-selective, so that the development of an armour layer is prevented /9/:

Laboratory experiments showed that transport of silty loamy materials becomes non-selective once a shear velocity of 0.03 ms^{-1} is exceeded/9/. Similarly, it could be shown that the transporting capacity of overland flow shows a sharp rise when the shear velocity exceeds ca. 0.03 m s^{-1}. This shear velocity value may therefore be a minimum threshold value for the initiation of rills on loamy soils. A first experimental validation of this proposal was provided by experiments carried out by Rauws /10/.

Later it was realised that this criterion could only be considered as a minimum threshold for rilling. Indeed, two important aspects of a normal, tilled surface were ignored: (1) agricultural soils are often cohesive, i.e. soil particles adhere to each other and do not behave as isolated grains and (2) a soil surface is often irregular.

The fact that a soil surface is rough has important implications for the erosive power of the water flowing over it. If a surface is irregular, the flow will be slowed down and its depth will increase. This automatically implies an increase of the flow's shear stress as well as of its shear velocity. Therefore, a procedure was developed to calculate the grain shear stress, i.e. the part of the shear stress which actually can be used to detach and transport sediment /11/. Later, rill initiation could successfully be related to the grain shear velocity /12/.

For the prediction of rill initiation, it was also necessary to find a useful measure of the resistance to erosion of a cohesive soil surface. Experiments, where both the soil mechanical strength and the hydraulic conditions were monitored at the moment when the first flow incision started, showed that the critical (grain) shear velocity for rill initiation could be linked to the vane shear strength of the soil surface /12, 13, 14/ (Fig. 1).

These relationships cannot be implemented in a model unless a methodology is developed (1) to estimate the grain roughness of a cohesive soil and (2) to predict the hydraulic characteristics of overland flow velocity on an irregular surface. The median diameter of the water-stable aggregate size distribution of the soil can be used to calculate the grain roughness and thus the grain shear stress /12/. Furthermore, several studies were conducted on the relationship between hydraulic roughness and measurable roughness characteristics of irregular soil surfaces or rills /15, 16/.

From the above, it may be concluded that the necessary methodology to predict rill initiation on agricultural soils has been developed in the course of the last decade, although it also has to be admitted that proper validation of some of the relationships proposed has yet to be carried out. One would therefore be tempted to conclude that, if more of such results become available from experimental studies within the years to come, a deterministic erosion model might be developed with a far better performance than the actually used lumped, statistical models.

However, the experimental results obtained not only provide us with a deeper insight into erosion mechanics: there are also findings which suggest that there are problems associated with event-based deterministic erosion modelling which may be of a rather fundamental nature. Below, a number of such problems are discussed and illustrated, more specifically in relation to sediment detachment by overland flow.

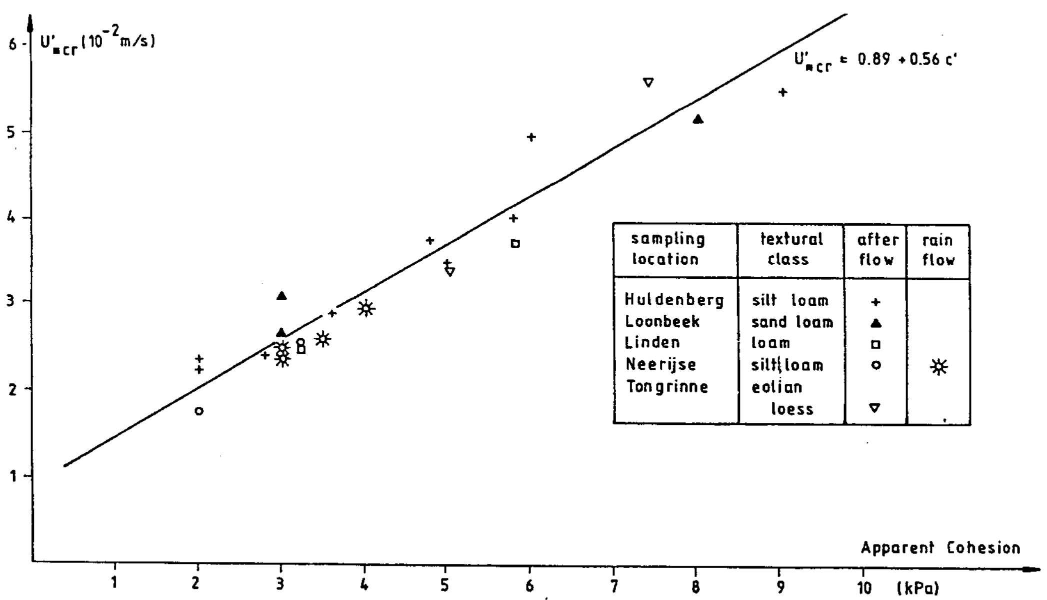

Fig. 1. Relationship between soil shear strength and critical grain shear velocity for rill initiation (after[12])

Spatial and temporal variability of parameter values

In order to operationalize any flow detachment model, information needs to be obtained on the detachability of cohesive soil materials (' K_r ' in equation (1)). As deterministic models are supposed to operate on an event basis, this information should also be available on an event basis.

The detachability of cohesive materials was studied experimentally in a large flume /17/. Figure 2 shows one of the main results of this study: the corrected runoff erosion resistance of the silt loam (as derived from the sediment concentration measurements at the downslope end of the flume) appeared to be strongly dependent on its initial moisture content. In an initially dry condition, the soil is at least an order of magnitude more erodible compared to a soil having a moisture content near field capacity. Possible mechanisms explaining this finding are slaking effects and structure disruption due to the swelling of the clay fraction.

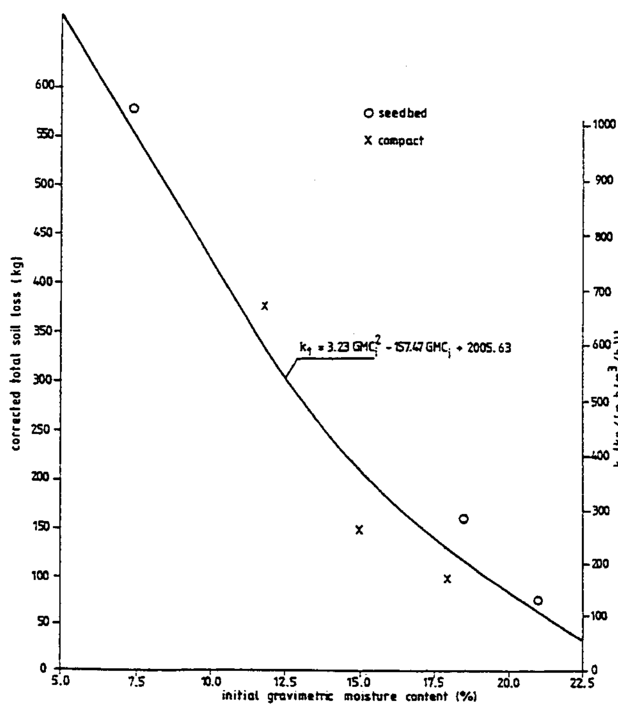

Fig. 2. Corrected total soil loss vs. initial moisture content of the soil (after[17])

This finding implies that the runoff erosion resistance should be treated as a dynamic factor in process-based event models. The strong sensitivity of the erosion resistance of a cohesive soil to its initial moisture content means that, if an event-based model is to be successful, high-quality input data on the soil moisture content of the upper centimetres of a soil before each erosion event is needed. Such data cannot be directly measured on a routine basis as this would be prohibitively laborious and expensive. Therefore they must be provided by a hydrological model. It can be questioned whether present-day hydrological models are capable of providing these data with a sufficient level of accuracy as the modelling of moisture contents and fluxes in the upper centimetres of soil requires precise information on the values of soil physical parameters near the surface. These vary continuously in time due to processes like tillage, consolidation, crusting and bioturbation. So, even if a perfect model available to predict the detachability of a cohesive material, the applicability would be seriously hampered by the lack of sufficiently accurate input data.

The problem is not limited to the assessment of the initial soil moisture content. Over 20 soil properties were reported to affect a cohesive soil's resistance to water erosion /18/. Some of these factors are complex and difficult to measure: it is clear that, at least in the near future, we shall not be able to measure or predict these factors with a sufficient spatial and temporal resolution.

A problem which further complicates matters is the fact that erosion processes appear to be associated with a high level of spatial variability. An overview of various data shows that experiments and field measurements of soil loss typically show coefficients of variation of 15-40%, independent of the scale of measurement /19/.

No measured factor allowed an explanation of this variation which is not systematic in time. With the present state of knowledge it has to be considered as random. This poses severe problems with respect to model calibration and validation: it means that long measurement campaigns on replicate plots or parcels are necessary to determine adequate average parameter values. However, it is well known that a number of relevant parameters (e.g. runoff detachability) also display a high temporal variability: this would mean that a very great number of replicate measurements is needed to establish an adequate average value for a single event. Of course, this implies that we shall only be able to predict a spatially or temporally averaged output as well as the possible margin of error. Indeed, if the unexplained variability of the input parameter values for the model is high, there is no possibility of making accurate predictions for a single hillslope within a single storm.

The lack of deterministic submodels

With respect to many parameters in so-called deterministic erosion models, there is no fundamental model available to predict or calculate them from basic physical and chemical soil processes. In the latest version of the WEPP-model the following equation is proposed to predict the runoff detachability of a cohesive soil /20/:

$$K_{radj}=K_{rb}(CK_{rbr})(CK_{rdr})(CK_{rlr})(CK_{rsc})(CK_{rft}) \quad (2)$$

where:

K_{rb} = the baseline erodibility parameter of a soil ($s\ m^{-1}$),

which is calculated as:

$$K_{rb}=0.00197+0.03vfs+0.03863e^{-184orgmat} \quad (3)$$

where:

vfs = the fraction of very fine sand in the surface soil
$orgmat$ = the fraction of organic matter in the soil.

The equation above is only used for soils containing more than 30% sand. For soils containing less than 30% sand a similar equation relating the baseline erodibility to the clay content is used. The other terms in equation (2) are adjustment factors of K_r for residue, roots, sealing and crusting and freeze and thaw respectively. As for the base erodibility, their values can be obtained from empirical regression equations.

Most of these equations do not the result from a fundamental analysis of the physical and chemical factors governing detachment; they are derived from a regression analysis of a set of experimental data on runoff erodibility collected throughout the USA. This situation is certainly not unique for the WEPP-model: other models, like EUROSEM /21/, rely on similar parameter estimation procedures.

These submodels have the disadvantages of any regression model: extrapolation of the model to soils for which it was not tested is dangerous and the model does not offer any fundamental insights in the processes involved. It may therefore be stated that present-day erosion modelling is not a solution of the problems associated with statistical, lumped models. Instead of the disadvantages of a lumped analysis being eliminated, they are now no longer explicitly present in the model structure, but hidden in the many procedures for parameter estimation. At best, our present models have a deterministic look: the problems of lumped models have not been solved.

Validation of a spatially distributed, multi-parameter model

The greater detail with which present-day models describe erosion processes inevitably causes the number of parameters to be estimated to multiply. Furthermore, these models are spatially distributed, so that parameter values must be provided for each node or grid point. This leaves the modeller with an enormous choice of input parameter values as well as of their spatial and temporal distribution. In an hydrological application, it was clearly demonstrated that is often possible to generate very similar model output with quite different combinations of input parameters /22/. There is no reason to assume that this should not hold for erosion models as well: as experimental research has shown that problems of spatial and temporal variability are probably at least as important with respect to erosion modelling. Philip /23/ already stated the problem in a general way: "The process of inferring the spatial (or sometimes temporal) variation of the parameters of the system from the output runs into the problem that any one of a large (even an infinite) number of assumed modes of variation may yield approximately the same output...the investigator who believes in the physical reality of the parametric values he infers does so at this own peril".

Analogous to Grayson and Moore, one could therefore state that 'There appear to be some major problems in the modelling of surface hydrology (or soil erosion) by so-called distributed-parameter, physically based models'. It is important to understand that these problems are not due to some kind of fundamentally unpredictable behaviour in the system: although under some circumstances geomorphic systems may show genuine chaotic behaviour the general belief is that such systems are in principle deterministic and predictable. The problems are not caused by genuine chaos, but by our inability to cope with the enormous input demands of such models and the spatial and temporal variability of the system to be modelled.

Conclusions

Although experiments have been a very successful tool of research over the last decades, it is also clear that the results obtained did not fulfil the initial goal completely. Our insight in erosion processes has considerably increased but experimental research has also identified a number of problems which appear difficult to solve without an enormous investment of time and money. This necessitates a reflection about the role of field and laboratory experiments and their relation to soil erosion modelling and some criteria for efficient and relevant experimentation may be put forward:

- Experiments should concentrate on those areas where critical knowledge about processes is lacking. Although this may seem obvious, experiments do not always meet this criterion. Detailed studies of grain velocities in overland flow /24/ do allow to gain a better insight in the hydraulics of overland flow and the mechanics of grain movement, but the relevance of such results, in terms of increased capability of predicting erosion processes is limited. In the field, far too many parameters remain unknown, in order to be able to apply this kind of detailed process knowledge. On the other hand, the estimation of the sediment delivery to streams, which is crucial if one wants to estimate the off-site effects of soil erosion, requires information on both erosion and deposition. Although the importance of the deposition process was already stressed some time ago /25/, detailed sedimentation studies are almost non-existent and current deposition and sediment delivery routines in erosion models remain largely untested /26/. It follows from this that problems like the delivery of nutrients and pesticides to streams by erosion cannot be adequately dealt with at the moment. Similarly, tillage erosion has received virtually no attention. Nevertheless it is clear from recent experimental data as well as from the modelling of long-term sediment budgets that soil movement on sloping arable land in Western Europe is probably often dominated by this process (Fig.3) /27/.

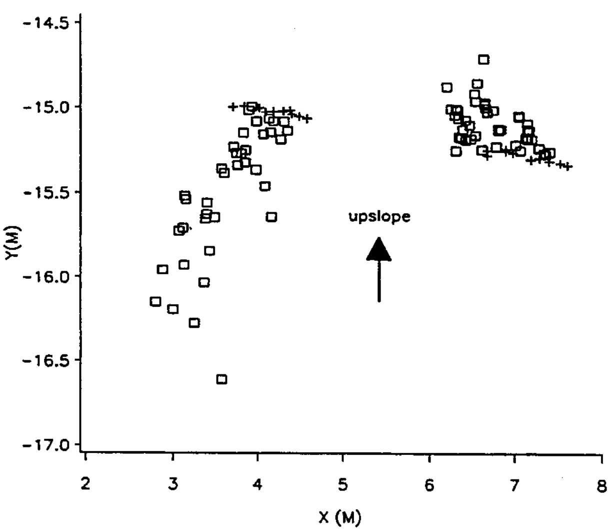

Fig. 3. Movement of soil material by tillage as indicated by tracers on a 0.2 slope: the crosses indicate tracer positions before tillage, while the squares indicate displaced tracers. A single moldboard plough tillage operation was carried out in downslope direction between 2 and 5 m and in the upslope direction between 5 and 8 m. Note that the tracers moved considerably further in the downslope direction. This leads to a net soil loss from the upslope field area (after[27]).

- Experimenters should give more attention to problems of spatial and temporal variability. Hydrologists have, over recent years, carried out several systematic studies on the spatial variability of variables like the saturated hydraulic conductivity of a soil. Similar studies with respect to variables controlling the erosion process (e.g. hydraulic roughness, runoff erosion resistance...) are almost totally lacking. Nevertheless this information is necessary if predictions based on dynamic, spatially distributed models need to evaluated.

- Field and laboratory experiments should be designed so that their results can be incorporated in the model structure which is going to be developed or used. This requires that it must be possible to measure the parameters used to describe experimental relationships in the field with a reasonable effort of time and money. Here, a problem will arise as the thorough scientific understanding of erosion processes may require the study of parameters which can only be measured in a laboratory situation using sophisticated equipment. In such situations additional research will be needed to develop 'transfer functions' to relate these parameters to parameters which can be measured in field situations.

- The modelling technology should be adapted to the problem under study. Many problems, e.g. the identification of long-term net sediment sources and sinks in a landscape, do not require the application of a fully dynamic model. For such problems much more simple steady-state approaches may be used, describing sediment movement through the landscape in terms of the key variables involved /28, 29/. The latter have the advantage that they are much more easy to implement and validate as far less parameters are involved This type of model was successfully applied to evaluate the relative contribution of water erosion and tillage erosion to soil redistribution on agricultural land /30/. The study of the spatially distributed erosion/ sedimentation result of a simple erosion model also revealed that it is in many cases more important to apply appropriate routing techniques than to further refine process descriptions in models in order to obtain acceptable results /31/.

Even if these considerations are taken into account, accurate event-based soil erosion modelling may still be out of reach for the near future: however, by carrying out well-designed experiments and by using appropriate modelling technology we will at least be able to understand better why our predictions are associated with such an important uncertainty and on what temporal and spatial scales more reliable predictions may be obtained. We may then also be able to devise strategies to cope with these problems, at least to some extent. Although this may not be the result which was expected when experimental research was initiated it is probably at least as important.

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Surface runoff, erosion and loss of sediment and phosphorus - Danish plot studies

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Introduction

Rain and melt water running of the surface of agricultural land erode the surface soil. This so-called water erosion carries soil and phosphorus (P) to the aquatic environment. The soil amount lost may be insignificant to the farmers in the short term /1/. However, the P and perhaps also pesticides attached to the fine eroded material may pose an environmental problem. Lots of effort and money have been spent on reducing P outlet from point sources to the Danish water bodies by improving the treatment of municipal and industrial waste water. The effect of these measures may be endangered by diffuse contributions from the agricultural land. To obtain a high quality in the aquatic environment, P losses caused by water erosion have to be controlled.

Few studies of water erosion and P losses have been made in Denmark, so our present knowledge of the magnitude, frequency and rate of these processes is limited.

Material and methods

In the autumn of 1989 eight erosion-study plots (22.1 x 3.0 m) of the Wischmeier type /2/ were installed at Foulum and Ødum on soils containing 9 and 11 % clay respectively and both having about 10% of slope. A detailed description of these experiments can be seen in Schjønning et al. /3/. Surface runoff, erosion and losses of P and N from six different cropping systems were followed for three years from 1989 to 1992. A tractor was used for all tillage operations. The trials were interrupted for one year (1992-1993) where all plots, except the GRS-plot were grown with a winter rape crop.

When the trials were started again in 1993 the layout was changed to that of Fig. 1. The number of plots was increased from eight to nine and the width of the plots reduced to 2.6 m. One treatment, CCR (spring barley undersown with a catch crop of rye grass) was left out and two new treatments were introduced, i.e. STB (spring barley leaving the stubble untreated during winter) and HRW (spring barley followed by a stubble cultivation after harvest). Instead of a normal tractor, a special tool carrier was introduced for the tillage of the plots. Only the sowing and ploughing is now done by tractor, which have reduced the wheeling in the plots to a large degree.
Soil moisture content is measured in four of the Foulum plots by time domain reflectometry (TDR). Vertical probes in 0-20 cm and 0-50 cm depth and horizontal probes 20 cm below the soil surface are used.

Results and discussion

During the first three years (1989 to 1992) winters were very mild and almost without snow. Only one minor incident of snow melt occurred in February 1991. However, there were several instances of heavy rainfall. Particularly the autumn of 1990 was very wet causing large and early surface runoff. That winter rills were present in arable fields all over the country in winter wheat fields. Schjønning et al. /3/ have given a detailed presentation of the results from 1989-92.

The first real winter with snow and hard frost since the eighties occurred in 1993-94. The average temperature of February was between -1 and -2 °C.

Plot no.	9	8	7	6	5	4	3	2	1
Treatment	WAC	WUD	PLG	WUD	HRW	FLW	STB	HRW	GRS
Crop	Winter wheat	Winter wheat	Spring barley	Winter wheat	Spring barley	Black fallow	Spring barley	Spring barley	Permanent pasture
Ploughing	---------------- Ploughed in the autumn-------------				---------------- Ploughed in the spring ----------				-None-
Winter surface	Winter-wheat drilled across	Winter-wheat drilled up-down	Ploughed up-down	Winter-wheat drilled up-down	Stubble harrowed up-down	Harrowed up-down Weeds removed	Stubble untreated	Stubble harrowed up-down	Grass cut october

Fig. 1. Treatments of plots in Foulum and Ødum since 1993.

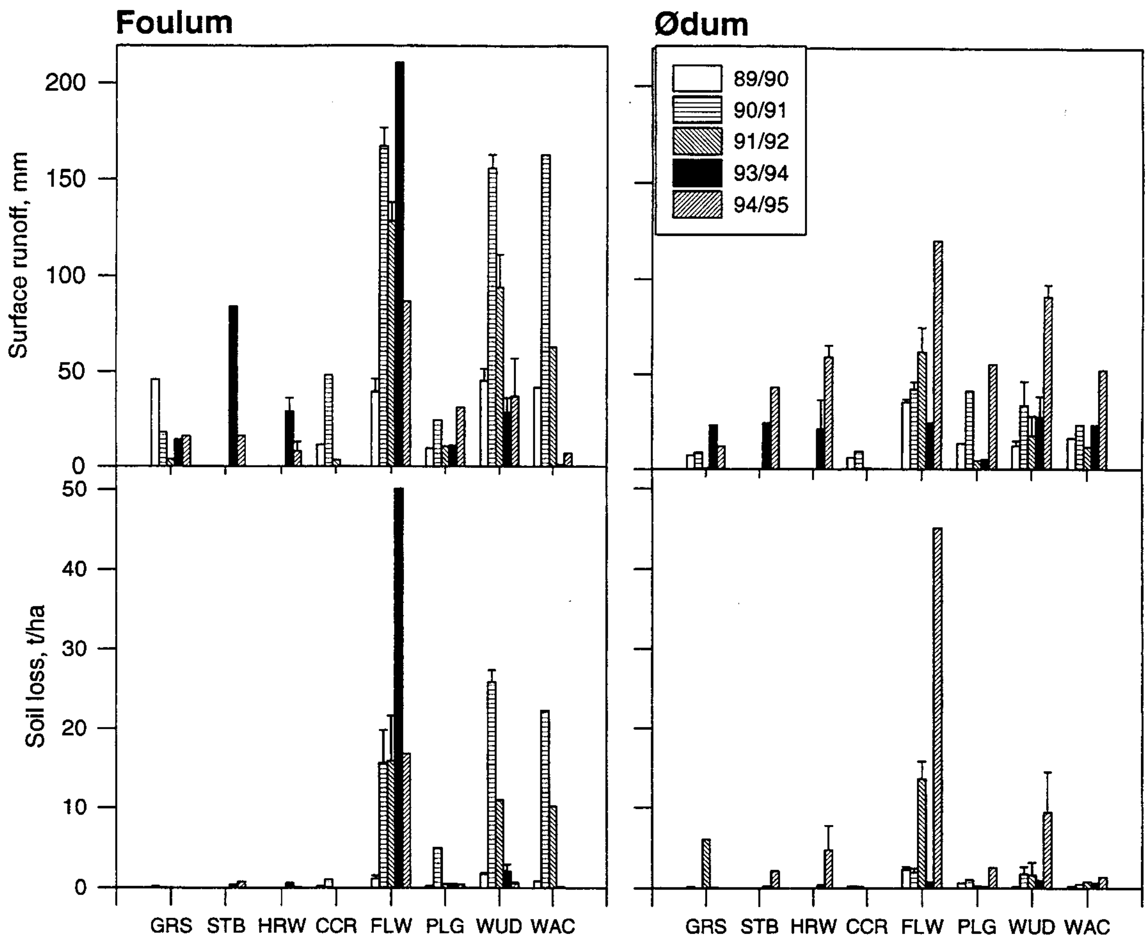

Fig. 2. Surface runoff and soil loss. The catch crop treatment CCR was only present the first three years and the STB and HRW treatments were only present the last two. The error bars indicate maximum values of two replicates. For explanation of treatments, see fig. 1.

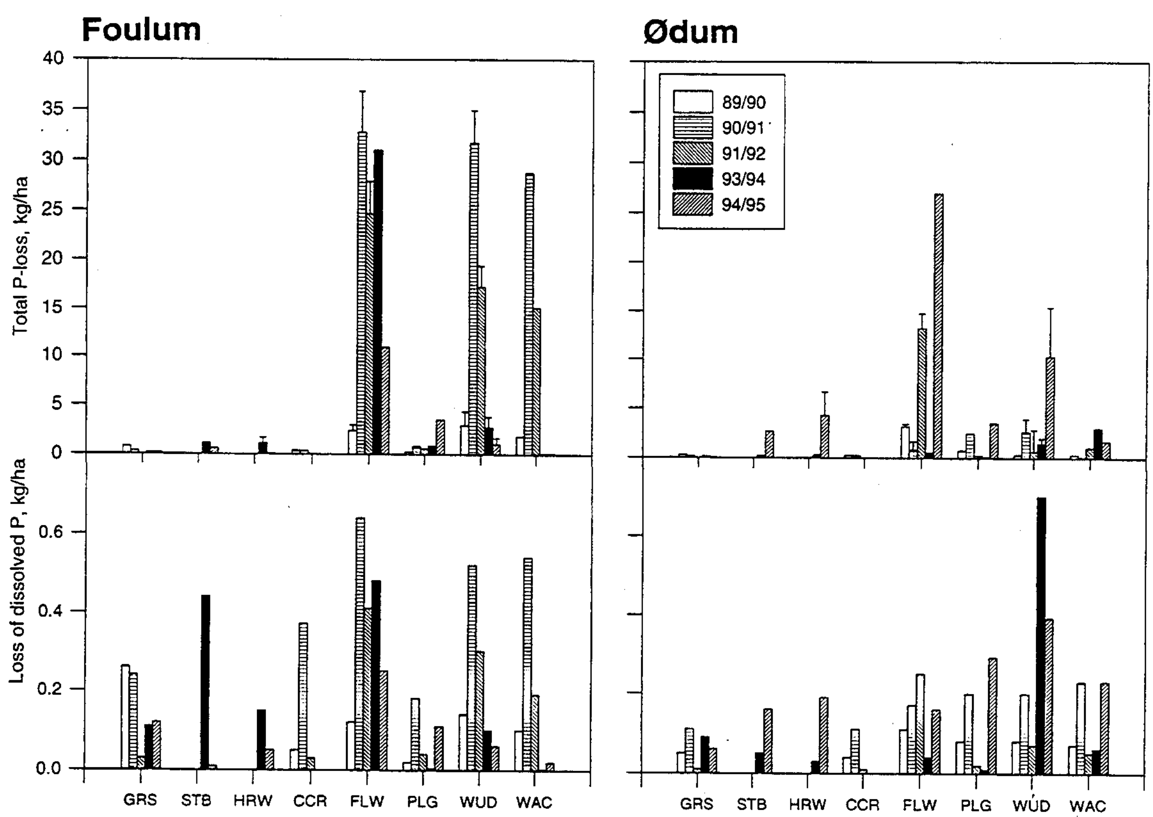

Fig. 3. Loss of phosphorus. The catch crop treatment CCR was only present the first three years whereas the STB and HRW treatments were only present the last two. The error bars indicate maximum values of two replicates. Notice the different scales on the vertical axes. For explanation of treatments, see figure 1.

At the beginning of March a sudden rise in temperature caused a heavy snow melt, when soils were still frozen. When comparing treatments for this year it should be born in mind that the plots were grown with winter rape 1992-93. Therefore the treatments HRW, FLV and STB were not ploughed in the spring as appearing from the plan (Fig. 1). And the only difference between FLW and HRW was, that crop residues were removed prior to the harrowing on FLW but not on HRW.

Plots covered with grass (GRS, CCR) and the ploughed plot (PLG) produced markedly less surface runoff, than those with winter wheat (WUD, WAC) or being fallow (FLW) (Fig. 2). This was probably caused by differences in surface roughness and infiltration capacity /3/. The very high runoff from STB 1993-94 was most likely due to the absence of spring ploughing in 1993. Degradation of soil structure and compaction of the plough layer had been going on since the sowing of winter rape in autumn 1992 which caused a decrease in the ability of the soil to infiltrate and store ponding water on the soil surface. Surface runoff from GRS occurred generally when the soil was frozen. A high capacity for storage of ponding water was probably the reason for the small surface runoff from PLG. The importance of residues is striking, when comparing treatments FLW with HRW 1993-94. Residues had been removed from FLW but not from HRW prior to harrowing.

Generally, the larger the surface runoff, the larger the concentration of sediments in the surface runoff and the loss of sediments. For a given amount of surface runoff, soil loss was least from GRS, CCR and PLG and highest from WUD, WAC and FLW. The larger erosion from the winter wheat plots than from the ploughed plot was probably due to crushing and compaction of the surface soil and the levelling of the soil surface during preparation of the winter-wheat seed bed. This reduced the storage capacity for ponding water and ability to infiltrate water of the winter-wheat soil /3/.

Direction of drilling is the explanation for the large difference between WAC and WUD. In spite that the surface runoff was higher from GRS than from WAC, GRS lost the least amount of sediment. Still, considering the magnitude of the difference and the magnitude of variation between replicates, this difference may not be significant.

Distinct differences regarding surface runoff and erosion were found between years and locations and between treatments. The highest losses of soil were found at Foulum except for 1994-95 where losses were highest at Ødum because of heavy precipitation early autumn 1994. At Foulum the order of sediment losses was FLW > WUD > WAC > PLG > HRW > STB > CCR > GRS. The ranking at Ødum was somewhat different, being FLW > WUD > PLG > WAC > STB > HRW > CCR > GRS.

Total P loss followed the pattern of sediment loss (Fig. 3) simply because the contribution from dissolved P, was small compared to that of particulate P. When ranking treatments according to losses of P the order was FLW > WUD > WAC > PLG > CCR > GRS at Foulum. At Ødum the ranking was similar except that PLG and WAC changed places.

In the treatments GRS and CCR the dissolved P made up a large proportion of the total loss of P, in accordance with the fact that about half the P content of plant litter can be washed out easily /4/. When looking at individual years at Foulum, losses of dissolved P from PLG, WUD, WAC and FLW accounted for 1 to 9% of the total P loss. At Ødum, where soil losses were less, dissolved P made up from 2 to 43% of the total P loss.

If results from 1993-95 are compared with those of 1989-92 for the treatments that were present both periods, generally less sediment was lost in 1993-95 except for GRS. Part of the reason may be the changed traffic in the plots. As mentioned before a tool carrier is now used for most tillage operations, a tractor is used only for sowing and ploughing. Thereby the number of wheelings in the plots has been reduced, i.e. soil conditions are now close to those of normal farming conditions.

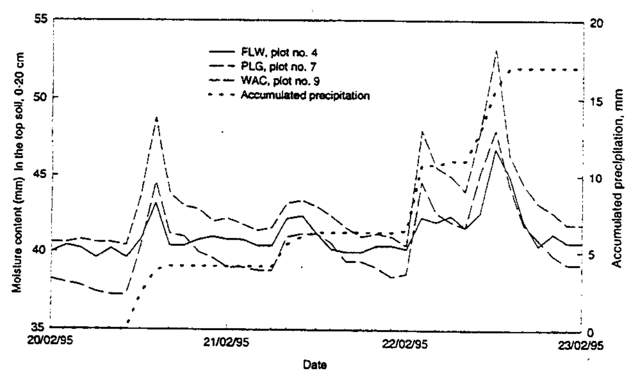

Fig. 4. Accumulated precipitation and moisture content in the top soil (0-20 cm) of 4 plots at Foulum. Moisture content was measured with vertical TDR probes.

An example of the soil-moisture measurements by TDR at Foulum is given in Fig. 4. It shows that the moisture content of the top soil (0-20 cm) increased quickly when it rained and dropped again when the rain stopped. However, the ability of the top soil to absorb water differed between treatments. It was much less in the FLW soil than in the soil of the other treatments, probably due to different degrees of compaction. This may be one of the main reasons for the high surface runoff from the FLW plot (Fig. 2).

Conclusions

Based on the plot experiments conducted in 1989-92 and 1993-95 we conclude that the surface runoff and erosion and loss of P vary with site, year and cultivation. Surface runoff and losses of sediments from the applied treatments generally followed the order: black fallow > wheat drilled up-down > wheat drilled across > barley, ploughed during winter > barley, catch crop during winter > permanent grass. Large erosion was accompanied by large losses of primarily particulate P.

Soil erosion can be significantly reduced by permanent grass as well as a catch crop of rye grass undersown a main crop of spring barley.

The importance of the direction of drilling was underlined. Erosion was much less when winter wheat was drilled along contour lines than when drilled up and down the slope.

Crop residues on the soil surface seemed beneficial for reducing erosion.

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An automatic system for continuous monitoring of rill development, surface runoff and delivered sediments

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Abstract

Formation of rills can cause transport of sediment and associated nutrients and pesticides over longer distances. This project aims at a better understanding of rill formation and evolution under Danish conditions. A system for continuous monitoring of surface runoff erosion and sediment from a plot is presented. Total weight and volume of runoff and sediment are recorded continuously and used for triggering a transport proportional sampling system with a newly developed sampler and a video-recording system. Manual measurements were used for control and for computation of rill volume.

Introduction and aims

Soil erosion has until recently been considered a minor problem in the Northern part of Europe. However, recent studies have demonstrated that in certain areas quite severe erosion takes place /1/. Severe erosion is very often found to be caused by rill erosion /2,3/. In Denmark erosion has been studied within the framework of the NPO (Nitrogen, Phosphorus and Organic matter) Programme. Results from that investigation /4/ confirmed the findings of the authors mentioned above. It was shown that the development of rills could be responsible for long distance transport of both sediment and nutrients and that rills were often found in fields with winter wheat tilled up and down the slope. A new Danish environmental programme (The Danish Environmental Research Programme 1992-1996) aims at a further understanding of the processes leading to eutrofication. The effect of tillage has been studied earlier in plots /5,6/. These investigations are continuing within the new programme. In order to develop and test soil erosion models the investigations have become more processoriented and a better time resolution is achieved in order to validate models and create a better understanding of causes and effects. In nine plots with different crops and tillage typical for Danish conditions /7/ the developed rills are registered after larger erosion events, approximately once a month. One plot with winter wheat drilled up and down the slope is equipped in order to study rill development and related sediment transport in detail. Layout and instrumentation of erosion plots were studied within the STEP Programme and by visiting research stations in different parts of the world. However, none of the methods used were able to fulfil the requirements of this investigation. Besides the manual monitoring of rills it is the aim of this project to construct a system for continuous/semicontinuous registration of the flow of water and sediment from a plot in order to investigate the conditions under which rills are initiated and further developed. The collected data should also be used for testing of erosion models e.g. EUROSEM and erosion models included in the SHE model /8/. The system has to fit into the existing research set up and the measuring routines. Therefore, certain specifications stated below must be fulfilled.

Requirements to the plot installation

1. The system should allow manual registration and collection of sediment after each runoff event as in the other plots. Therefore, a collecting tank should be placed in a measuring cellar.
2. The inlet should be sedimentation free so that all aggreates and grain size fractions released from the plot are carried to the measuring system without delay caused by sedimentation.
3. The ground water level around the measuring cellar should follow the natural level as in the other plots.
4. The volume of water and sediment in the collecting tank should be measured continuously.
5. The weight of water and sediment in the collecting tank should be measured continuously.
6. Flow proportional samples of water and sediment should be collected.
7. The runoff and erosion on the plot during an event should be recorded on a video-recorder.

Description of the recording system
The recording system is shown on Fig. 1 and 2. The description follows the order stated in the above-mentioned listing of the requirements:

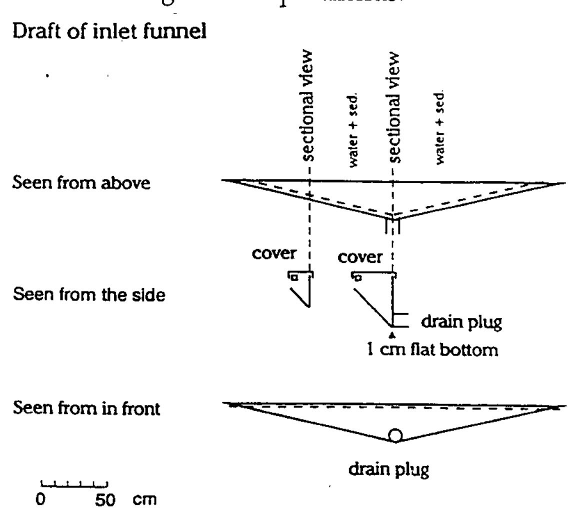

Fig. 1 Sketch of inlet trough.

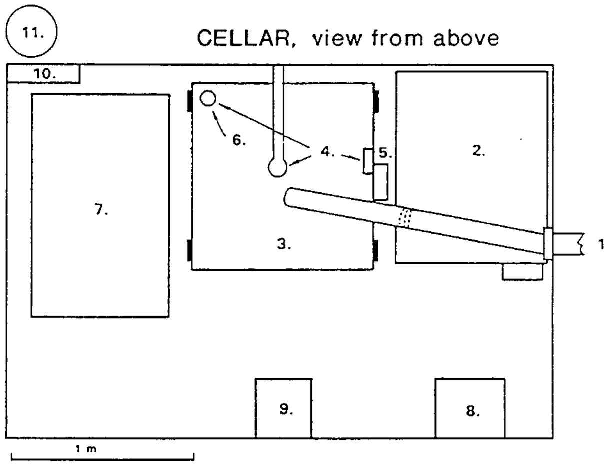

Fig. 2. Layout of instrumentation in cellar: 1. Connection tube, 2. Sampling system, 3. Collecting tank with Thompson weir, 4. Water level sensors, 5. Balance, 6. Sensors for water and air temperature, 7.Overflow tank, 8. Steering system (dataloggers) with housing, 9. Manuals and spare parts, 10. Pump for flood protection, 11. Rainfall recorder.

Collection tank

The dimensions of a single plot is 22.1 x 2.6 m and water and sediment are collected in plastic tanks with a volume of 1.4 m^3 and a depth of 0.9 m. They are emptied before spillover during an event or after major events. The system collecting tank is constructed of 4 mm stainless steel plates welded together in a cube with inner dimensions of 1,000 mm. The bottom area is therefore 1 m^2 so that 1 mm increase in stage corresponds to 1 litre. The cube is strengthened with flanges so that its volume is stable during filling and the tank can be lifted by a crane. If the water level rises above 1 m the surplus water is carried off to an overflow tank via a Thompson weir. The tank is emptied for water by use of an electric pump and the settled sediment is collected manually and dried and weighed as for the other collecting tanks.

Inlet

The inlet (Fig. 1) is constructed of stainless steel. Its sides sloping towards a tube, which leads to the cellar. The inlet is covered by a (steel) plate to prevent rain on the inlet to reach the cellar. All water and sediment leaving the lower edge of the plot will enter the cellar after a few seconds.

Cellar

The cellar is 2.5 m deep, 3 m long and 2 m wide. The bottom of the cellar is placed 2.4 to 2.1 m below the surface in order to obtain sufficient slope from the inlet to the collecting tank, to prevent sedimentation. The natural groundwater normally rises to approx. 1 m below the surface during the winter. This causes a significant boyancy on the cellar. The cellar is kept in place by flanges on the bottom held down by the weight of the overlying soil. The cellar made of steel is constructed watertight, allowing the ground water to rise without having to pump to keep the cellar dry.

Volume of water and sediment:

The idea was to measure volume and weight of water and sediment in the collecting tank continuously. Assuming a density of the sediment of 2.65 t/m^3 it should be possible to compute the volume of water and the amount of sediment accumulated during a timestep by solving the two equations with two unknowns. The total volume is measured by recording the stage of the water in the tank. Two to three independent systems are used for that. A Campbell (England) pressure transducer with a resolution of 1.3 mm, a TDR sensor /9/ with a resolution of 1.4 mm and an ultra-sonic sensor manufactured by Endress and Hauser (Switzerland) with a resolution of 1 mm. Theoretically the accuracy could be improved to 0.1 mm by averaging frequent measurements of the water stage.

Weight of water and sediment

The total weight is measured by an electric balance consisting of three load cells manufactured by Bizerba (Denmark), the max. capacity of the balance is 2 t and the resolution is 0.1 kg. The resolution, however, puts restraints on the minimum length of the timestep depending on the amount of sediment in the water because the computation involves a difference between two measurements.

Sampling water and sediment

Sampling of water and sediment are also applied. Measurements of the actual sediment concentration can be used to calibrate and control the computed concentrations. Furthermore, the sampling system ensures a time resolution if it is not possible to obtain a satisfactory one with the computation procedure. The best way to obtain samples is to be on the spot and collect them manually directly at the outlet from the plot. This procedure is possible in studies using artificial rainfall. If, however, as in the present case, natural conditions are to be studied on remote locations, automation is needed. A survey showed that no available commercial sampler was able to meet the demands of this investigation. Therefore, it was decided to construct a new one. The sampler should be able to collect "in stream" samples without causing sedimentation and cross contamination and it should be able to collect a large number of samples within a short time. External triggering and choice of sampling time should be possible.

The constructed sampler (Fig. 3) operates in the following way: Water and sediment enter the cellar through a tube from the inlet structure. The tube is connected to a funnel which is able to swing back and forth. While not sampling, the funnel is in position above a flume leading to the collecting tank. When sampling is triggered, the funnel swings into a position above a sample bottle placed underneath. The water and sediment then run from the funnel into the 250 ml PVC bottle. After a drip of time of 8 seconds a new bottle moves into position within 20-30 seconds. This allows a sample to be taken every 30-60 seconds. The bottles are supported by a chain that moves on a table. The capacity is 149 bottles.

Steering, videorecording and TDR

In order to monitor the system and to collect flow proportional samples the steering is provided by a programmable Campbell 21X datalogger (England). A scan interval of 30 seconds was chosen to fit the execution time of the programme. Every 10 minutes the average, the max. and the min. of water level and weight are recorded. For each scan the present water level and/or weight are compared with the previous and the difference is computed. If the accumulated difference is larger than a prechosen increment e.g. 7.5 l or 7.5 kg, the datalogger computes an opening time for the sampler that fills the bottles without overflow, and triggers the sampler. The actual time, sampling time, weight and water level are recorded. Simultanously, a modified Canon E60 videorecorder and a car headlight, which lights up the plot, are triggered. A "grey box" overrides the autofocus mechanism and copies date and time into the images. As a security procedure the water level measured by the pressure transducer is recorded independently on a Grant (England) 12 bit datalogger. As an experiment the water level was also recorded by the TDR technique in connection with measurements of soil water. Rainfall and temperature are also recorded on the Grant logger at 5 minutes intervals.

The installation is visited fortnightly in the winter halfyear and monthly during the summer. In connection with these inspections the dimensions and spacing of the rills developed are measured.

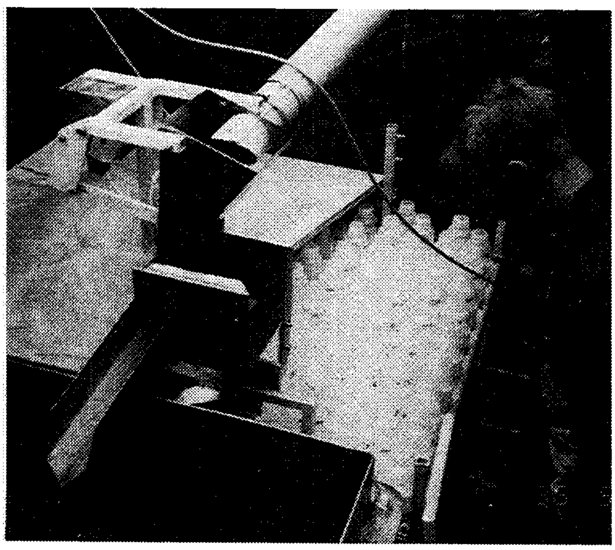

Fig. 3. Sediment sampler.

Experience and preliminary results

The project was ready to start in the autumn of 1993. Tillage and sowing of winter wheat take place in September after which the "erosion season" starts. Due to late arrival of different parts the various installations had to be put into operation as soon as they were constructed so the first field season was therefore partly considered as a pilot study. The cellar and the collecting tank was installed early in October together with the pressure transducer and the first balance and the ultrasonic sensor was installed late in October. The prototype sampler was installed late November. Not surprisingly such a complicated system suffered from a lot of failures that had to be corrected. The first problem was that the cellar did not seem to be watertight, because heavy rainfall caused preferential flow along the outside of the tube connecting the inlet and the cellar. After tightening this problem disappeared. The corregated plastic roof of the cellar caused condensation; this problem was, however, solved by using a plastic folio as a double layer. Together these moisture problems caused erratic readings of the sensitive balance and of the ultrasonic sensor. After installation of a thermostatic heating of the logger box the readings stabilized. The cellar endured a groundwater level about 1.5 m above its floor, however, the welding was not strong enough to prevent minor swelling of the floor. Therefore, the balance had to be placed on iron bars attached to the walls of the cellar. The first balance was not to able to keep the resolution of 0.1 kg as promised by the manufacturer, so a new one was installed in late December. The system functioned well in January and February. At the beginning of March several cases of stochastic sampling occurred, caused by periodic breakdown of the ultrasonic sensor. An accurate measurement of the waterlevel with a resolution of better than 1 mm was not possible with this technique, partly because the plunging water causes oscilations of the surface. The triggering is now steered by the balance which seems more stable.

The total volume of water and sediment recorded from the plot from 1 October 1993 to 20 April 1994 was 2,089 l or 36.4 mm /7/ with the surface runoff constituting about 10% of the precipitation. Twenty surface-runoff events could be distinguished in total. The largest event taking place from 3-5 March 1994 contributed 1190 l or 20.7 mm while the second largest taking place on 18 January contributed 150 l or 2.6 mm. Both events were caused by simultaneous rain and snowmelt. The total amount of sediment was 16,495 kg equal to 2.9 t ha^{-1} and corresponding to a mean concentration of 7,896 mg/l Mean concentration from 4 January to 10 February was 29,449 mg/l and from 10 February to 7 March 3,097 mg/l. The concentration found by sampling during a runoff event on 18 January proved to be much less than the concentration found by emptying the collecting tank. This lead to the conclusion that the construction principle applied in the first version of the sampler /10/ caused particle separation, leading to an under-representation of larger heavier grains.

Based on that the present version was constructed. The first occurrence of rills was recorded on 14 December, two small rills with a total volume of 0.0012 m^3 were formed. The rills grew during the snowmelt periods in January and March. At the end of the season the total rill volume was 0.0045 m^3 or approx. 6.7 kg, assuming a dry soil density of 1.5 t m^{-3}. The estimated rill erosion accounted for 41% of the total amount of sediment eroded from the plot.

Discussion and conclusion

The monitoring system is a prototype which is still not tested to its limits. Many problems have emerged and have been solved. The measurements of water level did not facilitate computation of flow for small timesteps, as the accuracy (±2mm) was not sufficient.

Accumulated weight seems to be more accurate (±0.2kg); these recordings are therefore better suited for steering the sampling and as data values for testing erosion models. Altogether the system gives a good time resolution that is useful for the study of the surface runoff and erosion processes.

The number of rills formed are less than in earlier investigations /6/. Also the eroded volume is less. This was puzzling, because surveys on selected fields showed that the number of rills and their volume were larger in 1993/94 than in previous years. Rills were formed on newly tilled seedbeds during rainstorms in September and early October. As the plot studies started later, these events were not included.

This indicates that early tilling may increase the risk for rill formation due to more frequent rainstorms. On the other hand, the rills on the plot did not grow much during the two snow melt periods, as was the case on surveyed fields. A snow drift covered the lower end of the plot. Runoff from the upper part was filtered through the snow, and meltwater from the drift had to run only a short distance to the inlet. This was clearly revealed on the videorecord.

It may therefore be concluded that the results from this year underestimate the amount of rill erosion.

Acknowledgement

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Water erosion on cultivated areas - Field monitoring of rill erosion, sedimentation and sediment transport to surface waters

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Introduction

Soil erosion due to surface runoff is an environmental problem throughout parts of northwestern Europe. Despite moderate elevations and mild climates, rills and even gullies can be found in Denmark on fields having long and gentle slopes, e.g. in dry valley bottoms (thalwegs), in wheel tracks, and between rows of drilled crops. Although loss of soil can be substantial, the main concern here is the loss of phosphorus to the aquatic environment accompanying erosion.

Erosion studies in plots give insight into the detailed mechanism of erosion and are valuable for comparing the effects of different soil management practices /e.g. 1/. However, findings from plot studies are difficult to upscale to whole fields and catchments. There the overall topography and linear features, such as dead furrows and wheel tracks, play a significant role for formation and concentration of surface runoff and the resulting erosion and sedimentation.

Earlier erosion studies have shown that mainly the three following situations may cause soil erosion by surface runoff:

1. A short, but very intensive rainfall, e.g. a thunder storm, which exceeds the infiltration capacity of the soil /1, 2, 3/.
2. Rain of low intensity but long duration, which saturates the surface layers whereby the infiltration capacity becomes less than the rain intensity /1, 2, 3/.
3. Thaw events, i.e. melting snow or rain when the soil is frozen and therefore may have a very low infiltration capacity /1, 4/.

This field study attempts to link water erosion and concomitant transport of soil and particulate phosphorus to surface waters by water erosion to topography, geology, soil physical properties, crop and soil management and configuration of riparian zones. The objectives of the study are:

- to elucidate and quantify water erosion and sedimentation on Danish arable land.
- to create an expert system suitable for predicting erosion.
- to estimate the erosional transport of soil and particulate phosphorus to the aquatic environment.
- to create a data base for validation of soil erosion models on a catchment scale level.

Study areas, slope units and measurements

Ten agricultural study areas, bordering on to streams or lakes, were selected throughout Denmark representing a range of topographical and geological zones where erosion could be expected and covering a variety of climatic conditions, surface management conditions, vegetations and riparian zones. Each study area was divided into smaller units, called slope units. A slope unit is a small independent discharging area as regards surface runoff. It has the size of a field or less and is separated from other slope units by permanent barriers, e.g. roads, ditches and water divides and/or by impermanent barriers such as dead furrows and crops. The presence of impermanent barriers implies that the total number of slope units changes from year to year. The selected slope units are not representative of Danish agricultural areas in general.

In the late autumn when fields are left for the winter and again in the early spring prior to tillage, erosion is estimated by measuring the size and number of rills. Losses of sediment to water courses are estimated by subtracting the volumes of the outwash fans from the measured rill volumes. At the same time the following physical parameters are measured:

- water infiltration from tension infiltrometer (8 cm diameter) near saturation
- penetration resistance (hand held penetrometer),
- *in situ* bulk soil shear strength at water saturation (hand held shear vane),
- index of surface roughness (nylon string with small lead pellets attached),
- bulk density, volumetric water content, air permeability, porosity, and water field capacity of the plough layer,
- texture and total-P of sediments.

Once during the project period texture, pH, organic matter content and total-P content are measured in the plough layer soil. The soils will be classified according to the U.S.D.A classification system and examinations made of impermeable layers like e.g. plough pans. Management of the fields are registered, regarding crops and fertilization and so are the orientation of the various linear features originating from tillage operations.

Results and discussion

Soil properties

Soil physical properties affecting the extent of water erosion may, in simplified form, be grouped into three different categories:
- the infiltration capacity,
- the water storage capacity of the soil surface, due to surface roughness,
- the soil shear strength.

Conditions supposed to be affecting these soil properties and thereby the extent of water erosion are texture, soil management and time of year. For the time being we have grouped the different management practices considered into the following four categories:

1. Autumn tillage:
   Soil ploughed, stubble cultivated or similar in the autumn after harvest.
2. Winter crop cover:
   Soil covered by an autumn sown crop during the winter period, e.g. winter wheat.
3. No autumn tillage:
   No soil tillage in the autumn and no crop cover during the winter, e.g. untreated stubble.
4. Grass:
   Grass cover during the winter period (only three soils).

Analyses of variance gave the following significant differences:

Water infiltration from a tension infiltrometer near saturation seemed to be influenced strongly by the soil texture together with tillage and crop cover during the winter period (Fig. 1), while the time of the year seemed to be of minor importance (data not shown). The infiltration generally decreased with increasing clay content. Moreover, the ranking of the infiltration with respect to soil management was: No autumn tillage >> Winter crop cover > Autumn tillage.

Soil surface roughness, which governs the surface-water storage capacity, was affected by soil management and time of the year (data not shown). The order of surface roughness relative to soil management was: Autumn tillage > No autumn tillage > Winter crop cover. Surface roughness was largest in the autumn and decreased during the winter.

The soil shear strength generally increased with clay content (Fig. 2), but was also affected by soil management and time of the year (data not shown).

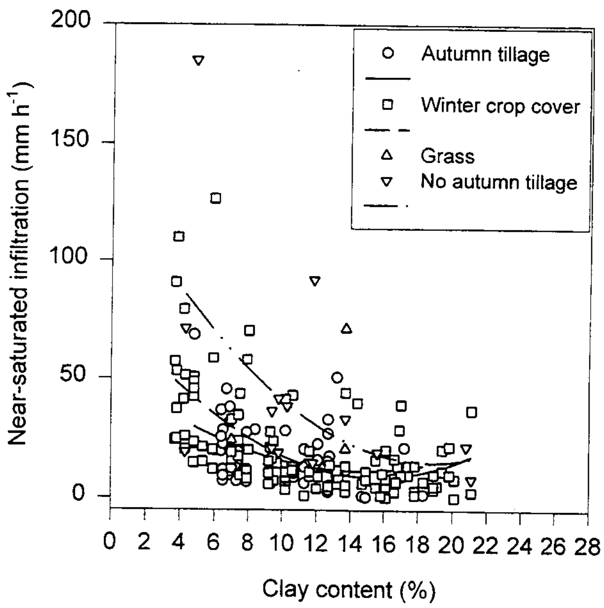

Fig. 1. Infiltration from a tension infiltrometer as a function of texture of plough-layer soil. Measuring position: up-slope. Data from spring and autumn of 1994 and spring 1995.

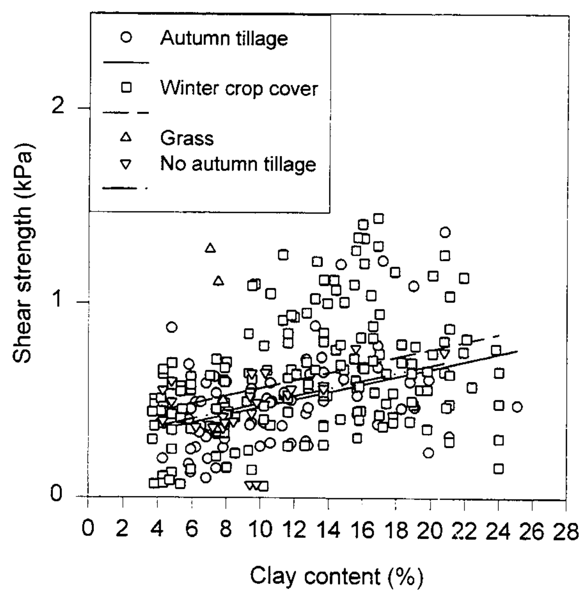

Fig. 2. Shear strength related to soil texture of plough-layer soil. Measuring position: mid-slope. Data from spring and autumn of 1994 and spring 1995.

Soils with Winter crop cover had a higher shear strength than both Autumn tillage and No autumn tillage soils. The shear strength increased during winter for fields with autumn tillage and winter crop cover.

The content of soil organic matter of the experimental areas ranged from about 1% to 5%, but did not show any significant effect on the soil physical properties discussed here.

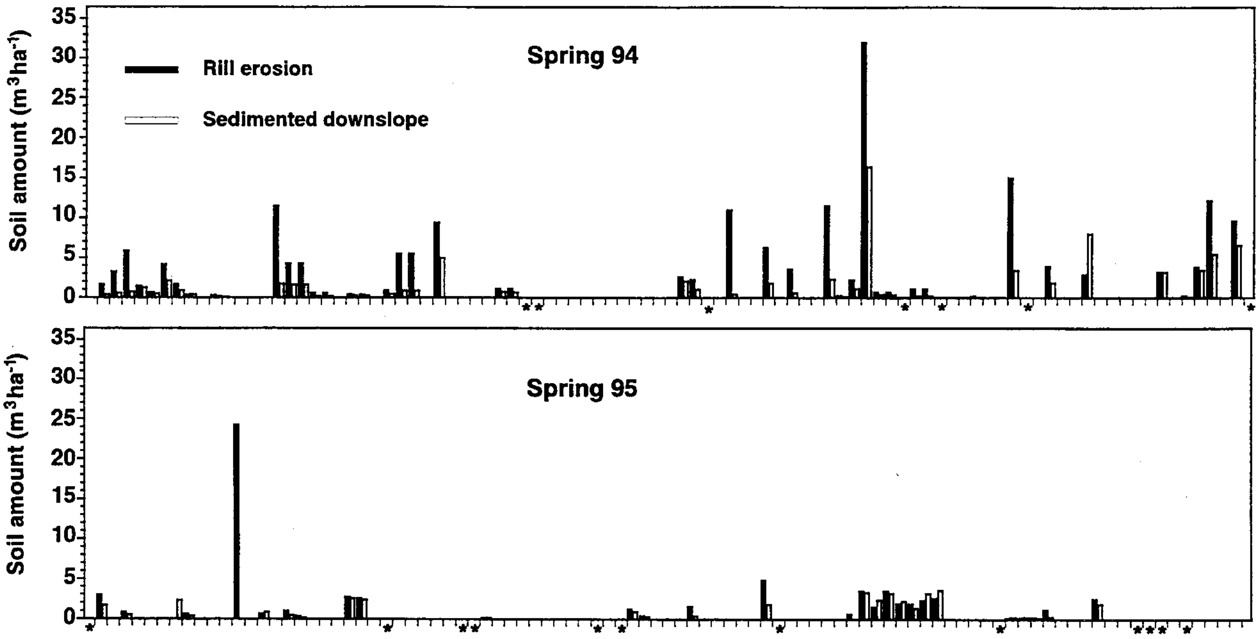

Fig. 3. Estimated rill erosion and down-slope sedimentation on 95 slope units (same position on x-axis both years) of which 7 were not included in spring '94 and 12 not included in spring '95 (marked with stars). No bars indicate no erosion or sedimentation.

