Soil indicators for critical source areas of phosphorus leaching

W.J. Chardon and H.G. van Faassen

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The loss of phosphorus from farm land can lead to eutrophication of surface waters, causing algal blooms, anoxia of the water and bottom sediments, the growth of toxic organisms, fish mortality and a decrease of attractiveness for recreation. This study, conducted as a part of The Netherlands Integrated Soil Research Programme, concerns an analysis of pathways of soil phosphorus to surface waters and of soil indicators for the identification of critical source areas. The study is based on a literature review, an inquiry among experts of the COST 832 network and the analysis of already available data sets. For the Netherlands leaching of phosphorus via the subsoil to surface waters is the main cause of phosphorus pollution of surface water by farm land. The best soil indicator of critical source areas for the Dutch situation is clearly water-extractable soil-P (Pw) of the topsoil or subsoil, depending on the local situation.

Project implementation:
DLO Research Institute for Agrobiology and Soil Fertility, Bornsesteeg 65, P.O. Box 14, NL-6700 AA Wageningen (The Netherlands); phone: (31)(0)317-475700; fax: (31)(0)317-423110.

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Abstract

An excess of nutrients like nitrogen and phosphorus (P) in surface waters can cause eutrophication. Since points sources (e.g. sewage treatments plants, fertilizer industries) in general succeed in reducing their output, diffuse sources of P like agricultural fields become more important. Within agricultural areas differences may exist in the contribution of different fields to the total loading of e.g. P to surface water. Areas that have a high contribution are called Critical Sources Areas (CSA). Factors that are important for CSA's in surface runoff are relatively well known: the main factors are tendency to produce runoff (slope) and soluble P in the top soil, related to plant available P. Both factors are relatively simple to determine in practise. An important condition for such a CSA is a hydrological link between a field and an adjacent surface water, so the field has to border the surface water or must have a link via (artificial) drains.

Detecting CSA's for P transport through the subsoil to surface water is investigated less intensively. One reason for this is that, until recently, it was believed that leaching of P hardly occurs since soils have a high P binding capacity. However, the insight is now gaining that the sorption capacity for P of almost all soils is limited, and that some factors may accelerate P transport through the soil profile. Examples of these factors are preferential flow and fast transport of organically or particle bound P.

This report deals with an evaluation of indicators that can be used in the field for detecting CSA's for subsurface P loss, also denoted as leaching. The study was performed by literature research, by analysing existing data sets and by an inquiry among partners in the European COST network on contribution of agriculture to eutrophication with P.
Parameters that are mentioned or used as indicator for CSA's via leaching are:
- P content in the soil solution;
- soil P content (total P, plant available or water-extractable P);
- P saturation index;
- soil hydrological characteristics, leading to a high risk of preferential flow;
- P content of tile drainage water;
- P content of the bottom sediment, or of the surface water adjacent to the field.
An important condition for a useful indicator is that it has a limited temporal variability. Otherwise, the time of determining the indicator would influence the result too much.

P content of the soil solution gives a direct indication of the downward moving solution through the soil profile. However, sampling of this solution is a time consuming process, and the conditions of sampling, storage and further treatment have a large influence on the results. Due to the complexity of obtaining reliable data of P in the soil solution, sampling of the soil solution has so far hardly been used as an indicator of CSA's. As an alternative, *extraction of homogenized soil samples* with water or a dilute salt solution has been used to simulate the soil solution.

Total top soil P content gives only a first indication of potential CSA's. Without further knowledge on the binding capacity of the subsoil this figure is meaningless. *Plant available P content* gives a comparable first indication needing subsoil data. For CSA's, it will be an advantage when the method used is directed on readily soluble P compounds, since these are most mobile in soil.

A *saturation index* for a soil was originally developed for acid sandy soils, and was based on oxalate extractable P, Fe and Al. The relation between this index and e.g. water soluble P is reasonable, but differs for different soil types. Several other saturation indices were proposed recently, but testing for different soil types has still to be done.

Soil hydrological characteristics, leading to a *high risk of preferential flow*, are most important for choosing a sampling depth. Preferential transport will cause rapid transport of P from the top soil to drains, and thus make the condition in the subsoil less relevant. In that case field samples must be taken from the top soil. Preferential flow occurs in cracking clay soils; dried organic soils; soils with continuous macropores, e.g. created by earthworms, and water repellent soils.

Although *P content of tile drainage water* gives an indication of the direct input from a field to surface water, its usefulness for detecting CSA's is limited. The main reason for this is the fact that tile drain P content can be very variable in time, both within hours or days in case of rain storm flow and within months due to changing seasons. Thus, sampling time will have a too large influence on the result.

A high temporal variability was also found for the *P content of ditch water* close to agricultural fields, which make it probably not an ideal parameter for detecting CSA's. Resuspension of *sediment* during periods of high discharge rates makes the sediment a dynamic compartment, with variable composition. However, data of P in ditch sediments are limited, and its usefulness for CSA's cannot further be evaluated yet.

It was concluded that *water soluble P* is the most promising parameter as indicator for CSA's. It is correlated with soil solution P content, with P content in artificial runoff, and with drainage water P in case of preferential flow. Care must be taken that sufficient time is taken between sampling and the last application of any fertilizer. Sampling depth must depend on hydrological conditions. In case of a very shallow water table or preferential flow the plough layer should be sampled, otherwise samples should be taken from layers near the water table or artificial drainage depth.

Samenvatting

Het verlies van fosfaat (P) vanuit landbouwgronden kan leiden tot eutrofiëring van oppervlaktewater. Gevolgen van eutrofiëring kunnen zijn: het optreden van algenbloei, het vóórkomen van toxische organismen, anaërobie van het oppervlaktewater en/of het sediment, en een verminderde aantrekkelijkheid voor recreatie. In het algemeen wordt het verlies van P gedomineerd door runoff en erosie. In sommige streken, zoals Florida (USA), West-Australië en Nederland, wordt P vooral via ondiep grondwater getransporteerd naar het oppervlaktewater. In de regel bepaalt de transportweg de omvang van het jaarlijkse verlies: bij erosie kan het jaarlijkse verlies meer dan 20 kg P ha^{-1} bedragen, bij runoff ligt het meestal tussen 1 en 5 kg ha^{-1}, en bij uitspoeling meestal lager dan 1 kg ha^{-1}. Het feitelijke P-verlies hangt onder meer af van de belasting van de bodem met P in het verleden en van de capaciteit van de bodem om P te binden.

Internationaal gezien is het besef dat uitspoeling van P belangrijk kan zijn relatief nieuw. De bodem werd altijd in staat geacht om fosfaat goed vast te kunnen leggen; vrij algemeen wordt nu echter erkend dat de vastleggingscapaciteit eindig is. Op plaatsen waar lange tijd meer P is toegediend dan door het gewas wordt afgevoerd kan deze capaciteit zijn verbruikt. Bodems rijk aan organische stof, zoals veengronden, werden altijd als een uitzondering gezien: hier vindt relatief veel uitspoeling plaats. Het onderwater zetten van een grond bevordert de uitspoeling sterk, doordat bij anaërobie ijzeroxiden in oplossing kunnen gaan waaraan P is vastgelegd. In kleigronden en droge organische gronden kan preferentieel transport door macroporiën plaatsvinden.

In veel studies waarin uitspoeling van P is onderzocht is het moeilijk om deze te correleren met gehalten in de bodem zelf, in de ondergrond of de bouwvoor. Dikwijls is dit het gevolg van onvolledige gegevens. Echter, de weg waarlangs het water zich door de bodem beweegt is variabel, evenals de periode waarin dit gebeurt. Een snelle afvoer van percolatiewater via drainagebuizen vermindert de contacttijd met de bodem, waardoor er minder P via vastlegging aan de ondergrond kan worden verwijderd. Dit heeft tot gevolg dat het veel lastiger is om de plaatsen te voorspellen waar uitspoeling zal plaatsvinden dan in het geval van verlies via oppervlakkige afspoeling.

Voor veel stroomgebieden geldt dat een groot deel (meer dan 75%) van de jaarlijkse afvoer van P plaatsvindt vanuit een klein deel (tot 5%) van het totale bodemoppervlak. Dit worden ook wel *critical source areas (CSA's* of *fosfaatlekkende gronden)* genoemd. Dit kan zowel opgaan voor transport langs het bodemoppervlak als via uitspoeling. CSA's voor oppervlakkig transport zijn uitgebreid onderzocht, en relatief goed bekend. Veel minder is bekend over CSA's waar uitspoeling de oorzaak is van verlies van P; voor Nederland geldt dat P verlies vanuit de bodem door uitspoeling veel belangrijker is dan oppervlakkig P-transport. Analoog aan CSA's kan gesproken worden van *critical periods and events*, aangezien een of enkele regenperioden verantwoordelijk kunnen zijn voor

meer dan de helft van het totale jaarlijkse verlies aan P. Dit geldt met name voor het optreden van oppervlakkige afspoeling van P en voor het vóórkomen van preferent transport door het bodemprofiel (zie par. 4.4). Het treedt op bij een al grotendeels met water verzadigde bouwvoor, een hoge regenintensiteit en een hoog gehalte aan desorbeerbaar P van de bodem.

Meer inspanning is nodig om plaatsen op te sporen binnen een stroomgebied waar onevenredig veel uitspoeling van P plaatsvindt. Op deze plaatsen kunnen dan extra maatregelen worden genomen om uitspoeling te beperken, omdat dit een hoog milieurendement heeft.

In deze studie zijn indicatoren geëvalueerd die bruikbaar zouden kunnen zijn voor het aanwijzen van CSA's voor uitspoeling van P. Dit is gedaan door middel van literatuuronderzoek en aan de hand van (ongepubliceerde) databestanden. Daarnaast zijn vragen gesteld aan Europese onderzoekers die betrokken zijn bij COST Actie 832: 'Quantifying the agricultural contribution to eutrophication' [with phosphorus].

Bij de evaluatie was een belangrijk criterium de variatie in de tijd van de indicator. Een bruikbare indicator levert een vrij constant beeld op in de tijd, omdat anders (te) hoge eisen worden gesteld aan het tijdstip van meten.

Parameters die worden genoemd of gebruikt als indicator voor CSA's via uitspoeling zijn:
- P-gehalte in het bodemvocht;
- P-gehalte van de bodem (totaal P, plant-beschikbaar of water-extraheerbaar P);
- P-verzadigingsgraad;
- hydrologische eigenschappen van de bodem, bepalend voor de kans op preferent transport;
- P-gehalte van drainwater;
- P-gehalte van sediment, of van oppervlaktewater nabij een perceel.

Het P-gehalte van het *bodemvocht* is een directe indicatie voor de oplossing die zich neerwaarts door het bodemprofiel beweegt. De monstername is echter tijdrovend, en de wijze van monstername, bewaren en verdere behandeling hebben een grote invloed op de gemeten P-gehalten. Omdat het moeilijk is om betrouwbare gegevens te verkrijgen over P in bodemvocht wordt dit gegeven zelden gebruikt als indicator voor CSA's. Als alternatief worden gehomogeniseerde grondmonsters geëxtraheerd met water of een verdunde zoutoplossing om bodemvocht te simuleren.

