National Environmental Research Institute
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Delivery of suspended sediment and associated phosphorus and heavy metals to small rural Danish streams

PhD thesis

Anker R. Laubel

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Contents

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Preface

This thesis is submitted to the Faculty of Science and Engineering, University of Southern Denmark (USD). It represents my work from 1999-2002 at Department of Freshwater Ecology, the National Environmental Research Institute (NERI), Silkeborg, and at Department of Biology, University of Southern Denmark. My research supervisors were Henning S. Jensen (USD) and Brian Kronvang (NERI). The Danish Research Agency, and the National Environmental Research Institute financially supported the project.

I will thank all the people who inspired and helped me during the project. I am especially grateful to Brian for his constant encouragement, advice and enthusiasm. Also a special thank to colleagues at NERI who helped and who's company I have appreciated. Thanks to Uffe Mensberg, Henrik Stenholt, Mette Thomson and Susanne Husted for field- and laboratory work. Thanks to Anne-Dorthe Villumsen, Kathe Møgelvang and Tinna Christensen for technical and graphical design. Many thanks to Søren Larsen and Lars Svendsen. Also Ian Foster and Anthony Chapman in Coventry, UK, were very supportive and good company.

Silkeborg, February 2004

Anker Laubel

English Summary

The aim of this study is to examine delivery pathways for suspended sediment, and particulate phosphorus (P) and heavy metals from open rural areas to small Danish streams. A further aim is to quantify the contribution from different pathways and source areas. Such studies are useful as a basis for considering measures to reduce diffuse pollution of the aquatic environment.

The first paper describes subsurface transport of suspended sediment and P in plot studies and at field scale within the Gjern stream catchment in central Denmark. Macropores and tile drainage systems acted as delivery pathway from the fields to the stream. During storm events, rapid macropore flow transported water to the drains. Suspended sediment and P concentrations were highest in the initial drainage flow of the storms. Loss rates of suspended sediment and particulate P were in the same range in the plot experiments as at field scale. Tracer analysis using ^{137}Cs revealed that the sediment in the drainage water was derived from the topsoil.

Paper 2-4 investigates soil erosion and bank erosion at an extended number of study areas representing landscape and climatic conditions of Denmark. The sites, however, are located so that they have higher risk of surface runoff and soil erosion than what is normal for Denmark.

Agricultural management practice had a significant influence on soil erosion rates (paper 2). Hence, soil erosion rate during the 6-year study period was highest on fields with winter cereals (0.36 m^3 ha^{-1}) and lowest on fields with grass (0.00 m^3 ha^{-1}). Large temporal variations in soil erosion rate were observed. Site specific mass balances illustrate the potential of buffer zones for sediment and P retention.

Soil erosion rate was compared with bank erosion rate over an 11-month study period (paper 3). Bank erosion was roughly estimated to deliver a ten-fold higher sediment input to the streams than that from soil erosion. Further, optimal strategies for applying erosion pins in bank erosion studies was discussed.

Spatial variation in bank erosion rate was investigated and empirically modelled in paper 4. Bank erosion rate over a 2-year study period was significantly related to a number of site-specific characteristics. An empirical model based on these characteristics yielded a 55% explanation of the observed spatial variation in bank erosion rate. The total P content of bank material was high (0.64 g P kg^{-1}) and at the same level as found in agricultural topsoil along the streams. Comparison between bank erosion rates and total catchment losses suggested that bank erosion was a major contributor of suspended sediment and P.

Paper 5-7 encompasses catchment studies. The effectiveness of a sampling device ('Sub-Marie') for collecting time integrated sus-

pended sediment samples was assessed at the southern Danish Lillebæk stream (paper 5). The results indicated that the in-situ sampler did collect representative time-integrated suspended sediment samples during the period of deployment. Furthermore, spatial and temporal variations in the composition of suspended sediment was evaluated based on the use of the in-situ sampler. The collected samples were also applied for fingerprinting the dominant sources areas in the catchment. This data is included in the introduction chapter below.

Paper 6 investigates heavy metal and pesticide content in streambed sediment in relation to catchment use and sediment properties of 30 small Danish streams. Pesticides were significantly related to catchment size, soil type and hydrological regime, and one group of heavy metals (Cr, Cu, Pb, V and Zn) was related to urban activity and soil type.

Paper 7 includes quantitative data regarding sediment, P and nitrate pathways in Danish agricultural catchments. Particulate P constituted ca. 62% of the total P-export from the investigated catchments (total P export: 0.52 kg P ha^{-1}) during the 9 year monitoring period. The majority of the particulate P export seemed to derive from bank erosion. Surface runoff and soil erosion is likely to have contributed significant amounts of particulate P in wet years with snow melting on frozen soils. We estimated that particulate P contribution from subsurface tile-drainage water on average contributed 6% of the particulate P export from the loamy catchments.

Paper 8 is an associated paper investigating sediment and P deposition on a 5000 m^2 flood plain within the Gjern stream catchment in central Denmark. The trapping efficiency during overbank flooding, calculated as deposition divided by the transport in the river, varied from 3.6-23.9% for sediment and from 1.2-6.5% for particulate P. Hence, even small inundated flood plains play a positive role by reducing the export to downstream water bodies.

Dansk sammenfatning (Danish Summary)

Formålet er at undersøge transportveje for suspenderet stof, partikulært fosfor (P) og tungmetaller fra det åbne land til små danske vandløb. Derudover er det et mål at kvantificere bidraget fra de forskellige transportveje og kildeområder. Sådanne undersøgelser kan bruges som baggrund for en vurdering af, hvordan man kan nedsætte den diffuse forurening af vandmiljøet.

Den første artikel beskriver transporten af suspenderet stof og P under jordoverfladen i plotforsøg og på markskala i Gjern Å oplandet i det centrale Danmark. Makroporer og drænsystem fungerede som transportvej fra marken til vandløbet. Under kraftige afstrømningshændelser blev vandet hurtigt transporteret gennem makroporer ned til drænene. Koncentrationen af suspenderet stof og P var højest i det første vand, der strømmede. Tabet af suspenderet stof og partikulært P var i samme størrelsesorden i plotforsøgene som på markskala. Sporanalyser med ^{137}Cs viste, at sedimentet i drænvandet stammede fra overjorden.

Artikel 2-4 undersøger jorderosion og brinkerosion på et større antal lokaliteter, som repræsenterer landskab og klimatiske forhold i Danmark. Lokaliteterne ligger dog, så der er en højere risiko for overfladisk afstrømning og jorderosion, end hvad der er normalt for Danmark.

Landbrugspraksis havde en signifikant indflydelse på jorderosion (artikel 2). Jorderosionen var således højest på marker med vinterkorn (0.36 m^3 ha^{-1}) og mindst på græsmarker (0.00 m^3 ha^{-1}) gennem 6 års perioden. Der var store tidslige variationer i jorderosionen. Massebalancer for de enkelte lokaliteter viser bræmmernes potentiale for at tilbageholde sediment og P.

Jorderosion blev sammenholdt med brinkerosion gennem en 11 måneders periode (artikel 3). Et groft estimat peger på, at brinkerosion bidrog med cirka 10 gange mere sediment til vandløbene, end der kom fra jorderosion. Derudover diskuteres der optimale strategier for anvendelse af erosions-pinde i undersøgelser om brinkerosion.

Den rumlige variation i brinkerosion blev undersøgt og empirisk modelleret i artikel 4. Brinkerosion var over en to års periode signifikant relateret til en række lokale faktorer. En empirisk model gav baseret på disse faktorer, en forklaringsgrad på 55% for den rumlige variation i brinkerosion. Indholdet af total P var højt (0.64 g P kg^{-1}) i brinker langs markerne, og på niveau med indholdet i markernes overjord. En sammenligning mellem brinkerosion og oplandstab tyder på, at brinkerosion leverer en væsentligt andel af vandløbenes suspenderet stof og P.

Artikel 5-7 indeholder oplandsanalyser. Et apparat ('Sub-Marie') til prøvetagning af suspenderet stof blev afprøvet i sydfynske Lillebæk.

Resultaterne indikerer, at apparatet faktisk opsamlede tids-integrerede (repræsentative) prøver af suspenderet stof igennem prøvetiden. Rumlige og tidslige variationer i sammensætningen af suspenderet stof blev undersøgt, baseret på prøver fra apparatet. Prøverne blev også anvendt til at genfinde 'fingeraftryk' af de dominerende kildeområder i oplandet.

Artikel 6 undersøger indholdet af tungmetaller og pesticider i bundsediment i 30 små danske vandløb i relation til oplandsforhold og sedimentets egenskaber. Indhold af pesticider var signifikant relateret til oplandsstørrelse, jordtype og hydrologisk regime, mens indholdet af en gruppe tungmetaller (Cr, Cu, Pb, V og Zn) var relateret til bymæssig bebyggelse og jordtype.

Artikel 7 indeholder kvantitative data om transportveje for sediment, P og nitrat i danske landbrugsoplande. Partikulært P udgjorde ca. 62% af total P tabet – som var på 0.52 kg P ha^{-1} - fra de undersøgte oplande gennem en 9 års måleperiode. Størstedelen af tabet af partikulært P menes at stamme fra brinkerosion. Overfladeafstrømning og jorderosion bidrog sandsynligvis med store mængder af partikulært P i våde år med snesmeltning på frossen jord. Vi estimerede, at bidraget fra landbrugsdræn med partikulært P var cirka 6% af oplandstabet fra de lerede oplande.

