Risk Assessment of Erosion and Losses of Particulate Phosphorus

A series of studies
at laboratory, field and catchment scales

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Risk Assessment of Erosion and Losses of Particulate Phosphorus. A series of studies at laboratory, field and catchment scales

Abstract

Phosphorus (P) losses from agricultural land are considered a major contributor to eutrophication in many aquatic ecosystems. Areas more vulnerable to losses of P need to be identified in order to effectively apply mitigation measures aimed at reducing total loads of P. This thesis focuses on the identification of soils and fields vulnerable to losses of particulate P (PP) due to erosion. Two simple soil dispersion tests to estimate the initial risk of soil and P mobilization (DESPRAL and SST) were tested and compared in the laboratory. The outcome was combined with data relative to source (soil P content) and transport (unit stream power length-slope topographic factor calculated from a high resolution digital elevation model) risks to establish probable causes of P losses at field scale and to target critical source areas at catchment scale.

DESPRAL showed higher precision and shorter execution time than SST, in addition to its already proven validation and reproducibility. Also, compared with other methods, the test returned a wider range of values for each textural class, allowing the differentiation of soils within these classes. This is especially important for fine-textured soils, which are the most sensitive to the mobilization of particles. The study of long-term P and sediment losses from five fields confirmed the relevance of adequately identifying the source and transport conditions within fields when assigning appropriate countermeasures. Finally, the assessment of long-term losses from two contrasting catchments highlighted how transport and mobilization risks have a greater effect on P losses due to erosion than P accumulation in soil. When ranking the fields within both catchments according to this prioritization of factors, a greater number of high-risk fields were found in the catchment with more pronounced transport pathways.

The outcome of this thesis is the proposal of methodology whereby easily obtainable data can be used in risk assessments to identify fields and catchments vulnerable to PP losses. The knowledge gained provides a good starting point to improve these assessments by incorporating means for prioritizing different mitigation measures currently not performed in Sweden.

Keywords: erodibility, erosion, critical source areas, high resolution digital elevation models, mobilization risk, phosphorus, particulate phosphorus, soil dispersion, transport risk

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For nitrates are not the land, nor phosphates and the length of fiber in the cotton is not the land. Carbon is not a man, nor salt nor water nor calcium. He is all these, but he is much more; and the land is so much more than its analysis.

John Steinbeck, *The Grapes of Wrath*

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List of Publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I Villa, A., Djodjic, F., Bergström, L., Wallin, M. (2012). Assessing soil erodibility and mobilization of phosphorus from Swedish clay soils – Comparison of two simple soil dispersion methods. *Acta Agriculturae Scandinavica, Section B – Soil & Plant Science* 62 (Supplement 2), 260-269.

II Villa, A., Djodjic, F., Bergström, L. (2014). Soil dispersion tests combined with topographical information can describe field-scale sediment and phosphorus losses. *Soil Use and Management*, DOI: 10.1111/sum.12121.

III Villa, A., Djodjic, F., Bergström, L. Ranking risk areas in two catchments for sediment and phosphorus losses to improve prioritization of mitigation strategies. Manuscript.

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The contribution of Ana Villa to the papers included in this thesis was as follows:

I Planned the study together with co-authors and had the main responsibility for part of the soil sampling. Was responsible for performing laboratory analyses of erodibility and suspended solids. Had the main responsibility for data analyses and interpretation, as well as writing the paper, with assistance from all co-authors.

II Planned the study together with co-authors and had the main responsibility for planning the sampling campaign, collecting the soil samples, and performing laboratory analyses of erodibility and suspended solids. Had the main responsibility for data analyses and interpretation, as well as writing the paper, with assistance from all co-authors.

III Planned the study together with co-authors and had the main responsibility for planning the sampling campaign, collecting the soil samples, and performing laboratory analyses of erodibility and suspended solids. Had the main responsibility for data analyses and interpretation, as well as writing the paper, with assistance from all co-authors.

Abbreviations and Terms

CSA	Critical source area
DESPRAL	Environmental soil test to determine the potential from sediment and phosphorus transfer in run-off from agricultural land
DRP	Dissolved reactive phosphorus
FWC	Flow-weighted concentration
K	Erodibility factor from the USLE/RUSLE
LiDAR	Light detection and ranging
LS	Length-slope topographic factor
NTU	Nephelometric turbidity unit
P-AL	Ammonium lactate-extractable phosphorus (indicator of plant available P)
PER	Phosphorus enrichment ratio
PI	Phosphorus index
PP	Particulate phosphorus
RUSLE	Revised universal soil loss equation
SS	Suspended solids
SST	Soil suspension test
TP	Total phosphorus
UP	Unreactive phosphorus (equivalent to PP)
USLE	Universal soil loss equation
USPED	Unit stream power-based erosion/deposition

1 Introduction

The 20^{th} century demonstrated how an exponential growth in the use of phosphorus (P) increased agricultural productivity, but also saw a rise in new environmental, social and political issues that need attention. Phosphorus (from the Greek *phosphoros*, meaning light-bringer) is a non-metallic element from the nitrogen (N) family in Group 15 or 5A of the periodic table and most of it occurs as minerals in phosphate rock. Due to the high reactivity of its most common elemental form (white P), it cannot be found as a free element on Earth, as it spontaneously combusts in air. As phosphate (PO_4^{3-}), it is an essential component of all genetic material (DNA), and is used by cells to carry energy in the form of adenosine triphosphate (ATP). As hydroxyapatite ($Ca_5(PO_4)_3OH$), it is the main component of bones and teeth. While P has been used in many applications in modern industry (*e.g.* production of detergents, pesticides, baking powders, matches and nerve agents), the vast majority of the extracted phosphate rock (approximately 90%) is destined for food production. Meeting increasing demand due to world population growth and diet change (increasingly meat intensive) might be complicated in the future by the fact that the majority of phosphate rock reserves are concentrated to a few areas of the world, mainly Morocco and Western Sahara (Cordell *et al.*, 2009).

Awareness of the environmental issues linked to P losses increased significantly during the second half of the 20^{th} century and has continued to increase ever since. While the amounts of P lost from fields to water bodies through different processes may seem small compared with the amounts of P actually retained in soils, from an environmental point of view these losses are sufficient to cause significant damage to aquatic ecosystems. In the case of eutrophication, nutrient enrichment in water bodies causes rapid growth of undesirable algal populations, ultimately leading to oxygen depletion of those aquatic ecosystems when the dead algae decompose. The presence of an excessive amount of P accelerates the slow natural process of eutrophication, turning it into what is called 'cultural' eutrophication (caused by human activity). The largest anthropogenically induced hypoxic area (*i.e.* area with reduced oxygen conditions) in the world is located in the Baltic Sea, with recent research estimates showing a total of 60 000 km^2 of dead sea bottom. The main driver of this process is considered to be the increased inputs of nutrients from land (Carstensen *et al.*, 2014).

The increased awareness of the issue in recent decades has resulted in 'Zero eutrophication' becoming one of the 16 Environmental Quality Objectives established by the Swedish government for the new millennium (Gov. Bill 2000/01:130). This is in line with the main requirement of the EU Water Framework Directive "achieving good ecological status in surface waters" (Directive 2000/60/EC). Reducing the eutrophication of water bodies involves dealing with P and N, the main question being which measures to prioritize, *i.e.* controlling the levels of P, the levels of N or a combination of both. In general, it is more or less accepted that control of P should be the main focus for inland waters (Schindler, 2012), while N is more important in marine environments (Howarth & Marino, 2006). In the case of the Baltic Sea, a thorough examination and evaluation of different mitigation strategies was made by an expert group appointed by the Swedish Environmental Protection Agency (SEPA), which resulted in a broader agreement on the fact that P inputs should be reduced, while no definitive consensus was reached regarding N (Boesch *et al*., 2006).

In response to the abovementioned assessments, a wide research on P related issues was started. A major part of this research is supported by the Swedish Farmers' Foundation for Agricultural Research, starting in 2009 with the focus on developing mitigation strategies for improved fertilizer utilization and reduction of eutrophication. A wide variety of projects have been conducted since then, focusing on areas such as feed conversion and sustainable livestock management (*e.g.* Parvage *et al.*, 2013), soil processes and transport routes (*e.g.* Andersson *et al.*, 2013; Paraskova *et al.*, 2013; Djodjic & Spännar, 2012) and fertilization strategies to reduce P losses at source (*e.g.* Liu *et al.*, 2013; Riddle & Bergström, 2013; Svanbäck *et al.*, 2013) or along water routes to reduce P transport (*e.g.* Kynkäänniemi *et al.*, 2013; Johannesson *et al.*, 2011). The research presented in this thesis pertains to the issue of soil processes and transport routes and aims at improving the tools and methods used for the identification of soils, fields and areas most vulnerable to losses of P driven by erosion. Ultimately, one of the main objectives of the studies presented in this thesis was to devise simple methodologies and tools to be used by farmers and land managers for the effective placement of mitigation measures.

2 Background

2.1 Agriculture as a Source of Nutrients

The importance of agriculture as a non-point source of nutrients has grown as control of point sources such as wastewater has improved following the requirements of the EU Urban Wastewater Treatment Directive (91/271/EEC). In Sweden, over 95% of urban waste water undergoes biological and chemical processing and the removal of P reaches levels of 95% (Swedish EPA, 2014). In many areas of the world, agriculture is considered an important source of nutrients. In the USA, it is considered to be the main non-point source of P (Carpenter *et al.*, 1998), while in Finland, agricultural P losses account for 62% of all P entering surface waters (Valpasvuo-Jaatinen *et al.*, 1997). In the North Sea, the annual loads of P from anthropogenic diffuse sources comprise 46% of the total inputs of P (European Environmental Agency, 2005). In Sweden, agriculture accounts for 48% of the total anthropogenic P discharges to the Baltic Sea (SMED, 2011) and is thus its largest single source from land to sea. The majority of the agricultural land in Sweden is located in the southern and south-central parts of the country, which is also where the main problems of eutrophication occur.

During a great proportion of the last century, high amounts of nutrients were added to farmland in Sweden in efforts to increase the productivity of agricultural land in the region. The increased inputs of nutrients, frequently a mix of both fertilizers and farmyard manure, generally exceeded plant uptake, thus causing a high surplus of P in soils. This is particularly evident in areas with intensive livestock farming. Nowadays, while excessive amounts of P are still applied to some fields of the country, the overall P average shows that inputs are in equilibrium with outputs (approx. 12 kg P ha^{-1}) (Statistics Sweden, 2013).

2.2 Transfer of Phosphorus from Agricultural Land

Transfer of P from land to water takes place through different processes (erosion, leaching and incidental losses) and pathways (*e.g.* overland flow, matrix flow, preferential flow) and in different forms (*e.g.* dissolved P, particulate P, organic P) (Haygarth & Sharpley, 2000). The main forms of P referred to throughout this text are dissolved P or dissolved reactive P (DRP) and particulate P (PP). The latter is also referred to as unreactive P (UP), calculated as the difference between Total P (TP) and DRP. The terminology refers to the molybdate reaction used in the analysis of the different P fractions in water[1].

The proportions of the different forms of P in water are closely related to the type of transport process that initiates the transfer of P from land to water. Transport processes are mainly dependent on soil type, hydrological conditions, climate and agricultural production. For instance, erosion is more likely to be the dominant process for P losses from fine-textured soils located on a sloping field, in which case P will mainly, but not exclusively, be lost as P attached to particles (PP). In contrast, leaching and losses of DRP are likely to dominate in sandy soils with low sorption capacity located in a flat area. In the latter case, the values vary between different locations. In north-eastern Europe, the ranges of DRP content (as percentage of TP) have been reported to range between around 20-60% for the United Kingdom, 9-23% for Norway and 20-80% for Ireland (Ulén *et al.*, 2007). In Sweden, the variation is also wide, with DRP comprising 20-85% of TP. The differences in variation are correlated with the occurrence of transport processes. In some areas, the main process occurring is soil erosion (higher percentage of PP), whereas in others it is leaching (Bergström *et al.*, 2007). In the different river tributaries to Lake Mälaren in south-central Sweden, P associated with particles accounts for approximately 64% of TP (Persson, 2001). In addition, in the 21 monitoring catchments across Sweden which are part of the national monitoring programme for agricultural land, 13 have PP as the main fraction (>60% of TP) (Figure 1). However, in many of the cases where PP is the dominant fraction in TP, high concentrations of DRP have also been observed (*e.g.* catchments E23, E24, K32, O14, O18 and U8 in Figure 1). In other words, the presence of one fraction does not exclude the presence of the other.

[1] The terminology used throughout this thesis essay for referring to unreactive phosphorus (UP) is particulate phosphorus (PP). The main reason for this is that PP can be more easily visualized by a broader audience. However, the term UP is used in Papers I-III, as it accounts more accurately for its calculation, namely the difference between total reactive P (TP) and dissolved reactive P (DRP). This includes all kinds of P attached to particles, as well as dissolved fractions which are not reactive with ammonium molybdate, such as organic P fractions.

In Sweden, TP concentrations and loads are generally considered to be low. Long-term TP flow-weighted concentrations (FWC) in the national monitoring catchments range from 0.047 to 0.38 mg L^{-1}, while TP load range from 0.1 to 1.0 kg ha^{-1} yr^{-1} (Figure 1). For reference, the corresponding data for micro-catchments in the Nordic-Baltic region show mean annual TP losses ranging from 0.1 to 4.7 kg ha^{-1} yr^{-1}, while losses in macro-catchments in Europe range from 0.1 to 6.0 kg ha^{-1} yr^{-1} (Kronvang *et al.*, 2007). Losses from the Chesapeake region (USA) range from 3.2 to 24.2 kg ha^{-1} yr^{-1} (Boynton *et al.*, 1995).

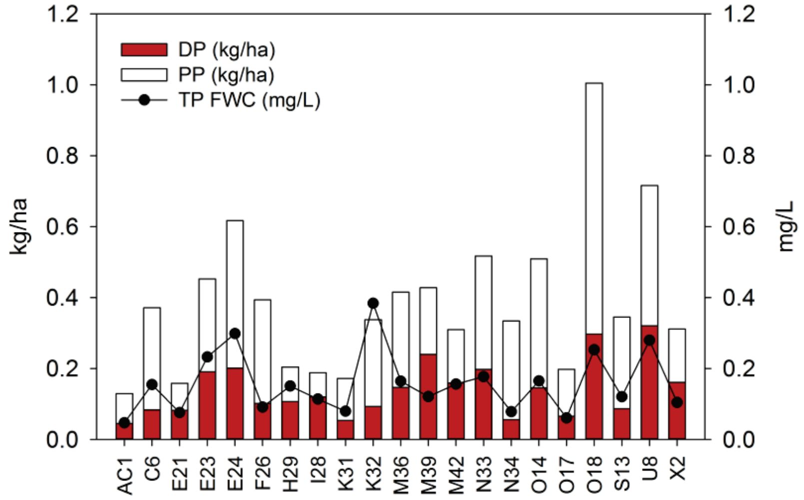

Figure 1. Mean annual phosphorus (P) transport and mean flow-weighted P concentration at the outlet of 21 Swedish monitoring agricultural catchments in the period 1996/1997-2012/2013. The red bars represent the dissolved fraction and the white represent the particulate fraction (PP). The dotted line represents mean annual flow-weighted concentration (FWC) of total phosphorus (TP). The letters in the catchment codes represent Swedish regions. Data extracted from the Swedish monitoring program investigating nutrient losses in catchments dominated by agricultural land.

Losses of P due to erosion have been detected in a number of agricultural areas dominated by clay and silty soils across Sweden (Djodjic *et al.*, 2012; Djodjic & Bergström, 2005; Ulén & Jakobsson, 2005). At field-scale, Ulén *et al.* (2001) have reported a significant correlation between TP losses and soil texture and showed that suspended solids (SS) concentrations were higher in water samples from fields with clay content >35%. Similarly, Kyllmar *et al.* (2006) reported higher P losses from catchments dominated by clay loam and clay soils than from catchments dominated by loamy sand or sandy loam soils.