Het *totaalgehalte van de bouwvoor* geeft slechts een eerste indicatie van mogelijke CSA's. Zonder kennis over de bindingscapaciteit van diepere bodemlagen heeft dit gegeven weinig betekenis. Het voor de plant beschikbare P vormt een vergelijkbare eerste indicatie, met dezelfde beperking. Het is een voordeel wanneer voor plant-beschikbaar P slechts makkelijk oplosbare P-verbindingen worden geanalyseerd, omdat die het meest

mobiel zijn. Dit is bijvoorbeeld het geval met het Pw-getal, de in Nederland gebruikte methode voor bouwland.

De *verzadigingsgraad* van een bodem is oorspronkelijk ontwikkeld voor zure zandgronden, en is gebaseerd op met oxalaat extraheerbaar P, Fe en Al. Er bestaat een vrij goede relatie met bijvoorbeeld water-oplosbaar P, maar deze is afhankelijk van bodemeigenschappen zoals pH. Inmiddels zijn er verschillende verzadigingsindexen in omloop, maar deze zijn nog niet uitgebreid getest op verschillende bodemtypen.

Bodemeigenschappen die leiden tot een grote kans op *preferent transport* zijn van belang voor de keuze van de bemonsteringsdiepte. Preferent transport zorgt voor een snel transport van bouwvoor naar drains en oppervlaktewater, en maakt kenmerken van de ondergrond van minder belang. In een dergelijk geval moet monstername in de bouwvoor gebeuren. Preferent transport treedt op in: (i) kleigronden waar scheurvorming plaatsvindt, (ii) uitgedroogde organische stofrijke bodems, (iii) percelen waar ononderbroken macroporiën voorkomen, bijvoorbeeld als gevolg van activiteit van regenwormen, en (iv) waterafstotende gronden.

Hoewel het P-gehalte van *drainwater* een indicatie is voor de directe uitstroom van P richting oppervlaktewater, is de bruikbaarheid voor CSA's gering. De voornaamste reden hiervoor is het feit dat dit gehalte sterk kan variëren in de tijd, zowel binnen uren of dagen in het geval van hevige regenval als binnen maanden door seizoensinvloeden. Het tijdstip van monstername zal dus een (te) grote invloed kunnen hebben op het resultaat.

Een grote variatie in de tijd werd ook gevonden voor het P-gehalte van *oppervlaktewater* nabij landbouwpercelen, wat dit minder geschikt maakt als indicator. Het opwervelen van sediment gedurende perioden van verhoogde afvoer maakt de waterbodem tot een dynamisch compartiment. De beschikbaarheid van gegevens van waterbodems van sloten is echter beperkt, en er kan dus geen eindoordeel gegeven worden over de bruikbaarheid ervan als indicator.

Geconcludeerd wordt in dit rapport dat *wateroplosbaar P* de meest bruikbare indicator lijkt voor het vaststellen van CSA's. De parameter is gecorreleerd met het P-gehalte van bodemvocht, met het P-gehalte van (kunstmatige) runoff, en met P in drainwater in geval van preferent transport. Er moet voor worden gezorgd dat de monstername voldoende lang (enkele maanden) gebeurt na de laatste toediening van meststoffen, in welke vorm dan ook. De bemonsteringsdiepte is afhankelijk van de hydrologische omstandigheden. In het geval van een hoge grondwaterstand (nabij bouwvoor) of van preferent transport dient de bouwvoor te worden bemonsterd, en anders een bodemlaag nabij de gemiddeld hoogste grondwaterstand of de drainagediepte.

List of abbreviations

COST	CO-operation in Science and Technology
COST 832	code number for COST-action on P loss from agriculture
CSA	Critical Source Area
MHW	Mean Highest Water-table
MRP	Molybdate Reactive P (indication for biological available P)
ox	oxalate
P	Phosphorus
PSC	Phosphorus Sorption Capacity
PSD	Phosphorus Saturation Degree
P-water	water-extractable soil-P (general)
Pw	water-extractable soil-P (Dutch method, 60:1 water:soil ratio)

1. Introduction

The loss of phosphorus (P) from soils can lead to eutrophication of surface waters. Consequences of eutrophication can be e.g.: the occurrence of algal blooms, the growth of toxic organisms, anoxia of the water or sediments, and decrease of attractiveness for recreation.

In most regions, runoff and erosion are dominant causes of P loss from soils. However, in some regions of the world, e.g. Florida (US), Western Australia and the Netherlands, P is mainly transported from soils to surface waters by subsurface drainage (Sharpley & Rekolainen, 1997). In general, the amounts of P lost varies between the different pathways: data reported for erosion can be in the range of a yearly loss > 20 kg P ha^{-1}, data for surface runoff are mostly between 1 and 5 kg P ha^{-1}, and for leaching < 1 kg P ha^{-1}.

Leaching of P from soil by drainage was previously considered to be of little significance, other than in a few specific cases, such as poorly drained soils high in organic matter (Brookes et al., 1997). This belief was mainly based on the large capacity of many soils to adsorb surplus amounts of P, added as manure and fertilizers. The *phosphorus sorption capacity* (PSC) of soils can indeed be very large, but is not unlimited. Generally, the P concentration in water percolating through the soil profile is small, due to sorption of P by subsoil with a high PSC. Exceptions are acid organic or peaty soils, where the affinity and sorption capacity for P are low, due to the predominantly negative charged surfaces and the complexing of Al and Fe by organic matter. Thus, P losses in drainage waters from organic soils were noted up to 37 kg ha^{-1} yr^{-1} (Duxbury & Peverly, 1978; Miller, 1979; Kuntze & Scheffer, 1979; Cogger & Duxbury, 1984). Soil management can be an important factor affecting P leaching: manuring of high moor grassland strongly increased P leaching compared to mineral fertilizer use (Bartels & Scheffer, 1987).

Furthermore, relatively large amounts of P can be lost by leaching from:
- sandy soils with a low PSC;
- soils which have become waterlogged;
- clay type soils or dried organic soils with preferential water flow through macropores;
- so-called *P-saturated soils*, where the PSC is reduced by large additions of fertilizers and manure over long periods.

Recent reviews (e.g. Brookes et al., 1997; Sims et al., 1998) assess the significance of P losses in drainage water by subsurface flow as a contribution towards eutrophication, and the possible mechanisms involved in these losses. In much of the work reviewed it is difficult to relate P loss to concentrations of soil P, in either the plough layer or the subsoil, due to insufficient data. Because of the variable pathways and period of water flow through a soil with subsurface drainage, factors controlling P concentrations in subsurface waters are far more complex than for surface runoff. Rapid removal of percolating soil water by

mole and tile drains decreases the contact time with the subsoil and thus less P can be removed through sorption by the subsoil.

The critical P concentration for triggering eutrophic effects in lakes can be as low as 0.02-0.035 mg P L^{-1} (Brookes et al., 1997). Thus, the movement to the aquatic environment of even relatively small quantities of P by subsurface drainage of soils can therefore pose a threat to surface water quality, as shown in several publications.

In many catchments, more than 75% of annual P export from soils is lost from only 5% of the land area, called *critical source areas (CSA's)*, both through surface runoff and subsurface drainage (Sharpley & Rekolainen, 1997). However, in catchments where both soil conditions and long-term loading with P are less variable, an opposite pattern is found. This is illustrated in figure 1, where estimations by Schoumans and Kruijne (1995) for the Schuitenbeek area in The Netherlands are compared with a hypothetical case.
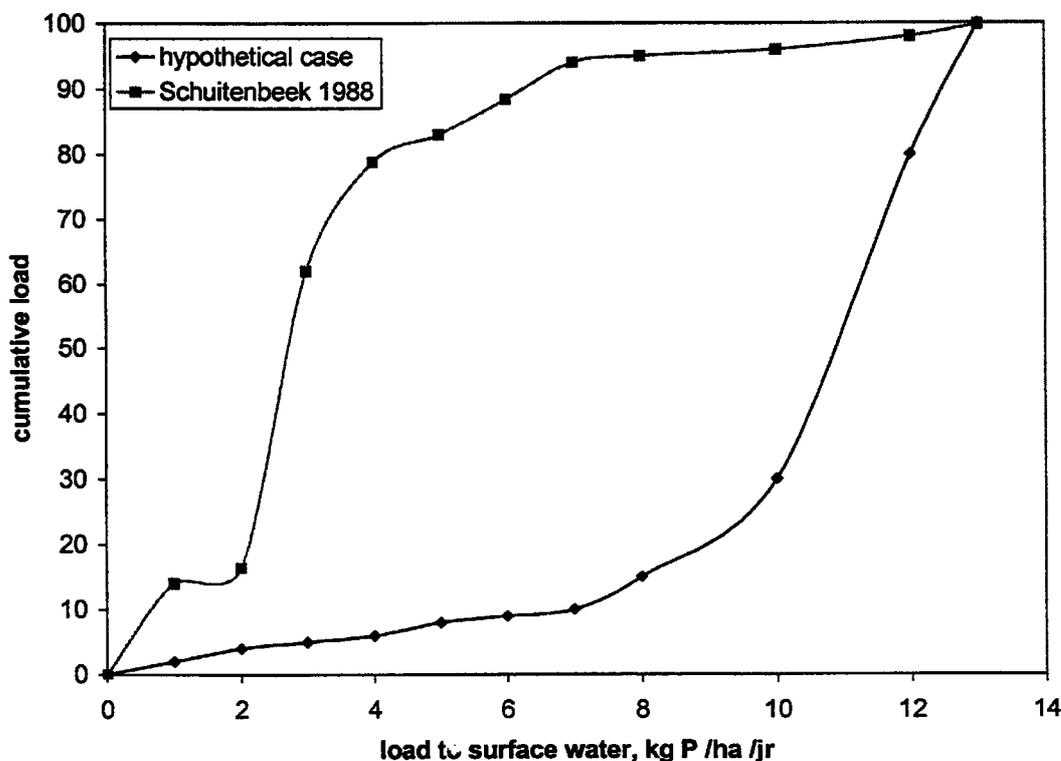
Figure 1: *Cumulative load of surface water as a function of areal loss. Estimations for the Schuitenbeek area were made by Schoumans and Kruijne (1995).*

For the Schuitenbeek area total cumulative loss was divided over fields with a comparable areal loss (kg P ha^{-1} yr^{-1}). It was shown that over 75% from total loss originates from fields with an areal loss of less than 4 kg P ha^{-1} yr^{-1}, and less than 10% from fields with a loss above 8 kg P ha^{-1} yr^{-1}. Thus, for this specific catchment there is no question of CSA's, but P-loss is highly diffuse. Figure 1 also shows a curve for a (hypothetical) catchment where a large part of the cumulative loss originates from fields with a high areal loss. In this case fields with a loss > 10 kg P ha^{-1} yr^{-1} show to be responsible for

70% of cumulative loss. Since P loss is less diffuse in this catchment there is a better opportunity to attack total P loss, by taking measures on the fields with a high areal P loss. Producing a curve as shown in figure 1 can be part of a procedure to evaluate possibilities for P loss reduction in a specific catchment.