Soil erosion and sedimentation results

Rill erosion was greater and more widespread during the winter season of 1993-94 than of 1994-95 (Fig. 3) but varied tremendeously between slope units both seasons. The fraction sedimented on land and the fraction lost to surface waters also varied much. In a few cases the estimated sedimentation exceeded the estimated rill erosion. The reason could be a simultaneous sheet erosion, which was not measured, or more likely a measuring error. Of the slope-units examined in spring 1994, 63% had rills to an extent that enabled estimates of soil loss, 7 % had insignificant erosion and 30% no erosion. In the spring 1995 40% had significant erosion rills, 5% had insignificant erosion rills and 45% no erosion. There was no correlation between the two winters as regards erosion on the various slope units.

The greater erosion during 1993-94 than 1994-95 was mainly caused by several thaw events during 1993-94, the first winter for several years with significant thaw events. Only a few thaw events occurred during 1994-95.

The relations between rill erosion and sedimentation and soil properties and landscape features have not yet been fully analyzed. However, the majority of the soil erosion events was observed on slope units covered with autumn sown crops or tilled in the autumn. Least erosion was observed on slope units not tilled in the autumn or covered with catch crops or grass. Clayey soils with low infiltration capacity or fine sandy soils with very low shear strength seem to be the Danish soil types most exposed to soil erosion, even though the infiltration capacity of fine sandy soils and the shear strength of clayey soils usually are high. Our study has furthermore confirmed the importance of linear features in the landscape such as dead furrows and wheel tracks collecting water and eventually causing erosion /5/. The most serious cases of erosion were all found in dry valley bottoms (thalwegs) /6/.

Potential utilization of the results

The final goal is an expert system that can be used as a predictive tool, pinpointing which combinations of topography, geology, soil type and crop management that may lead to soil erosion. Such an expert system could be used to advise farmers on how to alleviate or avoid soil erosion problems.

Combining data from this soil erosion project with existing data bases of soil parameters, climate data and satellite pictures will facilitate an improved mapping of areas susceptible to soil erosion. The targeting of such areas will encourage the best uses of, for example, limited resources, and the set-aside policy initiated by the European Union.

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Erosion, transport and control of peat sediments from peat mines

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Abstract

This paper presents the main results of research done on peat erosion, transport and control from 1991-1994. Results on settling, transport and erosion processes responsible for peat erosion and delivery are presented. The results obtained are used to identify a conceptual model that is used in control assessment and to further study the erosion and delivery system. The benefit of different control alternatives such as controlled drainage, sedimentation basin, and artificial floodplain is discussed. The final part of the paper focuses on a measurement project started in Central Finland during the spring of 1995 in order to further study the sediment erosion-transport, nutrient delivery, and the best control alternatives.

Introduction

Peatlands constitute a large part of cold regions. They form 33.5% of the land area of Finland and 18.4% of Canada. Peat is formed in areas, where the rate of production of organic material by living organisms exceeds the rate of degradation resulting in organic deposits. Wet peatlands, as bogs and fens, are often drained to reclaim land for forestry and agriculture; some land has also been drained for peat mining. The drainage has resulted in higher runoff peaks, sediment and nutrient yields /1/.

Previous studies on peat soils (e.g., 1-3) emphasize the importance of surface runoff in generating erosion from the soil surface and producing sediment transport. The low hydraulic conductivity and the hydrophobity of peat result in surface runoff even for low intensity storms /4/.

The main environmental problem with peat mining is the erosion of peat and the subsequent transport of sediments into downstream rivers. During heavy storms peat is eroded from the mine surface and the channel bottom /3/. Due to high drainage intensity and steep slopes, intense rain turns into intense runoff. As a result, the eroded peat is transported downstreams. The increase in discharge reduces the sedimentation efficiency in sedimentation ponds traditionally used to control sediment transport from peat mines /5/. Consequently, poor efficiencies have been reported /6-7/.

This article presents results of research done on peat erosion, transport and control during 1991-1994. The study is part of a larger project on peat mine hydrology, hydraulics, and drainage water treatment. The aim is to develop methods that reduce sediment and nutrient transport. The project is described in detail elsewhere /8, 3, 5, 9/.

Methods

The erosion from the mine surface during extreme rainfall was studied by simulating rain on two 10 m by 10 m erosion plots. Simulated rain was allowed to fall for 15 minutes in two trials on plot 1 with a rain intensity of 4.2 and 6.0 mm/min., respectively, making a total of 63 and 90 mm. Plot 2 was wetted for 15 minutes with a 6.0 mm/min. rain. The surface runoff was monitored downstream of the erosion plot 1 with a v-notch weir. From plot 2 the discharge was not monitored. Water samples (250 ml) were taken at 3 minute intervals from surface runoff and the sum of rill and interill erosion was estimated. The contribution of channel erosion was estimated from water samples taken from the v-notch weir immediately downstream of plot 1. Discharge was measured at intervals of 2.5 and 5 minutes. On plot 2 sediment concentration was measured only in surface runoff. During and after the experiments the rill network was registered by visual inspection.

Settling of peat treated in different ways was studied in laboratory with a settling column. First, the settling of base soil peat, which resembles the soil eroded during intense rainfall /10/, was studied in detail. Dry peat was taken from the peat mine surface and wetted for 12 hours. Peat and water were mixed to produce an approximate concentration of 20 or 200 mg/l. Then effect of temperature was studied by settling in a temperature controlled room at temperatures approximately 0 and 250° Celsius. The settling of 0.5 days wetted peat was followed by a study of settling of base-soil peat wetted for 5 days. The characteristics of the wet sediment would be similar to the peat in surface runoff from saturated surfaces. In the third settling experiment, peat sediments dredged from a peat-mine settling tank was settled. This sediment simulated the fine material. The settling column had a height of 1.5 m and volume of 48 l. The concentration was measured versus time at depths of 0.4, 0.8 and 1.20 m. The settling velocity (cm/h) was calculated according to /11/.

Results and discussion

The variation of peat erosion from erosion plots during constant precipitation showed three stages in all experiments (Fig. 1). First, the sediment discharge rate (kg/s) increased as the flow discharge increased. Simultaneously the sediment concentration (mg/l) in the water dropped dramatically. When the surface runoff equaled the precipitation, the transport capacity remained constant, but the erosion rate decreased gradually due to depletion of erodable material. When the rain ceased, the decrease in erosion became more pronounced.

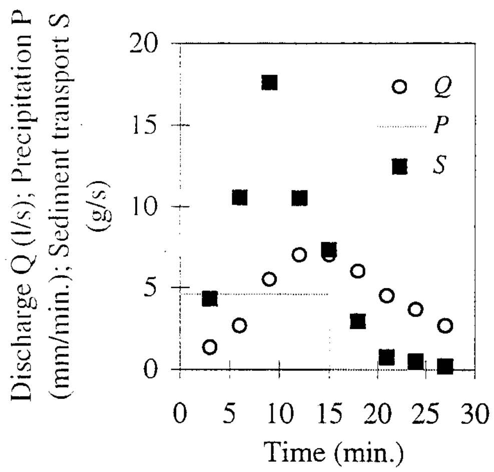

Fig. 1. Observed surface runoff and sediment transport during constant precipitation on plot 1.

The total sediment yields from the erosion plots are shown in Table 1. A considerably higher sediment yield was observed from plot 2 than plot 1 during equal rain intensity of 6.0 mm/min. In part, this was because very loose material had already eroded during the first event on plot 1 when the rills were formed. A second rain simulation did not considerably alter the rill patterns formed during the first rain event. The contribution from the channel was small during the first rainfall on plot 1. However, during the second rainfall the channel contributed a large sediment yield. This was probably due to erosion of deposited material that settled during the previous event.

Plot no: & Rain int. (mm/min.)	Rain during exp (mm)	Rain previous (mm)	Erosion soil surf. (kg)	Erosion ditch (kg)
plot1, 4.2	63	0	4.9	0,2
plot1, 6.0	90	63	2.6	1,4
plot2, 6.0	90	0	9.7	-

Table 1. Sediment yields observed during rainfall simulations on plot 1 and 2.

The effect of temperature on settling of base soil peat deviates from Stoke's law. The average settling velocity and temperature during the first settling hour was 57.9 cm/h in 1.2 C° (replicates n = 6) and 54.6 cm/h in 23.5 C° (n=6) water, and the portion of settled material 72% and 67%, respectively. The results obtained here are similar to those obtained on flocculating particles; a decrease of settling velocity due to repulsive forces as the temperature increases are observed /12/.

The average settling velocity and sediment concentration during the first settling hour was 56.4 cm/h in 12.8 mg/l (n= 6) and 56.1 in 123.6 mg/l (n= 6) water. The effect of concentration on settling was not significant in the range (20-200 mg/l) studied. Similar results have been obtained elsewhere /13/.

The settling experiments showed that the settling of peat depends on peat particle size and the wetness of the peat (Fig. 2). The lowest settling velocity was observed for the fine material dredged from the settling basin. The base soil peat wetted for 0.5 days (n=12) settled considerably slower than the peat wetted for 5 days (n=1).

Sediment control assessment

Model for sediment control assessment

A conceptual model for peat erosion and sediment transport is shown in Figure 3. It computes sediment transport caused by heavy storms on peat mines. It consists of two parts. The upper part is the hydrological part transferring precipitation (P) into surface runoff (Q_s) and erosion (C_s). The lower part is the hydraulic part routing water and sediment through the bed ditch (BD), collector ditch (CD), settling basin (B) and artificial floodplain (FP) and out of the watershed. Settling of peat (S) increases the amount of deposited material (D). The modeled is discussed in detail in /9/.

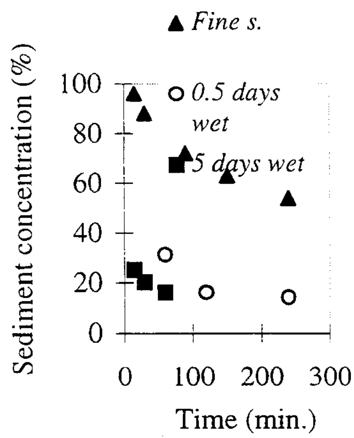

Fig. 2. Relative decrease of sediment concentration in settling column experiments.

Comparison of different water protection alternatives

The model presented in Figure 3 was used to evaluate the relative variation in maximum runoff and total sediment yield from a 0.5 km^2 peat mine caused by a 30 mm rain of 30 minutes duration during different water treatment alternatives. The results are shown in Figure 4. The reference control alternative, simulation number 1, against which all the other control alternatives are compared, consists of a 10 cm diameter pipe at the inlet of each bed ditch, a settling basin and a v-notch weir at the outlet. In simulations 2 and 3, a detention structure is added to the outlet. In simulations 4 - 7 only bed ditch pipes are used for runoff detention. In simulations 8 - 11 detention is achieved by using both bed ditch pipes and sedimentation basin outlet structures. In simulation 12 the settling basin is doubled. In simulations 13 and 14 two floodplains of different dimensions are added to the simulation alternative 8.

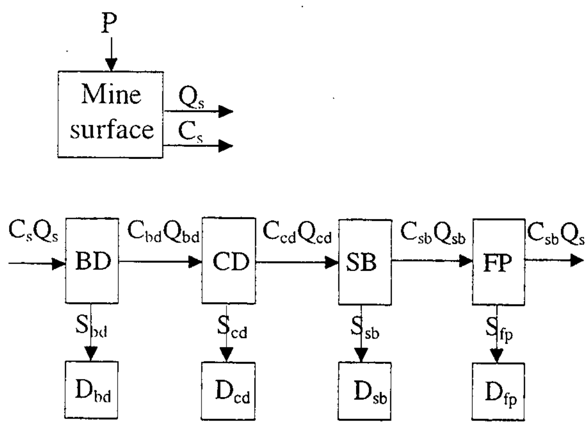

Fig. 3. Flow chart of the sediment transport model.

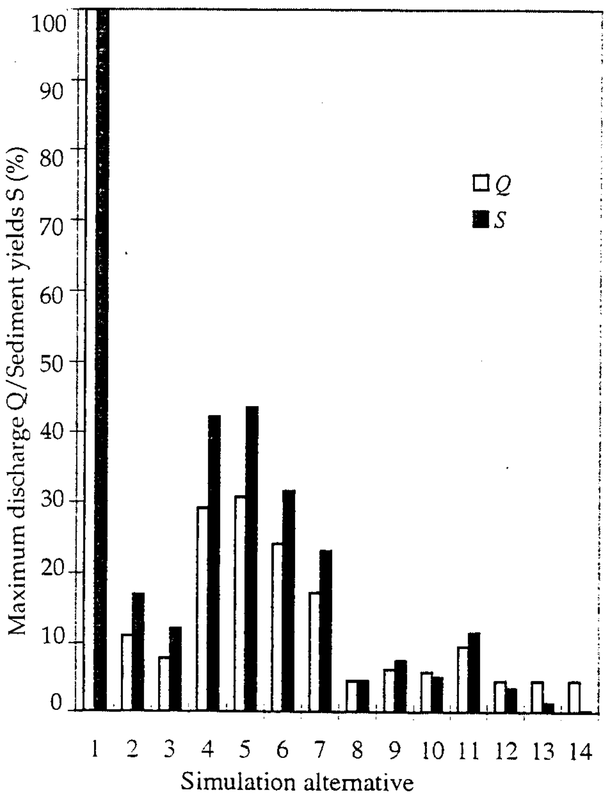

Fig. 4. Simulated maximum flow (white bars) and sediment yield (black bars) caused by alternative detention and settling structures.

The results show that low sediment yield is associated with small runoff peaks; detention is clearly the most important factor controlling the sediment yield. The lowest sediment yield is obtained with simulation alternative number 8 where detention at the drainage area outlet is combined with efficient detention pipes in the bed ditches. The sediment yield is reduced by 95%. The sedimentation ponds have a minor effect on the transport of sediment. Doubling of the settling basin reduces the total sediment yield to 1%. The floodplain reduces the total sediment yield by 3% for the smaller floodplain and 5% for the larger floodplain.

The alternatives 1, 4, 5 and 6 represent currently used sediment control alternatives. The results show that runoff detention instead of sedimentation basins should be the basic control alternative. Sedimentation basins should not be designed to trap suspended material, which they are not able to trap, but designed to trap bed load material and floating peat. If increased settling during storms is required, shallow settling basins such as artificial floodplains /5/ or wetlands /7/ should be built. In shallow settling basins the turbulence is low and settling of small particles possible /5/. Because small particles are more reactive than larger particles, they could have a large influence on the water quality downstream. Consequently, their removal is important. The trapping efficiency of the floodplains can be further increased by planting vegetation /14/. The reduction of humic acids by sunlight /15/ would be more effective in shallow floodplains than in deep settling basins.

Perspectives

Several of the results of experiments presented here are based on a limited amount of data and need to be expanded. Especially rainfall simulations with realistic rainfall intensities, which are much lower than the intensities presented here, need to be performed to test whether dynamics observed here is true also for low intensity rains. The origin of sediment during rainfall runoff events should be studied. Control of erosion of base soil peat of low settling velocity from the mine surface during intense summer storms would require a long settling time and profound detention in the ditch network. On the other hand, if most of the sediment load results from erosion of channel deposits the focus should be on controlling these processes. An estimate of the portion of suspended load, bed load and floating material load is needed to evaluate the importance of different control structures. Data on processes is needed to expand the model presented here and to calibrate and validate it. Measurements are also needed to evaluate the effect of controlled drainage on sediment and nutrient transport.

An extensive monitoring program was initiated during the summer of 1995 at Pohjansuo, Jämsänkoski, Central Finland. The aim of the project is to gain more information about peat mine hydrology, hydraulics, sediment and nutrient transport in order to further develop methods to reduce the pollution load. The measurement consists of measuring rainfall, surface water quantity and quality, and groundwater elevations. Measurements are made on two 400 m^2 peat mine surfaces and two 0.06 km^2 drainage areas.

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Application of EUROSEM at the catchment scale

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Abstract

Within the research programme "Economics and Ecology", nutrient losses and erosion are calculated for different agricultural management systems. Based on economic constraints and measures, agricultural practice, including crop rotation, tillage, fertilizer or manure applications, is simulated for individual farming systems. This paper describes the application of the deterministic event based erosion model EUROSEM in combination with a digital elevation model GRIDSEM, to analyse erosion at the catchment scale. The GRIDSEM uses a grid size of 30*30 m and includes a simple routine for routing runoff and sediment between cells. For each day with runoff during a 20- year simulation period the soil erosion level is estimated for each cell, based on 1) the permanent landscape related characteristics of slope and soil type, and 2) parameters determined by the agricultural management system, including parameters for plant height, soil covered by plants, surface roughness and soil cohesion, and 3) actual weather data. A distinction is made between rainfall and snow melt induced runoff events. Changes in erosion level could be estimated from the output of GRIDSEM and by analyses of the estimated erosion levels for the cells. Preliminary results show that this method is capable of a) estimating changes in overall erosion levels, b) estimating the relative benefits of different agricultural management systems to total erosion and c) showing the unpredictability and importance of extreme erosion events, in both time and space.

Introduction

The work presented in this paper is part of an interdisciplinary research project where economists and ecologists work together. The project "Economics and Ecology - Resource management and pollution in agriculture" (RPMA) is a five-year research programme, financed by the Research Council of Norway and staffed with both economists and natural scientists /1/. Its operational aim is to analyse measures for reducing the loss of nutrients (nitrogen and phosphorus) and erosion from agricultural land into streams, lakes and coastal waters. The programme aims to combine insights from both economics and natural sciences and to promote a systems view which takes into account important processes in other sectors. The programme is divided into three main substudies:

A. An analysis of the nitrogen and phosphorus cycles affecting the main sectors of the Norwegian economy.
B. Valuation of environmental effects resulting from varying methods of agriculture, concentrating on the effects of reduced outlets of N, P and loss of soil.
C. Evaluation of actions and public measures which aim to reduce pollution from agriculture and create a more ecologically adapted industry - a search for the most cost-effective strategies.

This paper deals with the analysis of erosion within substudy C. Figure 1 represents the system to be analysed. Based on given social economic conditions (product/fertilizer prices, production costs, subsidies, interest rate, labour costs, etc.) and the natural resource basis (farm size, soil type, climate, expected crop production, crop rotation, etc.) the farmer optimizes the management system for his farm.

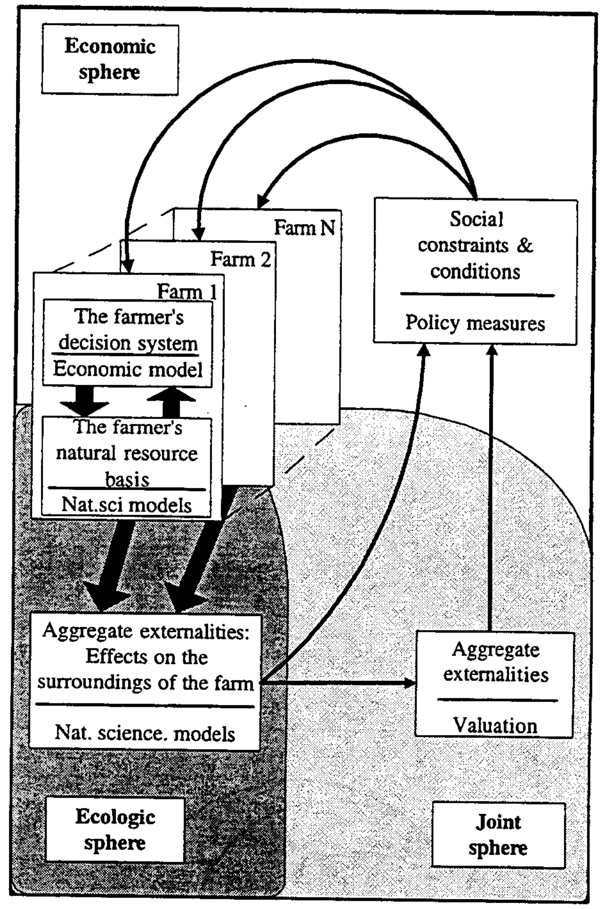

Fig. 1. Schematic presentation of the farmers interactions with the economic sphere and the environment showing the links between agro-ecology and economics.

Every management system will have externalities, i.e a side effect on the environment in form of nutrient and soil losses, but the level will vary with both the management system and the actual weather conditions. If optimal farming systems cause environmental effects at a level that is unacceptable for society, then authorities can introduce economic constraints or regulations that favour the choice of a farming system with lower externalities. To analyse the system a series of simulation models, or new ones created from the "shelf" are used. The close cooperation between economists and natural scientists has achieved consistence through the whole system simulated. This paper presents the method in which EUROSEM (EUROpean Soil Erosion Model) and GRIDSEM are used to analyse erosion and P-losses from management systems within the aims of the described project.

Model description

The economic model ECOMOD

The economic model is an optimization model running for a period of 20 years for different model farms (see definition on the following page). Besides economic outputs the model simulates the optimal management practice the farmer will choose for each year, given the prevailing conditions. Output variables used further in the erosion analyses are crop, catch crop (yes/no), type and date for soil tillage, date of spring sowing (weather depending) for each field of the model farms. Different scenarios are realised by changing the conditions under which the farmer has to make his decisions. This may be a change in relative prices, subsidizes on postponing soil tillage until spring or requirements to use a catch on e.g 50% of the areal basis of the farms. A more detailed description of the economic model will be published in the final report of the project /2/.

EUROSEM

The European soil erosion model (EUROSEM) is a process-based erosion prediction model designed to predict erosion in individual events and to evaluate soil protection measures /3/. The model uses a mass balance equation to compute sediment transport, erosion and deposition over the land surface. The model simulates the volume of rainfall reaching the ground surface as direct throughfall, leaf drainage and stemflow. The rate of detachment of soil particles by raindrop impact is computed as a function of the energy of the direct throughfall and leaf drainage, the detachability of the soil and the depth of surface water. The detachment of soil particles by runoff is determined as a function of the difference between transport capacity and existing sediment concentrations in the flow, simultaneous deposition of sediment from the flow and the cohesion of the soil. Compared with other similar models, EUROSEM simulates tillage and crop cover effects in a dynamic way and accounts for soil protection measures by describing the soil microtopographic and vegetation conditions associated with each practice. EUROSEM must be linked to a hydrological model capable of predicting and routing runoff. The present version is linked to KINEROS, a kinematic runoff model /4/.

GRIDSEM

The GRIDSEM modelling system /5/ was originally used with the USLE, but with the intention of extension to other erosion models at a later date. The GRIDSEM concept forms a data management and general computation "platform". It applies the principles of the various erosion models to individual cells (small rectangular areas) of a catchment or slope, and also takes account of some of their global interdependences. However, different erosion models have different strengths and disadvantages. In this application erosion values derived by EUROSEM are transposed directly as a grid cell factor. This in contrast to applying USLE where erosion is calculated for each cell based on their factors.

The digital elevation model, or DEM, (sometimes called a digital terrain model, or DTM) represents the surface, and forms the basis for several parts of the modelling process. As such it has an important function, so a necessary demand must be that it accurately simulates the area's topography. E x amples of the importance of the accurate representation of the surface include modelling of sediment generation, export from cells and the possibility of deposition in others. The following demands are made on the DEM: a) high resolution, b) high degree of accuracy in representing the 3-dimensional surface and c) fast algorithm.

Ad. a. High resolution

Erosion can occur in a serious degree on very short distances from stable soils. Studies on erosion have also often been conducted on quite small (1 or 2 m. square) trial areas. Initially it was felt that it would be most appropriate if the resolution for the model was as fine as possible, while still being of a size compatible with the capacity of the computer. In the end we settled for cells of 30 m, which gave about 180.000 cells for Mørdre area and about 600.000 cells for Auli area (see 3.1). This was well within the capabilities of the system.

Ad. b. High degree of accuracy for surface

All of the data points available are used in the model. Interpolation is done by piecewise linear functions orthogonally on the Z/X, and Z/Y axis, in the Euclidian space bounded by the minimum/ maximum values of the 3 axis (X,Y,Z). Representation of the slopes, their length, gradient and direction of slope, have a higher priority than relative height differences.

The central data management and computation modules are the engine of the system. Other modules also form logical parts of the engine, including licensed software, scanners etc. The use of EUROSEM or USLE means that erosion mapping units must be defined. These are areas or elements which have spatial homogeneity with respect to soil series, slope gradient and length, crop rotations and tillage. In the application of EUROSEM, a previously calculated potential erosion value is given to each erosion mapping unit. The erosion mapping units are operationally defined here as the cells of the model.

Numerical analysis is concerned with approximating solutions to mathematically expressed problems. This model consists of a raster of square cells, of size defined by the user, which has a particular shape and area, delineated by the user in a perimeter data file. The shape can in fact be irregular, with the important advantage over other raster models that there is no «dead» area of redundant cells between the actual perimeter and the rectangular limit of the raster. This saves computer and storage space, and cuts out search operations on redundant cells. This model is relatively simple, with certain intra- and inter-cell relationships. Nevertheless, some of the relationships are simplified to give a more efficient numerical model.

The model has as input, or can generate, the following attributes for each cell:

SPATIAL: Coordinates X, Y of the centre of the cell.
TERRAIN: Coordinate Z (height).
FACTOR: erosion (as simulated with EUROSEM).
FACTOR: Slope (segment attribute).
FACTOR: soil type
Road: Whether the cell is road surface (or other special type)
Channel: Whether the cell is a water channel
Deposition: Whether the slope is conducive to deposition in the cell
Direction: The direction of slope
Class: The class or "mother field" to which the cell belongs (see the next section)

The model is built up as a database with data objects representing the cells of the model. This enables the model to be processed and updated, without unnecessary re-computation of the respective layers each time. As previously stated, the structure of the system is modular. The modules are independent of each other, but in several cases they pass parameters and/or data. The overall control is exercised by the system GRIDSEM.

Method description

In practice it is impossible to run EUROSEM for the large number of grid cells and a method has been chosen where EUROSEM is run beforehand for all possible combinations of properties a grid can have, including weather input.

Landscape and farms within the RPMA-project
The analyses are done for two areas in southern Norway. One study area called «Mørdre», situated 50 km NE of Oslo, and the study area called «Auli» situated 100 km SW of Oslo. Mørdre represents one landscape area and Auli is divided into two landscape areas. The areas are bordered by watershed divides. An overlay of soil, topography and cadastral map are taken into GRIDSEM forming the basic grid cell properties. Based on agricultural statistics for the areas, 4 and 10 model farms with different main production and of different sizes have been defined to represent all the farms. Depending on the size of the farm, 3 to 7 fields with uniform soil type are defined such that the distribution of soil types within the model farms equals the distribution in the whole landscape. Further, each individual map figure in the study areas (achieved by an overlay of soil type and cadastral map) is assigned to be alike a field of a model farm and so will have a management system equal to the "mother field". This information is also stored in GRIDSEM.

Crop systems
EUROSEM needs information about the surface conditions of the simulated area. Besides the necessary soil type dependent physical parameters (hydraulic conductivity, erodibility, cohesion etc), a series of necessary input parameters depend on the crop grown and/or soil tillage have to be given. For each of 62 possible crop systems defined in ECOMOD, a table is created with a daily value for plant height, plant surface cover and a code for surface topography (seed bed, harrowed or ploughed). The economic model determines a crop system for each year in the simulation period. Combining this information with the available crop system dependent surface parameters, the actual surface conditions are defined for each day of the 20 years simulation period. For crops sown in spring, these values depend on the date of sowing (estimated by the economic model) and the 3 parameters are calculated thereafter.

Weather input data and winter hydrology
EUROSEM is not build to simulate snowmelt erosion. However, it is possible when selecting adequate parameter values and giving snowmelt runoff as driving input variable instead of precipitation combined with an infiltration rate of zero /6/. As a consequence summer and winter have to be treated separately. From simulations with the hydrology and heat model SOIL /7/, daily runoff (mm) for both summer and winter is available for the three main soil types (clay, silt, sand) in the two study-areas for the study period. Weather data input for SOIL has been daily values for temperature, precipitation,

wind speed, air humidity and cloudiness measured at both study sites.

Summer
Precipitation has to be given in a series of cumulative rainfall-depth (mm) for a single event. For the two study areas only daily precipitation values were available. However, for a weather station close to the area «Mørdre» pluviograph observations with a resolution of mm/min were available for a 5 year period, but only for the frost/snow free periods of the years (May-October). Single events (> 6 hours without rain in between) in pluviograph data were divided into 9 classes based on total precipitation per event. Further, each main class was divided into 3 subclasses (low, medium, high) based on maximum precipitation intensity per 10 minutes, resulting in a total of 27 precipitation classes. Finally, from the complete pluviograph data series, for each class the event was selected that matched best the class characteristics: total amount, maximum intensity and mean intensity of precipitation.

These 27 selected natural precipitation events are used as input in EUROSEM for simulating erosion caused by precipitation. In the series of measured daily precipitation, each event resulting in runoff (from SOIL) is assigned a precipitation class based on total amount of precipitation and random selection of intensity.

Winter
Runoff is defined as snowmelt runoff as long as there exists a snow cover (SOIL-output) on the soil surface. Based on the simulated runoff amounts (mm/day) 8 runoff classes were accepted. It is known from field observations that erosion of a frozen soil differs from erosion from a thawing soil /8/ and the snowmelt runoff classes are divided into two, based on temperature in the soil surface (<0°C or >0°C). From the simulated snowmelt runoff data series, within each class one event is randomly selected to represent that class. Looking at measured snowmelt hydrographs /8/, a time distribution for the daily snowmelt amount is estimated and used as input for EUROSEM. Finally, each simulated winter-runoff event is assigned a snowmelt- runoff class.

Erosion estimate for the landscape area
EUROSEM has until now a user interface that did not allow batch mode running of the model directly based on the output of the economic model. It is therefore chosen to run EUROSEM beforehand for fields that equals the grid size in the GRIDSEM (30x30 m). A set of input parameter files is built for winter, with and without frozen soil, and summer with each of the possible combinations of surface conditions as defined by the crop systems and for three soil types. Further the parameter files were duplicated for 6 different slope classes. These parameter files are then run in combination with the appropriate precipitation or snowmelt input files. All together about 1500 runs were needed. EUROSEM is calibrated with field measurements /8, 9/ as much as possible. For parameter and weather input conditions not represented in field data series, the relative level of soil loss is compared to what would be expected.

In the final step daily information on surface conditions for each field, the daily weather (summer precipitation with runoff, or snowmelt runoff) and soil type is used to select 6 soil loss values (one for each slope class) out of the 1500 EUROSEM outputs. These daily soil-loss values for one event are then carried over to all the grid cells belonging to the same "mother field" taking into account the actual slope class and soil type of the cell. Erosion can then be analysed in GRIDSEM taking into account routing in the landscape and independent of the landscape by analyses of grid cell properties.

Table 1: Potential soil erosion (kg/ha) from grid cell sized plots (30x30 m) simulated with EUROSEM for two summer precipitation events classes (EC), two slope classes and two soil types for different soil surface conditions. For sandy soils, no erosion was simulated for these cases.

	Slope class 2 (2-4 %)				**Slope class 5 (16-25 %)**			
	EC 5.2[1]		EC 9.2[1]		EC 5.2[1]		EC 9.2[1]	
Soil surface conditions	clay	silt	clay	silt	clay	silt	clay	silt
New prepared seed bed	1.2	0.0	1657	598.0	4.6	0.0	4352.0	4954.0
Fully developed crop	0.0	0.0	25	0.1	0.0	0.0	110.0	5.4
Stubble, straw removed	0.0	0.0	265	0.2	0.0	0.0	1106.0	837.0
Stubble, with straw	0.0	0.0	16	0.1	0.0	0.0	84.0	5.5
Grass	0.0	0.0	0	0.0	0.0	0.0	0.2	2.2
Ploughed	0.1	0.0	1524	0.2	0.2	0.0	4122.0	2815.0

[1] EC 5.2: total 8.8 mm; maximum intensity: 1.6 mm/10 min.
EC 9.2: total 74 mm; maximum intensity: 2.6 mm/10 min.

Table 2. Example figures of potential erosion levels (1000 kg/ha) during snow melt runoff events (EC) of two magnitudes over thawing soil under different soil surface conditions and two slope classes.

		Slope class 2 (2-4%)		Slope class 5 (16-25%)	
Surface conditions	Soil type	EC4.2	EC 6.2	EC 4.2	EC 6.2
Ploughed	clay	0	0	0.7	1.0
	silt	0	0	6.8	10.0
	sand	0	0	8.9	22.0
Grass	clay	0	0	0.4	0.5
	silt	0	0	3.9	6.0
	sand	0	0	5.9	14.0

[1] EC 4.2: total 6.1 mm; EC 6.2: total 12.2 mm;

Evaluation of the method

Estimating erosion on the scale of a watershed/landscape is difficult because of the spatial and temporal variation that always exists. Because a large landscape can not be simulated as a whole by deterministic models, several methods have been proposed to overcome the problems spatial modelling rises and the method described here is one alternative. It is our duty to give the strength and weaknesses of the method, to estimate the errors and to evaluate for what kind of analyses the method is appropriate to be used for.

In the method described erosion is first estimated for small plots and individual events, based on a detailed description of the soil surface, surface slope and real precipitation events or simulated snowmelt runoff events. Under the condition that the model is well calibrated these estimates can be considered to be correct. Adding up the results for single cells for the whole landscape introduces the need for routing of both runoff and sediment.

In GRIDSEM only simple routing routines could be used on the whole watershed/landscape and the main error in total erosion is introduced here. Another error introduced in up scaling is that spatial variation in precipitation during an event and variation in soil water content at the start of an event, is not taken into account. What is achieved by the method is a distribution of erosion levels over the landscape. Changes in the distribution caused by the introduction of measures to control erosion can be analysed. Further, the change in relative erosion levels for different scenarios can be analysed based on grid cell properties. The effect of different means to reduce erosion in a watershed can be evaluated in terms of cost-benefit, taking into account the economic aspects for both the farmer and society.

The advantage of the method is its basis in measured daily weather data instead of average values, thus keeping a high resolution in time. Seldom extreme events can be identified and their importance for total erosion levels can be evaluated. Erosion can be analysed for a long time period (20 years) but the analyses are still based on actual daily values for plant and soil surface conditions.

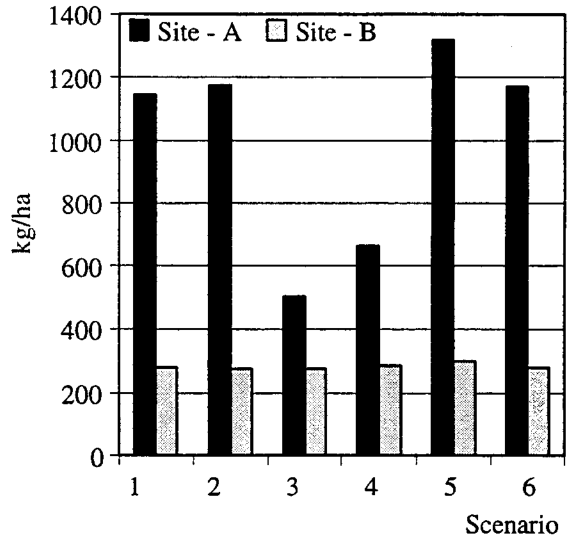

Fig. 2. Mean erosion (kg/ha/year) for six different scenarios and two sites in southern Norway simulated with EUROSEM-GRIDSEM over a 20-year period - Scenario 1: A 33% proce reduction on grains; Scenario 2: Manure storage for 12 month; Scenario 3: Subsidy to abandon fall tillage; Scenario 4: 50% arable requirement land with cath crops; Scenario 5: Basic scenario (1993 situation); Scenario 6:100 % tax on N in mineral fertilizers.

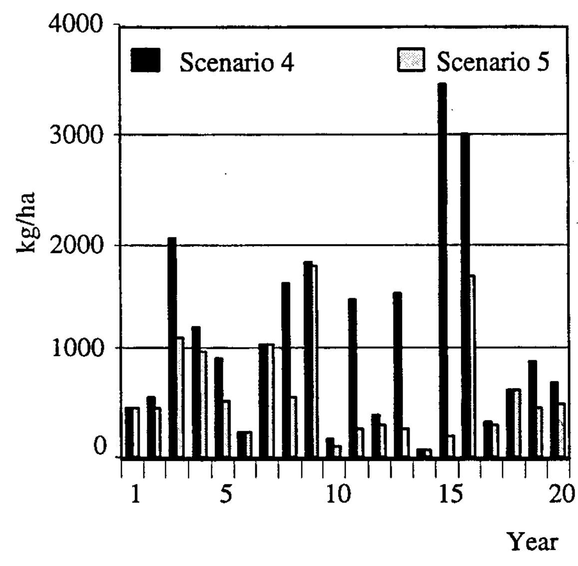

Fig. 3. Mean erosion (kg/ha/year) for a twenty year period for site A in southern Norway simulated with EUROSEM-GRIDSEM for scenario 4 and 5 (see figure 2 for description of scenario).

Results

Depending on soil surface conditions, slope and event characteristics (amount, intensity), a large spread in potential erosion values was found. Table 1 shows some potential erosion values for moderate and steep slopes with a medium strong and a heavy rainfall event. Table 2 shows some examples for potential snowmelt runoff erosion on thawing soil. These values are high, but it has to be remembered that infiltration has been put to zero in the model. The output from GRIDSEM showed that at average about 50% of the potentially eroded sediment reaches the streams. Figure 2 shows the mean soil loss for six different scenarios and two study sites. The modelling system showed both remarkable differences between the different scenarios, but also between two study sites that differ in main soil type, topography and main agricultural practice. Figure 3 indicates that a long simulation period is needed to show differences between scenarios. For some years no differences are found while in other years soil loss differs considerable between scenarios. For a more detailed discussion of results the reader is refered to /2/.

Acknowledgement

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Erosion plot studies 1989-92: Fit of a simple model to surface runoff from agricultural land

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Introduction

Tremendous efforts have been made to develop and evaluate models describing surface runoff and erosion. Consult papers in e.g. /1/. It seems as if this task is very difficult, as no models until now have succeeded in giving really good predictions of the processes. However, even if such complex models containing a lot of parameters, have problems in describing the processes on a catchment scale, it may be useful to apply simple models to controlled environments, as e.g. small plots, to get an impression of which key parameters determine the processes. The present paper reports such a model, which was fitted to surface runoff data from plot studies /2/.

The plot investigation

Surface runoff, erosion, losses of soil and phosphorus (P), and related soil physical properties were studied in differently cultivated plots on 10% sloping areas at Foulum (loamy sand) and Ødum (sandy loam) in Denmark, 1989-92. The treatments were as follows:

Code	Winter surface	Treatment
GRS	Grass	Permanent ryegrass cut four times per year.
CCR	Catch crop	Spring barley followed by a ryegrass catch crop during winter, ploughed in spring.
PLG	Ploughed	Spring barley ploughed in autumn.
WUD	Wheat up-down	Winter wheat drilled up and down slope, ploughed in autumn.
WAC	Wheat across	Winter wheat drilled across slope, ploughed in autumn.
FLW	Fallow	Fallow, ploughed in spring and harrowed from time to time to remove weeds.

Surface runoff, erosion and P losses varied tremendously with season, year, cultivation and site. By far the greatest rates were recorded from October to March. The surface runoff, soil and P losses varied from 0.6 to 167 mm y^{-1}, 0.004 to 26 t soil ha^{-1} y^{-1} and 0.01 to 33 kg total P ha^{-1} y^{-1}, respectively. The magnitudes generally followed the order: fallow > wheat-up-down > wheat-across >> barley-ploughed > barley-catch crop > permanent ryegrass. Consult /2/ for a detailed report of the investigation.

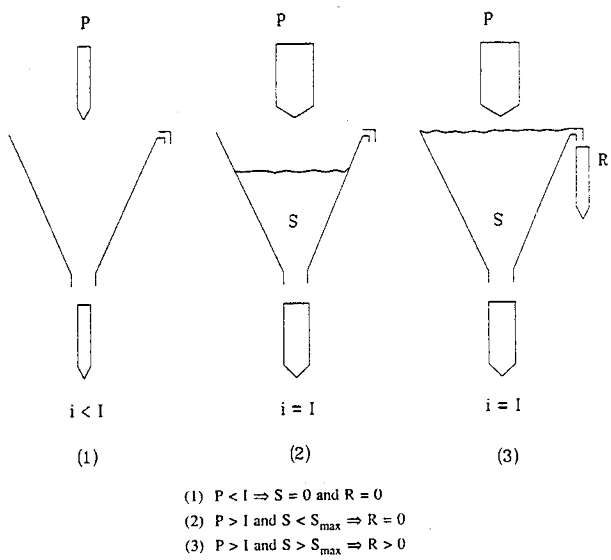

Fig. 1. Principle in model function for a specific soil surface. Three situations with different precipitation rates, P, but with constant infiltration capacity, I and depression storage capacity, S_{max} 'i' is actual infiltration.

The model of surface runoff

Model principle

A simple water-balance model for surface runoff on sloping land can be written as

$$S = P \times T - I \times T, \quad S > S_{max} \Rightarrow R = S - S_{max}$$

where S = actual amount of water on the soil surface (mm), P = precipitation rate (mm min^{-1}), T = time (min), I = infiltration capacity (mm min^{-1}), S_{max} = surface-depression storage capacity (mm) and R = runoff (mm). The principle of the model is illustrated with a funnel representing S_{max} (Fig. 1).

Simulation of surface runoff

Simulation of accumulated surface runoff from treatments PLG, WUD, WAC and FLW is given for a 60 minutes period (Fig. 2). The simulated shower causes surface runoff from treatments WAC, WUD and FLW, while the combination of infiltration capacity and surface depression storage capacity /2/ for the PLG treatment results in no surface runoff. Notice that only the FLW treatment is sensitive to the precipitation during the 32-42 minute period. The order of succession in amount of surface runoff is the same as measured in most years /2/.

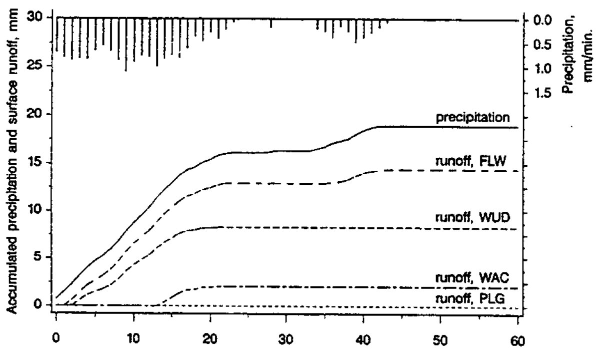

Fig. 2. Simulation of a 3/4 hour shower totalling about 19 mm of precipitation. Time step is one minute, using the values for infiltration capacity and depression storage capacity actually measured in the trial plots (Tables 5.1 and 5.5 of Schjønning *et al.*, 1995).

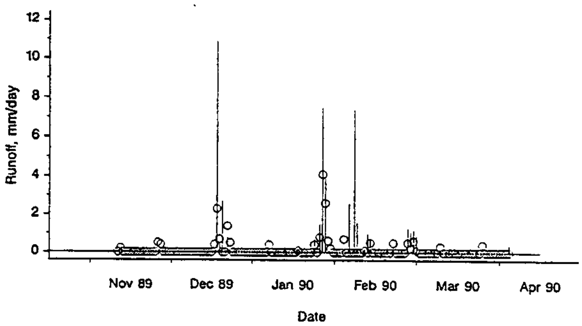

Fig. 3. Surface runoff at Foulum 1989-90, from WUD, as measured by water level recorder technique, and fitted by the model, using matching factors of 0.2 for the infiltration capacity and 0.05 for the depression storage capacity. Needles indicate observed values. Circles indicate model predictions.

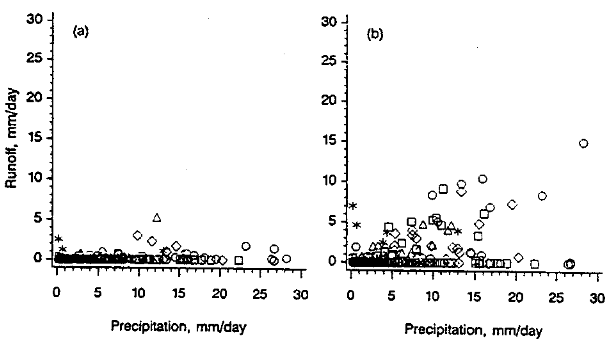

Fig. 4. Surface runoff versus precipitation for each single day during a 3-year period at Foulum, for treatments PLG (a) and WUD (b). Rain amount the previous two days is indicated by: ○ = < 3 mm, □ = 3-10 mm, ◇ = 10-20 mm and △ = > 20 mm. The symbol '*' indicates frozen soil.

Prediction
The predicted runoff for each time step was summed up to daily values for WUD at Foulum, 1989-90, and then related to the runoff measured with water level recorders (Fig. 3). Generally, the model predicted runoff in periods of recorded runoff. However, a matching factor of 0.2 was necessary for the infiltration capacity and one of 0.05 for the depression storage capacity.

Furthermore, the model gave a poor fit in periods with continuous, small-intensity rain (e.g. at the beginning of February). This could be due to deeper soil horizons with less hydraulic conductivity than the plough layer, e.g. a plough pan. When the plough layer soil was completely saturated, then the hydraulic properties of deeper soil layers might have governed the infiltration capacity.

Weather conditions and soil parameters
When relating recorded surface runoff to precipitation on a daily basis during the three years, different patterns appeared for the different treatments (Fig. 4). Runoff often occurred after a period of several rainy days. No significant effect of preceding rain could be detected at Ødum. At Foulum, however, the effect was statistically significant for the PLG treatment ($P=0.012$) with decreasing significance for WAC ($P=0.120$), WUD ($P=0.269$) and FLW ($P=0.523$). The significant effect of continuous, small-intensity rain for the ploughed soil probably reflected the existence of deeper soil layers with smaller hydraulic conductivities than the surface.

Conclusions

A simple model of surface runoff, including only precipitation, water infiltration capacity and soil surface depression storage capacity was shown to give relative estimates of surface runoff corresponding to observed values in differently treated plots in the field.The need of matching factors indicates that better estimates of water infiltration capacity and - especially - the depression storage capacity are needed.

Poor model simulation of situations with continuous, small-intensity rain may be due to deeper soil horizons with smaller hydraulic conductivity than the plough layer.

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NORPHOS, NOrdic Research project on losses of dissolved and particulate PHOSphorus from arable land to the aquatic environment

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Abstract

NORPHOS is a research project including experimental work, field measurements and modelling of phosphorus transformation and transportation in cultivated soils in the Nordic countries. NORPHOS will contribute to a better understanding of the processes behind losses of dissolved and particulate phosphorus from arable land. Knowledge obtained will be used to develop a SOIL-Phosphorus model on top of a well defined and tested hydrology model for the soil-plant system.

The close cooperation under NORPHOS during model development, process studies and field scale measurement will stimulate the exchange of ideas and methods between the Nordic countries, and facilitate greater cooperation between the scientists involved. The main objective of the project is to increase our understanding of processes related to the loss of phosphorus from agricultural soils and to incorporate this knowledge into mathematical models enabling more acurate prediction of phosphorus losses from arable land under different climate conditions and management practices, and to increase the potential for regulating these losses. The integration of Nordic research on P in a network, that can continue after NORPHOS is finished, is in itself an important objective.

Introduction

NORPHOS is a research project including experimental work, field measurements and modelling of phosphorus transformation and transportation in cultivated soils in the Nordic countries. NORPHOS will contribute to a better understanding of the processes behind losses of dissolved and particulate phosphorus from arable land. Achieved knowledge will be used to develop a SOIL-Phosphorus model on top of a well defined and tested hydrology model e.g SOIL /1/ for the soil plant system. This model will be connected to an erosion model e.g EUROSEM /2/ The close cooperation under NORPHOS between model development, process studies and measurements on a field scale will stimulate exchange of ideas and methods between the Nordic countries, and elaborate cooperation between scientists.

The project started June 1, 1993 and will finish May 31, 1997. Four Nordic countries are represented in NORPHOS with 6 scientists from Danmark, 5 from Finland, 8 from Norway and 5 from Sweden. Scenior scientists coordinate the main research topics, and sub-projects are most often carried out by junior scientists. The main research projects are not restricted to one country and in this way a closer cooperation between Nordic scientists is achieved. NKJ/SNS has appointed Dr. Sjoerd van der Zee, Agricultural University of Wageningen, NL, as referee for the project

The total budget for NORPHOS is DKK 6.1 million of which about 3 million is covered by a NKJ/SNS project and 3.1 million by others. The NKJ/SNS support is a recommendation for the national research councils and does not guarantee support from them. This lead to much frustration at the start of the project.

Main objective

To increase our understanding of processes related to the loss of phosphorus from agricultural soils. To incorporate this knowledge into mathematical models to enable better predictions of phosphorus losses from arable land under different climate conditions and management practices, and to increase our potential for regulation of these loss.

The specific **scientific** objectives of NORPHOS are the following:

i. To increase our understanding of the processes related to phosphorus transport through the soil system.
ii. To increase our understanding of the processes related to losses of phosphorus by surface runoff and erosion.
iii. To quantify the seasonal variation in bio-availability of phosphorus in runoff and eroded sediment in relation to agricultural management.
iv To examine the effect of cold climate on the processes studied.
v. To model and measure transportation of P-compounds in the soil plant system.
vi To stimulate a closer cooperation between Nordic scientists, doing research on phosphorus in the soil-plant system.

With the available economic and human resources not all possible questions can be answered. Therefore, the integration of Nordic research on P in a network, that can continue after NORPHOS is finished, is by itself an important objective.

Scientific background

Due to the improved control of point sources, the phosphorus (P) load from non-point sources becomes increasingly important. Data obtained in recent years have indicated that an essential improvement of the environmental status of e.g lakes cannot be achieved by the control of point sources alone, but requires a concomitant reduction of the diffuse loading from the open land.

Overland loss of P from arable land is strongly related to soil erosion /3,4/, but may also be due to loss of P from the vegetation during snowmelt /5/. In areas specialised in animal husbandary, loss of dissolved P can be related to the spread of manure or to leaching from overwintering grassfields in spring. Also significant loss of both dissolved and particulate P through artificial drains has been observed /6/. To this date few studies have adressed both overland and subsurface losses. However, the sources of P from a typical clay soil watershed (50 km^2) delivering to the Lake Erie Basin have been quantified /7/. It was concluded that effluent from the subsurface drains constituted 60% of the annual runoff. Overall, only 34% of the total P from subsurface drains was sediment associated. The transport of particulate P through drains under Danish soils account for about 25% of the total P in drainage waters /8/.

Phosphorus is the main cause for eutrofication of fresh waters, but the bio-availability of P may be related to its origin. To find the most effective measures to reduce the P-load in fresh water, we should increase our understanding of the processes leading to losses of bio-available P-compounds from arable land. Bio-availability of P can be measured in laboratory experiments /9/. Little research has been done on how bio-availability of P in runoff and sediment varies between seasons and agricultural management practices. An extrapolation of laboratory analysis to natural fresh waters is still difficult. The ultimate effect in the water depends on total nutrient load, pH, depth, and the micro and macro biological population, in other words the total fresh water ecosystem. Seasons cause a regular rythm with high biological activity in spring and summer and low activity during winter. This cycle affects the bioavailable amount of P in the system. NORPHOS will not include research on eutrofication.

There exists a common understanding that erosion and nutrient losses at a site are the result of a complex interaction between soil type, topography, climate and agricultural management. Soil structure strongly affects infiltration capacity and water storage capacity of the soil. Weather factors as temperature, precipitation, and potential evaporation directly affect crop development and the hydrological cycle. Erosion and losses of nutrients depends on precipitation intensity and the amount of surface runoff, infiltration rate and runoff through drain pipes. The bio-chemical processes in the soil depend both quantitatively and qualitatively on soil water content and soil temperature. Little is known about the P turnover in the living microbial biomass. Leaching of P from permanent grassfields in spring is known, but the chemical-biological processes behind it are not understood. Chemical and biological processes determine the amount of phosphorus available for transport, while the amount transported depends on the water flows in the system.

A consequence of preferential flow in a soil profile is that the residence time of the porewater is so low that it may be far from chemical equilibrium with the surrounding soil /10,11/. With regard to the transport of particulate P the macropores may act as channels for colloidally bound P. Analogous to this colloidal macropore transport of DDT in soil developed on clayey till has been demonstrated /12/.

Field observations in Norway, Sweden and Finland have shown that besides autumn rainfall, snowmelt is the other main cause of runoff and erosion, most times with a higher P-load than during the rest of the year. Therefore, both the hydrological and biochemical processes under winter conditions have to be studied, also with regard to crack formation in freezing soils. Good estimates of soil erodibility and the effect of freezing and thawing on this parameter are important too.

Overview of planned activities

Based on the NORPHOS' objectives and taking into account ongoing research activities at the national level, the following activities, divided into 5 main groups, are planned. Sub-projects within each main group are connected to different countries.

I. Process studies of P transformation and transportation in soils
Coordinator Jakob Magid, DK

I.1 Leaching of dissolved and particulate P from clay soils - laboratory studies.
Project leader: Jakob Magid,
Collaborators: Hans Christian Bruun Hansen (DK), Peter Jørgensen (DK), and Georgia Destouni (S).

I.2 Loss of particulate P to the aquatic environment t h rough microbial biomass.
Project leader: Roar Linjordet (N).

I.3 Phosphate interchange between water and surface soil during surface runoff and soil erosion.
Project leader: Erik Sibbesen (DK), collaborators: Helinä Hartikainen (SF) and Markku Yli-Halla (SF).

II. Field measurements of P-load in runoff and eroded sediment from arable land.
Coordinator: Seppo Rekolainen, (SF) from August 1994 Petri Ekholm (SF).

II.1 Phosphorus release from overwintering plants and plant material.
Project leader: Barbro Ulén (S).

II.2 Loss of P from arable land with different soil's and climate in Norway.
Project leader: Trond Knapp Haraldsen (N), collaborators: Hugh Riley (N), Ragnar Eltun (N) and Kristen Myhr (N).

II.3 Phosphorus fractions in surface runoff from agricultural fields in Finland.
Project leader: Petri Ekholm (SF), collaborators: Kari Kallio (SF).

II.4 The influence of weather conditions and soil type on the interior erosion of P.
Project leader: Barbro Ulén (S), collaborators: Kristian Wennberg (S).

III Field measurement methods and data storage (Lillian Øygaarden, N).

III.1 Development of a sedigraph monitoring system.
Project leader: Bent Hasholt (DK), Collaborators: Erik Sibbesen (DK).

III.2 Standardisation of field measurement methods and data storage.
Project leader: Lillian Øygarden, collaborators: Barbro Ulén (S), Helge Lundekvam (N), Petri Ekholm (SF) and Erik Sibbesen (DK).

III.3 Establishment of a database for measurements of nutrient losses and erosion.
Project leader: Peter Botterweg, collaborators: Holger Johnson (S), Trond Knapp Haraldsen (N).

IV. Development of a mathematical model simulating P transformation and transportation in soils.
Coordinator: Peter Botterweg (N).

IV.1 Modelling observations from column experiments (I.1)and evaluation of their ability to explain the observations from fieldscale investigations.
Project leader: Jakob Magid, collaborators: Hans Christian Bruun Hansen (DK), Peter Jørgensen (DK), and Georgia Destouni (S).

IV.2 Modeling phosphorus transport in surrface runoff.
Project leader: Kari Kallio (SF), collaborators: Petri Ekholm (SF), Helinä Hartikainen (SF), Markku Yli-Halla(SF), Erik Sibbesen (DK).

IV.3 Development of a model for simulating losses of dissolved and particulate phosphorus from arable land.
Project leader: Peter Botterweg (N), collaborators: Per-Erik Jansson (S), Lillian Øygarden (N), Trond Knapp Haraldsen (N) and all the others.

V. Seminar program.
Coordinator: Jakob Magid (DK).

Relation between the different sub-projects

A central activity in NORPHOS is to develop a mathematical model for P-transformation and transportation in the soil-plant system. To achieve this a better understanding and quantification of underlying processes is necessary. Model development will be based on existing hydrology models (SOIL), and will be connected to the erosion model EUROSEM. The modelling effort by the Danish group (IV.1) will concentrate on modelling P-transport through the soil profile, with and without preferential flow. In sub-project IV.2 the objective is to model P-losses with surface runoff, especially in relation to snow melt periods. The third sub-project (IV.3) will solve problems related to the interface with a hydrology and erosion model, and will be responsible for the coordination between the modelling groups, and integrate the results into an entity.

The sub-projects I.1 - I.3 will give important information about bio-and physio-chemical transformation processes and about P-fractions transported through the soil profile. The model will also estimate the P-load in surface runoff and eroded material. To be able to test the model on a field scale, reliable measurements of P-load in runoff and sediment have to be available from fields with different soil types, agricultural management, and climate conditions.

Measurement series from the different Nordic countries (sub-projects II.1-II.4) make it possible to validate the model and test its generality. To get reliable and comparable measurements from the different countries, a standardisation of measurement methods and development of new methods is necessary (sub-projects III.1-III.3). The measurements have to be coordinated with the demands of the model(s) in relation to time resolution, variables measured and dimensions. Data from measurement series have to be stored in a standard format to increase usefulness for others. A database with information about available measurement series will increase the use of these series in both scientific and and management applications. An integration of model work with the experimental work and field measurements is realised by that scientists working on model development are involved in the other activities too. The seminar program will create a platform for exchange of experience and knowledge between the participating scientists and external experts.