They also reported significantly higher mean annual N losses in the latter catchments. More than half (55%) of all agricultural soils in Sweden contain more than 15% clay and 15% of all soils can be classified as heavy clay (>40% clay content) (Eriksson *et al*., 1999). According to Rekolainen *et al.* (1997), about 39% of all soils in Sweden contain more than 30% clay.

The impact of P on water quality depends on its availability to algae as not all of the TP loads leaching to a water body are available to the biota. Bioavailable P can be defined as the sum of immediately available P (*i.e.* orthophosphate or DRP) and any form of P that can be transformed into an available form by natural processes (*e.g.* desorption, dissolution, enzymatic degradation) (Boström *et al.*, 1988). Particulate phosphorus potential bioavailability is complex as it is a dynamic function of different chemical-physical-biological phenomena occurring during transport of P through a catchment (Dorioz *et al.*, 1998; Sharpley *et al.*, 1992). As a reference, the potential bioavailability of PP from agricultural losses in Scandinavia varies between 5 and 41% (Rekolainen *et al.*, 1997). In Sweden, bioavailability assessments have shown that an average of 41-45% of PP from tributaries to Lake Mälaren can become available to algae (Persson, 2001). In Finland, Uusitalo *et al.* (2003) found that PP makes a significant contribution to total bioavailable P losses, although previous Finnish studies had shown a low percentage of potential PP bioavailability (Ekholm, 1994). Similar studies in the USA, produced estimates of potential bioavailable PP from runoff in agricultural, forested, urban and mixed use streams of 24, 17, 73 and 26%, respectively (Ellison & Brett, 2006). In addition to all the estimates cited above, the possible long-term threat of P in sediments has been stressed by Stigebrandt *et al.* (2013), who suggest the existence of a temporary internal source of DRP stemming from anoxic sediments in the Baltic Sea. According to their results, P has been accumulating in the oxic sediments of the Baltic Sea due to *e.g.* accumulation of P associated with iron (Fe) oxides, burial of P in organic matter (OM) or P accumulation in bacteria, until the actual anoxic conditions have favoured the release of P as DRP, thus making it bioavailable.

2.2.1 Soil Erosion

As mentioned above, soil erosion is one of the processes through which P can be lost from agricultural areas. Soil erosion (from the Latin *erodere* – to eat away) is a two-part process by which soil particles are detached from the soil and transported by the action of an erosive agent (*e.g.* water, wind, tillage). Deposition of particles occurs when there is no more energy to continue the transport. Human-induced erosion is closely linked to agricultural practices, characterized by the replacement of natural vegetation by arable land with very little or no vegetation to protect the soil surface during a large proportion of the year. Historically, attention has mainly been given to the effects of soil erosion at source with the focus on soil degradation and decreased agricultural productivity, which affects food security in many areas of the world (Pimentel, 2006). However, the effects of soil erosion can also be seen offsite, as soil particles can pollute surface waters and act as carriers of other pollutants, such as pesticides or nutrients.

Soil erosion can have several degrees of intensity (*e.g.* sheet, rill, gully). The most common form in Sweden is sheet erosion, which is the uniform loss of a thin layer of topsoil. While not necessarily very noticeable, this form of erosion can in fact be the most harmful in terms of nutrient losses, due to the preferential detachment of finer-sized soil particles, which have higher specific surface areas for adsorption of P (Sharpley, 1985). Another form of erosion observed in Sweden is rill erosion, which occurs when small channels (only a few centimetres deep) are formed by small intermittent water courses. Finally, gully erosion occurs when these small channels grow into deeper channels that cannot be removed by normal cultivation. Gully erosion is not usually seen in Sweden although gullies have occasionally been recorded after extreme flow events (see Figure 2).

Figure 2. Severe erosion in the agricultural area of Krusenberg (region of Uppland, Sweden), April 2013. Photo: Faruk Djodjic.

According to Ulén (2006), roughly up to 15% of the arable land in Sweden can be assumed to be a source of soil erosion. This estimate is based on the percentage of clay soils and of soils that might have limited natural or artificial drainage capacity. In Sweden, it is not uncommon to see surface water ponding due to overland flow ending in small depressions in fields due to the limited capacity of drainage systems (*i.e.* spring flow or autumn) as seen in Figure 3. This can increase P losses in the affected areas, both because macropore flow is favoured under ponded conditions (Skaggs *et al.*, 1994) and because surface water ponds can overflow to nearby streams, carrying over high amounts of particles and nutrients.

Contrary to what one might expect, there are few existing studies on losses of SS and P due to erosion in Sweden. The few that have been carried out have resulted in the detection of severe forms of erosion, especially during snowmelt and thawing of frozen soil (Alström & Bergman, 1990). In one such study covering a 90 km^2 area in southern Sweden it was estimated that 7% of the study area was affected by serious soil degradation (losses of 0.001-120 t ha^{-1} yr^{-1}) and that interrill erosion losses varied between 0.001 and 16 t ha^{-1} yr^{-1} (Alström & Åkerman, 1992). As reference, the mean estimated soil erosion rate from plots across Europe with different land uses is 8.76 t ha^{-1} yr^{-1}, with bare soil having the highest mean rate (23.4 t ha^{-1} yr^{-1}) (Cerdan *et al.*, 2006).

Figure 3. Surface ponding overflowing to a stream in an agricultural catchment in the region of Östergötland. Photo: Anuschka Heeb

3 Assessment of Particulate Phosphorus Losses through Erosion

Estimation of P losses from agricultural land requires an understanding of the journey of P from its application as a fertilizer or manure to its fate in receiving water, via its release from soil and subsequent transport from the release point to the water body. This is the basis of the P transfer continuum (source-mobilization-delivery-impact) proposed by Haygarth *et al.* (2005) and illustrated in Figure 4.

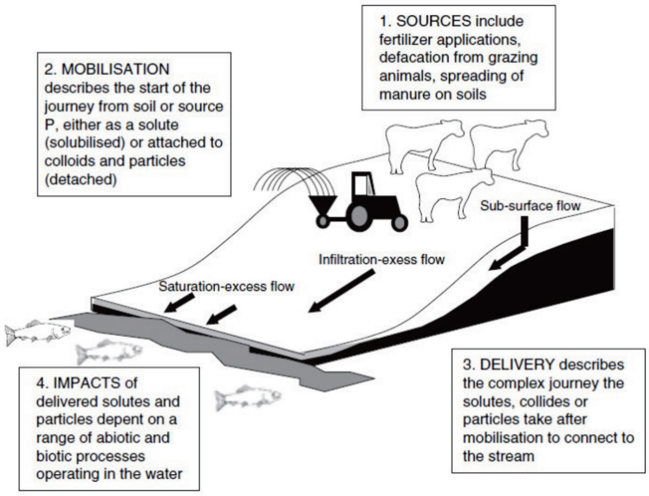

Figure 4. Phosphorus transfer continuum. Source: Withers and Haygarth (2007)

In the past, agronomic soil test P (*e.g.* P-AL, Olsen P, Mehlich-3 P) were used for environmental purposes to identify thresholds for the prediction of surface water pollution (Maguire *et al.*, 2005). Agronomic tests provide valuable information, especially when assessing DRP losses, but they need to be complemented with other information regarding the susceptibility of a site to overland flow and erosion (Weld *et al.*, 2001). This is the idea behind concepts such as the P index (PI) (Lemunyon & Gilbert, 1993) and Critical Source Areas (CSAs) (Pionke *et al.*, 2000), which emphasize that P losses from watersheds originate from areas where high levels of P at the source overlap with high potential for P transport. As an example, observations of TP losses from agricultural catchments in the Nordic countries have shown that the losses are not related to surplus P in soils (Kronvang *et al.*, 2007), but that other factors such as mobilization risk, overland flow risk, connectivity to the watercourse and hydrological processes have a greater influence on the form and intensity of P losses.

3.1 Estimation of Mobilization Risk

Mobilization of P attached to particles is called detachment. Erodibility is the inherent vulnerability of soils to detachment. The erodibility (K) factor defined in the Universal Soil Loss Equation (USLE) is determined by measuring soil losses from plots under natural or simulated rainfall. As expressed by Foster *et al.* (1981), erodibility is the rate of soil loss per unit of R or EI (rainfall erosivity) for a specific soil, as measured on a unit plot (22.1 m length of uniform 9% slope maintained in continuous clean-tilled fallow). This is the most accurate method to assess erodibility, but it is also expensive and requires large amounts of labour and time. These cost issues have led to the development of a nomograph constructed from numerous soil data collected across the USA, which provides a solution to an equation based on percentages of silt, sand and OM content, soil structure and permeability (Wischmeier *et al.*, 1971). Values of the erodibility factor (K), converted to SI units[2], range from 0.007 to >0.05 h ha ha^{-1} MJ^{-1} mm^{-1}. There are other methods available to determine erodibility but, as already noted by Harris (1971), a method that produces accurate and reproducible measures is still lacking. An alternative to the K factor from the USLE/Revised USLE (RUSLE) is soil testing based on aggregate stability, as this is probably the soil property most closely related to erodibility (Amézketa, 1999). Two types of tests describe aggregate stability, those that refer to macro-aggregate stability (*e.g.* wet sieving) and those that refer to micro-aggregate stability (*e.g.* soil dispersion tests). Some examples of these methods are presented in Table 1.

[2] Empirical units (U.S customary units) converted to SI units by multiplying by 0.1317.

Table 1. *Examples of methods currently available to estimate sediment mobilization risk*

Type	Name	Comments	Reference
Field	Erodibility index (K)	Mean annual soil loss per unit of R[a]	(Wischmeier & Mannering, 1969)
Nomograph based on field measurements	Erodibility index (K)	Equation using soil properties (texture, OM, stability, permeability)	(Wischmeier *et al.*, 1971)
Macro-aggregate/ Laboratory	Wet sieving	Aggregation (aggregate size distribution)	(Yoder, 1936)
Macro-aggregate/ Laboratory	Wet single-sieve: WSA[b]	Aggregation (stability)	(Kemper & Rosenau, 1986)
Laboratory	Mechanically dispersed clay	Dispersible clay after imposing different mechanical energy inputs	(Watts *et al.*, 1996)
Micro-aggregate/ Laboratory	Dispersion ratio	(%silt+%clay in undispersed soil) / (%silt+%clay after dispersal)	(Middleton, 1930)
Micro-aggregate/ Laboratory	Aggregation Index (AI)	100 (1 – WDC[c]/clay)	(Rhoton *et al.*, 2007)
Micro-aggregate/ Laboratory	Index of Structure	100 (1 – natural clay/total clay)	(Harris, 1971)
Micro-aggregate/ Laboratory	DESPRAL	Soil dispersed calibrated with soil loss from lysimeters	(Withers *et al.*, 2007)
Micro-aggregate/ Laboratory	SST	Soil dispersed calibrated with soil loss from lysimeters	(Udeigwe *et al.*, 2007)

[a]R is the rainfall erosivity factor, which considers rainfall amount and intensity; [b]water-stable aggregate percentage; [c]WDC is water dispersible clay

In the soil dispersion tests, the dispersed particles in a soil suspension are quantified at a specific time and depth calculated according to Stokes' law[3]. More interestingly, tests such as DESPRAL (Withers *et al.*, 2007) and SST (Udeigwe *et al.*, 2007) have been calibrated with results from rainfall simulation experiments, in the way that the amount of soil dispersed is correlated with the amounts obtained in surface runoff from rainfall lysimeters. This means that the soil dispersion test does not provide an absolute result of SS losses, but rather a relative value which is useful to rank the vulnerability of different soils. In addition, the recovered aliquot is useful for analyzing the different P fractions that are mobilized and relating those to the properties of the corresponding soil. The two tests are fairly recent and have been used in several studies in Europe (DESPRAL) and USA (SST). An overview of the

[3] Mathematical equation expressing the settling velocity of small spherical particles in a fluid medium (Encyclopædia Britannica)

two is presented in Table 2. Some of the flaws of soil dispersion tests discussed by Bryan (1968) are that they do not consider the possibility that high-velocity raindrops disperse previously undispersed material, and that they do not accurately reflect the mobilization risk of soils with a high sand content. Under Swedish conditions, neither of these is an issue, as rainfall is usually not severe and sand particles have less risk of being mobilized.

Aggregate stability has been compared to field erodibility, proving to be a good indicator of soil susceptibility to runoff and erosion. For example, Barthès and Roose (2002) found that erodibility estimated in the field from Mediterranean soils was correlated with aggregate stability, specifically with slaking, usually represented through macro-stability tests. On the other hand, Middleton (1930) concluded that more erodible soils were those most susceptible to being dispersed. More recent studies comparing macro- and micro-stability tests have shown diverging results, some reporting a correlation between these tests (Perfect *et al.*, 1990; Pojasok & Kay, 1990) and others not (Withers *et al.*, 2007).

Table 2. *Studies using soil dispersion tests to describe sediment mobilization risk*

Test	**Origin**	**Reference**	**Range of SS dispersed**	**Range of TP dispersed**
DESPRAL	EU	(Withers *et al.*, 2007)	0.28 - 2.68 g	0.3 - 2.5 mg (approx.)
	UK	(Withers *et al.*, 2009)	0.18 - 1.5 g L^{-1}	0.16 - 3.0 mg L^{-1}
	Italy	(Borda *et al.*, 2010)	0.31 - 2.75 g L^{-1}	0.12 - 3.30 mg
	Italy	(Borda *et al.*, 2011)	0.85 - 1.41 g L^{-1}	1.58 - 2.01 mg L^{-1}
	UK	(Scholefield *et al.*, 2013)	Not available	0.10 - 4.4 mg L^{-1}
	UK	(Zhang *et al.*, 2013)	Not available	Not available
SST	USA	(Udeigwe *et al.*, 2007)	50 – 750 NTU (approx.)	Not available
	USA	(Udeigwe & Wang, 2010)	Not available	Not available

In Sweden, studies reporting soil erodibility and sediment mobilization risk are scarce. This lack of soil erodibility data in Sweden has also been pointed out by Panagos *et al.* (2014), who reached this conclusion after an extensive literature review on soil erodibility. Some studies performed in Sweden on aggregate stability and soil dispersion are listed in Table 3. Many of these refer to the readily dispersible clay (RDC) test (Etana *et al.*, 2009), which is based on Dexter (1988) and measures the clay fraction that potentially disperses in water after a small amount of mechanical energy is applied. This type of test has been specially used to study the impact of tillage on particle mobilization

Table 3. *Examples of studies estimating sediment mobilization risk in Sweden*

Test	Type of study	Reference
DESPRAL[a]	Soil comparison	(Villa *et al.*, 2012)
	Field-scale	(Villa *et al.*, 2014)
	Catchment-scale	(Villa et al., Paper III)
SST[a]	Soil comparison	(Villa *et al.*, 2012)
Readily dispersible clay (RDC)[a]	Plots	(Etana *et al.*, 2009)
	Plots	(Ulén *et al.*, 2012a)
	Long-term fertility fields	(Kirchmann *et al.*, 2013)
	Field & lysimeter	(Ulén & Etana, 2010)
	Catchment-scale	(Ulén *et al.*, 2011)
	Plots	(Myrbeck *et al.*, 2012) following (Czyz *et al.*, 2002)
K (RUSLE)	Soil comparison	(Villa *et al.*, 2012)
	Catchment-scale	(Ekologgruppen, 2012)
	Catchment-scale (GIS modelling)[b]	(Larsson, 2011)
K (USLE)	Catchment-scale (GIS modelling)[c]	(Sivertun & Prange, 2003)
Aggregate stability: single-sieve	Long-term fertility fields	(Gerzabek *et al.*, 1995) following (Murer *et al.*, 1993)
Dry aggregate stability	Plots	(Myrbeck *et al.*, 2012) following (Dexter & Kroesbergen, 1985)

[a]Measured as turbidity (NTU units); [b]use of 6 values of K based on a texture map produced by the Swedish Geological Survey; [c]use of 5 values of K based on texture classes (clay, silt, sand), organic soils and gravels/hard rock.