As will be further discussed in par. 2.1, criteria to identify CSA's for P loss by surface runoff have been intensively studied, and relatively well known (Pionke et al., 1997; Lemunyon & Gilbert, 1993; Gburek & Sharpley, 1998). Much less is known of the criteria to identify CSA's for P loss by subsurface drainage.
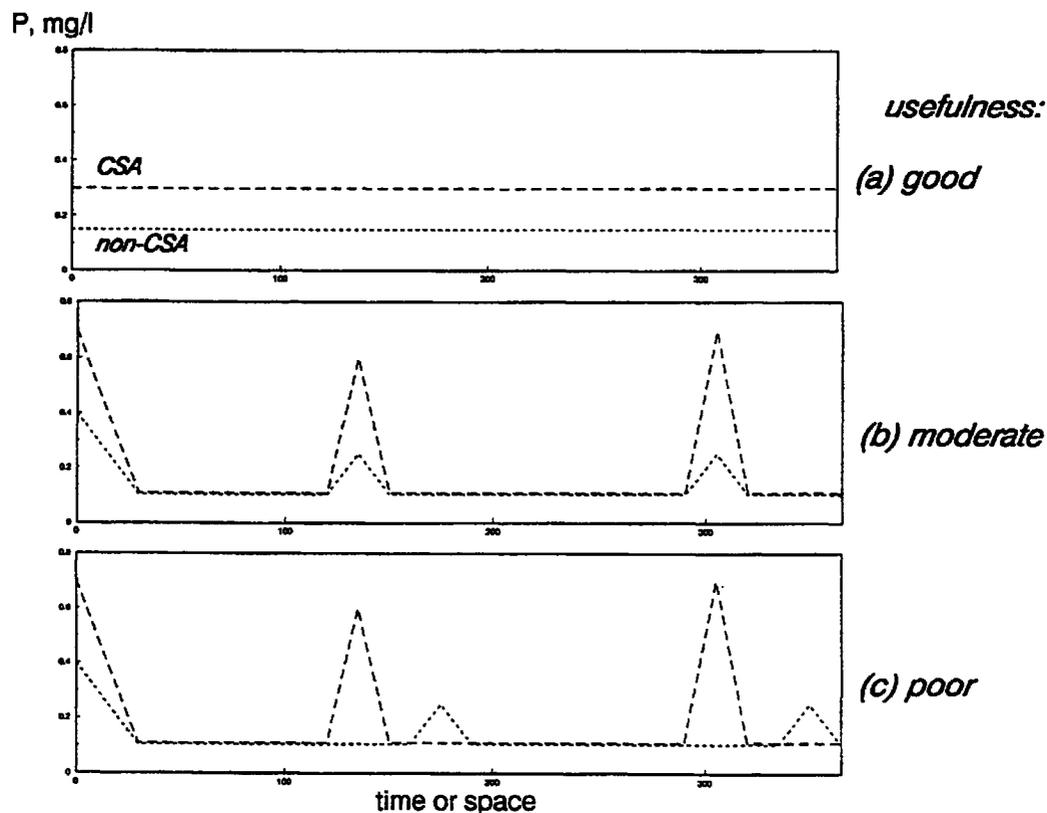
Figure 2: *Schematization of usefulness of indicators for CSA's.*

A useful indicator shows a difference between a CSA and a 'normal' field which is (more or less) constant in time, as schematized in 2a. Figure 2b shows a less useful indicator: most of the time the CSA and the 'normal' field do not show a difference, but only during certain periods a large difference is found. Only during these periods measurements will lead to detection of the CSA. Figure 2c shows an indicator which can hardly (or not) be used: during certain periods the value of the indicator is lower for the CSA than for the 'normal' field. This would necessitate continuous measurements, which in most cases will be impossible due to costs.

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Analogous to critical source areas, there can be *critical source times and events*, since one or a few events can contribute more than 50% of the annual P loss: Pionke et al. (1996) reported that about 70% of water phase P was exported from a watershed during 10% of the time defined as storm flow. However, in this study surface runoff was the main P loss mechanism; no data are available for regions where P loss by leaching was predominant.

Finally Withers (1996) stated that "further work is required for identifying high risk areas within catchments for implementation of specific land management control strategies where appropriate".
In this study it is attempted to evaluate indicators of CSA's for loss of P through subsurface transport, by evaluating the literature and (unpublished) datasets. Additionally, expert views have been asked from European scientists who are involved as members of the COST Action 832: 'Quantifying the agricultural contribution to eutrophication' [with phosphorus].

In the evaluation an important criterion will be the temporal and/or spatial variability of such an indicator. The influence of temporal variation on the usefulness is schematized in figure 2. Spatial variation is equally important: a parameter that has a large spatial variability is less useful than a parameter that is more or less constant within a field.

2. Compilation of literature

Recent book reviews about phosphorus losses from land to water were edited by Tiessen (1995) and by Tunney et al. (1997a). The contribution of agricultural phosphorus to eutrophication of surface waters has been reviewed by Foy and Withers (1995). Pathways and key factors affecting phosphate transport at catchment, field and plot scale have been reviewed by e.g. Sharpley et al. (1995), Heathwaite (1997), and Haygarth (1997). Phosphorus losses from soils to water must be examined at the national, regional and catchment scale, as well as at farm and field scale, because different key factors can be operative at the different scales. For instance, hydrological processes that dominate at the catchment scale can control which farms and fields contribute most to the exported phosphorus from a catchment.

In the following sections the literature is summarized on (i) indicators of critical source areas for surface runoff, (ii) the regional approach, and (iii) the P balance on farm level.

2.1 Indicators of critical source areas for surface runoff

In the past, it was assumed that surface runoff is occurring more or less uniformly over a watershed (Gburek & Sharpley, 1998). However, Ward (1984) developed the concept of a 'variable source area' (VSA). According to Gburek & Sharpley (1998), the basic premise of VSA is that "there is a dynamic contributing subwatershed within a topographically defined watershed that expands and contracts seasonally, as well as during a storm, as a function of precipitation, topography, soil type, geology, groundwater levels, and watershed moisture status".

For the estimation of site vulnerability to P loss via surface runoff the P index system was developed in the USA (Lemunyon & Gilbert, 1993). It considers two transport factors (erosion and runoff) and five P source factors (soil P test, P application rates with fertilizer and manure and P application methods). Each factor is given a value between 0.5 and 1.5 and next multiplied by an appropriate P loss potential weighting factor of 0, 1, 2, 4 or 8. Eventually, a product sum value is calculated for all combined factors to indicate the site vulnerability. The P index has had limited field testing and validation, but results from application to 26 grassed and cropped catchments in Oklahoma and Texas showed a close relationship between total P loss in runoff and the P index rating (see figure 3; Sharpley & Lemunyon, 1997).

To identify and implement phosphorus control practices for manure application by large dairy farms, McFarland et al. (1997) added vegetation, grazing and P application timing as important factors to calculate the P (risk) index. Haygarth & Jarvis (1997) also suggested that a period of high P export by surface runoff from grazed grassland could be explained from a combination of the presence of cattle (excreta returns and poaching), timing of inorganic P fertilizer additions and intensity of rainfall.
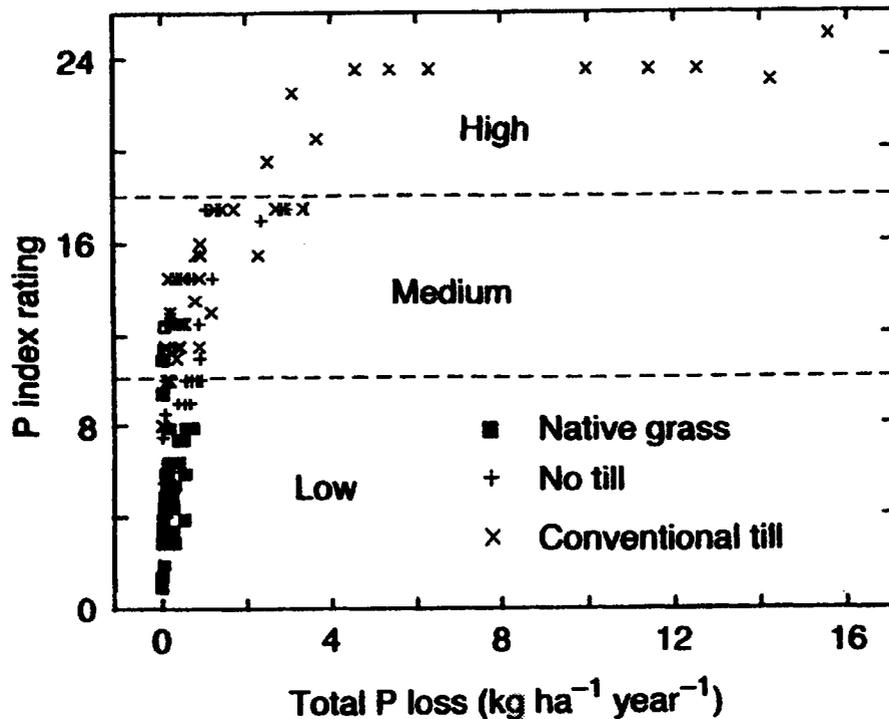
Figure 3: *Relationship between P index rating and mean annual total P loss in runoff as a function of catchment management (Sharpley & Lemunyon, 1997).*

In summary, runoff and erosion CSA's are characterized by slope induced runoff or erosion and a high content of desorbable P as time-independent factors. Time-dependent factors are (i) the fact that runoff or erosion only occur when the rainfall intensity exceeds the infiltration capacity of the soil, and (ii) that loss of P along this pathway is greatly enhanced by application of manure or fertilizers or the presence of animal excreta in periods of high rainfall intensity.

2.2 Regional approach

Examples of the regional assessment of areas where P leaching can be expected are available from several countries.

In the Netherlands, much work is done since the mid 80's at the Winand Staring Centre (formerly Stichting voor Bodemkartering) based on the P saturation of the soil profile above the mean highest water-table (MHW). As will be further discussed in par. 4.3, the P saturation of a soil is based on the molar ratio between P and (Fe+Al) in an oxalate extract of a soil sample. For a regional estimation of P sorption capacity, data on Fe, Al and MHW were derived from a data base of representative soil profiles for soil survey mapping units (Breeuwsma et al., 1986). Data for P_{ox} are calculated from natural P content and data on animal production, fertilizer use and crops (Breeuwsma et al., 1997). Combining these data leads to soil maps giving the estimated percentage of saturation of the sorption complex

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above the water table. From these maps regions can be derived where high P concentrations in the groundwater can be expected.

A different approach of studying regional differences in the Netherlands was used by Neutel (1994). It was based on data of soil samples sent to a soil analytical lab for P-fertilizing advise. For 13 agricultural areas it was calculated which percentage of the samples was classified as low, average or high with regard to P level. For the period 1990-1992 large differences were found between regions: a region with a low density of animal husbandry had 24 % of the samples with a high P status, and a region with a high density had 72% of the samples classified as high. For the whole country, between 1972 and 1992 the fraction of samples classified as low or very low reduced drastically, and the fraction (rather) high increased. A comparable approach was followed by Sims (1998) for the USA as a whole and for the State of Delaware. Based on fertility test results areas could be indicated where the majority of the samples are rated 'high' or even 'excessive' in P, due to long term fertilization or manuring.

