REPORT

Screening risk areas for sediment and phosphorus losses to improve placement of mitigation measures

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Received: 6 May 2015 / Revised: 30 May 2015 / Accepted: 9 June 2015 / Published online: 27 June 2015

Abstract Identification of vulnerable arable areas to phosphorus (P) losses is needed to effectively implement mitigation measures. Indicators for source (soil test P, STP), potential mobilization by erosion (soil dispersion), and transport (unit-stream power length-slope, LS) risks were used to screen the vulnerability to suspended solids (SS) and P losses in two contrasting catchments regarding topography, soil textural distribution, and STP. Soils in the first catchment ranged from loamy sand to clay loam, while clay soils were dominant in the second catchment. Long-term SS and total P losses were higher in the second catchment in spite of significantly lower topsoil STP. A higher proportion of areas in the second catchment were identified with higher risk due to the significantly higher risk of overland flow generation (LS) and a significantly higher mobilization risk in the soil dispersion laboratory tests. A simple screening method was presented to improve the placement of mitigation measures.

Keywords Erosion · Critical source areas · Phosphorus · Risk screening

INTRODUCTION

Transfer of sediment from agricultural areas can have negative impacts on water quality, causing changes in the physical, chemical, and biological properties of water bodies. In addition, sediment can act as a carrier of phosphorus (P), which when present in excess can cause enhanced eutrophication in river, lake, or marine ecosystems (Carpenter et al. 1998). Losses of P originate from a small proportion of a catchment known as critical source areas (CSAs), where a high risk of P mobilization and transport coincides with high soil P content. Accurate

identification of areas that contribute to such diffuse pollution in a catchment should be highly prioritized for control, treatment, and remediation purposes (Pionke et al. 2000). This is in line with the EU Water Framework Directive (WFD) (2000/60/EC) requirement of cost-effective approaches in the process of achieving “good ecological status” of water bodies.

In Sweden, many of the mitigation strategies to date have focused on controlling losses by restricting P sources (e.g., optimum soil test P, limited animal density) without consideration taken to where hydrologically active areas occur. While the regulation of soil P content is relevant from a sustainability point of view, it may not automatically lead to a reduction of P inputs in aquatic systems. Buffer strips are in Sweden one of the mitigation measures aimed at controlling P transport for which farmers can obtain subsidies. However, the current requirements for the establishment of buffer strips refer to the geographical location (which includes all regions in central-southern Sweden) and to the length/size of the strip (Swedish Board of Agriculture 2014), without taking into account the identification of actual transport pathways and landscape variability. The use of methodologies that are able to successfully combine source and transport factors is imperative in order to select high-risk catchments, sub-catchments, or fields.

Different approaches exist to target CSAs, ranging from process-based models (Davison et al. 2008) to more static P indices (PI) (Lemunyon and Gilbert 1993). The former have succeeded in determining critical areas at the catchment scale, but they usually require a high amount of input data and are dependent on calibration data (e.g., long-term series of water flow and water quality). These are often difficult, and thus costly, to obtain for non-monitored catchments. The static indices, such as the PI, were first

developed as a practical set of tools to be used directly by land managers and farmers in the USA (Sharpley et al. 2003). In Sweden, a PI has been developed and tested at the field scale (Djodjic and Bergström 2005) and at the catchment scale (Djodjic et al. 2002). It has been shown to be efficient in explaining long-term losses at the field scale, but is yet to be fully implemented at end-user level by the regulatory authorities, as it requires more input data to be effective (Foged 2011). Finally, Geographic Information System (GIS)-based tools are an increasingly popular method which ranks grid cells relative to the tendency of generating surface runoff (e.g., Heathwaite et al. 2003). These are usually simpler to use and provide detailed maps that are employed by decision makers to better allocate resources. One such example of these is the Unit Stream Power Erosion Deposition model (USPED) (Mitasova et al. 2001), a 3-dimensional improvement of the Universal Soil Loss Equation (USLE) built-in GIS that can accurately predict areas of erosion and deposition at the catchment scale (e.g., Djodjic and Spännar 2012; Djodjic and Villa 2015). It uses the concept of upslope contributing area (Moore and Burch 1986), profiting from the accuracy of high-resolution elevation data, which are becoming increasingly available. However, such modeling also requires considerable amounts of both time and data.