(Czyz *et al.*, 2002; Watts *et al.*, 1996) and is a good alternative to field erodibility estimations. However, the tests are more difficult to reproduce and are more time consuming than simple tests based on dispersion with water after shaking (*i.e.* SST and DESPRAL), which can more easily be used within routine environmental tests and are already well validated.

3.2 Estimation of Transport Risk

Phosphorus follows a complex journey since it is mobilized until it reaches a recipient water body. The complexity increases on moving up the scale, from laboratory/plot to field and catchment scales. Losses of P are temporally and spatially dependent (Pionke *et al.*, 1996). Different factors affect delivery of mobilized P during transport, such as hydrological events, the balance between particulate and dissolved fractions, the different pathways taken and the effects of land use and land management on both the transport and fractionation of P (Beven *et al.*, 2005). For instance, not all of the PP that is mobilized will directly reach a stream, as some of it will be deposited along the way and possibly be re-suspended in another, future event (Ballantine *et al.*, 2009). Identification of how overland flow is generated is also important in order to identify critical transport areas within a catchment. There are two types of overland flow based on how they are generated, infiltration-excess flow (Hortonian flow) or saturation-excess flow. The former occurs when the rate of precipitation exceeds the rate at which water can infiltrate (Horton, 1933). In the second case, the soil becomes saturated and any additional precipitation causes runoff, irrespectively of its intensity (Dunne & Black, 1970). Saturation-excess flow is common in the Nordic countries, especially after autumn rain, when the soil usually becomes saturated (Ulén *et al.*, 2012b), and in spring, during snowmelt (DeWalle & Rango, 2008).

Topography is a key factor determining the spatial variation in hydrological conditions (Sørensen *et al.*, 2006). There has recently been a considerable increase in the use of geographic information system (GIS)-based tools to identify transport pathways for nutrient pollution through topological representations (Shore *et al.*, 2013; Galzki *et al.*, 2011; Strauss *et al.*, 2007; Heathwaite *et al.*, 2005). These kinds of models describe the geospatial variation in transport risk in a very direct and intuitive way, and are useful as a decision support tool in catchment management. One example is the USPED model, which is a 3D improvement of the USLE/RUSLE models. In USPED, the Length-Slope (LS) parameter is derived from unit stream-power theory (Moore & Burch, 1986) and is a combination of the slope and flow accumulation (or upslope area) grids. This new LS parameter benefits from the ability to use higher-resolution elevation data compared with the LS factor in the USLE/RUSLE equations. One of the main limitations of the USLE/RUSLE models is that they only account for net erosion and obviate sediment deposition by only considering unidirectional flow (Kinnell, 2004). In contrast, the USPED model accounts for flow convergence/divergence. There are also concerns regarding the use of USLE outside the USA conditions under which the model was calibrated (Kinnell, 2010) and the fact that it might not be useful to predict sediment losses at the catchment scale (Boomer *et al.*, 2008). Another example of the use of topographical attributes derived from DEM in the literature is the Topographical Wetness Index (Beven & Kirkby, 1979), which is an indicator of soil moisture variability over a surface and is also based on upslope area and slope. There are many variations on this index, mainly differing in the calculation methods used to compute upslope area, slope and stream cell representation (Sørensen *et al.*, 2006).

4 Aim and Objectives

The overall aim of this thesis was to improve current assessments of P losses due to erosion by devising more accurate methods for the identification of soils and fields vulnerable to erosion and PP losses. Specific objectives in Papers I-III were to:

1. Evaluate two soil dispersion tests (DESPRAL and SST) for the estimation of sediment and P detachment risk, and study the effect of soil sample storage duration on soil dispersion (I).

2. Assess the mobilization of sediment and P from different types of soils using a single soil dispersion test (I-III).

3. Establish probable causes of long-term SS and PP losses at field scale combining source and transport factors (II).

4. Target and rank two agricultural catchments and critical source areas within these catchments which are more vulnerable to SS and PP losses (III).

5 Materials and Methods

5.1 Study Sites

The majority of the selected sites were located in southern Sweden (below latitude 60°), in predominantly agricultural landscapes (Figure 5). Only one soil used in the laboratory studies (I) was from a field located in a more northerly region. All of the soils form part of different national monitoring programmes with the aim of studying losses of nutrients from arable land (Ulén *et al.*, 2012c; Kyllmar *et al.*, 2006; Kirchmann, 1991). Due to the voluntary participation of farmers in the programmes, the exact location of the fields and catchments is not disclosed here. Information regarding crop management and fertilization strategies was gathered through yearly interviews with farmers from the different areas. In addition, long-term discharge and nutrient and SS concentrations at the outlet were recorded. The soils ranged from clay to loamy sand, of which clay was the dominant type (approximately 30% of all samples), followed by silty clay. A summary of selected soil properties of samples from all the studies is presented in Table 4.

The five fields selected in Paper II are part of the national monitoring programme '*Nutrient losses from arable land*' (Ulén *et al.*, 2012c), which started in 1972 with the main aim of studying the impact of different cultivation and fertilization strategies on nutrient losses under farmers' normal operations. The programme included in the national environmental monitoring is commissioned by the SEPA. The fields selected ranged in size from 5 to 28 ha and had textures varying from silty clay loam to clay. The fields had varied topography, from flat (20E) to relatively sloping (1D) or with a deep ravine (11M). They were selected from the 12 fields of the programme to study losses of SS and PP driven by overland flow. Fields 1D, 11M, 20E and 4O were four of the five fields that presented mean annual TP FWC and annual transport above the overall mean for all fields. In many of the other fields, SS and PP concentrations were very low, meaning that the main process for losses was probably leaching rather than erosion. Long-term TP and PP losses from the fields showed no significant trend, although a decreasing trend in DRP was found in fields 11M and 4O, where an internal buffer strip was placed (Ulén *et al.*, 2012c). This was surprising, given that buffer strips are meant to retain PP but have been shown to become a potential source of DRP in the long-term (Uusi-Kamppa, 2005; Daniels & Gilliam, 1996). Other studies on these fields can be found in the literature (Ulén *et al.*, 2012c; Ulén & Etana, 2010; Ulén *et al.*, 2008; Djodjic & Bergström, 2005; Ulén & Snall, 1998), as well as in yearly reports from the monitoring programme which are prepared at the Department of Soil and Environment (SLU) and published online through the publication series *Ekohydrologi* (Swedish University of Agricultural Sciences, 2013).

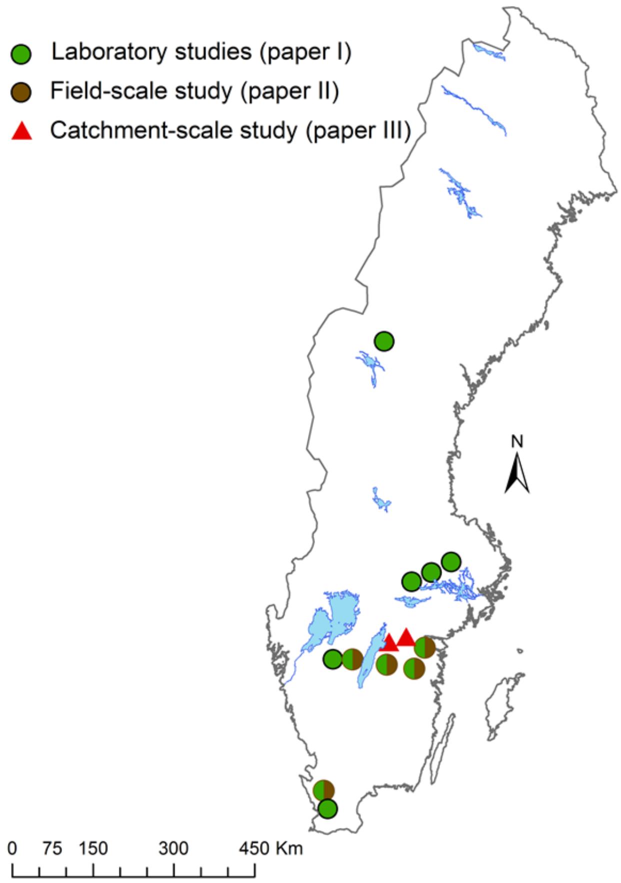

Figure 5. Location of the study sites in Sweden.

The two catchments selected in Paper III (E21 and E23) are part of the Swedish monitoring programme for agricultural land (Kyllmar *et al.*, 2006). Research on the monitoring catchments was started in the 1980s by the SEPA, with the aim of determining the relationship between different cultivation practices and water quality in runoff. Small monitoring catchments for nutrient losses are also used in many other countries such as Ireland (Fealy *et al.*, 2010), Australia (Government of Western Australia, 2014) and Norway (Deelstra *et al.*, 2011). In addition, catchment E23 is one of the three pilot catchments used in the Swedish advisory programme '*Focus on nutrients*' which was started in 2001 to reduce emissions of greenhouse gases and nutrient leaching and to ensure safe use of plant protection products (Focus on Nutrients, 2014). Catchment E21 is part of the national pesticide monitoring programme, which comprises four agricultural streams and two rivers (Adielsson & Kreuger, 2007). Both catchments are located in the same geographical region, only approx. 60 km apart, and both drain to the Baltic Sea. These catchments were selected due to their contrasting SS and PP loads observed at the outlet in spite of similar mean annual precipitation and similar discharge pattern. Long-term TP concentrations from the two catchments were below the 25th percentile (E21) and above the 75th percentile (E23) for 23 Swedish agricultural catchments (Heckrath *et al.*, 2008). Other reported mean annual TP FWC from the Nordic and Baltic countries range from 0.12 to 0.93 mg L^{-1} in Norway, 0.073 to 0.23 mg L^{-1} in Denmark, 0.11 to 0.68 mg L^{-1} in Finland and 0.04 to 0.36 mg L^{-1} in Estonia (Vagstad, 2001).

More information regarding these monitoring catchments can be found in the yearly reports of the monitoring programme (Swedish University of Agricultural Sciences, 2013) and in different studies (Ghafoor *et al.*, 2013; Ulén *et al.*, 2012d; Ulén *et al.*, 2011; Ulén *et al.*, 2004).

Table 4. *Selected soil properties of samples from the different study areas*

Study	**No. of samples**	**Clay**	**Silt**	**Sand**	**OM**	**pH**	**P-AL**	**Soil TP**
		(%)					(mg 100g^{-1})	
Laboratory	21	18-57	24-65	4-58	2.3-17.6	6.1-7.6	3.5-23	-
Mean (SD)		40(14)	43(10)	17(17)	4.7(4.6)	6.7(0.4)	8.2(6.4)	-
Fields	44	20-63	33-67	4-41	1.3-4.8	-	-	34.7-112.0
Mean (SD)		42(12)	46(9)	12(8)	2.7(0.8)	-	-	61.1(19.7)
Catchments	89	4-77	8-54	2-87	1.8-18	5.6-7.9	1.6-24.5	36.9-148.5
Mean (SD)		33(20)	27(8)	40(23)	4.0(2.6)	7.0(0.5)	8.1(4.7)	67.1(17.4)

5.2 Data Compilation

Throughout the studies, quantitative data from primary sources (Papers I-III) and secondary sources (Papers II-III) were used. Soil dispersion tests were an important component of all the studies, but they were complemented with other data as the scale of the study increased.

5.2.1 Soil sampling and analysis

In all cases, soil samples were collected as composite samples (10 sub-samples per 1 m^2 approximately) from the top 20 cm. Soil dispersion was estimated with the environmental test DESPRAL (I-III), following the procedure proposed by Withers *et al.* (2007) and with the SST test (I), following the procedure described by Udeigwe *et al.* (2007). In addition, the K factor from RUSLE (Paper I and in the present thesis essay with combined data from Papers II and III) was calculated using the following equations from Renard *et al.* (1997):

$$\mathrm{K}\left({}^{t\,h}/_{MJ\,mm}\right) = 0.0034 + 0.0405\, e^{-0.5\left(\frac{\log Dg+1.659}{0.7101}\right)^2} \quad (1)$$

$$\mathrm{Dg}\ (\mathrm{mm}) = e^{0.01\,\sum f_i \ln m_i} \quad (2)$$

where Dg is the mean geometric diameter of soil particles (mm), f_i is the primary particle size fraction (%), and m_i is the arithmetic mean of the particle size limits of that size fraction.

The alternative formula based on the nomograph proposed by Wischmeier *et al.* (1971), using texture, OM, structure class and permeability, was only used in Paper I due to the difficulties in finding information regarding permeability. In addition, hydraulic conductivity was considered to be high overall in Sweden and, thus was not used as a determining factor in the calculations.

5.2.2 Topography

The new Swedish digital elevation model (2-m grid) produced using airborne Light Detection and Ranging (LiDAR) was used to describe topographical attributes of the study sites (Papers II-III). These are of help in describing the potential sediment transport by overland flow.

LiDAR is a remote sensing technology analogous to radar. In airborne LiDAR, the distance (range) to an object is measured by the delay time between the emitted laser pulse and the detection of the reflected signal. The new national digital elevation model (DEM) was produced after a commission appointed by the Swedish government to study the effects of climate change on

Swedish society recommended an updated and more accurate elevation model to complement the previous 50-m grid model in order to better estimate the risks of climate change (Lantmäteriet, 2014). The new national grid has a mean elevation error of less than 0.5 m for a 2-m grid and is planned to be completely finished by 2015, covering all of Sweden. Some of the applications of the technology lie in vegetation mapping, flood and pollution modelling, urban planning and archaeology. The accuracy of the new model is high and details such as small ditches in agricultural fields or trails in the terrain are captured.

The topographical attributes from the DEM (*i.e.* slope, flow accumulation and unit-stream power LS) were calculated for every 2-m grid cell in ArcGis 9.3. Slope is the maximum change in z-value for each cell and describes the overland flow velocity. It can range from 0° to 90°. Flow accumulation or upslope contributing area is the drainage area of any cell and indicates the overland flow paths. Both slope and flow accumulation parameters were calculated with the Spatial Analyst tools in ArcGis. The sediment transport index (combination of slope and flow accumulation) or LS, was calculated following the formula from Mitasova (2001):

$$\text{LS}\ (\mathbf{r}) = (m+1)\left[\frac{A\ (\mathbf{r})}{22.13}\right]^{m}\left[\frac{\sin\beta\ (\mathbf{r})^{n}}{0.0896}\right] \tag{3}$$

where A (**r**) is upslope contributing area per unit width, β (**r**) is the steepest slope angle, **r** = (x, y), and m and n are parameters dependent on the type flow and set to 0.6 and 1.3, respectively.

5.2.3 Water Quality Data

Long-term water quality data from monitoring fields (II) and catchments (III) were extracted for evaluating losses from agricultural land (Swedish University of Agricultural Sciences, 2014). Agrohydrological years (*i.e.* 1 July-30 June) were selected, in the case of Paper II from July 2000 to June 2011, and in the case of Paper III from July 1994 to June 2010 (catchment E21) and from July 2002 to June 2010 (catchment E23). Although the monitoring programmes started in the 1970s and 1980s, the present study period was selected based on the start of the DRP analysis after filtration (membrane filter with 0.2 µm pore size), which has been shown to be effective in retaining most colloidal clay particles that are important for Swedish conditions (Ulén, 2004). Water samples were taken manually every fortnight, or more frequently during high flow periods. Throughout 2008-2009, the fields were equipped with data-loggers that automatically recorded flow. Automatic flow-proportional water sampling was also introduced and ran in parallel with manual sampling for two years. Flow-proportional sampling in the catchments started in 2004 (E21) and

2008 (E23) and ran in parallel with manual sampling for 6 and 4 years, respectively.

A brief evaluation of the parallel measurements in the fields showed that higher concentrations and transport were recorded when sampling was performed flow-proportionally (Figure 6). Mean TP annual transport increased in all fields, from 16% in field 11M up to 45% in field 7E. Increases in SS transport measured with flow-proportional sampling were higher in some fields (7E, 1D, 4O), whereas no apparent change was seen in other fields (11M and 20E).