Large scale sampling of West European regions with a high livestock density showed that a high percentage of the soils in those regions had become saturated with phosphate to a high degree, and often down to the drain depth. This caused phosphate concentrations in drainage water above the limit for eutrophication of surface water or even above the limit for the production of drinking water. A geostatistical approach, that could identify CSA's at a scale of 25 ha, showed that from the soils in four Belgian regions with a high livestock density 2-21% could be classified as P risk area and only 0.2-2.2% as P saturated area (VLM, 1997). However, to identify CSA's for P losses with drainage water the scale of sampling should go down to the average field size of the farms involved, which at least is expensive. Until now a relationship between *phosphorus saturation degree* (PSD) of the total soil profile and soil solution P concentration at the bottom of the sampled soil profile has been used (Breeuwsma et al., 1997; De Smet et al., 1996a,b, 1997; Schoeters et al., 1997; Scokart et al., 1997). Taking into account the distribution of the PSD over the soil profile in relation to drain depth could improve the identification of fields as CSA.

Catchment studies that quantify the phosphate loads of streams and rivers are complementary to the regional approach that estimates the inputs of phosphates from different sources to surface water. Pionke et al. (1997) and Sharpley et al. (1995) reviewed the hydrological, landscape and chemical controls on phosphorus loss from catchments. They conclude that P export from a catchment must be examined at the catchment scale, as well as at farm and field scale. First, hydrology related processes that dominate at the catchment scale can control which farms or fields contribute most to the exported P. Second, catchment elements between field and catchment outlets may alter the timing, the amount and the concentration of bioavailable P exported. These boundary or transitory elements can include wetlands, channels and reservoirs, and may act as storage buffers or filters (Pommel & Dorioz, 1997; Hillbricht-Ilkowska et al., 1995).

The contribution of agriculture to P discharge of seven selected European catchments varied between 1% and 42%, with area runoff varying from 0.14 to 3.9 kg P ha^{-1} yr^{-1} (Iversen et al., 1997). In general, the relationships between P concentration levels and agricultural activities in the catchments are complicated. The P load from diffuse sources ranged between 0.10-0.45 kg P ha^{-1} for subcatchments of the Odense fjord, and between 0.09-2.43 kg P ha^{-1} for subcatchments of River Avon. In catchments with a high population density the P levels are dominated by the discharges from point sources (e.g. sewage treatment plants). A good relationship between P discharge and agricultural activity and intensity can only be established in catchments where such inputs are small (Iversen et al., 1997). Since the transport of P from artificially drained catchments mainly occurs during storm events, reliable estimates of P losses can only be obtained using a sampling strategy that incorporates intensive sampling during these events (Grant et al., 1996).

A catchment study of the Poekebeek watershed, in a Belgian region with a high percentage of P saturated soils, showed the difficulty to distinguish between the contribution of diffuse arable land sources and from other sources (Hartmann et al., 1997; VLM, 1997). Mean ortho-P concentrations of streams ranged between 0.5-1.0 mg L^{-1} during wet periods and between 0.6-1.5 mg L^{-1} in dry periods. In spite of the extra P contribution by agricultural drainage water in wet periods, P concentrations of the brook water were lowered by rainfall. Since discharge rates were higher, P export was also higher during wet periods. During dry periods, P from point sources accumulated in the bottom sediments within the catchment, and was mobilized during very wet periods when drainage waters from arable land also contributed. A combination of high P concentrations and high water discharge rates caused very large fluctuations (range 3-1060 kg) in daily P export from the catchment.

In The Netherlands, data on P use (feed and fertilizer) and soil fertility level are available only on a municipal level, since the privacy of individual farmers is protected. For regional studies this is insufficient detailed.

In summary a regional approach has proven its validity for the identification of CSA's for surface runoff in the USA. For a first identification of CSA's in the case of P leaching a regional approach is useful, but for defining CSA's on field scale the availability of data on hydrology and on the P sorption capacity or P saturation of the subsoil are necessary as well.

2.3   Phosphorus balance at farm level

A phosphorus balance, the difference between P import and P export, at the farm level can give a first impression of potential P losses to surface waters. Long periods of annual P surplus caused considerable regional P enrichment of soils in Western Europe. As an illustration, average values for P surplus for several European countries are summarized in Table 1.

Table 1: *Surplus of total-P (kg P ha⁻¹ yr⁻¹) for the average farm of some European countries and regions in 1990/1991. Data from Brouwer et al., 1995.*

	country	region
United Kingdom	6	
England West		8
Scotland		3
Denmark	8	
Spain	12	
Galicia		20
Extremadura		10
Greece	15	
Ireland	15	
Switzerland	16	
Germany	21	
France	28	
Bretagne		37
Limousin		13
Belgium	36	
Netherlands	**40**	

The table shows that large differences can be found in the surplus on the P balance, not only between countries, but also between regions within a country. Eventually, such enrichments may lead to increased P losses to surface waters, from soils with a high degree of phosphorus saturation.

Differences in P balance between individual farms and farm types also received attention. Statistical nutrient budgets of Dutch farms showed mean annual P surpluses (kg P ha⁻¹ yr⁻¹) of 23 (range: -3 to 58) for arable farms, of 31 kg (range: 7-64) for dairy cattle farms, and 37 kg (range: 0-106) for mixed livestock farms (Poppe et al., 1995). Haygarth et al. (1998) compared P budgets of two contrasting grassland farming systems in the UK; on an intensive dairy farm a P surplus of 26 kg ha⁻¹ yr⁻¹ was calculated, for a hill sheep farm this was only 0.28 kg ha⁻¹ yr⁻¹.Thus, great differences exist between farms, especially related to the livestock density.

3.   Results of inquiry

An inquiry was made among participants of an European network entitled: COST 832, 'Quantifying the agricultural contribution to eutrophication'. Information was asked about their experience with indicators for CSA's. The text of the letter can be found in Appendix 2. The information received is summarized below, and will further be used in chapter 4.

Dorioz, Thonon-Les-Bains, France: Intensive studies are done on processes in watersheds. Export of P during base flow and rain storm flow are compared; P is essentially exported during rain storm flows. A high temporal variation is found in the P concentration of the surface water leaving the watersheds studied. No direct research was done on indicators for soil CSA's.

Hartikainen, Univ. Helsinki, Finland: In the past, most research was done on surface runoff and erosion, but recently more attention is paid to leaching. Rainwater bypassing a field is assumed to receive P, detached or desorbed from soil particles. This mechanism can be applied both on surface runoff and leaching. In general, the degree of P saturation of Fe/Al hydroxides is low, in Finnish soils; its determination using oxalate can be disturbed by dissolution of calcium phosphates. Water-extractable P (1:60, filtered over 0.2 μm) is assumed to be a readily available fraction of soil P which can be used to estimate the amount carried off from a field by surface runoff (Hartikainen, 1982). In cracking clay soils indications were found of P transported from the plough layer into drainage pipes by preferential flow.

Haygarth, North Wyke, UK: A number of reprints was received on transport mechanisms of P, preferential flow and influence of artificial drainage.

Hens, Leuven; Hofman, Gent, Belgium: A report was received on an inventory of P saturation of sandy soils in Belgium (VLM, 1997). The saturation of the soil profile with P was used as an indicator of potential CSA's. A regional soil sampling was performed following a grid, and soil maps were constructed using interpolation. The concentration of ortho-P in drainage water increased with the saturation degree of the soil profile on the same location, but showed a large temporal variation (in: Brookes et al., 1997, p. 266).

Leinweber, Vechta/Rostock, Germany: Reprints were sent of publications on regional inventories of P contents and saturation in intensively fertilized parts of Germany. Also, a pre-print was received of a study correlating P content of lysimeter drainage water and soil P analytical data.

Sibbesen, Tjele, Denmark: A report by Simmelsgaard (1996) was sent entitled: Plant nutrients in subsurface drainage water and soil water (in Danish). Movement of P to surface water was mainly due to surface runoff, and its amount was correlated with soil P content. A positive correlation was found between NH_4-N and P in drainage water, which indicates macropore flow.

Stamm, Zürich, Switzerland: A copy was received of a PhD-thesis (Stamm, 1997). The main subject of the work is rapid transport of P in intensively managed drained grassland soils, caused by preferential flow. High P concentrations were found in tile-drain effluent, before the peak of the water flow. After manure application, almost immediately high P concentrations were found in the effluent.

Werner, Bonn, Germany: A research report was received on soil P content and P saturation, in relation to drainage and surface water.

Withers, Winchester, UK: In a Rothamsted field experiment, the concentration of P in drainage water and in a $CaCl_2$-extract was found to increase exponentially with increasing Olsen-P content (>60 mg kg^{-1}; Brookes et al., 1997). Now this relation is investigated on other soil types; it appeared that the sharp increase in $CaCl_2$-P differed between soil types (18-100 mg kg^{-1}). This variation created some doubt on the direct applicability of Olsen-P as an indicator of P leachability. Nevertheless, an exponential relationship was found between Olsen-P and P extracted from suction cups.

Yli-Halla & Jansson, Jokioinen, Finland: In a field experiment both plant-available P in the plough layer and the P content of drainage water was determined. No significant correlation was found between both parameters.

4. Evaluation of indicators

The following characteristics of a soil or field have been used, or suggested, as indicators for CSA's:
- P content in the soil solution;
- P content of the soil (total P, plant available or water-extractable P);
- P saturation index;
- soil hydrological characteristics, leading to a high risk of preferential flow;
- P content of tile drainage water;
- P content of the bottom sediment, or of the surface water adjacent to the field.

4.1 Soil solution P content

Dissolved P in the soil solution often represents less than 0.1% of total P in the soil. After the addition of P fertilizer or manure, increased P concentrations can be present in the soil solution for some time. Temporary changes in soil conditions (pH, redox potential and ionic strength) can cause mobilization of inorganic and/or organic P and thus the actual P content of the soil solution.

Thomas et al. (1997) measured P in the soil solution, sampled with porous cups, from lysimeters installed in a silty clay loam soil. The cups were installed at 60 cm distance from pipe drains. Soil solution P content at drainage depth was much lower than P content of drainage water, but a close correlation was found between soil solution P content in the plough layer and P in drainage water. This led to the conclusion that preferential flow contributed to a large degree to P loss from the lysimeters (Thomas et al., 1997).

Ritchie and Weaver (1993) state that the potential short term leaching of P can be estimated from water soluble P in the soil solution just before winter rainfall. The potential yearly loss of P, when no fertilizers are applied, can be estimated with bicarbonate-P.

Except for the topsoil layer, limited data are available about actual concentrations of dissolved P in the soil profile. When sampling of the soil solution is attempted, a strict protocol should be followed, including analysis within 24 hours, to limit the risk of artefacts caused by sampling, sample preparation and storage (Chapman et al., 1997). The soil solution can be seen as a highly variable natural matrix containing soluble salts, colloids, and organically complexed substances. When its total P content is determined, the way of digestion can strongly influence the P content found (Rowland and Haygarth, 1997). Centrifugation of undisturbed soil cores has been used to isolate soil solution from the soil

pores involved in water transport. Homogenization of soil cores before the centrifugation can change the composition of the soil solution considerably (unpublished results AB-DLO).

Due to the complexity of obtaining reliable data of P in the soil solution, sampling of the soil solution has hardly been used as an indicator of CSA's. As an alternative, extraction of homogenized soil samples with water or a dilute salt solution has been used to simulate the soil solution (see 4.2).