Artikel 8 er en associeret artikel, som undersøger aflejring af sediment og P på en 5000 m^2 eng i Gjern Å oplandet. Tilbageholdelseseffekten under oversvømmelse af engen varierede fra 3.6-23.9% for sediment og 1.2-6.5% for partikulært P. Tilbageholdelseseffekt blev beregnet som aflejring divideret med transporten i vandløbet. Selv små enge kan altså have en positiv betydning ved at mindske transporten til vandområder, som ligger længere nedstrøms.

Introduction

1. Background

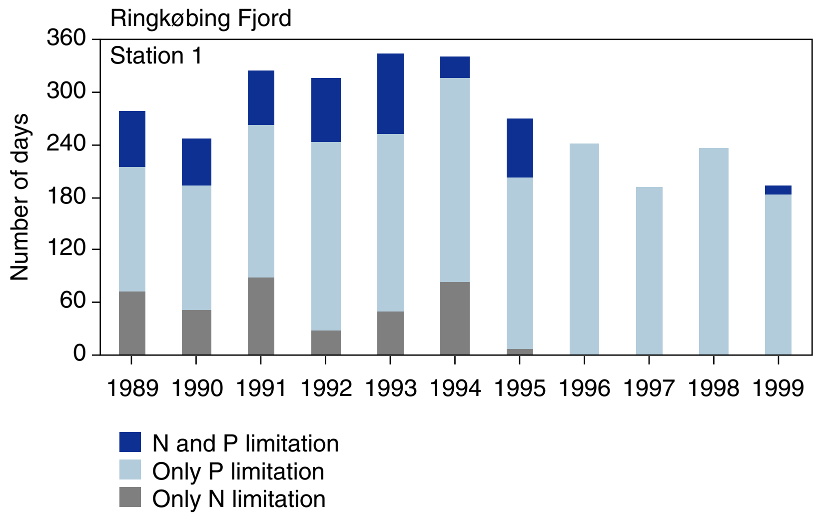
The negative impacts from high concentrations of suspended sediment in streams and rivers have been respected since the first sewage regulations a century ago (Newson, 1994). Industry is capable of producing effluents with a high suspended sediment load. Its content of organic matter, inorganic particles and toxic elements have different and often severe impacts on the aquatic environment ranging from acute toxic impact on organisms to change in benthic community, reduction in diversity, and reduced plant photosynthesis (Hellawell, 1986).

In rural areas where sediment input from sewage is a minor source, other sediment sources and pathways are active (e.g. soil erosion and bank erosion), also with negative impacts on the aquatic environment. Sediment input to streams and rivers has an effect on in-stream biological habitats. For example, spawning grounds can be damaged because their pores become blocked with migrating fine sediment and faunal species composition can be altered if a gravely substratum becomes covered with sand (Madsen, 1995). Sediment-associated pesticides deposited as fine sediment can be toxic to biota, particularly benthic invertebrates, and the assimilated pesticides can be transmitted through the food chain (Larson et al., 1977). Heavy metals associated with fine sediment may also exhibit a similar unwanted influence on biota although they are to a higher extent associated with urban or mining areas than with rural areas.

Part of the phosphorus (P) that is transported in the freshwater environment is associated with fine sediment (Svendsen et al., 1995). The mobilisation and delivery of sediment-associated P to the aquatic environment often have a harmful impact. Eutrophication of lakes and estuaries is largely controlled by phosphorus input from point and diffuse sources in many European countries (European Environment Agency, 1999).

In the Danish lakes and estuaries, phosphorus plays a major negative role in eutrophication by stimulating algae growth. Phosphorus is the limiting factor for algae growth in most of the 120,000 shallow lakes in Denmark. Also for several of the Danish estuaries, phosphorus is today the main limiting factor whereas it was nitrogen 10-15 years ago (Conley et al., 2001) (Figure 1).

Figure 1 Number of days with nitrate and phosphorus limitation for algae growth in the Ringkøbing Fjord estuary from 1989-1999.



From an economic point of view, sediment delivery is associated with undesirable expenses on land and also downstream in the fluvial systems and coastal waters. The transport and delivery from sheet and rill erosion, for instance, often result in reduced topsoil depth, diminished fertility and crop yield. Similarly, the downstream transport of eroded sediment may give rise to problems of stream channel and reservoir sedimentation, and harbour siltation (Walling, 1988). Brown (1948) estimated half a century ago the economic costs due to sediment erosion and input to United States fluvial systems to be 175 million $, which in present day value converts to 1-1.5 billion $. Brown estimated that agricultural areas were responsible for about one third of the total economic cost. Also today, the costs are huge.

Box 1 - Suspended sediment

In stream water, the sediment particles are transported along the bed (bed load) or in suspension (suspended sediment)

Fine particles such as clay and fine silt are transported as suspended sediment. Coarse particles such as sand and pebbles are most often transported as bed load. Considerable variation exists, however, in the particle size of suspended sediment at the global scale. In the Australian Barwon River, for instance, suspended sediment contained more than 80% clay sized particles whereas in the Chinese Huangfy River suspended sediment contained more than 60% sand sized particles (Walling & Moorehead, 1988). Particle size fractions are defined from the size of the aggregates or particles. Suspended sediment mostly consists of aggregates rather than individual particles (Walling, 1988).

Particle size fractions	**Size (mm)**
Pebbles	2 – 20
Sand	0.063 – 2
Silt	0.002 – 0.063
Clay	< 0.002

On a world-wide basis, suspended sediment play a major role in transporting elements such as metals and phosphorus from land to the coastal waters (Table 1). The mobilisation, transport and delivery of suspended sediment and different associated elements to stream and river systems vary according to natural and anthropogenic influences, with some elements such as heavy metals commonly associated with urban areas. Major urban areas are normally located in the downstream sections along main river channels.

Table 1. The role of suspended sediment in the global transport of major elements from land to the oceans by rivers (Walling, 1988, citing other authors).

Element	Sediment-associated transport / Total transport	Typical sediment content
	(%)	(mg g^{-1})
Al	>99	90
Fe	>99	52
Ti	>99	6
Mn	98	1.0
Si	96	280
P	95	1.2
K	87	21
F	80	0.97
B	60	0.07
N	57	1.2
Mg	56	11
Ca	43	25
C	34	20
Sr	32	0.15
Na	28	7.1
S	13	2.2
Cl	1	0.5
Inorganic C	28	10
Organic C	45	10
Organic N	68	1.2
Organic P	94	0.45
Inorganic P	97	0.70

Rural areas often dominate the upstream part of fluvial systems and the stream drainage density is relatively high. Here, the streams are smaller and the topography has a higher slope as compared with downstream fluvial systems. Small streams are, therefore, closer linked to the terrestrial landscape and the risk that eroded sediment from an arable field escapes to the small stream is much greater than further downstream where river valleys form natural barriers between the field and river. Small streams (1st-3rd order streams) are far the most dominant lowland stream type constituting more than half the total length of channel network in Denmark (Table 2). Therefore, special attention needs being directed to the understanding of the dominant transport and delivery processes bringing sediment and sediment-associated substances into small Danish streams.

Table 2 Total length of small, medium, and large Danish streams, as based on the national AIS database.

Small streams (< 2 m wide)	Medium streams (2-8 m wide)	Large streams (< 8 m wide)	All streams
49,400 km	14,700 km	1,600 km	65,700 km

2. Factors influencing diffuse sediment and sediment-associated substance pollution

Pollutants are substances causing damage to 'targets' (Holdgate, 1979) as they are emitted from sources and reach the target through delivery pathways (Figure 2). This definition includes the useful concepts of sources (which have spatial locations), pathways (linking locations), and targets (which also have locations). The 'target' can for instance be the air, water or soil within a region. In this thesis, I will focus on water bodies as targets, and especially on streams.

Figure 2. Diagram showing the Holdgate (1979) concept of pollution. Pollutants reach the target area by transport from a source area.

Pollution and pollution control exhibit the inherent space/time frameworks enjoyed by geographers and pollution has patterns and processes which are open to geographical treatment – it can be mapped and modelled (Newson 1991). Much of the scientific work is carried out by focussing on separate sources or processes (Newson, 1994). By focussing for instance, on pollution via subsurface runoff from a specific agricultural area, our knowledge on the transport process involved can be increased and possibly also incorporated into our understanding and models dealing with the broader theme of diffuse pollution from rural areas.

Diffuse pollution, often referred to as non-point pollution, refers exclusively to pollution from diffuse sources. Point sources such as urban areas and industry are not included. In practical terms this implies that sampling and investigations on diffuse pollution are not as easily undertaken as when dealing with point sources. Diffuse pollution to the aquatic environment is governed by large spatial and temporal extent and therefore difficult to assess and manage (Kronvang, 1996). Diffuse sources consist in contrast to point sources of extended geographical areas. Major types of diffuse sources are agricultural land, forested land, natural land, mining areas and atmospheric deposition. All other sources of pollution can preferably be regarded as point sources. Sources such as freshwater fish farms and scattered dwellings are according to the above definition regarded as point sources.

