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USING A LANDSCAPE APPROACH TO INTERPRET DIFFUSE PHOSPHORUS POLLUTION AND ASSIST WITH WATER QUALITY MANAGEMENT IN THE BASINS OF LAKE CHAMPLAIN (VERMONT) AND LAC LÉMAN (FRANCE).

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ABSTRACT

Diffuse pollution should be recognized as a landscape-level phenomenon. As such, it requires an observational approach consistent with the complex structure and function of the landscape system. We developed a landscape-level approach to study the transfer of phosphorus in rural areas of the Lake Champlain and Lac Léman basins. We began by developing a concept of P dynamics that captured some of the diversity and complexity of P movement through the land (transfer system). Given this initial concept of the diffuse pollution in the landscape, we adopted a synoptic watershed sampling strategy to begin the quantitative description of diffuse P pollution. Data from these types of studies were then analyzed using multiple regression to infer connections between activities on the land and phosphorus flux to surface waters. Our inferences include: 1) land cover determines phosphorus flux during high flow but not during low flows periods, 2) during high flow events, natural wetlands are a significant sink for diffuse phosphorus in surface waters, 3) fluxes and concentrations are higher when the basins are intensively plowed, 4) in the context of plowed areas, agricultural practices as opposed to land cover is a more important determinant of phosphorus flux in watersheds, and 5) the position of elements in the landscape is an important factor controlling diffuse phosphorus pollution. The method and basis for arriving at these conclusions are discussed. We suggest that synoptic sampling of water quality over extensive areas in a landscape, coupled with multiple regression to analyze relationships among P fluxes and landscape variables, is an appropriate tool for determining driving factors, analyzing the diversity of processes, and finding generality in complex landscape systems.

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1. INTRODUCTION

Human activity in the landscape has a substantial power to alter the quality of surface waters and ecosystems dependent on that quality. In the last two decades, as control of point sources has progressed, diffuse sources of pollution have been recognized as a major cause of this water quality degradation (U.S. Environmental Protection Agency, 1995; Pucket, 1995; CIPEL, 1988; Novotny and Olem, 1994). In temperate regions in freshwaters, phosphorus is generally the water quality parameter of greatest threat relative to eutrophication because it limits aquatic plant productivity (Vollenveider, 1968; Barroin, 1980; Hecky and Kilham, 1988). However, progress in controlling diffuse P pollution of surface waters has been slow (Pucket, 1995). In the watersheds of many large lakes, diffuse phosphorus has been identified as an important pollutant to control because of the threat of continued P loading further driving eutrophication (Lake Champlain Management Conference, 1996; CIPEL, 1984).

Understanding of the mechanisms of diffuse pollution is a critical step in achieving reductions. Developing this understanding at a landscape scale is key (e.g., Whigham et al., 1988; Johnston et al., 1990; Kling et al., 2000; Jones et al., 2001), because the transfer of diffuse P is a landscape-level phenomena primarily involving 1) erosion and runoff from a complex diversity of landscape surfaces differentiated by a variety of factors including soil, topography, land cover, land use practices (Sharpley and Halvorson, 1994), and 2) transfer and attenuation through ditches, stream, rivers, and wetlands.

Initial approaches for describing the export of diffuse phosphorus pollution in rural regions have excluded much of the complexity and diversity inherent in landscapes. Data from experimental work at the field-scale are very useful for elucidating processes of P export from land surfaces (Koro and Bernard, 1995; McIsaac et al., 1995). The watershed unit has also been used in many scientific studies to study the hydrological factors and to evaluate the role of agriculture in exporting phosphorus to surface water (Dorioz and Ferhi, 1994; Jordan-Meille et al., 1998; McDowell et al., 2001). However, because these studies are representative of only a small portion of actual field conditions, results can not be accurately extrapolated to the diversity of situations in a typical landscape.

Because of this landscape diversity, the challenge in managing diffuse P pollution is to understand P loading and transport dynamics at the appropriate scales. Mechanistic models (e.g., CREAMS-WT, Heatwole et al., 1988; AGNPS, Young et al., 1989; Lenzi and Di Luzio, 1997) require extensive data, which are not easily available, and their results are often disappointing at the scale of large and complex watersheds (Leite, 1990). This contrasts the relative success of empirical models, including P loading coefficients and export functions, for general land cover (e.g., Vollenveider, 1968; Bovay, 1980; Beaulac and Reckhow, 1982; Meals and Budd, 1994). These empirical averaging approaches are useful at a broad scale to 1) characterize the difference in diffuse pollutant export among regions (Dillon and Kichner, 1975; Omernik et al., 1976; Wendt and Corey, 1980), or 2) capture a long term trend (Cassell et al., 1998), but they neglect the diversity and complexity of the landscape and oversimplify the interaction of hydrology, land cover, management practices (e.g., Romkens et al., 1973; Gaynor and Findlay, 1995), biogeochemistry of the site (Sharpley et al., 1993), and the spatial relationship among landscape elements (Whigham et al., 1988). In addition, within broad classes of land cover, the variability of agricultural practices, which can control the intensity of erosion and surface runoff (Boiffin et al., 1988; Sharpley et al., 1994;Vansteelant et al., 1997), may play an important role in determining P fluxes. Thus, neither mechanistic or simple empirical models can adequately handle the complexity of P processes in the landscape.

Given the difficulty in describing P dynamics at the scale of the landscape, some researchers have begun to develop more complex, multi-scale, GIS-based, hybrid models to predict P load (or more broadly -- water quality) based on land cover and other landscape variables such as proximity to streams or riparian areas, presence of buffers, connectivity, and source-sink concepts (e.g., Tim et al., 1992; Soranno et al., 1996; Johnson et al., 1997; Tufford et al., 1998). We feel that a landscape approach is essential, and that it should consolidate concepts to include study of the 1) diversity field conditions that mobilizes phosphorus, 2) variety of routes phosphorus can take in moving from the field into the hydrologic network, 3) possibilities of attenuation in riparian areas and wetlands, and 4) hierarchy of transformation and storage mechanisms in first, second, third, etc. order streams. Furthermore, management of diffuse pollution requires an understanding of the human factors causing the pollution and creating, disturbing, or modifying the routes of P through the land. These human factors transcend the scale of the field and the farm, extending to the community and the region, and hence also dictate a landscape approach.