Risk screening of areas prone to P losses will help to initially identify high-risk areas and to detect priorities in the allocation of mitigation measures. This type of assessment can fill the current gap in the identification of focus areas, without demanding the large amount of data and time required in other types of assessments (i.e., PI or process-based modeling).

The aim of the present study was to develop simple methodologies to screen the risk of diffuse P pollution at catchment and sub-catchment scale, especially with the focus on soil erosion and overland flow. Two agricultural catchments were used to compare the risk and identify the catchments and parts of them where losses of P are most likely to occur. Specific objectives of the study were to: (1) evaluate the risk of diffuse P losses via overland flow in two catchments using three risk indicators, for source (STP), potential mobilization (soil dispersion), and overland flow transport [unit-stream power length-slope (LS) factor] as a combined risk assessment for sediment and P loss risk; and (2) screen the catchment and areas vulnerable to such losses.

MATERIALS AND METHODS

Study areas

The study was carried out in two catchments approximately 60 km apart, located in the county of Östergötland in south-

eastern Sweden. The catchments—hereafter referred to as E21 and E23—are part of a national monitoring program to determine the influence of agriculture on water quality (Kyllmar et al. 2014), meaning that long time series of water quality and discharge data at the outlet are available for both catchments. The mean annual precipitation in the catchments was below 600 mm, and the discharge at the outlet was similar in both catchments (Table 1).

Most of the agricultural fields in the catchments are artificially drained. The larger catchment, E21, comprised soils that range from loamy sand to clay loam and it is dominated by arable land used for cereal production. The soils in the smaller E23 catchment are overall finer in texture and vary from sandy to heavy clay soils. E23 has two distinct parts: a northern, downstream part, which is dominated by heavy clay soils, and a southern, upstream part, which has coarser soils, including sandy glacial till (Ulén et al. 2011). The crop production in this catchment consists predominantly of cereals and grass. The effect of different crop production systems on the risk of sediment and P losses was not considered, as the interest was in establishing the potential risk under worst-case conditions regarding soil crop cover. A totally culverted stream stretch is located in the southern part of E23, comprising 38 % of the total catchment area. The stream is open in the northern part.

Topographical analysis

Three topographical parameters: slope, flow accumulation, and unit-stream power LS, were determined using high-resolution Light Detection and Ranging (LiDAR) elevation data (2-m grid), which are already available in most parts of Sweden. The slope and flow accumulation attributes were calculated from the digital elevation model (DEM) for Sweden. Slope quantifies the maximum rate of change in value from each cell to its neighbors, while flow accumulation or upslope contributing area represents the drainage area of any cell and indicates overland flow paths. The LS parameter is calculated from slope and flow accumulation grids according to the equation:

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

where A is the flow accumulation or upslope contributing area per unit width, β is the slope angle ($^\circ$), $\mathbf{r}$ is the (x, y) point in the landscape, and m and n are the constants dependent on the type of flow and set to 1.6 and 1.3, respectively, as proposed by Mitasova et al. (2001).

In addition, the LS information for each agricultural block was summarized from the calculated 2-m cell values in order to operationalise the transport indicator used in this study. Agricultural blocks are established by the Swedish

Table 1 Summary of soil texture (clay, silt and sand), organic matter (OM), pH, and topsoil soil total P (soil TP) and P-AL from fields in catchments E21 and E23. The mean values are followed by SD in brackets

	Area (km ²)	Arable land (%)	Air temperature ^a (°C)	Water discharge ^b (mm)	N ^c	Clay (%)	Silt (%)	Sand (%)	OM (%)	pH _{H₂O} (%)	Soil TP (mg 100 g ⁻¹)	P-AL (mg 100 g ⁻¹)
E21	16.32	89	5.2	163	57	21 (10)	28 (9)	51 (17)	4.2 (3.1)	7.1 (0.5)	66.4 (18.4)	9.3 (4.9)
Range						4–45	8–54	12–87	2.0–18	5.6–7.9	36.9–148.5	4.1–24.5
E23	7.56	54	6.7	171	31	53 (16)	27 (6)	20 (17)	3.8 (1.5)	6.7 (0.4)	68.4 (15.7)	6.0 (3.7)
Range						11–77	16–36	2–68	1.8–7.9	6.0–7.7	44.2–128.7	1.6–18.8
Statistical significance						***	ns	ns	ns	ns	ns	***

ns not significant

^a Mean annual air temperature and precipitation for the period 1961/1990, at the nearest Swedish meteorological station