4.2 Total and water-extractable P content

Intensive use of P fertilizers during the last 40 years, in amounts larger than crop needs, has resulted in a general increase in total soil P content of many agricultural soils in Europe. The clear need for an improved, predictive understanding of the P release behaviour of these overfertilized soils has led to a comparative study of the soil test methods used most often, in relation to dominant soil properties (Barberis et al., 1996). Tests of available soil P showed highly positive correlations between the six P extraction methods used. The amount of P extracted varied, and increased in the order water-P < Olsen-P < calcium acetate-lactate P < Fe oxide strip P < mixed cation/anion exchange resin P < anion exchange resin P. A first evaluation of potentially releasable P could be made with these simple soil P tests in combination with soil pH (Barberis et al., 1996).

Total soil P
In regions in Germany with a high livestock density total soil P contents exceeding 8 g kg^{-1} have been found (Leinweber, 1996). Total P contents of mineral soils > 1 g kg^{-1} and of organic soils > 3 g kg^{-1} can be indicative of excessive historical P gifts, taking into account the P content of the parent material of the specific soil. Excessive historical P application can also be estimated from P budget calculations based on amounts of P applied through fertilizers and manure, corrected for P removal with harvested products. Groenenberg et al. (1996) distinguished six classes of historical cumulative P gift for Dutch soils (period 1970-1990), and showed that applications (kg P ha^{-1} yr^{-1}) ranged from 70-300 for maize land and from 40-100 for grassland and other land use. However, simulation modelling showed that the actual P losses to surface waters are strongly affected by hydrological conditions (mean highest groundwater table and drain water discharge) as well as by the degree of P-saturation (PSD).

Thus, high total soil P contents, as a result of excessive historical P applications, only can be used as a first indication of potential source areas for P loss to surface water. As a rule of thumb, it is assumed that if the amount of plant available or soil test P is at least twice as large as the accepted advised level for an average crop, substantial P losses can result from these soils by surface runoff or by leaching. Whether actual loss occurs will depend on the hydrology of the soil and on the degree of P saturation of the soil profile (Sharpley, 1995).

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Water-extractable P

The use of water-extractable P content (Pw) of a soil as an indicator for plant available P was introduced in The Netherlands by Sissingh (1971) and Van der Paauw (1971). Hartikainen (1982) postulated that water-extractable P is a readily available fraction of soil P that can also be used to estimate the amount carried off from a field by surface runoff.

For a silt-loam grassland soil, Pote et al. (1996) found that P content of runoff water correlated better with water-extractable P, with P extracted by FeO-paper (Sissingh, 1983) and with the oxalate saturation index than with traditional soil tests (Mehlich III, Bray I and Olsen). Also for grassland soils, Blennerhassett (1998) found a better correlation between P content in artificial runoff and water-extractable soil P content than with Olsen-P. However, under very dry soil conditions P in runoff increased much more than water-extractable P, which reduced the possibility to use a general applicable relationship between P in runoff and in a soil water extract. Pote et al. (1999) also showed an influence of runoff volume on the relationship between P in runoff water and water-extractable soil P. When comparing three Ultisols, a uniform relationship between both parameters could be found after correcting for differences in runoff volume (Pote et al., 1999).

Beauchemin (1996) found a weak correlation between P in drainage water and soil P status for 27 sites. However, when homogeneous soil classes were used, the correlation was better: a combination of Pw and [Pi/(Al+Fe)]ox gave the best result[1].
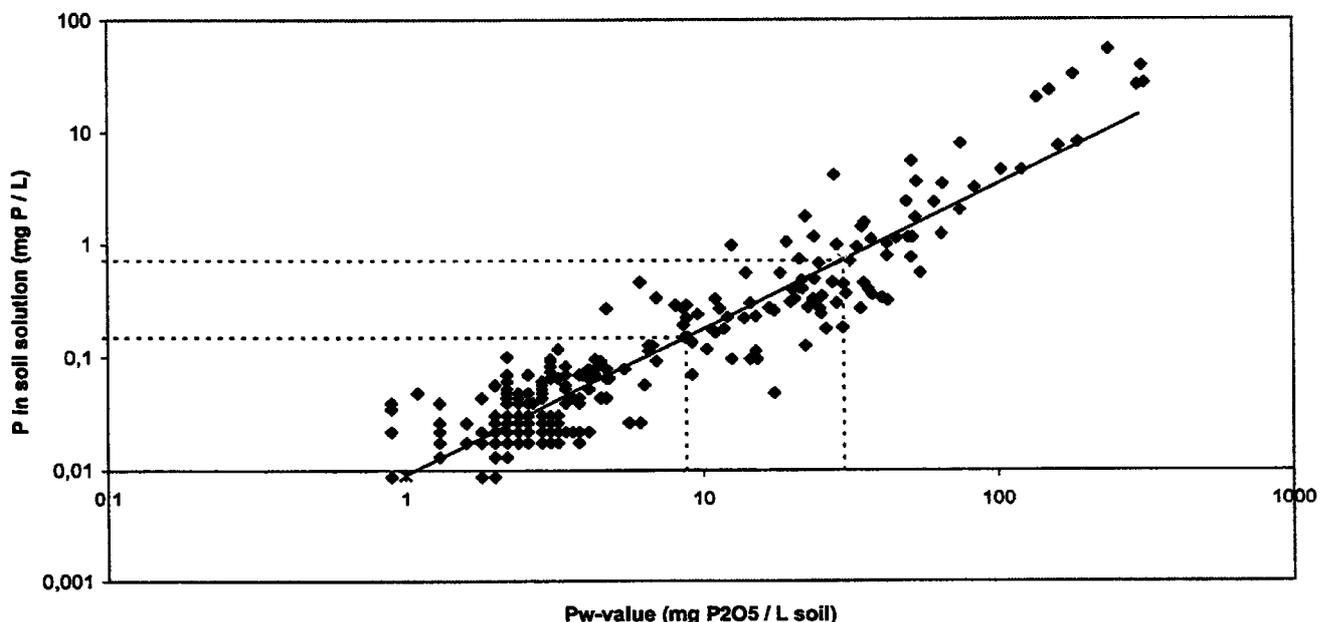
Figure 4:   *Relationship between Pw and P measured in centrifuged soil solutions (data from Blaauw et al., 1988). Drawn line corresponds to: $y = 0.0091.x^{1.29}$*

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[1]   [Pi]ox denotes P measured directly in an oxalate extract, without previous digestion.

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Blaauw et al. (1988) measured P content in the centrifuged soil solution and water-extractable P (Pw) of samples from 41 soil profiles, containing 6 to 9 layers. A total of 10 field plots were sampled, on clay, sand, loess, and reclaimed peat soils. Figure 4 shows the correlation between Pw and the P content of the soil solution of different soil types. Although the data are plotted on a double-log scale, it appears that Pw can be used as an indicator for soil solution P content.

In appendix 1 separate figures are shown for the different soil types. From these figures it appears that, when the relationship in figure 4 is used, for clay soil the P content of the soil solution is slightly over-estimated, and for reclaimed peat soils it is slightly under-estimated. As also shown in appendix 1, water-extractable P increases during dry periods, probably due to breakdown of organic material. Although fluctuations with time were found in water-extractable P, these were not very large, implicating that Pw is a relatively stable soil parameter.

Gächter et al. (1996, 1998) investigated P losses from drained grassland soils in Switzerland. Concentrations of P in drainage water and in surface runoff during an exceptionally heavy rain storm were about equal, frequently exceeding 1 mg P L^{-1}. This led to the conclusion that in periods of high discharge rates, soil solution from the plough layer moved to the drainage ditches via preferential flow through macropores. Gächter et al. (1998) proposed to use the water-extractable soil P content as an indicator of the P content of the soil solution of the topsoil layer, according to figure 4.

4.3 Soil P saturation degree (PSD)

For non-calcareous light-textured soils another approach has been followed by Van der Zee et al., (1990). A relationship was shown between the phosphate sorption saturation degree (PSD) of the soil profile between the soil surface and the groundwater, and the dissolved reactive P (DRP) concentration c (in mg L^{-1}) in the soil solution at a reference depth:

$$PSD = \gamma Kc / (1 + \gamma Kc)$$ (1)

where: γ = ratio between total fixed P and reversibly adsorbed P
  K = Langmuir adsorption constant

Assuming that the phosphate sorption capacity (PSC) of a soil can be calculated as:

$$PSC = 0.5 \times (Fe_{ox} + Al_{ox}),$$ (2)

where: Fe_{ox} and Al_{ox} are oxalate extractable soil iron and aluminium (mole kg^{-1}), PSD equals:

$$PSD = P_{ox} /PSC = P_{ox} /[0.5 \times (Fe_{ox} + Al_{ox})],$$ (3)

Note that in some publications calculation of PSC differs from eqn. (2), e.g. taking into account more crystalline forms of Fe (Borggaard et al., 1990) or giving an uneven weight to Fe and Al (Werner and Pihl, 1998).

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Using equation 1, with average parameters for Dutch sandy soils ($\gamma=3$; K=35 L/mmol; c=3 μmol/L), Van der Zee et al. (1990) calculated that a DRP equilibrium concentration of 0.1 mg L^{-1} in the soil solution near the mean highest groundwater level will be reached at about PSD = 0.24. This implies that the Dutch limit for DRP in groundwater or drain water would be reached long before the P sorption capacity has been fully used (PSD = 1.0). However, due to field heterogeneity, PSD should be interpreted with caution. Van der Zee et al. (1990) also assumed a uniform application of P to, and withdrawal from, the soil column. This does not have to be the case for a specific field, however.

Within a sandy loam soil region in Belgium, the PSD had a median value of 0.28 for the 0-90 cm soil profile for 296 locations. This meant that almost half of the sampled profiles were at, or above the critical PSD value and at these locations drain water DRP concentrations > 0.1 mg L^{-1} could be expected (De Smet et al., 1996a). The field data could be fitted rather well to equation 1, indicating that the model proposed was also valid for the Belgian sandy loam soils, although a somewhat lower value of K was found after fitting the data (Brookes et al., 1997).

Leinweber et al. (1997) determined P sorption capacities and P saturation of two regions with different livestock densities in Germany. As expected, both P_{ox} and PSD were highest in the region with the highest livestock density. However, it was found that both P_{ox} and Al_{ox} had a relatively large small-scale variability, constraining regional assessment of risks of P losses. The variability of P_{ox} was attributed to manuring history of a field; variation in Al_{ox} might be explained by anthropogenically caused differences in organic matter content. A large short-distance variability of both Fe_{ox} and Al_{ox}, and an influence of organic matter on Al_{ox} were also reported by Lookman et al. (1995, 1996). They also reported a large build-up of Fe_{ox} in the 0-30 cm layer near a ditch. The build up was attributed by Lookman et al. to transport of Fe^{2+} from the soil in between ditches to the ditch and precipitation as Fe^{3+} due to oxidation in its surrounding. Depositing dredged sediments on the borders of the ditch could also have caused this effect. Leinweber et al. (1999) calculated correlations between soil test values (0-30 cm) and P concentrations in effluents from lysimeters (1.25 m depth). The best correlation with effluent P was found for $NHCO_3$-P, oxalate-extractable P and PSD. No correlation was found between topsoil water-extractable and effluent P.