In this paper we review and discuss a series of our initial studies employing an empirical landscape approach to understand diffuse phosphorus pollution (including Weller et al., 1996; Windhausen et al., 2003; Vansteelant et al., 1997; Trévisan et al., 1995; Dorioz and Trévisan, 2001). The general question that forms the background to this work was how do the activities on and organization of the landscape contribute to the eutrophication of lakes (Lac Léman and Lake Champlain). We use broad-scale observations of the spatial and temporal patterns in surface water chemistry of watersheds in conjunction with multivariate analyses to evaluate the extent to which diffuse phosphorus fluxes are a function of 1) the content of the landscape (the distribution of land cover), 2) attenuating landscape elements (wetlands, riparian zones), 3) the practices that take place within each land cover type, and 4) the spatial position of the landscape elements. Because we feel that our approach deviates from the norms of experimental science in water quality research, we discuss the foundations of our inferences about diffuse phosphorus pollution.

In presenting syntheses of our previous work, we hope to demonstrate that our general approach using broad-scale observations is useful for understanding diffuse phosphorus pollution because it complements "classical" watershed research by emphasizing the diversity and position of landscape elements within the watershed. This additional knowledge can help form a basis for making management decisions to reduce P flux to surface water. The initial studies we present do no adequately describe the spatial and temporal patterns of nutrient fluxes in our landscapes, but only suggest use of a landscape approach to begin this difficult task.

2. AN APPROACH TO STUDYING DIFFUSE POLLUTION IN THE LANDSCAPE

We began by developing a concept that first, recognizes diffuse phosphorus transfer as a landscape-level phenomenon and second, captures some of the diversity and complexity of phosphorus movement through a heterogeneous landscape.

2.1. The Phosphorus Transfer System

Because there is no important gaseous phase for phosphorus in our landscapes, phosphorus can be considered to be a conservative element, accounted for using a mass balance. Occurring in inorganic, organic, soluble and particulate forms, phosphorus emission from land surfaces includes a combination of forms that can undergo a variety of transformations during transport (Dorioz et al., 1998). Total phosphorus from diffuse sources is dominated by the flux of particulate phosphorus (PP), primarily a surface phenomena closely tied to runoff and erosion (Ryden et al., 1973; Sharpley et al., 1993). This movement of total phosphorus is primarily a discontinuous process, in motion during surface runoff events (Verhoff et al., 1982). The potential for this runoff begins with any process reducing the infiltration rate of the soil surface. Impermeable surfaces (e.g., roads, houses, compacted areas) and undisturbed vegetated surfaces (e.g., forests, grasslands) comprise the extremes of a permeability gradient. Cultivated areas vary spatially and temporally in permeability, depending on soil type, soil cover (e.g., vegetation, mulches) and the degree of soil "crusting." "Crusting" is a precipitation-induced degradation of the surface porosity and concurrent evolution of a surface crust of fine clays (Boiffin et al., 1988; Auzet et al., 1990). Most of the cultivated soils in the Lac Léman area are sensitive to this crusting, which enhances surface runoff and associated phosphorus emission. Runoff can occur under very low rainfall intensities (Vansteelant et al., 1997).

Starting as a series of small rills at the field scale, surface runoff generally must connect with the network of temporary depressions and ditches in order to carry its load of phosphorus to streams and then higher order rivers (Jordan-Meille et al., 1998). Subsurface flow carries much less phosphorus due to fixation in the subsoil (van Riemsdjick et al., 1987; Dorioz and Ferhi, 1994; Gilliam, 1994), except in some specific conditions including sandy soils, long-term over-fertilization, and agricultural drainage (Sims et al., 1998). Highly permeable buffer strips of permanent vegetation (e.g., grass, shrubs, hedgerows) and riparian buffers can attenuate the P flux to the hydrologic network (Gilliam, 1994; Uusi-Kamppa et al., 2000). While in the network, particulate P can settle and soluble P can sorb onto the network surfaces or be taken up by biota (Wang et al., 1999). Resuspension and partial desorption from previously stored P can occur generally during stormflows (Dorioz et al., 1998). In addition, major obstacles like wetlands and lakes can store and/or transform the phosphorus emitted from upstream fields (Johnston 1991, Detenbeck et al., 1993).

Thus a set of interacting processes move P through the landscape in a "transfer system," that includes emission from sources (mainly soils or sediments) and transport (including transformation and attenuation), This transfer system comprise a highly diverse and complex set of landscape structures arrayed in a complex and diverse hydrologic hierarchy - and interacting with some of the human activities, especially agricultural practices and management of the hydrologic network. The notion of hierarchy and complexity are characteristic of the landscape science literature (e.g., Allen and Starr, 1982; Urban et al., 1987; Turner et al., 1995). Given this complex transfer system, the challenge is to design an observational approach that can capture enough of both the fundamental processes of phosphorus dynamics and the diversity of responses of the various landscape components (e.g., fields, buffers, streams, wetlands).

2.2. Data Collection in a Heterogeneous Landscape

Conceptual models and associated observation strategies are molded by the environments in which they are formed. The study areas are representative of rural areas in the Lake Champlain and Lac Léman Basins, which both cover a wide geographic area with considerable landscape diversity (Table 1). Each basin contains mountains and valleys with a rural-urban mix of land uses and land cover. Despite a large number of inhabitants, the basins are largely rural because the residents are mostly concentrated in urban areas. There are many hundreds of active farms (mainly dairy) and forestry operations arrayed in a complex and diverse mosaic pattern including many thousands of individual parcels.