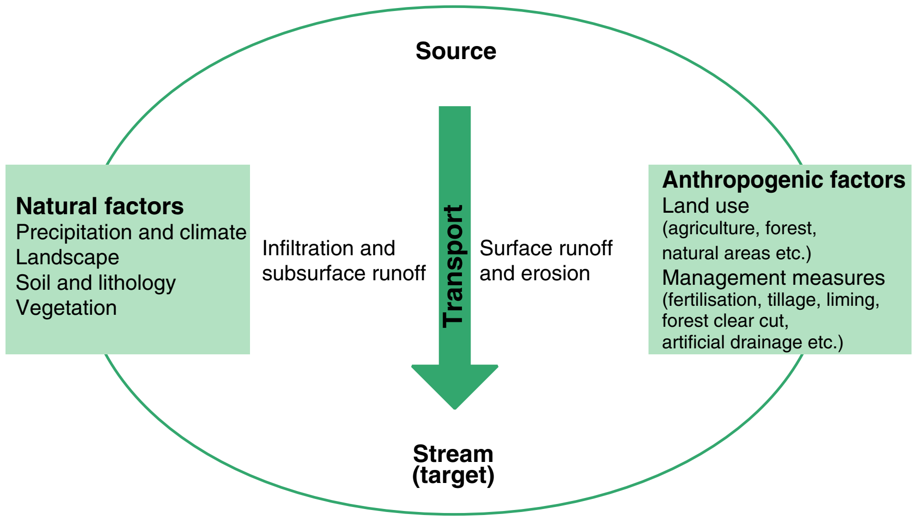
The factors influencing diffuse pollution can according to the definition on pollution be divided into two groups, i.e. those influencing:

the source and its content of pollutants
the transport from source to target

In practical terms a number of factors convey an influence both on the source and the transport. For instance, climate has a major influence on transport (via precipitation regulating the runoff) and also on the source content of elements, for instance by affecting the rate of soil weathering and organic matter production and break down.

The pollutant content of a source is often a product of both natural and anthropogenic influences (Figure 3). Agriculture and forest being diffuse sources are under influence from both natural and anthropogenic factors. Their metal and phosphorus content, for example, is the product of the natural metal and P content of parent soil material but also of anthropogenic factors such as fertilisation of soils with chemical fertiliser and manure. In many countries, the impacts from anthropogenic factors are more important than those from natural factors.

Figure 3 Diagram showing the natural and anthropogenic factors that influence source and transport of sediment-associated substances.



At the source, transport along the pathway starts with mobilisation of the sediment particles and aggregates. This mobilisation depends highly on the shear-strength of the sediment in place, which is mainly controlled by its particle size distribution and water content (Leopold et al., 1963). Several natural factors influence the mobilisation, transport and delivery to the stream (Figure 3). The most important natural influence on transport is climate, with precipitation as the main force driving the transport under the principle of gravity. Precipitation creates water runoff from the source area, either by infiltration of the soil and subsequent percolation or by surface runoff. Either way, sediment and sediment-associated substances may be entrained with the water and has the potential to be transported to the stream.

Anthropogenic factors also have great implications on the transport linkage between source and target. Drainage schemes for improving agricultural land has for instance a major impact on the hydrology of source areas by:

i) Decreasing evaporation and increasing surface runoff as opposed to forests and other permanently vegetated natural land;

ii) Detailed draining with tile drains and ditches which creates new pathways and better linkage between source and targets;

iii) Straightening and channelisation of streams and rivers improve drainage from adjacent agricultural areas.

Hence, agriculture is a main anthropogenic factor affecting transport of water, sediment and sediment-associated substances to streams. Agriculture has an influence on surface runoff, percolation, subsurface drainage and the connectivity from the source area to the stream, thereby increasing the potential of pollutants to enter the stream. Also stream bank erosion may be affected by agriculture.

The transport linkage between source and target changes over time, and a linkage that is normally not in use can during seldom events be very active. Along the transport linkages, numerous conditions may also interfere and possibly stop - or retain part of - the sediment and sediment-associated substances before they reach the stream. Soil erosion from surface runoff is an example of a process that in many cases only appear a few days a year. Sometimes the sediment enters the stream - but at other times it does not, possibly due to deposition in buffer zones along the stream.

Box 2 - Particulate and dissolved phases

Interactions occur between the particulate and dissolved phase of any pollutant on its way from source to target. At the target, the dissolved phase is generally regarded as having a higher bioavailability than the particulate phase. Nearly all the dissolved P, for instance, especially the dissolved reactive P is regarded bioavailable (Ekholm, 1991; 1994). On the other hand only a minor part of particulate P (less than half) is regarded bioavailable, or potential bioavailable. Particulate P is therefore, as compared to dissolved P, less harmful on the aquatic environment. For some lakes including most Danish lakes, however, particulate P that was deposited on the lake bottom years ago is today released from the bottom sediment to the lake water as dissolved P (Conley et al., 2001). Particulate P may thereby end up being more bioavailable than assumed from start.

For some purposes, it is useful to distinguish between different soil - or sediment - compartments within a source or catchment area. A differentiation between topsoil and subsoil compartments makes sense within in the source-transport-target concept. The topsoil compartment may in some cases (depending on the local conditions and the elements studied) be simplified as being the only source, or in other cases topsoil and subsoil compartments can be differentiated by their different content of elements.

In Denmark, the content of phosphorus is higher in agricultural soils as compared with forest soils, and also higher in topsoils than in subsoils (Table 3). The same applies for a range of heavy metals (Table 4). The delivery of sediment-associated substances to streams is generally higher from agricultural sources than from forest or natural sources - as both the source and the transport is higher from agriculture. When a source area is both high in its the element content and there also exists a high transport linkage from source to stream, the area is a Critical Source Area (Figure 4).

Table 3 Total P content in agricultural soils and soils under deciduous forest, Denmark 1986 (From Rubæk et al., 2000).

---------- Agriculture ----------		-------- - Deciduous forest ------------	
0-25 cm	25-50 cm	0-25 cm	25-50 cm
527	381	298	232

Table 4 Typical Heavy metal values in rocks, soils and fertiliser (Danish values from Jensen, 2000 and world wide numbers from Alloway 1995*)

	As	Cd	Cu	Cr	Hg	Ni	Pb	Zn
	----------------------------	$\mu g\ g^{-1}$	--					
Sandstone*	0,25	0,05	30	35	0,29	9	10	30
Granite*	0,04	0,09	13	4	0,08	0,5	24	52
Shales and clays*	13	0,22	39	90	0,18	68	23	120
Earth's crust*	0,07	0,1	50	100	0,05	80	14	75
Agricultural soils, Britain*	-	0,7	18	39	-	23	40	82
Agricultural soils, USA*	-	0,2	19	-	-	18	11	53
Agricultural soils, Denmark	3,6	0,18	7,8	10,7	0,04	5,7	11,3	29,1
Reference soils, Denmark	1,3	0,07	0,9	3,8	0,01	1,5	8,7	7,7
Forest soils, Denmark	2,3	0,09	2,8	7,0	0,04	2,9	12,1	18,9
Sewage sludge, Denmark	5,1	1,5	298	34	1,4	26	72	878
Animal manure, Denmark	0,3	0,66	636	3,6	<0,1	10	2,0	897
Animal manure*	3-25	0,1-0,8	2-172	1,1-55	0,01-0,36	2,1-30	1,1-27	15-566
Nitrate fertilisers*	2-120	0,05-8,5	-	3,2-19	0,3-2,9	7-34	2-27	1-42
Phosphate fertilisers*	2-1200	0,1-170	1-300	66-245	0,01-1,2	7-38	7-225	50-1450

The concept of Critical Source Areas opens up different possibilities related to management and remedial strategies. Critical source area management integrates all sources and all modes of transport, and focuses on either just one element or on a range of different elements. For phosphorus, the critical source area management approach is today an integrated management aspect in North America and is becoming more applied also in other parts of the world. In Denmark, it is an integrated part of management strategies presently considered in relation to the implementation of a third Danish Plan for the Aquatic Environment. Recently, McDowell et al. (2002) suggested the application of remedial measures in parts of central Pennsylvania, USA, where minimising the P export should focus on critical source areas, while minimising the nitrogen (N) export should be source based, concentrating on more efficient use of N by crops. Especially in relation to sediment-associated pollutants, the critical source area concept is useful as it offers an opportunity to integrate different aspects related to the different sources and different transport pathways.

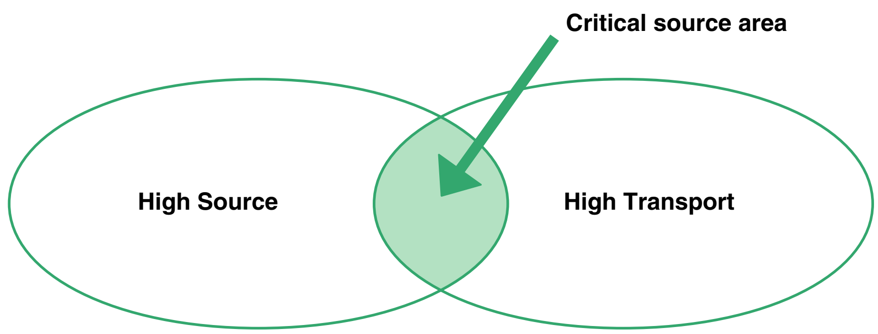


Figure 4 Critical source areas have a high pollutant content and a high risk for transport.

3. Delivery pathways

Sediment and sediment-associated substances are transported from source area to stream along a number of different delivery pathways (Figure 5) Along all pathways there is large spatial and temporal variation in the transport. In the manuscripts of this thesis, the delivery pathways of surface runoff (MS 2 and 3) , subsurface runoff (MS 1), and bank erosion (MS 3 and 4) are investigated. Different aspects of deposition are also examined (MS 5, 6 and 8), and different land uses and pathways are discussed at the catchment level (MS 6 and 7). A quantitative understanding of the different delivery pathways and the processes involved in mobilisation and transport is a helpful basis for discussing management and remedial measures aiming at minimising diffuse pollution to streams (targets) from the different pathways.