^b Mean discharge for the agrohydrological years 1988/1989–2010/2011

^c Number of observations

Significant differences between the two catchments are indicated, *** $P < 0.001$; ** $P < 0.01$; * $P < 0.05$

Board of Agriculture and are defined as a continuous land area with a unique identity. A total of 151 (E21) and 96 (E23) agricultural blocks were identified in the catchments. Their size ranged from 0.1 to 61.8 ha in E21 and from 0.09 to 55.6 in E23. Their median size is 9.4 ha (E21) and 3.0 ha (E23). In the present study, 49 (E21) and 24 (E23) blocks were sampled (1.2 soil samples approximately per agricultural block).

Water sampling and analysis

Water discharge at the outlet of the catchments was measured continuously using water level gauges with V-notch weirs (Kyllmar et al. 2014). Two sets of water quality data were used in the study. Firstly, water samples were taken flow proportionally at the outlet of the catchments, a few meters upstream from the weir section. When a set volume of water has passed the measurement station a sub-sample of approximately 20 mL is taken by a peristaltic pump and stored in a 10-L glass bottle. Every 2 weeks a composite sample is taken from the glass bottle and sent to the laboratory for analysis. A more detailed description of the sampling procedure can be found in Kyllmar et al. (2014). The agrohydrological years 2004–2014 (E21) and 2008–2014 (E23) were used in the analysis. Total P (TP) was analyzed on unfiltered samples after digestion in an acid persulfate solution, while dissolved reactive P (DRP) was measured after filtration with a 0.2- μ m pore diameter filter (Scheleicher & Schüll GmbH, Dassel, Germany). Suspended solids were determined by filtration using the same filters (i.e., 0.2 μ m), as the increase in filter weight. Total P, DRP, and SS analyses were performed according to European standard methods (European Committee for Standardization 1996). Unreactive P (UP) was calculated

as the difference between TP and DRP. All data regarding water flow, TP, DRP, and SS were downloaded from a publicly available database (Swedish University of Agricultural Sciences 2014).

Secondly, measurements from synoptic water sampling were used to observe patterns regarding the contribution of different parts of the catchments to losses of P. Sampling was performed manually during both high and low flow months, and with the aim to capture as many as possible of the distinct high flow events (Ulén et al. 2012). In E21, synoptic sampling was carried out in four campaigns during 2009–2010 (November, March, June, and August), (Stjernman Forsberg, 2010 unpublished results) while in E23 sampling was carried out in 23 campaigns throughout 2007–2013. In E21, samples were taken from the main stream at the outlet (Hf16) and middle part of the catchment (Hf11). In E23, samples were also taken from the main stream at the outlet (HE69) and at the opening of the culvert (HE30) (Fig. 1). These locations were used as control points for SS and P losses from the different halves of the catchments (referred to as north and south parts). Information regarding properties of north/south areas of the catchments is detailed in Table 2.

Soil sampling and analysis

A total of 57 (E21) and 31 (E23) topsoil samples (0–20 cm) were taken in September–October 2012 following a grid sampling procedure which covered the different soil texture classes in the arable land of the catchments (Fig. 1). Each sample comprised 10 soil cores distributed over an area of 1 m². The soils were air-dried and gradually broken down by hand, and then sieved before the soil dispersion (test) analysis (<5 mm) and before texture, organic matter

Fig. 1 Soil and water sampling points in catchments E21 (left) and E23 (right)

Table 2 Main characteristics and mean soil property values for the different areas (north/south) in catchments E21 and E23

		No. of sub-catchments	Area (ha)	No. of soil samples	Water quality control points	P-AL (mg 100 g ⁻¹)	Mobilization potential (NTU)	LS
E21	North (upstream)	11	1101	38	Hf11	9.4 (ns)	677 (ns)	0.017 (ns)
	South (downstream)	5	529	19	Hf16	9.1	743	0.014
E23	North (downstream)	10	479	22	HE69	6.5 (ns)	1170***	0.026*
	South (upstream)	10	264	9	HE30	4.9	726	0.017

ns not significant

Significant differences between the two halves of the catchments are indicated, *** $P < 0.001$; ** $P < 0.01$; * $P < 0.05$