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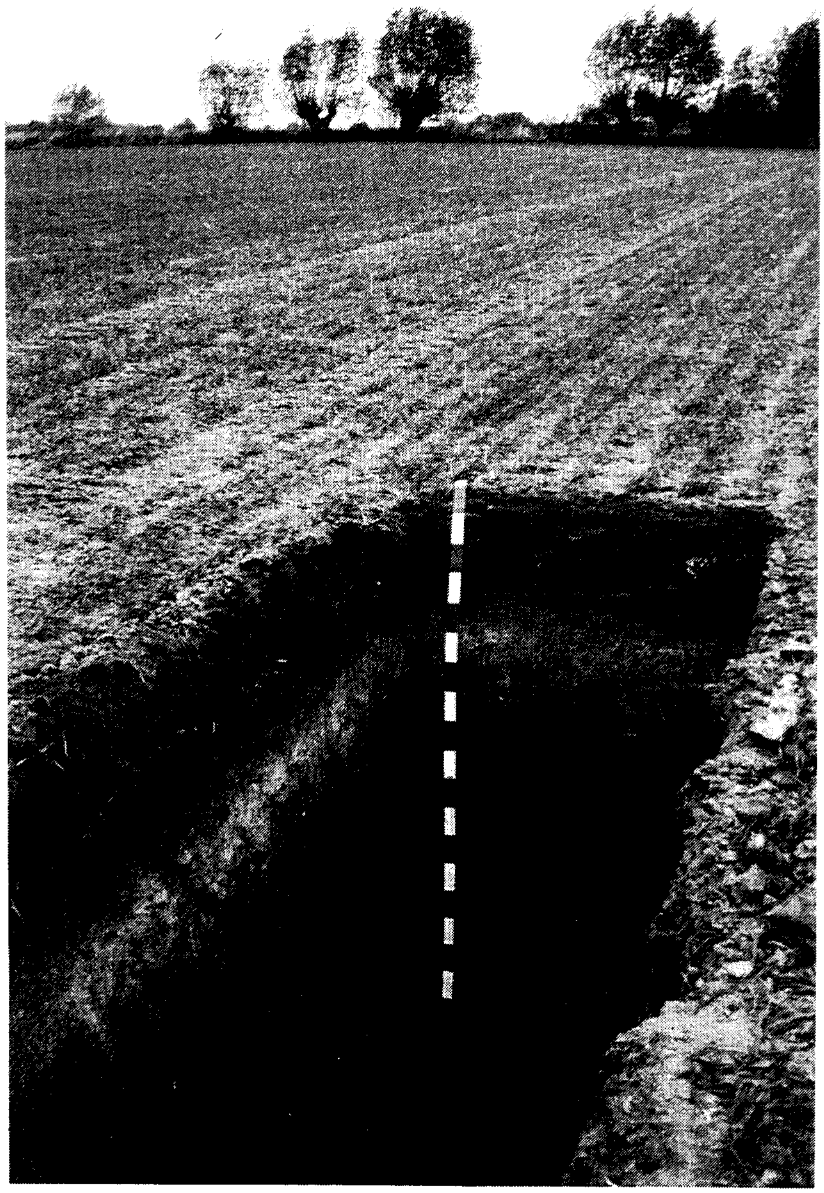
Technical Session II



Subsurface runoff and delivery of sediment and associated nutrients to watercourses

Soil erosion with preferential flow to drainage system in artificially levelled clay soils

L. Øygarden, J. Kværner and P. D. Jenssen
Centre for Soil and Environmental Research, JORDFORSK, Aas , Norway

Abstract

Runoff measurements from artificially levelled land in Norway have shown that rates of particle erosion to the subsurface system may be even higher than that from surface runoff. Drainage systems under autumn tillage sites gave high soil losses whilst no tillage reduced the losses significantly. In this study soil structures with macropores and cracks were examined to find out more about water and particle transport via preferential flow routes. The field site soil type was a silty clay loam. Artificial levelling at the site had exposed an unstable subsoil, susceptible to cracking. Large volumes of particulate transport have previously been recorded in drainage runoff from this site. A profile description of soil structure and CT scanning of monoliths from the plough layer, subsoil and backfill showed different density patterns and structural properties. The backfill consisted of clods and open voids with direct contact to the drainpipes. The soil down to 50 cm was cracked both vertically and horizontally and some cracks were leading into the backfill. Cracks of up to 10 mm width were found, indicating that transport of particles through them was possible. The measured field hydraulic conductivities varied with 3 orders of magnitude, indicative of a preferential flow network. Infiltration with a dye tracer visually demonstrated rapid flow of water through cracks leading directly to the drainage system. A hydraulic conductivity of 86 m day^{-1} was measured in the soil; and one of 504 m day^{-1} in the backfill. These results indicate that particles can be eroded from the plough layer and transported, through macropores and backfill, directly to drain pipes. Sawdust used as envelope material did not keep clay particles out.

Introduction

Erosion studies in Norway have focused on surface runoff processes. Recent studies have, however, shown considerable losses of phosphorus and particles through underground drainage systems on levelled clay soils /1-2/. Particle loss to the drainage system was strongly affected by soil tillage practices in autumn and it was even larger than sediment losses by surface runoff and erosion.

This study was made to examine soil structure, pathways of water flow and possibilities of particle movement to the drainage system in an artificially levelled, clay soil. Objectives were:

(i) to study water passing through the soil and the possibilities for preferential flow through cracks, macropores and backfill;
(ii) to study possible pathways for transport of eroded particles to the drainage system;
(iii) to try to find out if particles were eroded from the soil surface, plough layer, backfill or from the subsoil.

Materials and methods

Site description

The catchment (Fig. 1) is situated in Ullensaker about 40 km north of Oslo in Norway. The 0.9 ha area is mainly used for cereal production. Autumn ploughing along the slope contour has been the dominant tillage procedure. Marine sediments with silty clay loam and clay dominate, but some silt loam is also present. The area was grassland until it was artificially levelled in 1978. Except for one minor area that is slightly levelled, levelling has exposed deep clayey subsoils, with only a thin toplayer of the former topsoil material.

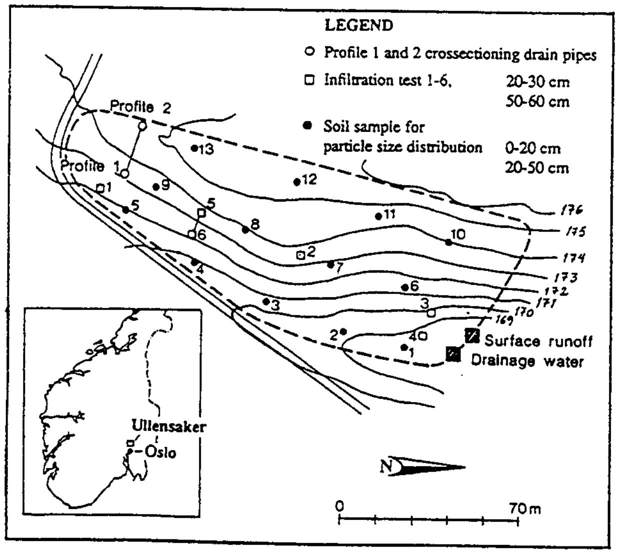


Fig. 1.Location of study area showing sites for profile description , soil sampling and measurement of hydraulic conductivity.

The field was drained in 1986 with a Raadahl digging-wheel machine. PVC- pipes of 48 mm and 83 mm diameters with 2-4 mm slot openings were laid at 0.7 to 0.9 m depth with 4 m drain spacing. The digging wheel made 20 cm wide trenches, and 8 cm depth of sawdust was used as envelope material. This work was completed in October after a dry summer and autumn. Therefore, the backfill consisted of fractured clay clods, resulting in an open structure with many voids. Many rainfall events were recorded soon after installation and, through the "open" backfill, probably caused some washing of the sawdust along the pipes.

Soil structure description

In October 1990 a 7 m long trench was dug, perpendicular to the drainage trenches, intercepting two drainage pipes. The new trench also crossed the border between levelled and slightly levelled land. From two cut profiles, soil structure was visually described and photographed; one profile representing levelled soil and the other soil less influenced by levelling. Macropores, cracks and fractures were marked out on a plastic sheet. Soil samples were taken from the different horizons for determination of particle size distribution, organic matter content and soil moisture retention curves. Variation in particle size distribution was also determined at 13 systematically distributed points across the entire catchment at 0-20 cm and 20-50 cm depth.

Computed tomography studies (CT)

Soil monoliths of 15-20 cm diameter and 30 cm height were extracted from the soil by a hand carving technique. The monoliths were taken from the plough layer (at 0-40 cm depth) and at 40-80 cm depth from the drainage backfill and subsoil between drainpipes. The monoliths were analysed by X-ray computer tomography (CT scanning) for determination of structure and possible macropores. Scanning was performed with a Siemens SOMATON 2, computer assisted Tomograph at the Agricultural University of Norway /3/. The machine setting was 125kV/320 mA. The pixel dimension was 1*1 mm and the slice thickness was 4 mm. Adjacent scans were made throughout the whole volumes with 2 mm distance between the scans. A computer programme was used to plot the density variations (CT values) of different slices from the monoliths. Plots of CT values lower than certain pre-set values were also made to study if the three cores had different macropore patterns. Statistical analyses were carried out on the data from an inner diameter of 16 cm in the soil cores, at between 8 to 17 cm depth. This inner diameter was used to eliminate the effect of none cylindrical core walls that would include values of air in the statistical analysis, from the core boundary.

Hydraulic conductivity

Hydraulic conductivity was measured using the constant head inverse auger hole method and equipment according to Jenssen /4/. The measurement strategy was carried out as follows:

1. A trench was dug, cutting through two drain pipes (Fig. 2) . Infiltration tests using a red dye tracer were performed at 20-30 cm depth and at 50-60 cm depth above a drainpipe, and at 1.2 m horizontal distance from the drainpipe. The soil was saturated before dye was added. The infiltration tests were performed with a constant 20 cm water head. Tensiometers were installed to measure hydraulic head. The path of the red dye through the soil profile was followed visually and the time before breakthrough of the red dye into the drain pipe recorded.

2. Hydraulic conductivity was also measured at 10-20 cm and 40-50 cm depth at four points.

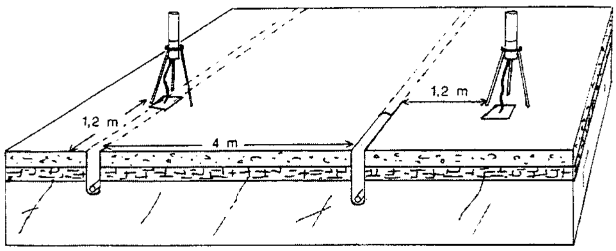


Fig. 2. Instrumentation and location of dye tracer test and measurement of hydraulic conductivity above and between two drain pipes.

Results

Computer tomogrammes of soil structure

A soil core cross section has both high and low density zones representing soil matrix and macropores. The pattern can be interpreted in terms of bulk density. Pixels with CT values below specified values are plotted (Fig. 3) for subsoil, topsoil and backfill.

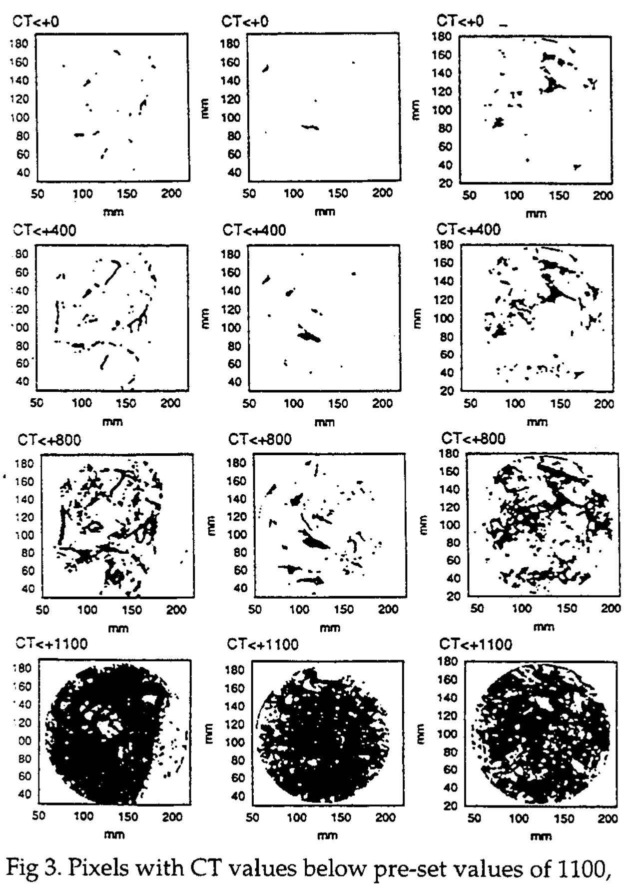


Fig 3. Pixels with CT values below pre-set values of 1100, 800, 400 and 0 for backfill (a) , subsoil (b) and plough layer (c).

The dark zones show pixels with low density. The plots show that all three cores have zones with values below 400 and 0, but that topsoil and backfill have more values, 400-0, indicating a more open structure. The border area made by the digging-wheel machine when installing drainage in the field can easily be seen in Figure 3a. This represents the area between denser subsoil and the more open structured backfill. Figure 3 illustrates the open structure in the backfill and also that all the cores have zones of low density. The pixel size of 1*1*2 mm restricts the scale of soil structural features that can be characterised. Mean CT values for the soil cores were 904 for subsoil, 895 for topsoil and 875 for backfill. Standard deviation was 178 for subsoil , 323 for topsoil and 263 for the backfill. Skewness was -1.4 for both topsoil and subsoil, but -2.2 for the backfill indicating more pixels with lower density values. In the backfill, dense matrix soil areas (with CT values units from a mean of 875 up to 1706) were broken up by a consistent pattern of points with CT values of between 0 and -980.

Hydraulic conductivity

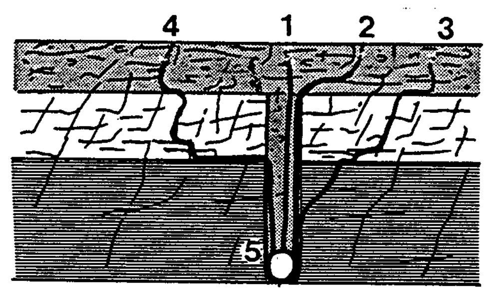
In the upper 20-30 cm the hydraulic conductivity varied between 0.6 and 25.3 m day^{-1} . In the subsoil the majority of tests gave values between 0.01-0.4 m day^{-1} . There were, however, some exceptions with higher values (2.65-12 m day^{-1}) assumed to be from the crossing of cracks or backfill.

In the test with infiltration of red dye tracer performed 1.2 m from the drainpipe the tracer flow was concentrated in a small number of cracks and macropores within and just below the plough layer. The red dye was observed coming out earlier from cracks at about 30 cm depth than from the ploughlayer. The red dye could be observed in water coming out from the drain pipe 20 minutes after the infiltration test was started, corresponding to a water flow of 86.4 m day^{-1}. The occurrence of the dye at the bottom of the profile wall after 1 hour and 10 minutes also showed a rapid transport through vertical cracks in the subsoil. In the test performed above the drain pipe the red dye was observed in the drain pipe 2 minutes after the test was initiated. It demonstrates rapid transport through the backfill with a k-value of 504 m day^{-1}.

Discussion

The levelling has caused exposure of original subsoil with high clay content, low organic matter content and unstable structure. In drought periods the soil is susceptible to substantial cracking and both the levelling and later drainage process may have caused structural, more permanent cracks. Both the profile description and the tomograph analysis showed this cracking structure. The levelled soil had an open cracking structure down to about 50 cm and some of the cracks were observed to lead directly into the backfill. The largest cracks and voids, up to 10 mm width, were found in the backfill, making a direct link between the topsoil and the drain pipes possible. Negative CT values of -200 to -980 were seen in all monolith cross sections. The subsoil had a denser structure without marked horizontal and vertical cracking. However, the CT values also showed zones of low density in the cross sections indicating larger macropores. Also the profile description showed single vertical cracks in the subsoil leading directly into the backfill.

The results from the soil structure description and the hydraulic conductivity measurements indicate an open soil structure with possibilities of preferential water transport in the trench backfill, along the border line between backfill and soil and in macropores and cracks in the uppermost 50 cm of the soil outside the backfill. Different pathways possible for water and particle movement are shown in Figure 4.



1. Detachment and transport from plough layer directly into backfill and down to drainage pipe.
2. Detachment from plough layer and transport along the border between soil matrix and backfill.
3. Detachment from plough layer and transport through vertical cracks leading into the backfill in the subsoil.
4. Detachment from plough layer and transport through vertical cracks down to 50 cm where horisontal cracks lead into the backfill.
5. Detachment of soil particles from the soil around the drain pipes

Fig. 4. Possible transport pathways for particles to drainage system.

A paper including more detailed results is in press.

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Drainage-associated losses of phosphorus and other nutrients from agricultural land following a conversion over to alternative production

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Abstract

In Sweden during recent years, there has been increasing interest in producing crops without using commercial fertilizer and pesticides, also referred to as "alternative production". Conversion over to alternative production has sometimes been mentioned as a way of reducing nutrient losses in runoff and leachate. An experimental field in Sweden with clay soil, studied since 1973, has shown very high losses of phosphorus and suspended solids. Phosphorus concentrations in the drainage water have far exceeded those considered to be acceptable from a limnological point of view. Five years of alternative production did not lead to a significant reduction in P losses. An early termination of the ley followed by autumn wheat led to moderate increases in P losses and strong increases in N losses. Lysimeter studies confirmed that the short-term effect of the production change on phosphorus losses was minimal. However, the soil type was an extremely important factor in this respect.

Table 1. Soil types, and mean values of easy soluble phosphor (P-AL) and total phosphorus of the tilled soil layer (mg/100g) at start (year 0) and after four years (year 4) together with number of lysimeters.

P content	Soil type	Humus	Clay	Sand	Fertilized sand
P-AL	year 0	4.2	5.0	4.8	4.0
	year 4	2.8	3.6	10.7	47.0
TotP	year 0	122	68	68	130
	year 4	122	75	70	130
number		3	3	4	5

Introduction

During the years 1989-1995, a changeover from conventional to alternative agriculture was made on more than 40 thousand ha of arable land in Sweden. Alternative agriculture, i.e. farming without the use of commercial fertilizer or pesticides, is also referred to as ecological farming in the Nordic countries. Nutrient needs are met mainly by supplying manure, compost, and urine, and added nutrients are organically bound to a large degree. Nitrogen fixation is enhanced by cropping lucerne and peas. The change in fertilization practices accompanying the conversion over to alternative agriculture undoubtedly influences nutrient losses to water, but knowledge concerning such changes is limited. The aim of this study was to document the phosphorus losses occurring during the first years following such a conversion.

Methods

Field studies

Phosphorus losses in drainage water from a conventionally farmed field in Central Sweden had been followed for 16 years (1973 to 1989) prior to a changeover to alternative agriculture. Thereafter, the monitoring was continued for another 5 years (1989 to 1994). About 45% of the cultivated area was covered with ley during both periods. However, after the conversion, the ley had a greater proportion of legumes, and its duration was two years instead of three or four years. Other frequently cultivated crops were winter wheat, oats, and barley. Research in the field was limited to making observations; i.e. no experimental treatments were carried out, and the conventional and alternative management regimes followed were of standard types. Precipitation falling on the field could go from the field to the drainage system both directly or surface runoff and through the soil profile. Water discharge was measured continuously. Drainage water was sampled once every fortnights except during periods with large amounts of runoff, when more frequent samplings were made. Contents of total phosphorus, phosphate phosphorus, suspended solids and other nutrients and constituents were determined according to the Swedish standards.

Lysimeter studies

Undisturbed soil monoliths, 3 or 10 dm long and 3 dm in diameter, were sampled according to /1/. The soils differed in type and phosphorus status as shown in Table 1. The monoliths with fertilized sandy soil came from the southwest of Blekinge County, where a large amount of mink manure is used together with commercial fertilizer. The other soils were from Central Sweden. The clay soil monoliths were taken from a field next to the observation field. The lysimeters were placed at a lysimeter station at Uppsala and cultured as shown in Table 2. After the first two years, some lysimeters received commercial fertilizer. Other "alternative lysimeters" received manure and urine with the exception of the heavily fertilized sand to which composted grass was added.

Table 2. Fertilization and crop removal (kg P/ha) in lysimeter studies (mean values).

Year	Crop	Fertilization	Removal by crop
1	ley (timotey and clover)	12	5
2	ley	0	5
3	spring wheat with catch crop	22	9
4	oat with catch crop	6	6
Sum		40	25

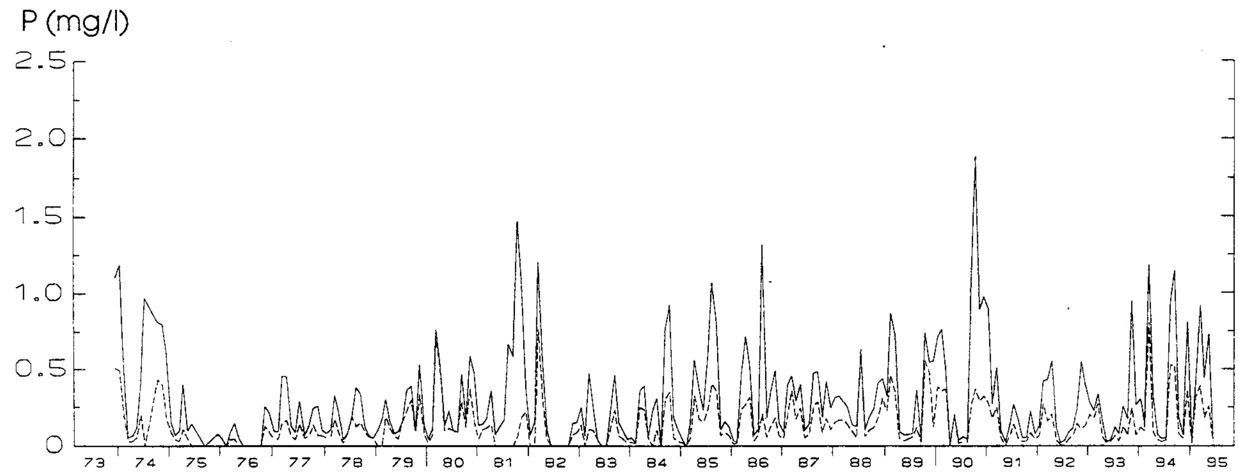

Fig. 1. TotP- (solid line) and dissolved PO_4 P- (broken line) monthly mean concentrations of drainage water (monthly transport divided by montly discharge).

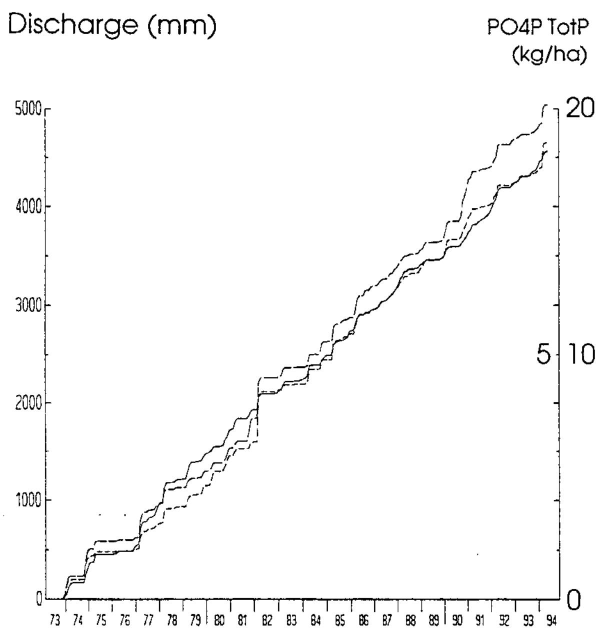

Fig. 2. Accumulated monthly discharge (solid line) and transport of total P (long broken line) and dissolved PO_4P (small broken line).

Results

Field studies

Phosphorus concentrations in drainage water (Fig. 1) from the observation field have consistently exceeded the upper limit that the Swedish National Board /2/ desires from a limnological point of view, namely 0.05 mg totP/l. Although this is often the case for clay soils in the eastern part of Sweden, concentrations in water draining from the observation field were unusually high. One reason for this could be that as a result of the hilly topography and dense clay, much of the surface water runs into the drainage system. In the upper parts of the field there is till in the soil surface and groundwater contributes to the flow of drainage water. In most cases, the dynamics of accumulated phosphorus transport followed that of the accumulated amount of discharge (Fig. 2). Phosphorus transport increased markedly relative to discharge during two periods: relative increases in both totP- and dissolved PO_4P-transport were observed in spring 1982, and a relative increase in the transport of totP minus PO_4P occurred during winter 90/91.

Nutrient-flow dynamics during two periods, one before the conversion over to alternative agriculture and the other afterwards, are illustrated in more detail in Figure 3. In autumn 1980, a three-year-old ley was terminated by ploughing. This was a good termination done in late November. Neither the phosphorus nor the nitrogen concentrations rose much. In 81/82 manure was added twice. The manure was tilled into the soil in autumn and spread in late winter (5 March). Phosphorus concentrations remained high for a long time after the second fertilization during the snowmelt period in 1982.

The alternative-farming period started with the termination of a ley. The harvest was complete by August. Thereafter, manure was applied and winter wheat was sown within two weeks. As a result of the manure application, there was so much organic nitrogen that not all of it could be mineralized. Consequently, unusually high nitrogen concentrations were measured in drainage water that year as well during the following year (Fig. 3).

Relatively high phosphorus concentrations were registered during 89/90, and they were even higher by 90/91. The autumn wheat stubble was ploughed under in early October 1990, and three days later, after 36 mm of precipitation had fallen, extremely high concentrations of suspended material (5000 mg/l) were observed together with very high phosphorous concentrations (2.7 mg/l). Most of the phosphorus (85%) was in forms other than dissolved PO_4P. In 1991, a spring-spread manure harrowed into the soil did not affect nutrient concentrations in the drainage water.

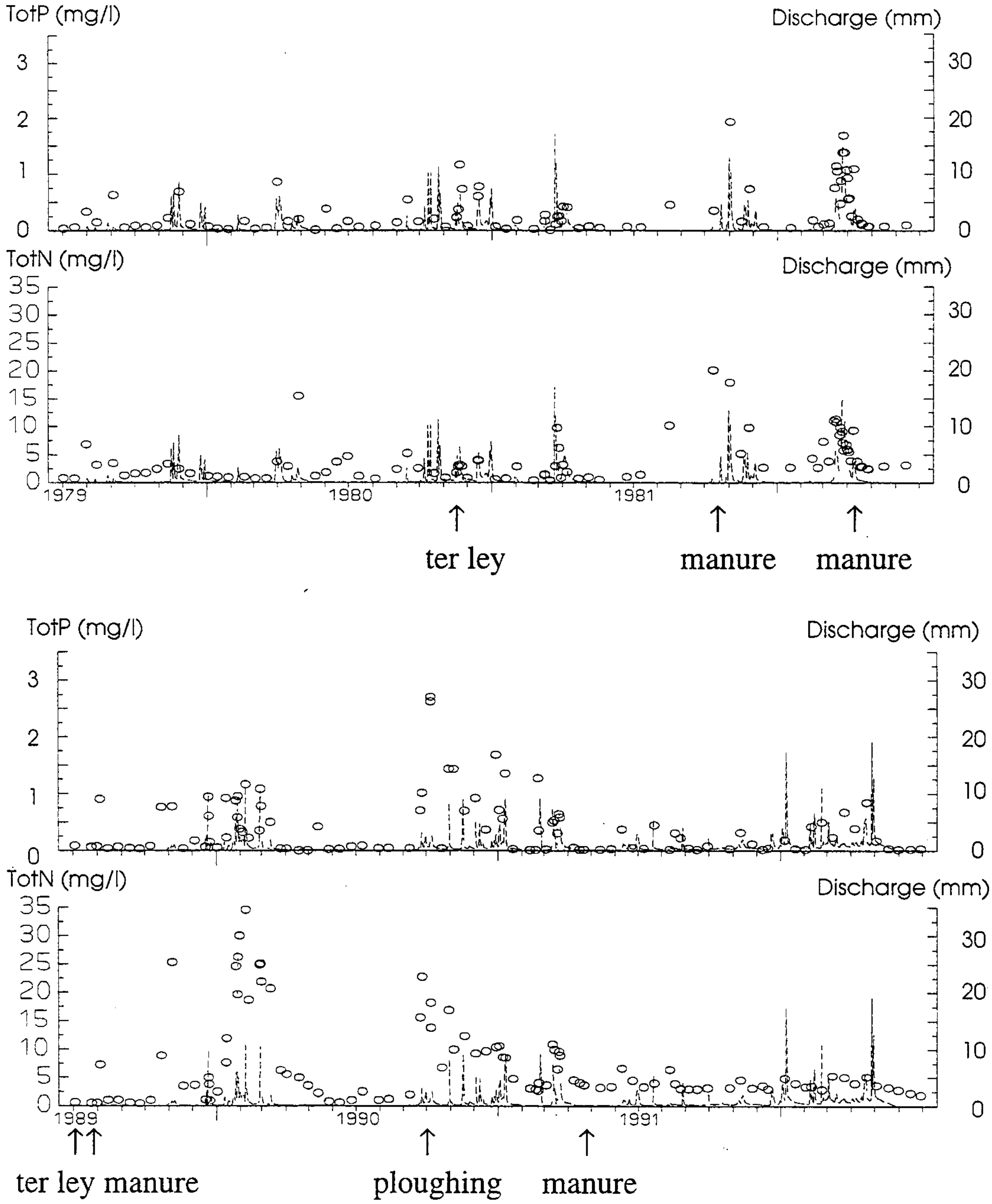

Fig. 3. Total phosphorus and total nitrogen concentrations and daily discharge in connection to some managements. Ter ley = termination of ley.

Losses of phosphorus from the sandy soil were low during the second ley year, by which time the grass had become fully developed. The fertilized sandy soil had already established a well-developed grass cover by the middle of the first growing season. Phosphorus losses from both types of sandy soils were highest during the wet year of 94/95. Total phosphorus losses were small from the humus soil. Most of the phosphorus was in a form other than phosphate and was probably bound in organic compounds.

Phosphorus losses from the clay soil averaged around 0.2 kg/ha · year, and most of the phosphorus was in a form other than phosphate. The high losses occurred simultaneously with high concentrations of suspended material. Phosphorus losses from moderately fertilized sandy soil varied in spite of the moderate phosphorus status (Table 1) of the soil. From the very phosphorus-rich fine sand, losses were extremely high. As was the case for the other sandy soil, practically all phosphorus was in the form of phosphate.

The highly fertilised soil was the only soil from which removal by crops exceeded the supply of fertilizer-P. The other soils had positive phosphorus balances. The P-AL content of the sandy soils in early summer increased in the sandy soils, but decreased somewhat in the other soils. Changes in the total phosphorus contents of the soils over the 4-year experimental period were minimal (Table 1). Phosphorus losses from "alternative" lysimeters were sometimes larger and sometimes smaller than those from "conventional" lysimeters (Table 3) and no consistent differences were found between the two. By contrast, differences were very pronounced between the different soil types (Fig. 5).

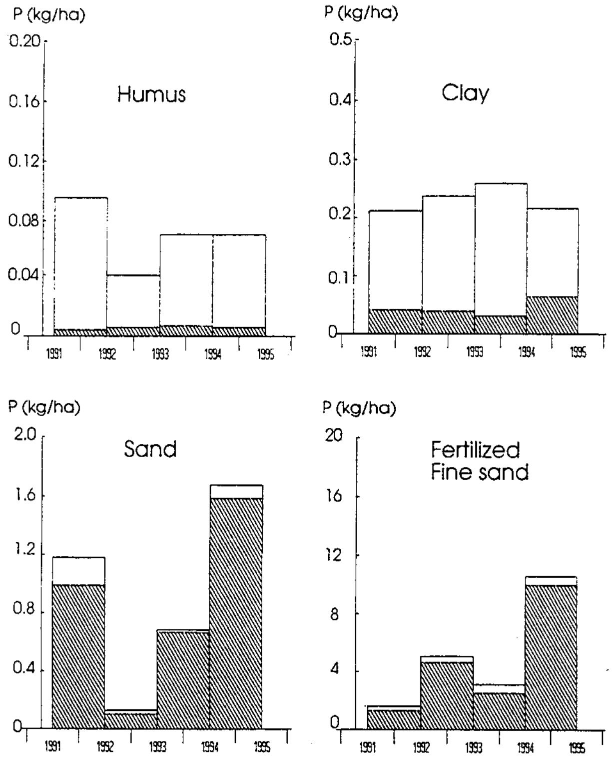

Fig. 4. Drainage losses of total P (entire bar) and PO_4P (filled bars) of different soil types in lysimeters.

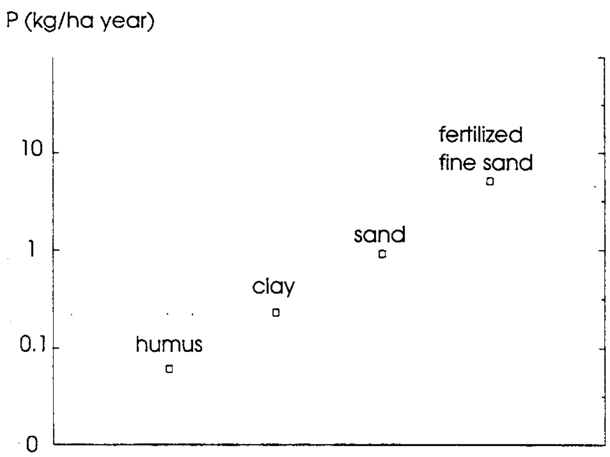

Fig. 5. Mean yearly leaching of total phosphorus during four years through different soil types (logarithmic scale).

Table 3. Mean values of total phosphorus transport in conventional (con) and alternative (alt) cultured lysimeters in kg/ha together with standard deviation (s) for all lysimeters representing a certain soil type.

Year	93/94	93/94	93/94	94/95	94/95	94/95
	con	alt	s	con	alt	s
Humus	0.07	0.07	0.01	0.19	0.07	0.10
Clay	0.25	0.27	0.02	0.77	0.16	0.38
Sand	0.61	0.75	0.29	1.84	1.51	0.42
Fertilized sand	3.42	2.95	0.46	9.12	11.26	1.71

Discussion and summary

Phosphorus losses from the studied field were unusually high on two occasions, the first in connection with the winter spreading of manure and the second after a conventional autumn ploughing in connection with very unfavourable weather conditions. Other management practices, such as covering the soil with a ley, had no effect on phosphorus losses. In connection with a conversion over to alternative agriculture it may be necessary to terminate many leys. Their termination, especially in conjunction with manure applications, can result in unacceptably high nitrogen losses. Mean phosphorus losses from the studied field were 1.00 kg totP and 0.47 kg PO_4P per ha and year which are high for Sweden. The P loss through free drainage in the lysimeter from the same soil type was lower (0.25 kg/ha). This discrepancy suggests that there either was a strong groundwater influence or that drainage losses in the lysimeters were under-estimated. The macropore structure of the 7-dm^2-wide lysimeters may have been less developed than that of the natural soil.

Transport of phosphorus from the observation field with sandy, fertilized soils reached 2.6 kg/ha · year /5/. Phosphorus leaching from the lysimeters showed no tendency to decrease, but the phosphorus concentration was still very high after four years of moderate fertilization. Therefore, compared with the type of production system, soil type and the fertilization history of the soil seem to be much more important as factors determining the magnitude of phosphorus leaching, at least in the short term.

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Transport of sediment and phosphorus in the arable Gelbæk catchment, Denmark: I. Transport through the soil in irrigated plots

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Abstract

Sediment and phosphorus loss to drainage water was studied in two small experimental field plots on structured loamy soil in November 1994 and March 1995. Irrigation of the plots varied between 25.2 and 37.3 mm and lasted 2-3 hours. The drainage flow response was rapid indicating the existence of preferential flow pathways in the soil. The concentration of particulate matter (PM) and particulate phosphorus (PP) was highest in the initial drainage flow (63-334 mg l^{-1} and 0.177-0.876 mg P l^{-1}, respectively).

PM and PP concentrations both decreased during the rising phase of the hydrograph, and continued to decrease after peak runoff had been reached. The PP loss coefficient at the two plots was 1.02 and 0.57 g P ha^{-1} mm^{-1} during the rapid-flow response in the November experiments, as compared to 0.45 and 0.41 g P ha^{-1} mm^{-1} in the March experiments. Sediment median particle size ranged from 2-5 μm in the majority of the drainage water samples from the March experiments. Tracer studies indicated that the particles originated from the topsoil.

Introduction

The movement through the soil to drainage systems of particles and particle-associated substances such as phosphorus and pesticides can pose a threat to the aquatic environment. This pathway has traditionally received little attention as surface erosion is generally regarded as being more important, especially in the case of loamy or clayey soils /e.g. 1/. Several studies indicate that the loss of particulate phosphorus via drainage systems is of particular importance /2,3,4/, it being known that eutrophication of surface waters in Denmark and other European countries is often caused by excess P loading /5/.

Clay translocation from upper to lower soil horizons is well known in Denmark and in other temperate regions, especially in loamy and clayey soils /6,7/. Translocation of mixed clay, silt and organic matter can also occur on agricultural land /8/. Preferential migration of fine particles via soil macropores has been reported for Danish till /9/, and has also been observed during precipitation events /10,2/. However, there have been few reports of fine particle migration based on field plot experiments /e.g. 11/.

The present paper focuses on the transport of particulate matter and particulate phosphorus through structured loamy soil to drainage water in two small field plots located within a larger tile-drained catchment area.

Study area

The study was undertaken at a tile-drained field located in the catchment area of Gelbæk stream in eastern Jutland. The field is comprised of clayey till deposited during the late Weichsel glacial period (approx. 18,000 B.P.). The soil is a Typic Hapludalf of sandy loam to sandy clay loam texture /12/. Soil structure is well developed with very coarse prismatic structure in the deeper soil horizons. Local interfingering of albic soil material takes place into the underlying clay accumulation horizon. Tile drains were installed in the field at a depth of approx. 1 m between 1944 and 1950.

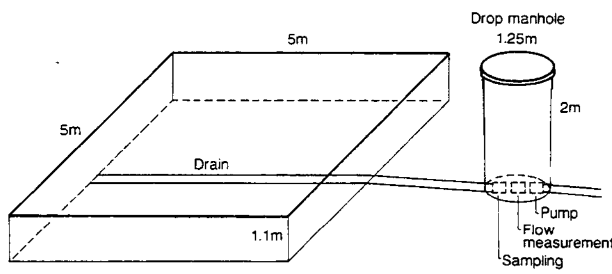

Fig. 1. Diagram of a field plot indicating the vertical plastic sheeting, the drain, and the drop manhole with sampling equipment. A pump leads the water from the drop manhole into the main tile drainage system.

Two experimental plots (5 x 5 m) were established on the soil in 1993 and sown with grass in April 1994. Each plot was isolated from the surrounding soil by vertical plastic sheeting penetrating to a depth of 1.1 m (Fig. 1). Both were located in the upper part of a drainage catchment that has been monitored since May 1993. The plots are representative of the upper part of this drainage catchment, whereas the lower part near the stream is comprised of fluvial sand and clay and is used for grazing cattle. The soil type at the field plots is common throughout the Gelbæk stream catchment, however.

Methods

Four experiments were conducted in all, one at each plot in November 1994 and one in March 1995. Precipitation was mimicked by irrigation, the plots being protected by rain shelters. Approximately 3-4 days prior to each experiment the plots were irrigated to moisten the soil. At the onset of each experiment the volumetric soil water content was approx. 25% in the topsoil and 30% in the subsoil; drainage flow was very low. The plots were then drip irrigated for 2-3 h (Table 1) using a perforated water hose array, the drip height being 30 cm and the distance between drip holes approx. 10 cm. The groundwater table was measured by piezometers.

Drainage outflow was measured automatically using a liquid level sampler actuator connected to an ISCO sampler placed in a drop manhole between the plots and the stream (Fig. 1). Water samples were collected manually from the drain pipe outlet in the drop manhole and analyzed for particulate matter (PM), total P (TP) and total dissolved P (DP), with particulate P (PP) being determined as TP minus DP. Particle size distribution in the water samples collected during the March 1995 experiments was measured using a LUMOSED photo-sedimentometer /13/ on sieved samples (63 μm mesh) dispersed using $Na_4P_2O_7$ and 15 min ultrasonic treatment. The point of separation of the rapid-flow response and the slower flow response was identified on the hydrograph by means of standard hydrological methods /e.g. 14/.

Results

PM and PP concentrations were found to be linearly related (Fig. 2). The P content of the particulate matter was similar in the March experiments (0.22% and 0.34%) and November experiments (0.28% at both plots). DP was not measured in all March samples, however, and the missing PP concentrations were therefore calculated as a fixed percentage of the PM concentration. During storm events the P content of particulate matter in the water draining from the whole drainage catchment was even higher than at the field plots (0.66%; calculated on the basis of nine pooled storm flow events during winter 1993-94).

In each experiment peak flow was reached 2.8 to 3.0 hours after the start of irrigation (Fig. 3), thus indicating very rapid flow down to the drains via macropores, as was also observed during an experiment where $CaCl_2$ was spread on the plots /15/.

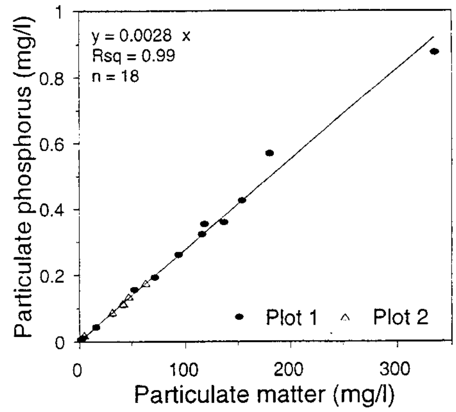

Fig. 2. Relationship between particulate phosphorus (PP) and particulate matter (PM) concentrations in drainage water from the two experimental plots in November 1994.

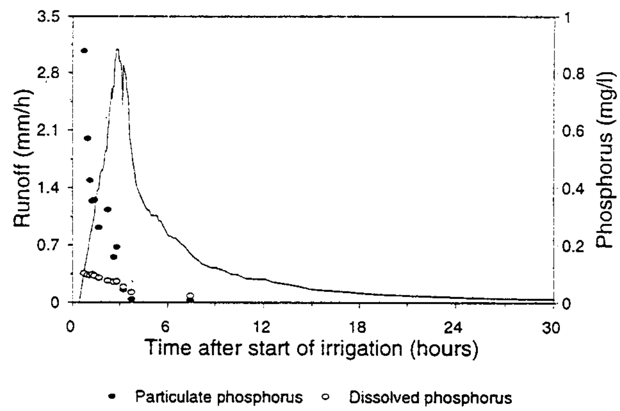

Fig. 3. Runoff, particulate phosphorus and dissolved phosphorus concentrations during the November 1994 experiment at plot 1.

Since PM and PP concentrations are linearly related, the event sensitivity of PM and PP fluxes through soil to drainage water can be revealed by focusing on just one of the two. PM and PP concentrations were highest in the first sample collected in each experiment (63-334 mg l^{-1} and 0.177-0.876 mg P l^{-1}, respectively), thereafter decreasing during the rising phase of the hydrograph as well as after peak runoff had been reached (Fig. 3). The relationship between PP (or PM) and runoff forms a clockwise hysteresis response. DP concentrations were generally much lower (0.014-0.103 mg/l), and also decreased with time. PP amounted to 67-82% of TP during the rising phase of the hydrograph in the four experiments as compared with 28-73% during the falling phase.

Irrigation in the four experiments varied between 25.2 and 37.3 mm, with runoff during the first 24 h period varying between 3.7 and 23.4 mm (Table 1). In the March experiments the plots were equally irrigated and flow responses were very similar; however, runoff differed markedly (23.4 mm d^{-1} vs 9.5 mm d^{-1}).

Table 1. Irrigation, irrigation time, runoff, PM and PP loss coefficients from four field plot experiments carried out in November 1994 and March 1995.

		Irrigation	Irrigation time	Runoff	PM loss coefficient	PP loss coefficient
		mm	min	mm d^{-1}	g ha^{-1} mm^{-1}	g P ha^{-1} mm^{-1}
Nov 94.	Plot 1	35.4	165	17.9	365	1.02
	Plot 2	25.2	135	3.7	196	0.57
Mar 95.	Plot 1	37.3	170	23.4	204	0.45
	Plot 2	36.9	180	9.5	122	0.41

It is therefore likely that in plot 2 part of the water escaped under the plastic sheeting and away from the drain pipe. The loss coefficient for PM and PP per unit runoff calculated for the rapid-flow response is shown in Table 1. That the loss coefficients for plot 1 in the November experiments were nearly two-fold greater than those for plot 2 is probably attributable to the difference in irrigation (Table 1). With each plot the loss coefficients for PM and PP were higher in the November experiments than in the March experiments.

Particle size analysis of the March samples revealed that in all samples the median particle size was less than 5 μm, and usually in the range 2-5 μm. No particles greater than 63 μm were present.

A preliminary tracer study indicates that particulate matter in the drainage water from the drainage catchment as a whole originates from the topsoil /18/. We therefore believe that soil structure, in particular macropores, has an important influence on PM and PP transport from the topsoil to the drainage water from artificially drained areas of the Gelbæk stream catchment. The process of fine particle translocation from upper soil horizons to tile drains in the deeper soil might be similar in nature to the processes that take place during the development of agric and glossic soil horizons. In future experiments we intend using methods such as irrigation with water containing colloids to further evaluate the mechanisms and time frame of particulate matter transport within the soil.

Discussion

The decrease seen in the concentration of PM and PP during the rising phase of the hydrograph and after peak runoff had been reached is explicable by a decrease in the availability of fine particulate matter in the macropores or in the tile drain itself. Similar exhaustion of PM has been documented for the drainage catchment as a whole during an early autumn storm event /3/, as well as in streams during storm events /16,17/. A similar but seasonal decrease might account for the higher PM and PP loss coefficients seen in the November experiments than in the March experiments.

That there was a good linear relationship between PM and PP concentrations at each plot indicates particulate matter homogeneity, as was confirmed by the particle size analysis. The higher P content of particulate matter in the drainage water from the drainage catchment as a whole compared to that from the field plots is probably due to the difference in soil type and land usage in the lower part of the drainage catchment.

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Transport of sediment and phosphorus in the arable Gelbæk catchment, Denmark: II. Drainage water

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Abstract

Intensive measurements of dissolved phosphorus (DP) and particulate phosphorus (PP) from two artificially drained catchments on sandy loam in Gelbæk were conducted during a one year period. Annual total phosphorus (TP) loss amounted to 0.098 and 0.627 kg P ha^{-1}, respectively, from the two catchments, the highest loss occurring from the area which received high inputs of manure and where parts of the catchment were low-lying riparian areas with a permanently high groundwater table. The major part of TP loss was accounted for by DP in the case of the catchment with the highest loss (71%) and by PP at the other catchment (55%).

Annual loss of PP amounted to 0.054 and 0.182 kg P ha^{-1} for the two catchments, respectively. Significant linear relationships were established between the concentrations of PP and particulate matter (PM). Evidence was found indicating exhaustion of PM availability during storm events, either in the soil macropores or in the drain itself. A study using a tracer technique showed that PM in drainage water most likely originates from the topsoil, and is transported to drains via macropores.

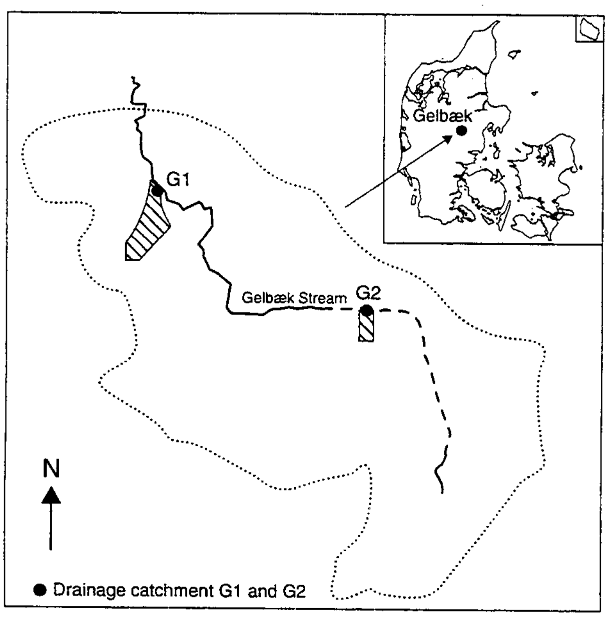

Fig. 1 Map of Denmark showing the location and a diagram of the Gelbæk stream catchment indicating drainage catchments G1 and G2.

Introduction

Eutrophication of many shallow surface waters in Denmark is controlled by diffuse loss of phosphorus (P) from arable land, and efforts to abate diffuse loading need to be enhanced /1/.

Over the last decade attention has focused on soil erosion as the major source of diffuse P loss to the fresh water environment as well as on the development of appropiate management strategies dealing with the problem /2, 3/. However, downwashing of P through macropores in clayey soils may also contribute to P loss via subsurface drainage systems /3/. Furthermore, phosphorus transport in the soil has been found to be extremely precipitation sensitive /4/. In Denmark, however, previous studies have been based on infrequent sampling, the available data is therefore inadequate for determining the impact on P losses of factors such as climatic conditions, drainage conditions, agricultural practice and fertilization practice.

The aim of the present study is therefore to accurately describe and quantify the seasonal and short term variation in diffuse P loss from artificially drained arable catchments. The contribution of dissolved P (DP) and particulate P (PP) to P transport in drainage water is quantified.

Study area

The study was undertaken at two artificially drained catchments on structured moraine soils, G1 (13.3 ha) and G2 (4.4 ha), located in the catchment area of Gelbæk stream in Eastern Jutland (Fig.1). Soil texture is mainly sandy loam /5/. The drains are comprised of clay tiles. Distance between drains is about 20 m and mean drainage depth is mostly between 1.2 and 1.4 m, but 0.8 m at the low-lying part of catchment G1 near the stream. The ground water table fluctuates during the year, that at catchment G2 often being more than 3-4 m below the soil surface during summer and from 0.3 m down to drainage depth during winter. Conditions are similar at catchment G1, except that the lower part of the catchment has a permanently high groundwater table. Both drainage catchments are intensively farmed, G2 being cultivated with rotational crops while G1 is largely comprised of rotational crops but with permanent grass used for grazing in the low-lying part (15%). Animal manure and slurry are regularly applied to both catchments. Further descriptions are given in /6/.

Methods

Field sampling was carried out between May 1993 and April 1994. One monitoring station was established at each drainage catchment. This consisted of a V-notch sharp crested weir in a drop manhole at the outlet of the drainage network. The station was equipped with automatic ISCO-samplers and datalogger for continuous logging of water level. Sampling was carried out according to two strategies: 1) hourly sampling pooled weekly and 2) storm flow sampling (24 samples per 24 hours). Storm sampling was more frequent during rising stage than during falling stage. The water samples were analysed for total P (TP), total dissolved P (TDP) and particulate matter (PM) as descibed in /6/. Particulate P (PP) was determined as TP - TDP.

Whether PM lost to drainage water originates from the topsoil or the subsoil can be determined using a tracer technique developed for identifying and quantifying soil erosion rates with Cs-137 as the tracer /7/. Source material was collected from three depths of one soil profile within drainage catchment G1 and PM in drainage water was collected during one major storm event; all samples were analysed for Cs-137 according to the method described by Kronvang et al. /8/.

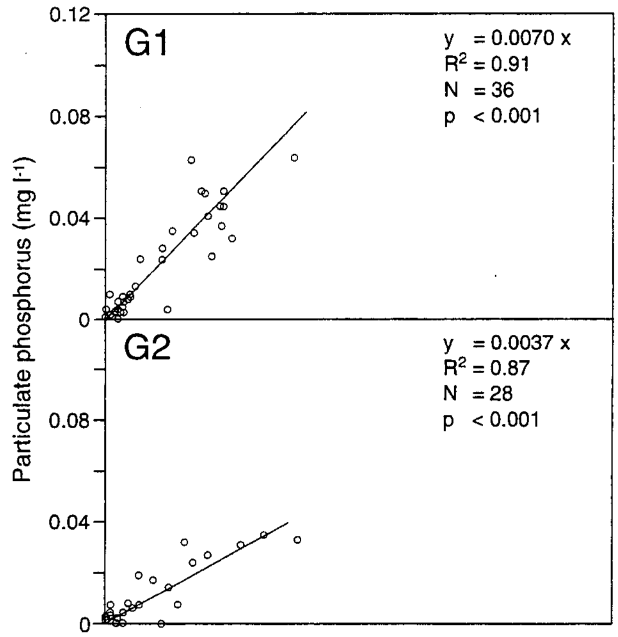

Fig. 2. Relationship between pooled weekly concentrations of particulate phosphorus and particulate matter in drainage water for the two drainage catchments.

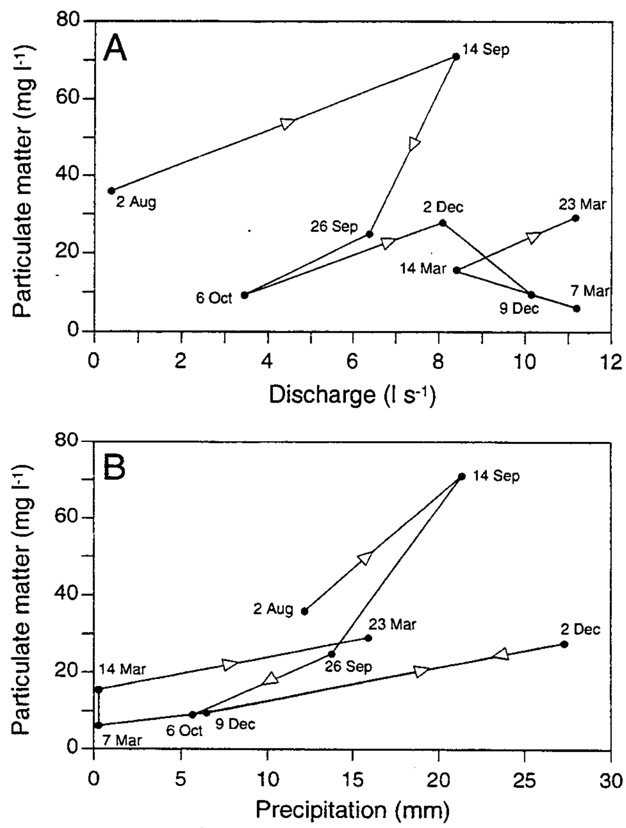

Fig. 3. Pooled storm flow concentrations of particulate matter in drainage water from drainage catchment G1 versus discharge (A) and precipitation (B) during nine storm events in the period May 1993 to April 1994 (snow melting occurred on 14 and 23 March 1994).

Results and discussion

Hydrology

Annual precipitation during the study year was 1,087 mm which was considerably more than the long-term average for the period 1961-90 (893 mm). Annual runoff amounted to 510 mm at catchment G1 and to 305 mm at catchment G2 (Table 1). The higher runoff at G1 is primarily attributable to the main discharge period being longer, although groundwater flow from the low-lying area also played a role at G1.

Table 1. Annual runoff, average concentrations and loss of dissolved and particulate P from the two drainage catchments during the period May 1993 to April 1994.

	Catchment	
	G1	G2
Runoff, mm	510	305
Concentrations:		
Dissolved P, mg l^{-1}	0.079	0.010
Particulate P, mg l^{-1}	0.028	0.012
Loss:		
Dissolved P, kg ha^{-1}	0.445	0.044
Particulate P, kg ha^{-1}	0.182	0.054

P loss from arable land
Annual TP loss amounted to 0.627 kg P ha^{-1} at G1 and 0.098 kg P ha^{-1} at G2. The higher P loss at G1 is attributable partly to the higher runoff and partly to a higher concentration level (Table 1). High P concentrations in catchment G1 may be attributable to the permanently high groundwater table in the low-lying part of the catchemnt, it being known that extended periods under reducing conditions increase the susceptibility to P movement in the soil /9/. Also high inputs of manure from grazing cattle may have caused increased concentrations of especially DP in drainage water.

The major part of TP loss was accounted for by DP (71%) at catchment G1, but by PP (55%) at catchment G2. The values for P fractions reported in the literature also vary. Thus, for moderately to highly fertilized Ontario clay soils it was found that DP accounted for on average 66% of the P content of drainage water and concentrations of DP being high /10/. In contrast most other studies of drainage water report much larger proportions of PP similar to what we found for catchment G2 /11/. Common for these studies were the lower P concentrations of drainage water. It seems that at the lower concentrations P is strongly bound to soil particles and hence mainly transported as PP.

Mechanisms of particulate phosphorus transport in the soil
Loss of PP varied throughout the year and occurred mainly during storm events /6/. Annual loss of PP amounted to 0.182 kg P ha^{-1} at catchment G1 and 0.054 kg P ha^{-1} at catchment G2. Significant linear relationships were established between concentrations of PP and PM for both catchments (Fig. 2), the PM being more enriched with P in drainage water from catchment G1 than from G2 (Fig. 2).

The episodic transport of PP through soils to drains has been interpreted by studying trends in PM concentrations. The PM concentrations were highest (35-70 mg l^{-1}) during the first two storm events, thereafter decreasing to a level between 6 and 30 mg l^{-1} (Fig. 3). The relationships between PM concentration and discharge during the first four storm events revealed a simple clockwise loop response (Fig. 3A). This pattern can be explained by an exhaustion of PM accumulated in macropores or drain pipes during the summer. Similar trends have been found for PM concentrations in streams /12/. Following these events, the average PM concentration was low and linearly related to precipitation, this probably being attributable to the fact that eroded fine particles from the soil are transported to the drain by preferential flow through macropores (Fig. 3B). That the same pattern was not found with respect to PM concentration and discharge is probably due to a general increase in baseflow during winter.

The tracer study showed that the topsoil had a much higher concentration of Cs-137 than the subsoil, and that PM in drains had a concentration level similar to that of the topsoil (Table 2). This indicates that PM lost to drainage water most likely originates from the topsoil and is transported to drains through macropores.