For individual soil samples, a correlation was found between the PSD and P concentrations in either water extracts (Lexmond et al., 1982; Schoumans et al., 1991; Chardon, 1994; Pihl & Werner, 1993; 1995) or 0.1 M KCl extracts (Lookman et al., 1996). However, as shown in figure 5, a (large) scatter can be found around regression lines, which can be explained by differences in soil type, pH, and history of the P addition. For separate locations or soil types a better correlation was found than after combining data from different soil types (figure 6). Note that in both figures the molar ratio P/(Fe+Al) in oxalate extracts was used instead of the PSD.
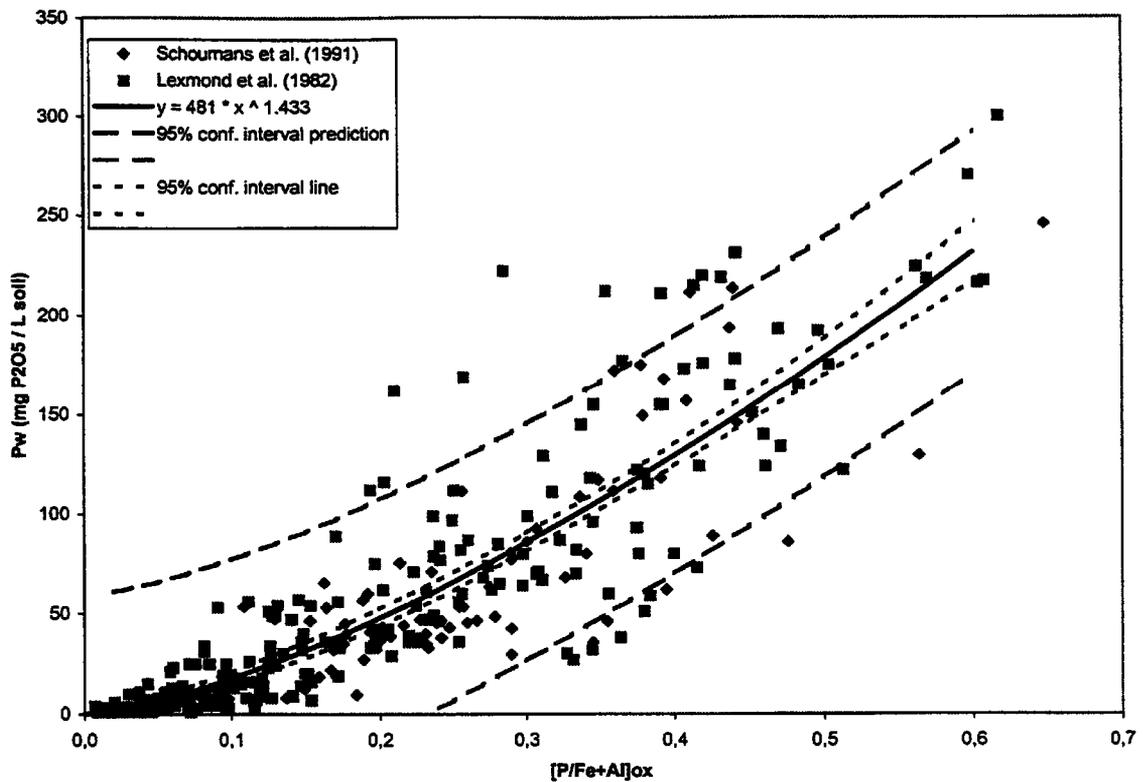
Figure 5: *Relationship between molar ratio P/(Fe+Al) in oxalate extracts and water-extractable P (Pw, 60:1 water:soil ratio), for different soil type (Chardon, 1994; data Lexmond et al., 1982, and Schoumans et al., 1991).*
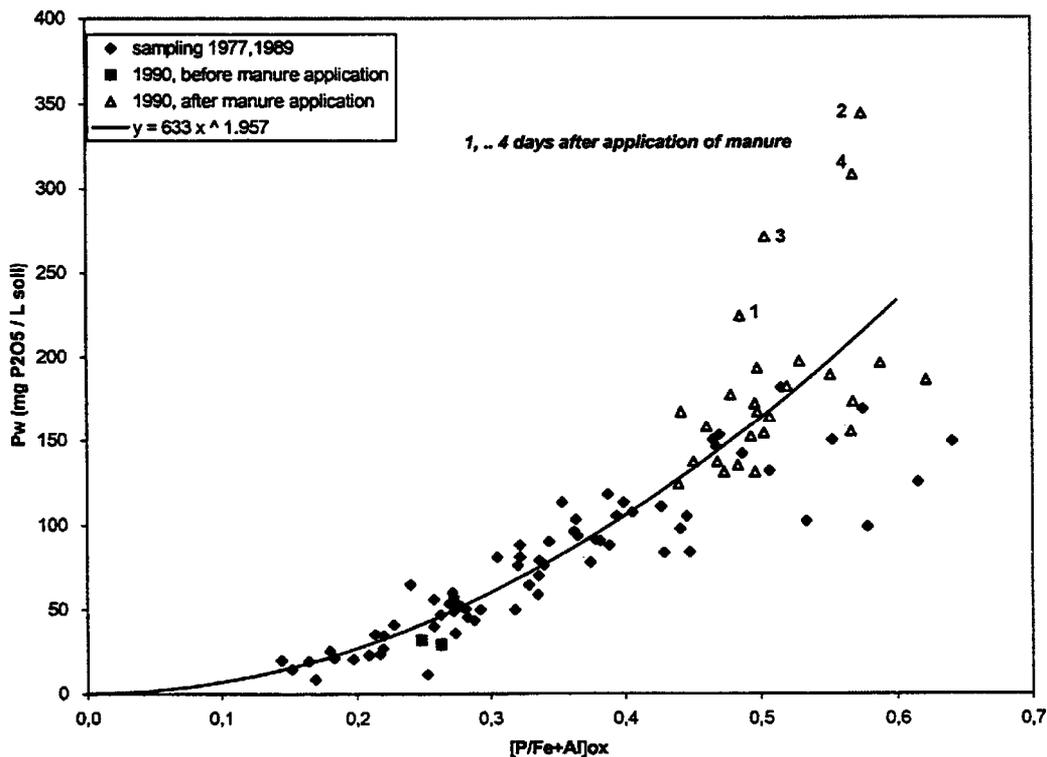
Figure 6: *Relationship between molar ratio P/(Fe+Al) in oxalate extracts and water-extractable P (Pw), for a sandy loam soil (Chardon, 1994).*

The scatter mentioned above makes it difficult to choose a widely applicable threshold value for PSD that yields acceptable P concentrations in soil solution or drainage water. If the PSD value of a soil would be used, it seems most appropriate to use the soil layer just above the groundwater level or around artificial drains for soils where no preferential flow occurs. In soils where bypass or preferential flow through macropores is the major transport pathway to the drain, the PSD value of a P rich surface layer may be more appropriate to predict P concentrations in drain water.

Beauchemin (1996) stated that risk assessment of P leaching should include both an intensity parameter of P desorption (e.g. P-water) and a capacity term related to P accumulation (e.g. PSD). Thus, the higher the PSD of the A-horizon is, the lower the P_w value of this horizon should be to prevent contamination of drainage water.

4.4   Soil hydrological influences

Rainfall and hydrology can be considered the driving parts of the P transfer system. Pathways of P transfer are associated with hydrological pathways, which can be considered at the scales of soil (centimeters to meters), slope (meters to hectares) and catchment (kilometers and square kilometers). Haygarth (1997) gave a pictorial view of the P transfer pathways at different scales and defined the terms involved at these scales: At the catchment scale rivers form high order drainage networks, which receive all drainage water other than that which has gone to groundwater. Streams and ditches form a low order, small scale surface drainage pattern and are often fast flowing. In times of heavy rainfall roads and agricultural tracks enhance water transfer of the land. At slope scale surface runoff is transport which occurs exclusively over the surface during periods of heavy rainfall. This is important in P transfer through soil erosion. Subsurface lateral flow occurs along or above an impermeable subsoil horizon. Drainage by pipe and mole drains involves the placement of artificial outlets for water in the subsoil.

Artificial drainage can promote preferential flow (or macropore flow), the transport mechanism by which relative high volumes of water flow through a small portion of the soil volume (Simard et al., 1999). Preferential flow can either be caused by (i) a local low hydraulic conductivity, forcing the water to soil parts with a normal conductivity, or (ii) by soil parts that have a high hydraulic conductivity, attracting water from the surrounding soil.
A low hydraulic conductivity can be caused by soil material with a low porosity, e.g. clayey material, or by organic matter causing water repellency of a soil (Dekker, 1998; Ritsema, 1998). According to Ritsema (1998) water repellency occurs in natural and agricultural soils all over the world, but its influence on transport of agrichemicals (like P) is poorly understood due to insufficient knowledge about the water flow and transport processes.

A high hydraulic conductivity can be found in macropores (> 100 μM), originating from e.g. soil cracks, root channels, and earthworm borings (Edwards et al., 1993). Preferential flow through these pores only occurs when the rainfall intensity exceeds the infiltration capacity of the bulk soil (Simard et al., 1999). Macropores can be destroyed by soil tillage. In Ontario, Canada, conservation tillage, e.g. zero- or ridge tillage, is considered the best management practice to control soil erosion from agriculture (Gaynor and Findlay, 1995). It was found that total P loss from fields treated with conservation tillage was larger than on conventionally tilled fields, which was attributed to preferential flow in the former. Large peaks of ortho-P in tile drain water were found both with zero and ridge tillage, indicative of preferential flow (Gaynor and Findlay, 1995). Simard et al. (1999) compared P content of drainage water under fields treated with either mouldboard ploughing or with ridge tillage. The latter treatment showed a much higher amount of P in the drain water than the former, attributed to preferential flow through the macropore network in the soil.

Drainage studies on a silty clay loam soil at Rothamsted showed that total-P concentrations in drainage water were less than 0.3 mg L^{-1}. However, this concentration strongly increased on fields with Olsen-P (0.5 M $NaHCO_3$-extractable P) contents in the plough layer exceeding 60 mg kg^{-1} soil (Heckrath et al., 1995). In the UK, this Olsen-P content is classified as very high (Tunney et al., 1997b). The observations were in agreement with a bypass flow mechanism in the subsoil, causing soil solution from the plough layer, with elevated P concentrations, to reach the drains without much reduction in P concentration through adsorption by the large P sorption capacity of the subsoil (Thomas et al., 1997).

Simmelsgaard (1996) found a positive correlation between NH_4-N and P content of drainage water. This is an indication of macropore flow, since leaching of NH_4-N is usually inhibited by nitrification in the top soil.

Jensen et al. (1998) studied P transport in undisturbed subsoil blocks in the laboratory. They compared P mobilized by either distilled water or by a dilute salt solution (simulating rain water and soil solution, respectively). Strong indications for preferential transport of P were found in the columns; a decrease of the ionic strength of the solution lead to an increase of P transport.

Turtola and Jaakola (1995) found peaks of P in drainage water soon after fertilizer application on a heavy clay soil followed by a drain discharge of only 7-15 mm, which indicated preferential P movement.

Hooda et al. (1999) measured losses exceeding 5 kg total P ha^{-1} yr^{-1} from grassland soils. These amounts were far more than earlier astemations. Subsoils did clearly not retain a significant proportion of P leaving the topsoil, despite having a large P retention capacity. This was ascribed to preferential flow, e.g. through worm-channels. Stamm et al. (1998) measured water discharge and P content at the outlet of a drainage system of permanent

grassland in Switzerland. Concentration peaks of P in the drain water coincided with peaks in water discharge, indicating preferential transport through the soil profile. Vertical worm burrows to a depth of 80 cm were the dominant structures that explained this fast transport (Gächter et al., 1998). Since population densities of deep dwelling earthworms are often very high in permanent grassland under Swiss environmental conditions (Daniel, 1992), it was postulated that preferential transport can be the normal case rather than the exception (Flury et al., 1994; Stamm et al., 1998).