(OM), pH, and soil TP analyses (<2.5 mm). Texture was determined by the pipette sedimentation procedure (Ljung 1987) and OM by loss on ignition. Soil pH was measured in a water solution according to the standard method SS 028122-2. Soil samples were then extracted after oxidative acid combustion with nitric acid following the Swedish standard method SS 28311 (Swedish Standards Institute 1997) to determine soil TP and with the ammonium lactate method (Egnér et al. 1960) to determine P-AL, which is the method used in Sweden to estimate plant-available P. These extracts were analyzed with inductively coupled plasma atomic emission spectrometry (ICP-AES, OPTIMA 3000DV; Perkin Elmer, Waltham, USA). Risk of sediment and P mobilization was determined with the soil dispersion test DESPRAL. The test recovers an aliquot of the clay and fine silt fraction (<0.02 mm) after shaking (10 rev min⁻¹, end-over-end) for one minute the air-dried soil in distilled water (1:50 soil-to-

solution ratio) (Withers et al. 2007). The test was performed in triplicate for each of the soil samples. Suspended solids, TP, and DRP were determined on this recovered aliquot following the same methods as in the water monitoring program. Turbidity was measured on post-dispersion aliquots using a Hach 2100AN (Hach, Loveland, CO) and expressed as Nephelometric Turbidity Units (NTU).

Data analyses and statistics

Mean annual flow-weighted concentration was estimated by dividing the annual load by annual discharge. The annual load was calculated by interpolation of daily concentrations from biweekly sampling data, multiplied by daily water discharge.

Differences in soil properties, soil dispersion, and topographical parameters between the two catchments and

between the two halves of each catchment were studied with a one-way ANOVA test and, when necessary, with the non-parametric Mann–Whitney *U* test. The first test was applied to study differences in means when the residuals fulfilled the assumption of normality (silt, sand) or when the residuals fulfilled the assumption of normality after logarithmic transformation (turbidity, SS, TP, UP, P-AL, and LS). When normality could not be achieved through transformation (DRP, soil TP, clay, and OM), a non-parametric test was applied (Mann–Whitney). The relation between turbidity (potential mobilization) and other soil properties in each catchment was studied through multiple linear regression. To avoid multicollinearity only the variables clay, OM, and pH were included in the model. The independent variable (turbidity) was transformed so the residuals fulfill the assumption of normality.

Screening of the blocks was done using three indicators that represent transport (mean LS per agricultural block), potential mobilization (mean turbidity from three replicates measured in the aliquot obtained from the dispersion test for each soil sample), and source (soil P test for each soil sample, P-AL). To screen fields for erosion and risk of UP losses, fields were sorted, in the order of priority, first by the LS parameter (highest priority, from high to low values), then by the soil dispersion risk and, finally, by P-AL values (lowest priority). To do this, the three parameters were previously normalized (0–1). The function “sort by more than one column” in Microsoft Excel 2010 was used. This function groups data by the same value in one column, and then sort another column or row within that group of equal values. In this way the blocks were ordered and classified into four groups depending on their position in the first, second, third, or fourth quarter. Statistical analyses were carried out in JMP Version 10 (SAS Institute Inc., Cary, NC, USA). The calculations of topographical attributes were performed in ArcGis 10.2.1. (©1999–2013 Esri Inc.).

RESULTS

Losses of P and suspended solids

A summary of mean annual flow-weighted concentrations and loads of P and SS is presented in Table 3. Total P concentration at the catchment stream outlet was low in E21 compared with the 10th percentile (0.052 mg L⁻¹) and high in E23 compared with the 75th percentile (0.23 mg L⁻¹) for long-term TP values in 21 Swedish agricultural catchments (Kyllmar et al. 2014).

Phosphorus transport at the catchment outlet in E21 was low throughout the year, with slightly higher loads during months with high discharge, while in E23 the P load was driven by periods of higher discharge (i.e., December to March–April) (Fig. 2). In E23, a higher proportion of UP in the TP exported corresponded to high discharge events (i.e., snowmelt and autumn rainfall events).

Single UP concentration values from synoptic sampling varied from 0.019 to 0.044 mg L⁻¹ in E21 (locations Hf16 and Hf11, respectively) and from 0.020 to 1.27 mg L⁻¹ in E23 (locations HE30 and HE69, respectively). In E21, DRP and UP mean concentrations were higher in the upstream location (northern half), Hf11, than at the outlet of the catchment (Table 3).