Table 2. Tracer analysis of source material (topsoil and subsoil) and particulate matter in drainage water collected during a storm flow in January 1995 at catchment G1. Soil sample represents particle fraction < 0.02 mm.

	Cs-137
Topsoil 12-20 cm	19.8
Subsoil 36-44 cm	<1.5
Subsoil 110-118 cm	<1.5
Drainage water particulate matter	24.2

Conclusion

The average annual TP loss of 0.368 kg P ha^{-1} for the two drainage catchments during the wet study year was similar to the average TP loss of 0.34 kg P ha^{-1} reported for three Danish streams draining small loamy arable catchments /13/. Since loamy soils cover almost 32% of the Danish arable area, and since more than 50% of this is artificially drained, P loss from artificially drained loamy soils represents an important source of diffuse P loss to the Danish aquatic environment.

Our experiment indicates that fine particles are transported from the topsoil through macropores to drainage water; thus giving rise to P rich PM in drainage water.

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Trends in phosphorus concentrations in stream Gelbæk

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Abstract

Concentrations of phosphorus have been measured since 1974 in Gelbæk stream, a small tributary to Gjern Å. Gelbæk stream drains an intensively farmed catchment area with no large point sources.

The time series of phosphorus concentrations were analysed for trends by Kendall's seasonal test, which is a nonparametric statistical method for testing monotonic trends in water quality. Slope of trend, given as change per year, was estimated using a nonparametric estimator.

The hydrograph was divided into two strata. Stormflow which are periods with increasing discharge due to heavy precipitation or snow melting, and baseflow which are periods with stable or decreasing discharge. Time series of phosphorus concentrations were divided into two series according to whether sampling took place during storm- or baseflow.

Kendall's seasonal test detected significantly decreasing phosphorus concentrations during both stormflow and baseflow conditions.

Introduction

Phosphorus pollution of Danish streams and lakes is derived from both point and non-point sources. The phosphorus load to freshwater from point sources has strongly decreased during the last two decades /1/. The diffuse load has consequently increased in importance.

Despite the reductions in the phosphorus load from point sources, the water quality of Danish lakes has not generally improved /2/. Improved knowledge on how to control the load from non-point sources is therefore of great importance.

This paper investigates the trend in total phosphorus concentrations during the twentyone-year period (1974-94) in a small Danish lowland stream without large point sources. A considerable proportion of the annual total phosphorus transport in the Gelbæk stream is sediment-associated and connected with short storm events /3/.

Study Area

The catchment area of stream Gelbæk is located in the eastern part of Central Jutland, Denmark. The catchment area is small (11.6 km^2) and is intensively farmed. Approx. 98% of the catchment is agricultural land. The only point sources in the catchment area are scattered dwellings, amounting to about 150 person equivalents (PE). Deposits from the Weichsel glaciation cover the catchment and soils consist of 44% sandy loam and 55% loam /4/. The slope of the Gelbæk stream is between 1-10‰ and the stream is piped and strongly channelled in its upper parts /3/.

Methods

Discharge and total phosphorus concentrations have been measured since 1974. Water samples were collected weekly to monthly by simple dip sampling. The concentration of total phosphorus and other variables were measured in the laboratory. Neither field nor laboratory procedures have been changed since 1974.

Both discharge (Q) and water stage (H) have been measured regularly throughout the period. Daily means of Q were estimated from the relationship $Q=\alpha H^{\beta}$. The seasonal development of submerged macrophytes were taken into consideration in the estimation of α and β.

Hydrograph separation

The hydrograph was separated into two time strata. The first includes the periods with increased discharge due to heavy precipitation or snow melting. The strata is called stormflow. The second stratum includes periods with stable or decreasing discharge conditions (baseflow).

Separation was done in accordance with the following rules. If discharge increases with more than 5 l s^{-1} from one day to the other then a storm-flow period begins. During stormflow the discharge from the latest day with baseflow is increased by 2% per day. A stormflow period ends when this increased baseflow is equal or greater than the actual discharge.

The time series of phosphorus concentrations were divided into two disjoint time series according to whether sampling was done during storm- or baseflow. During stormflow periods phosphorus is mainly delivered to the stream from soil erosion, surface runoff, drainage tubes, instream erosion, and to a smaller degree from groundwater and point sources. The main part of the phosphorus is sediment-associated during stormflow periods. During baseflow periods phosphorus is delivered from groundwater, subsurface runoff and point sources.

Statistical analysis
The statistical analysis consisted of both exploratory and confirmatory methods. Time series plots and Box-Whisker plots /5/ of the collected data were studied as a first step of data analysis. The exploratory methods help to identify seasonality and possible trends. The trends and relationsships seen in the plots were tested as a second step of analysis. Trends were tested by applying Kendall's seasonal test for monotonic trends /6,7/. This nonparametric test is robust with respect to missing values, serially correlated measurements, seasonality and measurements below detection limits. Traditional statistical methods, such as ordinary regression can be seriously misleading when data are serially correlated, which is usually the case when measurements are taken closely in time.

Kendall's seasonal trend test operates on average seasonal values. Typically, season is taken to be month, but also weekly, biweekly or quarter-yearly seasons can be applied. Ttest statistics are then calculated for each season and then combined to an overall test statistic for the time series.

If a time series shows a significant trend, the change per unit time is estimated by the non-parametric Kendall's seasonal slope estimator /6/. The trend is implicitely assumed to be linear by using this estimator.

Results

The complete time series of average monthly total phosphorus concentrations is shown in Figure 1 together with Box-Whisker plots of average monthly concentrations by year and month. A decreasing trend can be seen in the plot of the time series and to a clearer extent in the Box-Whisker plot. The decrease in average monthly phosphorus concentration is mainly occurring after 1980. After 1986 the monthly phosphorus concentration is more stable and without many high peaks. The monthly phosphorus concentration shows a weak seasonality with higher concentrations during the warm and dry months (May-September) than during the winter months (Fig. 2C). From 1974-94 the complete time series decreases significantly (P=0.01%). The annual decrease is estimated to be 0.012 mg P l^{-1}.

The monthly concentration during stormflow periods seems to decrease in the study period (Fig. 2). Samples with very high concentrations are fewer after 1986 (Fig. 2A).

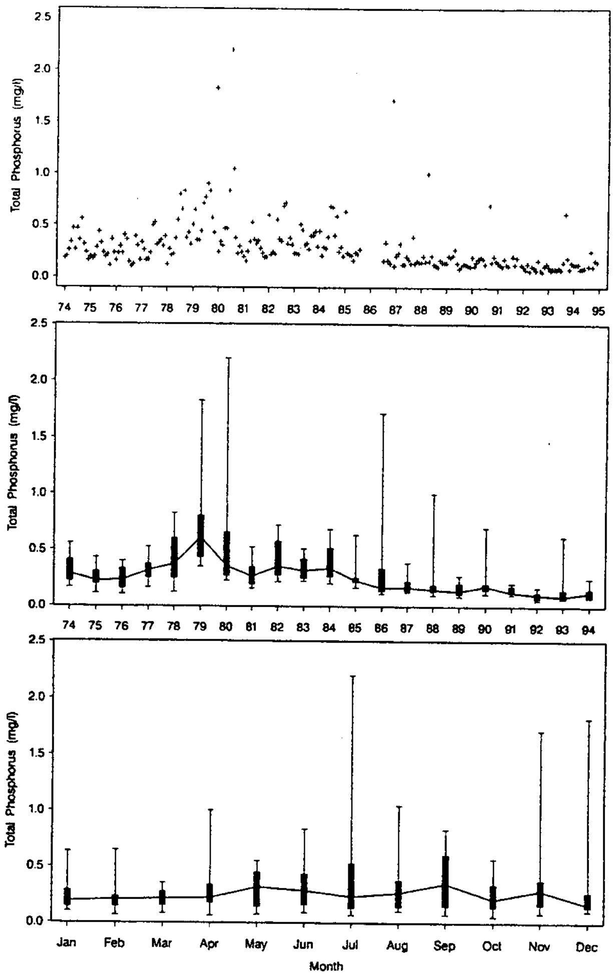

Fig. 1. Time series plot (A), Box-Whisker plot per year (B) and Box-Whisker plot per month (C) for average montly concentrations of total phosphorus.

The stormflow phosphorus concentrations reveal the same seasonality as for the complete time series (Fig. 2C)..The time series decreases significantly throughout the period (P=0.03%) and the slope is estimated to be -0.008 mg P l^{-1}.

The baseflow concentrations show an even more pronounced decrease than the storm-flow concentrations (Fig. 3). The baseflow time series reveals the same seasonality (Fig. 3C) and the decrease happens mainly after 1980 (Fig. 3B). A significant decrease is found (P<0.01%) with an estimated change per year of 0.015 mg P l^{-1}.

Discussion and conclusions

All three time series of total phosphorus showed a downward trend. The strongest decrease in monthly phosphorus concentration was found during baseflow periods giving a total of 0.315 mg P l^{-1} during the period 1974-94.

In comparison stormflow concentrations decreased 0.168 mg l^{-1} during the study period. It can be concluded that erosional sources of phosphorus showed a smaller decrease than other phosphorus sources.

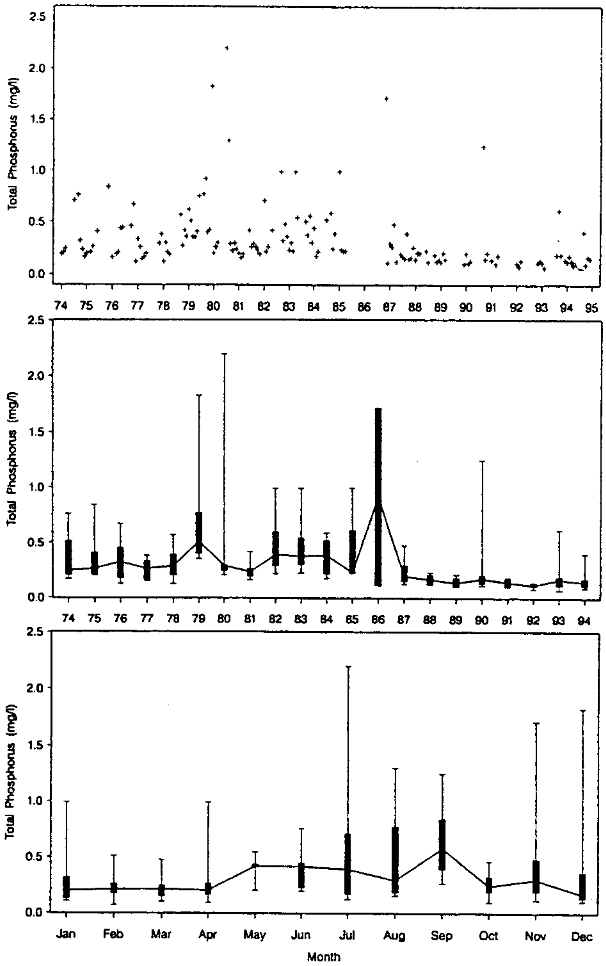

Fig. 2. Time series plot (A), Box-Whisker plot per year (B) and Box-Whisker plot per month (C) for average monthly concentrations of total phosphorus during baseflow.

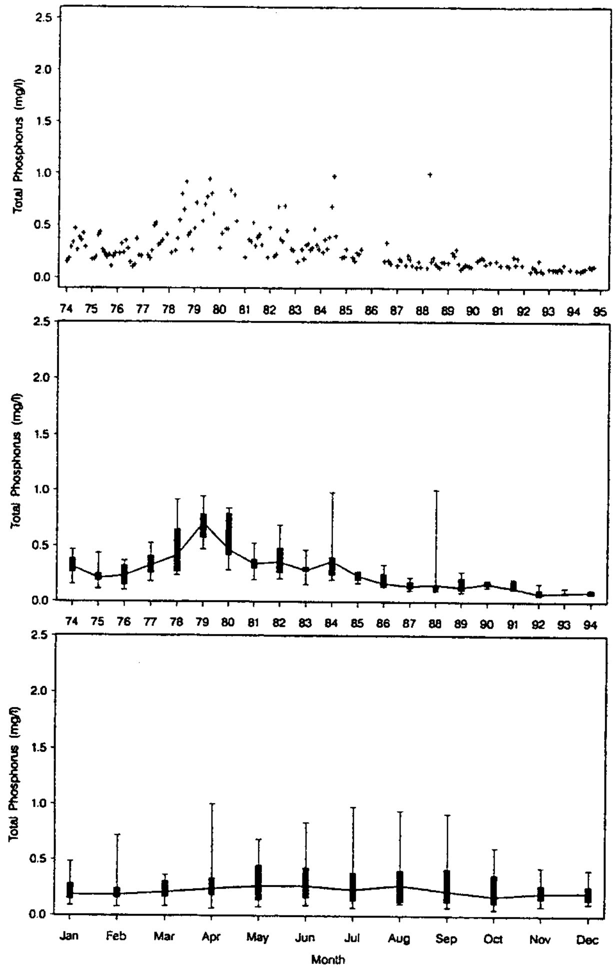

Fig. 3. Time series plot (A), Box-Whisker plot per year (B) and Box-Whisker plot per month (C) for average montly concentrations of total phosphorus during baseflow.

Time series of the discharge at exactly the same dates, when water samples were taken, were also analysed for trends. Neither the complete time series of discharge nor the disjoint time series of discharge during stormflow or baseflow periods showed any trends. Therefore, changes in discharge cannot explain the trends in observed phosphorus concentrations.

The observed decrease in phosphorus concentrations after 1986 could be due to waste water from the small villages in the catchment area being disconnected from the stream in 1986/87. Increased usage of washing detergents without phosphate during the last decade, e.g. decreased load from scattered dwellings, is also a possible explanation for some of the observed trends. The phosphorus load from 1 PE has decreased from 4 g P d^{-1} to 2 g P d^{-1} since 1980 /8/. If it is assumed that all the phosphorus load from the 150 PE reaches the stream, the total decrease amounts to 0.174 mg l^{-1} in the period 1980-94.

Finally, the load of phosphorus from farms in the catchment has possibly decreased during the study period due to the building of storage facilities for manure.

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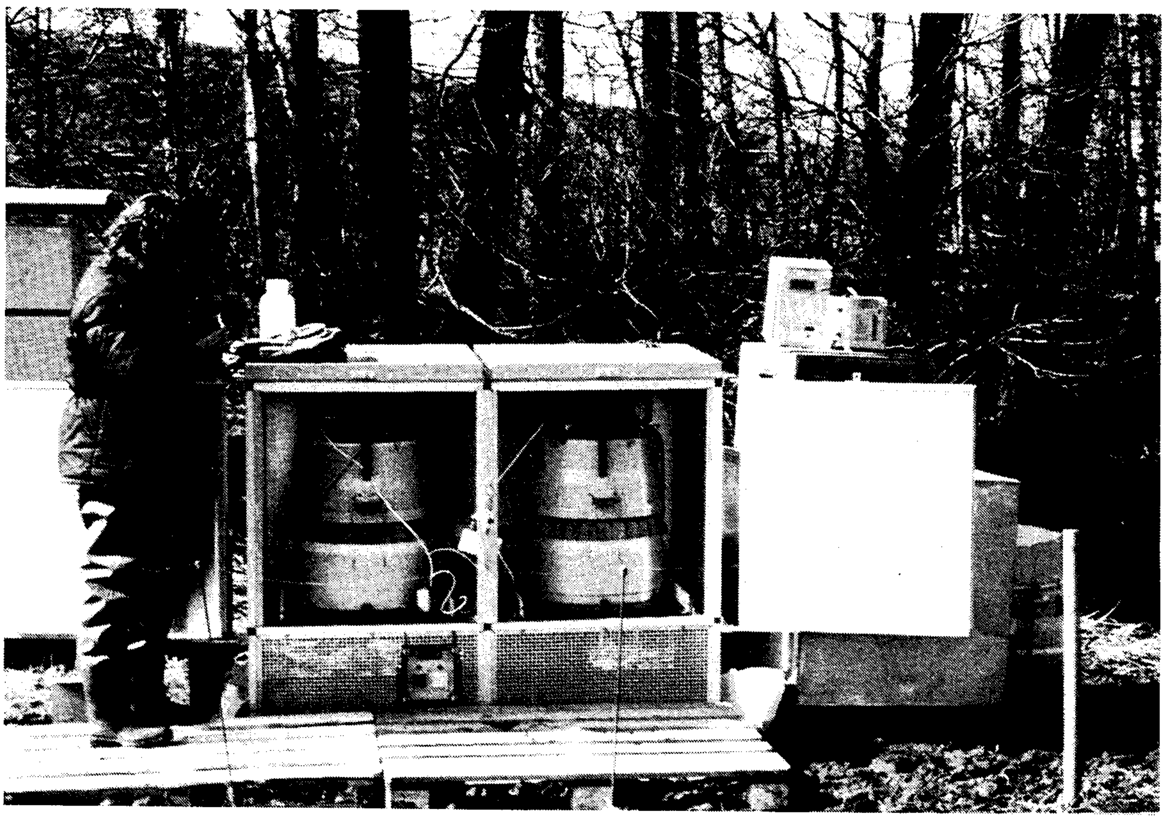
Technical session III



Transport of sediment and nutrients in streams

Phosphorus storage transport and transformation in river systems of the lake Leman basin

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Abstract

The work reported here focused on understanding and quantifying the storage, transformation and transfer of phosphorus in river systems draining rural watersheds within the Lake Léman basin. The transfer of phosphorus from watersheds to lakes appears to result from a series of in-stream processes including: alternate periods of deposition and resuspension of sediment and sediment-P, P sorption and release from sediments, and concentration and dilution of P in the water. All these processes affect not only the transport of total-P but also the dissolved-P to total-P ratio, the speciation of P, and P bioavailability.

Introduction

The impact of a watershed on a lake depends on many parameters, not only the total load of P at the outlet /1/. The ratio soluble P / total P, the bio-availability of particulate P and fixation capacity of associated suspended matter and the distribution of P inputs over the annual cycle control the use of P inputs by the lake ecosystem.

Consequently, managers who have the responsibility for implementing phosphorus reduction plans for lakes must know the origins of phosphorus within the watersheds feeding the lake as well as the amount, speciation, and properties of the phosphorus ultimately delivered to the lake. An understanding of the origin and speciation of P requires knowledge of how P is transferred from its origins through the watershed, to the lakes.

The transfer of phosphorus through watersheds to lakes must be considered as a discontinuous process /2,3/ that includes the storage and transport of phosphorus within and through the watershed drainage net /4/. The work reported here focused on understanding and quantifying the storage, transformation and transfer of phosphorus in river systems draining rural watersheds within the lake Léman basin /5,6/. It is based on the results of 3 years of intensive studies in two watersheds, the Redon (33km^2) and the Foron (51,5 km^2).

Watersheds and sampling strategy

Lake Léman, is one of the largest lake in western Europe. Concerns for the quality of the lake's water were expressed as early as the 1960's, when an increase in algal blooms prompted a closer look at the inputs of P to the lake /7/. Presently, since the point sources from the cities around the lake are better known and controlled, attention has shifted to the sources of P in rural areas. The overall objective is to decrease the P concentration in the lake to 30 µg.l^{-1} before the year 2000.

In the rural areas around lake Léman, phosphorus sources include

1) points sources (known sewage discharges from houses or agricultural operations)
2) numerous small sewage pipes scattered all along the river and streams (discharges largely unknown)
3) and surface runoff from villages and vegetated and cultivated land areas.

Table 1 - Some characteristics of the Redon and Foron watersheds.

	Total area (km^2)	Agricultural areas (km^2)	Inhabitants	Discharge (m^3.s^{-1}) annual mean value	Discharge (m^3.s^{-1}) extremes values	Fluxes of phosphorus total-P (annual mean value) t.year^{-1}	Ratio point sources/ diffuse sources (annual mean values) %
Redon	33	18	3000	0,5	0,08 to 12	2,6 to 5,1 (1983-1988)	0,7
Foron	51,5	24	6000	0,7	0,1 to 14	5 to 6,7 (1989-1993)	0,8

Our two experimental watersheds (Redon 33 km^2 and Foron 51,5 km^2; Fig. 1, Table 1) are representative of French rural areas around lake Léman. The sources of the rivers are located in steep slopes of the upland forests of the calcareous Prealps. The lower part of the stream cross a morained region of a gentle relief used for agriculture (70% grasslands; 30% wheat and corn). In this lower part the banks are usually vegetated and the hydrographic network receive point sources discharges from villages. The waters and the sediments are always rich in calcium. The rivers has a torrential flow regime /8/ with suspended sediment content varying from 0,005 to 1,5 $\mu g.l^{-1}$. The flow regime prevents development of macrophytes.

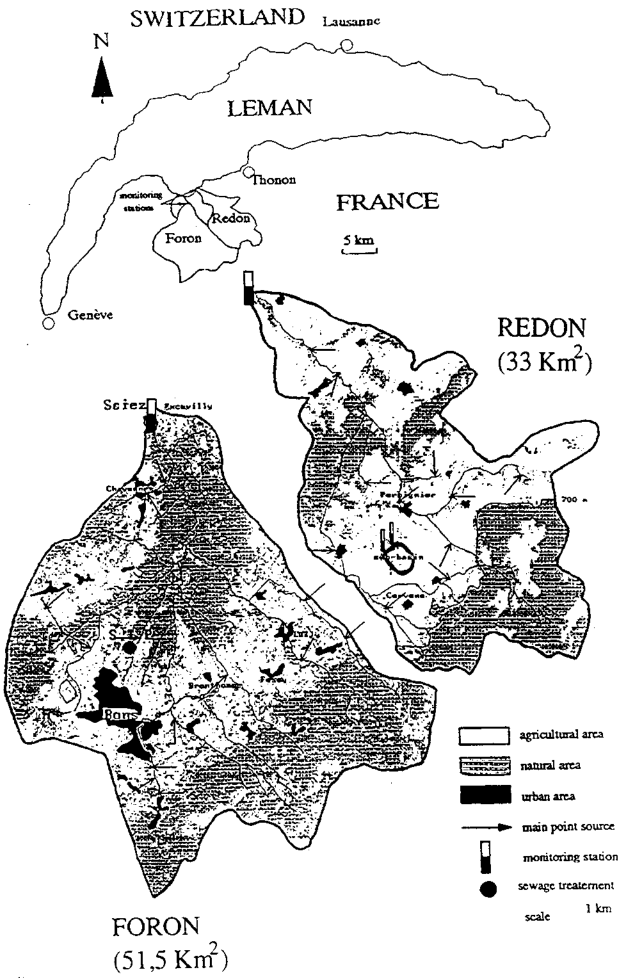

Fig. 1. Studied sites : Redon and Foron rivers watersheds

The monitoring of the watersheds /9, 10/ included

1) a continuous record of P loads and concentrations at the outlets of the two watersheds (for three years the water was sampled every 30 minutes ; P analyses were performed on flow proportional composite samples to obtain the loads ; during numerous stormflows discrete samples were analysed,
2) complementary and grab sampling of sediments from typical locations along the rivers and streams, 3) seasonal inventories of water quality in springs, rainfall, and waste waters. The same approach /11/ was used during 1 year study of a small agricultural subwatershed (14 ha, 50 % cultivated soils, 50 % grasslands) and of a wetland (2 ha).

P analyses were performed according the method of Murphy and Riley /12/. Total-P (TP), total dissolved-P (DP) after filtration at 0,45 µm and soluble ortho-P (dissolved molybdate reactive P, after filtration) were measured.

Sediments samples were taken in the superficial layers (0 to 5 cm) of the material deposited in the pools of the rivers. Analyses were performed with the fraction < 200 µm in order to compare

1) the P speciation,
2) the content of easily exchangeable phosphorus (Ex-P) and bioavailable P (Bio-P),
3) and the P fixation capacity (FC).

Three complementary analytical technics were used

1) chemical extractions to determine the organic-P content, the P released in water or in stronger extractants /13, 14/,
2) standard bioassays was used to test the bioavailability /6/,
3) isotopic exchange with PO_4^{32} /15/ was used to test the capability of sediment particles to fix and exchange ortho-P with water. All the results are expressed in elemental P ($mg.l^{-1}$ or $mg.kg^{-1}$ sediment).

Overview of mass balance of phosphorus in the watersheds

The total annual P loads measured (Table 1) at the outlets varied greatly according to the hydrological conditions of the year. These outputs were very high compared to the natural inputs but very low considering the total amount of P which were "handled" within the watersheds.

Throughout the watersheds agriculture applies an average of 30 $kgP.ha^{-1}.yr^{-1}$, manure and chemical fertilizers, which for the Redon was about 55 t of P. The domestic inputs of waste and detergent contribute 4 g per day per inhabitant /16, 17/ corresponding roughly to 5 t per year for the Redon and 7 t per year for the Foron. A part of this domestic P is discharged into the river system directly or with no P removal. Agriculture also contributes to point source discharges in unknown amounts.

The input of P to the river from agricultural land has been estimated at 0,2 - 0,3 $kg.ha^{-1}.year^{-1}$ for the two watersheds /11, 16/. However the load measured in the small agricultural sub watershed had a higher, mean value of 0,8 $kg.ha^{-1}.year^{-1}$. The 2 ha wetland reduced by 80% the inputs of diffuse P that it transferred to the river system /11/.

Mass balance and P retention during base flow periods

During base flow periods the water discharge is stable over time and at its lowest level (very dry periods).

The phosphorus inputs originate largely from point source discharges that vary over the day but are constant at the scale of the week. These inputs are made up with more than 60% dissolved-P. It is difficult to estimate exactly the total quantity of sewage discharges because, despite intensive full survey, some small sources of sewages often remain unknown. Natural background inputs are negligible (less than 15 µg.l^{-1} total-P in springs and phreatic water, comprising 95 % of dissolved-P).

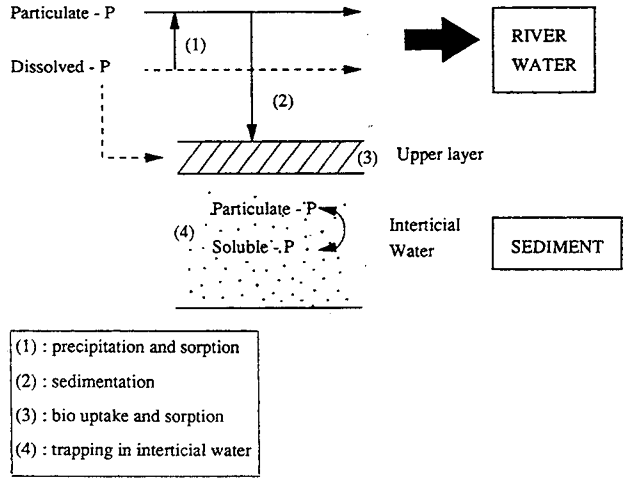

Fig. 2. Schematic representation of mechanisms contributing to P retention in rivers.

Table 2 - Mass balance during base flow periods.

	Minimum outputs during the lowest flow periods (kg per week)	Known (1) sewage inputs (kg per week)	Discharge ($m^3.s^{-1}$)
Foron	28	>60	80-100
Redon	17	>60	70-75

At the outlet concentration of suspended sediments were very low (< 10 mg.l^{-1}) but total-P concentrations were high (up to 0,5 mg.l^{-1}), with 95% of D-P (including 85 % of ortho-P). During these periods the weekly average export of total-P was only one third and one half of the known inputs of P for the Redon and the Foron respectively (Table 2a), regardless of the season.

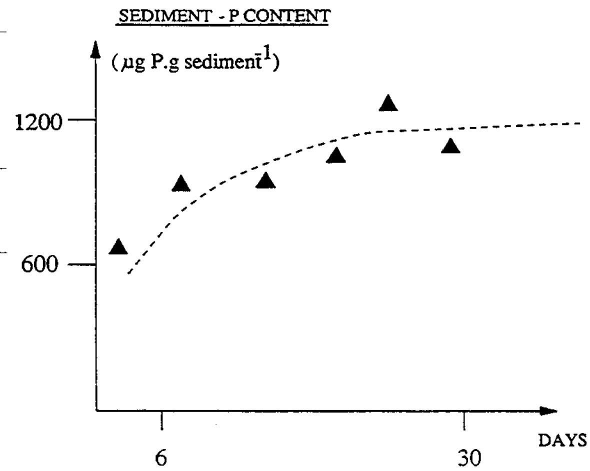

Fig. 3. Total-P content of sediment at the Foron river and duration of base flows (sediments<200 µm).

Table 3 - Sediments (<200 µm) in the Redon and Foron watersheds : P content, speciation and fixation capacity for P.(ppm = mgP.kg^{-1} sediment).

	General survey of sediments of Foron and Redon (10 samples/ location)	Representatives sampled in the Redon river (mean values)						
Locations	Total-P content (ppm)	Intersticial water soluble-P (mg.l^{-1})	Total-P (ppm)	Organic-P (ppm)	Bio-P (ppm)	Immediately exchangeable (ppm)	Fixation capacity index (% maximum)	Number of samples in Redon
Unpolluted area (forested areas, natural grasslands)	300-500	0,051	415	60	60	1,1	90	3
Agricultural subwatersheds	120-950	0,120	600	290	90	2,8	93	6
Outlet of wetlands	300-2000	0,027	780	250	60	2,0	100	1
Down stream, far from point sources (outlet)	800-1500	1,30	800	110	220	8,0	50	4

All these processes are controlled by time, discharge, concentration gradients, speciation of P and consequently can vary from one reach to another in a river ecosystem. For example the attenuation of total-P seemed to be highest in the vicinity of sewage discharge where the total-P content of sediment increased to 2000 ppm (x 5 compared that of natural sediment). But in these rivers, even at many km downstream from sewage discharges, the sediments were statistically richer (x 2) in total-P than the sediment of natural origins (Table 3).

At the outlet, the sediment enrichment was progressive over time (Fig. 3), and the maximum P content was reached only after about 10 days. At this time the content in DP of intersticial water of river bed sediment was very high (up to 1,3 $mgP.l^{-1}$; x 50 compared to the sample collected in a natural area).

Speciation and properties of the accumulated particulate-P

To evaluate the magnitude of the changes in P The portion of the total-P input that was not exported and consequently was stored in the hydrographic network, corresponded to significant amounts of P. For example, the P stored during one or two weeks of base flow (≈50 kg P) was equivalent to the mean weekly export or to the load of a medium stormflow /10/. In the Redon, during a five week period of base flow the inputs were stable 17 ± 2 kg of P. The P speciation data for the inputs and outputs indicated that both soluble P and particulate P were stored /17/.

Retention or storage of P in a stream system (Fig. 2) is due to a number of mechanisms including :

- sedimentation of the particulate P in discharges into the river and of the particulate P created in stream coagulation processes.
- biological uptake of DP /3/ which can be a significant removal mechanism but appeared to be negligible in the Foron and Redon since there was no macrophyte, and a low density of periphyton.
- the ortho-P, in theory, might be expected to precipitate with Ca, because the Redon and Foron rivers were over-satured with respect to the Ca-P precipitation /10/. However using Xray it has not been possible to detect apatite in suspensions, perhaps because of the formation of poorly crystalline form, as in soils /18/.
- adsorption of dissolved-P onto particles including previously settled ones /19/.
- the dissolved-P contained in interstitial waters is trapped in river bed sediment. speciation and properties induced by the retention of P from sewage discharges in the river bed sediment, a comparative study was made of the P content of typical sediments of the Redon river /6/.

Following a three weeks period of base flow during the winter, a total of 11 sediment samples has been collected at representative locations as determined by a general mapping of P content in sediment. The main chemical forms of P, sediment P fixation capacity and bioavailability, are summarized in Table 3.

All properties including bioavailability, varied according the origin of sediment. Sediment originating from natural areas had a low contents in Bio-P, organic-P and exchangeable-P and a high fixation capacity. In other words, these sediments possessed the capability to take up phosphorus from the overlying water but little ability to contribute P. With the sediments originating in agricultural areas and in the wetland, the total-P content increased, but the fixation capacity remained very high. These sediments were also very rich in organic-P /20/. Near the outlet of the Redon the interstitial waters of the sediments contained a high content of dissolved-P while the sediments showed the highest ability to release P into the water by exchange or biouptake. This diversity of sediment and phosphorus characteristics along the river is important to consider. Erosion of natural areas, river banks, agricultural soils and ditches in wetlands, input to the river system sediments with a high fixation capacity. As these sediments move downstream, they trap P discharged by point sources and may approach saturation. These enriched sediments have a high level of exchangeability and low fixation capacity. They accumulate during base flow periods and represent an important source of P to be exported during subsequent high flow.

Dynamic of P storage, transport and export during other hydrologic conditions

Low flow in absence of surface runoff

The low flow condition in absence of runoff is defined as a period of at least one week, with no rain, and a continuous decrease or a stabilized water discharge. After a stormflow, the start of this low-flow condition is indicated by the return to background values for conductivity (>500 µS) and suspended sediment content (<30 $mg.l^{-1}$). The concentration in total-P at the river outlet during these periods is very high (0,2 to 0,5 $mg.l^{-1}$) in comparison to that in drainage from natural areas, agricultural subwatersheds and springs (< 0,03 mg l^{-1}) /10/. The soluble forms of P and especially ortho-P, dominate (≥ 90 % of TP). Under these conditions mainly ground waters contribute to the flow. The P from sewage discharges is the main source of P entering the river. The weekly average values of these inputs from sewages are essentially constant from one week to another.

Table 4 shows, for the Redon in 1983, a relationship between P load exported at the outlet and mean weekly discharge during low flow periods. The data suggest a plateau at a flow QL > 0,6 to 0,7 $m^3.s^{-1}$ and an output of 70 kg $P.week^{-1}$. Since the point sources inputs was constant the P export increased with weekly discharge. With the lowest flow discharges, a large part of the P entering the river was retained (Table 2). As the average weekly discharges increased less and less P was retained. When the plateau was reached, there was an equilibrium between weekly inputs and outputs so that all the point source inputs were exported from the river system.

Table 4. P exports at the outlet of the Redon watershed, during the low flow periods of 1983.

Date (months)	Mean Q ($m^3.s^{-1}$)	Outputs ($kg.week^{-1}$)
02	0.08	19
07	0.11	27
11	0.15	30
03	0.20	37
06	0.27	36
10	0.40	38
05	0.64	70
07	0.72	75
07	0.80	70
09	0.90	72
09	1.00	74

Table 5. Mass balance during lowflow periods in absence of runoff and estimation of total input from sewage discharges : (1) results of a field survey (2) maximum output - natural input.

	Known (1) sewage inputs (kg per week)	Maximum output (kg per week)	Natural inputs (kg per week)	Total sewage input (2) (kg per week)
Foron	>60	80-100	8	82-92
Redon	>60	70-75	5	65-70

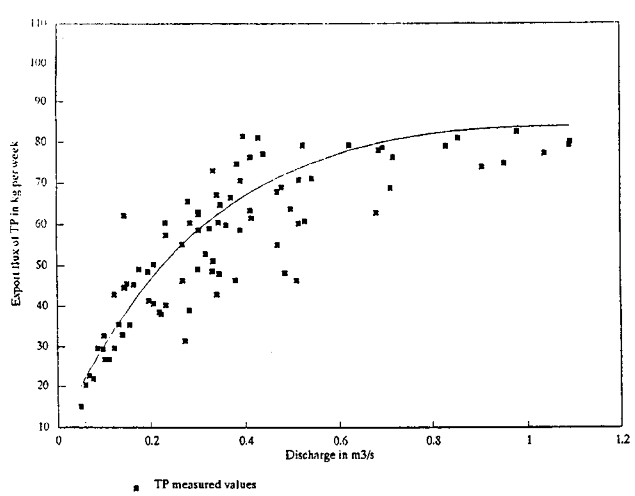

Fig. 4. Weekly mean discharge (q) and export of total-P during low flow periods in absence of runoff; (Foron river, 1990-1993); flux F = 90.(1 - exp(-3q)-10q).

The same type of relationship was observed in both the Foron and Redon watersheds, during the 3 years studied (Fig. 4, for the Foron). These relationships may be used to estimate the total inputs from all sewage discharges (Table 5). The average maximum output on the plateau is equal to the total input including all the sewages discharged to the river (known and unknown) and the background inputs (product of the mean concentration in the unpolluted parts of the basins and the discharge at the outlet). Estimates of the P load from total sewage discharges found in this way were very close to those from our field survey of the Redon, but substantially higher than survey result for the Foron river. There are many small sewage pipes in the Foron watershed not accounted in the field survey (scattered habitat) and it is believed that these unknown discharges made up the difference.

Significant relationship between weekly mean concentration at the outlet and weekly mean discharge has been established to validate these observations (Fig. 5, for the Redon). The equations of regression didn't follow a standard dilution law because the dilution effect is disturbed by the antagonistic effect due to the increase of transport of P inputs from sewages with the increasing discharge /5/.

Dynamic of P during stormflow events

General caracteristics

Most particulate P was exported during stormflow events. According to the stormflow export loads of particulate-P varied between 10 to 250 kg. Particulate-P concentration ranged up to 1,5 $mg.l^{-1}$ /5 , 9/. There was no consistent relationship between water discharge and total-P or DP, either in concentration or in flux. The same lack of relationship has been observed in larger watersheds. This suggest that the stormflows might not be considered as an homogeneous set of events, considering P dynamic.

Typology of stormflows

The patterns of P dynamic and mass balance observed during stormflow events appeared to be of two main types according to the origins of P delivered to the river, to the area of the basin that contribute to increases of flow and to the hydrologic state, prior to the event. Figure 6 shows in detail two characteristic stormflows of the Redon river. The first (Fig. 6a) occured in December at the end of a rainy period during which there was numerous small stormflows. The second stormflow (Fig. 6b) occured in march after a 3 month base flow period (initial Q << QL). The hydrographs and total-P loads were different : the december stormflow exported 170 kg and had a peak flow of 9,1 m^3s^{-1}, whereas the march stormflow exported 61 kg of total-P with a peak flow of 2,2 m^3s^{-1}. The DP (mainly ortho-P) loads were similar being 52 kg and 42 kg, respectively.

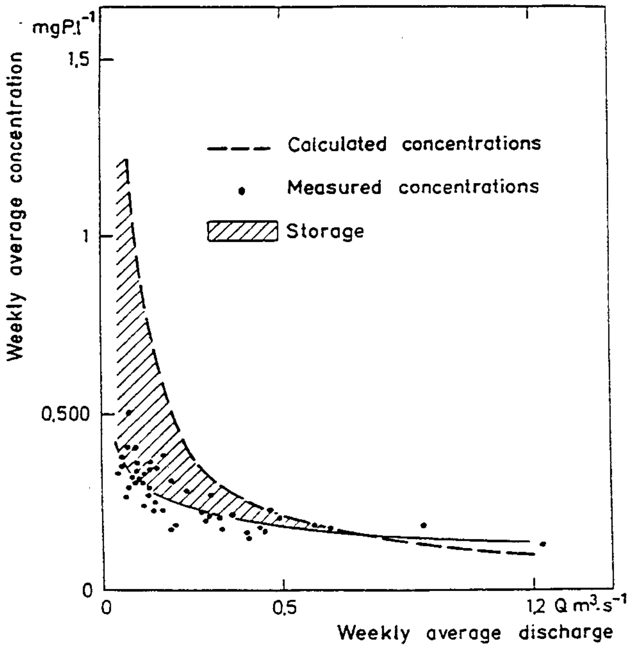

Fig. 5 - Weekly average discharge and weekly average total-P concentration during low flow period in absence of runoff ; Redon river watershed 1984-1987. Calculated concentrations correspond to the dilution law: known inputs from points sources/discharges.

The conditions in which P transport occured were quite different. The stormflow of March showed peaks of concentration of particulate-P and total-P as high as those of the December stormflow. But the peak of particulate-P appeared at the very begining of the event and decreased rapidly many hours before the maximum of water discharge. This pattern was also observed for sediments and is typical of the transfert of resuspended sediment originated from the flushing out of the P stored in the river bed enriched sediment. In the December event, little P was stored in the river (initial Q > QL), the P level in the sediment was at its background level. The particulate-P concentration increased and dicreased gradually with the discharge. The delivery of particulate-P was controlled in this case by the water supply to the river, a pattern typical for erosion. The different patterns between the two stormflows events for DP concen-tration (dilution in December, increase in March) and P content of suspended matter (higher in March) also suggest a difference in origin for the transported P.

Based on the above observations, the 21 stormflows periods studied in the Redon basin, were classed into two categories. We noticed the relationships between the mean concentration of total-P of the stormflows and the total volume of discharge that were completely different /5/. For stormflows like the type represented by March event the average total-P concentration (Cm) was directly proportional to the amount of P stored (S) during the previous low flow periods, and inversly proportional to the total volume of flow (V) :

$$\text{LogCm} = 0{,}03\ S/0{,}001\ V$$

(Cm in mgl^{-1}; S in kg; V in m^3 ; r = 0,83 ; n = 13 ; 1 %).

This correlation suggest a model in which an important amount of P is flushed from the river bed and then diluted by the volume of the flow. Total-P mass transported is a function of (S), while total-P concentration is a function of both (S) and (V).

For the other type of stormflow like the December event, a more classifical relationship was found in that the average concentration of total-P (Cm) was directly proportional with the volume of water (V)

$$\text{Cm} = 0{,}5.V + 1{,}8$$

(Cm in mgl^{-1}; V in m^3; n = 14; r^2 = 0,73; 5 %).

This suggest a transfer controlled by water discharge and a total-P that originated mainly from erosion.

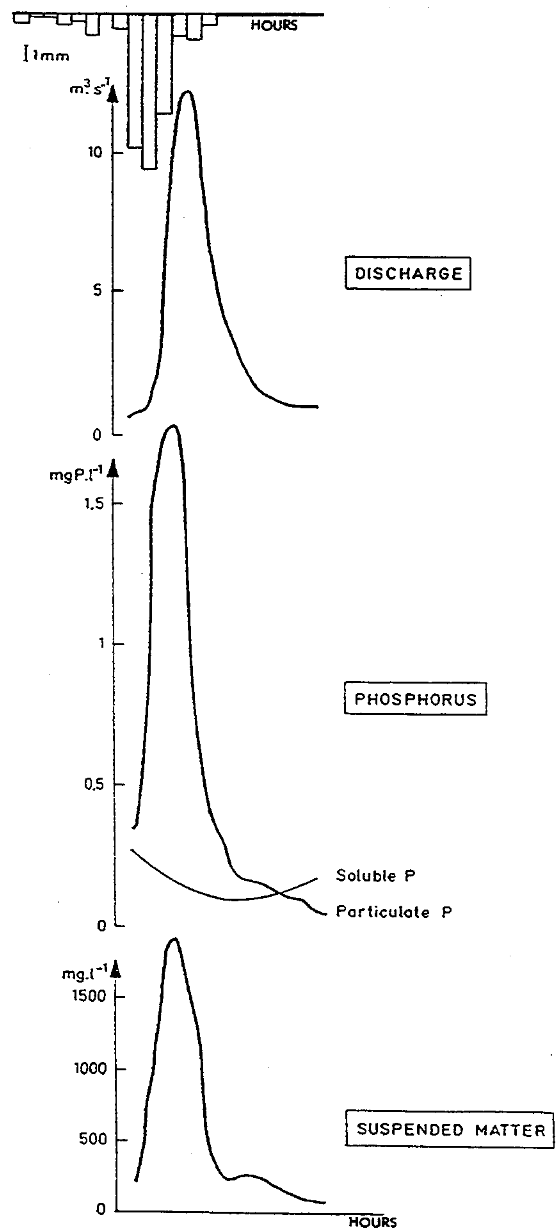

Fig. 6a. Discharge, suspended sediment, phosphorus during a typical stormflow of a wet period (Redon river watershed, December).

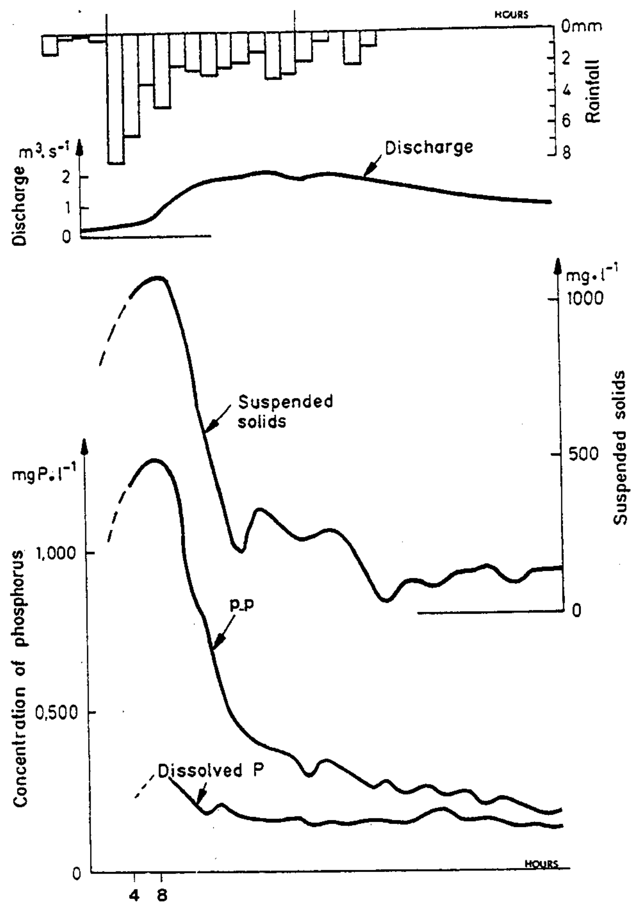

Fig. 6b. Discharge, suspended sediment, phosphorus during a typical stormflow occuring at the end of a dry period (Redon river watershed, March).

Discussion

Diffuse sources are active only during stormflows /21/ and include urban runoff and runoff from natural and agricultural areas, 2 sources of sediments and total-P which ordinarly doesn't operate at the same time. Point sources discharge continuously mainly DP. A portion of this P is transported to the outlet and a portion is stored in the river system during low flows. As previously mentionned the P stored constitutes a significant source of P for many stormflows.

During base flow and low flow periods the watershed exports total-P that originates from point source discharges. Exported P predominately soluble and bioavailable has a maximum potential impact on the lake.

At the end of low flow or base flow periods, large amounts of P can be stored in the river system and the soils and drainage ditches from agricultural areas are dry. Thus, when the rainfall occurs the hydrologically active part of the basin is largely the urban impermeable areas and roads. The runoff washes the dry deposits accumulated on these surfaces into the river and the increased flow resuspends some or all of the sediment and total-P stored in the river system. The sediments transported are very rich in Bio-P and has the lowest fixation capacity observed. Resuspension may also increase the soluble-P due to release of the intersticial water of the river sediment or due to release from particulate-P. Total-P exported is largely particulate but highly bioavaiable. Moreover the transfer to the lake take place generally during the growing season for algi.

During rainy period soils are wet and a large part of the basin area is hydrologically active. Before stormflows the water discharge is high enough to prevent storage of P from point sources in river bed sediments, and dry atmospheric deposition doesn't accumulate due to frequent rainfalls. Thus, the P stored in the river and on urban surfaces is low and at its background level. Consequently the P outputs originate mainly in the diffuse sources from the land. The total-P exported during the stormflow of these periods is composed mainly of particulate-P. But this particulate-P originates from natural areas, river banks and agricultural soils that have relatively low Bio-P and high fixation capacity. This could explain the low values of dissolved-P / particulate-P ratio and the mean low content of in suspended sediment observed during these periods. In the Foron river these stormflows set the stage for a renewal of river system P capacity to accumulate which increased attenua-tion of DP in the water for the following low flow periods /9/. Finally, we assume that the impact of these stormflows on the receiving bodies is low, even if large amounts of P are exported.

/17/ and /16/, evaluated the mass balance of TP over time in the rivers Redon and Foron respectively and showed that the storage of TP in enriched sediment was a substancial portion (respectively 40 and 50 %) of the total amount of TP exported by the watersheds. They estimated at least that 35 % of the particulate-P exported resulted from resuspension (during the studied periods).

Conclusion - Summary

In the watersheds studied dissolved-P inputs originated mainly from various sewage discharges. During low flow periods the DP in the water was attenuated and the TP content of the in-stream sediments was increased. These in-stream sediments played a major role in the storage of phosphorus within the stream ecosystem. The magnitude of the TP accumulation in the sediments can be estimated by analysing total-P mass balances during low flow periods. This accumulation appeared to be controlled by the magnitude of streamflow and the duration of the low flow period. During stormflow events, the sediments were resuspended and moved downstream. The ratio of dissolved-P to particulate-P was different during stormflow events when the P associated with the enriched in-stream sediments was the main source of P.

Regardless of the origins of the P that enters the rivers, the transfer from watersheds to lakes appears to be the result from a serie of in-stream processes including, alternate periods of deposition (enrichment) and resuspension (dilution), P retention and release from sediments. Biotransformation and biocycling of P within the in-stream ecosystem must also be important. All these processes affect not only the transport of total-P but also the timing of P export and the speciation such as the soluble-P to total-P ratio, and the speciation of P, as well as P bioavailability associated with particulate-P. The relative importance of these processes in the transfer of phosphorus from watersheds to lakes varies temporally and spatially and is dependent on the hydrology and size of the watershed. Future researchs will study the importance of these processes as they impact on receiving bodies of water.

Acknowledgement

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Transport of sediment and phosphorus in the arable Gelbæk catchment, Denmark: III. Quantification of sources

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Abstract

Using a multi sampling strategy approach the annual loss of suspended sediment and particulate phosphorus (P) from a Danish arable catchment, Gelbæk, from May 1993 to April 1994 was estimated to be 71 kg ha^{-1} and 0.32 kg P ha^{-1}, respectively, most of which occurred during infrequent storm events. Thus 40% of annual sediment loss and 28% of particulate P loss took place during 2% of the year. The sampling strategy followed enabled us to determine the accuracy of estimating suspended sediment and particulate P losses on the basis of discrete and intensive monitoring.

Identification and quantification of the different pathways of sediment and particulate P at the catchment level is very important with regard to appropriate catchment management. Our experience shows that in artificially drained and geologically complex catchments such as the Gelbæk comparative monitoring of different sources (e.g. drainage, surface runoff, internal erosion) is one important mean to reliably discriminate between the various diffuse sources of sediment and phosphorus losses. Of the annual export of sediment and phosphorus from Gelbæk catchment drainage water accounted for 18% and 22%, respectively, albeit that seasonal variation was large, while soil erosion (rill erosion) accounted for 19% and 7%, respectively. Internal stream bank and bottom erosion was therefore the major source of diffuse losses of sediment and phosphorus from this arable catchment in the study year.

Introduction

The transport and fate of sediments in watersheds is an important aspect of environmental research as large quantities of nutrients, pesticides and other contaminants are transported in association with the sediment /1,2/. However, precise and correct estimation of watershed losses of sediments and sediment-associated substances is difficult due to the large spatial and temporal variation in the factors governing sediment delivery and transport /3,4,5/.

Another problem is that the mechanisms governing the flux of sediments in watersheds are still poorly investigated, especially with regard to the establishment of source budgets and the physical and biological processes affecting in-stream sediment transport /6/.

Most of what we know is based on sediment and phosphorus losses from small-scale field plots, such studies having provided valuable information on mobilization and transport mechanisms. That catchment-level sediment and phosphorus budgets have rarely been established is mainly attributable to the large spatial and temporal variability in mobilization, delivery and transport systems seen at the catchment level. Such research is essential, though, because of the difficulties in upscaling from field to catchment level.

In recent years indirect methods applying a suite of different "fingerprinting" tools to potential source compartments have proven successful /7/. Fingerprinting with ^{137}Cs, ^{210}Pb and C, N and P content in combination with simple or more complex mixing models seems especially promising /8/. Another important tool is direct quantification of sediment and phosphorus sources by simultaneous and comparative measurement of different delivery routes. Although use of the method has been limited, it may nevertheless be an important tool in combination with indirect measurements.

In the present paper we will focus on the difficulties of correctly measuring sediment and phosphorus fluxes in watercourses and comment on the accuracy of the different methods available. In addition, the importance of drainage water and rill erosion as diffuse sources of sediment and phosphorus will be quantified on an event, monthly and annual basis.

Study area and study methods

The stream Gelbæk drains an intensively farmed (>90%) catchment in central Jutland, Denmark, comprised of sandy loam soils /9/. The study is based on intensive and comparative measurements of suspended sediment and particulate P content of drainage water from two sub-catchments (4.4 and 13.3 ha) and the stream draining the whole catchment (1160 ha). The study period was from May 1993 to April 1995. Water samples were taken by simple dip sampling as well as by automatic samplers (ISCO 3900) installed in termostated boxes /9/. Three sampling strategies were conducted and used to estimate suspended sediment and particulate P transport: infrequent sampling (fortnightly); hourly sampling (pooled weekly); storm event sampling combined with infrequent sampling. The samples were kept dark and refrigerated prior to analysis for sediment and phosphorus fractions, which was conducted as described in /6/.

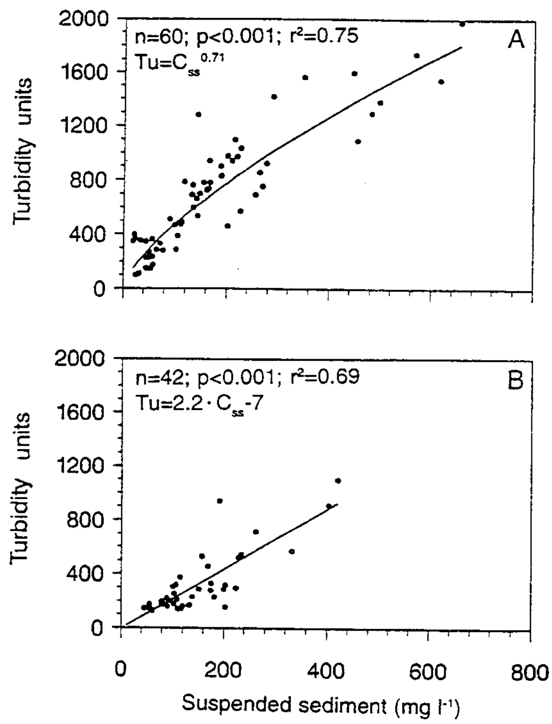

Fig. 1. Relationship between turbidity and suspended sediment concentration during four autumn storms (A) and four winter storms (B) in Gelbæk stream during 1994/95.

Results and Discussion

Measurement of sediment and phosphorus transport in streams

Infrequent versus intensive sampling
Reliable measurements of the transport of suspended sediment and particulate P in streams are essential when establishing source budgets, several recent studies having shown that the reliability of load estimates is closely linked to the hydrological regime of the stream in question, and possibly also to the extent of local source areas.

The accuracy of different sampling strategies was evaluated in Gelbæk stream during the year May 1993 to April 1994 by comparing infrequent sampling every fortnight with hourly sampling (pooled to obtain a weekly average concentration) and more intensive sampling during individual storm events (daily basis).

Comparison of the three sampling strategies revealed that infrequent sampling underestimated the annual transport of suspended sediment and particulate P as compared to pooled weekly sampling and storm event sampling (Table 1). The bias of suspended sediment transport and hence transport of particulate P was even more pronounced on a monthly basis ranging from 80% to -160%. Multiple regression analysis proved that the monthly bias was significantly correlated to runoff ($n = 12$; $p<0.01$; $r^2 = 0.68$) and a baseflow index (BFI-index) ($n = 12$; $p<0.05$; $r^2 = 0.13$), thus indicating that the hydrology and hydrological regime in the Gelbæk stream plays a major role in controlling sediment and particulate P mobilization and transport.

Table 1. Annual transport of suspended sediment and particulate phosphorus estimated based on infrequent (fortnightly) sampling, hourly sampling (pooled weekly) and storm event sampling.

	Infrequent sampling	Hourly sampling	Storm event sampling
Suspended sediment	62,950 kg	82,165 kg	78,413 kg
Particulate phosphorus	359 kg	365 kg	390 kg

Use of indirect methods
Although turbidimeters have been widely used for indirect monitoring of suspended sediment concentration, the relationship between turbidity and concentration is potentially confounded by variation in particle size, particle composition and water colour (e.g. 10). Nevertheless, turbidimeters provide a very detailed picture which is difficult and costly to obtain by other means.

In the present study we used an infrared Partech turbidimeter to measure turbidity every 5 minutes, thereby enabling us to record detailed fluctuations in suspended sediment concentrations that would not otherwise be recorded, e.g. during single storm events.

Comparison of simultaneous measurements of suspended sediment concentration and turbidity during eight storm events in September 1994 to March 1995 revealed a highly significant linear relationship ($n = 101$; $p<0.0001$; $r^2 = 0.70$) that could be improved by separately developing relationships for the four autumn storms (September to December 1994) and the four winter storms (January to March 1995) (Fig. 1A and B).

The relationships show that turbidity was much higher in Gelbæk stream during autumn storms than during winter storms, this probably being attributable to differences in particle size and composition since organic content of sediment was higher in autumn than in winter.

The concentration of particulate P proved to be significantly linearly related to the concentration of suspended sediment ($n = 39$; $p<0.0001$; $r^2 = 0.87$), irrespective of season or rising and falling stage conditions, with the P content of the sediment being 0.4% at all suspended sediment concentrations (Fig. 2).

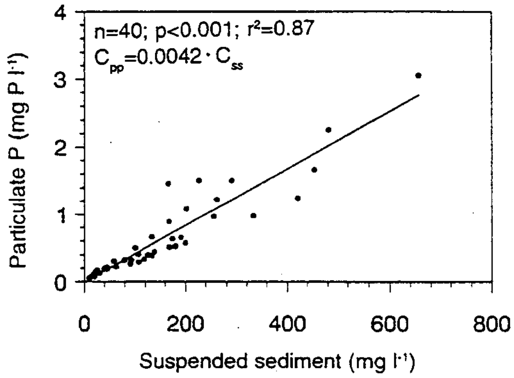

Fig. 2. Relationship between particulate phosphorus and suspended sediment concentrations during eight autumn and winter storm events in Gelbæk stream during 1994/95.