With typically complex bedrock geology due to periods of uplift and weathering, both regions are further complicated by a glacial surficial geology. Glacially compacted subsoils are fairly impermeable leading to an abundance of streams, which drain much of the precipitation (>60 cm/yr) from the landscape (Hamid et al. 1989). This extensive and diverse land cover creates a considerable challenge for measurement and sampling. We focused on predominantly rural areas with no point sources of pollution. The land cover included in the studies cover the range of dominant agricultural practices.

Table 1. Parameters of the study areas.

Parameter	LaPlatte Watershed	Chablais	Eight eastern watersheds
Basin context	Study 1	Study 2	Study 3
Location	Lake Champlain Basin - Vermont, New York, USA; Quebec, Canada.	Lac Léman Basin - Haute Savoie, France; Geneva, Vaud, Valais, Switzerland.	see study 1
Basin size	21,326 km^2	7,393 km^2	see study 1
Approx. population		1.5 million	see study 1
Maximum elevation		4,634 m	see study 1
Trophic state of lake	mesotrophic	mesotrophic	see study 1
Study area			
Dependent variables	TP, SRP, TSS, concentration and g ha^{-1} day^{-1}	TP, PP, TSS concentration	TP, kg ha^{-1} yr^{-1}
Independent variables: type	general land use, details on wetland types	general land use, cultural practices by field, aggregate indices	general land use, details on wetland types
Independ. variables: number	15	30	17
Spatial dimension: number of watershed units	15	14	8
Temporal dimension: number of samples or intervals	18	7	1
Time frame	annual	winter: January - March	annual
Sampling regime	events, base flow	seasonal high flows	periodic over the year
Range of watershed sizes	149 to 1396 ha	15 to 244 ha	3058 to 21,005 ha

2.3. A Phosphorus Observation Strategy Employing Watershed Networks

To observe phosphorus movement in the landscape, we chose small watersheds as the basic unit of study. Watersheds were minimally sized to provide a measurable signal (discharge and concentration) that can be related to variables describing the state of the basin. The complexity of the P transfer system led us to emphasize collection of data at broad-scales using networks of watersheds. While it would be ideal to know the chemical flux for many points in the landscape over a period of time (preferably replicate observations from homogeneous landscape elements, e.g., multiple corn fields with a particular slope, soil, agronomic practice combination), this detail and quality of data is prohibitively expensive to collect. An alternative, continuous monitoring of water quality from one watershed, while providing detailed and accurate estimates of phosphorus fluxes and mechanisms of transfer, would not capture the diversity of P dynamics at the landscape scale. Thus extrapolation of this site intensive information would be difficult to support. The compromises between intensive sampling of one (or a few) watersheds and a few periodic observations from a network of watersheds depend on the structure of the system (Figure 1). If the landscape diversity is not high, then extrapolation of intensive studies is justified. If the chemical quality of stream water is fairly constant, then infrequent but more spatially extensive sampling is not rendered useless by high temporal variation.

Given the enormous landscape variability of the lake basins in our studies, and what we knew about the pattern of temporal variations (Dorioz et al., 1991), we developed a phosphorus sampling strategy with the following characteristics:

(a) sample extensively:
- many watersheds (10-100 ha)
- watersheds with varying land cover and agricultural practices, include a diversity of cover types and their proportional representation

(b) sample infrequently, but differentiate:
- wet period events
- seasonal base flows

(c) use grab samples of surface water:
- collections are “near” synoptic, describing one point in time
- during declining limb of events to avoid the high variability in the beginning of the chemograph
- focus on chemical concentration, obtain flux if possible
- collect approximate flow information, water budget indices, etc.

We considered a variety of indicators of phosphorus movement including total phosphorus (TP), soluble reactive phosphorus (SRP), total suspended solids (TSS), and particulate phosphorus (PP), analyzed via Standard Methods (APHA 1985). The speciation of P is of great relevance to understanding P dynamics as well as P impact on receiving waters (Bostrom et al., 1988; Logan et al., 1979). For each measure of water quality, an appropriate unit needs to be selected (Table 2). The social question revolves around eutrophication of receiving surface water, thus implicating P flux (e.g., kg yr^{-1} or kg ha^{-1} yr^{-1}) as the unit of interest. In order to calculate these numbers, intensive monitoring of stream water using automated gauging stations is necessary. In Lake Champlain these data were sometimes available but generally we sampled water quality