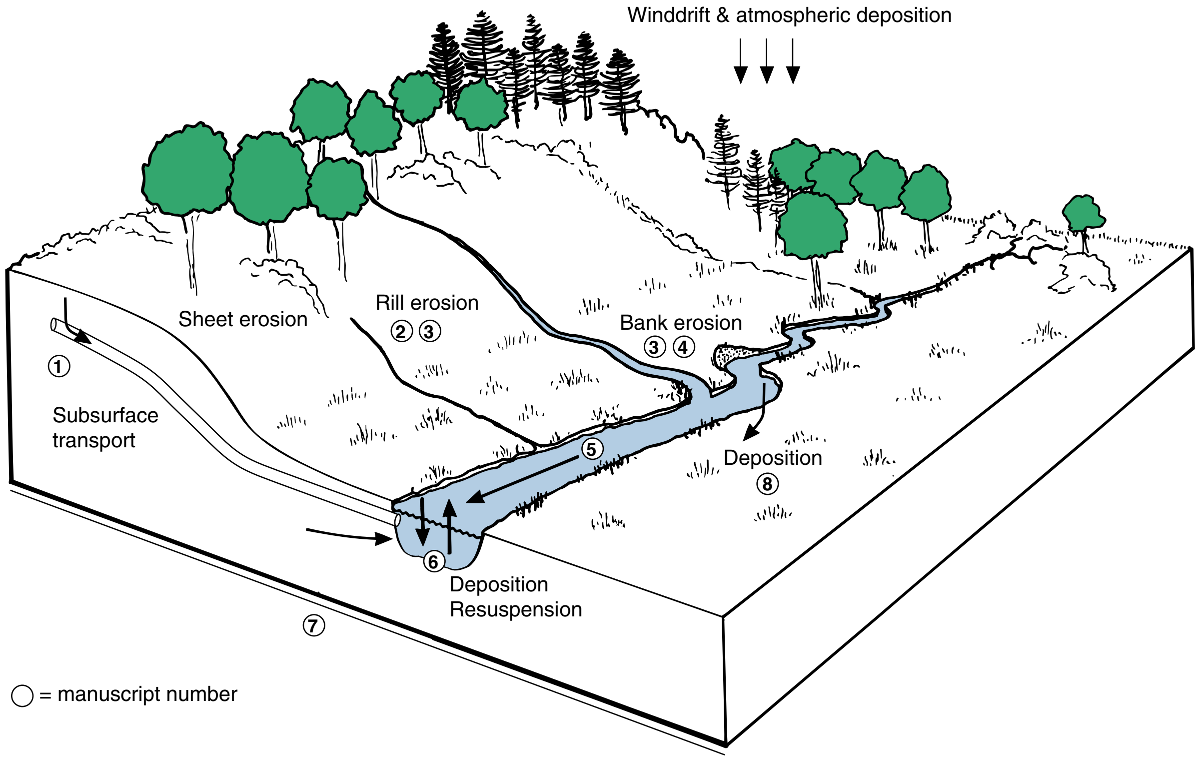


Figure 5 Delivery pathways for sediment and sediment associated substances of diffuse sources.

3.1. Soil erosion

Processes

The process of soil erosion by water can be described in two stages, i.e. detachment and transport. Detachment results from raindrop impact or from overland water flow, in both cases with shear stress acting on the soil. Raindrop impact causes a local intense stress and similarly, overland water flow causes a stress that may be high enough to exceed the critical shear stress, and the sediment is detached. In Denmark, detachment occurs both by raindrop and overland flow. Especially when winter rain melts snow on the ground, overland flow and sediment transport can be high.

There are three main types of soil erosion processes:

1. Sheet erosion
2. Rill erosion
3. Gully erosion

Sheet erosion is evenly distributed across a slope and creates a uniform detachment and transport of sediment particles and aggregates from the soil surface (Hairsine & Rose, 1992). Rill erosion occurs when water in overland flow moves along preferential pathways forming an easily recognisable channel (Rose, 1993). Rill formation is controlled by the cohesive strength of the soil and the shear forces exerted on the soil. Sediment that originates from sheet erosion (interrill sources) is often transported further down the hill slopes in rills. The contribution from sheet erosion is variable. Some studies have indicated that soil loss from rill erosion is often higher than that from inter-rill erosion (Govers & Poesen, 1988).

Gully erosion, in contrast to rill erosion, forms channels that are too deep to be wiped out by cultivation (Rose, 1993). Gully erosion differs from sheet and rill erosion in that raindrop impact is not an important factor in its formation, and that subsoil sediment is a major part of the sediment being transported.

The three types of erosion processes may occur in isolation from each other but in many cases they act in combination. Rill and gully erosion will occur with increasingly extent as the length of the slope increases. Hence, the dominant erosion process would be expected to follow a downslope sequence of detachment-sheet-rill-gully (Loch & Silburn, 1996).

Soil erosion rates

The rapid erosion of soil has been a problem since man began cultivating the land. Especially semi-arid and semi-humid areas of the world are vulnerable to soil erosion, as can be seen in China, the western USA, the former Central USSR, the Mediterranean countries, and India (Table 5). Northwest European countries such as Belgium, the UK, and Denmark generally have low erosion rates due to the climate. The mean annual erosion rate on Danish arable slope units amounted to 0.05 kg m^{-2} yr^{-1} as estimated based on annual average rill erosion volumes from 88 selected slope units over a 6-year period (MS 2).

Table 5 Rates of erosion in selected countries (kg m^{-2} yr^{-1}) according to Morgan (1986) citing different authors.

	Natural	Cultivated	Bare soil
China	< 0.20	15 - 20	28 - 36
USA	0.003 - 0.30	0.50 - 17	0.4 - 9.0
Ivory Coast	0.003 - 0.02	0.01 - 9.0	1.0 - 75
Nigeria	0.05 - 0.10	0.01 - 3.5	0.3 - 15
India	0.05 - 0.10	0.03 - 2.0	1.0 - 2.0
Belgium	0.01 - 0.05	0.30 - 3.0	0.70 - 8.2
UK	0.01 - 0.05	0.01 - 0.30	1.0 - 4.5

Within any area there is considerable variation in erosion rates but if the rates are grouped according to vegetation (natural vegetation, cultivated land, bare land), each group follow a broadly similar pattern of global variation (Table 5). In passing from a global scale to a national or regional scale, gradual changes occur in the dominant factors influencing soil erosion. Climate is dominant at the global scale but at smaller scales where climate is fairly uniform, other factors such as slope, slope length, soil type, crop type and tillage become important for soil erosion rates.

Factors influencing soil erosion

Important factors which influence the rate of soil erosion are precipitation, runoff, soil, slope, slope length, vegetation cover, tillage and management practice (Morgan, 1986). These and other related factors can be grouped according to energy, resistance and protection (Figure 6). The energy related factors include the potential ability of precipitation and runoff to cause erosion. Also those factors that directly affect the power of the eroding agent (slope and length) are included amongst the energy-related factors. The main factor related to resistance is the erodibility of the sediment, whereas the protection related factors mainly are related to vegetation cover.

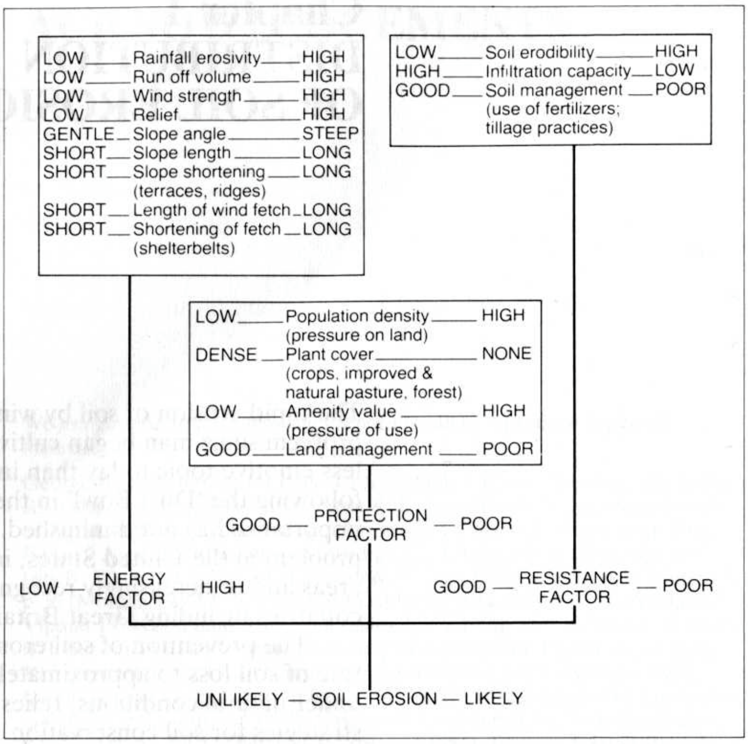


Figure 6 Factors affecting soil erosion by water and wind (Morgan, 1986).

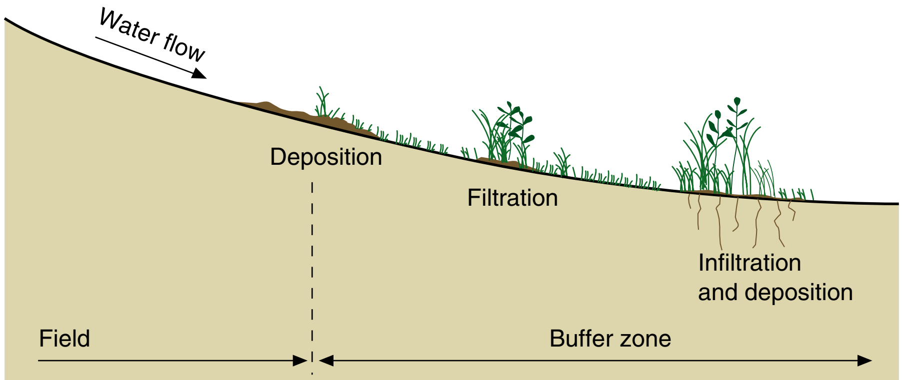
In agriculture, measures for reducing soil erosion are based on the role of vegetation. Because of differences in their density and morphology, plants differ in their effectiveness in protecting the soil from erosion. Row crops are generally the least protective and give rise to more serious erosion problems. Particular problems are associated with maize. The inclusion of grasses and legumes in a rotation can reduce erosion and in addition increase the yield of the main crop, for instance maize (Morgan, 1986, citing other authors). On Danish slope units, the lowest erosion rates were observed from grass and highest erosion rates from winter cereals that similarly to maize is a row crop. Soil erosion rates were lower on bare, ploughed soil as compared to winter cereals due to higher surface roughness being able at retaining more water and reducing surface runoff (MS 2). It has been suggested that the change from spring sown cereals to winter cereals in Danish agriculture included in the first Plan for the Aquatic Environment to reduce nitrogen leaching had a negative effect on phosphorus losses due to an increase in soil erosion.