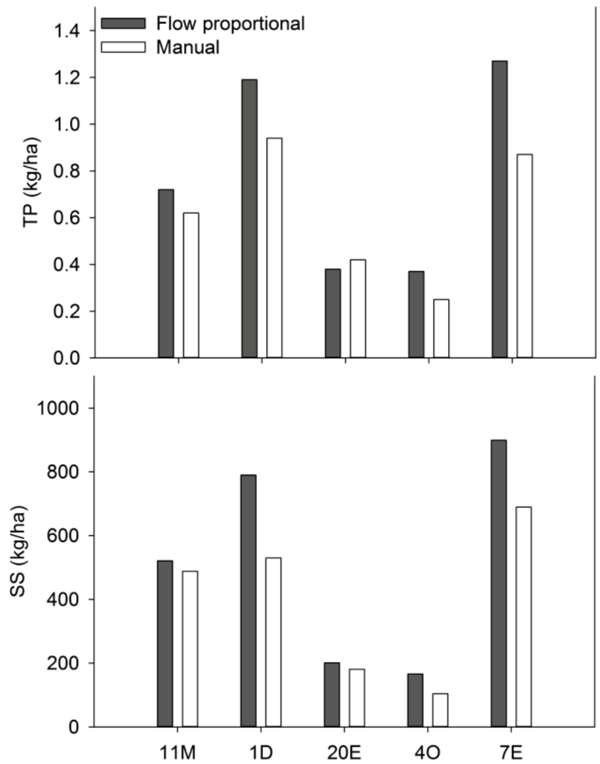

Figure 6. Transport of total phosphorus (TP) (above) and suspended solids (SS) (below) from the five monitoring fields, as measured with flow-proportional sampling and manual sampling for the agrohydrological year 2009/2010.

Comparison of loads and FWC measured with manual or flow-proportional sampling gave different results in catchments E21 and E23 (Table 5). In E21, annual loads and FWC were higher when measured with manual sampling. The opposite pattern was observed in E23 where loads and FWC were overall higher when measured with flow-proportional sampling. Large differences were found in SS, which could be expected being an event-responsive compound. Other studies have reached similar conclusions. In Norway, one study showed lower P and SS loads when sampling was performed once a week than when performed more continuously with flow-proportional sampling (Haraldsen & Stålnacke, 2006). In Finland, Rekolainen *et al.* (1991) obtained the best results by combining flow-proportional sampling during high flow events with regular interval sampling during the rest of the year.

Table 5. *Mean annual transport and mean annual flow-weighted concentrations of the different phosphorus (P) fractions, as measured with manual and flow-proportional sampling for the years 2004-2010 (E21) and 2008-2012 (E23)*

Catchment	Sampling type	TP	DRP	PP	SS	TP	DRP	SS
		(kg km^{-2})				(mg L^{-1})		
E21	Manual	14.12	6.90	4.90	3189	0.09	0.04	22.93
	Flow prop.	10.31	4.84	3.25	1506	0.06	0.03	9.69
E23	Manual	50.77	24.77	20.37	16096	0.27	0.13	84.33
	Flow prop.	55.88	24.35	25.33	23938	0.30	0.13	129.45

5.2.4 Data Analysis

In addition to the analysis carried out in each of Papers I-III, an overall analysis of the results from soil dispersion test and other erodibility estimations was performed within this thesis essay. The results are presented and discussed in section 6.1. Data from Papers II and III were combined for this purpose and are referred to as 'combined data'. A classification tree was used to identify the conditions in which the soil dispersion test DESPRAL and K_{RUSLE} yielded similar results. Each sample was assigned to one of four groups, where group 1 had low DESPRAL and low K_{RUSLE}, group 2 had low DESPRAL and high K_{RUSLE}, group 3 had high DESPRAL and high K_{RUSLE}, and group 4 had high DESPRAL and low K_{RUSLE}. The K_{RUSLE} threshold for low/high was set at 0.026, which is commonly considered to be a moderate value. The DESPRAL threshold selected was 1000 NTU (median value from combined data). The four groups were classified based on the independent variables clay, silt, sand and OM content, and clay and silt/organic C (OC) ratio, which has shown in other studies to be an important variable controlling soil physical properties (Dexter *et al.*, 2008). The 'best split option' was used to classify the groups. The analysis was carried out using JMP 10.0 software.

In addition, to calculations of K_{RUSLE}, the erodibility factor calculated in a 500 m grid in Europe and based on the LUCAS (Land Use/Cover Area frame

Statistical Survey) dataset (Panagos *et al.*, 2014) was extracted in ArcGis for every sample in Papers II and III. This database uses the algebraic approximation of the nomograph developed by Wischmeier *et al.* (1971) including texture, OM, coarse fragments, structure and permeability to calculate erodibility (Renard *et al.*, 1997).

6 Results and Discussion

6.1 Sediment and Phosphorus Mobilization Risk

6.1.1 Experimental Evaluation of Soil Dispersion Tests (I)

A comparison between the two methods that estimate soil mobilization risk by soil dispersion, DESPRAL and SST, showed that both tests were significantly correlated and that the ranking for the 10 test soils was similar. The DESPRAL test gave a smaller variation within replicate measurements than the SST test. This could be attributed to the higher dilution ratio used in SST, as higher dilution rates might introduce experimental errors (So *et al.*, 1997). In addition, the SST test uses a much smaller amount of soil which might lead to a worse representation of the properties and variations in soils. In summary, both the DESPRAL and SST proved to be simple, easy to perform tests but the former was less time consuming, in addition to being more precise and reproducible, as previously proven (Withers *et al.*, 2007).

The use of turbidity as a substitute measurement for SS in the aliquot recovered from the dispersion test (DESPRAL) was successful in the different studies (r^2 = 0.85 in Paper I, r^2 = 0.82 in paper II; r^2 = 0.89 in Paper III, at P<0.0001). In the combined data, the prediction accuracy decreased (r^2 = 0.67, P<0.0001) due to the different slopes obtained in Papers II and III. The use of turbidity as an alternative method to estimate SS concentration has been proposed, as it is a quicker and cheaper method than conventional measurements. The advantages of using turbidity as a surrogate for SS have also been seen when measured in the field, where peaks driven by storm events can be captured thanks to the possibility of measuring turbidity continuously (Grayson *et al.*, 1996). However, several concerns have risen related to the potentially confounded relationship between turbidity and SS concentrations caused by variations in particle size, particle composition and water colour (Gippel, 1995). This might explain the site-specific character of the surrogate relationships, as well as their seasonality (Jones *et al.*, 2011). In the present case, *i.e.* using turbidity as a surrogate in the aliquot from the soil dispersion test, the concerns relate to particle size distribution and not flow, as the aliquots were obtained under the same conditions.

The effect of soil storage on soil dispersion was tested with the DESPRAL test and proved to be inconsistent. Five of the 11 soil samples showed non-significant variations after 15 weeks of storage while six samples showed a significant decrease after only 8 weeks. The significant difference in variation was observed in finer-textured soils, similarly to results reported by Coote *et al.* (1988). There are only few studies in the literature addressing this effect but some have reported an increase in stability with increasing storage duration (Murer *et al.*, 1993; Kemper & Rosenau, 1986; Kemper & Koch, 1966). Such variation has been occasionally been attributed to the residual microbial activity which may cause agglomeration in air-dried soil samples (Orchard & Cook, 1983), but data regarding the cause are still too scarce to reach a definitive conclusion. In all the above studies the recommendation is to perform the analysis immediately after air-drying, which was also considered in the present thesis work for all subsequent analyses that were performed.

6.1.2 Evaluation of Sediment and Phosphorus Mobilization Risk (I-III)

A wide range of soil dispersion values were obtained from all soil samples. Turbidity in the recovered aliquot ranged from 177 NTU to 6003 NTU, while SS content ranged from 0.09 g L^{-1} to 2.8 g L^{-1}. This range is in a similar order of magnitude as obtained in other studies (Table 2). In the combined data, the mean calculated value for K_{RUSLE} (calculated using equations 1 and 2) was 0.033 t h MJ^{-1} mm^{-1} (SD 0.009), which is close to the mean soil erodibility for Europe estimated to be 0.032 t h MJ^{-1} mm (SD 0.009) (Panagos *et al.*, 2014). Mean soil erodibility calculated for Sweden in that same study was 0.025 t h MJ^{-1} mm^{-1}. Ranges for low/medium/high soil dispersion must be established if this measurement is to be used in management or modelling tools. In the same way, we now know that a K_{RUSLE} value of around 0.05 t h MJ^{-1} mm^{-1} is moderately high or, conversely, that 0.015 t h MJ^{-1} mm^{-1} is rather low. The results obtained would suggest that values lower than 550 NTU (~25th percentile) correspond to very low dispersion risk and, conversely, values higher than 1500 NTU (~75th percentile) correspond to a medium to high dispersion risk (Figure 7). At the moment, it is difficult to establish appropriate ranges for soil dispersion due to the low number of values available (N=133, from the combined data), but low and high values are proposed.

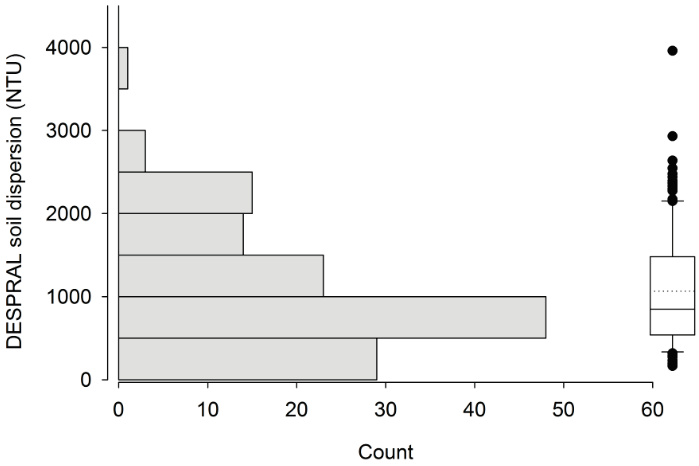

Figure 7. Distribution and boxplot of soil dispersion (NTU) for the combined data (Papers II and III). The lower and upper boundaries of the box indicate the 25th and 75th percentiles, respectively. Whiskers (error bars) above and below the box indicate the 90th and 10th percentiles, respectively. Inside the box, the dotted line indicates mean value, while the solid line indicates median value. Every outlier is represented by a filled dot.

The erodibility factor, K_{RUSLE}, and the soil dispersion (termed as $K_{DESPRAL}$ here) were significantly correlated, although the relationship could not be explained through a linear regression model, suggesting different relationships at various levels (Figure 8). The two methods were significantly correlated (r = 0.55, P<0.0001), although a significant correlation could not be found with the extracted estimated K factor from the European project (Panagos *et al.*, 2014). Results from the partitioning model showed the conditions under which different or similar results were obtained from the two methods. It was found that low values in both methods were obtained mainly for sandy soils (>57% sand). On the other hand, both methods showed high values mainly for samples with <57% sand and a silt/OC ratio >23. Finally, different results from the two were obtained for several groups of samples (*e.g.* silt/OC < 23 and clay </> 65%). The use of the clay/OC and silt/OC ratios as factors influencing the soil dispersion response was explored after the conclusions drawn by Dexter *et al.* (2008). They showed that non-complexed clay to OC is more easily dispersed in water than complexed, and that for the soils they studied a complex was formed between 10 g clay and 1 g OC. In our case, in the combined dataset, a significant but weak correlation was found between soil dispersion and clay/OC (r = 0.24, P<0.0001), and between soil dispersion and silt/OC (r = 0.45, P<0.0001).

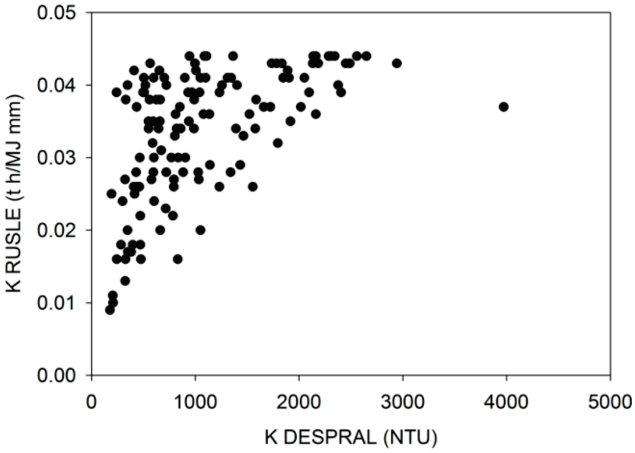

Figure 8. The K factor from RUSLE versus soil dispersion from the DESPRAL test (N=133)

Soil dispersion ($K_{DESPRAL}$) and K_{RUSLE} showed a similar ranking order for the different textural classes (five classes from the European Soil Database, ESDB[4]). In addition, $K_{DESPRAL}$ showed a wider range of values within each group. The highest mean value was obtained from the medium-fine textural class, followed by the fine textural class (Figure 9). On the other hand, the mean values were lowest in the coarse group. Similar results have been observed in other studies (Panagos *et al.*, 2014; Torri *et al.*, 1997), which agree with the general assumption that coarse particles are usually too heavy for transport, while very fine particles have usually high cohesion strength and are not thus prone to soil detachment. Accordingly, medium-fine soils have the highest risk of erosion (silt content was >55% in all samples from this group). The soil dispersion test showed a wider variation of values for medium and fine soils, whereas results from K_{RUSLE} showed a very small range for all soil samples within these groups, which might be a problem when differentiating finer-textured soils. The erodibility factor under Swedish conditions might have a greater sensitivity to total erosion risk than, for example, rainfall intensity, as saturation-excess overland flow, which is less dependent on rain intensity, usually prevails over infiltration-excess overland flow. The extracted K values from the soil erodibility European database also showed that medium-fine soils had the highest erodibility (Figure 9). However, the coarse group showed higher erodibility than expected. Moreover, the extracted estimated

[4] The definition of the five soil textural classes from the ESDB is: Coarse > 65% and clay <18 %; Medium <35% clay and <65% sand; Medium fine clay between 18 and 35% and <15% sand, or <18% clay and sand between 15 and 65%; Fine <35% clay <60%; Very fine >60% clay

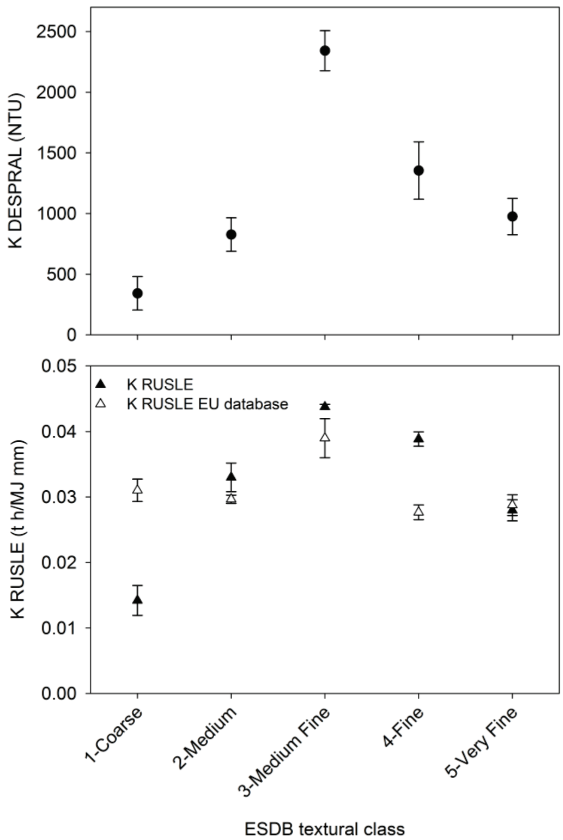

Figure 9. Soil dispersion or $K_{DESPRAL}$ (above) and K_{RUSLE} calculated and extracted from the soil erodibility European database (below), for the five soil texture classes established in the European Soil Data Base (ESDB). Data from Papers II and III (combined data). The number of observations in each group is: coarse (N=10), medium (N=53), medium-fine (N=8), fine (N=44), very fine (N=18). The bars represent 95% confidence interval.

values were very similar across most of the groups, probably due to the greater scale used for the estimation (500-m grid), with the uncertainties this entails.