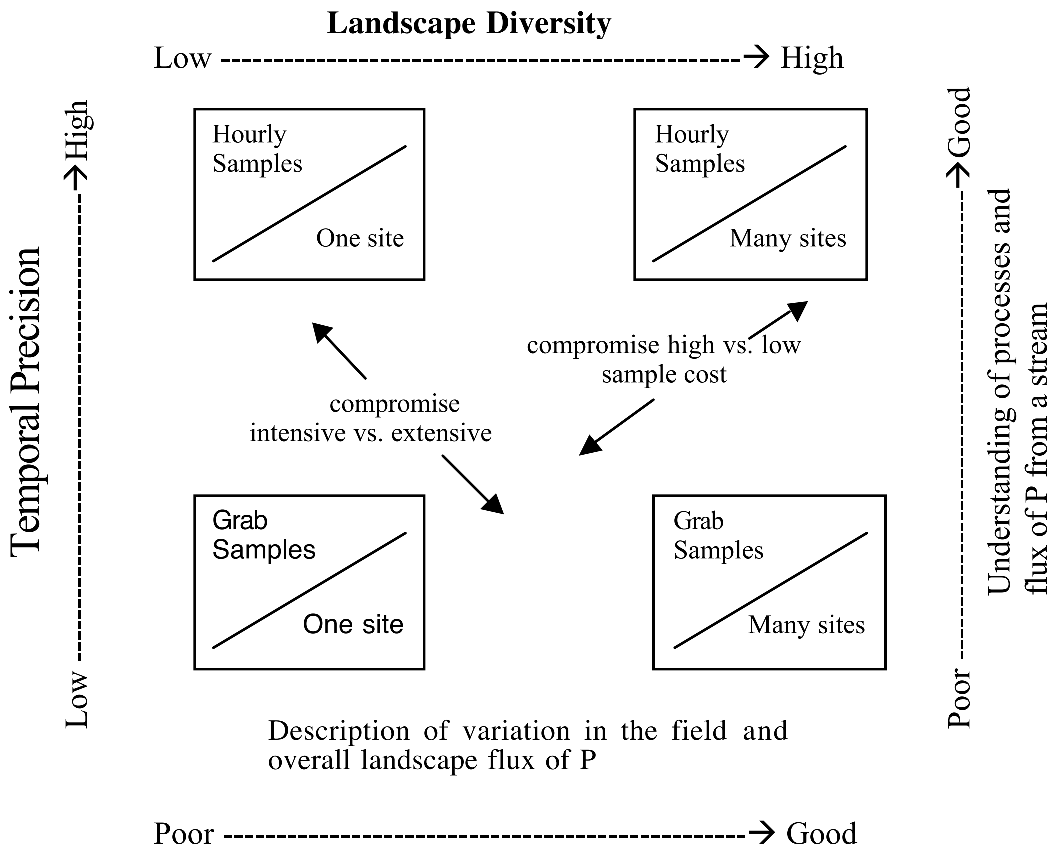

Figure 1. Trade-off of intensive vs. extensive observations, an example from study of the structure of the P transfer system.

(mg l^{-1}) along with concurrent estimation of Q (l sec^{-1}) to yield an instantaneous estimate of P flux (kg min^{-1}, extrapolated to kg day^{-1}). Without a good estimate of Q, phosphorus must be expressed as simple concentration.

The relevance of each unit of measure (kg min^{-1} or mg l^{-1}) is dependent on 1) the question being asked, and 2) the water yield from precipitation across the sampled watersheds. If the percentage of precipitation reaching the stream in each watershed is similar, then concentration is a good surrogate measure of instantaneous flux when expressed on a unit area basis (kg ha^{-1} min^{-1}). In addition, knowing the P flux at the time of sampling is really the primary question if you want to know when in the year the highest risk of P flux exists. Knowing only the concentration of P in a grab sample is not nearly so useful as knowing flux, yet it can still provide information about the relative risk of P export from the land.

Table 2. Measures of P Flux - Compromises between extensive and intensive observations

Observation	Unit	Comment
Annual flux	kg yr^{-1} or kg ha^{-1} yr^{-1}	sample intensive, expensive to collect, best measure for estimating P flux from the landscape
Instantaneous flux	kg min^{-1} or kg ha^{-1} min^{-1}	extensive sampling possible, good measure of flux at a point in time, annual calculation not possible
Volume-weighted concentration	mg l^{-1} for the year/period	extensive sampling possible, suggestive of a time-cumulative impact
Average concentration	mg l^{-1} for the year/period	extensive sampling possible, some reduction in sample variance, difficult to relate to absolute flux, allows relative comparisons among watersheds
Concentration	mg l^{-1}	maximum extensive sampling possible, allows relative comparisons among watersheds

2.4. A Landscape Observation Strategy

In our studies, observations were made 1) directly, and land condition noted on a map, or 2) remotely, employing satellite imagery, air photographs, and GIS to help classify the landscape. Our classification of the land surfaces began with simple cover types (crop, forest, residential developed, etc.). However, the cover type is not the only characteristic important for the potential emission of P, and other classifications were tried. Specific practices leading to that land cover may be a critical factor in diffuse flux. Other land characteristics such as a particular conservation tillage of a crop cover may be a more important determinant of surficial emission of P via runoff than whether the crop is corn or cereal, or whether the soil P concentration is high or low (Vansteelant et al., 1997). In addition, soil "crusting," a precipitation-induced degradation of the surface porosity and concurrent evolution of a surface crust of fine clays, leads directly to runoff and associated phosphorus emission. Thus crusting could also provide a good predictor of P flux. In a similar way, impermeable surfaces generate large volumes of runoff during rainfall events and thus cause resuspension of stream sediment and attendant P fluxes. These agricultural practices and watershed characteristics were described using a survey of farms and fields.

We used a variety of approaches to quantify the amount of landscape that is potentially emitting or attenuating phosphorus (Table 3). Each has relevance to different expressions of the phosphorus observed and often has a different conceptual basis. The simplest and most direct measure is area, but this is not necessarily the unit best suited to expressing the landscape to phosphorus relationship. For example, the emergent vegetation at the periphery of an open-water wetland may be the active unit of P processing, thus suggesting the use of perimeter as an appropriate measure. Furthermore, all positions in the landscape may not be equally as important. For example, riparian areas may be more functional quantities to consider for both P emission and attenuation. On the contrary, surfaces without nearby connection to the hydrologic network may not participate in the landscape dynamics of phosphorus.

Table 3. Some possible measures of landscape

Observation	Unit	Measure of
area	ha	a land cover, or land use/practice, of a soil with a certain P concentration or in a certain physical condition
area	%, of total watershed	land cover, physical condition, etc. normalized as a percentage of the total area of a watershed
perimeter	km	the landscape element, e.g. wetlands
number	count	landscape elements, most useful if the landscape elements are of the same size
active area	ha, within bounds	land cover, etc. that contributes to an elemental flux, e.g., within 10 m of a flowing stream.
weighted area	ha, adjusted for effectivenes s	land cover adjusted for its potential contributions, e.g., land within 10 m of a stream may contribute 2x the P of land 20 m from the stream

Each landscape metric was used only with compatible units of phosphorus. For example, area as hectares is the appropriate unit where the unit of P flux is kg of P per watershed. The percentage of the total watershed area of a contributing landscape element is compatible with P expressed as concentration or specific flux (kg ha^{-1} min^{-1}).