Transport and delivery of sediment and phosphorus

Storm events

The transport of suspended sediment and particulate P in Gelbæk mainly takes place during short periods where intensive precipitation or melting snow generates runoff thereby mobilizing sediment in the various source areas. Thus 40% of the annual suspended sediment transport and 28% of the annual particulate P transport took place during nine major storm events together lasting 2% of the year. Suspended sediment and particulate P losses from the two drained sub-catchments and the catchment as a whole were measured simultaneously during these nine storm events. High suspended sediment and particulate P losses occurred in both the drainage water and the stream (Fig. 3).

Table 2. Estimated contribution of drainage water to suspended sediment and particulate P export from the Gelbæk stream catchment during nine storm events in 1993/94.

Date of storm event	Suspended sediment	Particulate phosphorus
19 September 1993	30%	25%
14 September 1993	40%	33%
9 October 1993	20%	23%
9 December 1993	30%	46%
19 December 1993	17%	41%
27 January 1994	28%	35%
2 February 1994	69%	51%
7 March 1994	3%	5%
23 March 1994	20%	18%
Average	29%	31%

The percentage of total suspended sediment and particulate P losses from the catchment accounted for by drainage water (50% of the catchment being drained) averaged 29% (range: 3-69%) and 31% (range: 5-51%), respectively, during the nine storm events (Table 2). The loss of suspended sediment and particulate P from drainage water during single storm events was found to be significantly related ($n = 9$; $p<0.001$; $r^2 = 0.44$) to precipitation within the catchment (Fig. 4).

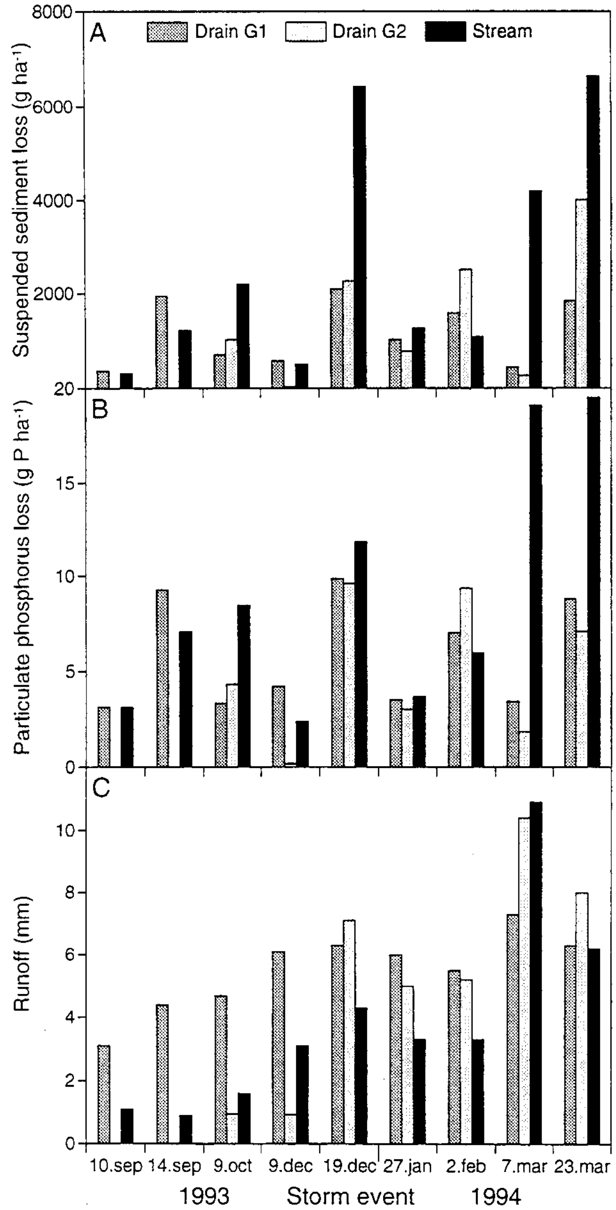

Fig. 3. Losses of suspended sediment (A), particulate phosphorus (B) and runoff (C) from two drained sub-catchments (G2: 4.4 and G1: 13.3 ha) and Gelbæk stream catchment (1160 ha) during nine storm events in 1993/94.

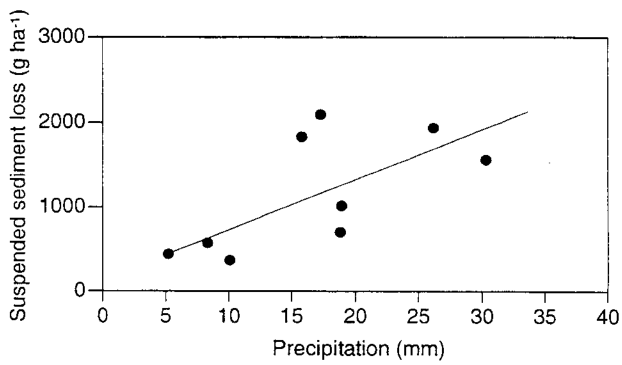

Fig. 4. Relationship between turbidity and suspended sediment from drainage water and precipitation in one of the sub-catchments in Gelbæk stream during nine storm events in 1993/94.

Monthly and annual

Monthly maximum loss of suspended sediment and particulate P was 4.68-12.6 kg ha^{-1} and 23.9-62.2 g P ha^{-1}, respectively, for the two drained sub-catchments as compared to 25.5 kg ha^{-1} and 105 g P ha^{-1} for the catchment as a whole. Drainage water accounted for 1% of suspended sediment and 0.4% of particulate P in the dry period (July 1993), in both cases increasing to 24% in the wet period (February 1994 and November 1993, respectively).

Annual losses from Gelbæk catchment during the period May 1993 to April 1994 amounted to 71 kg ha^{-1} suspended sediment and 0.32 kg P ha^{-1} for particulate P. Drainage water accounted for 18% of the suspended sediment and 22% of the particulate P while soil erosion accounted for 19% of the suspended sediment and 7% of the particulate P. Internal stream bank and bed erosion was therefore the major source area of suspended sediment (63%) and particulate P (71%) losses from the catchment.

Perspectives

The study clearly illustrates the need to use an appropriate sampling strategy when establishing source budgets for sediment and phosphorus at the catchment level since transport is concentrated in very small periods of the year.
Turbiditimeters seem to be a valuable method for reliably determining short-term temporal variation in suspended sediment concentration during single storm events, and are therefore likely to improve sediment transport estimates.

Moreover, the highly significant correlation between the concentrations of suspended sediment and particulate P in streams enables estimates of particulate P transport to be improved by using a turbiditimeter. However, careful calibration at the seasonal or perhaps even storm event level seems to be necessary.

Subsurface drainage water is an important source of suspended sediment and particulate P loss to watercourses during storm events, as well as on an annual basis. The early arrival of peak suspended sediment concentrations together with its high content of nitrogen and organic matter, high content of ^{137}Cs and the significant correlation between precipitation and loss rates during single storm events is consistent with the hypothesis that fine particulate matter from the surface soil is transported to drainage water through macropores in the soil matrix /9/. Such sediment mobilization poses the threat of eutrophication to surface waters because of the phosphorus enrichment of surface soils that has taken place over the last decades. The direct transport link from surface soil to drainage water also poses the potential threat of contamination of the aquatic environment with other harmful substances (e.g. heavy metals and pesticides).

We believe that comparative studies of the mobilization, transport and fate of sediment and sediment associated substances at the catchment level is a scientific task for the future which could benefit from further development of the "fingerprinting" technique for different catchment types. A holistic catchment approach is highly needed for improving our understanding of the mechanisms involved in the mobilization, transport and fate of sediments in order to be able to establish the most appropriate management practices at the catchment level.

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Discharge measurement, sampling techniques and their influence on calculated sediment and phosphorous loss from agricultural areas

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Abstract

JORDFORSK has carried out a measurement programme in two agricultural catchments, which showed that different methods for discharge measurement and water sampling techniques can have a large effect on the calculated sediment and phosphorous loss. As the standard method at our measuring stations, discharge is measured continuously whilst volume proportional composite water samples are extracted automatically. This method is considered to give the "true" figures and is compared with the alternative methods. These include point water sampling techniques with different time intervals combined with continuous - and point discharge measurements. Point water sampling techniques lead to deviations from the "true" sediment and phosphorous loss estimates. Increasing the point water sampling interval had a negative effect. However, increased access to hydrological data improved the estimates. A sampling frequency of once every 4 weeks could lead to deviation ranging from - 100% up to + 230% of the "true" values.

Introduction

Calculation of the amount of sediment and phosphorous transported in a watercourse requires the data on water chemistry and discharge.
Exact calculation of the losses of sediment and phosphorous requires continual recording of discharge and sediment and phosphorous concentration. This is not possible in most cases, primarily due to the costs involved.

JORDFORSK has developed a system based on continuous recorded discharge data and the extraction of discharge-volume proportional, composite water samples. Knowing that sediment and phosphorous concentrations can vary with discharge, this system is considered to give the most accurate values when estimating the transport of sediment and phosphorous in a watercourse /1/.

Most monitoring schemes rely on the regular extraction of point samples and recording of water head at the measurement structure. Many adopt trigger systems that initiate a period of greater sampling frequency when a pre-decided water head is exceeded (i.e. at the beginning of a storm event).

This paper presents part of the results from a Norwegian Research Council funded project comparing the JORDFORSK sampling method with point sampling systems in operation /2/. The project was carried out in five catchments during 1993, two of which are reported in this paper.

The catchments are both located in the south east part of Norway, but they are different with respect to size, land use and soils. Catchment number one, Mørdre, covers a total area of 680 ha, of which approximately 70% is arable and the rest is forest. Soils consist mainly of silt and clay.

The second catchment, Stokke, covers an area of approximately 180 ha, of which 40% is arable land. The other 60% consist of forest, bogs and urban areas. The soils are mainly silty sand.

Sampling methods

JORDFORSK's System

The JORDFORSK system requires a control module (such as a datalogger) on site. Water head is recorded continuous and the discharge calculated from the known head-discharge relationship for the measurement structure.

One discharge (i.e. volume) proportional composite water sample is a mixture of a series of smaller sub-samples taken over a known time period, called a composite sample period. A sub-sample (of equal volume) is taken each time a given volume of water has flowed through the measurement station, so that the sampling frequency is dependent on the runoff intensity. The sub-samples are collected in a sample container. Sediment and phosphorous transport (T) for a sample period is calculated on the basis of the water volume (V_Q) that has passed the measurement device and the sediment and phosphorous concentration in the composite sample.

Thus:

$$S = c_{comp} \times V_Q$$

where:

$$V_Q = \left(\int_{t=1}^{t=n} Q \times dt \right)$$

The total yearly sediment and phosphorous transport is the sum of the sediment and phosphorous transported during all the composite sample periods in the year.

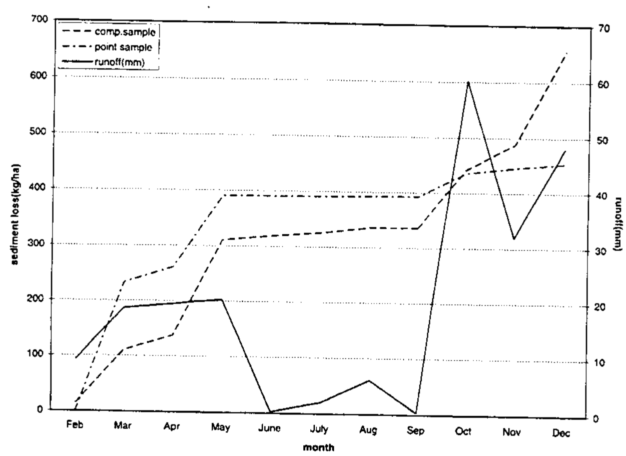

Fig. 1. Accumulated sediment loss(kg/ha) at the Mørdre catchment calculated on the basis of composite- and single sample techniques and continual recording of discharge.

Point Sample Systems

Calculation of sediment and phosphorous transport based on chemical analysis of point samples can be executed in different ways, dependent on the access to hydrological data and sampling frequency. This study refers to three levels of access to hydrological data, strategies 1, 2, and 3.

Strategy 1: No discharge data available. Estimation of runoff based on precipitation corrected for evapotranspiration. To obtain the total loss, average concentrations of all the point samples are multiplied by the total yearly water discharge, estimated using meteorological data.

Thus:

$$S = V_c \times \overline{\sum c_n}$$

Strategy 2: Water discharge measured at the time of sample extraction.

In *strategy 2a* substance transported (T) is calculated based on the discharge (Q_n) and concentration (c_n) recorded at the moment of sampling (t_n). They are assumed to have remained constant in the entire sampling period.

$$S = Q_n \times (t_n - t_{n-1}) \times c_n$$

Strategy 2b uses the mean values for discharge and concentration between two consecutive samples.

$$S = \overline{Q_{(t-1,\,t)}} \times (t_n - t_{n-1}) \times \overline{c_{(t-1,\,t)}}$$

Strategy 3: Water discharge recorded continuous. *Strategy 3a* and *3b* are equal to *2a* and *2b* respectively, except that calculations use the continually recorded discharge data.

Point sampling was carried out at weekly intervals throughout the study period. From these data, sediment and phosphorous transport calculations are based on sampling intervals of 1, 2 and 4 weeks.

In the investigation, the composite sample values (*JORDFORSK's system*) are represented as the "true" values for transported sediment and phosphorous. Total sediment and phosphorous transport for the whole of 1993 is calculated for the different point sample strategies and compared with the results from the composite samples.

Results and discussion

Figure 1 shows a characteristic picture of the runoff and the cumulative sediment loss based on composite samples and weekly point samples for the Mørdre catchment during an eleven month period in 1993. The point sample calculations are based on access to continuous recorded discharge data (Strategy 3b). The figure illustrates that a deviation on a yearly basis of calculated sediment loss between the different sampling strategies can conceal a much greater deviation in particular periods.

Soil erosion is a typical event based process. Due to the climatic conditions in Norway, most of the soil erosion occurs during the snowmelt periods in winter and springtime. Rainstorm erosion is generally of minor importance. However, it is well known that heavy rainfall on autumn ploughed fields could imply substantial erosion.

Previous studies have shown changes in sediment concentration in agricultural runoff in the order of at least 1 to 10 within a time period of a few hours /3/. The sediment delivery to the primary recipient will in many cases behave as flushes. This indicates that major episodes of soil erosion might occur without being recorded by water sampling rutines based on for instance weekly intervals.

This is clearly illustrated in tables 1 and 2. The figures present the maximum error in calculated sediment and phosphorous loss for the eleven month period in 1993 for different point sampling strategies relative to the composite sample results.

Generally, the deviation seems to be larger for sediment loss than for the loss of phosphorous. Irrespective of sampling frequency, there has been a risk of severely underestimating the sediment loss by using point samples. However a few combinations overestimated the sediment loss. For instance, a 4-week sampling routine resulted in a calculated sediment loss ranging from almost -100 % to approximately + 230 % of the "true" figures.

Table 1. The maximum error in calculated sediment loss in 1993 for the different point sampling strategies relative to the composite sample results based on measurements at Stokke and Mørdre catchment.

	1- week % deviation	2- weeks % deviation	4- weeks % deviation
Strategy 1	-84 < >- 46	-93 < > - 42	-100 < > - 7
Strategy 2a	-85 < >+26	-90 < > +72	-100 < > +230
Strategy 2b	-93 < >- 25	-99 < > + 3	-100 < > +100
Strategy 3a	-90 < >- 30	-99 < > - 34	- 37 < > - 26
Strategy 3b	-93 < >- 49	-99 < > - 56	- 46 < > - 45

Table 2. Deviation in calculated phosphorous loss in 1993 for the different point sampling strategies relative to the composite sample results based on measurements at Stokke-1 and Mørdre catchment.

	1- week % deviation	2- weeks % deviation	4- weeks % deviation
Strategy 1	-42 < >+58	+27< >+83	- 8 < > +124
Strategy 2a	- 4 < >+18	-20 < >+30	- 81 < > +143
Strategy 2b	-11 < >- 17	- 2 < >- 23	- 71 < > + 48
Strategy 3a	- 1 < >+18	-45 < >+26	- 60 < > + 41
Strategy 3b	-27 < >+ 7	-49 < >+14	- 64 < > + 12

The tendency of underestimation is not as clear for the phosphorous loss as for the sediment loss. However, the deviation by using point samples is still high, and seems to increase when reducing the sampling frequency. The sediment loss in the two catchments is relatively moderate, and phosphorous loss through subsurface drainage is more dominant here than in catchments with a large proportion of highly erodible fields.

Conclusions

Generally, methods chosen for monitoring sediment and nutrient losses have to be adapted to the dynamics of the processes involved.

In principle, the presented results are only applicable to the catchments and time period included in the investigation. However, they indicate clearly the risk of inaccuracies when using inappropriate sampling strategies and calculation methods methods when estimating sediment and phosphorous loss from small agricultural catchments.

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Fingerprint methods under Danish conditions: sources and delivery routes.

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Abstract

A large range of possible fingerprinting properties was tested in a part of the Gjern å system, Denmark. Because of the glacial geology and the role of secondary sources the discrimination is poor. Precise quantitative results are therefore not yet possible, but the agreement between independent methods supports the qualitative conclusions. Results indicate that almost all the suspended sediment flux in the stream derives directly or indirectly from the surface soil and that the contribution from the subsoil is insignificant. Sediment delivery via macropores and drains is found to play a surprisingly large role.

Methods

The Gelbæk basin, part of the Gjern å system (Fig. 1), was chosen for its small size and because it is already well-documented /e.g. 1, 2, 3, 4, 5/. The range of sources is shown schematically in Fig. 2. Since this was a pilot study a large number of fingerprint properties was tested: N, P, O, grain size, radioactive isotopes, heavy metals and other trace elements. The study took place from autumn

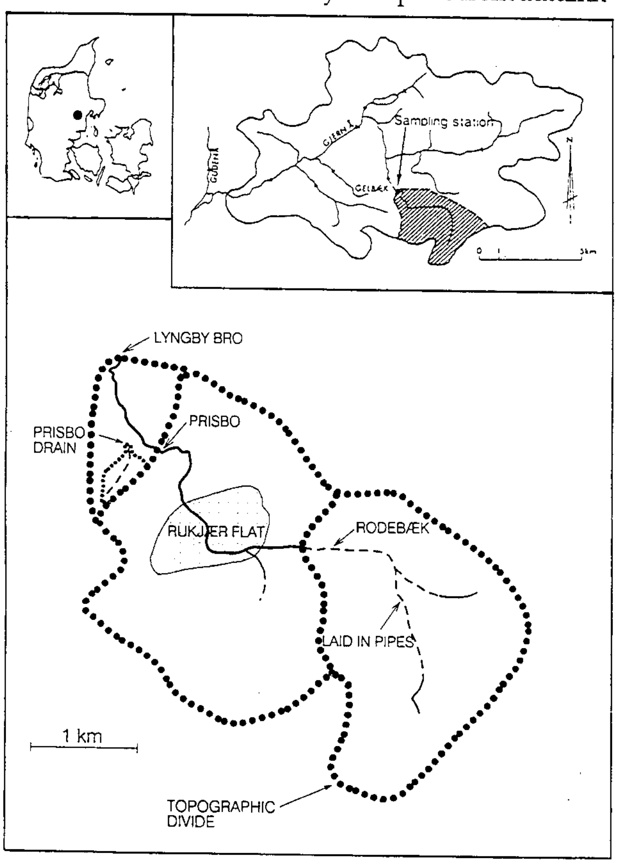

Fig. 1. Maps of the Gjernå system and of the Gelbæk subcatchment.

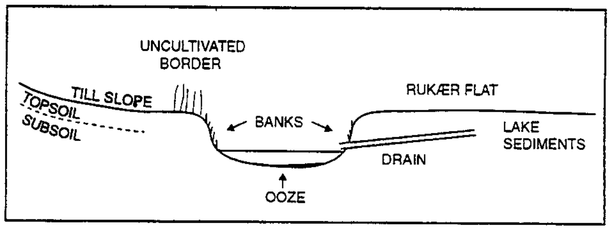

Fig. 2. Schematic illustration of the sediment sources.

In each source environment composite soil samples of up to 20 subsamples were taken at each of several locations and depths. Stream discharge was determined from continuous stage records at Prisbro, later at Lyngby bro, see Fig. 1. Water samples were taken at intervals as short as 5 minutes by a stage-triggered ISCO sampler. Larger samples were collected in jerry cans several times during storms. Occasional 200 litre samples were collected in 2 bathtubs.

Paticulate matter content and loss by ignition were determined according to DS 207 /6/. Suspended sediment for chemical analysis was obtained from water samples by decanting and filtering after storage in the dark at 4°C for one to two weeks. Both suspended sediment and source materials were wet-sieved at 63 µm, dried at 60°C and homogenized by crushing with a tungsten-carbide grinder before chemical analysis.

P was determined as orthophosphate by spectrophotometry (DS 291 = 7), and P-fractions by a modified version of the Hieltjes and Lijklema method /8/. The Kjeldahl method was used for N /9/, and the Leco principle for C /9/. Heavy metals and trace elements were determined by ICPMS /10/. Radioisotopes were analysed by the gamma ray method at the Research Centre, Risø. Grain sizes were determined by laser diffraction /11/.

Results and Discussion

Analysis of the samples from the field sites showed that the discriminating power of fingerprint properties can be poor or even nil (Table 1). Furthermore, enrichment in some cases cannot be adequately explained in terms of the selective transport of particular grain sizes, implying other, unknown, enrichment processes. Fingerprint properties have therefore often been used in pairs as dimensionless ratios, partly to enhance the discrimination, and also on the assumption that the pairs have been subjected to the same enrichment processes. Similarly the potential sources have in many cases been compared in pairs.

Table 1. Examples of fingerprint properties to distinguish between topsoil and subsoil as sources. The properties range from excellent to poor.

		Topsoil	Subsoil	Suspended sed.
Cs^{137}	mean	24.79	0.92	21.11
(Bq/kg)	st. dev.	8.60	0.18	1.60
P	mean	2.00	0.64	3.73
(g/kg)	st. dev.	0.65	0.33	0.77
C	mean	5.36	1.92	7.28
(%)	st. dev.	1.29	1.68	1.46
N	mean	0.49	0.17	0.75
(%)	st. dev.	0.13	0.16	0.22
C/N	mean	10.90	11.83	9.72
	st. dev.	1.39	3.96	1.29
Fe	mean	10.52	7.79	16.63
ppm	st. dev.	9.40	5.09	8.18
Pb	mean	26.95	14.52	45.29
ppb	st. dev.	8.08	3.28	39.62

Topsoil/subsoil: The ^{137}Cs concentration in the topsoil is at least 20 times greater than in the subsoil, but equal to that in the suspended stream sediments. This implies that the subsoil is not a significant source. This conclusion is supported both by the P-fraction data and the C and N concentrations. The subsoils were therefore largely excluded from further study.

Till/Rukær flat topsoils: 22 elements could be used to distinguish between these source groups. 9 of these (Ag, Hg, Mo, Tl, Li, Sc, Sn, Zr, Y) were used in pairs in order to improve the discrimination between sources. An example is shown in Fig. 3.

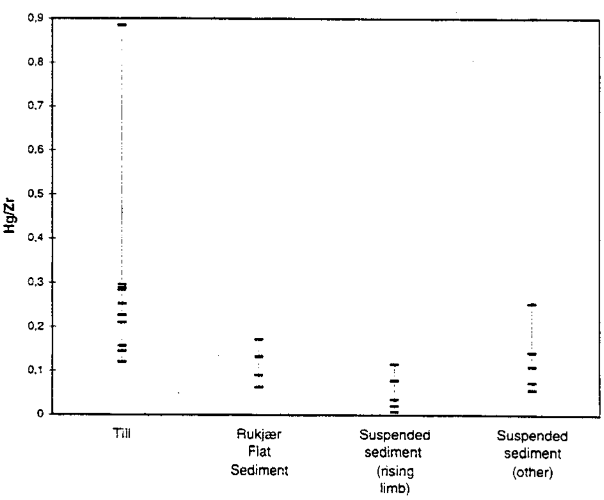

Fig. 3. An example of the use of trace element ratios to distinguish between sediment sources.

A comparison with the suspended sediment at the Prisbro station showed that the suspended sediment flux is dominated by the Rukær source. The lanthanides, which clearly differ from other trace elements (see below), suggest that this Rukær dominance is greatest on the rising limb of the hydrograph. This is hardly surprising, since Rukær is not far upstream from the sampling station. (This proximity may also account for some of the apparent overall dominance of the Rukær source: signals from poorly discriminated distant sources can be masked by the large amount of noise). There is little doubt that Rukær is a major source, and substantial bank erosion was observed in that reach, but the fingerprint calculations are not quantitatively reliable.

Ooze/surface erosion: The stream bottom ooze has an average Fe- and Al-bound P (Ox-P) content of 1.17 mg/g as against 0.5 mg/g in the topsoil. During the first storm in the autumn of 1992 the Ox-P content of the suspended sediment was as high as 2.25 mg/g, as against 0.89 mg in later storms. Given the notorious difficulty in sampling the ooze this only suggests that the ooze played a greater role during the first storm. However, this view is supported by direct observation of the stream bed which was virtually clean after the one event, as well as by numerous studies of resuspension in the Gjernå system /3, 4, 5/ and elsewhere.

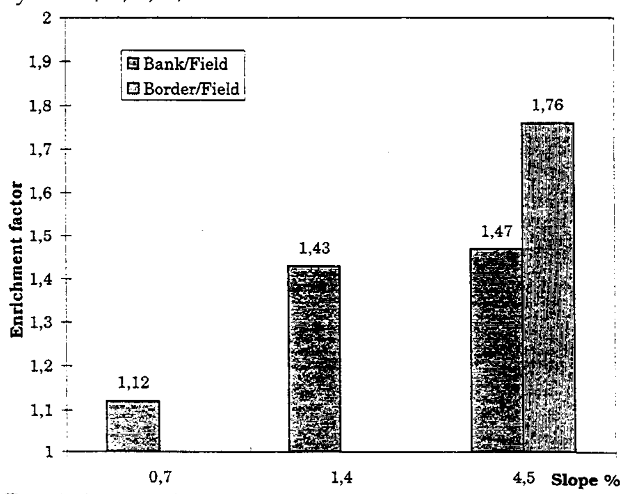

Fig. 6. Average bank and border enrichment for all trace elements as a function of the slope of the adjacent fields.

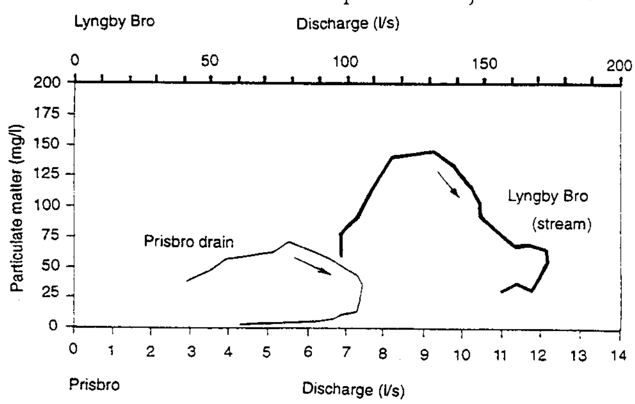

Fig. 5. Sediment concentration hysteresis at the Lyngby Bro station and in the Prisbro drain, 26-27. Sept. 1993.

Surface erosion/lateral erosion: The banks and borders contain higher concentrations of most of the fingerprint tracers than either the suspended sediment or the ooze. This enrichment can thus not be explained simply by the accumulation of ooze or suspended sediment during floods or during stream maintainance, though these no doubt play a role. Selective enrichment from surface wash is almost certainly a major source, a view supported by the fact that enrichment is greatest at the foot of steep slopes, Fig. 6. Whatever the ultimate sources, long-term storage is involved and the banks and borders act as secondary sources when subjected to lateral erosion and collapse. Major collapse up to 10 metres long was, in fact, observed after a storm, as well as smaller ones spaced as close as 5-10 metres. Some high sediment and tracer concentrations late in the storm hydrograph might be accounted for by collapse during the recession /12/.

Sediment delivery from drains: The storm hydrographs exhibited the usual hysteresis with high sediment concentrations during the rising limb (Fig. 5). This hysteresis occurred not only in the first storms of the autumn, when resuspension of river bottom ooze can play a major role, but also in the later storms, when direct observation showed that all the ooze had been flushed out. Resuspension is therefore not a general explanation for the hysteresis. Similarly, during several of the storms there was little evidence of surface wash flowing direct into the stream. In contrast, both fingerprint properties and direct measurements of the Prisbro drain show that a major part of this sediment delivery takes place through drains. Fingerprint data show that the source of almost all of this sediment is the topsoil. Furthermore, a correlation with rainfall intensity suggests that rainsplash may be a major mobilizing process.

The Prisbro data can be used to calculate the total sediment and phosphorus contribution from drains. Admittedly the monitored area is very small (13.3 ha) but identical concentration data were obtained from the 7 km^2 Rodebæk system, the major part of which is drained. Unfortunately, there are no discharge data from Rodebæk, so the Prisbro data have to be extrapolated. Such a scaling-up is very sensitive to the underlying assumptions. Assuming drain contributions from the entire 60% drained part of the Gelbæk basin then during the 4 monitored events in 1993/94 out of a total export of 2.7 tons sediment the drains supplied a maximum of 2.3 tons (86%) and 9 out of 12 kg P (75%). This is almost certainly unrealistic since substantial parts of the artificial drainage system do not become active till later in the season. Further, there may be some storage en route. A minimum estimate results from assuming only 42% of the area as actively drained and a delivery ratio of 75%. This yields respectively 28% for sediment and 24% for particulate during these four storms.

Multiple sources: It was stated above that there are serious difficulties in applying mixing models to the fingerprint data in this catchment, and these difficulties increase with the number of sources used in the model. Nevertheless an attempt was made to use particulate matter concentration and total phosphorus contents in a three-source model of the 4 first autumn storms of 1993. The calculation suggested that 43% of the sediment flux derived from the topsoil, 11% from the subsoil, and 46% from the stream bed. Of the surface sediment 36% arrived via drains while the remaining 7% together with the 11% subsoil must be ascribed to lateral erosion and collapse.

Conclusions

Ideally fingerprinting methods involve properties which are specific to particular sources which themselves are homogeneous. Such conditions cannot be expected in the glacial and proglacial geology of Denmark. Only the topsoil-specific shortlived radioisotopes satisfy these requirements. For many of the other properties the differences between the sources are small and/or the variations within each source are great (see table 1). To this must be added the problem of enrichment, in which processes other than grain-size-specific sorting are involved. Several of the sources, such as banks, borders and stream bed ooze, are secondary, that is, they are buffers which vary in time. The conditions needed for accurate mixing models are therefore almost entirely absent.

The question therefore arises whether there is any scope for using tracer methods. The answer is yes, and is tied to the very problems which raised the question. Given the very poor data sets, new information is valuable, even if it is only semi-quantitative or even purely qualitative. Such information can be used to simplify the complexity of the conceptual system (e.g. by showing that subsoil is a very minor source) and to focus attention on specific sources or delivery paths (e.g. banks, borders; flow in drains and soil cracks). Further, although individual methods only produce imprecise and doubtful results, the broad agreement between results from separate methods used in this study suggests that the collective output permits some reliable conclusions.

Since it is desirable to use several methods in conjuction it follows that no single method can be recommended at the expense of all others. In particular, the cumbersome and expensive trace element analyses are probably not justified by the quality of the output, but some avenues (such as the peculiar behaviour of the lanthanides, see Fig. 4) still need to be explored. With the exception of the very useful radio isotopes it seems most useful to concentrate on the results of traditional laboratory analyses.

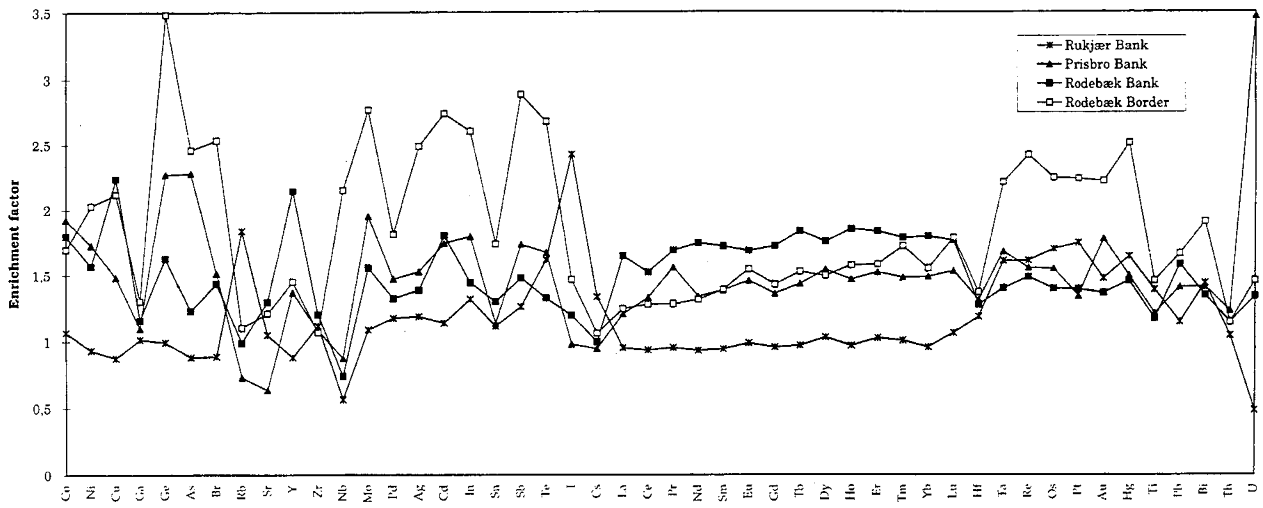

Fig. 4. The enrichment of trace elements in three banks and one border as compared with the content in the topsoil of the adjacent field. Note the anomalous behaviour of the lanthanides.

Apart from the conclusions about fingerprinting methods, this study has highlighted several problems which need further examination. The first of these is the remarkable importance of the delivery of surface sediments via macropores and drains. Much of the cultivated area of Denmark is drained and it is the aim of good farming practice to reduce surface runoff to a minimum, even in areas of heavy till. The soil structure that permits this interflow obviously also permits the transport of fine sediments. The processes involved are not clear. Hydrochemical studies suggest that even rapid interflow is of the piston type /13/, new water replacing old in the soil, but the sediment content suggests that direct throughflow is a major factor. Similarly, the role of temporary storage in soil cracks and in drains needs to be investigated.

The second problem is the role of buffers in general. Depending on the time scale the river bed ooze or the banks and borders can either be considered as secondary sources or as temporary buffers in the delivery system. Present environmental and agricultural policy will almost certainly lead to longer and wider uncultivated borders, so in the long term this "source" (which is also a sink) will play an increasing role in the flux of sediment and of P.

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Flow proportional water sampling - a method to estimate phosphorus transport

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Abstract

This project compares three sampling methods for the estimation of phosphorus (P). Flow proportional sampling, time proportional sampling and manual weekly discrete sampling. In cooperation, The County of Funen and Hedeselskabet have site-tested an intelligent datalogger system in the River Lillebæk on the island of Funen. The datalogger was programmed to collect flow proportional water samples in an unregulated river. During the test a water sampler collecting time proportional one hour samples was also operated. Both sampler systems were took composite samples with a seven days' cycle. In addition, water samples were collected manually with an interval of seven days. All samples were analyzed for ortho-phosphate (DRP), total-phosphate (TP) and suspended solids. The tests showed that by using the flow proportional sampling method rather than the time-proportional sampling method, the P-transport was 15 to 20% higher. Manual discreet weekly sampling strategy also resulted in transport levels lower than the flow-proportional sampling method and the results proved to be highly sensitive to the time of the manual sampling. Transport of suspended solids yielded a corresponding tendency when collected with the sampling strategies.

Introduction

In connection with the Action Plan of the Aquatic Environment (Danish) a nationwide monitoring programme was initiated in 1988. The Counties are responsible for a major part of this programme. In rivers the monitoring programme includes 262 stations with a typically sampling strategy of 12-26 manual samplings a year. The monitoring tasks include transport calculations, determination of load sources and monitoring of the effects of initiated reduction measures. When making P-load calculations, the sampling strategy appears insufficient.

When estimating the contribution and influence on the total P-load from the upstream sources, it is necessary with an accurate determination of the P-transport out of the basin. In spite of some uncertainties in the discharge calculations, these are negligible compared to the uncertainties introduced by the lack of representability, by which water samples are collected for nutrient concentrations. An example is manual or time dependent sampling.

In this study we compare transport calculations based on three sampling strategies in rivers:
- flow proportional sampling (automatic sampling)
- time proportional sampling (automatic sampling)
- traditional manual sampling

The flow proportional sampling method is based on automatic water sampling with a sampling frequency proportional to the flow. On-site discharge calculations based on measured water level transformed by a stage discharge relation are used to start an automatic water sampler.

Test site

The stream Lillebæk drains an area of 4.7 km^2 in the south eastern part of the island of Funen (Fig. 1). The drainage basin has been selected as one of the catchments with great favour in the Nationwide Monitoring Programme. Load from cultivated areas and single habitations contribute a considerable part of the nutrient transport to the river.

The county of Funen has operated a discharge station in this river since 1988. As key figures the mean discharge is 40 l/s, the maximum daily mean is 700 l/s and the minimum approx. 2 l/s. (period 1990-1994) /2/. At this location the P-transport has been calculated from manually collected water samples until the time proportional water sample method was introduced in 1993.

The river profile is affected by bed erosion and consequently changing stage discharge rating curve. In return no vegetation, due to shade from surrounding trees, influences the discharge capability. The stage discharge rating-curve is considered as stable within six months periods.

Method

The measurement system for collecting flow proportional water samples consists of a programmable datalogger, a pressure transducer for measuring the water level and a software programme that handles necessary information to a water sampler. The setup is shown in fig. 1, where the flow- and time proportional measurement systems were positioned site by site with water intake at a joint location in constantly flowing water.

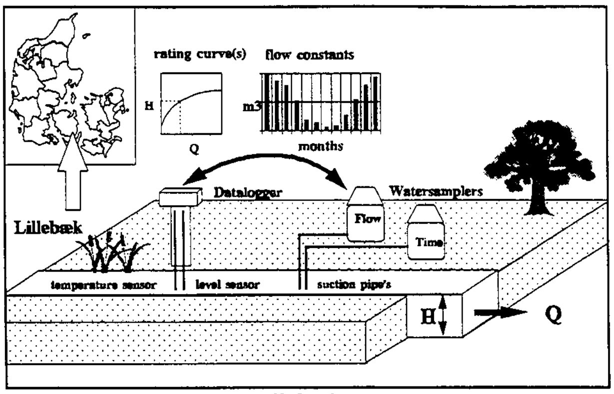

Fig. 1. Test site setup, Lillebæk

Every 15 minutes the mean of one minute recorded stage (m) is logged. The actual discharge (l/s) is calculated by the stage discharge relationship. Finally, the flow (m^3) of the previous 15 minutes, passed a cross section in the stream is calculated.

Twelve flow constants are predefined in the software, one for each month of the year. The constants are calibrated to known flow regime to meet the demand - taking optimum amount of water samples in low flow and peak flow situations without overfilling the collecting capacity. The number of samples each day should be high enough to fulfil the description of a possible diurnal variation in concentration.

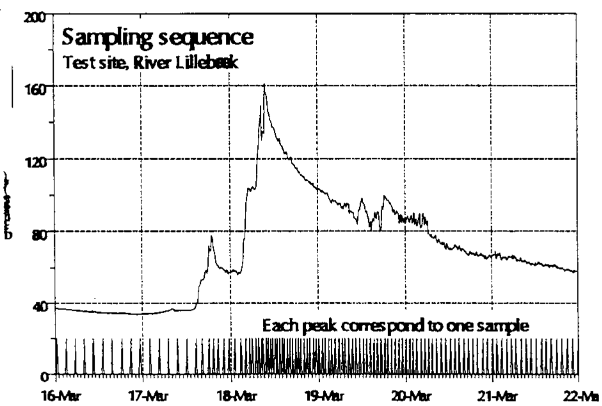

Fig. 2. Selected sampling sequence, March 1995

The calculated 15 minute flow is cumulated until the exceed of the actual flow constant. The exceed triggers a signal to the automatic water sampler and a sample will be taken. An excess deficit flow is stored and used in the next calculation period. An example of a sampling sequence in March 1995 during a rain event is shown in figure 2.

On-line connection by telemodem is used to update the stage discharge relationship, optimize the flow constants if necessary, to give a status of the amount of collected water, to give a status on the discharge, serve as alarm function or to reprogramme the datalogger if required. More detailed information on the measurement system is given in /3/.

One litre of the composite samples taken both the flow and time proportional way, as well as a one litre manual sample, were analyzed. The flow- and time proportional determined transports were calculated as the measured weekly period-mean concentrations multiplied with the weekly flow. The manual sampling transport was calculated as mean daily transport by using the C-linear interpolating method. This method interpolates the daily concentration linear between two weekly analyses and multiplies with the daily mean discharge.

Results

The total P-transports during the test period October 1994 - May 1995, compared for the three sampling methods, are summarized in Table 1.

Table 1

	flow method	time method	manual method
DRP (Ortho-P) (kg)*	146*	127*	136
TP (Total-P) (kg)	328	278	271
Susp. materials (kg x 10^3)	77	48	67

* The DRP results must be considered with a certain reservations when comparing the composite sampling methods with the manual sampling method. Durability tests have shown a reduction in orth.-p when storing the water samples. /4/. In this test the composite samples have been stored for seven days, while the manual sample has been stored for only one day before analyzing.

The transported DRP and TP were 15 and 18% higher, respectively, based on the flow proportional samples, compared with time proportional sampling. Comparing the flow method with the manual method the corresponding figures were 7% and 20%. The resulting transport of suspended solids supports the tendency, comparing the three methods.

The accumulated transport of DRP calculated the three sampling methods is illustrated for the test period with the discharge in figure 3. TP and suspended solids are illustrated in the same manner in figure 4 and 5.

Figures 2 to 4 show higher transport, especially in peak situations based on the flow proportional method. The manual method shows the lowest transport during most of the test period, but also periods with the highest transport e.g. the first week of February, when the cumulated transport curve turns greater than the time proportional sampling curve.

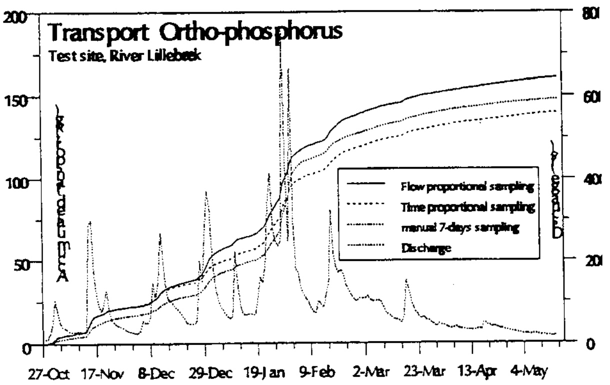

Fig. 3. Transported ortho-phosporus

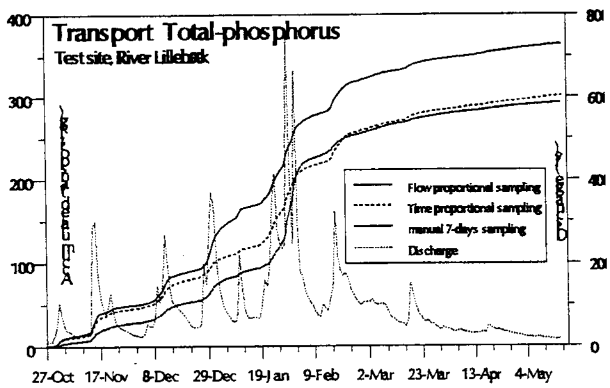

Fig. 4. Transported Total-phosphorus

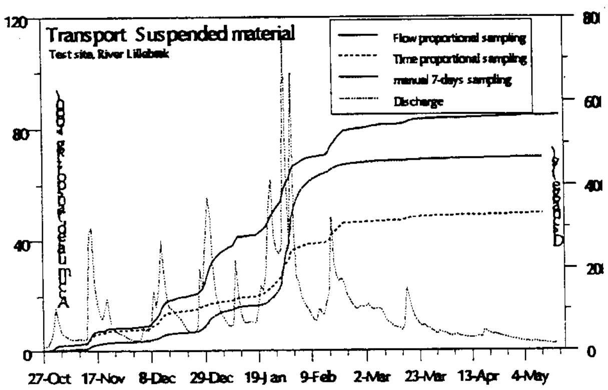

Fig. 5. Transported suspended solids

Calculated on a monthly basis, the manual sampling strategy indicated both positive and negative differences compared with the other sampling methods.

Discussion

This investigation does not exactly quantify the difference in calculated transport, when using the flow criteria instead of the time criteria as sampling strategy. As to the short test period and with reference to expected minor differences in larger rivers /1/, it seems that time proportional sampling in Lillebæk underestimates the P-transport. Manual sampling in Lillebæk is proved to be sensitive, due to short term variations in concentrations.The sensitivity of the manual method is obvious when comparing with the other methods on a monthly basis, where both over- and under estimations occur.

Flow proportional sampling is, if used correctly with respect to the discharge calculations, a method with full control of every sampling and its representability. Every sample is weighted in the right scale to the total discharge. It is so far only a matter of how detailed the 'in between periods' to the samples are described. In this test period there were no days with less than eight samples to describe the variation in concentration, and these days were low flow situations on the discharge recession. It can be a question how representative a water sample is to the former period, especially in a time sequence with major variations in the concentrations, as during a storm event. The project does not test these circumstances. Further, the equipment has no limits concerning the sampling frequency during a storm event, as long as the same sampling strategy is fixed since dealing with one composite sample.

Vital for the sampling representability is the certainty of the direct calculations of the discharge. In rivers with changing stage discharge relationship caused by aquatic vegetation or changes in the river bed, a frequent update of the discharge capability is important. In rivers with permanent weirs or other permanent constructions, only few discharge calibrations and controls are necessary.

Conclusion

This investigation resulted in higher calculated phosphorus transport, when using the flow proportional sampling method, compared with time proportional and manual method. The higher/increased??? transport is especially related to major precipitation/discharge events.

This indicates that the traditional sampling methods underestimate the transport and that a flow related automatic sampling method should be used to determine a more correct picture and minimize the uncertainties of estimated P-transport.

The benefits of the flow proportional sampling method should be considered in the approaching revision of the Danish Aquatic Monitoring Plan.

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Transport and fate of sediments and phosphorus in an equatorial catchment: the influence of geology, climate and land use

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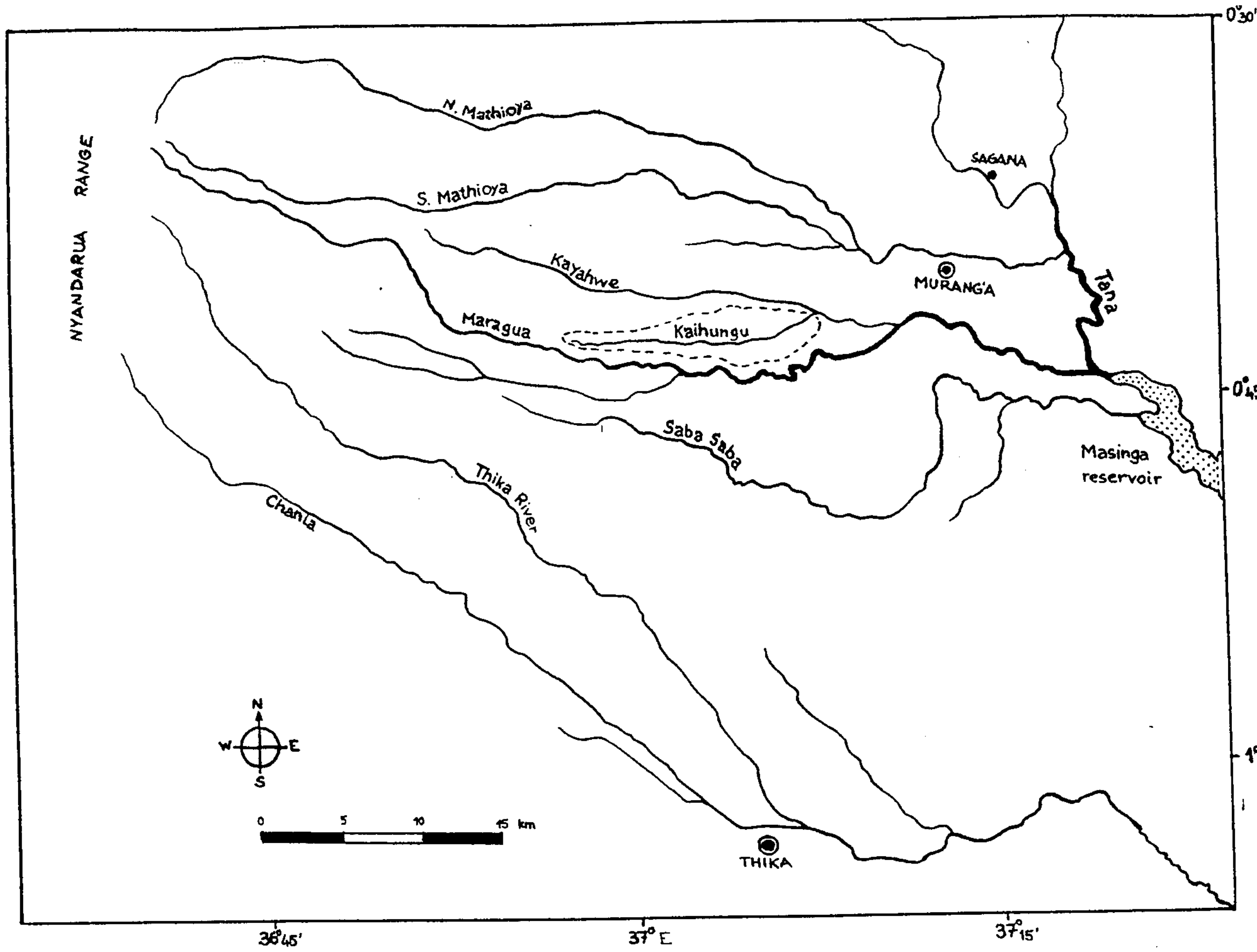

Fig. 1. Location of the Kaihungu catchment at the mid slope between the Nyandarua Range (>2,000 m a.s.l.) and the lowland Masinga reservoir (1,056 m a.s.l.).

Introduction

The impact of fertiliser nutrients on terrestrial agricultural and aquatic systems is linked to soil properties and transport pathways within the ecosystem. Studies on phosphorus (P) export from agricultural land in temperate regions stressed the need for an understanding of the main factors implicated in P mobilisation and transport for an efficient management of P fluxes in agricultural systems /1,2/. In tropical Africa the quantitative chemical relationships between land and water are as yet virtually unstudied /3/. Mineralogical research, fertiliser trials and greenhouse tests have proved that phosphorus is low in Kenyan soils and is commonly the primary limiting nutrient /4,5/. Phosphorus limitation is common in tropical agricultural systems and relates to specific properties of tropical soils /6/.

This article presents results of a study on P export from a small agricultural upland catchment in the Kenya Highlands. The study is part of a wider research project on soil erosion, nutrient runoff and reservoir limnology carried out in the Upper Tana River Catchment (UTRC) between 1991 and 1993 /7,8/. This included the intensive monitoring of an upstream 24 km^2 sub-catchment (this paper), a simultaneous study of 10 rivers in the wider UTRC (7335 km^2) and the monitoring of in-lake P dynamics in the lowland Masinga reservoir /8/.

Land character and management

The Kaihungu stream (17.5 km in length) rises on the eastern slopes of the Nyandarua Range in Murang'a District (Central Province). This 24 km^2 catchment is part of the Maragua river basin (370 km^2) which represents one of the most productive and densely populated regions in the country (Fig. 1). The landscape is characterised by narrow valleys with steeply dissected ridges forming a dendritic drainage with numerous small streams with moderate to steep gradients.

The Nyandarua range is formed by Tertiary and Quaternary basalts and basaltic agglomerates (phonolites, trachytes and olivine basalts) associated with the volcanism which created the Rift Valley. The soil profiles are deep, rich in clay and uniformly red in colour. Humic and eutric nitosols are common in the Kaihungu and in the Maragua Catchment. These friable clays, have low erodibility and exhibit a moderate water retention capacity. Nitosols represent an intermediate stage of ferralitic weathering and develop typically on well drained steep slopes. When stripped of their vegetation cover they tend to dry out, harden and eventually erode forming deep gullies along the prevailing slopes (20-30°). Population densities exceed 600 inhabitants·km^{-2} with an estimated growth rate of 3%. The largest part of the population is distributed within the farmland with no large urban settlements. As a consequence the whole Maragua catchment is covered by an intricate network of footpaths, unsurfaced roads, cattle tracks and other bare ground areas which tend to widen and grow into gullies.

The catchment comprises the Upper Coffee Agro-ecological Zone (1600 m a.s.l.) located between the Tea Zone above 1700 m and the Lower Coffee Zone at the foot of the volcanics (1200 m a.s.l.). Farmers grow a mixture of cash and subsistence crops such as tea, coffee, maize and beans. Fertilisers are applied on cash crops. More than 50% of the catchment surface is cultivated. Agroforestry is practised on some of the steeper slopes as a measure to reduce soil erosion. The rains are highly erosive and occur during two main rainy seasons (April-June, October-December). Mean rainfall is around 1500 mm·year^{-1}. Floods occur mainly at the beginning of the Long Rains. During the study period the 1st of May was set as the beginning of the water year. The baseflow of the Kaihungu stream (0.2 m·s^{-1}) is fed by permanent spring situated higher in the catchment. The yearly water load is determined primarily by storm runoff conveyed to the channel by numerous steep gullies. The maximum measured discharge during the study period was 5.4 m·s^{-1}.

Methods

Water samples were abstracted by wading into the river and dipping a bottle at the mid channel just below a riffle to ensure good mixing. The sampling station was set above the confluence of the Kaihungu and the Kayahwe streams at the eastern boundary of the catchment. Samples for the solute chemistry were filtered in the field (pre-washed Sartorius Cellulose Acetate filters, 0.45μ) and immediately analysed employing standard techniques adapted to a portable spectrophotometer (Hach Drel-2000). Soluble Reactive Phosphorus (SRP) was often undetectable by this technique and had to be measured on pre-concentrated samples after hexanol extraction (also carried out in the field). Samples destined to TDP analysis were filtered in the field and immediately fixed with 0.1% Chloroform (analytical reagent grade). Total Phosphorus (TP) and Total Dissolved Phosphorus (TDP) were measured on evaporated samples digested in a muffle furnace at 500°C /9/. Particulate Phosphorus (PP) was estimated from TP-TDP. Suspended Solids (SS) were estimated by filtering samples in the laboratory through pre-weighed, oven-dried GF/C glass fibre filters; then drying the filters at 105°C and weighing. Sampling frequency was weekly to bi-weekly during low runoff conditions and daily to weekly during rainy seasons. During storm events we carried out hourly sampling using an automatic liquid sampler. By the end of the project 75% of the TP and SS samples collected from the catchment were obtained during the rainy seasons. The data set collected during the 2 year-long sampling programme totalled 141 and 214 measured concentrations for TP and SS respectively.

Instantaneous discharge was estimated by the velocity-area method using an Ott current meter. Discharge measurements were related to a stage fixed in the river bed. Stage readings were taken daily during low runoff and hourly during storm events. An automatic stage recorder was set by the river bank but continuous measurements could not be ensured due to sediment deposition within the stilling well.

For load calculation, the Kaihungu discharge data were interpolated to obtain a daily discharge series for the period May 1992-May 1993. Instantaneous P and SS loads (g·s^{-1}) were computed by multiplying single measured TP and SS concentrations (mg·l^{-1}) by the discharge value (m^3·s^{-1}) corresponding to the sampling day. Total P and SS rating curves were produced by linear regression of logarithmic load values against discharge. Estimates of yearly yields (t·km^{-2}·year^{-1})of TP, SS and discharge for the Kaihungu catchment were derived applying the rating curve equation to the extrapolated daily discharge data.

Results

Chemistry of runoff

The general composition of runoff reflected selective leaching caused by the interaction of tropical rains with mature, leached ("lateritic") tropical soils /8/. Soluble Reactive Phosphorus and Total Dissolved Phosphorus ranged between 0 and 30, and 15 and 50 μg·l^{-1} respectively. Suspended Solids varied between 3 mg·l^{-1} during very low runoff to more than 5 g·l^{-1} during floods. Table 1 illustrates median river water concentrations during the dry and the rainy season for some relevant parameters.

While TP values were dependent on episodic events of high transport of particulate material in suspension, SRP and TDP concentrations showed a relatively even distribution reflecting the buffering of phosphates by adsorption onto clay and organic matter.

Table 1. Median concentrations of suspended solids (SS), total phosphorus (TP), total dissolved phosphorus (TDP), soluble reactive phosphorus (SRP) and conductivity (Cond) in the Kaihungu stream.

	Dry Season	Rainy season
SS($mg \cdot l^{-1}$)	7.5	224.0
TP($mg \cdot l^{-1}$)	0.043	0.248
TDP($mg \cdot l^{-1}$)	0.020	0.024
SRP($mg \cdot l^{-1}$)	0.008	0.012
Cond($\mu S \cdot cm^{-1}$)	150	139

The median values for SRP are comparable to the average SRP concentration of rivers in humid tropical regions estimated as 12 $\mu g \cdot l^{-1}$/10/. The measured concentrations of TDP accounted, on average, for 10 to 25% of the concentration of TP. This proportion is under the influence of dilution and the effects of increased P sorption during high discharge periods. During baseflow SS concentration was low and TDP was sometimes as much as 50% of TP. Such periods, however, had little influence on the annual P export budget.