Correlations of soil dispersion with selected soil properties showed that texture and OM were two important properties affecting soil dispersion. Soil P content (P-AL or soil TP) only significantly affected soil dispersion in Paper III. In Paper I, turbidity (soil dispersion) was positively significantly correlated with clay content, and negatively correlated to sand and OM content. In Paper II, turbidity was positively correlated with silt, and negatively correlated with clay and OM. In Paper III, soil dispersion was positively correlated with clay silt, pH and soil TP, and negatively correlated to sand and P-AL. The positive correlation between soil dispersion and clay content found in Papers I and III might be due to the low number of heavy clay soils included, as it is generally assumed that these soils form very stable aggregates. Other studies on soil dispersion have found correlations with OM, pH and clay (Withers *et al.*, 2007) and with clay content (Udeigwe *et al.*, 2007). Borda *et al.* (2010) established that 66% of the variance in SS is explained by clay, silt and Olsen P (soil test P which indicates plant-available P). To fully establish the properties that affect soil dispersion, a wider range of soil samples needs to be studied and other soil properties that have a demonstrated effect on soil dispersion need to be included in the analysis. Amézketa (1999) reviewed the factors controlling soil dispersion/flocculation [5], such as electrolyte concentration (EC), sodium adsorption ratio (SAR) and soil pH. Low EC and high SAR (*i.e.* high concentrations of Na^+ versus concentration of Mg^{2+} and Ca^{2+}) (Panayiotopoulos *et al.*, 2004) and higher pH values (Suarez *et al.*, 1984) lead to higher dispersion of particles.

Overall, potentially mobilized P was mainly attached to particles (94% of all TP mobilized). The amount of DRP was more strongly correlated with P-AL than was the amount of PP. However, the amount of PP was better correlated with soil TP content. The amounts of TP and DRP dispersed in the soil dispersion test ranged from 0.17-3.2 and 0.01-0.41 mg L^{-1}, respectively, which are similar to ranges obtained in other studies (Table 2). In Paper I, linear correlations showed that DRP was significantly and strongly correlated with P-AL and more weakly with the stronger extraction (P-HCl), which is closer to the soil TP content. In the combined dataset, DRP was only correlated with P-AL (r = 0.71, P<0.0001) and PP was only weakly correlated with P-AL (r = 0.27, P < 0.05). The relationship of DRP and soil test P has been reported previously and has been used to predict DRP losses (Maguire *et al.*, 2005; Sims *et al.*, 2000), while the relationship between PP and soil TP indicates that extraction with stronger acids recovers less soluble forms of P.

Phosphorus enrichment ratio (PER) showed a wide variation and was negatively related to the amount of clay content and SS dispersed. Phosphorus enrichment ratio is the enrichment of eroded particles in P content, calculated as the content of P in SS to that in soil (Ryden *et al.*, 1974). It ranged from 0.5-6.7 (if one extreme value of 10.5 was excluded), with the majority of the values being in the range of 1-3. The range is similar to ranges obtained previously in field and catchment studies (Sharpley, 1985) and in mobilization studies (Borda *et al.*, 2010; Withers *et al.*, 2007). The highest value of 10.5 was observed in catchment E21, from a sandy loam soil with very high P-AL

[5] Floculation is the stabilizing mechanism opposite to dispersion. It refers to the agglomeration of particles in to clusters or clumps of bigger size.

content, in combination with very low turbidity and amounts of SS dispersed. Phosphorus enrichment ratio exponentially increased with decreasing soil clay content, as seen in other studies (Borda *et al.*, 2011; Gburek *et al.*, 2005). The threshold was observed around 20-25% clay content, meaning that clay soils with lower clay content are more enriched in P than those with a higher clay content. Strongly significant correlations were found between PER and dispersed SS (r = -0.76, P<0.0001) and more weakly significant correlations between PER and P-AL and soil TP content (r = 0.60 and -0.55, respectively, P<0.0001). Other studies have reported that runoff and rainfall energy and soil P status have a greater effect on PER than soil physical properties (Cooke, 1988; Sharpley, 1980) and have found significant correlations only with soil TP (Withers *et al.*, 2007). While the relationship between PER and the amounts of SS dispersed was exponential, as in other studies (Borda *et al.*, 2010), the relationship with P-AL seemed less clear (Figure 10). Menzel (1980) found that on sandy textured soils, sediment concentration has less effect on PER, which might explain why enrichment was higher in catchment E21 (Paper III), where P-AL might be driving the variation. The dispersion test provides a means to calculate the enrichment of P in eroded material and to analyze the different forms of P (*e.g.* labile P attached to Fe oxides) which is important when assessing the environmental impact of eroded soils (Diaz *et al.*, 2013).

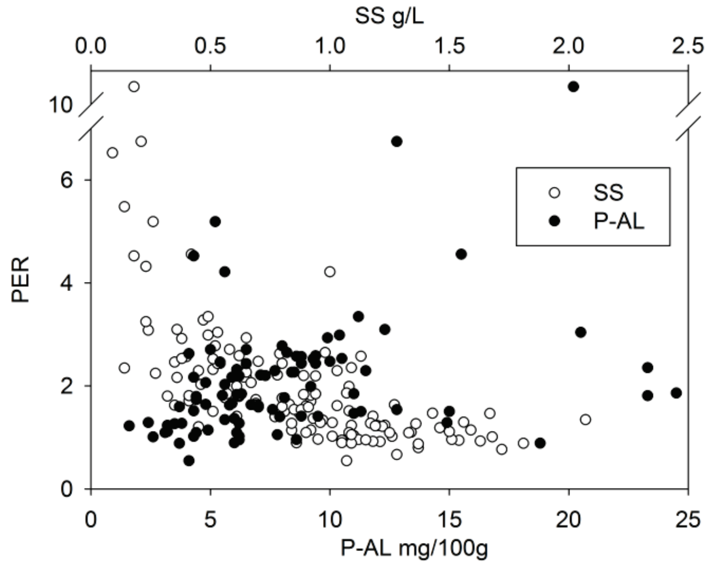

Figure 10. Variation in phosphorus enrichment ratio (PER) with suspended solids dispersed (SS) and plant-available P (P-AL) for the 133 samples in the combined data (Papers II and III).

6.2 Assessment of Field-Scale Sediment and Phosphorus Losses (II)

Sediment and P losses from the five fields were explained by source and transport favourable conditions (fields 11M and 1D), source limited conditions (fields 20E and 7E) or transport limited conditions (field 4O). The different situations are described in the following paragraphs.

The greatest long-term SS and TP losses observed occurred in fields 11M and 1D, which could be explained by a high risk of sediment and P mobilization, as established with the soil dispersion test DESPRAL, together with favoured transport conditions (high LS factor). These fields could therefore be classified as source and transport favourable. The highest soil TP content was observed in field 1D and, combined with the high mobilization and transport risks, led to similar long-term TP concentrations and loads as in field 11M, even though long-term SS loads were lower. This emphasizes the relevance of P content at the source in situations of high transport risk with regard to its potential environmental impact.

The smallest long-term SS and P losses were observed in field 4O, despite it showing the highest SS mobilization risk and medium P mobilization risk. The low losses at the outlet could be explained by adverse topography, which limits the potential transport of particles through overland flow. This field is long and flat – and thus characterized by a low LS factor – together with the presence of perennial fallow grown in the section closest to the outlet, which retained particles. This buffer area counteracted the high risk of PP mobilization observed with DESPRAL, effectively acting as a filter (Proffitt *et al.*, 1991) and retaining coarser particles, thus allowing only finer and enriched particles to reach the outlet. This was reflected in the PER, which was higher in particles from drainage water than in mobilized particles. Consequently, this field was classified as transport-limited, meaning that more material is mobilized that can be transported (Morgan, 2005).

Small SS and TP losses were observed in fields 20E and 7E, resulting from their low mobilization risk at the source (source-limited). Despite having higher P content at the source, field 20E had no effective overland transport pathways, which translated into PP loads similar to the levels found in field 4O. On the other hand, field 7E showed the highest discharge levels of all the observed fields, which translated into medium loads of SS and P, suggesting that source limitation can, to a certain degree, be surpassed by high transport risk as a predictor of PP losses. The introduction of flow-proportional sampling showed a considerable increase in TP loads from this field (Figure 7), indicating once more the importance of flow episodes driving P losses. This being said, the question of possible losses from fields in which a source limitation is accompanied by very high transport risk should be studied further. For instance, recent research on a field outside the monitoring programme indicated the possibility that mobilization risk could be surpassed by high transport capacity (Djodjic & Villa, unpublished) as a determining factor. The results from that study suggested that the control exerted by topography over hydrology and overland flow concentration may prove to be more important than susceptibility to mobilization.

The present results show the importance of identifying source and transport-prone fields for the correct placement of suitable mitigation measures. For instance, mitigation measures intended to control P losses at the source could be especially effective in fields 11M and 1D. Such measures could consist of application of lime products ($CaCO_3$, CaO, $Ca(OH)_2$) or gypsum ($CaSO_4 \cdot 2H_2O$) to improve soil structure. Phosphogypsum application has been shown to considerably decrease soil losses from dispersive soils and moderately decrease losses from nondispersive soils (Ben-Hur *et al.*, 1992). Gypsum application has shown the potential to decrease PP losses from clay soils (Jaakkola *et al.*, 2012) by increasing particle aggregation as well as decreasing DP losses by favouring P adsorption with the increase on the ionic strength. Measures aimed at controlling transport could be useful in fields such as 7E, where transport capacity in the form of discharge might be driving losses. Of these measures, buffer strips are one of the most common. A buffer strip 10 m wide can reduce up to 95% of the total PP load to streams, as well as increasing the diversity of flora and fauna (Vought *et al.*, 1995).

Differences in soil dispersion between fields were statistically significant, despite of the variations within fields. Overall, the values in each field were spread around the same percentile range and, thus, their classification in terms of lower or higher mobilization risk did not change. The variation of soil dispersion within fields was greater for the two largest fields, 11M and 7E. Given the design of the study, it was difficult to isolate the different factors that might be driving variability within fields, such as soil texture, OM content and land-use history. In field 11M, the values that stood out from the rest (938 and 3972 NTU) were obtained at points located along the same slope (approx. 25 m apart from each other). Although there were no differences in soil texture, there still was a unit difference in OM which has been proven to be enough to generate significant decreases in erodibility in other studies (*e.g.* Fullen, 1998). The highest (discordant) value observed in field 7E may be due to that sample showing the highest silt (54%) and a low sand content (8%) combined with the lowest OM content in the whole field (2.1%).

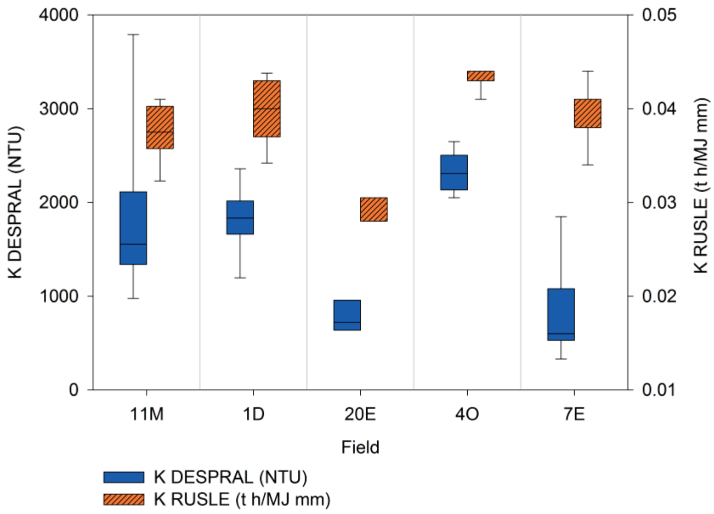

Figure 11. Soil dispersion ($K_{DESPRAL}$) and K_{RUSLE} in fields from Paper II.

The use of K_{RUSLE} to estimate erodibility for the fields gave a similar picture in terms of higher and lower erodibility, with the exception of field 7E (Figure 11), which showed low soil dispersion risk whereas the calculated K_{RUSLE} value was high. If this is true, it is difficult to explain the rather low long-term SS observed at the outlet given such high discharge. Furthermore, given that the soil dispersion test DESPRAL is designed to represent the P mobilization risk under adverse conditions, it will usually tend to overestimate rather than underestimate the risk of detachment. All of this would suggest that K_{RUSLE} may not be properly calibrated for Swedish conditions, as pointed out in Paper I.

6.3 Ranking Areas Vulnerable to Sediment and Phosphorus Losses (III)

Comparison of the SS and PP losses in the two catchments suggested that factors governing transport of SS and PP exert greater control over losses at the catchment scale in spite of lower plant-available P values in soils across the catchments, which is in line with findings in other studies (Shore *et al.*, 2014; Jordan *et al.*, 2012; Buda *et al.*, 2009). Flow accumulation was similar in both catchments but the LS factor was higher in E23. In addition, mobilization risk was significantly higher in catchment E23. The co-occurrence of higher LS and higher mobilization risk lead to higher SS and P loads in the outlet of E23 than

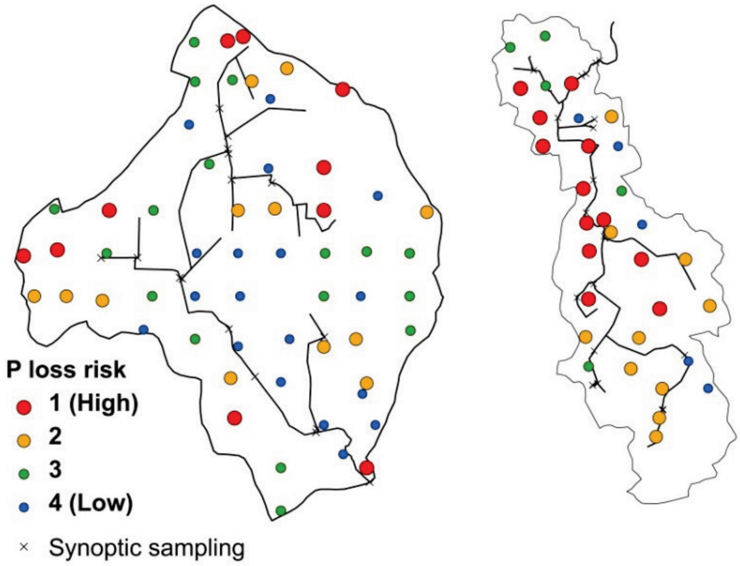

Figure 12. Risk of P losses in the agricultural catchments E21 (left) and E23 (right). Higher risk areas are represented by a larger red dots, while low risk areas are represented by small blue dots.

in that of E21, even though the overall soil P content was lower. A similar differentiation to the one between the catchments was observed between the two halves of catchment E23, where the north half had higher LS and mobilization values, leading to higher P concentrations in water across this section.

Ranking of fields across both catchments translated into a higher proportion of fields identified within the top 50% in risk being located in catchment E23 (Figure 12). Most of the high risk fields in this catchment were located near the main stream, with connected transfer pathways to the stream. The ranking was established prioritizing transport risk before mobilization and source risks, considering the results from the catchment comparison. The results from this study support the notion that implementation of mitigation measures should be prioritized in the areas showing the highest risk of P losses. The subsequent challenge for future research would then be to set the different thresholds for high or low risk areas. It is a widely accepted fact that the majority (~80%) of P losses originate from a small proportion of the catchment area (~20%) (Sharpley *et al.*, 2009). For instance, Tim *et al.* (1992) identified high risk source areas of soil erosion, sediment and P pollution in 15, 16 and 21% of a watershed area, respectively. Ghebremichael *et al.* (2010) found that 80% of

TP losses in a 71 km^2 basin originated from only 24% of the watershed. Busteed *et al.* (2009) found that 85% of the pollutant load came from only 10% of a 2400 km^2 basin.

As an example, buffer strips to mitigate PP losses were located along the main stream in E21, where according to the present study there was a low risk of PP losses, while almost no buffer strips were located in catchment E23, where the highest risks of PP losses were identified. The case of these two catchments can thus serve as an illustration of how resources should be allocated in a more balanced manner and in consistency with proper assessments of the risk of PP losses. Selecting the most vulnerable half of the catchments instead of focusing on identifying only the 20% most vulnerable areas (*i.e.* CSAs) within the catchments could be a good compromise in terms of effective management of the available resources for mitigation.