Given these considerations discussed above, our landscape observation strategy had the following characteristics:

(a) cover a diversity of landscape elements
- diversity of surfaces
- diversity of geo/topo/bio variables
- include attenuating and “sink” features

(b) consider surface conditions and diversity of agricultural practices
- reduction of permeability due to compaction and “crusting”
- cultural practices such a method of plowing, timing, etc.

(c) include relative spatial information
- connection or proximity to the hydrologic network
- stream order in the hydrologic network

This variety of ways of expressing and quantifying the different landscape surfaces resulted in many non-mutually exclusive sets of landscape descriptors that could be used to relate to phosphorus fluxes.

2.5. Analyzing Observations Of Phosphorus In The Landscape

In its simplest form, the analysis of the relation between P flux or concentration and the landscape requires a matrix with one dependent variable representing P from each watershed and several corresponding independent variables describing landscape attributes of each watershed. Our observations of phosphorus and the landscape generally resulted in a 3-dimensional data matrix of various chemical constituents expressing phosphorus or TSS, many times of sampling, and many different landscape attributes for each watershed. We interpreted the patterns within these spatial and temporal observations of chemical information to infer function at the scale of the landscape.

To evaluate our matrices, we generally relied on multiple regression to relate patterns of phosphorus (Table 2) to the structure of the landscape as measured by our various spatial representations (Table 3). Empirical models are of the form P = bo +b1(X1)+ b2(X2) + b3(X3); where Xi's are variables in each land cover category employed. We limited models to three land cover variables. Given the different possible expressions of phosphorus (e.g., TP, SRP, PP, kg ha^{-1}, mg l^{-1}, annual or seasonal average, volume-weighted mean, annual maximum, individual sample date) and the many possible ways of representing the landscape variables, choosing the analyses to conduct was not straight forward. The number of possible regressions, to relate P to the landscape, runs into the thousands. To reduce the combinations of regressions, clustering was used to group sample dates. Clustering was accomplished using a Ward's clustering algorithm (JMP 3.1, SAS Institute) on a date-by-watershed matrix of phosphorus values.

To further complicate choosing the specific regressions to run, statistical considerations create further options, which increases the number of possible regressions. For example, lack of homoscedasticity requires a data transformation in order to meet requirements for hypothesis testing. Some variables needed this transformation, others did not. We chose to use a process-based, intuitive selection of independent landscape variables, rather than attempt all possible combinations of dependent and independent variables. This choice has some ramifications for estimation of the significance of any relationship that we might discover and is discussed in section 4. Step-wise regression was also used to help with the selection of variables.

The results of our multiple regressions provide no direct information about causes and effects, only indications of statistical relationship, which could indicate spurious, indirect, or causal relationships. Despite the ambiguity of the method, we found consistent and statistically significant relationships (high and significant r^2), which we interpreted as evidence of a functional relationship between phosphorus flux and certain attributes of the landscape.

2.6 Brief Description Of Methods Employed

Methods are described in greater detail in the original sources (Weller et al., 1996; Trévisan et al., 1995; Vansteelant et al., 1997; Dorioz and Trévisan, 2001; Windhausen et al., 2003). A brief overview of study design, analytical, and statistical methods are provided here.

In Study 1, SRP, TP and TSS were observed in 15 subwatersheds in the LaPlatte River watershed from October 1994 to October 1995. Subwatersheds ranged in size from 149 to1396 ha (Table 1). Fluxes were estimated via synoptic grab samples and estimates of flow using a velocity-integrating flow meter according to the conventional mid-section method (Rantz, 1982). Discharge rating curves were developed for each sampling site according to Kennedy (1984). Subwatershed boundaries were delineated on U.S.G.S. 7.5 minute topographic maps and verified in the field, particularly in areas of low relief. GIS coverages were created by digitizing land cover and subwatershed boundaries using ARC/INFO based on 1988 1:5000 panchromatic orthophotographs, updated with field surveys in 1990. The overall accuracy of the linework was reported as +/- 25 m (Appleton, 1993). The land use coverage followed a classification system that was adapted for use in Vermont (Vermont Center for Geographic Information System, 1992) from the Anderson land use and land cover system (Anderson et al., 1976). Row cropland in the study area was mapped onto 1:5000 orthophotographs during summer field surveys in 1995 and then transferred to the GIS layers. Wetland classification (Cowardin et al., 1979) was based on interpretation of National High Altitude Photograph (NHAP) color infrared imagery, enlarged to a scale of 1:20,000 from an original 1:58,000.10% of the wetlands were field checked. Wetland boundaries were rectified using 1988 1:5000 orthophotographs. Hydric soils were mapped using criteria in the Federal Manual for Identifying and Delineating Wetlands (Federal Interagency Committee for Wetland Delineation, 1989). The area of these drainage classes in each subwatershed was determined from an existing soils coverage of the LaPlatte River basin (U.S.D.A. Natural Resource Conservation Service, 1995). TP was analyzed colorimetrically on a spectrophotometer using the molybdenum blue method following digestion in a sulfuric acid-persulfate solution (U.S. Environmental Protection Agency, 1983). SRP was measured on 0.45 μm membrane filtered samples colorimetrically by the molybdenum blue method within 48 hours of sample collection. Simple and multiple regression was used to assess relationships among dependent and independent variables. All combinations of explanatory variables were evaluated for multicollinearity using the variance inflation factor (VIF, Neter et al., 1989) and tested for normality (Shapiro-Wilks, $p < 0.05$).