P transport during storm events

Due to the lack of a trigger that would control the start of the automatic sampler, we missed the rising limb of the hydrograph in most storm events and we can only comment on the descending limb. During the initial phase of the storm the TP and SS concentration apparently increased in parallel with discharge and then continued to increase as the peak in discharge was reached (Fig. 2). As a result the concentration-discharge relationship describes an anticlockwise hysteresis loop (Fig. 3). This is consistent with transport-limited erosion supported by high availability of detachable material. This result was observed during several storms. The width of the loop varied with the specific intensity and duration characteristic of each storm.

Anticlockwise hysteresis loops can be explained as follows; sediments with high TP content are washed off the top of the soils and transported during the rising stage of the hydrograph, then the discharge peak mobilises coarse materials. These particles are less chemically active and contain a lesser amount of adsorbed P. The P content of sediment decreased during the peak in discharge and recovered its original level only after the storm runoff had passed (Fig. 4). High energy flows erode the soil less selectively (i.e. with a lower enrichment ratio). The observed effect could also be explained with the arrival into the stream channel of sediment originating from a distant source (delayed transport). These could be sediments originating from paths and bare ground areas, likely to be characterised by lower P concentrations than riverbank sediments or sediments originating from the erosion of topsoils.

Inter-storm variations are related to the seasonal development of the vegetation cover and are influenced by the duration of the storm event. A long, sustained rainstorm will tend to deplete transportable sediment producing clockwise loops; the TP content in such occasion should be comparatively lower. In the Kaihungu catchment and in the whole UTRC sediment TP content was negatively correlated to discharge /8/.

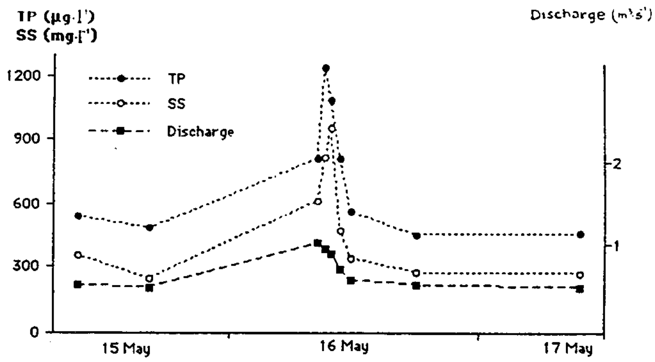

Fig. 2. Concentration of total phosphorus (TP), suspended solids (SS) and discharge during the storm event of the 15/5/93, Kaihungu stream.

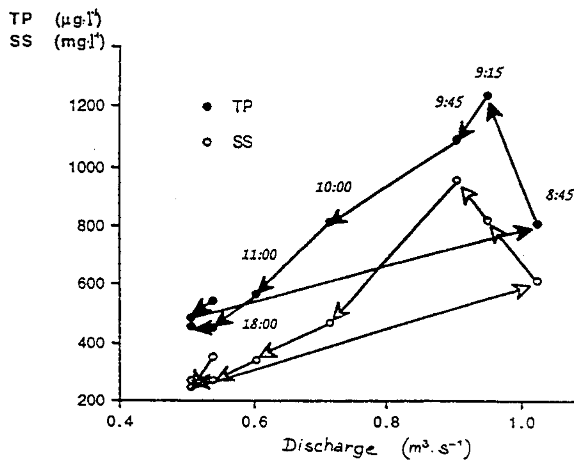

Fig. 3. Hysteresis loops described by the concentrations of total phosphorus (TP) and suspended solids (SS) plotted against discharge during the storm event of the 15/5/93, Kaihungu stream.

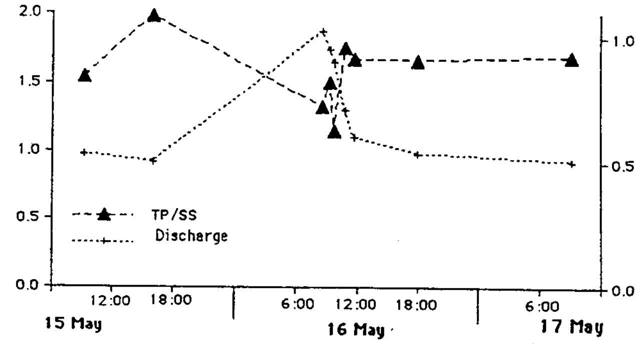

Fig. 4. Change in the sediment phosphorus content during the storm event of the 15/5/93, Kaihungu stream.

Suspended sediments were largely composed of kaolinitic clays and had a P content close to the specific adsorption capacity of kaolinite estimated to 1.2-1.3 mg P·g^{-1} /11/. In the soils of the UTRC the abundance of Al hydroxides is highly correlated to P-sorption while clay is plentiful and does not limit P-sorption system /4/. Fixation onto the aluminium and iron coatings of kaolinite is the main mechanism of P sorption, while organic matter is scarce and is not thought to play a significant role /8/.

Loads and yields

Results of the calculations are summarised in Table 2.

Table 2. Yearly total phosphorus (TP) and suspended solids (SS) areal yields and runoff in the Kaihungu subcatchment

SS yield (t·km^{-2}·year^{-1})	161
TP yield (kg·km^{-2}·year^{-1})	109
Runoff (mm)	346

Correlation coefficients (r^2) between TP, SS instantaneous loads and discharge were respectively 0.88 and 0.89. Such high correlation coefficients are partly due to the autocorrelation inherent to the estimate of loads by the rating curve technique as discharge is a component of both axes of the linear regression plot. Our results are supported by highly significant correlations between SS, TP concentrations and discharge (for SS $r^2 = 0.69$, n = 214, for TP $r^2 = 0.58$, n = 141).

The coefficient of variation of the 30-year monthly cumulated readings of discharge for the Maragua river is 49%. From historical records of discharge and SS concentrations we estimated a coefficient of variation of the yearly SS yield of 68%. When such coefficient is applied to the TP the estimate of the Kaihungu yearly TP yield becomes 35-183 kg P·km^{-2}·year^{-1}. During the course of the study period the discharge of the Maragua river was not significantly different from the long-term mean.

Infrequent heavy discharge events were responsible for the largest amounts of sediment transported. As a consequence of the fast variation of discharge not all events were monitored; the estimates of TP and SS load are considered conservative. Ten percent of the TP load (1992-1993) was transported in one single day when discharge exceeded 2.4 m^3·s^{-1}. In the case of SS a smaller fraction of discharge transported an even higher proportion of load. During the same water year 40% of the TP load and 64 % of the SS load were transported within 15 days of high discharge. The relationship between load and discharge however should be questioned. Monthly changes in TP and SS loads in the river during the year are influenced by the alternation of periods of deposition and resuspension within the catchment. Sediment deposition during baseflow creates a store of SS and TP that is easily mobilised by the first coming flood. Consequently sediment transport is expected to depend on the length of the previous accumulation period in the stream channel /12/.

Changes in the rates of erosion during the year due to vegetation cycles are also considered responsible for reduced sediment loads /1/.

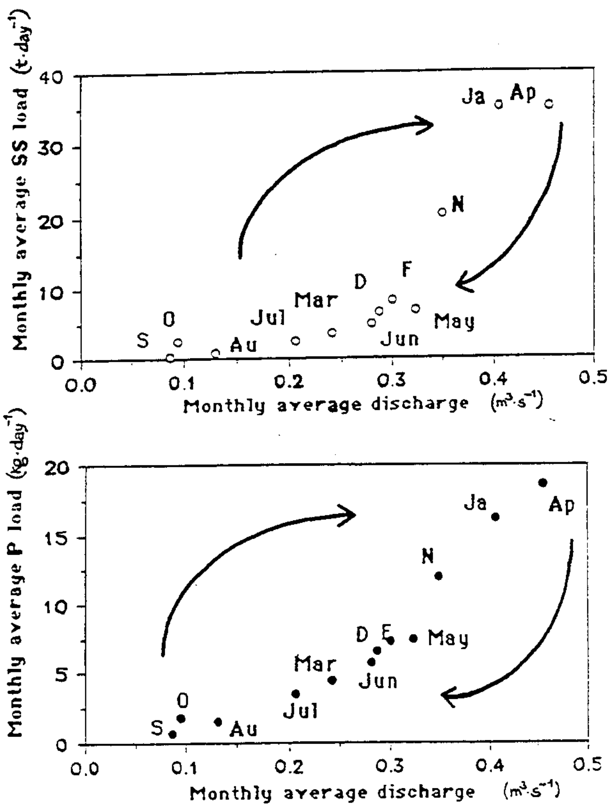

Fig. 5. Progression of the monthly average suspended solids (SS) load (t·day^{-1}) throughout the year 1992-93 in the Kaihungu stream.

In figure 5 the progression of monthly load values appears to be mainly a reflection of the seasonality of rainfall. A steady decrease during the dry season from July to October is followed by an increase in November (Short Rains) and then again a peak in January. February and March are both above the dry season levels (July-October). Even under close scrutiny our data were not able to show a significant seasonal change in SS or TP load that would not be explained by a change in runoff.

Discussion

Although the Kaihungu catchment enjoys relatively high rainfall, the large availability of sediment to transport gives it throughout the year the characteristics of a transport-limited, rather than sediment-limited, system. Soil conservation practices are not efficient in limiting the delivery of sediment to the river system. The cause is related to a wrong identification of targets. Unsealed roads and bareground areas provide unrestrained sources of sediment while agricultural plots are controlled /13/. This mechanism is consistent with the lack of sediment exhaustion at high discharges, the observation of anticlockwise hysteresis during storm runoff and the lack of seasonal exhaustion of transportable sediment observed in the field and by comparing the progression of monthly load with runoff.

Phosphorus export in the Kaihungu catchment was higher than in most watersheds of igneous and sedimentary lithology in North America studied by Dillon and Kirchner /14/ with the exception of those under the direct influence of urban pollution. It was comparable instead to their estimate of mean P runoff from basins of volcanic origin in Washington state (USA). Grobler and Silberbauer also stressed the importance of catchment lithology in determining P export /15/. Volcanic catchments give consistently higher P export loads than catchments characterised by granitic or metamorphic bedrocks but often lower P loads than erodible sedimentary catchments. This is related to the slope of the terrain found in catchments of volcanic origin, but it is also due to geochemical processes. In the UTRC significant P export was due to groundwater naturally rich in P /8/. The chemical composition of these springs is an expression of the intense chemical weathering undergone by these relatively porous rocks under the warm and humid tropical climate. In our study these P sources could not be easily distinguished from the impact of fertiliser P. The consumption of fertilisers in Kenya is far higher than the average consumption in other countries of the African continent. The P balance of Kenyan agricultural systems can be considered positive as far more P is put into the agricultural system than is exported as produce every year /16/. A large part of the added P is buffered by the soil adsorption capacity. Studies of P sorption potential in soils of the UTRC revealed moderate sorption indices /4/. Nonetheless these deep draining profiles favour P sorption. Fertilisation trials on nitosols in the Embu district (southern slopes of Mount Kenya), showed that application of superphosphate greatly increases the P pool occluded in Al and Fe (hydr)oxides /17/. Intensive weathering under the sustained high temperature which characterises the tropical climate, conduces to the formation of abundant amorphous Fe and Al and oxide-coated clay which in turn tend to store P. A study including 250 soils collected in 35 countries confirmed common P deficiency to crops in ferralitic and ferruginous soils /6/.

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Phosphorus loss to water from agricultural areas underestimated

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Abstract

Phosphorus transport from two small cultivated catchments has been estimated using very frequent sampling technique (sampling every hour throughout the year). Transport of total phosphorus (TP) was 0.5 kg P ha^{-1} - 1.3 kg P ha^{-1}, which was 1.5 - 2.5 times the transport calculated from 18 - 26 point samples. The difference between the transport calculated from frequent sampling and from point samples was due to a higher fraction of particulate phosphorus (PP) in the transport calculated from frequent sampling. High PP concentrations typically occurred during storm flow events of short duration.

Transport of dissolved inorganic phosphorus (DP) was 0.15 kg P ha^{-1} - 0.36 kg P ha^{-1}, and calculations bases on either frequent sampling or point samples only differed slightly.

The P load related to cultivation was the dominant source compared to both the natural background load and the load from sewage from scattered dwellings without sewage systems.

During a stormflow event in five streams more than 80% of PP was either organic bound phosphorus or associated to ferri- or aluminium compounds. These fractions can be readily transformed to phosphate in lakes and coastal waters and influence the eutrophic state.

Introduction

Most lakes in Denmark are limited by P /1/. Therefore it is essential to restrict the P-sources to improve the eutrophic state of the lakes. High loads of P from unpurified or partly purified sewage water has hitherto been the main cause for eutrophication. Improved treatment of sewage has reduced the loadings from this source, ie. in Aarhus County by 75% during the period 1989-1994 /2/. Consequently non-point P-sources including phosphorus runoff from agricultural areas, loadings from scattered dwellings without sewage systems and natural background load now plays a dominant role.

The P contribution from these diffuse sources has hitherto been underestimated due to the sampling technique. The sampling frequency in most Danish streams has typically been 18-26 times a year from 1989 - 1995.

In 1993 a new sampling technique including very frequent sampling by means of automatical samplers was implemented at a number of monitoring stations /3/. The monitoring stations represented streams in small cultivated catchment areas without wastewater outlets from cities. The aim of this investigation was to obtain a better estimate of the transport of P from these small catchments dominated by agriculture. The results were expected to be useful when calculating loadings from similar catchments elsewhere. Secondary a true estimate of P load was a prerequisite to have an estimate of the agricultural load, as agricultural load is calculated as the difference between transport and the other sources.

Study area

Two monitoring stations were established in Aarhus County, Jaungyde Brook, catchment area 10.6 km^2 with 94% cultivated area and Horndrup Brook, catchment area 5.5 km^2 with 74% cultivated area. The soil in the catchments of Jaungyde Brook and Horndrup Brook is dominated by clayey sand and sandy clay, respectively.

The discharge in Jaungyde Brook varied between 2 l/s and 2510 l/s during the study period (1993 - 1994) and in Horndrup Brook between 4 l/s and 1000 l/s. The hydrograph for both brooks showed big variations and only a smaller part of run off (37% - 48%) derived from baseflow-discharge /4/.

Methods

Phosphorus transport in streams was estimated on basis of frequent sampling (sampling every hour throughout the year) by means of an automatical sampler (ISCO 3700). Twelve samples were pooled in every bottle and two bottles represented one day. Samples were weekly collected from the sampler and brought to the laboratory. Water level was automatically registrated on a data logger. Moreover, contact to the data logger was established by a modem, making it possible to follow the fluctuations in water level from the office.

Water samples were pooled based on flow events. In weeks without sudden rises in water level, all samples were pooled. However, when sudden rises in water level exceeded 4 - 5 cm day^{-1}, samples from this event were pooled separately and samples before and after that event also were pooled separately. The shortest pooling interval was 24 hours.

Pooled samples were analysed for TP and DP. To determine DP water samples were filtered through Whatman GF/C glass-fibre filters. Transport of TP and DP in the brooks was estimated by multiplying daily discharge with concentrations, which were estimated for one day or longer time intervals depending on the pooling. Transport of PP was calculated as the difference between the transport of TP and DP.

The normal sampling strategy in the brooks with 18 or 26 point samples has been carried out at the same time as the frequent sampling. The transport based on these samples has been estimated by C-linear interpolation method /5/.

Once PP sampled during a stormflow event was analysed by phosphorus fractionation. Point samples were taking during a stormflow event in five streams representing small cultivated catchments (including Horndrup Brook and Jaungyde Brook) at 16th of March 1993. P was divided in different fractions: easily adsorbed P, organic bound P, Ferri- or Aluminium bound P and Calcium- or Magnesium bound P /6/.

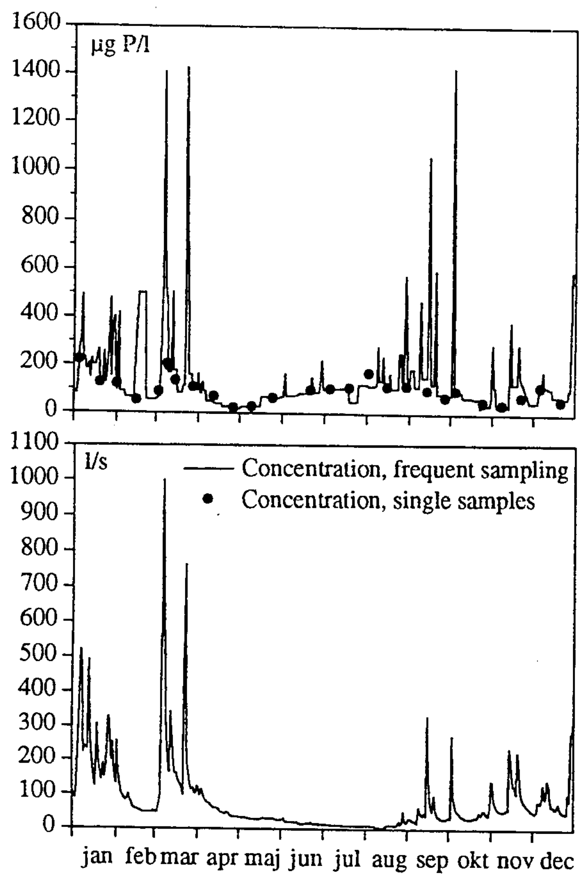

Fig. 1 Concentrations of TP as measured by frequent sampling and by 26 point samples in 1994 in Horndrup Brook. The contemporary discharge is shown.

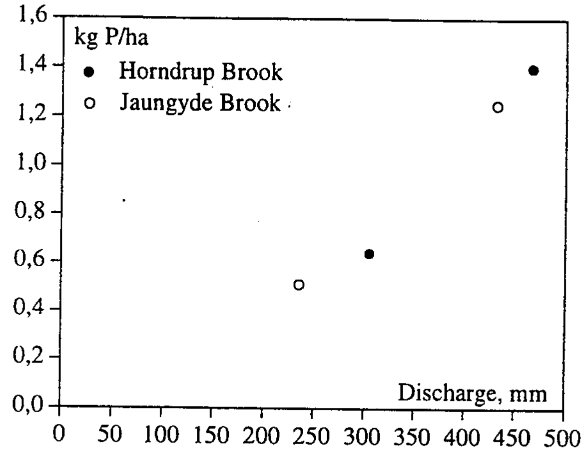

Fig. 2. The relationship between TP transport and discharge in 1993 and 1994 in Horndrup Brook and Jaungyde Brook.

Results

Transport

In Jaungyde Brook and Horndrup Brook the TP transport in 1993 and 1994 was estimated to values between 0.5 kg P ha^{-1} and 1.3 kg P ha^{-1} based on frequent sampling (Table 1). The concentration and the transport of TP was highly dependent on discharge. When discharge increased suddenly concentration increased relatively more than discharge (Fig. 1). When increase in discharge occurred after a period with low discharge, this was most evident. In Horndrup Brook there was a remarkable increase in concentration ultimo August 1994 when the first small increase in discharge during more months occurred.

The transport of TP per year was also closely related to the discharge of water (Fig. 2). Transport of TP based on point sampling was estimated to 0.33 kg P ha^{-1} - 0.72 kg P ha^{-1} (Table 1).

High TP concentrations during storm flows typically had short duration (Fig. 1). In 1994 more than 75% of TP was transported in both brooks within 10% of the time, which explains the significant difference in P transport calculated from the two sampling strategies. Transport of DP calculated by the two methods was approximately the same (Table 1). Transport of DP was 0.15 kg ha^{-1} - 0.36 kg ha^{-1}, and transport of DP maximally constituted 30 % of the yearly transport of TP.

Fractionation of P

PP from one stormflow event in five streams was divided in different fractions: easily adsorbed P, organic bound P, ferri- or aluminium bound P and calcium- or magnesium bound P. A significant part (more than 80%) of PP transported during this stormflow event was either organic bound P or associated to ferri- or aluminium compounds (Table 2 and Fig. 3).

Table 1. Transport of TP and DP in Horndrup Brook 1994 as calculated from frequent sampling and from point sources.

	Discharge	Frequent sampling		Point Samples	
		TP	DP	TP	DP
	mm	kg P ha^{-1}	kg P ha^{-1}	kg P ha^{-1}	kg P ha^{-1}
Javngyde Brook					
1993	236	0.51	0.15	0.33	0.14
1994	432	1.25	0.36	0.71	0.29
Horndrup Bro-ok					
1993	305	0.64	0.16	0.33	0.15
1994	467	1.40	0.30	0.56	0.26

Table 2. Relative contribution of different fractions of PP in a stormflow event 16th March 1994 in five streams.

	Lyngby-gaards River	Jaungyde Brook	Horndrup Brook	Knud River	Aarhus River
Easily absorbed P	5.3 %	2.4 %	3.0 %	7.7 %	8.4 %
Organic bound P	20 %	17 %	52 %	47 %	57 %
Ferri or Aluminium bound P	73 %	79 %	29 %	44 %	34 %
Calcium or Magnesium bound P	2.1 %	1.4 %	16 %	0.9 %	1.0 %

Discussion

The diffuse P load originates from different sources: a natural background load, waste water from scattered dwellings without sewage systems and a load affected by cultivation of the catchment. This load can derive from surface run off from agricultural areas or run off through drainpipes. Cultivation can also affect the erosional load from banks or bottom. Cultivation close to the banks can make them more unstable resulting in an increased erosional load. Cultivation can also affect the amount of sediment settled down slope close to the banks as well as its content of P and thus enhance the amount of material accessible for erosion. Finally, many streams have been regulated by straightening and deeping out which too can affect bank stability.

The background load is estimated using TP concentrations from streams in non - cultivated catchments. The background load estimated from these catchments is about 0.05 mg TP l^{-1} /7/, which is used in these calculations.

P - load from scattered dwellings is estimated using standard emission values per capita at 2 g P day^{-1} and 55% reduction on account of purifying and leaking from drains.

The P loadings affected by cultivation is estimated as the difference between the measured transport and the other sources. P loadings affected by cultivation is by far the dominant P source, which is emphasized by using TP transport based on frequent sampling.

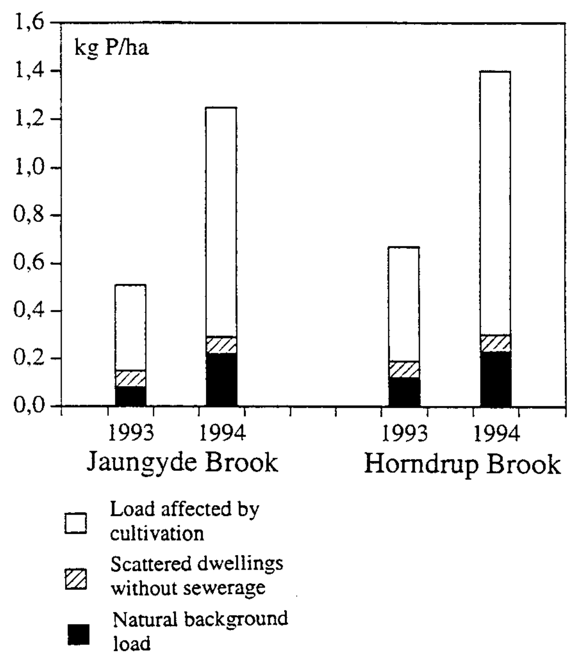

Fig. 3. Transport of TP from the two catchments, Jaungyde Brook and Horndrup Brook in 1993 and 1994. The contribution from the different phosphorus sources is shown.

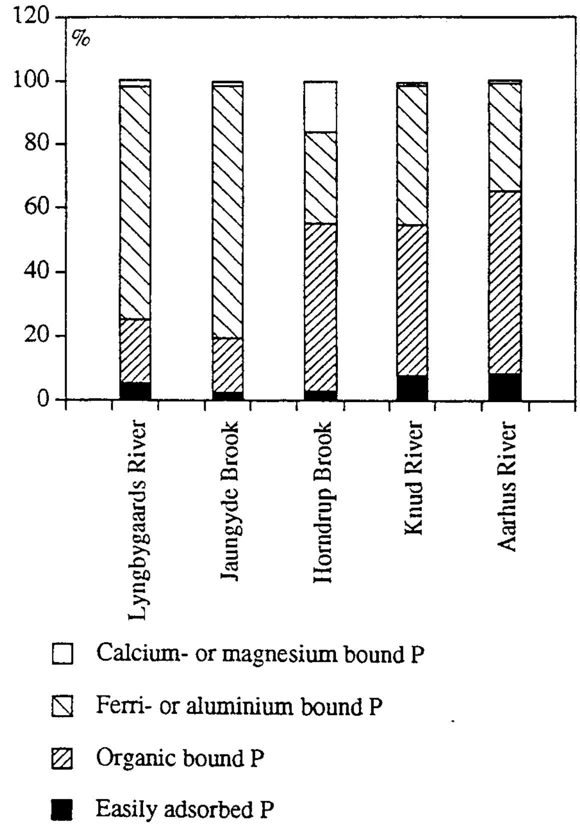

Fig. 4. Fractionating of PP from a stormflow event in five streams.

Fractionation of PP from one storm flow event in five streams showed, that more than 80% of PP was either organic bound P or associated to ferri- or aluminium compounds. Organic bound P can easily be transformed to phosphate in lakes. Under anoxic conditions, as typically found in sediments in many lakes, ferri- or aluminium bound P can too become accessible to algae. Consequently, the pulse transported during a stormflow in streams may influence the downstream lakes and coastal waters by increasing eutrophication.

Conclusion

Eutrophical level in most Danish lakes is affected by P load. Proper management of these freshwater ecosystems, including reducing external P load is based on estimates of the contribution from the various P sources.

This investigation emphasizes the necessity of having good estimates of transport in order to quantify the various sources. In smaller catchment areas the P load affected by cultivation has most often been under-estimated due to the sampling strategy /7/. Especially in small streams with big fluctuations in discharge it is essential to make sure, that P transport during stormflow events will be measured too.

Investigations of P runoff from agricultural areas handled in different ways are presently conducted /8/, as are investigations on the contribution from drains and other small catchments /9/. These investigations together with improved estimates of loadings will result in a better tool for deciding how and where to reduce P runoff.

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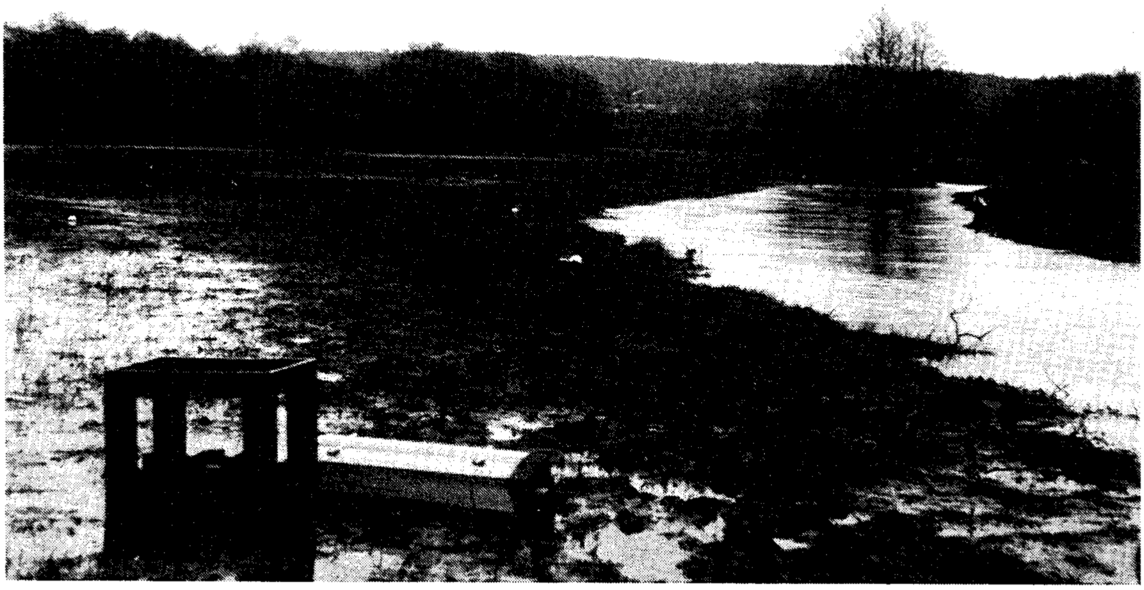

Fate of sediment and nutrients in watersheds

Evaluation of phosphorus retention in streams by means of radio active tracer

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Introduction

As part of a research project dealing with transport of phosphorus (P) in agricultural drainage brooks, particularly mathematical modelling of temporal and spatial trends, two complementary field studies have been conducted.

In connection with declining flow in an agricultural drainage brook (Uvered, /1/) in Sweden, about 50% of the total supply of P to the brook from a 8 km^2 large watershed was retained more than a week. The investigation was carried out during the later part of a washout event, induced by precipitation causing P transport from the farmland to the drainage system. The stream bed is almost saturated with respect to P and the retention is caused primarily by deposition of carrier particles during decreasing discharge. The surficial layer of temporarily deposited P-contaminated soil interacts with lower soil strata due to slower transport mechanisms such as advective induced diffusion and a mechanism denoted bioadvection /2/. Similar results of a high seasonal retention rate of P in drainage brooks on a seasonal basis has been found by Svendsen & Kronvang /3/. Their investigation showed that carrier particles were deposited primarily at sections vegetated by emergent and submersed macrophytes.

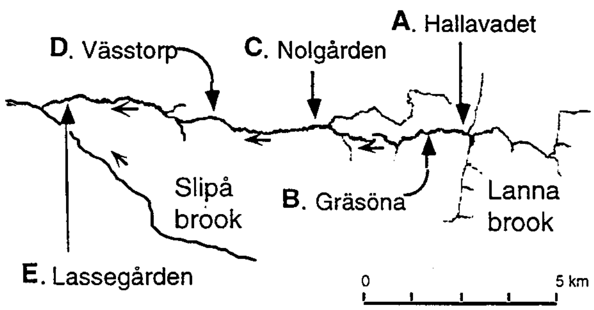

Fig. 1 Investigation reach of the Lanna brook (Skaraborg County in Sweden).

A tracer experiment makes it possible to raise the water concentration of the solute significantly above the saturation level of the bed and thereby facilitates evaluation of the vertical mixing process in the bed. Concentration profiles in the bed sediment should be measured at various occasions and locations. By choosing a tracer that has sufficiently low reactivity with respect to particulate adsorption, the impact of accumulation of tracer in the sediments will be negligible to the break through curves of the water column. This makes it possible to evaluate the dispersion mechanisms of the flow independently of the retention mechanisms.

The present study focuses on the transport and retention mechanisms of Lanna drainage brook. The primary objectives are to measure and theoretically describe (in terms of basic state variables of the system) the exchange of dissolved tracer between the water column and the bed sediment, the particulate exchange between the water column and the bed sediments and dispersion mechanisms of the brook. In this purpose, a tracer experiment with ^{51}Cr is conducted.

Field procedure and methods

Injection of the radioactive tracer, a $^{51}CrCl_3$ solution, into the quite humic (brownish) and neutral (pH 7.6) Lanna brook, was carried out on 10 April beginning at 12.45. The injection was kept at a constant rate of 41.5 MBq/min^{-1} for 4.5 hours.

Along the 11.2 km investigated reach five measurement stations were set up at the distances 70, 1 700, 5 100, 8 400, 11 200 meters from the injection point at station A (Fig. 1). The upstream section0-5 100 m is mainly located in coniferous forest whilst the downstream part flows through agricultural farmland. The total drainage area of the investigated reach is 28 km^2 and the discharge at the five stations were 146, 151, 268, 345 and 435 ls^{-1}, respectively. There were no precipitation during the experiment. An estimation of the bed sediment grain size composition gives a harmonic mean diameter, by volume, that is 2.5 μm.

Water samples were collected manually (10-20 cm below the water surface) and sediment cores were collected with a modified Willner core sampler (Ø 6,4 cm). The spacing in time of samples are presented in Fig. 2-8. Each marker represents one sample in Figs. 2-6. In Figs. 7-8 each marker represents arithmetic mean values of the core samles taken at the same cross-sectional transect. At every bed sediment sample event, a transect of three cores were collected and sectioned with depth (0-1, 1-2, 2-4 and 4-6 [cm]) to obtain a measure of the horizontal and vertical variability in tracer concentration, porosity, etceteras in the bed sediment. Additional bed sediment cores were collected during several subsequent months.

To obtain a measure of the discharge in the brook, the cross sectional area and the vertical mean velocity of the main stream were measured at each station. The velocity measurement was made according to the stream gaging procedure of the U.S. Geological Survey, by dividing the cross section in vertical strips by a number of successive verticals and determining the mean velocity in each vertical by measuring the velocity at 0.6 of the depth from the water surface.

Analytical methods

Determination of concentration of suspended solids (C_{ss}), expressed as mass dw per bulk volume, and concentrations of dissolved and particulate ^{51}Cr (C_d and C_p respectively), evaluated as radio activity per bulk volume, were performed in the following order. Water samples were first filtrated through a 0,45 μm membrane filter (Schleicher & Schuell, cellulose nitrate). To determine C_d and C_p filtration was performed immediately in the field while for C_{ss} this filtration was performed on a separate sample at the laboratory. C_{ss} was determined as the remains of the particulate matter on the filter after drying at 105° C for 24 h.

Radioactivities were measured on a gamma-counter (Packard 5000 series). The activities were corrected for background activities (i.e. from atmospheric and natural bed sediment activity), sample height (i.e. detector efficiency) and decay for ^{51}Cr (Halflife 27,706 days) for all samples. The natural bed sediment activity was obtained from cores taken before the tracer pulse reached the site. A relative error in the counting of the samples of less than 10% was achieved for most samples except for samples by which the difference between the sample and background activity was very small. A total number of 10 000 counts was generally counted for both sample and background. For statistical considerations about radio active counting see /4/ pp. 294-309.

Water content and organic mass content was determined by drying bed sediment (105° C for 24 h) and by combustion of dried bed sediment (550°C over night). The organic mass content was taken as 2.08 of the carbon mass content that was measured /5/.

Results

Break through curves of tracer in the water column are presented in Figs. 2 - 6, as well as the concentration profiles in the stream bed (Figs. 7 and 8). The retardation of the injection pulse was underestimated during the planning of the investigation. Therefore, the "tails" of the tracer pulse were not registered. Note that the breakthrough curve is interrupted earlier at station C than at D.

Concentration profiles of the stream bed at stations C, D and E has been omitted since the results merely show that no tracer has been retained in the sediment. The ^{51}Cr activity data shown in Figs. 7 and 8 is presented as arithmetic mean values (ws denotes wet substance), and does not reveal the variability in the transects. The reactivity data shows coefficients of variation up to several hundred percents within a transect.

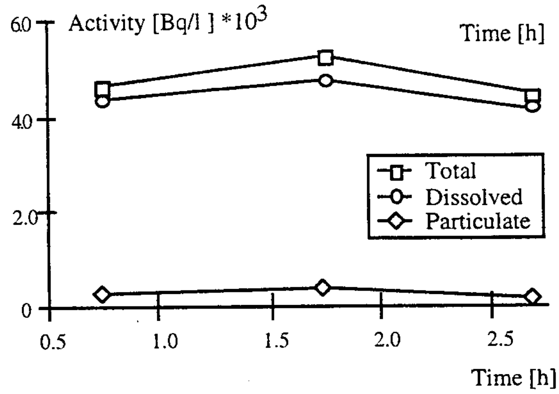

Fig. 2 Radio activity of ^{51}Cr in the water column versus time at station A (+70 [m]).

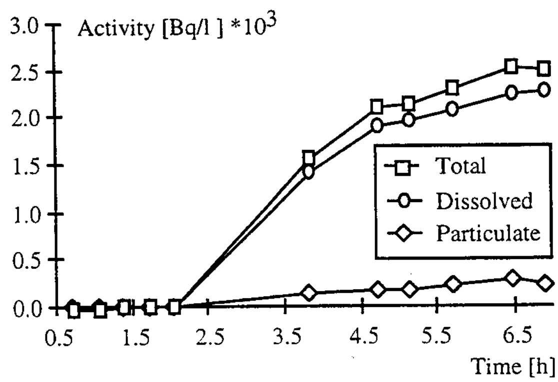

Fig. 3 Radio activity of ^{51}Cr in the water column versus time at station B (+1700 [m]).

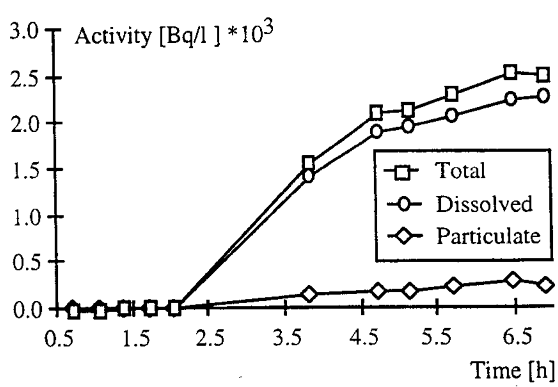

Fig. 4 Radio activity of ^{51}Cr in the water column versus time at station C (+5100 [m]).

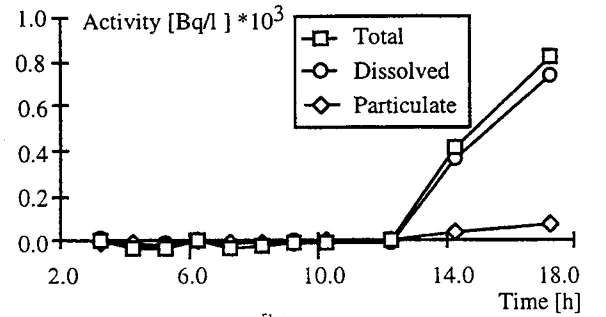

Fig. 5 Radio activity of ^{51}Cr in the water column versus time at station D (+8400 [m]).

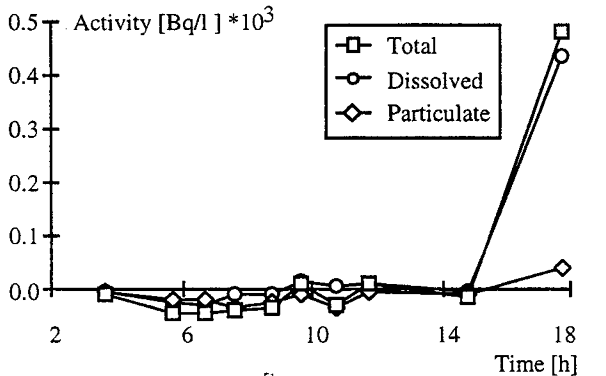

Fig. 6 Radio activity of ^{51}Cr in the water column versus time at station E (+11200[m]).

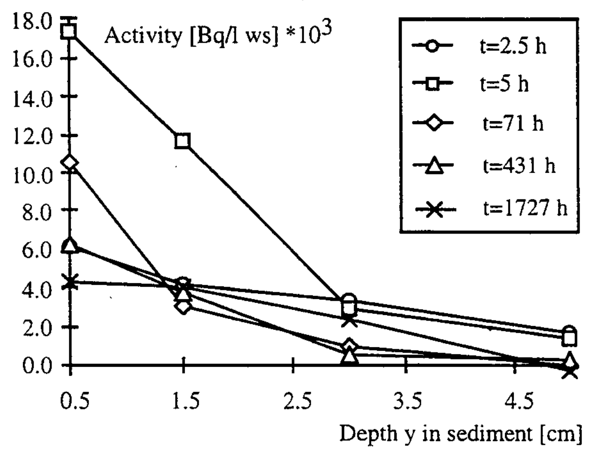

Fig. 7 Radio activity of ^{51}Cr versus depth in stream bed at station A (+70[m]).

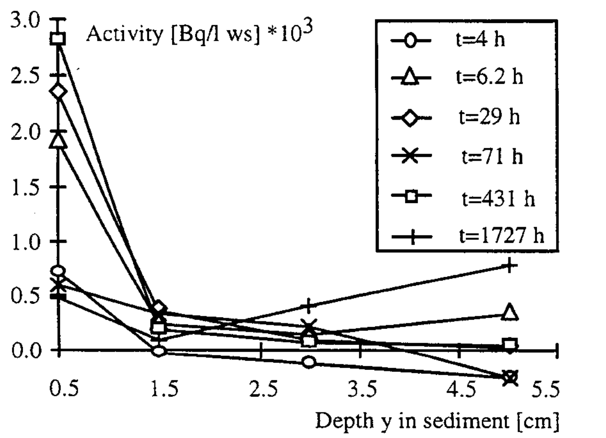

Fig. 8 Radioactivity of ^{51}Cr versus depth in stream bed at station B (+1700[m]).

As indicated in Fig. 9, the organic content decreases with distance from the injection point. Porosity for the bed sediment is shown in the same figure.

A partition coefficient that is normalised with particle concentration is defined as

$$K_D = \frac{1}{C_{ss}} \frac{C_p}{C_d} \qquad (1)$$

Evaluated K_D data did not show any clear dependency on elapsed time after injection, as indicated by Fig. 11. The rapid equilibration is consistent with the findings of a laboratory pre-study with ^{51}Cr and natural sediment as well as water from a nearby location (the Uvered brook). Equilibrium between particulate and dissolved phases was reached after only a few hours, and the major part of the exchange occurred during the first 30 minutes. The water from Uvered Brook in which C_{ss} = 23 [mg dw/l^{-1}] resulted in a K_D that was ten times larger than in Lanna brook, in which the particle concentration varied between 5-14 [mg dw/l^{-1}]. The partition coefficient for phosphorus with a C_{ss} between nearly 0 and 200 [mg dw/l^{-1}] in the Uvered brook was, however, of the same order of magnitude ($K_D \approx 0.01$ l mg^{-1} dw) /1/.

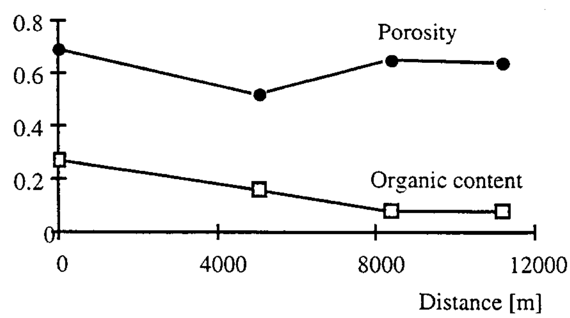

Fig. 9 Porosity and organic content of sediment versus distance from injection point.

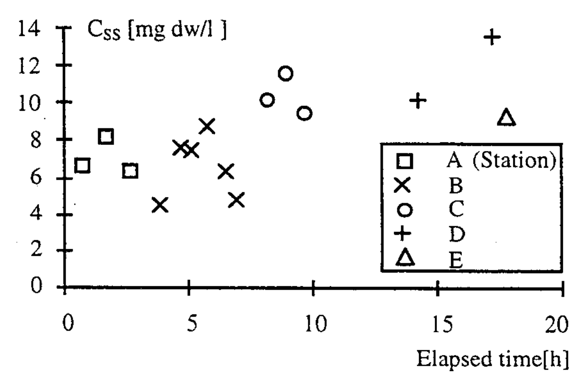

Fig. 10. C_{ss} for ^{51}Cr in the water columnversus elapsed time after injections at stations A-E.

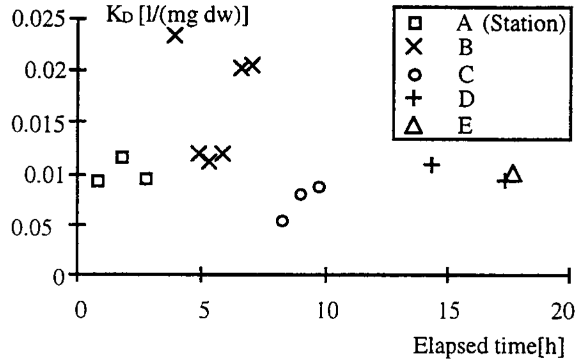

Fig. 11. K_d for ^{51}Cr in the water column versus elapsed time after injections at stations A-E.

Theoretical approach and evaluation of results

Dissolved exchange

An evaluation of the dissolved exchange between the water column and the sediment bottom was made slightly downstream of the injection point (+70 m). At this location the exchange was considered to be caused solely by ^{51}Cr in solute phase caused primarily by advective induced diffusion, AID /6,7/. AID, or pumping, is driven by pressure gradients caused by irregularities along the stream bed surface. The vertical transport also depends on the partition coefficient of the bed:

$$K_{bed} = \frac{C_p}{C_d} \qquad (2)$$

where the concentrations are defined as mass per bulk volume of bed. K_{Bed} is not necessarily the partition coefficient at chemical equilibrium.

The vertical concentration profile in the sediment bottom which is caused by dissolved exchange can be analytically expressed as /2/:

$$\frac{C_b}{C_b(y=0)} = erfc\left[\frac{y}{2\sqrt{\frac{D_e}{1+K_{Bed}}t}}\right] \qquad (3)$$

where C_b is the total concentration of the bed. The analysis is based on the assumption that the concentration is constant at the sediment/water interface and the initial concentration throughout the sediment medium is zero. erfc[ ] is the complementary error function and the effective diffusion coefficient $D_e = D_a + D_I + D_m$. D_a is the advective diffusion coefficient, D_I is the irrigation diffusion coefficient according to Berner /8/, D_m is the molecular diffusion coefficient, all in units $[m^2 s^{-1}]$.

The bed layer thickness active in the accumulation can be parameterized as /2/:

$$\lambda = \frac{2}{\sqrt{\pi}}\sqrt{\frac{D_e}{1+K_{bed}}t} \qquad (4)$$

Slightly downstream of the injection point (station A), the following field data is obtained after 5 hours (Figs. 2, 7): $C_b(y = 0.005\ [m]) = 17\,300\ [Bq\ l^{-1}]$ and $C_d(y = 0) = 5\,000\ [Bq\ l^{-1}]$, where y denotes depth in bed and y = 0 coincide with the level of the sediment/water interface. Inserting these values in eqs (2) - (4) for t = 18 000 s (5 h) gives $K_{Bed} = 3.1$, $D_e = 7.2\times10^{-8}\ [m^2 s^{-1}]$ and $C_b(y = 0) = 21\,000\ [Bq\,l^{-1}\,ws]$.

If the total uptake of tracer is expressed as $M = \lambda C_b$ the uptake rate in units $[kg\ m^{-2}\ s^{-1}]$ is obtained as /2/

$$\delta_d = \frac{dM}{dt} = C_d\sqrt{\frac{D_e(1+K_{Bed})}{\pi}}t^{-0.5} \qquad (5)$$

A representative value of δ_d is calculated with the aim to compare the solute flux into the stream bed with the particulate flux. The values above yield a uptake rate of dissolved ^{51}Cr of $\delta_d = 11.4\ [Bq\ m^{-2} s^{-1}]$.

Particulate exchange

The particulate exchange of ^{51}Cr between water column and bed sediment can be evaluated based on deposition rates of particles and the particulate concentration of ^{51}Cr. Estimation of the rate of deposition can be made by assuming a terminal settling velocity based on the measured particle size and density. The exchange of carrying particles in units $[kg\ m^{-2}s^{-1}]$ is

$$\dot{S}_D = wC_{ss} \qquad (6)$$

where w is the terminal settling velocity and C_{ss} is the measured concentration of particles in mass per volume.

The exchange in units $[kg\ m^{-2}\ s^{-1}]$ of adsorbed ^{51}Cr is (eqs (1) and (6)):

$$\delta_{p,D} = wC_{ss}\,K_D C_d = w\,C_p \qquad (7)$$

Field data from station A after 5 hours is obtained as (Figs. 2 and 10): Cd = 5 000 $[Bq\ l^{-1}]$ $K_D = 10\ [m^3\ kg^{-1}\ dw]$, $C_{ss} = 0.0065\ [kg\ dw\ m^{-3}]$ and $w = 4\times10^{-5}\ [m\ s^{-1}]$. Inserted in (7) the data yields a particulate exchange of $\delta_p = 13.0\ [Bq\ m^{-2}\ s^{-1})]$. Thus, one may conclude that the dissolved and particulate exchanges are of the same order of magnitude.

However, the particulate exchange may affect the solute exchange. As the particulate exchange is limited to a thin layer on top of the stream bed and the dissolved exchange penetrates deeper into the bed, the effect of the former can be considered to affect the boundary condition to the latter. This means that an increased surface concentration must be compensated by a greater K_{Bed} to obtain a similar concentration profile as evaluated from field data. (The tracer concentration in the very upper layer is not known since the sediment sample had a thickness of one centimetre).

The impact of resuspension is probably substantial, but is difficult to estimate because the depth of resuspension/hydraulic mixing is unknown. Small mixing depths suggest fast turnover, high surface concentrations and low stream bed accumulation with respect to particulate exchange. Larger mixing depths, on the other hand, increase the turnover times, decrease the surface concentrations and increase the accumulation capacity.

To obtain an estimate of the upward particulate exchange one may equate the mass balance including the observed inventory I with the above estimated exchange rates (δ_d = 11.4 and $\delta_{p,D}$ = 13.0 [Bq m^{-2} s^{-1}]). The inventory

$$I = \delta_d + \delta_{p,D} - \delta_{p,R})\Delta t = (\text{Total uptake})/(LW) \qquad (8)$$

where L is the distance 1 700 m between stations A and B, W=2 m is an estimated average width of the brook, $\delta_{p,R}$ is the upward particulate exchange rate due to resuspension and Δt = 5 h is the exposure time. The measured total uptake was about 730 Mbq, see below. Solving eq (8) yields the rate $\delta_{p,R}$ =12.5 [Bq m^{-2} s^{-1}].

One may thus conclude that the upward particulate exchange is almost equal to the downward particulate exchange in the investigated case.

Pulse retardation by hyporheic exchange

Based on the measured flow velocities and distances between the stations, expected arrival times for the tracer pulse could be estimated. Comparison between these expected arrival times and actual arrival times evaluated from the break through curves (Figs. 3-6) reveals that the pulse is subjected to a marked retardation. Such a retardation can be caused by a momentaneous transient storage caused by exchange with hyporheic zones such as pockets at the brook sides, slowly recirculating zones along the brook or more rapidly mixed recirculating zones behind flow obstructions like boulders or vegetation.

From the mathematical formulation of solute transport in streams, including hyporheic exchange of Bencala & Walters /9/, one may show that a momentaneous equlibration between the main stream concentrations, (C_p + C_d), and that of the hyporheic zone, C_{hyp}, in the sense that $C_p + C_d = C_{hyp}$ implies that the advection velocity, ξ, of the tracer relates to the main stream velocity, V, as

$$\frac{\xi}{V} = \frac{1}{1+K_R} \qquad (8)$$

The retardation coefficient

$$K_R = \frac{M_{hyp,q}}{M} = \frac{A_{hyp,q}}{A} \qquad (9)$$

where $M_{hyp,q}$ and M are the quick hyporheic and the main stream tracer mass respectively, and $A_{hyp,q}$ and A denotes cross sectional area in hyporheic and main stream zone, respectively. A similar reduction of the influence in the transport equation operates on the dispersion term and other source terms of the equation.

The K_R values presented in Fig. 12 are evaluated in two ways. The maximum value of ξ was obtained from the transport distance between two stations divided with the time of the initial increase of radioactivity that was registered in the stream. This value was related to the average of the maximum stream velocities of the cross sections at the two adjacent stations in order to calculate K_1. Further, the advection velocity was determined based on the arrival time of the inflection point of the break through curves. The inflection point ($\delta^2C/\delta t^2 = 0$) was considered an average of the forefront of the pulse. This velocity was related to the mean velocity of the stream in order to calculate K_2.

The evaluated variations in retardation may be supported by a comparison between Fig. 12 and the characteristics of Lanna brook.

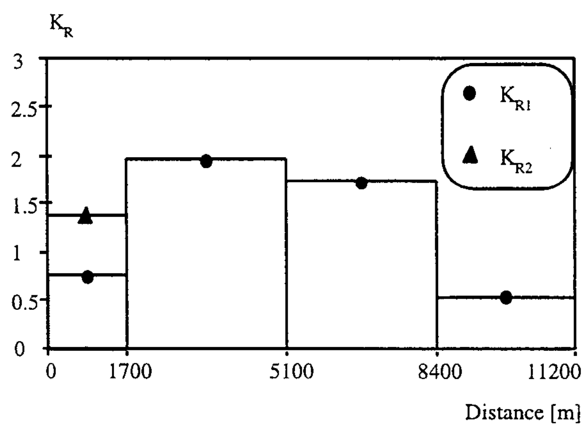

Fig. 12 System retardation factors versus distance between injection point and measurement stations.

The area between 8 400 and 11 200 meters is situated in arable land and is mainly dug in straight lines through the landscape. The upstream section on the other hand, is naturally formed and situated in more erosive ground and thus permitting establishment of storage zones to a greater extent.

Effects of retention and retardation on regional trends in transport

Based on the vertical activity profiles of the bed sediments, the total inventory of ^{51}Cr that accumulated in bed sediments on the 11.2 km reach was at the maximum (after 5 - 8 hours) 730 MBq. The total supply of chromium corresponded to an activity of 11.2 GBq. Hence, the relative retention in bed sediments along the 11.2 km reach is about 6-7%. This means that the break through curves of ^{51}Cr in the water column can be evaluated independently of the retention in bed sediments.

In addition to the momentaneous exchange from the main stream to the hyporheic zone, a slower exchange from the primary hyporheic zone to adjacent water aquifers may have a significant impact on the dispersion of tracer. In particular, it may explain the typical tails of break throughs. Such an inert transient storage or hyporheic exchange is recognised and investigated by e.g. Bencala & Walters /9/.

The hyporheic exchange is represented herein by a quick (momentaneous) exchange associated with a concentration $C_{hyp,q}$ and a slow exchange associated with a concentration $C_{hyp,s}$ (Fig. 12). Since the quick exchange is imagined as momentaneous, the concentration of tracer in the yporheic zone equals that of the main stream. The slow exchange on the other hand, occurs between storage zones containing lower concentrations than that of the main stream.

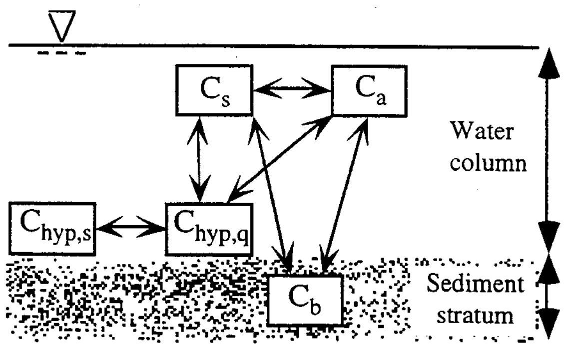

Fig. 13. Exchange mechanisms considered in the study.

For subsequent computations, flow velocities are assumed to vary linearly between measurement stations, the width of the brook is assumed uniform (2.0 m), depth calculated from Manning´s formula (taking the Manning $n^{-1} = 10$), the dispersion coefficient was taken as a typical value for a stream that is highly mixed by eddies generated by protruding boulders and vegetation D = 4 /10/ and the $A/A_{hyp,q}$ ratio is evaluated independently of the computation exercise using Fig. 12. The two parameters, α and $A_{hyp,s}$ are evaluated from calibration of calculation results with the measured break through curves and by comparison with data of Bencala and Walters /9/, where $A_{hyp,s}$ denotes cross sectional area of slow hyporheic zones and α denotes exchange rate in [s^{-1}].

The reduction of the maximum concentration during a break through can be reproduced computationally for an infinite numbers of pairs (A/A_{hyps}, α) that can be thought of as a function. However, only one pair of numbers results in "a best fit" (in some defined sense) to the data. Figs. 14 and 15 show simulated break throughs using a set of three equations similar to the formulation of Bencala & Walters /9/, though, with two hyporheic exchange zones, reflecting momentaneous equilibration and "slow" exchange. The rate coefficient for the slow exchange was found to be in the range of 0.2×10^{-4} [1 s^{-1}] < α < 0.5x10-4 [1/s] which correlates well with the findings of Bencala and Walters /9/.

There is a physical dependence between the hyporheic exchange, i.e. the determination of the A/A_{hyps} ratio, and the stream velocity. Since the variation of the stream velocity is not determined in detail, the determination of A/A_{hyps} is somewhat uncertain. Further, the distinction between a dispersion term in a Fickian sense (flux ~ D $\partial^2 c/\partial x^2$) and a hyporheic exchange is to some extent arbitrary. Taking these uncertainties into account, the ratio A/A_{hyps} was found to be somewhere in the wide range of $1 < A/A_s < 15$.

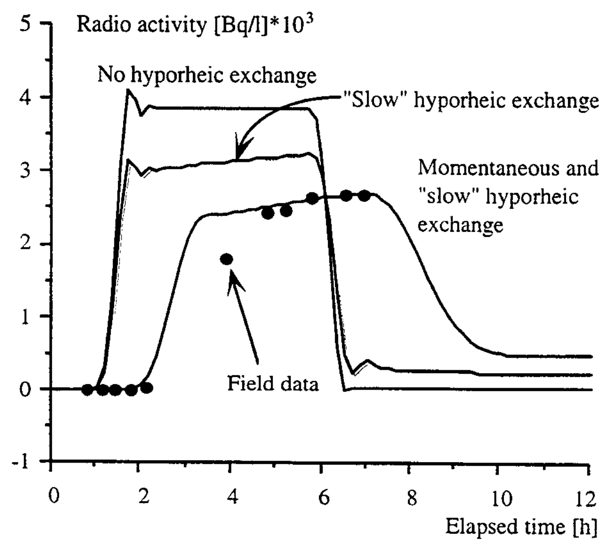

Fig. 14. Simulated break through curves compared to field data at station B.

From eq. (5), one may show that the uptake of tracer in dissolved phase (in the stream water column) to bed sediments relates to concentration and exposure time as $M \sim C_s (t)^{0.5}$. Thus, if the available tracer mass is constant, a maximum uptake is achieved if the bed is exposed to a maximum concentration during a corresponding minimum period. Therefore, if dissolved exchange dominates the tracer exchange between water and sediment, increasing hyporheic exchange and dispersion decrease the uptake of tracer in bed sediments. Accordaningly, increasing hyporheic exchange leads to decreasing uptake in bed sediment with distance from the injection point.