In E23, both DRP and UP mean concentrations were higher in the downstream location HE 69 (northern half), which is dominated by finer soils and where the stream is open, with the differences more pronounced in the case of UP (Table 3).

Topography

The cumulative distribution of flow accumulation and LS in catchments E21 and E23 is presented in Fig. 3. In

Table 3 Mean annual flow-weighted concentrations and mean annual transport of total phosphorus (TP), dissolved reactive phosphorus (DRP), unreactive phosphorus (UP), and suspended solids (SS) at the outlet of the catchments E21 (years 2004–2014) and E23 (2008–2014) and at two locations in the main stream. Samples from the synoptic locations were taken during 2009–2010 (E21) and 2007–2013 (E23). The mean values are followed by SD in brackets

	TP (mg L ⁻¹)	DRP (mg L ⁻¹)	UP (mg L ⁻¹)	SS (mg L ⁻¹)	TP (kg km ⁻²)	DRP (kg km ⁻²)	UP (kg km ⁻²)	SS (kg km ⁻²)
E21								
Outlet (long-term)	0.060 (0.023)	0.029 (0.014)	0.031 (0.013)	11.4 (3.9)	10.90 (7.75)	5.37 (4.72)	6.29 (3.55)	1965 (1017)
Hf11 (north)	0.096 (0.027)	0.066 (0.021)	0.031 (0.01)	14.0 (9.9)				
Hf16 (south, outlet)	0.084 (0.024)	0.059 (0.028)	0.025 (0.03)	14.0 (4.3)				
E23								
Outlet (long-term)	0.29 (0.018)	0.12 (0.011)	0.17 (0.022)	130.4 (45.10)	61.50 (15.37)	26.54 (7.95)	28.93 (11.14)	27,041 (7922)
HE69 (north, outlet)	0.38 (0.29)	0.13 (0.08)	0.25 (0.26)	191.8 (171.6)				
HE30 (south)	0.22 (0.16)	0.070 (0.05)	0.15 (0.13)	118.9 (123.9)

Fig. 2 Monthly export of dissolved reactive phosphorus (DRP) and unreactive phosphorus (UP), and discharge from **a** catchment E21 and **b** catchment E23. Data represent mean values for the period 2004–2014 (E21) and 2008–2014 (E23)

summary, both catchments had similar flow accumulation patterns but diverged as regards the slope and LS parameters. Significant differences in LS ($P < 0.01$) between the two catchments were found, with the risk of flow accumulation and initiation of surface runoff being higher overall in catchment E23. Catchment E21 is mostly flat, while E23 has areas with steeper slopes which are located in the proximity of the water course (Fig. 4). The mean LS values were higher in the northern half of the E23 catchment, but the difference between the halves was not significant.

The mean, median, and 90th percentile LS values for all of the agricultural blocks contained in E21 and E23 were highly correlated with each other. The Pearson correlation coefficient was 0.93 ($P < 0.001$) between the mean and the median, 0.98 ($P < 0.001$) between the mean and the 90th percentile, and 0.90 ($P < 0.001$) between the median and the 90th percentile. These correlations showed that the use of mean values to describe the LS parameter of each agricultural block was a robust method (i.e., the use of mean, median, or percentile values would give very similar rankings).

Fig. 3 Cumulative distribution of flow accumulation and LS in catchments E21 (filled triangles) and E23 (unfilled circles), calculated from LiDAR elevation data

Soil properties and mobilization potential of SS and P

Analyses of topsoil properties (Table 1) showed that soils in catchment E23 had a significantly ($P < 0.001$) higher clay content than soils in E21, which displayed a wider range of textural classes with a significantly ($P < 0.001$) higher sand content. A higher number of areas in the northern part of the catchment showed signs of intense fertilization. Mean topsoil P-AL concentration was significantly ($P < 0.001$) higher in E21 ($9.3 \text{ mg P } 100 \text{ g}^{-1}$) than in E23 ($6.0 \text{ mg P } 100 \text{ g}^{-1}$) (Table 1). Mean P-AL values in E21 fell within class IVa (of V, i.e., $8\text{--}12 \text{ mg P } 100 \text{ g}^{-1}$) and mean P-AL values in E23 within class III (i.e., $4\text{--}8 \text{ mg P } 100 \text{ g}^{-1}$) (Table 1) which is the recommended optimum soil P status according to the Swedish Board of Agriculture (2013). Additionally, no significant differences were found between the catchments in terms of soil TP, silt, and OM content. No significant differences were found between properties in the halves of E21, while significant differences were found between halves of E23 in potential mobilization ($P < 0.01$). The northern, upstream half had higher values of potential mobilization and LS.