Study 2, conducted in the Chablais, was part of a larger set of studies to understand sources of diffuse pollution in the Lac Léman Basin (CIPEL, 1984; Dorioz and Ferhi, 1994; Pilleboue and Dorioz, 1986). In the Chablais study, stream water was collected from 14 watersheds on seven dates in the generally high flow period from 19 January to 6 March 1995 (Table 1). This time period had the highest risk of runoff due to the condition of the fields after harvest in winter (little vegetative cover, low EVT, high water table, see Dorioz et al., 1998). For independent variables, data on 17 specific land cover and/or agricultural practices were collected using low altitude photography coupled with field surverys to identify planted crops, agricultural practice in use, area of roads, residential area, and natural cover in forest. Percentage of “crusted” soil (precipitation-induced degradation of the surface porosity and concurrent evolution of a surface crust of fine clays, Boiffin et al., 1988) in the watershed was determined by visual evaluation. The concentration of P in soils was analyzed by standard methods. All area units for the independent variables were converted to percent of total watershed area. In addition to the individual sample dates, a simple numerical average for each of the water quality parameters was calculated. Multiple regression was used to evaluate the relationship between dependent and independent variables.

In study 3 (Weller et al., 1996), annual TP flux was estimated for eight watersheds ranging in size from 3,058 to 21,005 ha. (Smeltzer and Quinn 1996, VT DEC & NY DEC 1997, unpublished report). Landscape and wetland variables were obtained from a variety of sources, but were predominantly interpreted from 1988 Landsat Thematic Mapper (TM) images. Primary land cover information was modified using a geographic information system (GIS). Again simple and multivariate analyses were performed.

3. RESULTS - THE RELATIONSHIP BETWEEN DIFFUSE PHOSPHORUS POLLUTION AND LANDSCAPE STRUCTURE

Using the approach described, we started with the simplest concept about P and the landscape (P flux is a function of land cover in the landscape) and then progressively modified that concept to include greater complexity (also a function of landscape attenuators of P transfer, and of landscape position).

3.1. Phosphorus Flux Is Related To The Area Of Different Land Cover

The notion that phosphorus flux is determined by the surface area of certain land cover within a watershed is supported by many watershed studies (Johnson et al. 1976, Beaulac and Reckhow, 1982; Meals and Budd, 1994). Does our broad scale approach of using synoptic grab samples across a diversity of watersheds support these other studies?

We first used the LaPlatte River watershed in Vermont to evaluate this question (Windhausen et al., 2003, this volume). For each water quality measure for each sample date, we ran regressions with simple land cover variables (total agriculture, row crops, forest, residential, wetlands, etc.). None of the more than 60 regression run for total suspended solids (TSS) concentration indicated that a relationship existed and these values are not reported. Soluble reactive phosphorus (SRP) concentration followed the same pattern as total phosphorus (TP) and on certain dates the two correlated very closely (up to r^2 = 0.99, and very often over 0.6, see Table 4). After obtaining this result, TP was analyzed more comprehensively than SRP.

To reduce the amount of data and sample variation, we calculated volume-weighted mean concentration for clusters of dates corresponding to different TP flux regimes (Table 5). The clusters consisted of four groups of dates that resembled each other in having similar TP fluxes values for a similar set of watersheds. In addition, a volume-weighted mean for the entire year of samples was calculated. These different ways of deriving a dependent variable related to TP resulted in a complex matrix of r^2's. Some of these values are included in Table 5.

Using the samples means, a very simple land cover model taking into account just the area of three very general categories of land cover (set 1 = % of agric land, developed land, and wetland) can “explain” 60 % of the variability in the dependent variable (flux or concentration). This increases to about 80% (set 2 or set 3) using a more precise descriptors of agricultural land. As a result of the similarity between SRP and TP samples, the r^2's observed for the relationship of SRP and TP export to land use also followed each other fairly closely (r^2= 0.74) suggesting that overall, the two forms of phosphorus flux are dominated by a similar process in the landscape: runoff.

In addition, although the r^2's are generally lower for TP concentration, concentration seems to tell the same story as TP flux, thus suggesting that concentration captures much of the relationship between landscape surface condition and the flux of diffuse phosphorus to surface waters. Using the empirical model with set 3 (X1 = croplands ; X2 = grasslands ; X3 = wetlands), produces an r^2 of 0.58 for concentration of P. This is an important result as it mirrors finding in study 2 (discussed below), where the landscape in

Table 4. R^2 values for (a) the relationship of SRP and TP concentration for 15 watersheds on 15 dates, (b) the relationship of SRP flux to land use variable set 1 (cropland, wetland, cows), and (c) the relationship of TP flux to land use variable set 1.

Date	(a) R^2 of [SRP] to [TP	(b) R^2 for SRP flux	(c) R^2 for TP flux
11/01/94	0.99***	0.47	0.56
11/29/94	0.82***	0.55	0.44
01/22/95	0.87***	0.31	0.11
02/25/95	0.45*** (0.93 excl. #8)[1]	0.63	0.34
03/08/95	0.52***	0.79	0.87
03/14/95	0.09 (0.61excl. #6)	0.65	0.73
04/13/95	0.72***	0.51	0.44
04/28/95	0.60***	0.31	0.34
05/18/95	0.40** (0.75 excl. #11)	0.19	0.05
06/12/95	0.83***	0.03	0.23
08/04/95	0.27* (0.97 excl.#5)	0.12	0.14
08/06/95	0.96***	0.43	0.21
08/12/95	0.80***	0.22	0.28
10/06/95	0.94***	0.42	0.40
10/15/95	0.76***	0.25	0.26

[1] watershed #8 produced a large amount of sediment on this sample date resulting in an outlier

in the [SRP] to [TP] relationship; excluding this value the r^2 is much higher, 0.93.

the Lake Léman basin is very similar to that of Lake Champlain. These results validate our broad scale approach: using synoptic grab samples across a diversity of watersheds is able to differentiate land cover types and their role in P export.

However, there is additional complexity inherent in this type of approach. For individual sample dates, many of the regressions were not statistically significant for any combination of independent land cover variables tried. The r^2 for most dates was generally low, except for on several occasions when streamflow was high due to heavy rainfall on wet or frozen soils in the spring. In these cases, agricultural land, cropland, the presence of cows in the stream, and wetland, generally appeared as good indicators of P flux. In addition, the r^2's for means for the spring high flow period and the whole year were significant and ranged as high as 0.95. Thus temporal variability in the relationship between land cover and P flux is high, probably due to large changes in the mechanics and ecology of P emission and transport under different hydrologic and seasonal conditions.