7 Evaluation of the Methods

The methods used in this thesis were intended to estimate losses of P by erosion and therefore only give a partial view of the broader problem of P losses in general. Losses of P by other processes such as leaching (*e.g.* losses occurring in sandy soils with low sorption capacity) are not fully represented. In such cases, P is mobilized by solubilization rather than by physical detachment of soil particles, and plant-available P has greater importance than topographical attributes. The methods presented here already use P-AL (plant-available P) as an indicator of source risk and could therefore be complemented with P sorption capacity (*e.g.* measures of Al-AL and Fe-AL), which has been proven to play an important role in estimating losses of P by leaching.

In order to perform the soil dispersion test, as with every other soil test performed in a laboratory, the soils were taken away from their natural environment and some of their properties may therefore have been subjected to slight modification, namely in the form of moisture content or structure variation. Thus, the tests will never fully reproduce the natural conditions of the soil, but this is largely compensated for by the fact that they are easier to perform and reproduce than erodibility measurements in the field. This also applies to many aggregate stability tests. Field-scale measurements of erodibility require a great amount of resources and time and are not feasible if the intention is to perform risk assessments at catchment or regional level.

All this being said, performing the soil dispersion tests also demands some use of resources for soil sampling and analysis. A cost-effective alternative to these tests could be the development of pedotransfer functions, for which a higher number of samples with a wider range of properties (*e.g.* pH, OM, EC, Al and Fe oxides) would need to be tested. Yet another alternative could be to limit soil sampling to previously targeted risk transport areas detected through GIS calculations with high-resolution DEM.

The possible variation in soil dispersion due to sample storage duration may be a logistic constraint when using these tests, as samples would have to be analyzed immediately after air-drying. The initial intention of using stored soils from different surveys located outside monitoring programmes had to be revised in light of the results from Paper I. This makes it difficult to scale up the results obtained in the study areas to regional and national level. More research on the variation of soil dispersion with sample storage duration would need to be performed to fully establish the test as a routine tool in risk assessments. Moreover, attention should be given to the variation in soil dispersion during different parts of the year. The values of soil dispersion in the present work were used as single values, although they might vary depending on the time of the year at which they were taken. In the thesis, samples were taken at the same time for each of the studies.

The use of fields and catchments as study units allows the analysis to get closer to the impact point. However, this also means that there is an increase in uncertainty stemming from moving up in scale, which is added to the greater complexity of different processes, sinks and sources interacting in the landscape. In addition, results are usually more difficult to reproduce than those obtained at smaller scales. This might be especially true for the results in Paper II, which were fundamentally descriptive and where confounding factors might arise if applied in other fields. For this reason, there is a clear need to develop the proposed methodology into a quantitative tool or model that would allow validation of the results obtained. A sensitivity analysis of the different parameters affecting SS losses would also be useful to verify whether the results from the soil dispersion test would substantially improve the model.

Using a simplified method on a larger scale would obviously not allow all the processes and mechanisms driving the observed P losses to be fully explained, but it could be a useful way of screening the risks and prioritizing implementation of mitigation measures and of meeting the demands for time- and cost-effective methods from the regulatory authorities.

Finally, the classification of risks within the two catchments (Paper III) should be interpreted with caution. The results should be viewed as relative indications rather than in absolute terms, as they were established comparing fields from the two catchments and presented as ranges in percentages. The ranking method used would also need to be validated in smaller subcatchments where the CSAs are controlled by soil and land management, as well as topography. The use of intensive synoptic sampling would be especially useful, as the results could not be properly validated due to the small variability across the catchments, which was probably related to the low number of episodes sampled.

8 Conclusions

In summary, this thesis assessed the risk of erosion and losses of PP for 10 soils, five fields and two catchments in agricultural areas in Sweden. The results could serve as the basis for proposing alternative methodologies to identify catchments/fields vulnerable to soil erosion and losses of PP to those currently in use. The main conclusions that can be drawn are:

- Comparison of soil dispersion tests showed that the DESPRAL test was more precise and less time-consuming than the SST test. Both tests provided an alternative to erodibility measurements and aggregate indexes to estimate initial sediment mobilization risk from agricultural soils, introducing wider ranges to differentiate soils within textural groups and especially within finer-textured soils, which are the most sensitive to mobilization.
- The soil dispersion test DESPRAL provided the means to estimate the different P fractions mobilized, as well as the enrichment in P of the eroded material (PER). The latter showed a significant negative correlation with the amount of sediment dispersed and was positively, although more weakly, correlated with the P content in the soil. In addition, the recovered aliquot could potentially be used to estimate the environmental impact of the eroded material by analyzing the release potential of the eroded P.
- Soil dispersion varied significantly with soil storage duration for some samples, while for others it remained stable over time. A probable cause of the variation is soil texture, as finer-textured soils showed most variation. More research needs to be done to clarify whether this variation is stable across all types of soils. The simplest short-term solution to this problem would be to analyze the samples directly after drying.
- Long-term field-scale losses were explained by a source and transport favourable situation in two fields, transport limitation in one field and a source limitation situation in two other fields. These types of qualitative

assessments are important for risk screening and are useful for proper placement of suitable mitigation measures.

- Losses of PP in the two catchments were driven by the presence of effective transport pathways, identified with the LS parameter, rather than by P accumulation in soils, suggesting that soil test P could not be used on its own to predict PP losses. However, P content in the soil could be expected to have greater importance in the case of two catchments with similar transport risk.
- A ranking scheme was proposed for the identification of vulnerable areas for PP losses from two catchments based on three indicators, of which transport risk (LS factor) was favoured over mobilization (soil dispersion) and source (soil test P, P-AL) risks, in that order. The results support the idea that mitigation measures should be prioritized in detected high risk areas for PP losses. Identification of the most vulnerable fields in a catchment would help to prioritize the allocation of mitigation measures, as no such prioritization is currently being made in Sweden.

9 Resumen (Summary in Spanish)

Las pérdidas de fósforo (P) desde tierras agrarias son uno de los principales factores que contribuyen al problema de la eutrofización de masas de agua. Una parte importante de estas pérdidas se produce debido a procesos de erosión en los que el P asociado a sedimentos (P particulado) es transportado a través de flujo superficial. Las áreas más vulnerables a dichas pérdidas deben ser identificadas adecuadamente a fin de establecer las medidas de mitigación correspondientes a nivel de campo y cuenca. El objetivo principal de esta tesis es contribuir al desarrollo de metodología para la efectiva identificación de suelos y campos vulnerables a pérdidas de P particulado mediante el uso de herramientas de fácil implementación destinadas a autoridades competentes y agricultores.

En primer lugar, se realizó un estudio metodológico a nivel de laboratorio de dos métodos de análisis de dispersión del suelo (DESPRAL y SST) para la estimación del riesgo inicial de movilización de sedimento y P. A continuación se estudiaron las pérdidas de P y sedimento en cinco campos pertenecientes al programa sueco de monitorización de campos agrarios, con el objetivo de estudiar su posible clasificación según los principales factores que determinan las pérdidas por erosión. Para ello, se propuso el uso de indicadores pertenecientes a las distintas etapas del continuo de transferencia del P: un indicador de fuente (contenido de fósforo en el suelo), un indicador del riesgo de movilización (vulnerabilidad del suelo a la dispersión) y un indicador del riesgo de transporte (el factor topográfico LS, “length-slope”). Este último indicador está compuesto por los atributos topográficos ángulo de pendiente y acumulación de flujo, los cuales fueron calculados mediante el sistema de información geográfica ArcGIS, a partir del modelo digital de elevación de alta resolución LiDAR de Suecia. Por último, todas las consideraciones metodológicas y herramientas de análisis estudiadas se aplicaron a dos cuencas de monitorización para identificar áreas de riesgo de pérdidas de P ("critical source areas").

De los dos métodos de análisis de dispersión evaluados, DESPRAL mostró una mayor precisión y un menor tiempo de ejecución, resultados que se añaden a su ya probada reproducibilidad y válida calibración. Además, en comparación con otros métodos, se comprobó que esta herramienta de análisis permite obtener rangos de valores de dispersión más amplios para cada clase de textura, lo cual facilita la diferenciación de los suelos en base a su riesgo de movilización. Esto último es especialmente relevante en el caso de suelos arcillosos y limosos, que son generalmente más vulnerables a los procesos de erosión. Por su parte, el estudio de los cinco campos de monitorización evidenció la importancia de identificar las condiciones en fuente y de transporte a la hora de seleccionar medidas de mitigación de pérdidas adecuadas, un paso fundamental a la hora de implementar las medidas de mitigación apropiadas a nivel de campo. Finalmente, la evaluación de dos cuencas de monitorización puso en evidencia que los factores de riesgo de transporte y movilización tienen un mayor efecto sobre las pérdidas de P por erosión que la acumulación de P en el suelo. En línea con esto, se comprobó que las pérdidas de P eran mayores en la cuenca con mayor riesgo de generación de flujo superficial a pesar de que el contenido de P en el suelo era significativamente menor y, en consecuencia, la identificación de los campos más vulnerables dentro de las cuencas se realizó priorizando el riesgo de transporte sobre la movilización y, a su vez, de esta sobre el contenido en P de los suelos. Esta clasificación de los campos de las dos cuencas dio pie a la identificación de un mayor número de campos de alto riesgo en la cuenca con mayor número de vías de transporte pronunciadas.

Esta tesis propone metodología a través de la cual datos fácilmente obtenidos puedan ser usados en un análisis de riesgo para la identificación de campos y cuencas vulnerables a las pérdidas de P particulado. El conocimiento adquirido es un buen punto de partida para mejorar estos análisis, al incorporar los medios necesarios para la priorización de las distintas medidas de mitigación, algo que actualmente no se lleva a cabo en Suecia.

References

Adielsson, S. & Kreuger, J. (2007). Environmental Monitoring of Pesticides in Sweden. *TemaNord,* 514, pp. 65-74.

Alström, K. & Åkerman, A.B. (1992). Contemporary Soil Erosion Rates on Arable Land in Southern Sweden. *Geografiska Annaler. Series A, Physical Geography,* 74(2/3), pp. 101-108.

Alström, K. & Bergman, A. (1990). *Water Erosion on Arable Land in Southern Sweden*. In: Boardman, J., Foster, I.D.L. & Dearing, J.A. (eds) Soil Erosion on Agricultural Land. Chichester: John Wiley & Sons Ltd.

Amézketa, E. (1999). Soil aggregate stability: A review. *Journal of sustainable agriculture,* 14, pp. 83-151.

Andersson, H., Bergström, L., Djodjic, F., Ulén, B. & Kirchmann, H. (2013). Topsoil and Subsoil Properties Influence Phosphorus Leaching from Four Agricultural Soils. *Journal of Environmental Quality,* 42(2), pp. 455-463.

Ballantine, D., Walling, D.E. & Leeks, G.J.L. (2009). Mobilisation and Transport of Sediment-Associated Phosphorus by Surface Runoff. *Water Air and Soil Pollution,* 196(1-4), pp. 311-320.

Barthès, B. & Roose, E. (2002). Aggregate stability as an indicator of soil susceptibility to runoff and erosion; validation at several levels. *CATENA,* 47(2), pp. 133-149.

Ben-Hur, M., Shainberg, I., Stern, R. & van der Merwe, A. (1992). Slope and gypsum effects on infiltration and erodibility of dispersive and nondispersive soils. *Soil Science Society of America Journal,* 56(5), pp. 1571-1576.

Bergström, L., Djodjic, F., Kirchmann, H., Nilsson, I. & Ulén, B. (2007). *Phosphorus from Farmland to Water - Status, Flows and Preventive Measures in a Nordic Perspective*. Report Food 21, 4. SLU, Uppsala, Sweden. ISBN: 978-91-576-7232-2.

Beven, K., Heathwaite, L., Haygarth, P., Walling, D., Brazier, R. & Withers, P. (2005). On the concept of delivery of sediment and nutrients to stream channels. *Hydrological Processes,* 19(2), pp. 551-556.

Beven, K. & Kirkby, M.J. (1979). A physically based, variable contributing area model of basin hydrology/Un modèle à base physique de zone d'appel

variable de l'hydrologie du bassin versant. *Hydrological Sciences Journal,* 24(1), pp. 43-69.
Boesch, D., Hecky, R., O'Melia, C., Schindler, D. & Seitzinger, S. (2006). *Eutrophication of the Swedish seas*. Report 5509. Swedish Environmental Protection Agency, Stockholm, Sweden. ISBN: 91-620-5509-7
Boomer, K.B., Weller, D.E. & Jordan, T.E. (2008). Empirical models based on the universal soil loss equation fail to predict sediment discharges from Chesapeake bay catchments. *Journal of Environmental Quality,* 37(1), pp. 79-89.
Borda, T., Celi, L., Zavattaro, L., Sacco, D. & Barberis, E. (2011). Effect of agronomic management on risk of suspended solids and phosphorus losses from soil to waters. *Journal of Soils and Sediments,* 11(3), pp. 440-451.
Borda, T., Withers, P.J.A., Sacco, D., Zavattaro, L. & Barberis, E. (2010). Predicting mobilization of suspended sediments and phosphorus from soil properties: a case study from the north west Po valley, Piemonte, Italy. *Soil Use and Management,* 26(3), pp. 310-319.
Boström, B., Persson, G. & Broberg, B. (1988). Bioavailability of different phosphorus forms in freshwater systems. *Hydrobiologia,* 170(1), pp. 133-155.
Boynton, W.R., Garber, J.H., Summers, R. & Kemp, W.M. (1995). Inputs, transformations, and transport of nitrogen and phosphorus in Chesapeake Bay and selected tributaries. *Estuaries,* 18(1B), pp. 285-314.
Bryan, R.B. (1968). The development, use and efficiency of indices of soil erodibility. *Geoderma,* 2(1), pp. 5-26.
Buda, A.R., Kleinman, P.J.A., Srinivasan, M.S., Bryant, R.B. & Feyereisen, G.W. (2009). Effects of hydrology and field management on phosphorus transport in surface runoff. *Journal of Environmental Quality,* 38(6), pp. 2273-2284.
Busteed, P.R., Storm, D.E., White, M.J. & Stoodley, S.H. (2009). Using SWAT to target critical source sediment and phosphorus areas in the Wister Lake Basin, USA. *American Journal of Environmental Sciences,* 5(2), p. 156.
Carpenter, S.R., Caraco, N.F., Correll, D.L., Howarth, R.W., Sharpley, A.N. & Smith, V.H. (1998). Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications,* 8(3), pp. 559-568.
Carstensen, J., Andersen, J.H., Gustafsson, B.G. & Conley, D.J. (2014). Deoxygenation of the Baltic Sea during the last century. *Proceedings of the National Academy of Sciences,* 111(15), pp. 5628-5633.
Cerdan, O., Poesen, J., Govers, G., Saby, N., Le Bissonnais, Y., Gobin, A., Vacca, A., Quinton, J., Auerswald, K. & Klik, A. (2006). *Sheet and rill erosion*. In: Poesen, J.B.a.J. (ed.) Soil Erosion in Europe. Chichester: John Wiley & Sons, Ltd, pp. 501-513. ISBN: 13 978 0-470-85910-0.
Cooke, J.G. (1988). Sources and sinks of nutrients in a New Zealand hill pasture catchment II. Phosphorus. *Hydrological Processes,* 2(2), pp. 123-133.
Coote, D.R., Malcolm-McGovern, C.A., Wall, G.J., Dickinson, W.T. & Rudra, R.P. (1988). Seasonal-variation of erodibility indexes based on shear-