In conclusion preferential flow can cause transport of soil solution from the topsoil layers directly to tile drains or to the surface water, without sorption of P to the subsoil. It is a phenomenon that can be widespread, especially in (permanent) grassland soils or in arable soil under reduced tillage. Indications of preferential flow are: (i) a fast response of P in drain effluent to the effluent hydrograph, and (ii) the absence of a relationship between soluble P in the subsoil and P in drain water. When preferential flow is important for transport from a given field, sampling of the topsoil will give the best indicator for CSA's.

4.5   Tile drainage water P content

Spatial variation
Detailed measurements at drain outlets showed significant differences in P discharges between drains, that reflected differences in the PSD within the field and in drain depth, (Hartmann et al., 1997; VLM, 1997). Thus, measurements made on a single drain reflect only P losses at the sub-field level, that may not be representative for the whole field. Considerable intra-field spatial variability of PSC and PSD were also found in a transect study around a field drainage ditch (Lookman et al., 1996).
Meinardi and Van den Eertwegh (1997) sampled drain water on farms situated on clay soils. A farm in the Dutch province of Zeeland (Colijnsplaat) showed significant differences in drain water composition between the northern side and the southern side of the fields. The drain water at the southern side showed higher Cl and P contents, especially during periods with Cl-contents near 200 mg L^{-1}. It was concluded that this was caused by periodically occurring discharge of nutrient rich seepage water from marine origin. Comparable results were obtained on another marine soil by IWACO (1995); see also next paragraph.

Temporal variation
Patterns of P loss with drainage water have been shown to vary considerably with flow rate. A small number of storm events, or intensive rainfall periods, are often responsible for the major part of annual P losses with drainage water. Bolton et al. (1970) found that annual P loss was positively correlated with annual drain flow rates, indicating that nutrient concentration varied less than flow rate. A large part of the temporal variation in loss rates was related to the interaction of short-term events with slurry spreading, fertilizing, or poaching of land by animals in the winter (Haygarth & Jarvis, 1997; Hooda et al., 1997; Johnston et al., 1965; Lennox et al., 1997; Stamm et al., 1997; Smith et al., 1995; McAllister & Stevens, 1980).

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A regular sampling programme with fixed intervals will result in sampling a high percentage of base flow, resulting from groundwater with a low P content. This can underestimate P loss by more than 50% (Grant et al., 1996). Continuously monitored P losses from a catchment were used to assess the errors associated with losses estimated from simulated discrete sampling programmes (Stevens & Stewart, 1981). They showed that P losses based on log-log rating curves consistently underestimated the true losses by around 50%. Therefore, quantification of total P loss via drainage water requires that P losses at high discharge events should be accurately assessed. However, such events can be short-lived and unpredictable in their occurrence. For this purpose drainage water should be monitored continuously in a flow-proportional way (Grant et al., 1996).

In Belgium, tile drainage water, from 19 fields receiving different amounts of P in the past was sampled during two winter seasons (Brookes et al., 1997). Large differences in P-content were found, both between fields and between individual samplings. However, differences between fields were not found during all samplings. Also in Belgium, VLM (1997) found a large temporal variation in P content of drain water of a field with a high soil P level. At the start of a rainfall period the P content of the drainage water was equal to the soil solution on drain level (0.5 mg P L^{-1}). After several weeks of rainfall, the water table between the drains raised to a level near the soil surface, and lead to a strong increase in drainage water P content (1.8 mg L^{-1}).

Stamm et al. (1998) found large variations in P content of drain water during a short time-interval (hours); minimum and maximum concentrations differed 2-5 fold. Although a log-log relation between drain water P content and drainage rate was found for different drainage events, the relationship varied between events. Haygarth and Jarvis (1995) found a two-fold increase in total-P content of drainage water within 3 hours during a storm; this increase was attributed to particulate forms of P. A seasonal variation was also found: in spring the MRP concentration was twice as high as during autumn. It was very low during summer.

R.M.Perrot (New Zealand, pers.commun.) also found a large temporal variation in the P concentration of drain water of grassland soils with drainage tiles on ca. 40 cm depth.

Beauchemin et al. (1998) measured total P in drain water of 27 sites in 1994 and 1995. On 24 sites P content was low (< 0.06 mg P L^{-1}). Due to temporal variation the field with the highest P content in the drain water in 1994 had the lowest in 1995. The difference between both years could be attributed mainly to particulate P (> 0.45 μm) in the samples.
Hergert et al. (1981a,b) measured P content of tile effluents from manured land, and found that P concentration increased and decreased with flow rate. The P content was negatively correlated with the Ca content of the drainage water. It increased sharply at Ca-concentrations below 5 mg L^{-1}.

Chardon et al. (1997) found a strong negative correlation between Cl and P content of effluent from maize grown, outdoor lysimeters. Highest total P contents (up to 0.3 mg. L^{-1}) were found when Cl was < 5 mg.L^{-1}. This was attributed to preferential flow of (nearly) undiluted rainwater, mobilizing colloidal forms of P (Chardon et al., 1997). Comparable correlations (high P, low Cl) were found by IWACO (1995) in soil solution samples from suction cups on farms situated on a sandy soil and a peat soil. A farm on a marine clay soil showed the opposite: highest P and N concentrations (> 3 mg P L^{-1}) were found when Cl exceeded 1000 mg.L^{-1}, which was attributed to seepage of nutrient rich water of marine origin from the subsoil.

The conclusion is clearly that although the P content of drain water gives some indication of CSA's, both a spatial and a temporal variation is found. For a given field, large differences can be exist between individual drains, and peak values are alternating with much lower values. A reliable comparison of P losses between fields would require continuous sampling of a number of drains, which is very expensive. Moreover, drain water can originate from a nutrient rich subsoil; in that case, treating the topsoil will evidently be useless. As a consequence, drain water P content does not seem to be a useful indicator of CSA's.

4.6 P content of adjacent surface water and the bottom sediment

After leaching to surface water P will either remain in the water phase or sorb onto the sediment. High P concentration in the water or sediment can be an indication of a CSA adjacent to a ditch.

Adjacent surface water
Oranjewoud (1992) investigated P contents of the surface water and the P saturation of the soils from adjacent fields on 37 locations in the Dutch Province of Friesland. No relationship could be detected between P saturation of the soils and P in either the groundwater or surface water.

Sallade and Sims (1997) investigated sediments and surface water in drainage systems in Delaware (USA). The P concentrations in the water were variable, being highest in summer, and lowest in winter; in spring total P strongly increased. Possible explanations are: (i) soil erosion or re-suspension of ditch bottoms during heavy storms, (ii) release of colloidal organic P upon decomposition of organic matter, and (iii) spring algal blooms (Sallade and Sims, 1997).
Dorioz et al. (1998) investigated P storage, transport and export dynamics in a river watershed. The P content of the river water strongly depended on (present and past) weather and discharge conditions. At low discharges P concentrations were also low. P originated from point sources, and a net P accumulation occurred in biota and sediment on the riverbottom. At higher flow rates urban runoff and release of sediment P were

added to the existing P burden by point sources. When soils were already wet before heavy rainfall, both urban and (especially) soil surface runoff were large. P concentrations remained low due to dilution, but export of P was large due to the high discharges.

In conclusion, due to the temporal variability of the P content, ditch water is probably not an ideal parameter for detecting CSA's.

Bottom sediment

Sediment P contents in agricultural drainage systems were analysed by Sallade and Sims (1997). It was found to decrease with depth. The sediment P contents, measured according to Mehlich-1, were much higher than those of adjacent subsoils, and were comparable with values for topsoils. Since the sediments were analysed only once in this study, no indications on temporal variability was given.

Dorioz (1996a) reported that P, settled in drainage ditches during periods of low discharge, can be resuspended and exported during periods of high discharge. Typical for the resuspension of sediment is that the peak of the (mainly particulate) P content of the water is observed before the peak of the water discharge (Dorioz, 1996b). Thus, sampling of the water bottom in a period when most of the sediment has been removed after a storm flow would give relatively low P contents. Sampling a long period after removal of sediment would give relatively high P contents of the sediment. In this way the P content of the bottom sediment of a ditch would be independent of the presence nearby of CSA's. For this moment, lack of data do not allow further conclusions on the usefulness of sediment P for determination of CSA's.

5.   Discussion and conclusions

In this study critical source areas (CSA's) in a watershed for P loss through the subsoil are defined as areas that contribute significantly more than other areas to the P loss to the surface water. After identifying these areas, specific control or remediation strategies can be implemented in order to reduce total P loss within the watershed. In order to detect CSA's, hydrological information is essential. A short distance from a field to the surface water, or the presence of artificial drainage, is a prerequisite for a field to be assigned as a CSA.

Total P content of the topsoil can only be used as a first indicator of a critical source area. For further interpretation, information is needed on the (remaining) binding capacity, especially of the subsoil.

A phosphate saturation index of the soil profile was found to correlate with P loss rates. A number of analytical methods have been developed for the determination of the index, but the various methods have often not been tested on different soil types.

The P content of the soil solution is a direct predictor of runoff or drain water P content. However, sampling is difficult, and sample treatment may have a large influence on the result. This aspect is responsible for the fact that soil solution samples are seldom used in practice on a large scale as a predictor for phosphate saturation of soils.

The P content of (artificial) drainage water is directly related to P loss, and could therefore be a good indicator. However, it was found that the P concentrations can drastically change on short term intervals, especially during periods of heavy rainfall. Also, a large difference was found between P contents of samples from individual drains. Continuous sampling of several drains is required for getting a complete picture of P loss from a field via drains. This makes it too costly for application in a watershed when differences between areas or fields must be detected. A complicating factor with drainage water is the fact that the water can also originate from nutrient-rich layers below the drains.

Water-extractable P content (P-water) of the soil has been widely used for the prediction of P in runoff water, and shows a good correlation. It was also found that P-water in the topsoil correlates with drain water P in case of preferential flow. Compared with other parameters the temporal variation is low; slightly increased values were found during dry periods only. Therefore it is concluded that water-extractable P content as a method is the most promising for the detection of critical source areas.

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Soil sampling for the determination of P-water should not be done shortly after fertilization. Sampling in spring, before any application of fertilizers, would be appropriate.

The sampling depth depends on the local hydrology. If there are no indications for preferential flow, samples should be taken from the subsoil, near the average highest groundwater level or near artificial drains. Preferential flow can be expected in cracking clay soils, in dried organic soil, in permanent grassland where earthworms have created deep and continuous soil pores, and in water repellent soils. In this case, samples should be taken from the topsoil.

In practice, large differences exist between methods used for the determination of P-water (e.g. use of field-moist or dried soil, soil-solution ratio, and filtration procedure), so some standardization is necessary on this point.

6.   References

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Appendix 1   Water-extractable P

In the following, more information is given on the relation between water-extractable P and P in the soil solution, and on the temporal variability of Pw.