Deposition of eroded sediment in buffer zones

Processes of deposition in the stream channel, in lakes and reservoirs and along the stream by flooding of river valleys reduce sediment delivery to downstream water bodies, such as estuaries. It acts on sediment from all pathways, not only that from soil erosion. Additionally, sediment delivery to the stream via surface runoff can be reduced by downhill deposition processes either on the field or in buffer zones along the stream channel. Permanently vegetated buffer zones along the stream can, therefore, increase sediment deposition. Buffer zones (narrow, often referred to as buffer strips) are bands of permanent vegetation, situated between source areas and streams or other water bodies. They are generally considered to be sediment sinks in catchments disturbed by human activity such as agriculture.

Historically, sediment control aims at reducing erosion at the source itself. Buffer zones, on the other hand, are meant to store sediment and other elements once they have left the source area. Buffer zones are nowadays an essential element of all diffuse pollution control programmes (e.g. Karr & Schlosser, 1978). The major sediment-associated removal mechanisms associated with buffer zones involve change in flow hydraulics which enhance the opportunity for sediment deposition, filtration of sediment by vegetation, and infiltration into the soil (Dillaha & Inamdar, 1997) (Figure 7).

Figure 7 Sketch illustrating three sediment removal mechanisms of buffer zones, i.e. deposition, filtration and infiltration.

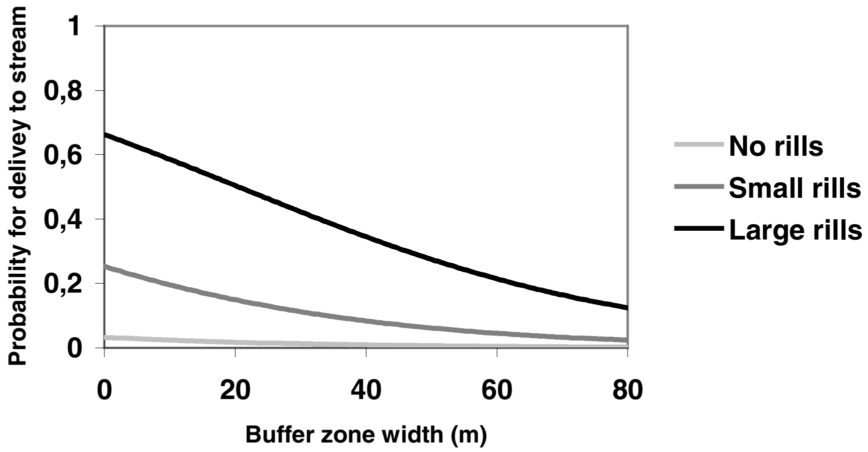


Infiltration is a significant removal mechanism affecting buffer zone performance. Infiltration is important since the finer sediment particles enter the soil profile along with infiltrating water and because it decreases surface runoff, thus reducing sediment transport capacity. Infiltration is probably most significant for clay sized particles. Buffer zones also remove sediment through the processes of deposition and filtration. Deposition is most significant for silt and larger sized particles and aggregates, and filtration only for the largest particles and aggregates (Dillaha & Inamdar, 1997).

The vegetation of buffer zones facilitates the retaining from overland storm water of suspended sediment and sediment-associated substances (Mithsch et al., 1979; Peterjohn & Correl, 1984). Grass or dense herbaceous vegetation is more effective at trapping particles from overland storm flow as compared with forest vegetation (Osborne & Kovacic, 1993; Parsons et al., 1994). For grass buffers, the fraction of sediment trapped in vegetation has been related to factors such as flow velocity, flow depth, particle size, and buffer width (e.g. Barfield et al. 1979). The majority of sediment deposition often occurs just upslope of the buffer and within the first metre of the buffer (Figure 7). The trapping efficiency can be high as long as the water flow entering the buffer zone is not concentrated in preferential pathways and vegetation is not submerged.

Buffer zone width – as measured from field edge to stream bank – has a clear impact on sediment trapping efficiency and sediment delivery to the stream. Many factors apart from buffer zone width, however, also play a role and therefore the influence from buffer zone width may vary considerable from place to place and from time to time. Magette et al. (1989) for example measured an increase from 52 - 75% in sediment trapping efficiency by doubling the width of a grass buffer zone from 4.6 - 9.2 metres. Kronvang et al. (2003, submitted) did not measure sediment trapping efficiency on individual fields but established instead an overall relation between buffer zone width and 'sediment delivery' to the stream, based on 394 observations on different arable fields and buffer zones. 'Sediment delivery' was defined as less than 100% trapping efficiency. The buffer zones were grass or herbaceous vegetated. The vegetation was short in most grass buffers due to intense cattle grassing. By increasing the buffer zone width from 2 - 20 metres, cases with sediment delivery decreased from 25 - 15%, or from 65 - 50%, depending on the severity of soil erosion on the field (Figure 8).

Figure 8 Probability for delivery of sediment from rill erosion on sloping Danish fields to small streams (Kronvang et al., 2003 submitted).



Sediment-associated substances are transported with fine particles. These are normally transported for the longest distances. Therefore, 100% sediment trapping efficiency is desirable in order to trap the sediment-associated substances. A 90% sediment trapping efficiency may for instance imply that as much as 60-100% of the fine particles and sediment-associated substances are not trapped but enter the stream.

Some investigations have reported that buffer zones are sometimes a source of sediment and sediment-associated substances instead of being a sinks (e.g. Smith, 1992). In addition, even though the buffer zone is a net sediment sink, streambank erosion may increase in order to satisfy the available sediment transport capacity of the stream water (Baker, 1992). Delivery of fine sediment to the stream by future bank erosion may possibly also increase, as fine sediments from soil erosion can have accumulated near the streambank and then by bank erosion later enter the stream. Harvesting of buffer zones is a mean to remove nutrients and increase vegetation density at the ground level, thereby enhancing the sediment trapping efficiency (Dillaha & Inamdar, 1997). Several authors have also shown that buffer zones can be a sink for sediment and sediment-associated P but a net source of dissolved P due to freezing of plant residues (release of P from organic tissue) and possible desorption of P from highly P-enriched soil material to surface water runoff through the buffer zone (Dillaha et al., 1989; Uusi-Kämppä & Yläranta 1992, 1996).

3.2. Subsurface transport

The movement of particles and particle-associated substances such as phosphorus, heavy metals and pesticides through the soil to groundwater and tile drainage systems has traditionally received little attention as surface erosion is generally regarded as being more important (e.g. Bengtson et al., 1984). Especially when artificial subsurface drainage systems are installed - hereby offering a direct linkage to the stream from source areas located some distance from the stream – it may play an important part in the transport of sediment and sediment-associated substances to streams.

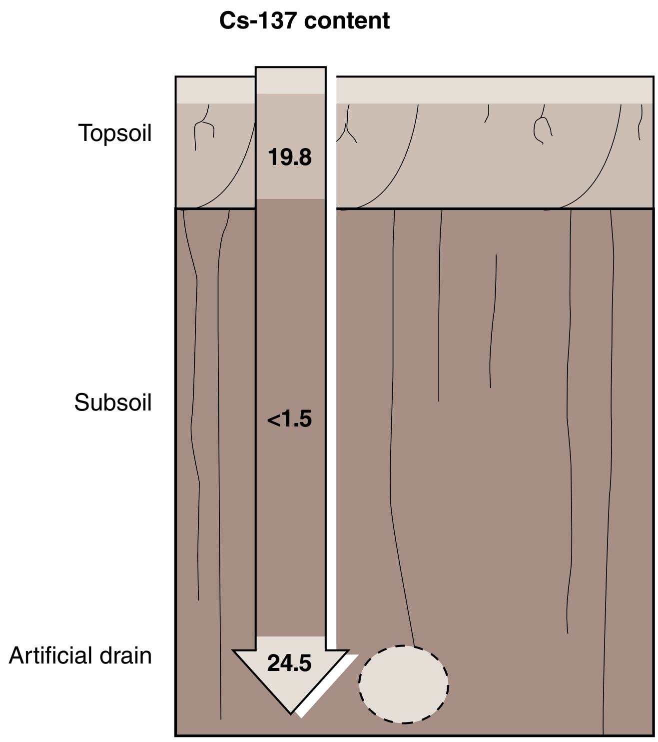
When no artificial subsurface drainage system is installed on the other hand, the most important effect of the natural subsurface flow is to bring about the accumulation of moisture in the soil in the foothill and concave parts of the landscape, thereby enhancing the risk of overland flow and surface soil erosion (Zaslavsky & Sinai, 1981). Roose (1970) showed in a study from Senegal that the natural subsurface transport only constituted about 1% of the total sediment eroded from a hillside. This sediment was mainly in the form of fine particles and minerals in ionic solution.