strength and aggregate stability in some Ontario soils. *Canadian Journal of Soil Science,* 68(2), pp. 405-416.
Cordell, D., Drangert, J.-O. & White, S. (2009). The story of phosphorus: Global food security and food for thought. *Global Environmental Change-Human and Policy Dimensions,* 19(2), pp. 292-305.
Czyz, E.A., Dexter, A.R. & Terelak, H. (2002). Content of readily-dispersible clay in the arable layer of some Polish soils. In: Pagliai, M. & Jones, R. (eds) *International Conference on Sustainable Soil Management for Environmental Protection*, Florence. pp. 115-124.
Daniels, R.B. & Gilliam, J.W. (1996). Sediment and chemical load reduction by grass and riparian filters. *Soil Science Society of America Journal,* 60(1), pp. 246-251.
Deelstra, J., Oygarden, L., Blankenberg, A.G.B. & Eggestad, H.O. (2011). Climate change and runoff from agricultural catchments in Norway. *International Journal of Climate Change Strategies and Management,* 3(4), pp. 345-360.
DeWalle, D.R. & Rango, A. (2008). *Principles of snow hydrology*. Cambridge: Cambridge University Press. ISBN: 978-0-521-82362-3.
Dexter, A. (1988). Advances in characterization of soil structure. *Soil and Tillage Research,* 11(3), pp. 199-238.
Dexter, A., Richard, G., Arrouays, D., Czyż, E., Jolivet, C. & Duval, O. (2008). Complexed organic matter controls soil physical properties. *Geoderma,* 144(3), pp. 620-627.
Dexter, A.R. & Kroesbergen, B. (1985). Methodology for determination of tensile strength of soil aggregates. *Journal of Agricultural Engineering Research,* 31(2), pp. 139-147.
Diaz, I., Carmen del Campillo, M., Barron, V., Torrent, J. & Delgado, A. (2013). Phosphorus losses from two representative small catchments in the Mediterranean part of Spain. *Journal of Soils and Sediments,* 13(8), pp. 1369-1377.
Djodjic, F. & Bergström, L. (2005). Phosphorus losses from arable fields in Sweden—effects of field-specific factors and long-term trends. *Environmental Monitoring and Assessment,* 102(1), pp. 103-117.
Djodjic, F., Hellgren, S., Futter, M. & Brandt, M. (2012). *Suspenderat material - transporter och betydelsen för andra vattenkvalitetsparametrar*. SMED, Norrköping.
Djodjic, F. & Spännar, M. (2012). Identification of critical source areas for erosion and phosphorus losses in small agricultural catchment in central Sweden. *Acta Agriculturae Scandinavica, Section B - Soil & Plant Science,* 62(sup2), pp. 229-240.
Djodjic, F. & Villa, A. (2014). Distributed, high-resolution modeling of critical source areas for erosion and phosphorus losses. Submitted manuscript.
Dorioz, J.M., Pelletier, J.P. & Benoit, P. (1998). Proprietes physico-chimiques et biodisponibilite potentielle du phosphore particulaire selon l'origine des sediments dans un bassin versant du Lac Leman (France). *Water Research,* 32(2), pp. 275-286.

Dunne, T. & Black, R.D. (1970). An experimental investigation of runoff production in permeable soils. *Water Resources Research,* 6(2), pp. 478-490.

Ekholm, P. (1994). Bioavailability of Phosphorus in Agriculturally Loaded Rivers in Southern Finland. *Hydrobiologia,* 287(2), pp. 179-194.

Ekologgruppen (2012). *Test av tre nordiska fosfor index för förhållanden i svensk jordbruksmark*. Ekologgruppen, Landskrona, Sweden

Ellison, M.E. & Brett, M.T. (2006). Particulate phosphorus bioavailability as a function of stream flow and land cover. *Water Research,* 40(6), pp. 1258-1268.

Eriksson, J., Andersson, A. & Andersson, R. (1999). *Åkermarkens matjordstyper – Texture of agricultural topsoils in Sweden*. (In Swedish, English summary). Report 4955. Swedish Environmental Protection Agency, Stockholm, Sweden

Etana, A., Rydberg, T. & Arvidsson, J. (2009). Readily dispersible clay and particle transport in five Swedish soils under long-term shallow tillage and mouldboard ploughing. *Soil and Tillage Research,* 106(1), pp. 79-84.

European Environment Agency (2005). *Source apportionment of nitrogen and phosphorus inputs into the aquatic environment*. EEA Report No 7/2005. European Environmental Agency, Copenhague, Denmark.

Fealy, R.M., Buckley, C., Mechan, S., Melland, A., Mellander, P.E., Shortle, G., Wall, D. & Jordan, P. (2010). The Irish Agricultural Catchments Programme: catchment selection using spatial multi-criteria decision analysis. *Soil Use and Management,* 26(3), pp. 225-236.

Focus on Nutrients (2014). *greppa näringen – rådgivning lantbruk och Miljö tjänar på*. Available from: http://www.greppa.nu/ [2014-06-26]

Foster, G., McCool, D., Renard, K. & Moldenhauer, W. (1981). Conversion of the universal soil loss equation to SI metric units. *Journal of Soil and Water Conservation,* 36(6), pp. 355-359.

Fullen, M.A. (1998). Effects of grass ley set-aside on runoff, erosion and organic matter levels in sandy soils in east Shropshire, UK. *Soil and Tillage Research,* 46(1), pp. 41-49.

Galzki, J.C., Birr, A.S. & Mulla, D.J. (2011). Identifying critical agricultural areas with three-meter LiDAR elevation data for precision conservation. *Journal of Soil and Water Conservation,* 66(6), pp. 423-430.

Gburek, W.J., Barberis, E., Haygarth, P.M., Kronvang, B. & Stamm, C. (2005). *Phosphorus mobility in the landscape*. In: Sims, J.T. & Sharpley, A.N. (eds.) Phosphorus: agriculture and the environment. Madison: American Society of Agronomy, pp. 941-979. ISBN: 0-89118-157-1.

Gerzabek, M.H., Kirchmann, H. & Pichlmayer, F. (1995). Response of soil aggregate stability to manure amendments in the Ultuna long-term soil organic matter experiment. *Zeitschrift Fur Pflanzenernahrung Und Bodenkunde,* 158(3), pp. 257-260.

Ghafoor, A., Koestel, J., Larsbo, M., Moeys, J. & Jarvis, N. (2013). Soil properties and susceptibility to preferential solute transport in tilled topsoil at the catchment scale. *Journal of Hydrology,* 492(0), pp. 190-199.

Ghebremichael, L., Veith, T. & Watzin, M. (2010). Determination of critical source areas for phosphorus loss: Lake Champlain basin, Vermont. *Transactions of the Asabe,* 53(5), pp. 1595-1604.
Gippel, C.J. (1995). Potential of turbidity monitoring for measuring the transport of suspended solids in streams. *Hydrological Processes,* 9(1), pp. 83-97.
Government of Western Australia. *Annual Sub-catchment Nutrient Reports.* Available from: http://www.swanrivertrust.wa.gov.au/the-river-system/evaluation-and-reporting/catchment-water-quality-monitoring-and-reporting/annual-sub-catchment-nutrient-reports [June 8, 2014].
Grayson, R., Finlayson, B.L., Gippel, C. & Hart, B. (1996). The potential of field turbidity measurements for the computation of total phosphorus and suspended solids loads. *Journal of Environmental Management,* 47(3), pp. 257-267.
Haraldsen, T.K. & Stålnacke, P. (2006). Methods for water quality sampling and load estimation in monitoring of Norwegian agricultural catchments. *Nordic Hydrology,* 37(1), pp. 81-92.
Harris, S. (1971). Index of structure: evaluation of a modified method of determining aggregate stability. *Geoderma,* 6(3), pp. 155-162.
Haygarth, P.M., Condron, L.M., Heathwaite, A.L., Turner, B.L. & Harris, G.P. (2005). The phosphorus transfer continuum: Linking source to impact with an interdisciplinary and multi-scaled approach. *Science of The Total Environment,* 344(1-3), pp. 5-14.
Haygarth, P.M. & Sharpley, A.N. (2000). Terminology for phosphorus transfer. *Journal of Environmental Quality,* 29(1), pp. 10-15.
Heathwaite, A.L., Quinn, P.F. & Hewett, C.J.M. (2005). Modelling and managing critical source areas of diffuse pollution from agricultural land using flow connectivity simulation. *Journal of Hydrology,* 304(1-4), pp. 446-461.
Heckrath, G., Bechmann, M., Ekholm, P., Ulén, B. & Djodjic, F. (2008). Review of indexing tools for identifying high risk areas of phosphorus loss in Nordic catchments. *Jounal of Hydrology,* 349, pp. 68-87.
Horton, R.E. (1933). The role of infiltration in the hydrologic cycle. *Transactions, American Geophysical Union,* 14, pp. 446-460.
Howarth, R.W. & Marino, R. (2006). Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: evolving views over three decades. *Limnology and Oceanography,* 51(1), pp. 364-376.
Jaakkola, E., Tattari, S., Ekholm, P., Pietola, L., Posch, M. & Bärlund, I. (2012). Simulated effects of gypsum amendment on phosphorus losses from agricultural soils. *Agricultural and Food Science,* 21(3), pp. 292-306.
Johannesson, K.M., Andersson, J.L. & Tonderski, K.S. (2011). Efficiency of a constructed wetland for retention of sediment-associated phosphorus. *Hydrobiologia,* 674(1), pp. 179-190.
Jones, A.S., Stevens, D.K., Horsburgh, J.S. & Mesner, N.O. (2011). Surrogate Measures for Providing High Frequency Estimates of Total Suspended Solids and Total Phosphorus Concentrations. *Journal of the American Water Resources Association,* 47(2), pp. 239-253.
Jordan, P., Melland, A.R., Mellander, P.E., Shortle, G. & Wall, D. (2012). The seasonality of phosphorus transfers from land to water: Implications for

trophic impacts and policy evaluation. *Science of The Total Environment,* 434, pp. 101-109.
Kemper, W.D. & Koch, E.J. (1966). *Aggregate stability of soils from Western United States and Canada*. Washington, DC: United States Department of Agriculture.
Kemper, W.D. & Rosenau, R.C. (1986). *Aggregate stability and size distribution*. In: Klute, A. (ed.) Methods of Soil Analysis. Part 1. Physical and Mineralogical Methods. Madison: American Society of Agronomy, pp. 425-442
Kinnell, P.I.A. (2004). Sediment delivery ratios: a misaligned approach to determining sediment delivery from hillslopes. *Hydrological Processes,* 18(16), pp. 3191-3194.
Kinnell, P.I.A. (2010). Event soil loss, runoff and the Universal Soil Loss Equation family of models: A review. *Journal of Hydrology,* 385(1–4), pp. 384-397.
Kirchmann, H. (1991). Properties and classification of soils of the Swedish long-term fertility experiments .1. Sites at Fors and Kungsängen. *Acta Agriculturae Scandinavica,* 41(3), pp. 227-242.
Kirchmann, H., Schon, M., Borjesson, G., Hamner, K. & Katterer, T. (2013). Properties of soils in the Swedish long-term fertility experiments: VII. Changes in topsoil and upper subsoil at orja and Fors after 50 years of nitrogen fertilization and manure application. *Acta Agriculturae Scandinavica Section B-Soil and Plant Science,* 63(1), pp. 25-36.
Kronvang, B., Vagstad, N., Behrendt, H., Bogestrand, J. & Larsen, S.E. (2007). Phosphorus losses at the catchment scale within Europe: an overview. *Soil Use and Management,* 23, pp. 104-116.
Kyllmar, K., Carlsson, C., Gustafson, A., Ulén, B. & Johnsson, H. (2006). Nutrient discharge from small agricultural catchments in Sweden: Characterisation and trends. *Agriculture, Ecosystems & Environment,* 115(1-4), pp. 15-26.
Kynkäänniemi, P., Ulén, B., Torstensson, G. & Tonderski, K.S. (2013). Phosphorus Retention in a Newly Constructed Wetland Receiving Agricultural Tile Drainage Water. *Journal of Environmental Quality,* 42(2), pp. 596-605.
Lantmäteriet (2014). *GSD-Höjddata, grid 2+.* Available from: http://www.lantmateriet.se/Kartor-och-geografisk-information/Hojddata/GSD-Hojddata-grid-2/ [2014-06-10]
Larsson, M. (2011). *Identifying risk areas for phosphorous losses due to surface runoff- a method development with GIS.* Master Thesis. Uppsala: Swedish University of Agricultural Sciences.
Lemunyon, J.L. & Gilbert, R.G. (1993). The concept and need for a phosphorus assessment tool. *Journal of Production Agriculture,* 6(4), p. 449.
Liu, J., Khalaf, R., Ulen, B. & Bergkvist, G. (2013). Potential phosphorus release from catch crop shoots and roots after freezing-thawing. *Plant and Soil,* 371(1-2), pp. 543-557.
Maguire, R.O., Chardon, W.J. & Simard, R.R. (2005). *Assessing Potential Environmental Impacts of Soil Phosphorus by Soil Testing*. In: Sims, J.T. & Sharpley, A.N. (eds) Phosphorus: Agriculture and the Environment.

Madison: American Society of Agronomy. pp 145-180. ISBN 0-89118-157-1.

Menzel, R.G. (1980). *Enrichment ratios for water quality modeling*. Pp. 486-492. In W.G. Knisel (ed.) CREAMS. A field scale model for chemicals, runoff and erosion from agricultural management systems. Conservation Research Report 26.U.S. Department of Agriculture, Washington, USA

Middleton, H.E. (1930). *Properties of soils which influence soil erosion*. Washington, DC: United States Department of Agriculture.

Mitasova, H., Mitas, L., Brown, W. M. Multiscale simulation of land use impact on soil erosion and deposition patterns. (2001). In: Mohtar, R.H. , Steinhardt, G.C. (ed.) *Proceedings of 10th International Soil Conservation Meeting*, Purdue University, West Lafayette, May 24-29 2001, pp. 1163-1169.

Moore, I.D. & Burch, G.J. (1986). Physical basis of the length-slope factor in the Universal Soil Loss Equation *Soil Science Society of America Journal,* 50(5), pp. 1294-1298.

Morgan, R.P.C. (2005). *Soil erosion and conservation* (Third Edition). Malden, MA: Blackwell Publishing Ltd. ISBN: 1-4051-1781-8

Murer, E.J., Baumgarten, A., Eder, G., Gerzabek, M.H., Kandeler, E. & Rampazzo, N. (1993). An improved sieving machine for estimation of soil aggregate stability (SAS) *Geoderma,* 56(1-4), pp. 539-547.

Myrbeck, A., Stenberg, M., Arvidsson, J. & Rydberg, T. (2012). Effects of autumn tillage of clay soil on mineral N content, spring cereal yield and soil structure over time. *European Journal of Agronomy,* 37(1), pp. 96-104.

Orchard, V.A. & Cook, F.J. (1983). Relationship between soil respiration and soil moisture. *Soil Biology and Biochemistry,* 15(4), pp. 447-453.

Panagos, P., Meusburger, K., Ballabio, C., Borrelli, P. & Alewell, C. (2014). Soil erodibility in Europe: A high-resolution dataset based on LUCAS. *Science of The Total Environment,* 479–480(0), pp. 189-200.

Panayiotopoulos, K.P., Barbayiannis, N. & Papatolios, K. (2004). Influence of Electrolyte Concentration, Sodium Adsorption Ratio, and Mechanical Disturbance on Dispersed Clay Particle Size and Critical Flocculation Concentration in Alfisols. *Communications in Soil Science and Plant Analysis,* 35(9-10), pp. 1415-1434.

Paraskova, J.V., Rydin, E. & Sjoberg, P.J.R. (2013). Extraction and quantification of phosphorus derived from DNA and lipids in environmental samples. *Talanta,* 115, pp. 336-341.

Parvage, M.M., Ulen, B. & Kirchmann, H. (2013). A survey of soil phosphorus (P) and nitrogen (N) in Swedish horse paddocks. *Agriculture Ecosystems & Environment,* 178, pp. 1-9.

Perfect, E., Kay, B.D., van Loon, W.K.P., Sheard, R.W. & Pojasok, T. (1990). Factors Influencing Soil Structural Stability within a Growing Season. *Soil Science Society of America Journal,* 54(1), pp. 173-179.

Persson, G. (2001). Phosphorus in tributaries to Lake Malaren, Sweden: Analytical fractions, anthropogenic contribution and bioavailability. *Ambio,* 30(8), pp. 486-495.