Fig. 4 Slope increase estimated with high-resolution elevation data (2-m grid) for catchments E21 (left) and E3 (right). Steeper slopes are represented with darker shades. Note the different slope range used for the two catchments

Fig. 5 Boxplot showing the estimated concentrations of dispersed suspended solids (SS), unreactive phosphorus (UP), and dissolved reactive phosphorus (DRP) for the catchments E21 and E3, from the laboratory DESPRAL test. The lower and upper boundaries of the box indicate the 25th and 75th percentiles, respectively. The line within the box marks the median and the error bars below and above the box indicate the 10th and 90th percentiles, respectively. The filled circles below and above the error bars indicate the 5th and 95th percentiles, respectively

A summary of the SS and P mobilization (soil dispersion) potential data obtained in the laboratory by the DESPRAL test (Fig. 5) revealed that the potential risk of sediment mobilization (measured both as turbidity and SS) was significantly ($P < 0.001$) higher in E23 than in E21, although the latter displayed wider range of values. UP was the main fraction in the dispersion test considering soil samples from both catchments, with DRP being one order of magnitude smaller than UP. In E21, no significant differences were found between the two halves of the catchments regarding SS or P mobilization potential. The risk of sediment mobilization in the two halves of E23 was significantly different, being higher in the northern part, with higher mean clay content, than in the southern part (Table 2). The P fraction bound to the dispersed soil (UP/SS) was significantly ($P < 0.001$) different in the two catchments and ranged from 0.72 to 7.33 mg g^{-1} (mean 1.6, SD 0.95) in E21, and from 0.60 to 1.92 mg g^{-1} (mean 0.94, SD 0.30) in E23. The extreme value (7.33 mg g^{-1}) observed in E21 was due to a very low mobilization (i.e., 327 NTU and 0.18 g L^{-1}) combined with a very high P-AL content (20.2 $\text{mg P } 100 \text{ g}^{-1}$).

In E21, potential mobilization (turbidity) showed a significant and positive relation to clay ($r^2 = 0.43$, $P < 0.001$) following the equation $\text{Log turbidity} = 6.36 + 0.39 \times \text{clay}$. In E23, potential mobilization showed a significant relation to OM, clay, and pH ($r^2 = 0.54$, $P < 0.001$). Of these three

parameters it was OM that showed the greatest effect on turbidity (coefficients = -0.39 , 0.29 , and -0.16 , for OM, clay, and pH, respectively). The regression equation obtained was $\text{Log turbidity} = 6.84 - 0.39 \times \text{OM} + 0.29 \times \text{clay} - 0.16 \times \text{pH}$. Potential mobilization decreased with the organic matter content and pH. Contrarily, potential mobilization increased with the clay content.

Screening of vulnerable areas

Ordering the agricultural blocks in terms of transport, mobilization, and source risks resulted in a higher number of blocks from catchment E23 identified with a higher risk for P losses (Table 4), which corresponded to a larger proportion of the studied area. In the case of E21, 22 blocks (corresponding to 43 % of the area) were found within the top half positions (groups 1 and 2). In the case of E23, 22 blocks were identified which corresponded to 71 % of the studied area.

DISCUSSION

The assessment method used was a simple but robust screening scheme which offered a transparent procedure with full insight in the factors included in the evaluation. The prioritization of transport factors was based on previous knowledge. For example, according to Pionke et al. (1997), sources with low P content and high runoff and/or erosion potential can be major P sources, whereas those with high P concentrations but low runoff and erosion potential may be minor P sources. Similarly, Sharpley et al. (2013) state that soil test P legacy build-up can be overwhelmed by the site hydrology when it translates to P loss. Also, the inherent susceptibility of soils to erosion (i.e., mobilization risk), has been suggested to be less important than convergence of the overland flow (Djodjic and Villa 2015) which is controlled by topography. The evaluation of the results of both soil analyses and water quality monitoring supports this hypothesis. UP export at the catchment scale was generally driven by the hydrological transfer potential and mobilization risk rather than by the P content

in the soils, which is in line with previous findings (e.g., Buda et al. 2009; Jordan et al. 2012; Villa et al. 2014).