Several studies have indicated that the loss of sediment-associated substances via artificial subsurface drainage systems is important (Bottcher et al., 1985; Ulén et al., 1991; Grant et al. 1996; Øygarden et al., 1996). The presence of macropores can enhance the mobility of particles because adsorption, sedimentation and sieving are less pronounced in large pores. A few studies have dealt with particle transport on undisturbed structured soils either based on soil column experiments or on field plot experiments (Pilgrim and Huff, 1983; Smith et al. 1985; McKay et al., 1993).

When applying the term 'macropores', Pilgrim & Huff (1983) did not specify what kind of large structures, fractures or pores they had in mind. They mentioned cracks, and root and animal wholes as possible kinds of macropores. They emphasised that a fast response to rainfall and high sediment concentrations can be observed in 'macropore' transport, and that macropore transport is probable to occur in many regions. McKay et al. (1993) later emphasised the role of natural fractures in a Canadian subsurface transport study, whereas for the Danish study area investigated in MS 1, Jacobsen et al. (1997) emphasised the role of worm wholes. Macropore flow has also been reported for similar Danish soils by Engesgaard & Jensen (1991) and Jørgensen & Fredericia (1992).

Some have used the term 'internal erosion' when referring to macropores transport (e.g. Ulén et al., 1991). This term may bear with it an indication that the macropores themselves and the soil below ground surface are eroded. It may, however, not be the case. The sediment may well derive from the surface where it was entrained by raindrop impact. Several studies have pointed to evidence showing that the sediment is derived from the topsoil (e.g. Pilgrim & Huff, 1983; MS 1). The evidence includes high levels of Cs-137 in the subsurface drainage sediment (Figure 9). Cs-137 was not present in the environment before the testing of nuclear weapons in the late 1950s and is therefore found in higher concentrations in topsoil compartments as compared with subsoil compartments.

Figure 9 Cs-137 content of topsoil and subsoil sediment (<20µm) and of sediment in the tile drainage water (MS 1).



The median grain size of sediment in macropores and artificial drainage systems is relatively fine, often below 5-8 µm (e.g. Pilgrim & Huff, 1983; MS 1). Relatively high amounts of sediment-associated substances are therefore also often transported with it. In The Danish Gelbæk catchment, for example, subsurface drainage accounted for 10-15% of suspended sediment, and 10-18% of particulate P from the total catchment loss (Kronvang et al., 1997).

Several investigations apart from the one by Pilgrim & Huff (1983) have showed a fast response to rainfall and high sediment concentrations in subsurface runoff from macropores. Maximum suspended sediment concentrations were observed at the very start of storm flow events, with the peak in suspended sediment preceding the peak in runoff at the plots and the tile-drained catchment of MS 1 (Figure 10). This indicates an exhaustion effect such as has been described for streams during storm events (Walling, 1974; Kronvang, 1990). Due to the very high temporal variation in suspended sediment concentration, the choice of sampling strategy has great importance for the accuracy of the estimation of the suspended sediment loss by tile drainage water. This also applies for phosphorus as was illustrated for four Danish tile-drained catchments (Grant et al., 1996). Here, an infrequent sampling consisting of 12 samples per year underestimated the annual P losses by 26-130% when comparing with a frequent automatic sampling (Table 6).

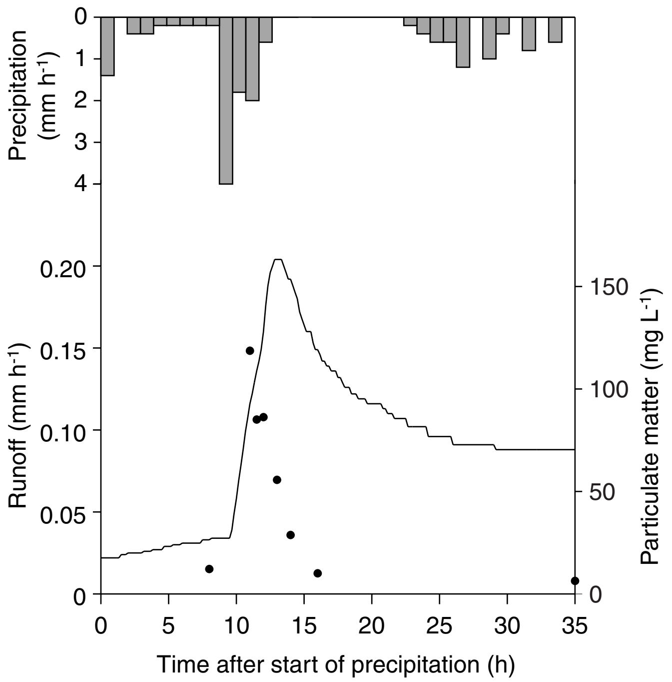


Figure 10 Precipitation, runoff and particulate matter (i.e. suspended sediment) during a storm event in a tile-drained catchment (MS 1).

Table 6 The accuracy (bias) of annual total phosphorus transport estimated by infrequent sampling (T_i) and frequent sampling (T_f) for four tile drained areas (Grant et al., 1996)

Tile drainage catchment number	T_i	T_f	Bias 100*((T_i-T_f)/T_i)
Catchment G1	3,965 g	8,343 g	110%
Catchment G2	188 g	433 g	-130%
Catchment L1	457 g	759 g	-66%
Catchment L2	125 g	159 g	-26%

3.3. Bank erosion

Bank erosion rates

Bank erosion can contribute significant amounts of sediment to fluvial systems and commonly accounts for more than 50% of catchment sediment export (Coldwell, 1957; Carson et al., 1973). Also Danish investigations indicate a significant contribution from bank erosion (Hasholt, 1988; Laubel et al., 1999; MS 4). Since the late 1950s, bank erosion has commonly been measured by use of erosion pins (Lawler, 1993) and a range of different measuring techniques can been applied (Lawler et al., 1997). Direct measurement of stream bank erosion over shorter time-scales requires much fieldwork and is subject to difficulties because of considerable spatial variation (Lawler, 1993).

World-wide, bank erosion rates have been measured in the range 0,01 – 1000 m yr^{-1} (Hooke, 1980; Lawler et al., 1997). Part of the reason that such relatively high rates have been measured is that most studies deal with sites of maximum bank erosion (e.g. outside meanders) and not with sites of representative erosion rates. Increase in catchment size has been related to increasing bank erosion rate (Hooke, 1980; Lawler et al., 1997; Bull, 1997). Hooke (1980) for instance, explained 53% of spatial variation in the bank erosion rate alone by catchment size. Bank erosion rates in small Danish lowland streams were measured in the range 0,00 – 0,16 m yr^{-1} (MS 4), and those of small streams within the River Severn catchment in Wales were in a comparable range, i.e. 0,01 – 0,06 m yr^{-1} (Bull, 1997).

The rate, periodicity and distribution of bank erosion are highly variable because many controlling factors are involved (Knighton, 1998).

Bank erosion processes

Bank erosion processes may be grouped into three categories: weakening processes, direct fluid entrainment, and mass failure (Lawler, 1992). Several recent studies have emphasised the interplay between weakening, fluvial erosion and mass failure in producing bank erosion and bank retreat (e.g. Hooke, 1979; Lawler, 1992). Relative dominance of fluvial erosion and weakening processes to bank failure has been described for streams of small size (Lawler 1992, 1995; MS 4). At larger streams further downstream in the fluvial system Lawler (1992, 1995) found a combination of processes, including frequent bank failures.

The seven main factors influencing the three erosion processes are (Knighton, 1998):

water flow properties
bank material composition
climate
subsurface conditions
channel geometry
biology
man-induced factors

Few studies have modelled spatial variation in bank erosion based on empirical data from a high number of banks and with many factors. Most of these have used banks of apparent maximum erosion. At 15 sites, Hooke (1980) for instance, found that erosion was controlled by seven variables (catchment size; silt-clay content; presence of gravel layer; width-depth ratio; radius of curvature; slope; bank height) yielding a 72% explanation of spatial variation in bank erosion rate. Others investigated bank erosion within single catchments. Knighton (1973) for example, emphasised variation and magnitude of flows in creating bank erosion. Lawler (1992, 1995) put forward a conceptual model describing downstream changes between the three main erosion process types as a response to change in stream power and bank

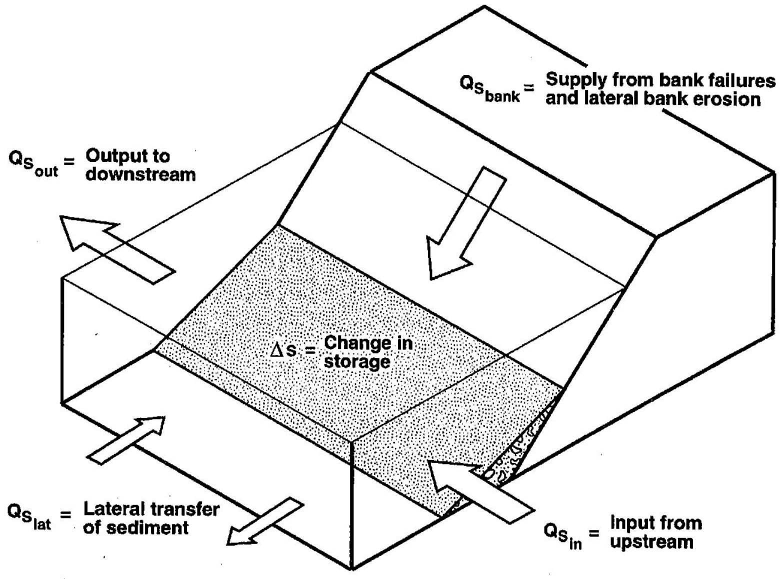
geometry and composition. In MS 4, four site-specific factors were combined in an empirical regression model for smaller Danish streams yielding a 55% explanation on the observed variation in bank erosion rate: bank angle, overhanging bank, total vegetation cover, and stream power.
The four site-specific characteristics included in the empirical model of MS 4 are directly associated with three out of seven main factors above (Knighton, 1998) influencing bank erosion, that is: flow, channel geometry and biology. Bank erosion is generally related to flow conditions, either expressed as near bank velocity (Odgaard, 1987) or as stream power (Nanson & Hickin, 1986). In MS 4, channel geometry primarily influenced bank erosion via bank angle and overhanging bank. Bank angle is important to bank stability (Lawler et al., 1997). Twidale (1964) found the highest erosion at the steepest part of banks. Overhanging banks are banks that have been undercut - and the description 'overhanging bank' therefore supplements the bank angle measure as a description of the maximum angle found at a bank. Bank vegetation affects practically all aspects of bank erosion, influencing near-bank flow conditions, bank erodibility and stability. The effect is probably greatest in small rivers (Lawler et al., 1997).