Table 5. R^2 values for regressions of TP concentration (mg/l) and TP flux (g ha^{-1} day^{-1}) vs. five land use variable sets for individual sample dates and means for clusters of dates. Volume weighted means are shown for concentration data. Models are of the form: Y=ß0+ß1(X1)+ß2(X2)+ß3(X3)+… where the X variables in each land use variable set are described below. Clustering was based on temporal patterns of TP flux across the study watersheds.

Sample	Land Use Var. Set 1		Land Use Var. Set 2		Land Use Var. Set 3		Land Use Var. Set 4		Land Use Var. Set 5	
	[TP]	TP flux	[TP]	TP flux	[TP]	TP flux	[TP]	TP flux	[TP]	TP flux
low flow* mean	0.55[1]	0.45[1]	0.54	0.35	0.52[1]	0.38	0.51	0.43	0.51	0.31
02/25/95	0.28	0.19	0.26	0.37	0.32	0.32	0.28	0.20	0.25	0.34
05/18/95	0.12	0.03	0.07	0.02	0.12	0.04	0.16	0.09	0.14	0.05
06/12/95	0.12	0.30	0.22	0.30	0.22	0.21	0.21	0.28	0.31	0.23
wint/summer mean	0.36	0.15	.036	0.29	0.42	0.29	0.37	0.17	0.39	0.30
11/01/94	0.26	0.45[1]	0.12	0.25	0.26	0.47[1]	0.27	0.55[2]	0.21	0.56[2]
11/29/94	0.50[1]	0.58[2]	0.16	0.26	0.49[1]	0.55[1]	0.52[2]	0.61[2]	0.29	0.44
01/22/95	0.52[1]	0.29	0.41	0.09	0.52[1]	0.27	0.53[1]	0.26	0.46	0.11
04/13/95	0.40	0.39	0.48[1]	0.36	0.50[1]	0.40	0.41	0.45	0.51[1]	0.44
04/28/95	0.29	0.29	0.29	0.26	0.15	0.21	0.30	0.43	0.29	0.34
10/06/95	0.13	0.27	0.06	0.28	0.14	0.18	0.25	0.40	0.25	0.40
10/15/95	0.45[1]	0.47[1]	0.51[1]	0.43	0.51[1]	0.28	0.52[1]	0.30	0.63[2]	0.26
spring/fall mean	0.44	0.42	0.30	0.29	0.44	0.42	0.57[2]	0.57[2]	0.50[1]	0.52[1]
03/08/95	0.46	0.68[3]	0.54[2]	0.80[3]	0.59[2]	0.84[3]	0.60[2]	0.76[3]	0.64[2]	0.87[3]
03/14/95	0.54[2]	0.47[1]	0.52[2]	0.67[2]	0.49[1]	0.65[2]	0.50[1]	0.53[2]	0.55[2]	0.73[3]
spring high flow mean	0.54[2]	0.66[2]	0.67[3]	0.84[3]	0.68[3]	0.86[3]	0.67[3]	0.80[3]	0.76[3]	0.95[3]
all sample mean	0.59[2]	0.68[3]	0.74[3]	0.77[3]	0.74[3]	0.83[3]	0.71[3]	0.85[3]	0.86[3]	0.94[3]

* r^2's for individual low flow samples not shown.

[1] significant at $p<0.10$ [2] significant at $p<0.05$ [3] significant at $p<0.01$

Land Use Variable Set 1: X1=% agricultural land, X2=% developed land, X3=% wetland

Land Use Variable Set 2: X1=% row cropland, X2=% developed land, X3=% wetland

Land Use Variable Set 3: X1=% row cropland, X2=% pasture and hay, X3=% wetland

Land Use Variable Set 4: X1=% agricultural land, X2=presence/absence of cows-in-stream, X3=% wetland

Land Use Variable Set 5: X1=% row cropland, X2=presence/absence of cows-in-stream, X3=% wetland

Another limitation of the approach is that the pattern of r^2 values in Table 5 can not be evaluated in a strict statistical manner with corresponding inferences about the role of agricultural land, cropland, wetlands, etc. in determining phosphorus flux. First, these models employ only three out of the many independent variables possible. Other sets of possible regression models could have been tested. Other combinations of two, three or more independent variables were calculated, and are not reported here. Some of these models also result in high r^2 values. Which is more "correct?" Second, the relationship that each regression might suggest can be very different, or even contradictory. For example, the high r^2 values observed in November of 1994 suggests that TP export may be due to the extent of agricultural and developed land (land cover variable set 1), but not the amount of row cropland and developed land (predominantly residential, land cover variable set 2) in the studied watersheds.

While the individual regression results in Tables 4 and 5 are perhaps dangerous to interpret quantitatively, the overall approach seems to have utility when the results are considered as a whole. Thus it is the pattern and consistency of r^2 values that provide the evidence of meaningful relationships. The highest r^2 values in Table 5 are found during the spring runoff period using either TP concentration or TP export as the dependent variable. This suggests that under these high flow conditions, runoff from cropland (or agricultural land in general) determines the amount of TP in the streams. The landscape elements apparently contributing P to the stream water seem to consistently include some aspect of agricultural activity, in particular more intensive activities such as row cultivation and unrestricted access of the stream corridor to cows. A closer examination of the regression coefficients in Table 6 reveals a surprising degree of consistency when the coefficients are highly significant ($p < 0.01$). Regardless of the date of sampling or the use of means, the use of concentration or flux, or the variable set combination selected, the ratio of the coefficient for row cropland to wetland only varies from 1.2 to 3.6. This is perhaps 2 to 5 times higher than the ratio for all agricultural land. Together these two observations suggest that row cropland is the dominant agricultural land use contributing to P flux to surface waters, and wetlands are modest sinks for P. If we attempt to squeeze the maximum amount of information out of the data, because the unit of the dependent phosphorus variable is taken as g ha^{-1} day^{-1}, we can estimate that each hectare of row cropland yields 140 to 400 grams of P each day under the observed conditions of high spring flow (Q on March 8 and 14 were somewhat similar at the time of sampling, thus allowing this Q-independent estimate of daily P flux).