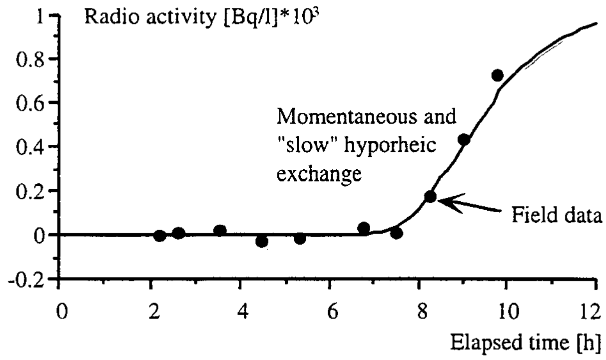

Fig. 15. Simulated break through curves compared to field data at station C.

Dilution is another factor that causes a decreasing uptake in bed sediments with distance from the injection point. However, even the combined effect of hyporheic exchange and dilution cannot explain the marked trend of decreasing uptake of tracer in bed sediments with distance from the injection point. According to the simulation results depicted in Fig. 14, the reduction of the maximum concentration at station B is approximately 50% of the level at station A and the exposure time at B is twice of that at A which, utilising $M \sim C_s\,(t)^{0.5}$, suggest a reduction of the uptake of tracer mass in the order of 70%. The maximum uptake in the bed sediments at station B is approx. 20% of the uptake at station A. At stations C, D and E, no uptake could be detected at a sufficiently high confidence level.

Probably, the marked decreasing uptake of the solute in bed sediments with distance from the injection point depends on the spatial variation of bed characteristics from permeable, organic sediments at the upstream section of the investigated reach to less permeable clay bottoms with significantly lower organic content further downstream (cf. Fig. 9). An increasing of organic content often leads to an increase of the particulate affinity for the solute, i.e. K_{Bed} increases. From eq.(5), the uptake of tracer relates to the partition coefficient of the bed sediments and AID as $M \sim [D_e\,(1 + K_{Bed})]^{0.5}$. A decrease of D_e one order of magnitude would, for example, be sufficient to explain the additional reduction of the uptake (particulate exchange omitted). Hence, future measurements should include the spatial variation of bed permeability, affecting the AID, as well as spatial variation of K_{bed}.

Conclusions

Different investigations of phosphorus transport in agricultural drainage brooks in Scandinavia show that a large portion of the P mass can be retained on a seasonal basis accumulated in bed sediments and biota. A tracer experiment using ^{51}Cr has been conducted in Lanna drainage brook in Skara County in Sweden with the aim to estimate the uptake rates in dissolved and particulate phases in bed sediments and dispersion mechanisms typical for the stream. The experiment made it possible to quantify the exchange between the solute in the main stream and hyporheic zones such as stagnant zones behind protruding boulders, in stream bends or connected ground water aquifers. A large portion of the exchange is practically instantaneous and leads to a retardation of the pulse propagation by a factor given by the ratio of the cross sectional areas for the main stream and the hyporheic zone. The retardation varies from 30% to 50% of the stream velocity in Lanna brook. In addition, an inert exchange with hyporheic zones could be quantified in terms of a rate and an equilibrium coefficient.

The hyporheic exchange alters the break through of tracer and decreases the uptake of tracer in bed sediments. If dissolved exchange dominates the tracer exchange between water and sediment, the uptake of tracer in dissolved phase (in the stream water column) to bed sediments relates to concentration and exposure time as $M \sim C_s\,(t)^{0.5}$.

The uptake of tracer can occur either in dissolved form, primarily by pressure or advective induced diffusion (AID), or in particulate form. In the present study the dissolved and particulate exchanges were expressed in terms of basic state variables of the system, such as bed partition coefficient, AID coefficient and fall velocities. The two exchanges were found to be of the same order of magnitude. The total retained tracer mass along a 11.2 km reach of the brook was approx. 6-7% of the total injected mass.

The partitioning between dissolved and particulate phases of ^{51}Cr was similar to the partitioning found for P in the nearby situated Uvered brook, $K_D = (C_p/(C_s\,C_{ss})) \approx 0.01$ l mg^{-1} dw. In this respect, the result can be considered representative to the P transport. However, the P retention is generally affected by fluctuations of the deposited sediment mass due to changes in flow conditions and the fact that the underlying, permanent bed stratum has a high saturated level with respect to P.

The marked decreasing uptake of the solute in bed sediments with distance from the injection point probably depends on the spatial variation of bed characteristics from permeable, organic sediments of the upstream section of the investigated reach to less permeable clay bottoms with significantly lower organic content further downstream. The uptake of dissolved tracer relates to the partition lcoefficient of the bed sediments and AID as M ~ $[D_e (1 + K_{Bed})]^{0.5}$.

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Fate of phosphorus in a Danish minerotrophic wetland

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Abstract

In a Danish minerotrophic wetland a phoshorus (P) mass balance showed that 7.69 and 8.87 kg P ha^{-1} yr^{-1} were exported to the aquatic environment as dissolved reactive P (DRP) and dissolved non reactive P (DNRP), respectively. In the groundwater recharge area of the wetland the sediment contained large amounts of reducible iron (Fe), probably as a result of autotrophic denitrification with pyrite as an e-doner. The pool of reducible Fe was unsaturated with P and incomming DRP was probably retained in this part of the wetland. Thus, exported P is likely to originate from within wetland mineralization processes. Tracing the groundwater flowline the groundwater DRP concentration was nearly constant (~1μM) throughout the wetland, but increased to 4~μM in surface water leaving the wetland. 90% of the P export occurred in surface water leaving the wetland. The nitrate (NO_3^-) concentration decreased from ~1800 μM in the recharge area to <25 μM 15 meters into the wetland while the sulphate (SO_4^{2-}) concentration increased from ~150 to ~800 μM across the wetland. Some of the NO_3^- probably disappears because of autotrophic and heterotrophic denitrification, the latter resulting in organic matter decomposition and P release. However, sediment DRP sorption further out in the wetland is minor because the reducible Fe pool was P saturated and because most sediment Fe appeared in pools (e.g. pyrite) inactive against P sorption. Up to 30% of wetland sediment P was associated with humic acids, but at present the exchangeability of this P pool is unknown. Large amounts of P was cycled within the wetland as compared to import and export. Thus, at peak standing crop the vegetation stored ~2000 mol P ha^{-1}. The sediment molar ratio between organic pools of carbon (C), nitrogen (N) and P was 670:44:1 when averaged from a sediment depth of 0 to 41 cm. Such a high C:P ratio indicates ecological unstability.

Introduction

Mass balance studies have often demonstrated that phosphorus (P) is retained efficiently within wetlands /1-3/ thereby improving the water quality of downstream aquatic ecosytems. However, studies of dynamic P processes within wetlands show, that the capacity of these ecosystems to retain P is limited since there is only a finite number of sorption sites in the mineral portion of the sediment. When the mineral matrix becomes P saturated, further P storage is dependent solely on biotic accumulation in plants and microorganisms /4-5/.

In wetland sediments cycling of other elements probably interfere with the dynamic P processes. Thus, if nitrate (NO_3^-) is reduced autotrophically using pyrite (FeS_2) as an energy source, ferric iron [Fe(III)] may be formed /6/. Hydrous Fe(III) adsorbs phosphate readily. In contrast, increased supply of NO_3^- should accelerate organic matter decomposition if heterotrophic denitrification replaces slower anaerobic decay processes /7/. This could result in P mobilization from sediment organic matter.

The objective of this study is to quantify the P retention efficiency of a Danish minerotrophic wetland by means of a mass balance for the year 1993 and to evaluate the result in relation to element processes within the wetland. We approched this question by measuring dissolved nutrient concentrations in groundwater throughout the wetland as well as in surface water leaving the wetland, by detailed studies of sediment chemical composition, and by quantifying nutrient storage in wetland vegetation.

Study site

The wetland (0.314 ha) is in the catchment area of River Gjern, Denmark (see excursion guide). The sediment profile (Fig. 1) is an upper peat layer followed by peaty sand. Gyttja forms an almost impermeable layer at a depth of 2 m. The plant community is dominated by grasses, sedges and other herbs. Because of groundwater discharge from the surrounding agricultural land (14.5 ha) the wetland is covered with 5-10 cm of water throughout the year. The surface water is drained through two rivulets to a nearby stream. Oxygen is not present in wetland groundwater, except very close to the hillslope.

Materials and methods

Hydrology
To ensure a well-defined wetland area sheet pilings were established (Fig. 2) to a depth of approximately 60 cm below ground. Rivulets were piped and the outflow of surface water recorded automatically every 10 min using electromagnetic flowmeters. Piezometers were installed at varying depths above the gyttja layer (Fig.1) along a presumed groundwater flowline from the hillslope towards the stream (Fig. 2). Hydraulic potentials were measured weekly in the piezometers. Water level in the stream was measured continuously.

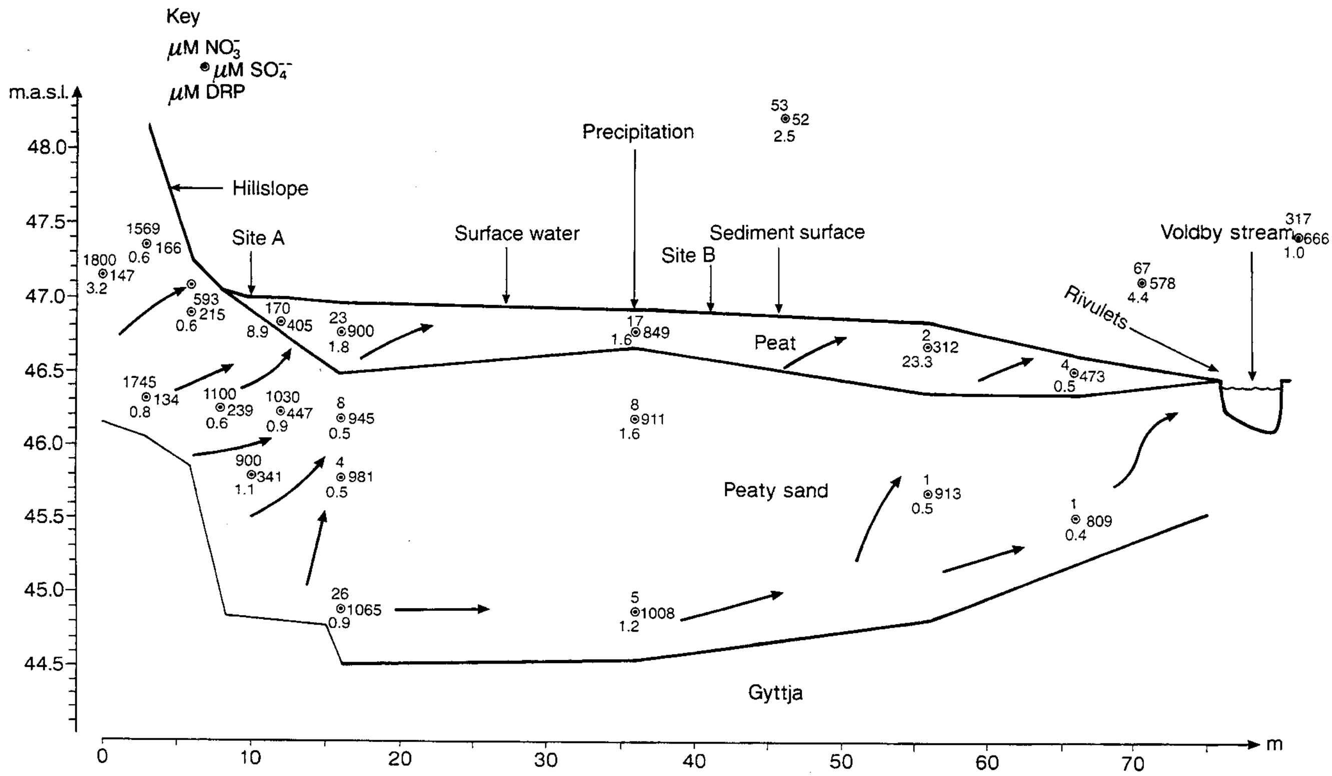

Fig. 1. Transection of the wetland showing concentrations (means of fortnightly samples in 1993) of dissolved reactive P (DRP), nitrate (NO_3^-), and sulfate (SO_4^{2-}) in groundwater at selected piezometers (⊙). Arrows (in bold) show the groundwater flow pattern. Also shown are the location of site "A" and "B" for sediment sampling.

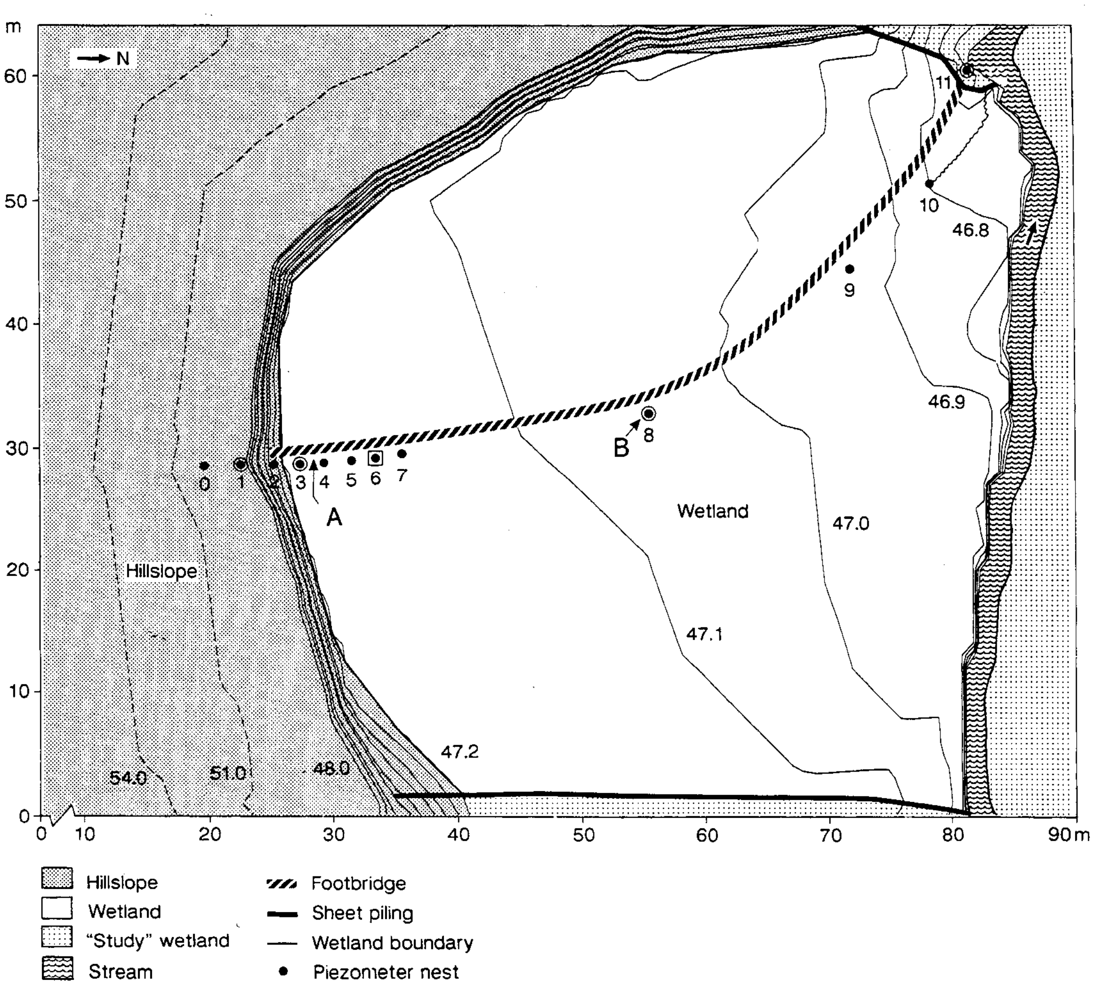

Fig. 2. Topographic map showing the study area. Numbers (0-11) indicate piezometer nests tracing the presumed groundwater flowline. Contour lines indicate level above sea (metres). "A" and "B" shows sites for sediment sampling.

Precipitation was sampled at ground level (fortnightly intervals) and actual evapotranspiration measured in the growing season using a weighable, vegetation-covered lysimeter /8/. The wetland water balance was calculated from actual evapotranspiration, precipitation, and overland discharge, while ground water discharge to the stream was calculated using Darcy's equation assuming a hydraulic conductivity of $4.1*10^{-6}$ m s^{-1} /8/. Groundwater recharge of the wetland was calculated by difference. The hydrology is described more detailed in /8/.

Water chemistry
Water was sampled from piezometers and rivulets fortnightly (n=1 per sampling site). To ensure that fresh ground water was sampled, piezometers were emptied the day before sampling. Samples were kept below 5°C and analysed within 24 hrs. Dissolved reactive P (DRP), nitrite (NO_2^-), and NO_2^- +NO_3^- were determined on filtered samples (1.2 μm) by automated standard methods. SO_4^{2-} was determined on filtered samples (0.2 μm) using ion chromatography. Dissolved non-reactive P (DNRP) in rivulet water was determined by the difference between total dissolved P (TDP) and DRP in filtered samples /9/. The methods used in the chemical analysis are described in /8-9/.

Sediment chemistry
Sediment cores (50 cm long, 5.2 cm i.d., n=4) were obtained close to the hillslope (site A) and in the middle of the wetland (site B) (Fig. 1). To preserve the redox conditions, the sediment was sampled in plexiglass tubes and in the laboratory all handling of sediment was carried out under N_2. Each core was divided into 5 sections: 0-5, 5-15, 15-25, 25-31, and 31-41 cm depths. All determinations were performed on 2 mm mesh sieved sediment.

Dry weight and organic matter content were determined by drying subsamples at 105°C followed by combustion at 550°C. Dried sediment was ground manually and organic carbon (C_{org}) determined by wet oxidation. The Kjeldahl method was used to determine organic nitrogen (N_{org}) content.

Particulate P fractions were determined by a wet extraction scheme, developed for use in wetland sediments rich in organic matter /10/. The method allows discrimination between seven different fractions of P including P associated with humic acids. The method is highly reproducible. Initially deoxygenated water was used to extract *loosely adsorbed DRP*. 0.11 M bicarbonate dithionite (BD) then extracted phosphate (denoted *Fe-P*) associated with reducible forms of iron and manganese oxides. The first two steps were performed under N_2 to avoid formation of Fe(III) and Mn(IV) oxides. Thirdly, 0.1 M NaOH extracted phosphate (denoted *Al-P*) adsorbed onto aluminum oxides and clay minerals. The extract was dark brown due to dissolved organic matter. A precipitate was formed after acidification (pH 1). This was interpreted as humic acids, and associated P (denoted *HA-P*) was determined after combustion and acid boiling of the precipitate. The forth extraction step used 0.5 M HCl to extract phosphate associated with calcium (denoted *Ca-P*). Finally, the residual sediment pellet was combusted and extracted with hot 1 M HCl. This P pool is believed to consist mainly of *refractory organic P*. DNRP may be leached in all of the first 4 steps. The sum of these P pools will be denoted *leachable organic P*.

DRP and TDP in filtered (1.2 μm) extracts were determined by standard methods. DNRP was calculated by difference. Fe and Al in filtered extracts were determined by reducing Ferrozine and atomic absorption spectroscopy, respectively /9/.

Plant chemical composition
Monthly, aboveground plant biomass was harvested at two "new" plots (each of $1m^2$) in a uniform plant community at site B. Plant material was sorted in living and dead portions and concentrations of C_{org}, total nitrogen (TN), and total P (TP) determined on dried and ground plant material.

Results

Hydrology
The groundwater flow pattern (Fig. 1) has been mapped from hydraulic potentials. Groundwater recharging the wetland from the hillslope is directed towards the sediment surface resulting in overland flow. Some flow also takes place in the peat and peaty sand horizons. Because the wetland is covered by water, precipitation does not infiltrate the sediment, but runs off as surface water. The various components of the water balance have been calculated on a monthly basis /8/. In general the size of the various contributions and losses varied little throughout the year (data not shown). Monthly the wetland received 30 to 110 mm of water as precipitation and between 580 and 1020 mm as ground water recharge. Monthly, overland discharge accounted for approximately 90% of the water loss (400 to 1100 mm) while minor losses occurred as groundwater discharge (50 to 100 mm) to the stream and as evapotranspiration (0 to 140 mm). There were no storage changes of importance.

Water chemistry
Yearly averaged concentrations of DRP, NO_3^-, and SO_4^{2-} in groundwater tracing the groundwater flowline as well as in surface water leaving the wetland are shown in Fig. 1.

There was a marked change in element concentrations along the flowline. DRP concentrations in recharging groundwater was generally around 1μM and did not exceed that level throughout the peaty sand layer. In the peat layer however, groundwater DRP concentrations were as high as 23 μM. Also the concentration in surface and rivulet water was generally high (~4 μM).

The concentration of NO_3^- was very high (~1800 μM) in groundwater recharging the wetland but 15 metres into the wetland the concentration did not exceed 25 μM. In rivulet water the concentration increased slightly to 67 μM.

The distribution of SO_4^{2-} in the groundwater was in general opposite that of NO_3^-. Thus, the concentrations were generally low in recharging groundwater (~150 μM) but increased steadily to more than 1000 μM in the zone where the NO_3^- concentration decreased. In the remaining part of the wetland the SO_4^{2-} concentration decreased slightly and was ~600 μM in rivulet water.

Table 1. P mass balance for the wetland (1993). P is dissolved reactive P (DRP) and dissolved non reactive P (DNRP).

	P-form	kg P $ha^{-1} yr^{-1}$
Input		
Ground water	DRP	3.47
Precipitation	DRP	0.49
Output		
Ground water	DRP	0.12
Rivulets	DRP	11.53
Rivulets	DNRP	8.86
Net loss	DRP + DNRP	16.55

P mass balance

The P mass balance was calculated on a monthly basis from the hydrological and chemical data. Overall, there was a net export of DRP and DNRP from the wetland to the aquatic environment equivalent to 16.55 kg P $ha^{-1} yr^{-1}$ (Table 1). Groundwater recharging at the hillslope was the dominant DRP source of the wetland, while precipitation and groundwater discharge were only minor components in the P mass balance. By far the largest amounts of DRP and DNRP were lost in overland discharge through the rivulets.

Sediment chemistry

Averaged for both sites (data not shown) sediment organic matter content decreased gradually with depth from 50.4±3.3% of dw (±SD, n=8) in the surface layer to 5.0±2.5% of dw at a depth of 31-41 cm depth. The corresponding concentrations of C_{org} decreased from 17.22±1.56 mmol C g dw^{-1} to 1.87±1.02 mmol C g dw^{-1}, while those of N_{org} decreased from 1.25±0.12 mmol N g dw^{-1} to 0.12±0.07 mmol N g dw^{-1}.

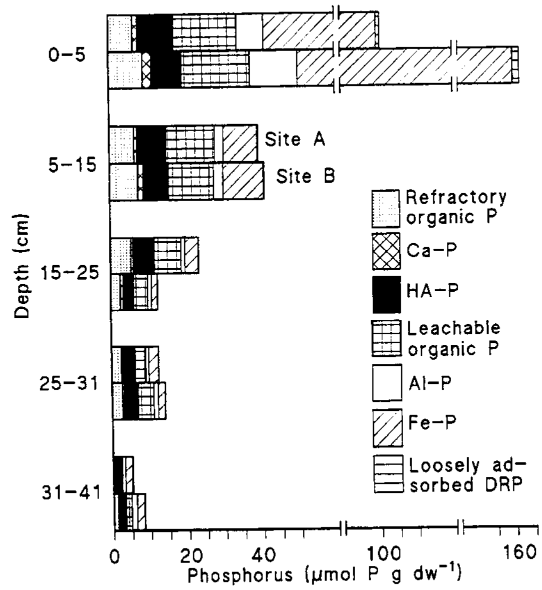

Fig. 3. P fractions in sediment from site A and B. Each value represents the mean of determinations from 4 individual sediment cores. For statistics see /9/.

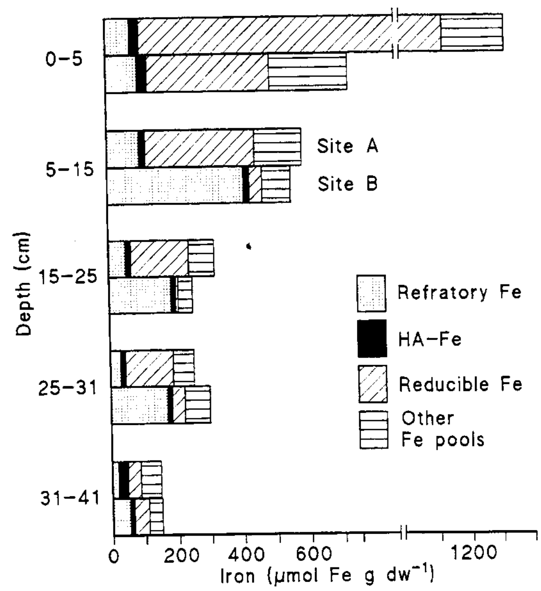

Fig. 4. Fe fractions in sediment from site A and B. Each value represents the mean of determinations from 4 individual sediment cores. Fe fractions corresponds to P fractions.

The composition of P fractions were almost the same at site A and B (Fig. 3). In the uppermost peat layer up to 80% of total P (sum of all P pools) was inorganic occurring primarily as Fe-P. However, in deeper sediment horizons the contribution of inorganic P to the total P pool decreased markedly and was only 22% at a depth of 15-25 cm. The contribution of Al-P to the inorganic P-pool was as high as 38%, while that of loosely adsorbed DRP and Ca-P was significantly less (3 to 10%). Although the concentration of organic P declined with sediment depth, the relative contribution of organic P (sum of HA-P, leachable organic P, and refractory P) to total P increased markedly below the uppermost peat layer. At a depth of 15-25 cm for example up to 78% of the total P content was organic. Leachable organic P was the most important organic P pool in the upper part of the sediment, but with depth, HA-P became increasingly important. The sum of P pools varied less the 10% from total P determined on separate samples /9-10/.

The concentration of reducible Fe, as extracted with BD, was much higher in site A than in site B sediments (Fig. 4) resulting in ratios between reducible Fe and Fe-P of 20-75 and 4-20, respectively. The concentration of refractory Fe was highest at Site B. In contrast to Fe, a considerable portion (~ 20%) of sediment Al was extracted together with HA-P (data not shown).

The molar ratio between organic pools of carbon, nitrogen and phosphorus did not vary between sites and was 511:37:1 in the uppermost peat layer. With depth, sediment organic matter became increasingly depleted in P and the ratio was 887:53:1 at a depth of 25-31 cm.

Plant chemical composition
At maximum standing crop (August) the living part of aboveground wetland vegetation stored approximately 31 kg P ha^{-1}, and the ratio between C_{org}, Total N and Total P was 482:20:1 (data not shown). At minimum standing crop (January), living aboveground vegetation only stored 14 mol P ha^{-1}. Nutrients were retained in standing dead biomass throughout the year, but this biomass was more depleted in P than living biomass (C:N:P=621:14:1).

Discussion

Based on wetland area the groundwater input of DRP is high (Table 1) when compared to other minerotrophic wetlands (0 to 0.49 kg P ha^{-1} yr^{-1}) where P mass balances are available /2, 11-13/.The high DRP input to our wetland is the result of at least two circumstances, resulting in large amounts of ground water being discharged to the wetland. Thus, the wetland area is relatively small when compared to the catchment area, and the catchment soil has a high hydraulic conductivity. Further, large amunts of P-fertilizer and manure are applicated to the fields in the catchment area. Input parameters in our P mass balance did not include DNRP (Table 1). Probably this only introduces a minor error in the overall P balance since DRP previously has been shown to be the primary P pool in ground water discharging into wetlands /2/.

There was a substantial net loss of both DRP and DNRP from our wetland to the aquatic environment (Table 1). Such losses have also been reported from other natural wetlands /e.g. 3, 13/, but at a much smaller scale (0.03 to 0.30 kg P ha^{-1} yr^{-1}). However, one can normalize the P output by expressing it on the basis of the catchment area. Such calculations reveal a loss of 0.36 kg P ha^{-1} yr^{-1} which is characteristic for agricultural watersheds in Denmark /14/.

Different pathways for wetland P release have been discussed /3, 11, 13/. It may happen as a result of mineralization processes within the wetland or because the P input occurs in periods with a limited biotic P demand. Also, in periods with high hydraulic through-flow the time available for chemical and biotic P fixation will be limited. In wetlands receiving high inputs of DRP, for example through wastewater, the efficiency of P removal decreases over time because the P adsorption capacity of the mineral sediment matrix becomes saturated /4-5/. In contrast, the absence of P retention in our wetland seems to be affected by the NO_3^- loading and NO_3^- removal processes. This is discussed below.

Tracing the groundwater flowline, the congruent decrease and increase in concentrations (Fig. 1) of NO_3^- and SO_4^{2-}, respectively, indicate that NO_3^- partially disappears from the groundwater due to autotrophic denitrification. A likely reductant is FeS_2 resulting in SO_4^{2-} and Fe(III) formation /6/. Indeed, the concentration of reducible Fe was very high in site A sediment (Fig. 4), where NO_3^- disappeared. The pool of reducible Fe was highly unsaturated with DRP suggesting that DRP in recharging ground water is retained efficiently in the sediment next to the hillslope. A molar ratio between reducible Fe and Fe-P greater than 8 is required for continued DRP adsorption /15/. In contrast, the low ratios at site B indicate that DRP adsorption on reducible Fe compounds is restricted in the outer part of the wetland. These results are consistent with the interpretation that the ability of sediments to adsorb DRP can be improved when NO_3^- is reduced trough FeS_2 oxidation /16/.

Decreasing SO_4^{2-} concentrations further out in the wetland (Fig. 1) indicate sulfate reduction. This process may have adverse effects on sediment DRP adsorption capacity due to increasing alkalinity (OH^- production) and formation of Fe-sulphide restricting the availability of reducible Fe compounds /17/. This is supported by the fact, that a major part of total sediment Fe at site B was extracted in the refractory organic pool (Fig. 4). Fe-sulphide dominates this Fe fraction (Jensen, pers. comm.).

Some of the NO_3^- disappearance (Fig. 1) in the wetland is probably accounted for by heterotrophic denitrification /18/, thereby accelerating organic matter breakdown and P release in the sediment close to the hillslope. Thus, the combined export of inorganic carbon from the sediments to the atmosphere and to the aquatic environment was more than two times larger at site A compared to site B /18/, where no NO_3^- loading occurred. The high organic C:P ratio in the sediments suggests microbial P immobilization during organic matter decomposition. Considering the total sediment P pool, however, the microbial contribution was minor (~9%) /9/, indicating that the microbial P flux is fast or alternatively, that small amounts of P are sufficient to sustain the microbial P demand. Therefore it can be assumed that both DRP and DNRP will be released during organic matter decomposition although initially, the release rate will be smaller than expected from the organic C:P ratio because of P immobilization.

Under the assumption that above- and below ground plant production is equal the P demand of the wetland vegetation is approximately 62 kg P ha^{-1} yr^{-1}, when estimated from the maximum standing aboveground crop. Even though plants retranslocate nutrients (e.g. N and P) for use in the next growing season, our results show, that the overall wetland P cycle is dominated by internal circulation between sediments and plants (max 62 kg P ha^{-1} yr^{-1}) while the exchange with the aquatic environment is somewhat smaller (16.55 kg P ha^{-1} yr^{-1}). Accounting for variation in sediment bulk density /9/ the total sediment (at a depth of 0-41 cm) P pool must turnover every 21-43 years to support plant production. This compares favourably with turnover times reported from other freshwater wetlands /19/.

Clay minerals and oxides of Al and reducible Fe are all important with respect to DRP exchange between porewaters and the sediment matrix. However, the humic acid fraction contained high concentrations of Al and also some Fe, suggesting that DRP can also be associated with these refractory organic compounds. The most probable mechanism for this association is DRP adsorption on the humic acids through bridges of Al and Fe oxides /9/, but the exchangeability of DRP on these complexes is unknown at present.

Overall, the results of this study suggest that P loss from a minetrophic wetland occurs because of NO_3^- loading and NO_3^- removal processes. Present knowledge about the relationship between P and N cycling in wetlands should be improved through detailed laboratory studies evaluating the temporal changes in pool sizes of the various components participating in the processes described above. Also, it is important to focus on the ecological stability of freshwater wetlands used for NO_3^- removal. Thus, these systems appear unstable with respect to P cycling and the effect on for example the plant community structure is unknown.

Acknowledgement

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Retention of particulate nutrients within aquatic macrophyte patches and on the floodplain

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Abstract

Daily accumulation of particulate nitrogen (PN) and particulate phosphorus (PP) in aquatic macrophytes in a small lowland stream was measured over a three-week period. Retention was greater in the submerged than in the emergent vegetation, and was strongly influenced by variations in the suspended sediment concentration associated with weed cutting and a storm event. Long-term storage of P on the floodplain was studied near the mouth of the same river. Deposition of particulate P took place, both during inundation via a drainage ditch and during flow across the floodplain during overbank flooding. During the highest flood which was monitored the entrapment astride 100 metres of river amounted to 5% of the particulate P exported from the watershed.

Introduction

During the last two decades increasing interest has been paid to the fluxes of nutrients through the freshwater drainage system. This is not only because of toxic and eutrophic effects within the streams and lakes themselves, but also because of the environmental impact in coastal marine areas /1, 2, 3/. Phosphorus (P) has in recent years received special attention as far too little is known about diffuse sources and pathways of what is thought in many aquatic environments to be a limiting nutrient. In particular, the ability of P to be bound to particles means that the flux can be highly discontinuous, with periods of suspension and transport alternating with periods of sedimentation and rest /4/. Such storage takes place at very varied time scales, ranging from days to centuries, and there is a similar, though smaller, range in space.

In lowland streams nutrient-rich fine sediments are trapped by vegetation within the channel during the summer, to be released again during high discharge events, mainly in the autumn and early winter /4/. The in-channel storage is thus mainly temporary, and can also be very local, but as a result the flux is concentrated into short periods of high sediment and water discharge, sometimes, though not always, associated with flooding and overbank storage. Such storage on the floodplain is both extensive and long-term. This paper examines both shorter, in-channel storage and overbank storage of sediment, PN and PP.

Study sites

Both study sites are on the 15 km long main branch of the river Gjern å, in central Jutland, Denmark. The catchment (114 km^2) has a gently hilly topography ranging in height from 135 m to 21 m above sea level. The soils are fluvio-glacial and glacial, ranging from clay-loams to sands.

In-channel storage was studied in a reach at Domdalsvej about 5 km from the headwaters, and some 2.5 km downstream from the shallow lake Søbygård. The lake sediments are a major long-term source of P in the Gjern å system /4/. The reach has stands of submerged and emergent macrophytes, dominated by *Batrachium peltatum* (Schrank) Presl. and *Sparganium emersum* Rehman.

Overbank flood deposition was studied at Smingevad near the confluence of the Gjern å with the large river Gudenå. The river here meanders through a floodplain which varies in width up to 200 to 300 m. Some degree of flooding takes place up to several times in most winters.

In-channel retention

Methods

The purpose of this part of the study was to monitor the day-to-day accumulation of N, P and O in an emergent macrophyte strip along the stream banks and a mid-stream submerged macrophyte patch during a 19-day period in August/September 1993. Gross sedimentation was measured by means of rows of sediment traps in the stream bed. Each trap consisted of a permanently emplaced outer cylinder and a removable inner tube (diameter 7 cm, height of opening above bed 4 cm). The submergent macrophyte patch was dominated by water crowfoot (Batrachium peltatum) and was large enough for 3 rows with respectively 4, 5 and 6 traps spaced at 10 cm (Fig. 1). Some 40 m downstream 2 rows of 12 traps were carefully buried between the emergent bur reed (Sparganium emersum) plants along the side. For adequate sample size the trapped sediment was pooled into one daily sample from each of the two environments. Stage was monitored continuously further downstream, where hourly water samples were taken with automatic sampling equipment. These samples were also pooled daily, and analysed for total sediment content, total N, and the four dissolved/particulate organic/inorganic fractions, i.e. DIP, DOP, PIP and POP. In the following "suspended sediment" will refer to the <250 μm fraction and for simplicity the two environments will be referred to as "patch" and "strip".

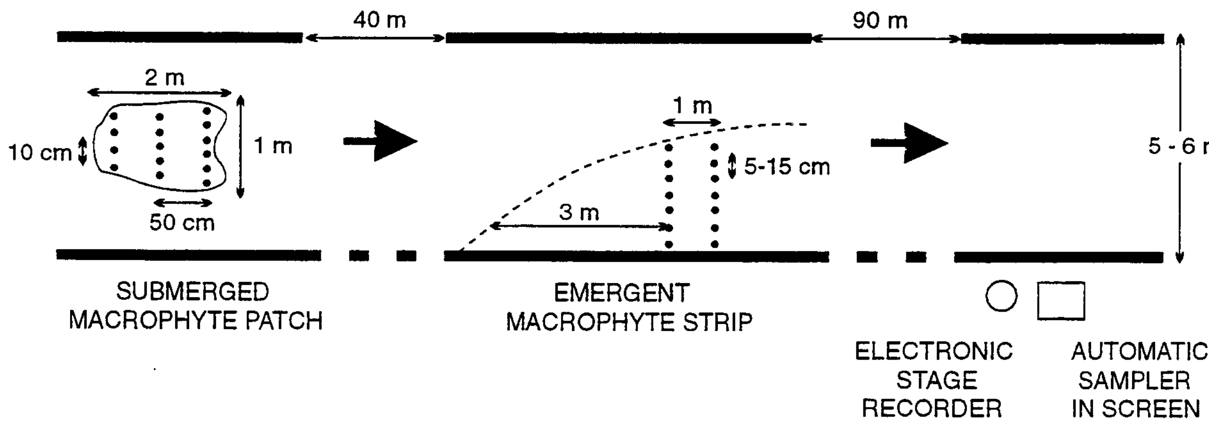

Figure 1. Schematic sketch of the sampling setup. The measurements and proportions are not to scale.

Results and discussion

During the first twelve day of the 19-day period daily discharge and daily accumulation rates were very steady (Fig. 2A and B). The daily sediment accumulation rates in the two environments was on average nearly the same being 1.2 g N m^{-2} and 0.7 g P m^{-2} in the patch and 1.0 g N m^{-2} and 0.7 g P m^{-2} in the strip.

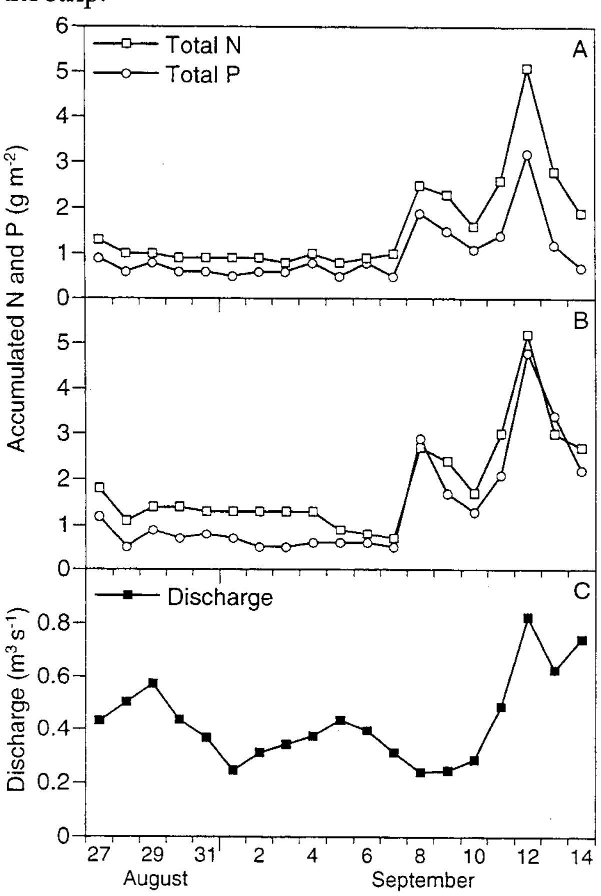

Fig. 2. Daily accumulation in the submerged macrophyte patch (A), the emergent macrophyte strip (B) and daily discharge (C).

On September 7th about 2/3 of all the aquatic vegetation was cut upstream and downstream of the sampling reach as part of the normal annual river maintainance, but the sampling reach was not disturbed. This more than halved the stage without any change in discharge (Fig. 2C). The weed cutting reduced the shelter, but, more important, the cutting and wading upstream of the sampling reach resuspended accumulated sediments. Consequently, the PN and PP accumulation during the three days period (8th to 10th of September) increased in both environments (Fig. 2A and B). The average accumulation during the three days period following weed cutting increased with a factor 1.9 for PN and 2.9 for PP in the patch and 2.1 for both PN and PP in the strip as compared to the pre-disturbance period. It is remarkable that no major change in the suspended sediment concentration could be observed at the monitoring station downstream, indicating that the resuspension was an entirely local phenomenon quickly damped in the streamwise direction, and therefore of no great significance in the downstream transport.

A two-day period of heavy rainfall started on September 11th, resulting in a markedly increased discharge (Fig. 2C). Suspended sediment concentrations at the monitoring station did not respond until two days later. From one day to the next increases were: suspended sediment x 3.7, N x 3.7 and P x 3.1, and the increase continued on the following day when the discharge peak had long passed. It is tempting to ascribe this sediment pulse to resuspension because of reduced shelter upstream due to weed cutting, but the delay was far too long for this small drainage basin. (The same argument applies to quickflow). The increase was therefore probably occasioned by bank collapse upstream during the receeding limb of the hydrograph. Although there was no increase in the suspended sediment and nutrient concentrations for the first two days of the storm discharge, the retention increased with the increased sediment flux caused by the rising discharge. The increase was on average for the four days period (11th to 14th of September) as compared to the average for the 12 days pre-cutting period greater for PP in the patch (PN x 2.9; PP x 4.4), than for PP in the strip (PN x 3.1; PP x 2.3).

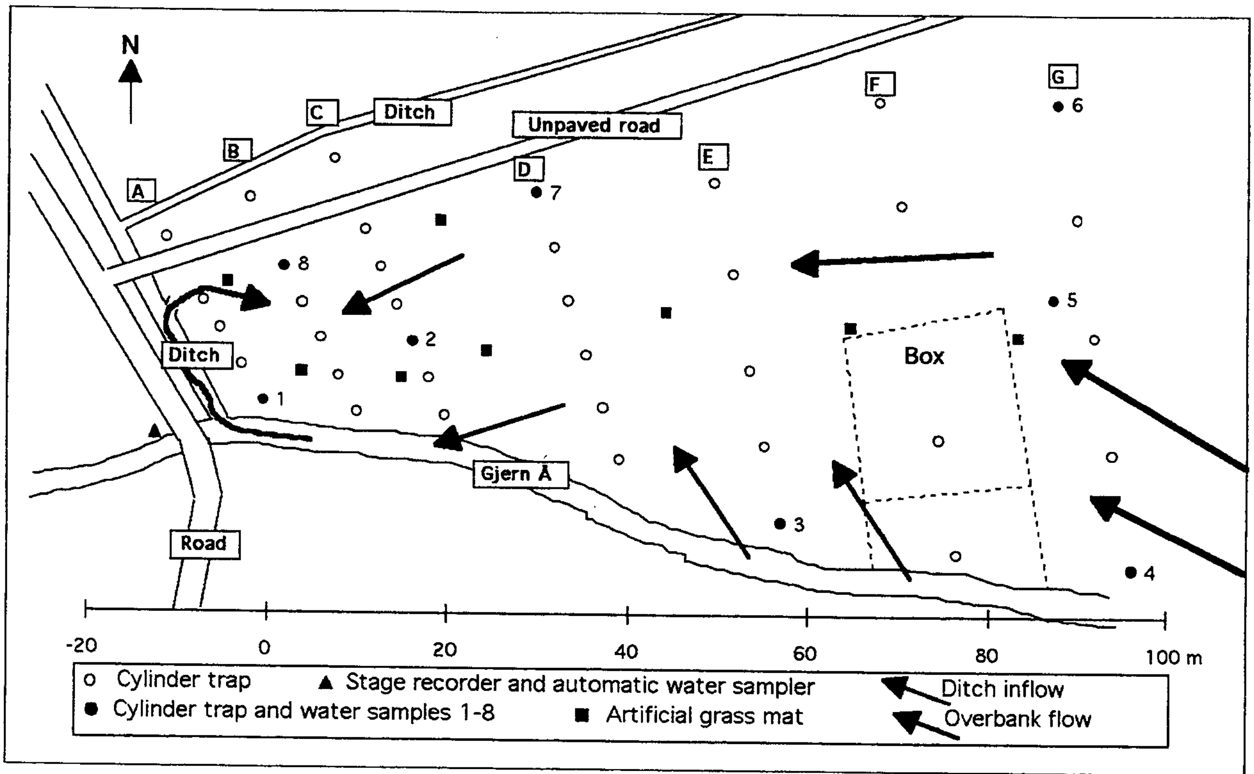

Fig. 3. The floodplain at River Gjern, Smingevad with a sketch of the sampling setup and the ditch and overbank flow directions.

There are two possible explanations for the somewhat greater PP retention in the patch than in the strip after weed cutting and the even greater difference during the storm flow. Firstly, the shelter is better in the patch: flow velocities measured immediately above the traps were lower in the patch than in the strip. Secondly, the greater streamwise extent of the strip probably meant that much of the sediment had already been retained in the upstream end of the strip and did not reach the samplers. Further studies must therefore look more closely at the spatial distribution of retention within the individual environments and/or monitor the lateral and longitudinal fluxes across the open/vegetated boundaries.

Retention on the floodplain

Methods

39 cylindrical traps were emplaced along 7 transects in the 5000 m^2 area of frequent flooding at Smingevad at the downstream end of the Gjern å system (Fig. 3). These traps measure gross sedimentation as they are too deep to permit resuspension. Net retention was measured with 9 artificial grass mats (tuft height 18 mm) /5 and 6/. Stage, and the concentration of suspended sediment, organic matter and total P were all monitored continously at a downstream river station (Fig. 3). Water samples were also taken during floods at selected trap sites and overbank flow velocities were measured during the highest floods.

Results

Flooding in this area takes place in two ways. "Ditch inflow" with flooding of the lowest areas taking place during moderately high stages, water entering and leaving via the main drainage ditch (Fig. 3). At very high stages there is overbank flooding with continuous flow over the floodplain. During the period Nov. 92 - Feb. 93 there were 2 irrigation floods and 3 overbank floods.

The pattern of deposition and the total amounts reained were calculated by two methods. In the "box method" bisectors were drawn between the traps and each trap was assumed to be representative of the whole box area (Fig. 3), although each trap was not necessarily at the centre of its box. In the "regression" method exponential regressions were fitted to the data from each line of traps (Fig. 4), on the assumption of a diffusion analogy /7/.

The two methods of calculation disagree on the total dry matter (Table 1). This is because the box method ascribes for too large an area to the samplers at the river bank which trap large amounts of coarse sediment. The exponential method also has drawbacks in that the high value obtained close to the stream dominated each fit, and, possibly as a result, the regressions on adjacent transects differ considerably. The two methods agree tolerably well on both organic matter and P, and the box method has been used in the following.

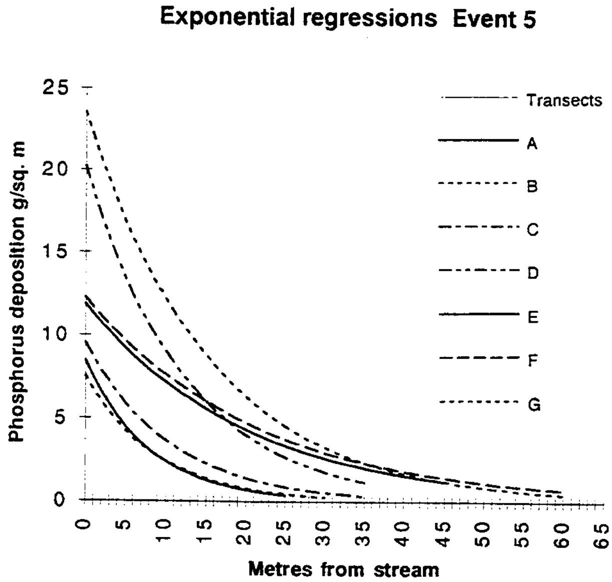

Fig. 4. Examples of transect regressions during one of the sampled flooding periods.

Table 1. Total retention during the 5 floods calculated by two methods.

	Total deposition in g m^{-2}		
	Dry matter	Organic matter	Particulate P
Regression	1,825	498	9.0
Box model	4,572	662	11.8

A comparison of the cylindrical traps and the grass mats showed that resuspension only occurs close to the river bank, so that gross and net retention are almost identical (Fig. 5).

It had been assumed that ditch inflow events only involved the filling and emptying of an otherwise stagnant basin. Calculations of the volumes and concentrations involved for each change of stage show that this picture is incomplete. Even allowing for 100% retention, the volumes involved can only explain 60% of the deposited PP and only 27% of the deposited dry matter. This implies that there is an exchange of water even during ditch inflow events, i.e. that flow volume exceeds basin volumen.

However, this is quantitatively not a major problem as a comparison with the totals from the 5 events showed that the total contribution of the two irrigation events was very small (Table 2 and 3): 2.1 % of the total dry matter and 5.5 % of the PP. Using stage and concentration data this can be extrapolated (very approximately) to the floods of the 10-year period 1982-92 to yield only slightly larger percentages, respectively 5.7 and 10.1%.

Flow measurements during the highest flood stage showed that overbank discharge took about one hour to flow over the 100 m long floodplain. In the process it lost 40% of its PP content, representing 10% of the fluvial P flux at the time. The entrapment efficiency of the floodplain during each of the 5 events is shown in Table 2 and 3. The weighted average for P for the 5 events together is 5.0 % of the total P export from the River Gjern watershed during the five events. Since the P flux mainly takes place during flood events this underlines the importance of even small inundated flood plains in restricting the P export to downstream water bodies.

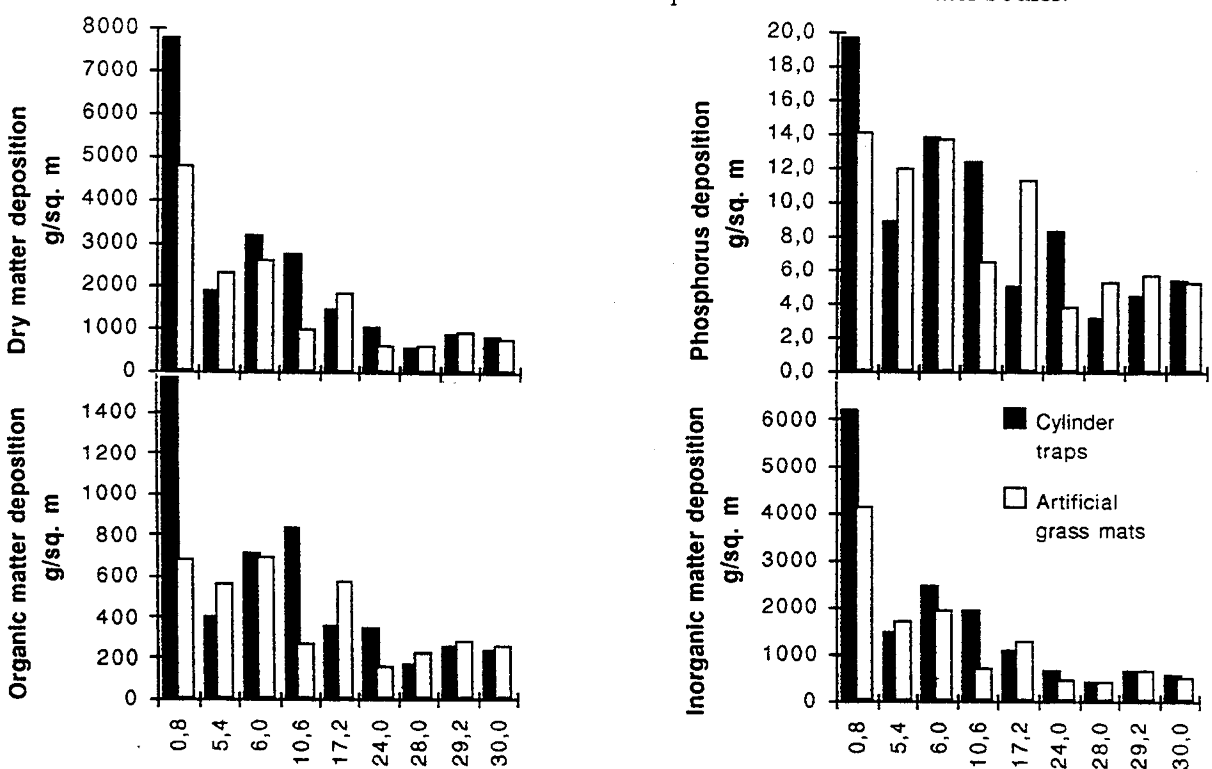

Fig. 5. Comparison of the deposition of dry matter, particulate P, organic matter and inorganic matter measured by means of cylinder traps and artificial grass mats with distance from the River Gjern.

Table 2. Deposition of particulate matter on an approx. 5,000 m^2 flooded area (100 m river reach) in the lowermost part of the River Gjern watershed as compared to the export of suspended sediment from the watershed during five irrigation and flood events in winter of 1992/93.

Period	Flood type	Sediment depostion (kg)	Export suspended sediment (kg)	Deposition of total export (%)
10 Nov. - 23 Nov.	Irrigation	211	18,122	1.2
24 Nov. - 2 Dec.	Overbank	1,279	22,771	5.6
3. Dec - 8 Dec.	Irrigation	342	9,455	3.6
11 Jan. - 20 Jan.	Overbank	6,067	53,512	11.3
21 Jan. - 9 Feb.	Overbank	15,115	63,336	23.9

Table 3. Deposition of particulate P on an approx. 5,000 m^2 flooded area (100 m river reach) in the lowermost part of the River Gjern watershed as compared to the export of total P from the watershed during five irrigation and flood events in winter of 1992/93

Period	Flood type	Phosphorus depostion (kg)	Export particulate phosphorus (kg)	Deposition of total export (%)
10 Nov. - 23 Nov.	Irrigation	0.84	243	0.33
24 Nov. - 2 Dec.	Overbank	5.92	217	2.7
3. Dec - 8 Dec.	Irrigation	0.96	101	0.95
11 Jan. - 20 Jan.	Overbank	18.9	380	5.0
21 Jan. - 9 Feb.	Overbank	32.7	606	5.4

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Fate of phosphate, nitrate and other elements during short-term flooding of a riparian meadow

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Abstract

Two irrigation experiments and one flooding experiment were performed in a riparian meadow. The experimental setup consisted of four plots, each 2 m x 2 m, encircled by sheet pilings. In the two irrigation experiments each plot received 1,000 and 1,500 l stream water, respectively. The water was sprinkled uniformly over the plots during a four day period. In the flooding experiment the plots received 1,000 l stream water for 30 min. The water was enriched with nitrate to a final concentration of 10 mg NO_3^--N l^{-1}.

Water samples were collected from surface water and from piezometer nests with slotted well points at six depths: 1-2, 2-3, 5-10, 10-15, 20-25, and 30-35 cm.

The three experiments all showed the same trends. Nitrate was reduced (denitrified) in the upper 0-2 cm of the soil profile. Phosphate concentrations declined with depth. Phosphate concentrations in inlet water were 0.15 - 0.17 mg P l^{-1}, but decreased with a marked gradient to 0.04 - 0.06 mg P l^{-1} at depth 2-3 cm in all three experiments. At depth 30-35 cm the phosphate concentrations further decreased to 0.01 - 0.03 mg P l^{-1}. Sulphate and iron concentrations increased with increasing depth probably due to denitrification with pyrite as e-donor.

Introduction

Phosphate removal in wetlands seems to be somewhat contradicting. Theoretically, retention of phosphate by different sorption mechanisms is not favoured in many wetland soils due to redox conditions and high content of organic matter with low sorption capacity /1,2/. Yet some studies show that phosphate is retained in wetlands /2,3/. Retention of phosphate by sorption mechanisms in organic wetland soils seems to be dependent on the mineral content, especially calcium and the two metals aluminium and iron /2,4,5/.

Phosphate retention in wetlands is not as pronounced as that of nitrogen, the percentage of phosphate retention ranging in most cases between 0 and 50 per cent. It is also subject to large seasonal fluctuations /6,7,8,9,10,11/. The more long-term research projects in wetlands influenced by phosphate rich water have shown, that after only a few years retention capabilities can fall to almost zero or - in some cases vanish completely /1,2,12/. A final point is that any given wetland's ability to retain phosphate will depend on its past phosphorus history, as pointed out by Peverly /8/ and Nichols /1/.

The purpose of this study was to elucidate the fate of nitrate and phosphate after flooding or waterlogging of wet meadows in summer, when the groundwater table is initially below ground. This situation permits an instantaneous downward flow of water and nutrients before saturation is reached.

Study site

A riparian meadow (3.5 ha) in the lower reaches of river Gjern Å (UTM 32 E542.550 m N6 230.425 m Jutland, Denmark) was chosen as study site. The meadow is flooded during winter, often for several months, and occasionally during summer following heavy precipitation. The meadow soil consists of hemic peat in the upper 0-40 cm. The peat also contains fine to medium grained sand and reddish brown nodules of iron. At a depth of 40 cm is a 10 cm band of gyttja or gyttja-like peat. Underneath the gyttja there are alternating layers of less decomposed peat and peaty sand to a depth of 150-200 cm, where sandy layers become dominant. Dominant plant species are *Poa praténsis ssp. praténsis*, *Glyceria flúitans*, *G. máxima*, *Ranúnculus repens*, and *Alopecúrus geniculatus*.