A classification system ('the Level III stream reach condition assessment') was developed by Rosgen (1996) as a tool for obtaining a quantitative basis for comparing bank erosion for similar types of streams. The system includes determination of 'bank erosion potential ratings' (BEP) and 'near bank stress estimates' (NBS), representing bank physical, vegetative and hydraulic factors. Rosgen (1996) found r^2 values of 0.93 and 0.87 in two study areas when relating integrated BEP and NBS estimates to log-transformed bank erosion rate. Harmel et al. (1999) independently tested the classification system on 36 banks and found that integrated BEP ratings and NBS estimates were poorly related to bank erosion rate. Separately though, the BEP ratings performed relatively well (r^2 = 0.16; $p<0.05$) in predicting bank erosion rate.

The temporal variations in bank erosion rates are considerable, with erosional events matching the temporal fluctuations in bank stresses applied by the controlling factors. An example from River Severn, UK points the importance of mass-instability rather than excess fluid shear stress, as maxiumum bank erosion took place 43 hours after discharge peak (Lawler & Leeks, 1992).

The concept of 'basal endpoint control' (Carson & Kirkby, 1972; Richards & Lorriman, 1987) is useful when discussing bank erosion processes, bank failure and fluvial erosion in relation to longstream sediment transport and sediment removal (Figure 11). Sediment may be supplied to the basal area of the bank (i.e. the lower bank) accordingly, partly from the upper bank - especially by bank failure - and partly from upstream sediment supply. Removal of failed material from the basal area depends on flow entrainment. When there is a balance between sediment supply and sediment removal, bank height and bank angle remain unchanged. If net accumulation occurs at the basal area, the result is a decrease in bank angle and height, and an increase in bank stability. If 'scour condition' prevails, however, all sediment is removed and the bed and lower bank are eroded, thereby leading to an increase in bank angle and bank height, and the bank stability decreases. Scour condition generally prevailed at the Danish banks investigated in MS 4.

Figure 11 Diagram illustrating sediment transport to and from stream bank basal zones (Thorne & Osman, 1988).

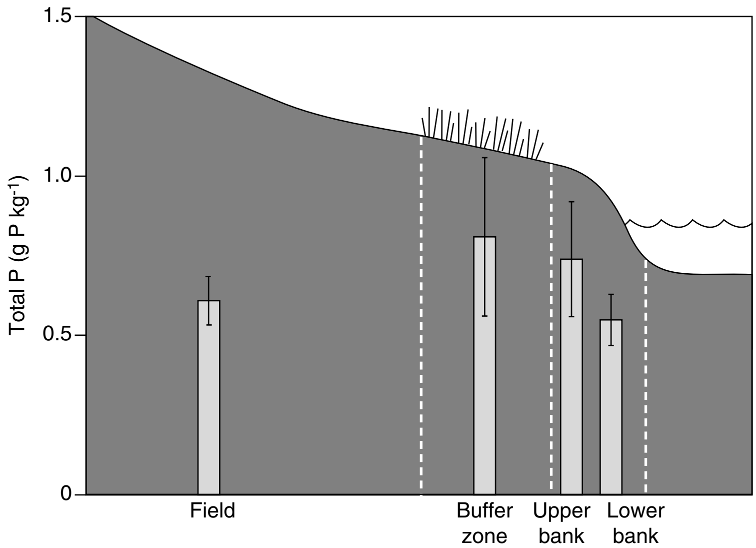


$$\Delta s = Qs_{in} - Qs_{out} \pm Qs_{lat} + Qs_{bank}$$

Sediment-associated substances in stream banks

The mean P content was found to be high (0.64 g P kg^{-1}) in small Danish stream banks (MS 4). Measurements of the P content of the different sediment compartments on arable land showed that the P content of the banks was similar to that of the field topsoil, which was unexpectedly high (Figure 12). Hence the bank material is a potential major contributor of P to the streams. The buffer zone tended to be even higher in P (Mean: 0.79 g total P kg^{-1}; 38 mg Olsen-P kg^{-1}) than the other compartments. The soil material from Danish arable buffer zone has the potential to be moved into the stream by future bank erosion and the buffer zone therefore constitutes a significant potential source for P delivery to the stream.

Figure 12 Mean total P content in field topsoil, buffer zone topsoil, and in upper and lower banks along 26 Danish stream reaches. Lower and upper 95% confidence limits of the mean are also indicated (MS 4).



Bank erosion was estimated to have been a major sediment and P source, probably contributing 40-70% of suspended sediment and 15-40% of total P during the Danish stream bank study (MS 4). In a number of other studies, bank erosion has been found to be a similarly important contributor to sediment or P transport (Carson et al., 1973; Svendsen et al. 1995; Laubel et al. 1999).

In some investigations, the differentiation between topsoil and subsoil compartments has been extended to apply also for the stream banks; some have simplified the entire bank face as being subsoil compartment (e.g. Owens et al., 1999). Whether such a simplification can be justified for the banks in question, rely on the local conditions. In MS-4, the investigated banks resembled more to topsoil than suboil compartment, and their content of phosphorus was at the same level as that on the fields along the streams (Figure 12). Other elements such as heavy metals may possibly also be at a similar level to that of the agricultural fields but this was, however, not investigated.

When discussing means to reduce bank erosion rates, it is not often in order to reduce the input of heavy metals to the steams and rivers. This was, however, the case for the Carson River catchment, USA. Here, the historic processing of metal ores and the substantial release of mercury to the valley floor has meant that the streambanks nowadays are exceptional high in mercury, i.e. 0.1-164 $\mu g\ g^{-1}$ (Miller et al. 1998).

3.4. Atmospheric deposition and wind drift

The atmospheric deposition of fine sediment and sediment-associated substances has for large water bodies and in the oceans a significant size. The direct deposition onto streams is negligible, however, as compared to other delivery pathway. The atmospheric depositions of sediment-associated substances such as phosphorus and heavy metals are larger over land than over the ocean as emissions are land based. For heavy metals, atmospheric deposition on land is often an important source of diffuse pollution. A major part of it is anthropogenic. Surface waters eventually receive a major part of atmospheric emissions and deposition of heavy metals via other delivery pathways (Foster & Charlesworth, 1996)

Wind is the basic factor starting the airborne drift of particles, and if the weather has been dry up to a storm the drift of particles is increased (Kuhlman, 1986). A well-developed soil structure counteracts wind drift and also particle size distribution has importance. The soil is considered vulnerable to wind drift if more than two thirds of the particles are above 0.84 mm (Dormaar, 1987).

The delivery by wind drift of sediment and sediment-associated substances to streams is relatively low in the long-term. During individual heavy windstorms, however, the delivery may be very high - especially if the soil is uncovered by vegetation. Wind drift may remove large amounts of fine sediment and sediment-associated substances from uncovered soils and the delivery to large water bodies and the ocean can be significant. The fine sediment and its sediment-associated substances are relatively easily wind borne and can be transported over large distances (Munkholm & Sibbesen, 1997).

4. Models at the catchment scale

The concept of pollution can as described above be understood using the 'source-pathway-target' definition offered by Holdgate (1979). Much scientific work has been done and still needs to be done in order for us to gain understanding of the individual sources and pathways contributing in the pollution of the aquatic environment. This thesis contributes with investigating some pathways related to rural areas. A next step is to combine our range of process understanding and our existing quantitative results for the individual sources and pathways into descriptions or models that operate at the catchment scale. Such models or tools are needed for catchment managers in pollution control, for instance in the development of River Basin Management Plans under the newly adopted EU Water Framework Directive (European Community Directive 2000/60/EC).

A variety of tools exist and the approaches vary from very simple field tools to empirical and complex mechanistic models. Schoumans & Silgram (in press) and Merrit et al. (2003) have each given their overview of a number of the quantification tools, and have discussed their strengths and limitations. The tools are able at describing either just parts of, or all, the different pathways and processes that govern sediment or nutrient cycling at the catchment scale.

4.1 Field tools

Before considering the range of quantification tools normally adapted at the larger catchment scale, it is worthwhile to consider using very simple field tools. Simple field tools can be used for direct measurement of the transport from different sources and via different pathways in a catchment. They offer reliable data for the measured sites and also for the catchment in question if the spatial and temporal variations of the anthropogenic pressures and physical catchment processes are covered. Normally they require much man work and the quantifications are not directly applicable for other catchments. For small high priority catchments they can be very useful, and furthermore, they often supply quantitative data used in empirical models (section 4.2 and 4.3) or index tools (section 4.4) used at the catchment scale.