In the present case, the STP could not explain the overall risk of P pollution for the studied catchments. Despite the fact that catchment E23 had significantly lower STP values, TP export was, according to the long-term water monitoring, higher in catchment E23 than in E21 both in UP and DRP. Even high levels of P-AL in E21 did not result into high TP loads or concentrations at the outlet. The higher overall P export in E23 was a result of longer/steeper slopes and higher soil mobilization potential than in E21, the former illustrated by the divergence in LS values between the catchments (Fig. 3).

Screening of vulnerable areas within catchments translated into a higher proportion of areas in catchment E23 with higher transport risk than in E21. This means that, in comparison, there were more areas in E23 which were more likely to be hotspots of P losses due to erosion and overland flow. Many of the high-risk blocks in E23 were located near the main stream in the northern part of the catchment, which was also indicated by higher UP and DRP concentrations based on weekly or biweekly water sampling (Ulén et al. 2012). High-risk blocks were more widely spread within E21 (Fig. 6).

In E21, the highest mean UP and DRP concentrations were observed in the northern part (upstream synoptic sampling location Hf11), where also a higher number of blocks were identified with a high risk for P losses due to erosion (Fig. 6). However, given that the main P fraction is in dissolved form and that soil P (STP) hotspots can be found especially in the north-east part of the catchment, priority at a first stage should be given to improving source P management rather than just controlling erosion. The risk of DRP losses in this catchment would increase if the P sorption capacity were to reach saturation levels due to long-term application of P to the soil (Schoumans and Groenendijk 2000). Long-term DRP concentrations in the outlet were much lower than those associated with the synoptic sampling (Table 3). Two values from the four observations were driving the high concentrations in the synoptic samples. These could be related to snowmelt

Table 4 Classification of the agricultural blocks according to their phosphorus (P) loss risk. The risk groups correspond to their position in the first, second, third, or fourth quarter after an ordinal classification. The range for length-slope (LS), turbidity (soil dispersion), and soil test P (P-AL) is given

P loss risk group	LS	Turbidity (NTU)	P-AL (mg 100 g ⁻¹)	Number of blocks		Area (%)	
				E21	E23	E21	E23
1 (high)	0.023–0.066	415–2488	2.7–23.3	10	12	17	31
2	0.017–0.023	323–2942	1.6–24.5	12	10	26	40
3	0.011–0.016	193–1340	3.2–23.3	17	5	33	21
4 (low)	0.0043–0.012	177–1736	3.5–20.2	18	4	24	9

Fig. 6 Phosphorus loss risk of observations in catchments E21 (left) and E23 (right). Class 1 in red and bigger sized circles represents the highest 25 % of all the observations, class 2 (yellow) the following 25 % highest observations, class 3 (green) and class 4 (blue) represent the lower 50 % of risk fields. Synoptic sampling locations are marked with a cross

(March) and to fertilization management (August). However, long-term measurements give a more reliable indication of overall catchment losses.

In E23, higher concentrations in both P fractions were observed in the northern downstream part of the catchment than in the south, both at the end of the culvert and in the open stretch at the end of the southern half (Table 3). The majority of high-risk blocks, according to the screening scheme, in E23 were also located in its northern half, where there is a concurrence of higher mean LS values and a significantly higher mobilization risk than in the southern half despite similar soil P content. In the same direction, farmers have previously reported a few erosion hotspots located mainly in the northern part of the catchment, which have also been identified by the USPED model (Ulén et al. 2012).

The results showed that clay content was an important variable affecting sediment dispersion when there was a greater range of textures (i.e., E21), while OM was more important when there was a predominance of finer-textured soils (i.e., E23). Soil texture, structure, and OM content are important parts of erodibility assessment (Wischmeier and Smith 1978; Withers et al. 2007). No significant differences were found in the UP mobilized by the laboratory test between the two catchments (Fig. 5), which suggest

that the amounts or loads of suspended material mobilized plays a stronger role in determining losses of UP at the catchment scale.