Field design

The experimental set up consisted of four 2 m x 2 m plots encircled by sheet piling (galvanized iron plate), which was forced 40-50 cm into the soil (Figure 1). Each plot was supplied with a sprinkler and a flow regulator connected to a 1,000 l water tank into which water was pumped from the river Gjern.

Piezometers were placed at six depths: 1-2, 2-3, 5-10, 10-15, 20-25, and 30-35 cm. All experiments were run in quadruplicate, each plot being subjected to the same treatment.

During late summer, early autumn 1994, three experiments were conducted. In week 30 (25.07.94 to 31.07.94) and 32 (08.08.94 to 14.08.94) the plots were sprinkled with 1,000 and 1,500 l of river water, respectively, during a four day period. In week 34 (22.08.94 to 28.08.94) each plot was flooded with 1,000 l nitrate enriched river water (10 mg NO_3^--N l^{-1}) for thirty minutes.

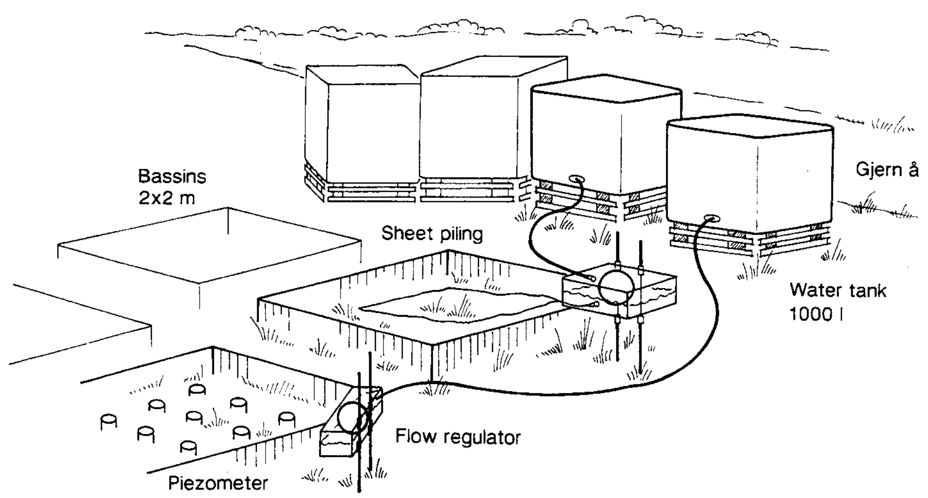

Fig. 1. Experimental setup.

Table 1. Estimated nitrate-N concentrations during week 30 and 32, with 95% confidence limits. (n = 177).

Date	NO_3-N mg l^{-1}	95% Confidence limit Lower mg l^{-1}	Upper mg l^{-1}	Compares equal to
25 Jul	0.361	0.054	1.421	all dates
26 Jul	0.861	0.458	1.504	25, 27, 8, 9
27 Jul	0.313	0.156	0.578	all dates
28 Jul	0.161	0.063	0.354	25, 27, 29, 9, 10, 11, 12
29 Jul	0.158	0.050	0.401	25, 27, 28, 9, 10, 11, 12
08 Aug	0.711	0.436	1.109	25, 26, 27, 9
09 Aug	0.408	0.242	0.656	all dates
10 Aug	0.241	0.131	0.416	25, 27, 28, 29, 9, 11, 12
11 Aug	0.253	0.140	0.430	25, 27, 28, 29, 9, 10, 12
12 Aug	0.189	0.095	0.344	25, 27, 28, 29, 9, 10, 11

Methods

Water samples were collected from the water tanks prior to the experiments and once daily during the irrigation experiments. In the flooding experiment two inlet samples were taken from the outflow tubes of each water tank during the 30 min flooding session. Soil water was sampled from the piezometers each morning for five or six days and analysed the same the day. Ammonia-N, nitrate-N, nitrite-N and phosphate-P were determined on filtered samples (Whatman GF/C) using a multichannel flow-injection analyser, QuikChem automated ion analyzer, (LACHAT INSTRUMENTS, U.S.), methods 10-107-06-3-D, 10-107-04-1-B (Nitrite, Nitrite + Nitrate), 10-115-01-1-B, respectively (Lachat instruction manual). Sulphate and nitrate-N (>1 mg-N l^{-1}) were determined by ion chromatography on a Shimadzu HIC-6A chromatograph using an IC-A1S anion column (Shimadzu Corporation, Japan) following pre-filtration (0.2 µm membranefilter, Schleicher and Schuell). Total iron was determined according to Danish Standards /13, 14/ and determined by atomic absorption spectroscopy (model 2380, Perkin Elmer). pH was measured using a PHM 93 pH meter (Radiometer, Denmark).

Statistics

Although the experiments were designed to be balanced, watersamples could not be obtained from all the piezometers every day and the data was therefore analysed by generalized linear models for unbalanced data /15/. To improve normality and homogeneity of variances prior to analysis the data was transformed by a Box-Cox transformation /16/. An overall significance level of 5% was accepted for all tests. All interactions mentioned below in the results paragraph are in the statistical sense of the word. Statistical analysis of the variables revealed no significant inter-plot differences and all four plots were therefore treated as replicates (n = 4). In tables and figures the results of the statistical analyses are shown after a transformation of the data back to original untransformed values. Consequently, it is the median values that are shown with 95% confidence limits (c.l.).

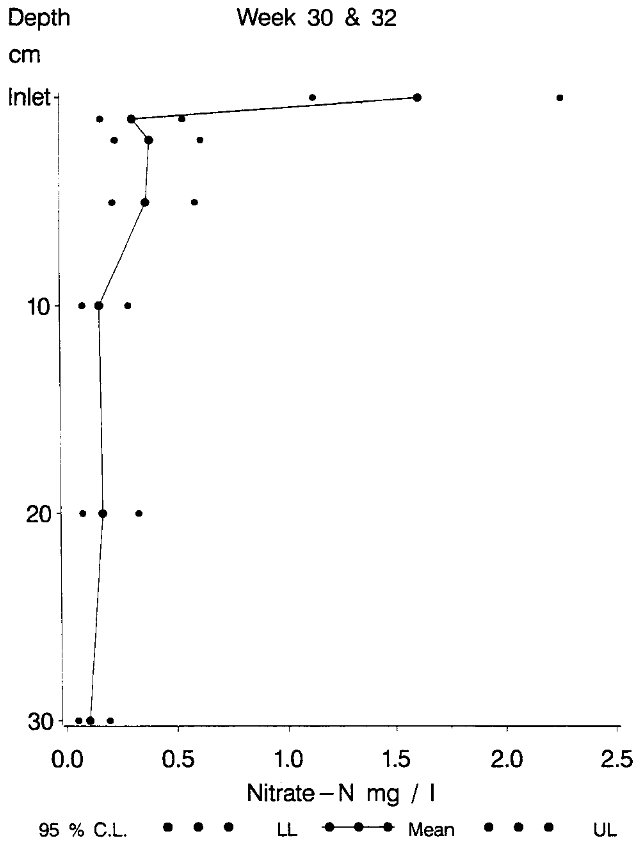

Fig. 2. Estimated nitrate-N concentrations at different depths with 95% c.l. from the two irrigation experiments (n=177).

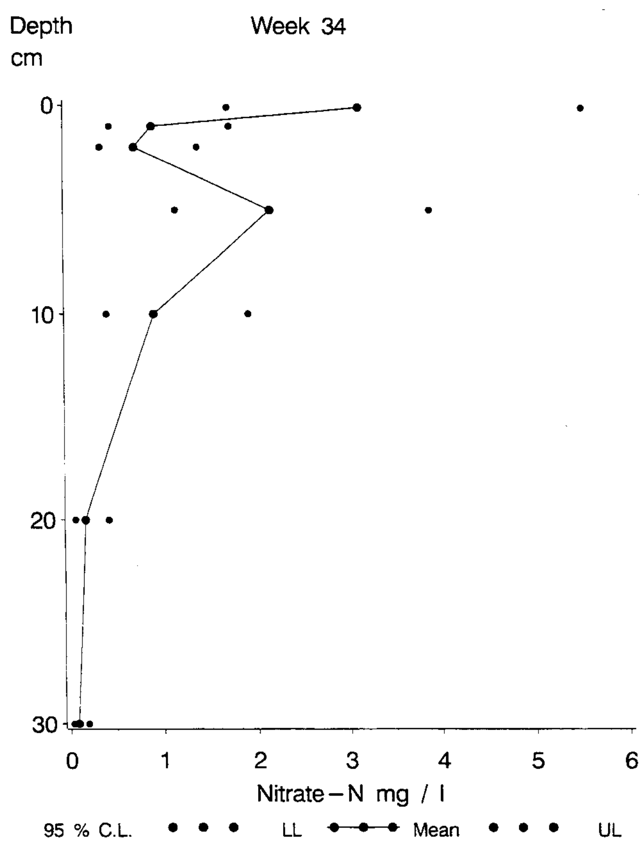

Fig. 3. Estimated nitrate-N concentrations at different depths with 95% c.l. from the flooding experiment (n=128). Surface water concentration (0) is shown instead of the inlet concentration.

Results

Analysis of variance (SAS/GLM) of the two irrigation experiments (week 30 and 32) did not reveal any significant inter-week difference with respect to nitrate as the dependent variable. Nitrate observations from these two experiments were therefore analysed together. The unadjusted (observed) mean in nitrate concentration in the inlet water was 1.755 mg NO_3^--N l^{-1} but decreased to 0.323 mg NO_3^--N l^{-1} at depth 1-2 cm, and further to 0.093 mg NO_3^--N l^{-1} at 30 cm. Nitrate data from the two irrigation experiments in week 30 and 32 was transformed to the power of 0.2 in order to improve normality and homogeneity. The expected value of nitrate could be described by the model: $NO_3{}^{0.2}$ = Depth, Date, r^2 = 0.61. The mediean values of depth and date visualised in figure 2 and table 1, respectively, show that the nitrate-N concentration decreased both with depth and date. The major part of the nitrate reduction, i.e. 82%, took place already in the upper 0-2 cm of the soil.

Also in the flooding experiment in week 34 the nitrate concentration could be described as being dependent on the two main factors depth and date. However, since the river water was added to the plots rapidly, i.e for 30 min, inlet water (enriched to a final concentration of 10 mg NO_3^--N l^{-1}) was omitted in the statistical analysis and instead surface water (depth 0 cm) , now present, was included. Mean observed nitrate concentration in the surface water was 3.106 mg NO_3^--N l^{-1}, while the concentration at depth 2-3 cm had decreased to 0.711 mg NO_3^--N l^{-1} . The nitrate level decreased gradually from 2.376 mg NO_3^--N l^{-1} on 22 Aug to 0.074 mg NO_3^--N l^{-1} on 26 August (Table 2). The statistical model: $NO_3{}^{0.1}$ = Depth, Date, r^2 = 0.70, is illustrated by the median values of depth and date in figure 3 and table 2, respectively. Although there was a significant increase (and difference) in nitrate concentration from depth 2 cm to depth 5 cm (Figure 3) there was no significant difference between nitrate concentrations at neither depth 0, 1, 5, and 10 cm nor at depth 0, 1, 2 and 10 cm. The overall result was two distinct sets of means: 0, 1, 2, 5, and 10 cm vs 20 and 30 cm.

Table 2. Estimated nitrate-N concentrations during week 34, with 95% confidence limits (n=128). Two distinct sets of means can be identified (22, 23,24, 25 Aug) vs (26 Aug).

Date	NO_3-N mg l^{-1}	95% Confidence limit Lower mg l^{-1}	Upper mg l^{-1}
22 Aug	2.136	1.248	3.559
23 Aug	1.781	1.041	2.965
24 Aug	1.001	0.566	1.716
25 Aug	0.458	0.247	0.819
26 Aug	0.074	0.034	0.150

Opposite to nitrate, sulphate concentrations increased with depth in all three experiments. There was a tendency towards a decrease in sulphate levels from start to end of the experiments. Sulphate concentration at the inlet was 51 - 53 mg SO_4^{2-} l^{-1} but at depth 10 cm the concentration had gradually increased to 274, 127 and 228 mg SO_4^{2-} l^{-1}, respectively for week 30, 32 and 34 (observed means). From depth 10 cm to depth 20 cm sulphate concentration decreased markedly to 100, 93 and 64 mg SO_4^{2-} l^{-1}, respectively for week 30, 32 and 34 (observed means). In all three experiments sulphate concentration again increased from depth 20 cm to depth 30 cm, observed means being 204, 165 and 133 mg SO_4^{2-} l^{-1}, respectively. As with nitrate, the statistical analysis revealed that sulphate concentration was dependent on depth and date (Table 3). Additionally, sulphate concentration was influenced by pH in week 32 and 34, i.e. sulphate concentration increased with constant (or decreasing) pH. Moreover, in week 34 there was an interaction between depth and pH which significantly influenced the expected sulphate concentration (Table 3). The interaction could be explained entirely by surface water, where sulphate concentration decreased with constant (or increasing) pH, while at the other depths sulphate concentration increased with constant (or decreasing) pH. Estimated median sulphate concentration with 95% c.l. at each depth is shown for each of the weeks in Figure 4. Two distinct sets of means could be identified in week 30: Inlet vs 1 cm, 2 cm, 5 cm, 10 cm, 20 cm, 30 cm, and in week 32: Inlet, 1cm, 2 cm, 5 cm, 10 cm and 20 cm vs 30 cm. In week 34 no distinct sets of means could be identified, although there were significant differences between some depths. Compared to the observed mean at depth 10 cm (228 mg SO_4^{2-} l^{-1}) the model prediction at depth 10 cm in week 34 is inaccurate.

Table 3. Analysis of variance with sulphate as response variable. Normalized by use of Box-Cox transformations. mg

Week				R^2	n
30	$1/SQRT(SO_4)$	=	Depth, Date	0.78	54
32	$1/SO_4$	=	Depth, Date, pH	0.81	98
34	$SQRT(SO_4)$	=	Depth, Date, pH, Depth*pH	0.86	120

Table 4. Estimated sulphate levels during week 30, 32, 34. Each week analysed separately. Week 30: n = 54, week 32: n = 98, week 34: n = 120

Date	SO_4 mg l^{-1}	95% Conf. Lim. Lower mg l^{-1}	Upper mg l^{-1}	Compares equal to
26. Jul	178.1	134.5	246.7	27, 29
27 Jul	130.6	105.4	166.0	all
28 Jul	111.7	89.1	144.1	27, 29
29 Jul	134.5	101.9	185.7	all
08 Aug	104.6	89.4	125.9	9
09 Aug	90.4	80.7	102.8	8
10 Aug	75.5	68.3	84.5	11, 12
11 Aug	72.5	66.2	80.3	10, 12
12 Aug	69.2	61.7	78.7	10, 11
22 Aug	81.3	65.6	98.7	All
23 Aug	90.0	76.2	105.0	22, 24, 25
24 Aug	95.3	79.9	112.1	22, 23, 25
25 Aug	86.4	73.4	100.4	all
26 Aug	70.4	58.6	83.2	22, 25

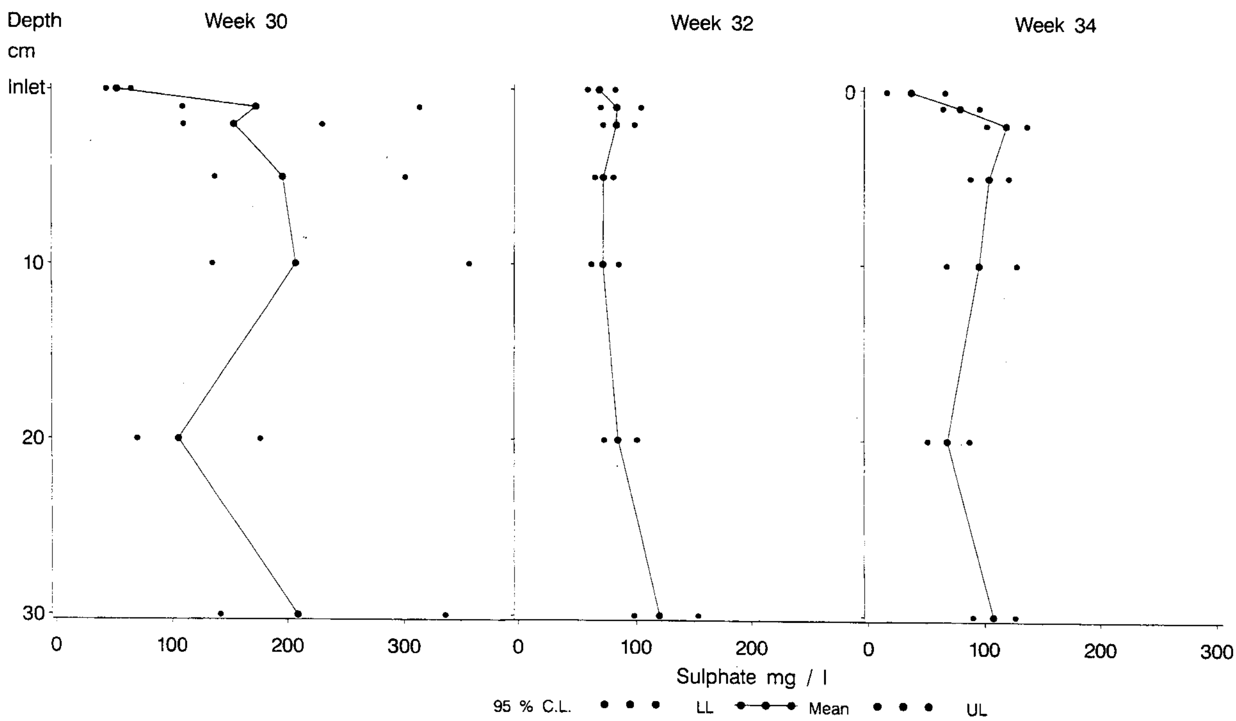

Fig. 4. Estimated Sulphate concentrations at different depths with 95% c.l. from week 30, 32 and 34, respectively. Week 30: n=54; week 32: n=98; week 34: n=120. In week 34 the surface water concentration (0) is shown instead of the inlet concentration.

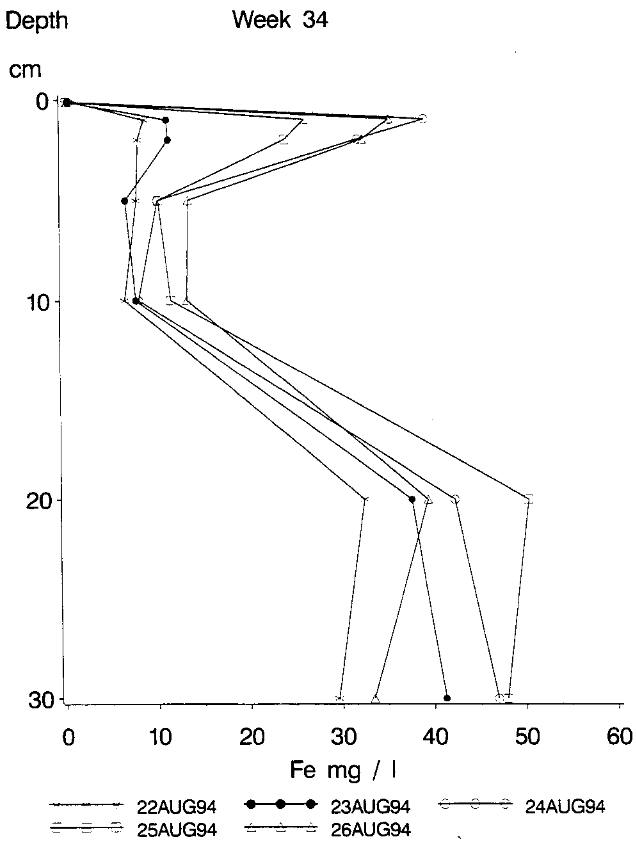

Fig. 5. Estimated iron concentration at different depths and dates in week 34 (n=121).

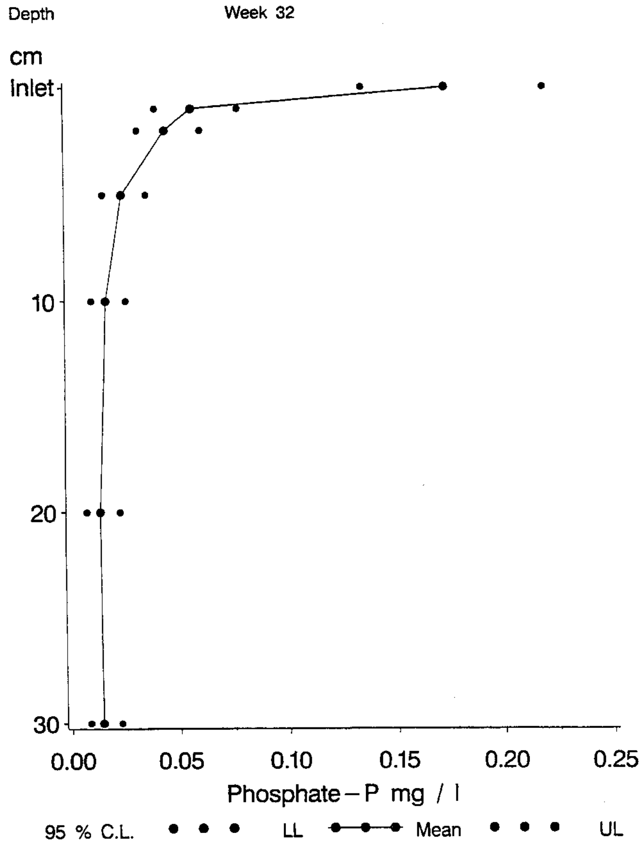

Fig. 7. Estimated phosphate-P concentration at different depths in the second irrigation experiment with 95% c.l. (n=140).

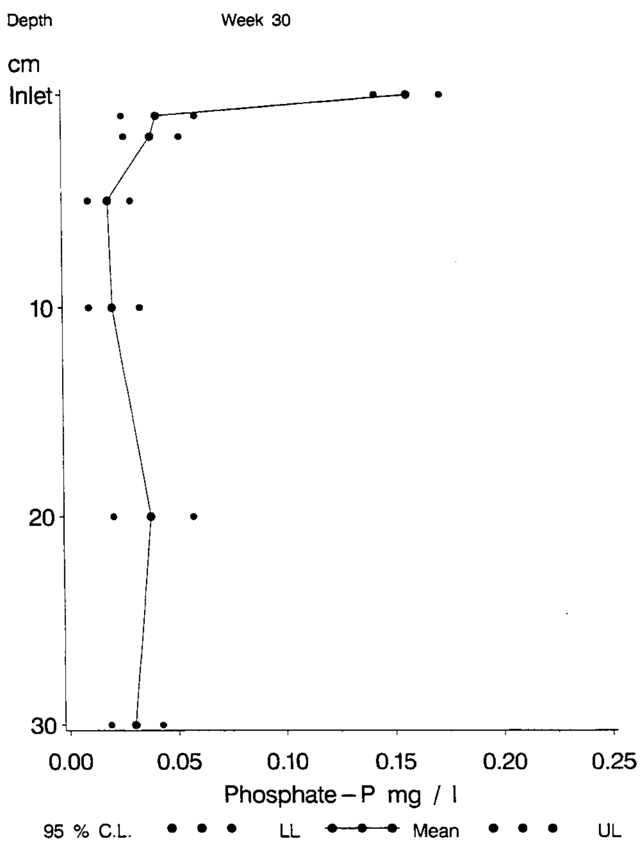

Fig. 6. Estimated phosphate-P concentration at different depths in the first irrigation experiment with 95 % c.l. (n=105).

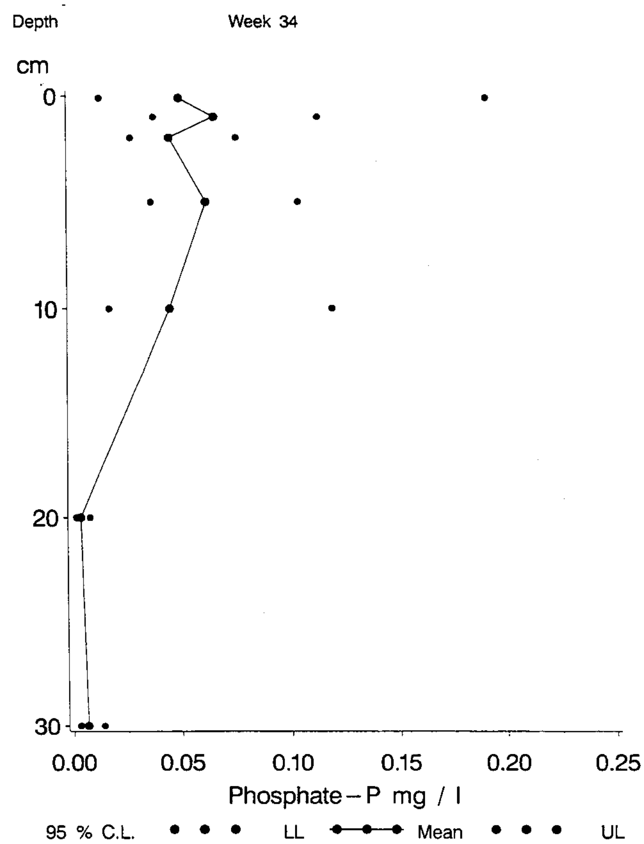

Fig. 8 Estimated phosphate-P concentrations at different depths with 95% c.l. from the flooding experiment (n=129). Surface water concentration (0) is shown instead of the inlet concentration

Table 5. Analysis of variance with Iron as response variable. Normalized by use of Box-Cox transformations.

Week		R^2	n
30	LOG10(Fe) = Depth, NO_3, pH	0.92	24
32	LOG10(Fe) = Depth, NO_3, SO_4*Date	0.79	98
34	LOG10(Fe) = Depth, Depth*Date, pH	0.90	121

Table 6. Estimated iron concentrations at different depths in week 30 (n=24) and week 32 (n=98). With 95% c.l. Two distinct sets of means can be identified in week 30: (Inlet) vs (5, 10, 20 and 30 cm) and in week 32: (Inlet, 1, 2, 5, 10 and 20 cm) vs (30 cm). No data available for depth 1 and 2 cm in week 30.

	Week 30			Week 32		
		95% Conf.lim.			95% Conf.lim	
Depth cm	Fe mg l^{-1}	Lower mg l^{-1}	Upper mg l^{-1}	Fe mg l^{-1}	Lower mg l^{-1}	Upper mg l^{-1}
Inlet	0.12	0.03	0.54	1.82	0.84	3.93
1	.	.	.	2.66	1.27	5.56
2	.	.	.	2.46	1.46	4.13
5	4.14	1.03	16.71	1.13	0.73	1.75
10	26.87	3.17	227.75	1.40	0.81	2.43
20	5.29	1.09	25.76	2.40	1.38	4.19
30	31.23	8.79	111.00	16.03	8.29	31.01

Table 7. Analysis of variance with phosphate as response variable. Normalized by use of Box-Cox transformations.

Week			R^2	n
30	$PO_4^{0.7}$	= Depth, Date	0.94	57
32	$PO_4^{0.3}$	= Depth, SO_4	0.78	118
34	LOG10(PO_4)	= Depth, SO_4, Fe, Depth*pH	0.70	115

Table 8. Estimated phosphate levels for each day in week 30 (n=57). 95% confidence limits included.

		Conf. Limits		
Date	PO_4-P mg l^{-1}	Lower mg l^{-1}	Upper mg l^{-1}	Compares equal to
25 JUL	0.058	0.035	0.084	All dates
26 JUL	0.057	0.047	0.068	25JUL, 27JUL
27 JUL	0.044	0.036	0.053	All dates
28 JUL	0.034	0.025	0.044	25JUL, 27JUL, 29JUL
29 JUL	0.034	0.024	0.045	25JUL, 27JUL, 28JUL

The observed concentrations of iron also increased with increasing depth in all three experiments. The inlet concentrations of iron in week 30, 32 and 34 were 0.65, 0.76, and 0.61 mg Fe l^{-1} , respectively. In the soil the corresponding iron concentrations increased gradually with depth to 24.31, 32.34, and 42.99 mg Fe l^{-1} , respectively, at a depth of 30 cm (simple means). Another general trend for all three experiments was that pH decreased from 8.0 - 8.1 in the inlet water to 4.2 - 5.2 at depth 10 cm and then increased again to 5.7 - 6.4 at depth 20 and 30 cm. The statistical analysis for iron very much resembles those for sulphate and nitrate. Thus in all three experiments, iron concentation varied significantly with depths (Table 5 and Figure 5).

In week 30 two other factors, pH and nitrate, also affected the expected concentration of iron. The resultant statistical model: LOG10(Fe) = Depth, NO_3, pH, explained 92% of the variation. Table 6 shows the estimated iron concentrations with 95% c.l. for inlet water and all depths in the irrigation experiments. In the week 32 experiment nitrate also had a significant influence on the iron concentration, as did an interaction between sulphate and date. Hence the statistical model was: LOG10(Fe) = Depth, NO_3, SO_4*Date, r^2 = 0.79. The interaction was a result of a relatively high sulphate concentration (level) and the highest iron concentration (level) on 12 August. The pattern for the other days showed that the iron level rose gradually from 2.66 ± 5.32 mg Fe l^{-1}, (± standard deviation, n = 17) on 8 Aug to 9.06 ± 16.05 mg Fe l^{-1}, (n = 23) on 12 Aug while the sulphate level decreased from 132.78 ± 71.46 mg l^{-1} (n = 24) on 8 Aug to 78.52 ± 45.88 mg l^{-1} (n = 27) on 11 Aug, but increased again to 89.92 ± 50.57 mg l^{-1} (n = 25) on 12 Aug.

The statistical model for the flooding experiment: LOG10(Fe) = Depth, pH, Depth*Date, explained 90% of the variation. The interaction between depth and date, which is illustrated in Figure 5, was the result of alternating shifts in iron concentration at the depths: 2 cm, 10 cm and 30 cm on different dates.

The inlet concentration of phosphate in week 30, 32 and 34 was 0.160, 0.176, and 0.152 mg PO_4^{3-}-P l^{-1}, respectively. The overall picture was that phosphate concentration decreased significantly with depth in all three experiments. Most notably was the change at the soil surface and in the uppermost 2 cm of the soil. In week 30 there was a slight increase in phosphate concentration from 5 cm to 20 cm but this was not significant according to the statistical model: $PO_4^{0.7}$ = Depth, Date; r^2 = 0.94 (Table 7, Figure 6). There were two distinct sets of means: inlet water vs 1, 2, 5, 10, 20 & 30 cm. Day to day variation showed significant differences between some of the dates and a tendency towards lower concentration at the end of the experiment (Table 8). In week 32 phosphate concentration was dependent on the two main factors depth and sulphate concentration. The model: $PO_4^{0.3}$ = Depth, SO_4, explained 78 % of the variation. Three sets of distinct means could be identified: inlet vs 1 & 2 cm vs 5, 10, 20 & 30 cm (Figure 7). For all depths phosphate concentration decreased with increasing sulphate concentration. In week 34 the inlet concentration of phosphate (0.152 mg PO_4^{3-}-P l^{-1}) was not included in the the statistical analysis due to the above mentioned experimental procedure. The concentration of phosphate in the soil water was markedly lower than the inlet concentration, while phosphate concentration in surface water showed great variation, (Figure 8). In week 34 phosphate concentration was further dependent on iron, sulphate and an interaction between depth and pH, which could be described by the following statistical model: LOG10(PO_4)= Depth, SO_4, Fe, Depth*pH; r^2 = 0.70. Compared to the observed means in surface water and depth 1 cm, which was 0.120 and 0.086 mg PO_4^{3-}-P l^{-1}, respectively, the model prediction is not very precise (Figure 8). The interaction depth*pH was due to decreasing phosphate concentration with increasing pH at depths: 2 and 30 cm while phosphate concentration increased with increasing pH in surface water and at depth 5, 10 and 20 cm (depth 1 cm being indifferent). The concentration of phosphate with depth showed two distinct sets of means: surface water, 1, 2, 5, 10 vs 20 & 30 cm (Figure 8).

Total-P concentration measured in the inlet water in the three experimental weeks was: 0.217, 0.341 and 0.335 mg P l^{-1}, respectively. The corresponding load was 54, 128 and 84 mg P m^{-2} $week^{-1}$. The load of phosphate was 40.0, 65.8 and 38.0 mg PO_4^{3-}-P m^{-2} $week^{-1}$ for the weeks 30, 32, 34, respectively. The retention of phosphate-P calculated as the difference between the inlet concentration multiplied by the hydraulic load and the concentration at depth 30 cm multiplied by the hydraulic load (250, 375 and 250 l m^{-2} $week^{-1}$, respectively) was 33.5 (84%), 60.0 (91%) and 27.4 (72%) mg PO_4^{3-}-P m^{-2} $week^{-1}$, respectively.

Discussion

The observed and statistically significant decrease in nitrate concentration with depth and date, which was concomitant with significant increases in both iron and sulphate concentrations with depth, is attributable to waterlogging initiated at the beginning of each of the experiments. Waterlogging creates a shift in redox conditions due to a greater demand for oxygen than what is present in the floodwater. As oxygen presumably is depleted from the topsoil very rapidly, nitrate becomes the next avaible electron acceptor and also nitrate is exhausted from the soil i.e used by denitrifying microbes. The increase in sulphate and iron concentration seems to indicate that at least part of the nitrate reduction was caused by autotrophic denitrification. Kölle et al. /17,18/ proposed the following two step reaction to take place:

$$5FeS_2 + 14NO_3^- + 4H \rightarrow 7N_2 + 10SO_4^{2-} + 5Fe^{2-} + 2H_2O \quad (1)$$

$$5Fe^2 + NO_3^- + 7H_2O \rightarrow 5FeOOH + 0.5N_2 + 9H^+ \quad (2)$$

In the present study, reaction 2, which is mediated by iron bacteria /19/, may have been of limited importance, since nitrate was rapidly depleted from the upper part of the soil. Further, during the laboratory work, it was noticed that many of the water samples appeared reddish and "thick" when they had been left in the laboratory for a couple of hours. Therefore, although only total iron was measured, it is most likely that most iron was in fact ferrous iron. At depth 10 cm the decrease in pH might have been caused by reaction 2. In all three experiments an increase in iron concentration from 10 cm to 30 cm was observed. This was presumably not due to autotrophic denitrification as nitrate was only present in trace amounts. The waterlogging conditions have instead exhausted the meadow soil of electron acceptors and thus the lowering of the redox potential was caused by a reduction in ferric iron already present in the soil.

Also, the decrease in sulphate concentration with time in all three experiments can be explained by the waterlogging of the meadow soil and the subsequent drop in redox potential resulting in sulphate reduction.

The alternating shifts in sulphate concentration, i.e decreasing from depth 10 cm to 20 cm, and increasing from 20 cm to 30 cm, which were seen in all three experiments, occurred together with observed increases in iron concencentration from 10 to 30 cm and also together with increases in pH. A possible explanation to these observations could be oxidation of thiosulphate to sulphate. Jørgensen (1990) has proposed the following reaction:

$$S_2O_3^{2-} + 8FeOOH + 14H^+ \rightarrow 2SO_4^{2-} \; 8Fe^{2+} + 11H_2O \quad (3)$$

as a possible (but hypothetical) pathway. According to Jørgensen /20/ the only well-known electron acceptor for anoxic thiosulphate oxidation is nitrate, but reactive ferric compounds or pyrite could have a similar function. Presumably the meadow soil consists of both ferric and ferrous compounds as the groundwater level fluctuates near the soil surface during the year - being above ground in winter and approximately 50 cm below ground in summer.

Compared to the inlet water the phosphate concentration at the soil surface and in the upper 2 - 3 cm of the soil decreased 75, 73 and 64% in the weeks, 30, 32 and 34, respectively. This meadow soil probably has a phosphate sorption capacity which is higher than other organic wetland soils. This can be ascribed to the (anticipated) high content of iron, seen by inspection of the soil profile that revealed iron nodules. It is furthermore worth noticing that the phosphate concentration decreased deeper in the soil, in spite of the waterlogging conditions created by the experiments. In this context it should also be noted that uptake of phosphate by the vegetation must have been of minor importance, as the study started at the end of the growing season.

Other studies have also reported phosphate retention. Similar to this study Gumbricht /21/ found a mean reduction in total phosphorus from 0.26 to 0.07 mg l^{-1} in a submerged macrophyte pond which was loaded with stream water. The corresponding removal rate was 0.04 g m^{-2} d^{-1}. In summer the reduction was even higher, reaching 82% /21/. Reddy and Graetz /22/ found the maximum reduction (36%) in ortho-P concentration following a residence time of six days in an organic soil loaded with agricultural drainage water.

In conclusion this study showed that short-term flooding or waterlogging of wet meadows during summertime can reduce the concentration of nitrate and phosphate in the infiltrating stream water. But at the same time the waterlogging conditions had decisive effect on the iron transformation processes, which were not fully elucidated in this study.

Acknowledgement

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Dynamic model for sedimentation and resuspension of phosphorus for stream reaches

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Abstract

The need for strategic water management tools has initialized the development of a modelling system (TRANS) for simulation of biological, chemical and physical processes affecting nutrient transport, accumulation and removal in streams, lakes and wetlands. This paper demonstrates that the stepwise development procedure from empirical models to dynamic models can be used in a description of the sedimentation and resuspension of particulate phosphorus (P) in streams. The calibration of the transformed empirical model in a dynamic modelling system shows that the accumulated phosphorus in the sediment can be simulated. However, when validating the model for another reach in the same watershed the results reveal that the model needs further improvement.

Introduction

The Danish Strategic Research Programme, which is a co-ordinated programme between several ministries, started in 1992 and continues until 1996. As a part of this programme empirical models describing the nutrient transport in Danish streams and lakes are developed /1/. These non-dynamic models, i.e. static models without any spatial and time resolution, are developed based on data from the Nationwide Monitoring Programme covering 270 Danish streams and 37 lakes and research data from the River Gjern watershed.

Dynamic models for the transport, accumulation and removal of nutrients in freshwater systems, however, are further developed during the programme in a joint NERI/VKI project. This project is aiming at implementing dynamic models for the nutrient transport in a one-dimensional modelling system and calibrating these models on an intensively monitored watershed (River Gjern) /1,2/. The calibrated modelling complex will hereafter be used in the simulation of different catchment management scenarios.

This paper describes the initial integration of static sub-models within a dynamic modelling system. In this first phase a model for the sedimentation/ resuspension of P in streams, developed from an analysis of data from three streams is implemented, tested and calibrated for two intensive stations in the River Gjern watershed.

Methods

Study Site
The 113.5 km^2 lowland watershed of th River Gjern, situated in central Jutland, Denmark, is monitored for discharge, nutrient concentrations and accumulation of particulate matter and associated nutrients at 35 representative stations /3/. Two of the stations are monitored intensively and measurements of nutrient concentrations are conducted every hour by means of automatic samplers. The two stations were chozen to represent a 1. order tributary (the Gelbæk stream) and the main river channel /3/. At each station the net accumulation of sediment associated nutrients in stream sediments are measured by random sampling every fortnight on a 250 m and 160 m sub-reach, respectively /4/. Moreover, the biomass of aquatic macrophytes is measured by means of monthly random sampling on the sub-reaches. The 33 extensive stations are measured for the same parameters though less frequent, i.e. monthly sampling. Moreover, cross-sectional dimensions of the whole river network have been surveyed /3/.

Modelling

The modelling work includes the following

- Development of empirical (static) models for general description of resuspension and sedimentation of phosphorus

- Setup and calibration of a hydrodynamic physical based model for the river network

- Implementation and test of the developed empirical models for phosphorus using the dynamic and hydrodynamic calibrated model as the underlying water transport mechanism (advection/dispersion)

Empirical models
Modelling of data from 77 small watersheds revealed significant empirical relationships between nutrient losses and various predictor variables /5/. The empirical description of the sedimentation and resuspension of phosphorus in stream sediments in question are developed through an analysis of data from 3 streams /6,7/. The development of these equations are discussed in detail in /7/. The final equations read

$$SED = \beta^*(p_1 + p_2{}^*BIO) * (1 - V_a{}^*V_a/V_{a-1})^*C \quad (1)$$

$$RES = (1-\beta)^*(p_3 + p_4{}^*BIO) * (V_a - V_{a-1})^{2*}CUM \quad (2)$$

$$\text{if } (V_a - V_{a-1}) > p_5 \text{ then } \beta = 0 \text{ else } \beta = 1 \quad (3)$$

where

SED = Sedimentation of P (g P day^{-1})
RES = Resuspension of P (g P day^{-1})
BIO = Biomass of macrophytes (g DW m^2)
C = Concentration of total-P in the water phase (mg P l^{-1})
V_a = Mean daily flow velocity (m s^{-1})
V_{a-1} = Mean flow velocity previous day (m s^{-1})
CUM = Accumulated phosphorus in sediment (g P m^2)
$p_1..p_5$ = Empirical parameters to be calibrated

Hydrodynamic model

The water flows of the watershed are simulated using the MIKE 11 modelling system. The MIKE 11 model developed at DHI/VKI is a well known comprehensive, one-dimensional modelling system for simulation of flows, sediment transport and water quality in estuaries, rivers, irrigation systems and other water bodies /9/.

The set-up for River Gjern includes 13.4 km of the main channel (River Gjern), 11 tributaries and 1 lake (Lake Søbygaard). The total length of the modelled stream system accounts to 45.3 km distributed in 213 calculation (grid) points. The hydrodynamic model has been calibrated against 8 internal discharge and 11 water level stations giving a trustworthy data basis for the subsequent P simulations.

Implementation of sedimentation and resuspension in the dynamic model

The dynamic model, MIKE 11, is based on the one-dimensional equation of mass conservation for dissolved or suspended material (advection/ dispersion) combined with a system of coupled differential equations describing the physical, chemical and biological processes occurring in waters. The advection/dispersion (AD) is calculated from the simulated hydrodynamics (HD). The HD and AD modules of MIKE 11 are some of the basic components of the modelling system and have shown to be effective in numerous applications /8/. Consequently, the implementation of the P processes deals with the sedimentation and resuspension equations only.

In order to couple the developed model (equation no. 1-3) to a dynamic modelling system such as MIKE 11 one must adjust some of the model components. This applies to the mean flow velocity in particular since no "mean" values are used in a dynamic system. In MIKE 11 the actual, site-specific flow velocity is calculated in each time step as forced by the local conditions from the previous time step, the time varying water inflow/outflow at model boundaries and the lateral input/output.

The calculated flows therefore reflects the "true" velocities more accurate than the estimated mean daily velocities and should thus be used in the dynamic model description. However, the basis on which the equations are developed should be maintained at the same time, and the principles in the empirical model must be transformed unchangeable to the dynamic approach.

This contrast is solved by using a "circulating" mean value for the flow velocity, i.e. the daily mean velocity, V_a, is updated in each time step using the velocity calculated during the previous 24 hours. The same technique is applied for V_{a-1}, here the "circulating" value is calculated from the foregoing 48 hours and up to 24 hours before actual time.

Results

The dynamic model described above is calibrated and tested for two intensively measured stream reasches of 160 m and 250 m, respectively. The flow and channel dimensions of the two reaches are extracted from the hydrodynamic model covering the whole watershed producing two very small model setups. The simulation period covers two years (1993 - 1994) and the time step is 1 hour. The short distances along the two sampled sub-reaches imply that no longitudinal changes in the hydrodynamics are modelled.

The biomass of macrophytes, BIO, is included in the model as a forcing function, i.e. the model is "forced" to use the actual recorded biomasses.

The concentration of phosphorus in the water phase, C, is one of the two components or state variables in the simple dynamic model. No significant differences in the concentration of P in the water phase have been measured over the sub-reaches, meaning that the phosphorus concentrations of inflowing water are nearly equivalent to the outflow concentrations. Thus the calibration does not include component C. The simulated C, however, must be examined for a complete accordance between the measured and simulated phosphorus concentration.

The second state variable, which is the only true calibration parameter, is the accumulated P in the sediment, CUM. The model is calibrated for the intensive station located in the main river channel of River Gjern. The 5 model coefficients ($p_1..p_5$) are carefully adjusted until an acceptable accordance between measured and simulated accumulated sediment P is found. The result of the calibration is presented in Figure 1. In Table 1 the calibrated values of the model coefficients for the main channel reach are shown.

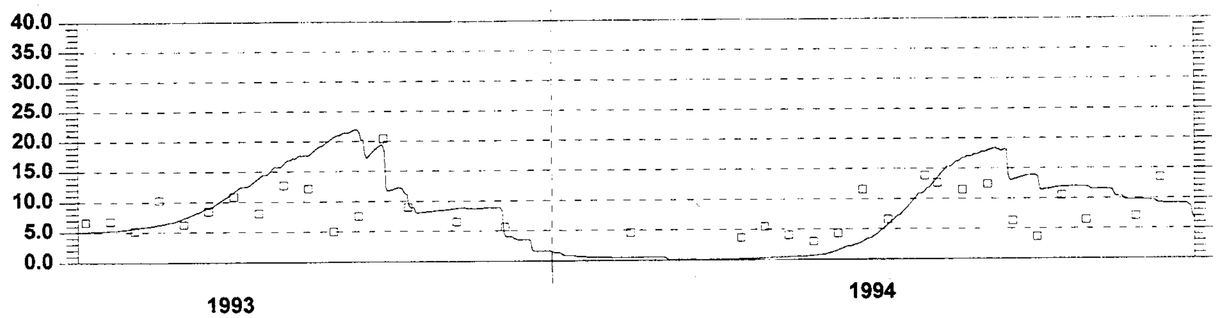

Fig. 1. Simulated and measured accumulated P in the sediment at the main channel reach. Unit is g P m^2.

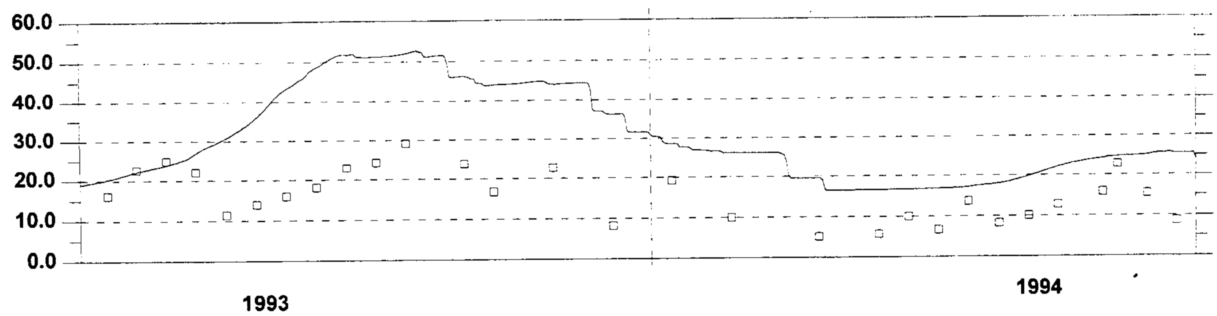

Fig. 2. Simulated and measured accumulated P in the sediment at the 1st order tributary reach, the Gelbæk stream. Unit is g P m^2.

Table 1. Calibrated value of model coefficients based on the sub-reach in the main channel of the River Gjern.

Model coefficient	Value
p_1	0.0007
p_2	0.0005
p_3	0.2
p_4	0.003
p_5	0.001

By inspection of figure 1 it can be seen that a relative good accordance between the simulated and the measured accumulated P on the stream bed exists. A strong seasonal variation with a net sedimentation during the summer period due to the presence of macrophytes and a dominating resuspension in the winter are observed.

By plotting the simulated rates of sedimentation and resuspension (not shown) it will be observed that resuspension is modelled with a considerably lesser frequency than sedimentation. However, the resuspension rate is about 10 times higher than sedimentation when occurring. No annual net accumulation is seen.

The simulated and measured P in the water phase (not shown) are identical. This confirms that the unmeasurable impact of the sedimentation/ resuspension rates on the water phase concentration during the short distance also is modelled.

The results of the subsequent validation against data from the other intensive station (the Gelbæk stream) in the watershed which represent a 1. order tributary reach are shown in figure 2. This station is characterized by very low flow velocities (about 10 times lower than in the main river channel during the summer period) and a much lower macrophyte biomass.

As will be seen from figure 2 the accumulated P on the stream bed is overestimated throughout the whole simulation period. The measured seasonal variation, however, is modelled rather closely indicating that the model description includes the controlling factors for sedimentation/resuspension.

The overestimation could be caused by a too high initial concentration of accumulated P in January 1993. The result shown in figure 2 is based on an initial (and measured) P-content of 19.2 g P m^2. A test simulation (not presented) setting the initial P-content in stream bed sediment to a value representing the situation during late April 1994 (5.0 g P m^2) shows that the overestimation is less pronounced. The peak measured during the spring of 1993, however, is not modelled.

This shows that the marked differences in flow velocities, channel dimensions and coverage of macrophytes between the calibration site in the River Gjern and Gelbæk stream is at the border of what the empirical relation is able to describe with the same set of parameters.

The spring peek of accumulated-P in 1993 is probably caused by a bloom of microbenthic algae, but unfortunately no measurements of chlorophyll were carried out in 1993-94 to verify this. Extreme high spring blooms of microbenthic diatoms have been observed in earlier investigations of the Gelbæk stream /9/. In these investigations the chlorophyll concentrations in the sediment were measured to be in the range of 500 to 1000 mg m^2 during April-May.

The spring peek of P disappears in May-June, at a time where emergent vegetation are developed, decreasing the light penetration to the stream bed.

The model does not include benthic microalgae and as such is not expected to include the increase in accumulated P in early spring 1993.

In the near future the model will be extended with a description of the physical-chemical immobilisation of P in sediment and a partition of total-P (C) into the dissolved and suspended phase. Furthermore, a sub-model modelling the growth of the aquatic macrophytes is incorparated. These extensions might improve the model performance so that the controlling P processes in streams are more adequately modelled.

Conclusion

The calibration shows that it is possible to transform empirical equations to a dynamic modelling system with an acceptable accordance between measured and simulated P content in the stream bed sediment at a main channel reach of 4th order. However, when applying the same set of calibrated parameters to another (and hydraulic different) part of the watershed (a 1st order stream), the process description is insufficient to cover the full range of phosphorus regulating processes, and a poor model verification is seen.

The future work to be carried out will reveal if it is possible to describe the P dynamics within a whole watershed using the same set of calibration parameters and a simple model.

In conclusion, the approach of a stepwise model development from regression analysis of a considerable amount of data to empirical models, before incorporation in dynamic models, has shown to be a promising method in the process of making strategic decision tools.

Acknowledgement

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The River Gjern Catchment

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Study area

The 113 km^2 River Gjern watershed is located in central Jutland, Denmark. The river system comprises approx. 85 km stream channels, which can be divided into four stream orders (*sensu Strahler, 1957*) (Fig. 1). The catchment has a gentle topography with elevations ranging from 20-135 a.s.l. Several monitoring stations ae situated in the River Gjern catchment (Fig. 1).

Soils in the 1st and 2nd order sub-catchments are dominated by clayey till deposits from Weichsel, whereas soils in the lower part of the watershed (3rd and 4th order) are mainly sandy glacio-fluvial deposits (*late Weichsel and early Holocean*), (Table 1 and Fig. 2).

River Gjern watershed is intensively farmed with an average 77% being agricultural land (Table 1 and Fig. 3).

Soil types and land use correspond to average Danish conditions. The climate is humid temperated with an annual mean temperature of 7.4°C and an annual mean precipitation of 770 mm during 1961-90. The precipitation shows only small seasonal variations with the highest amount in autumn and winter. The summer is cool and the winter mild.

Table 1. Description of the River Gjern catchment.

	Soil type		Land use				Hydrology	
Catchment area	Sandy	Loamy	Agricultural land	Forested land	Lakes and meadows	Towns and roads	Precipitation[1]	Annual runoff[2]
114 km^2	65%	35%	77%	14%	4%	5%	770 mm	333mm

[1]Average 1961-90
[2]Average 1974-94

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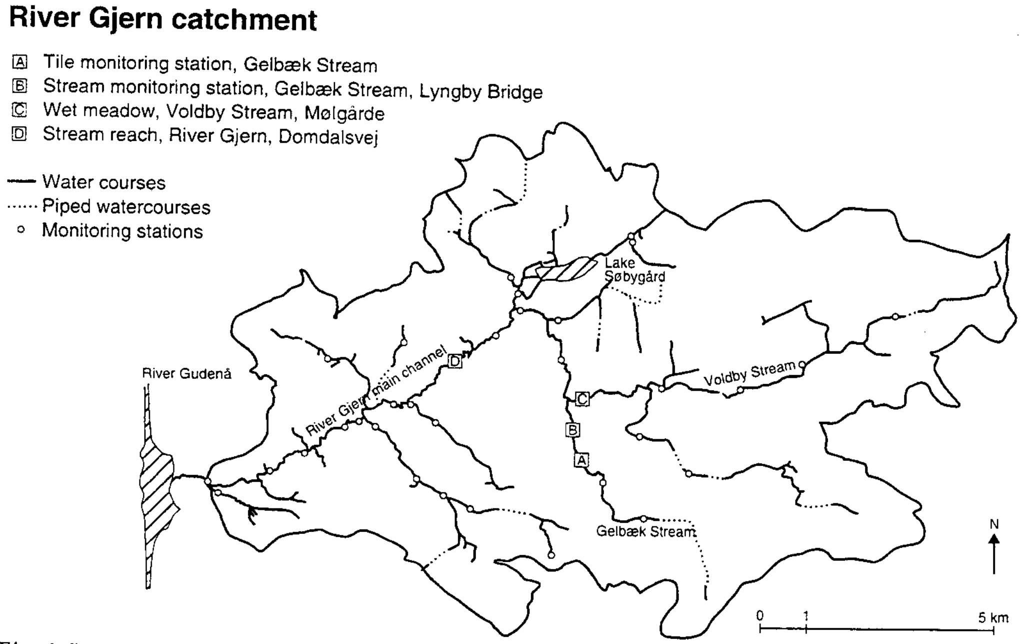

Fig. 1. Localities investigated in the River Gjern catchment to which some of the papers in these proceedings are referring.

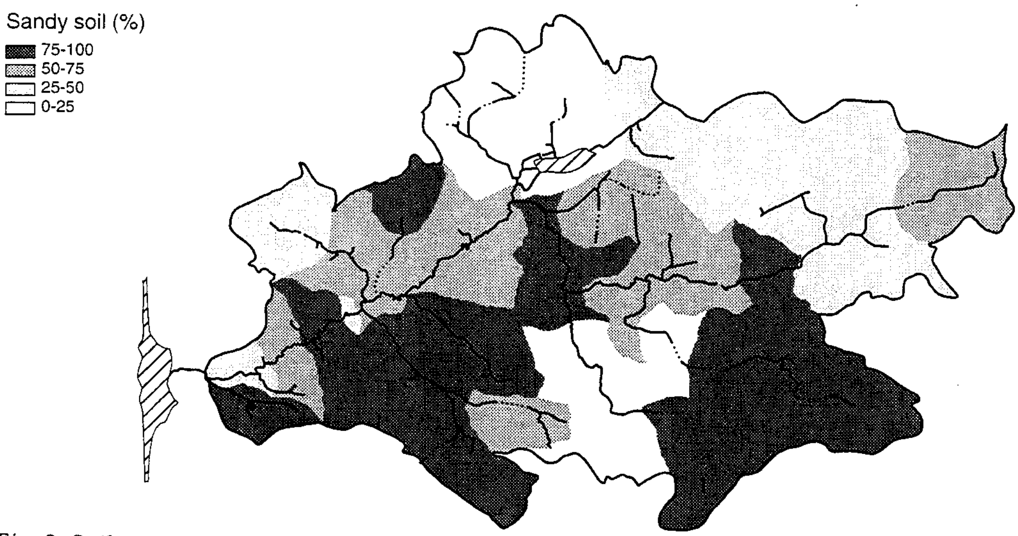

Fig. 2. Soil types in River Gjern catchment.

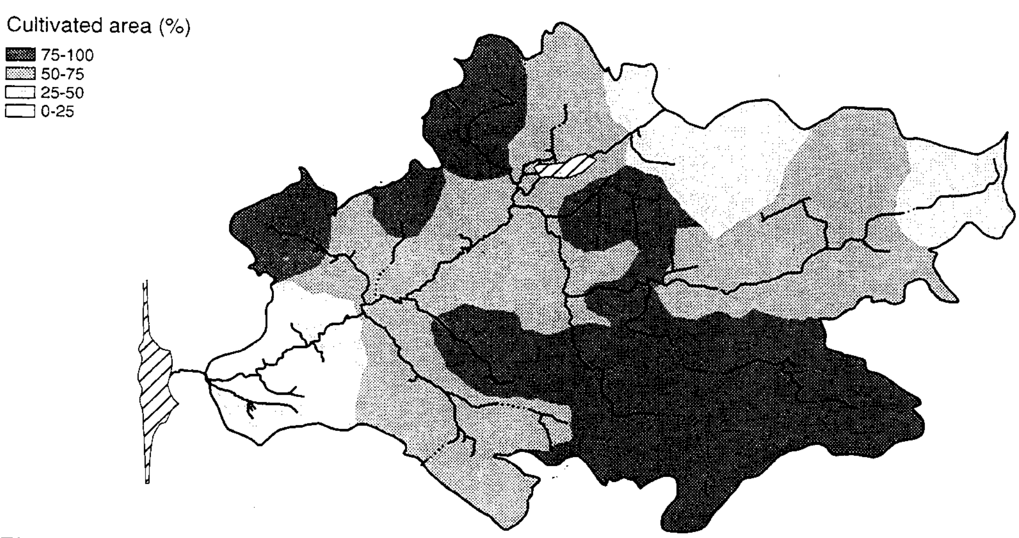

Fig. 3. Land use in the River Gjern catchment.

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