An alternative to measurements on the individual pathways acting here and now, is to apply a 'fingerprint' technique for identifying contemporary or historic delivery sources within catchments (e.g. Walling et al., 1993). Fingerprinting of sediments is dependent upon the presence of one or several traceable identifiers in the sediment compartments in the catchment. Numerous tracer properties can be applied, including mineralogy, particle size, geochemistry, mineral magnetic signatures and radionuclides. Preliminary results from the small, rural Danish Lillebæk catchment indicate a similarity between suspended sediment of the upper stream site and the topsoil and upper bank compartments (Table 7). Statistical methods can be applied for estimating the relative suspended sediment contribution from different sediment compartments (e.g. Foster, 2001). A successful estimation of relative contributions, however, demands a sufficiently strong difference in signatures between the sediment source compartments.

Table 7 Mineral magnetic signatures for suspended sediment of upper and lower stream sampling sites, and of different sediment compartments of the Lillebæk catchment (Laubel et al., unpublished). Differences in particle size were not fully taken into account.

Compartment	N	Xhf	Xfd	Xarm	SIRM	IRM	HIRM
		$\mu m^3\ kg^{-1}$	$nm^3\ kg^{-1}$	$\mu m^3\ kg^{-1}$	-------	$mAm^2\ kg^{-1}$	-------
Upper stream	5	0.34	11.2	2.4	3.6	2.6	3.1
Lower stream	6	0.24	9.5	2.0	2.3	1.6	1.9
Topsoil	12	0.28	14.7	2.0	2.4	1.5	1.9
Upper bank	10	0.27	8.3	1.4	2.4	1.5	2.0
Lower bank	10	0.23	5.2	1.0	1.8	1.1	1.5
Subsoil	12	0.23	6.0	1.2	1.9	1.0	1.4

The fingerprint technique is a field-based tool that can be used for identifying the contributing sources and pathways at the catchment level. The tool can, however, not be used for implementing management measures at a smaller scale (the field level) because it operates on a catchment or sub-catchment scale.

4.2. The strengths and weaknesses of different models

Models differ in their complexity, their resolution in time and space, and they need different levels of detail in terms of data requirements (Figure 13). The term 'quantification tool' may be used instead of 'models' (Schoumans & Silgram, in press). Quantification tools have often been applied at different scales and cover a wide range of tools from spatially lumped static tools to fully distributed process orientated dynamic models (Figure 13). Many quantification tools have only been applied to specific parts of the world, which means that they may not be able to handle the gradient in climate (e.g. frozen soils), hydrology (shallow groundwater), land use and/or agricultural practices existing in other parts of the world.

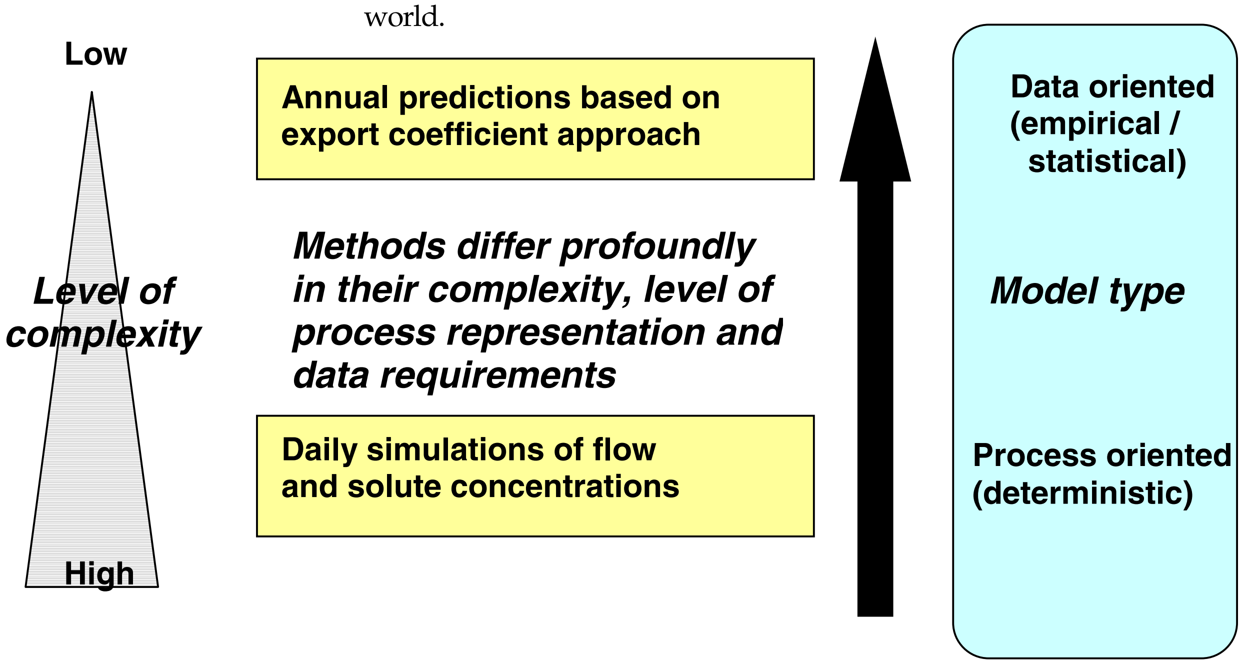


Figure 13 Diagram showing the relationship between complexity (left), output, and model type (right) (Schoumans & Silgram, in press).

Process-orientated dynamic quantification tools normally require large amounts of input data at a very detailed temporal and spatial scale. In many cases, such detailed data may not be available, at least not at the larger scale, and some assumptions or default values needs to be made. Empirical and quasi-empirical approaches may in such cases be viable alternatives. Even in this category there is a large variability in complexity (e.g. Grimvall & Stålnacke, 1996; Caraco & Cole, 1999). However, many empirical and quasi-empirical based models have the limitation that they may not be able to describe the dynamics in the fluxes. This trade-off between the complexity and applicability of these two approaches has been discussed by several authors (e.g. De Vries, 1994). Simple models are also often very weak in their possibility of predicting changes from management or climate outside the data range from which they emerge. Only few quantification tools have been applied in Denmark for the quantifying of sediment and sediment-associated substance transport to streams (Table 8).

Table 8. Pathways where models to my knowledge have been applied for quantifying losses of sediment and P at the catchment scale in Denmark

	Pathways					
	Mobilisation from soil erosion	Loss via surface runoff including deposition	Loss via matrix flow and subsurface runoff	Loss via preferential flow and subsurface runoff	Loss via bank erosion	Loss via groundwater flow
Sediment	Yes	No	No	Yes	Yes	No
Particulate P	No	No	No	No	No	No
Dissolved P	No	No	No	No	No	No

4.3 The P-index tool

The P-index was developed in USA during the 1990s as a tool for ranking the risk for P loss from individual fields to the stream. Forty-seven from fifty states in USA nowadays use the P-index in order to manage P in agricultural areas (Sharpley et al. 2003). The P-index tool is based on the concept of Critical Source Areas (Figure 4). The P-index integrates all sources and all modes of transport in ranking the risk for total P loss. It can be modified into making rankings separately for particulate P and dissolved P (pers. com., Douglas Beegle, Penn State University, USA). The P-index does not produce quantitative results but describes potential losses of P. It is a tool of low complexity and it may be used by regional authorities to pinpoint individual high-risk fields.

In Denmark, a modified version of the original P-index, i.e. the Pennsylvania P-index, has been suggested (Andersen et al., 2004). The most important modifications refer to artificially drainage from fields and to the potential leaching of dissolved P. Furthermore, some modification has been introduced in relation to the distance between field and stream, and to the potential effect from the buffer zone (Andersen et al., 2004). Similarly to the original P-index (Sharpley, 1995), the Danish P-index showed a significant, positive relationship between P-index and P loss (r^2=0.79) on a number of small sub-catchments. Such a 'set-up' dealing with sub-catchments rather than individual fields supplies some indirect indication of the precision of the ranking. In principle, however, it should be tested at the field scale rather than at the sub-catchment scale.

5. Perspectives

The identifying of contributing areas and pathways of pollution is the basis for developing management plans aiming at reducing pollution. Index tools supply such identification and also a description of the potential losses. They should preferably be used on a field-by-field basis in each catchment. In addition, tools or models for the actual quantification of the pollutant transport from sources and pathways are important for being able at applying cost-effective management strategies. Process-orientated dynamic quantification tools normally require large amounts of input data. If the difficulties and costs in achieving such input data prevent this type of quantification tool from a broader use, empirical and quasi-empirical approaches may be viable alternatives.

In relation to the development of River Basin Management Plans under the newly adopted EU Water Framework Directive (European Community Directive 2000/60/EC) there is a need for operational tools that can help focussing future management measures in order for many water bodies to improve their conditions.

Index tools similar to the Pennsylvania P-index lack fundamental ability in actual quantification of losses and impacts from changing management measures. On the other hand, they have a potential for broader use due to their simplicity and the fact that they are interlinking conditions related to different pollutant sources and pathways in an operational manor. The development of an index tool specially indented for assessing sediment-associated substances losses could be valuable due to its potential use as a sub-index for the particulate fraction of any pollutant - phosphorus, nitrogen, heavy metals, pesticides etc.

In order to improve the quantification tools for the catchment scale it is viable to continue investigations on the different sources and pathways. Partly because the data and understanding achieved from such studies is a core material for building and rebuilding quantification tools and index tools, and partly because any quantification tool need data for calibration, also due to national or regional differences in site-specific conditions.

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URL: http://www.dmu.dk

National Environmental Research Institute
Frederiksborgvej 399
PO Box 358
DK-4000 Roskilde
Denmark
Tel: +45 46 30 12 00
Fax: +45 46 30 11 14

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DK-8600 Silkeborg
Denmark
Tel: +45 89 20 14 00
Fax: +45 89 20 14 14

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Grenåvej 12-14, Kalø
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Denmark
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