In summary, the results from the comparison of catchments (E21 vs E23) showed that preconditions for overland flow pathways (LS in E23 > E21) and mobilization (soil dispersion in E23 > E21) are needed for P losses to occur. Comparison of catchment areas in E23 showed that for similar LS (when comparing different halves of E23), mobilization is determining the amount of SS that is lost. The importance of transport over source factors also supports the suggestion by some authors that CSAs should be determined by working the P transfer continuum from the impact to the source (Reaney et al. 2011). This is more in accordance with the requirement in current water legislation (e.g., the EU WFD and the US Clean Water Act) to prioritize measures when there is an effect on a valuable indicator species (Valinia et al. 2014).

At the moment, no prioritization of the location of the implemented measures is done in Sweden based on a general risk assessment. In fact, contrary to the observed distribution of the risk in the two catchments, the available information on buffer strips showed that in 2012, these were present along nearly all streams in E21, while barely

any vegetated strips were present in E23, where they would be needed the most. High-risk fields are the most sensitive to P erosion and thus, appropriate mitigation measures should be first implemented in these fields. Placement of buffer strips in areas with no or limited overland flow (i.e., the majority of E21) will be highly ineffective and thus costly. The example of the two catchments presented here can serve as an illustration of how resources could be allocated in a more balanced manner and in consistency with proper assessments of the risk.

Within the detected risk areas, priority should also be given to those with a higher connectivity and proximity to surface water. Transport of mobilized particles to surface waters can initially happen through overland flow, but may reach surface waters either through overland flow or via the tile drainage system. Conversion of overland flow into tile drains may occur either through macropores or surface water inlets which can act as a rapid connection between CSAs and surface water bodies. Nevertheless, attention should be paid to those fields where overland flow is generated, as mitigation measures are more effective as close as possible to the source. Prioritization and targeting of countermeasures to mitigate other causes of P losses (e.g., leaching losses) may need other approaches (Djodjic et al. 2002).

In addition to overland flow risk generation, water ponding could occur in areas with a limited capacity of the drainage system and when there is limited permeability of the soil, both of which have been reported in catchment E23 (Ulén et al. 2011; Kyllmar et al. 2013). Ponding was observed in small depressions in the lower part of the fields. An enhancement of preferential flow (Skaggs et al. 1994) and/or possible overflowing of ponds directly to the stream could occur. Even these features are, however, indirectly included to the results of LS calculations, as flow accumulation is an important part of the applied equation. The indicators used in this study would be most appropriate for fields/catchments with relatively poor drainage, where overland flow is the dominant transport pathway.

CONCLUSIONS

The use of a simple risk screening method could be an effective way to select catchments and areas vulnerable to SS and P losses in eroded form. Simple methodologies such as those proposed in this study are far easier to implement than the more complex and data-intensive tools, although they do not have the same level of detail. The background for this is that, at this time in Sweden there is no systematic methodology available to classify and rank the risks between and within watersheds and thus, no prioritization to allocate mitigation measures is done based on their effectiveness. In contrast, most risk assessment efforts

usually aim at precisely identifying only the top few percent of vulnerable areas within the catchments, but such assessments are data demanding, time consuming, and difficult to scale up and implement in a broad sense. In addition, there is a greater focus on P sources since these are more easily estimated but, as observed in this study, high P soils do not necessarily translate into high losses at the outlet of the catchment.

Alternatively, as we show in this study, targeting for example top 50 % of the most vulnerable fields in a catchment with simple risk screening method could be a good compromise in terms of more effective management and focusing of the resources available for mitigation. The screening assessment presented here could identify catchments or sub-catchments where P losses are more likely to occur.

The proposed method here would need to be validated in small catchments with intensive water quality measurements across the catchments, and then further integrated in a quantitative tool. Losses and risk scores would need to be compared in several catchments.

Acknowledgments This study was funded by the Swedish Farmers' Foundation for Agricultural Research, which is gratefully acknowledged. Monitoring of the catchments has been financed by the Swedish Environmental Protection Agency and the assessment of catchment E23 by a research project funded by Formas. Thanks to Anuschka Heeb at the County Administration of Östergötland, who is the local coordinator for the advisory program "Focus on Phosphorus". Thanks to Lovisa Stjernman Forsberg and Stefan Andersson for helping with data regarding synoptic sampling and sub-catchment delineation.

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