Figure 1a of this appendix shows the correlation between Pw and the P content of the soil solution for loess and reclaimed peat soils, figure 1b for sand and clay soils. Both figures contain the same regression line, calculated on basis of all data. From these figures it appears that, when the relationship from figure 4 of the main text is used, calculated on basis of all data, for clay soil the P content of the soil solution is slightly over-estimated, and for reclaimed peat soils it is slightly under-estimated.
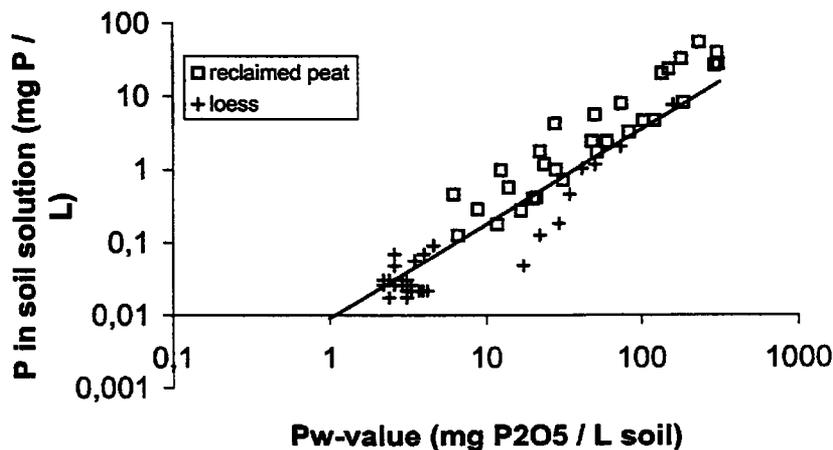
Figure 1a: *Relationship between Pw and P measured in centrifuged soil solution, loess and reclaimed peat soil (data from Blaauw et al., 1988).*
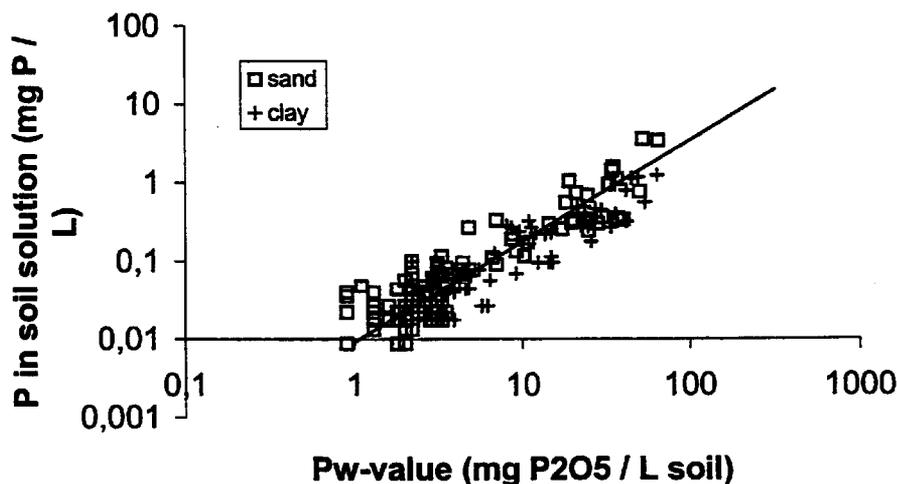
Figure 1b: *As figure 1a, sand and clay soils.*

Temporal variability of water-extractable P was investigated by Van der Paauw (1962) and Prummel (1974). Results of Van der Paauw (1962) are shown in figure 2 of this appendix, where both water-extractable P (10:1 water:soil, 24 hr extraction at 50 °C) and deviations of average rainfall for that location are given. It appeared that P-water increased during dry periods, and decreased during wet periods. Prummel measured monthly the Pw (60:1 water:soil, 24 hr incubation and 1 hr shaking, all at 20 °C) on a clay soil. Results are shown in the figure for a field that received no P and a field that received 120 kg P₂O₅ ha⁻¹ yr⁻¹. Fertilization increased Pw sharply, ploughing influenced Pw negatively. Results are shown in figure 3 of this appendix.
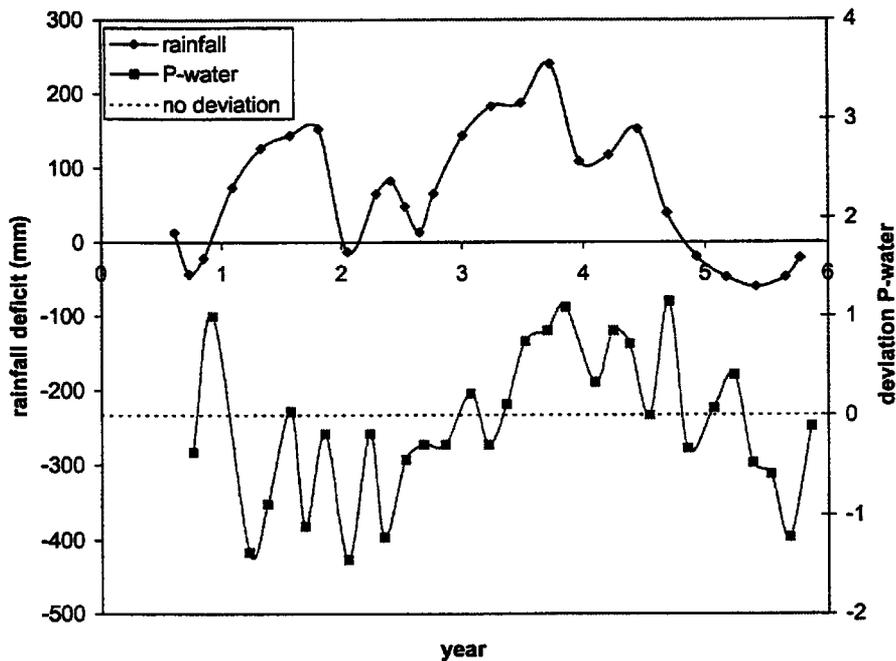
Figure 2: Change with time in water-extractable P (Pw) and deviation from average rainfall at 7 experimental fields; extractable P increases during dry periods (Van der Paauw, 1962).
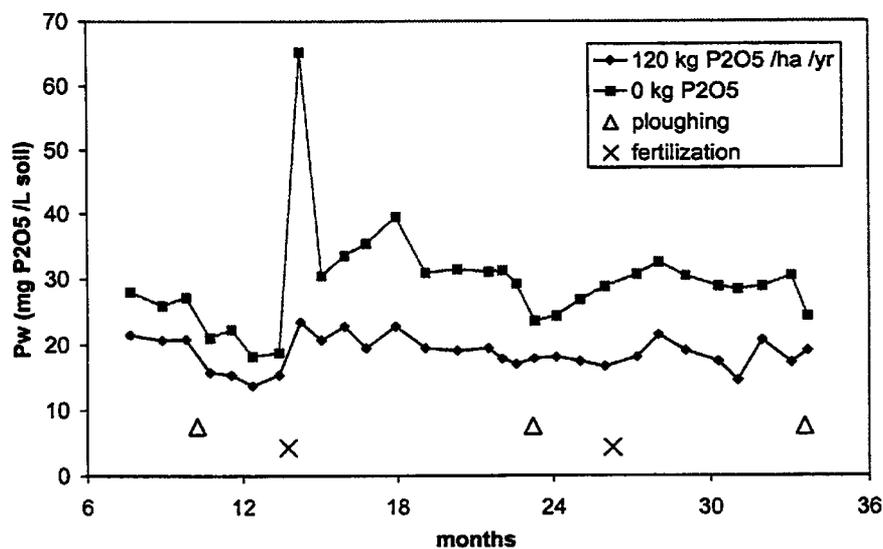
Figure 3: Change with time in water-extractable P (Pw) for fields receiving either 0 or 120 kg P₂O₅ ha⁻¹ yr⁻¹. The symbol P denotes fertilization, the symbol p ploughing (Prummel, 1974).

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Appendix 2   Text of inquiry on experiences with indicators

Dear colleague,

You receive this letter since you are, or will be involved in COST 832 (Quantifying the agricultural contribution to eutrophication). As a part of an ungoing study we would like to ask you some information.

In 1996 a Dutch survey was made of potential remedial measures for phosphorus leaking agricultural fields[1] (see appendix for an abstract). At present we make an inventory of field indicators which can be used to point out P leaking fields within a watershed. The term which can be used for these fields is 'critical source area [field]'. In the USA much work has been done on detecting areas which significantly contribute to P losses via surface runoff (e.g. Pionke, 1997)[2]. However, in the Netherlands, which is a flat country, the contribution of surface runoff to P losses is assumed to be small. Therefore we are looking for indicators which can be used to identify fields which loose unacceptable amounts of P via subsurface transport or via (tile) drainage. An overview of possible indicators, based on our present knowledge, is given in the table below.

TABLE   Possible indicators of P leaking fields:

- *soil P content (total P, plant available or water-extractable P) in the plough layer, or near the mean highest groundwater level (MHGL)*
- *P saturation index of the soil profile, or of soil samples near the MHGL*
- *P content in the soil solution in the plough layer, or near the MHGL*
- *soil hydrological characteristics, leading to a high risk of preferential flow*
- *P content of tile drainage water*
- *P content of the bottom sediment, or of the surface water adjacent to the field.*

Our questions are the following: (I) do you have any experience (both with positive and negative results) with the use of (one of) these indicators, and (II) do you know other indicators than those mentioned in the table? Both published[3] and preliminary results can be used for our study.

The usefulness of indicators will be studied in a literature survey and by using existing datasets. Next, a field study is planned to test the most promising indicator(s).
The results of this inquiry will be a part of the final report of the study (in English), which will be distributed among the respondents. Below you can find a list of colleagues who received this letter. If you can give names and addresses of persons who can also contribute to this inventory it would be highly appreciated.

Sincerely yours,
W.J. Chardon

On behalf of:

H.G. van Faassen	AB-DLO, Haren	B. Fraters	RIVM, Bilthoven
P.C.M. Boers	RIZA, Lelystad	O.F. Schoumans	SC-DLO, Wageningen

[1]Chardon WJ, Oenema O, Schoumans OF, Boers PCM, Fraters B & Geelen YCWM 1996. Exploration of options for management and restoration of phosphorous leaking agricultural soils (in Dutch). Report of the Netherlands Integrated Soil Research Programme, vol 8, Wageningen, Netherlands.

[2] Pionke HB, Gburek WJ, Sharpley AN & Zollweg JA 1997. Hydrological and chemical controls on phosphorus loss from catchments. pp 225-242. In: H Tunney, OT Carton, PC Brookes and AE Johnston (eds) Phosphorus loss from soil to water. CAB International UK.

[3] Publications in the following languages can be read: English, German, French, Spanish, Portuguese, Italian.

This inquiry was sent to:

E. Frossard, Switzerland
C. Gessa, Italy
F. Gil-Sotres, Spain
H. Hartikainen, Finland
P. Haygarth, UK
M. Hens, Belgium
G. Hofman, Belgium
R. Indiati, Italy
B. Kronvang, Denmark

P. Leinweber, Germany
C. Morel, France
A. Sharpley, USA
E. Sibbesen, Denmark
J. Torrent, Spain
H. Tunney, Ireland
N. Vagstad, Norway
P. Withers, UK