Pimentel, D. (2006). Soil erosion: A food and environmental threat. *Environment Development and Sustainability,* 8(1), pp. 119-137.

Pionke, H.B., Gburek, W.J. & Sharpley, A.N. (2000). Critical source area controls on water quality in an agricultural watershed located in the Chesapeake Basin. *Ecological Engineering,* 14(4), pp. 325-335.

Pionke, H.B., Gburek, W.J., Sharpley, A.N. & Schnabel, R.R. (1996). Flow and nutrient export patterns for an agricultural hill-land watershed. *Water Resources Research,* 32(6), pp. 1795-1804.

Pojasok, T. & Kay, B.D. (1990). Assessment of a combination of wet sieving and turbidimetry to characterize the structural stability of moist aggregates. *Canadian Journal of Soil Science,* 70(1), pp. 33-42.

Proffitt, A.P.B., Rose, C.W. & Hairsine, P.B. (1991). Rainfall detachment and deposition: Experiments with low slopes and significant water depths. *Soil Science Society of America Journal,* 55(2), pp. 325-332.

Rekolainen, S., Ekholm, P., Ulén, B. & Gustafson, A. (1997). *Phosphorus Losses from Agriculture to Surface Waters in the Nordic Countries*. In: Tunney, H., Carton, O.T., Brookes, P.C. & Johnston, A.E. (eds.) Phosphorus Loss from Soil to Water. Wallingford, UK: C.A.B. International, pp. 77-93.

Rekolainen, S., Posch, M., Kämäri, J. & Ekholm, P. (1991). Evaluation of the accuracy and precision of annual phosphorus load estimates from two agricultural basins in Finland. *Journal of Hydrology,* 128(1–4), pp. 237-255.

Renard, K.G., Foster, G.R., Weesies, G.A., McCool, D.K. & Yoder, D.C. (1997). *Predicting soil erosion by water: a guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE)*. Washington, DC: United States Department of Agriculture.

Rhoton, F.E., Emmerich, W.E., Goodrich, D.C., Miller, S.N. & McChesney, D.S. (2007). An Aggregation/Erodibility Index for Soils in a Semiarid Watershed, Southeastern Arizona. *Soil Science Society of America Journal* 71(3), pp. 984-992.

Riddle, M.U. & Bergström, L. (2013). Phosphorus Leaching from Two Soils with Catch Crops Exposed to Freeze-Thaw Cycles. *Agronomy Journal,* 105(3), pp. 803-811.

Ryden, J.C., Syers, J.K. & Harris, R.F. (1974). *Phosphorus in Runoff and Streams*. In: Brady, N.C. (ed.) Advances in Agronomy.Volume 25. Academic Press, pp. 1-45.

Schindler, D.W. (2012). The dilemma of controlling cultural eutrophication of lakes. *Proceedings of the Royal Society B-Biological Sciences,* 279(1746), pp. 4322-4333.

Scholefield, P., Heathwaite, A.L., Brazier, R.E., Page, T., Schaerer, M., Beven, K., Hodgkinson, R., Withers, P., Walling, D. & Haygarth, P.M. (2013). Estimating phosphorus delivery from land to water in headwater catchments using a fuzzy decision tree approach. *Soil Use and Management,* 29, pp. 175-186.

Sharpley, A.N. (1980). The Enrichment of Soil Phosphorus in Runoff Sediments. *Journal of Environmental Quality,* 9(3), pp. 521-526.

Sharpley, A.N. (1985). The selective erosion of plant nutrients in runoff. *Soil Science Society of America Journal,* 49, pp. 1527-1534.

Sharpley, A.N., Kleinman, P.J.A., Jordan, P., Bergstrom, L. & Allen, A.L. (2009). Evaluating the Success of Phosphorus Management from Field to Watershed. *Journal of Environmental Quality,* 38(5), pp. 1981-1988.

Sharpley, A.N., Smith, S.J., Jones, O.R., Berg, W.A. & Coleman, G.A. (1992). The Transport of Bioavailable Phosphorus in Agricultural Runoff. *Journal of Environmental Quality,* 21(1), pp. 30-35.

Shore, M., Jordan, P., Mellander, P.E., Kelly-Quinn, M., Wall, D.P., Murphy, P.N.C. & Melland, A.R. (2014). Evaluating the critical source area concept of phosphorus loss from soils to water-bodies in agricultural catchments. *Science of The Total Environment,* 490(0), pp. 405-415.

Shore, M., Murphy, P.N.C., Jordan, P., Mellander, P.E., Kelly-Quinn, M., Cushen, M., Mechan, S., Shine, O. & Melland, A.R. (2013). Evaluation of a surface hydrological connectivity index in agricultural catchments. *Environmental Modelling & Software,* 47, pp. 7-15.

Sims, J.T., Edwards, A.C., Schoumans, O.F. & Simard, R.R. (2000). Integrating soil phosphorus testing into environmentally based agricultural management practices. *Journal of Environmental Quality,* 29(1), pp. 60-71.

Sivertun, Å. & Prange, L. (2003). Non-point source critical area analysis in the Gisselö watershed using GIS. *Environmental Modelling & Software,* 18(10), pp. 887-898.

Skaggs, R.W., Breve, M.A. & Gilliam, J.W. (1994). Hydrologic and Water-Quality Impacts of Agricultural Drainage. *Critical Reviews in Environmental Science and Technology,* 24(1), pp. 1-32.

SMED (2011). *Beräkning av kväve- och fosforbelastning på Vatten och hav för uppföljning av miljökvalitetsmålet "Ingen övergödning".* Report Nr 56 2011, Sveriges Meteorologiska och Hydrologiska Institut, Norrköping, Sweden. ISSN: 1653-8102.

So, H.B., Cook, G.D. & Raine, S.R. (1997). An examination of the end-over-end shaking technique for measuring soil dispersion. *Australian Journal of Soil Research,* 35(1), pp. 31-39.

Sørensen, R., Zinko, U. & Seibert, J. (2006). On the calculation of the topographic wetness index: evaluation of different methods based on field observations. *Hydrology and Earth System Sciences Discussions,* 10(1), pp. 101-112.

Statistics Sweden (2013). Nitrogen and phosphorus balances for agricultural land and agricultural sector in 2011. (In Swedish with parts in English). *Statistiska meddlelanden* MI 40 SM 1301. Stockholm, Sweden

Stigebrandt, A., Rahm, L., Viktorsson, L., Ödalen, M., Hall, P.J. & Liljebladh, B. (2013). A New Phosphorus Paradigm for the Baltic Proper. *Ambio*, pp. 1-10.

Strauss, P., Leone, A., Ripa, M., Turpin, N., Lescot, J.-M. & Laplana, R. (2007). Using critical source areas for targeting cost-effective best management practices to mitigate phosphorus and sediment transfer at the watershed scale. *Soil Use and Management,* 23(s1), pp. 144-153.

Suarez, D., Rhoades, J., Lavado, R. & Grieve, C. (1984). Effect of pH on saturated hydraulic conductivity and soil dispersion. *Soil Science Society of America Journal,* 48(1), pp. 50-55.

Svanbäck, A., Ulen, B., Etana, A., Bergstrom, L., Kleinman, P.J.A. & Mattsson, L. (2013). Influence of soil phosphorus and manure on phosphorus leaching in Swedish topsoils. *Nutrient Cycling in Agroecosystems,* 96(2-3), pp. 133-147.

Swedish Environmental Protection Agency (2014). *Wastewater treatment in Sweden*. Report ISBN 978-91-620-8704-3, Swedish Environmental Protection Agency, Stockholm, Sweden.

Swedish University of Agricultural Sciences (2014). *Dataväardskap Jordbruksmark (agricultural land data hosting)*. Available from: http://www.slu.se/mark/dv [2014-05-15]

Swedish University of Agricultural Sciences (2013). *Publication series Ekohydrologi - Department of Soil and Environment*. Available from: http://www.slu.se/sv/institutioner/mark-miljo/publikationer/ekohydrologi/ [2013-11-21]

Tim, U.S., Mostaghimi, S. & Shanholtz, V.O. (1992). Identification of critical nonpoint pollution source areas using geographic information systems and water quality modeling. *Journal of the American Water Resources*, 28(5), pp. 877-887.

Torri, D., Poesen, J. & Borselli, L. (1997). Predictability and uncertainty of the soil erodibility factor using a global dataset. *CATENA,* 31(1), pp. 1-22.

Udeigwe, T.K. & Wang, J.J. (2010). Biochemical Oxygen Demand Relationships in Typical Agricultural Effluents. *Water Air and Soil Pollution,* 213(1-4), pp. 237-249.

Udeigwe, T.K., Wang, J.J. & Zhang, H. (2007). Predicting Runoff of Suspended Solids and Particulate Phosphorus for Selected Louisiana Soils Using Simple Soil Tests. *Journal of Environmental Quality,* 36(5), pp. 1310-1317.

Ulén, B. (2004). Size and Settling Velocities of Phosphorus-Containing Particles in Water from Agricultural Drains. *Water, Air, & Soil Pollution,* 157(1), pp. 331-343.

Ulén, B. (2006). *Sweden*. In: Poesen, J.B.a.J. (ed.) Soil Erosion in Europe. Chichester: John Wiley & Sons, Ltd, pp. 17-25.

Ulén, B., Alex, G., Kreuger, J., Svanbäck, A. & Etana, A. (2012a). Particulate-facilitated leaching of glyphosate and phosphorus from a marine clay soil via tile drains. *Acta Agriculturae Scandinavica, Section B - Soil & Plant Science,* 62(sup2), pp. 241-251.

Ulén, B., Bechmann, M., Fölster, J., Jarvie, H.P. & Tunney, H. (2007). Agriculture as a phosphorus source for eutrophication in the north-west European countries, Norway, Sweden, United Kingdom and Ireland: a review. *Soil Use and Management,* 23(s1), pp. 5-15.

Ulén, B., Bechmann, M., Øygarden, L. & Kyllmar, K. (2012b). Soil erosion in Nordic countries – future challenges and research needs. *Acta Agriculturae Scandinavica, Section B - Soil & Plant Science,* 62(sup2), pp. 176-184.

Ulén, B., Carlsson, C. & Lidberg, B. (2004). Recent Trends and Patterns of Nutrient Concentrations in Small Agricultural Streams in Sweden. *Environmental Monitoring and Assessment,* 98(1), pp. 307-322.
Ulén, B., Djodjic, F., Etana, A., Johansson, G. & Lindstom, J. (2011). The need for an improved risk index for phosphorus losses to water from tile-drained agricultural land. *Journal of Hydrology,* 400(1-2), pp. 234-243.
Ulén, B. & Etana, A. (2010). Risk of phosphorus leaching from low input grassland areas. *Geoderma,* 158(3-4), pp. 359-365.
Ulén, B. & Jakobsson, C. (2005). Critical evaluation of measures to mitigate phosphorus losses from agricultural land to surface waters in Sweden. *Science of The Total Environment,* 344(1-3), pp. 37-50.
Ulén, B., Johansson, G. & Kyllmar, K. (2001). Model predictions and long-term trends in phosphorus transport from arable lands in Sweden. *Agricultural Water Management,* 49(3), pp. 197-210.
Ulén, B., Johansson, G. & Simonsson, M. (2008). Leaching of nutrients and major ions from an arable field with an unfertilized fallow as infield buffer zone. *Acta Agriculturae Scandinavica Section B-Soil and Plant Science,* 58(1), pp. 51-59.
Ulén, B. & Snall, S. (1998). Biogeochemistry and weathering in a forest catchment and an arable field in central Sweden. *Acta Agriculturae Scandinavica Section B-Soil and Plant Science,* 48(4), pp. 201-211.
Ulén, B., von Bromssen, C., Johansson, G., Torstensson, G. & Forsberg, L.S. (2012c). Trends in nutrient concentrations in drainage water from single fields under ordinary cultivation. *Agriculture Ecosystems & Environment,* 151, pp. 61-69.
Ulén, B., Von Brömssen, C., Kyllmar, K., Djodjic, F., Stjernman Forsberg, L. & Andersson, S. (2012d). Long-term temporal dynamics and trends of particle-bound phosphorus and nitrate in agricultural stream waters. *Acta Agriculturae Scandinavica, Section B - Soil & Plant Science,* 62(sup2), pp. 217-228.
Uusi-Kamppa, J. (2005). Phosphorus purification in buffer zones in cold climates. *Ecological Engineering,* 24(5), pp. 491-502.
Uusitalo, R., Turtola, E., Puustinen, M., Paasonen-Kivekas, M. & Uusi-Kamppa, J. (2003). Contribution of Particulate Phosphorus to Runoff Phosphorus Bioavailability. *J Environ Qual,* 32(6), pp. 2007-2016.
Vagstad, N. (2001). *Nutrient losses from agriculture in the Nordic and Baltic countries: Measurements in small agricultural catchments and national agro-environmental statistics*. TemaNord 2001:591. Nordic Council of Ministers, Copenhagen, Denmark. ISBN: 92-893-0713-7.
Valpasvuo-Jaatinen, P., Rekolainen, S. & Latostenmaa, H. (1997). Finnish agriculture and its sustainability: Environmental impacts. *Ambio,* 26(7), pp. 448-455.
Villa, A., Djodjic, F. & Bergström, L. (2014). Soil dispersion tests combined with topographical information can describe field-scale sediment and phosphorus losses. *Soil Use and Management*, DOI: 10.1111/sum.12121.

Villa, A., Djodjic, F., Bergström, L. & Wallin, M. (2012). Assessing soil erodibility and mobilization of phosphorus from Swedish clay soils – Comparison of two simple soil dispersion methods. *Acta Agriculturae Scandinavica Section B-Soil and Plant Science,* 62(sup2), pp. 260-269.

Vought, L., Pinay, G., Fuglsang, A. & Ruffinoni, C. (1995). Structure and function of buffer strips from a water quality perspective in agricultural landscapes. *Landscape and Urban Planning,* 31(1–3), pp. 323-331.

Watts, C.W., Dexter, A.R., Dumitru, E. & Arvidsson, J. (1996). An assessment of the vulnerability of soil structure to destabilisation during tillage. Part I. A laboratory test. *Soil and Tillage Research,* 37(2-3), pp. 161-174.

Weld, J.L., Sharpley, A.N., Beegle, D.B. & Gburek, W.J. (2001). Identifying critical sources of phosphorus export from agricultural watersheds. *Nutrient Cycling in Agroecosystems,* 59(1), pp. 29-38.

Wischmeier, W., Johnson, C.B. & Cross, B.V. (1971). Soil erodibility nomograph for farmland and construction sites. *Journal of Soil and Water Conservation,* 26(5), pp. 189-193.

Wischmeier, W.H. & Mannering, J.V. (1969). Relation of soil properties to its erodibility. *Soil Science Society of America Proceedings* 23: 131–7

Withers, P.J.A. & Haygarth, P.M. (2007). Agriculture, phosphorus and eutrophication: a European perspective. *Soil Use and Management,* 23, pp. 1-4.

Withers, P.J.A., Hodgkinson, R.A., Barberis, E., Presta, M., Hartikainen, H. & Quinton, J. (2007). An environmental soil test to estimate the intrinsic risk of sediment and phosphorus mobilization from European soils. *Soil Use and Management,* 23(s1), pp. 57-70.

Withers, P.J.A., Jarvie, H.P., Hodgkinson, R.A., Palmer-Felgate, E.J., Bates, A., Neal, M., Howells, R., Withers, C.M. & Wickham, H.D. (2009). Characterization of Phosphorus Sources in Rural Watersheds. *J Environ Qual,* 38(5), pp. 1998-2011.

Yoder, R.E. (1936). A Direct Method of Aggregate Analysis of Soils and a Study of the Physical Nature of Erosion Losses. *Agronomy Journal,* 28(5), pp. 337-351.

Zhang, T., Page, T., Heathwaite, L., Beven, K., Oliver, D.M. & Haygarth, P.M. (2013). Estimating phosphorus delivery with its mitigation measures from soil to stream using fuzzy rules. *Soil Use and Management,* 29, pp. 187-198.

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