During most of the year, the observations in Table 5 suggest that land cover is not an important determinant of TP in stream water. This does not support our initial conception and suggests that at these times of year either 1) other aspects of the landscape control TP flux, or 2) perhaps in-stream phosphorus dynamics dominate TP flux.

3.2. Phosphorus Flux Is Attenuated By Wetlands

Our results (Tables 5 and 6) suggest that wetlands, as a general land cover type, can be an important sink for P emission from land surfaces. In addition, as a group, their unit area efficiency in P removal is not as great as the emission capacity of agricultural land, as suggested by comparison of regression coefficients. However, this may reflect the variability

Table 6. Coefficients in regression models of P concentration (mg l^{-1}) and export (g ha^{-1} day^{-1}) vs. land use variable sets. Concentration mean values are volume-weighted. Models are of the form: Y=b0+b1(X1)+b2(X2)+b3(X3)+e, where the X variables in each land use are described below.

		Land Use Variable Set 1				Land Use Variable Set 2				Land Use Variable Set 3		
Date	Variable	b1	b2	b3	-b1/b3	b1	b2	b3	-b1/b3	b1	b2	b3
03/08/95	TP concentration	-0.05	-0.58	-8.4**	ns	10.7	-2.1	-9.3**	ns	15.6**	-2.0	-9.5**
03/14/95	TP concentration	7.7**	7.9**	2.1	ns	12.4**	1.7	-4.7*	2.6	11.1*	-1.1	-6.9**
Spring high	mean concentr.	0.00046	-0.005	-0.0075	ns	0.010*	-0.002	-0.008***	1.2	0.014**	-0.001	-0.0084
Annual	mean concentr.	0.0015	-0.003	-0.0048**	ns	0.010***	-0.001	-0.005***	1.9	0.011***	0.0004	-0.0054
03/08/95	TP flux	0.71***	-0.13	-1.1**	0.65	4.0***	0.46	-1.4***	2.8	3.06***	0.35*	-1.4**
03/14/95	TP flux	0.63*	0.57*	0.16	ns	1.4***	0.10	-0.40**	3.6	1.4***	-0.9	-0.50*
Spring high	mean flux	0.47***	-0.04	-0.86***	0.55	2.7***	0.34	-1.00***	2.7	2.1***	0.19	-0.98*
Annual	mean flux	0.063***	-0.07	-0.12***	0.51	0.33***	0.02	-0.14***	2.3	0.26***	0.03	-0.14*

Land Use Variable Set 1: X1=% agricultural land, X2=% developed land, X3=% wetland
Land Use Variable Set 2: X1=% row cropland, X2=% developed land, X3=% wetland
Land Use Variable Set 3: X1=% row cropland, X2=% pasture and hay, X3=% wetland
Significant at * 0.10, ** 0.05, and *** 0.01 levels.

in P uptake capacity of specific wetland types that make up the total. Many individual studies of wetlands have demonstrated that wetlands can act as important filters of water pollution, including phosphorus pollution (Johengen and Larock, 1993; Tilton and Kadlec, 1979; Dorioz and Ferhi, 1994). The diversity of wetlands in our landscapes is very great, and these individual studies may not be justifiably extrapolated to wetlands in general. Some wetlands may be ineffective in attenuating P and some might actually release P under certain conditions (Peverly, 1982). A survey we conducted of inlet and outlet SRP from 10% sample of randomly sampled wetlands in the LaPlatte River Basin indicated that net flux was positive for about 30% of sampled wetlands, and negative for 70 % of wetlands on one day during spring snowmelt (Goldsmith 1994, University of
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Vermont, unpublished master's thesis). A similar observation was made for a selection of 20 wetlands of various morphological types in Lac Leman. No clear significant differences in phosphorus concentration between the inlets and outlets were observed (Parmeland, 1995).

To attempt to better understand wetland type and P dynamics, we developed a set of nine wetland variables, each capturing a slightly different aspect of the concept of wetland (Windhausen et al., 2003). Each of these variables was then successively included with the total agricultural land variable in regression models for each date sampled. The resulting r^2's continue to show that significant relationships between any expression of wetland and P flux only occur at high spring flows. On 8 March 1995, with the highest flows of the year, some differentiation among wetland types was observed (Table 7). The strongest relationship was observed for hydric soils, followed by an expression including all wetlands, lakes and ponds. Other r^2's were significant, but suggested a weaker relationship. Accepting, for the moment, a causal link to this relationship, we might infer that because the strongest relationships were observed for the most general wet surface classifications (hydric soils being the most general and including the greatest areal extent of land surface), perhaps wet places and topographic depressions, in general, serve to reduce P export. In this context, our efforts to detect which type of wetland may be the most efficient were not successful, perhaps due to the lack of an appropriate functional classification.

As observed in study 1, phosphorus flux, under some conditions, seems to be determined by land cover. This observation supports the common notion of export coefficients (Omernik, 1976; Beaulac and Reckhow, 1982). However, in almost every significant regression model relating TP flux to land cover, there are peripheral watersheds or outliers suggesting that land cover alone does not determine TP flux.

3.3 Phosphorus Flux Is Related To Agricultural Practices

Usually these points are taken as random error (sampling error, parameter variance, etc.) that gets in the way of the multiple regression analyses. However, also embedded in this residual term are the processes that determine TP flux that are not land cover related. For example, in the LaPlatte River watershed, one subwatershed consistently deviated from all regression models with unusually high P fluxes. Closer field observation of this watershed revealed one farm with consistently poor management practices, including poor management of stream banks and animal concentration areas in the surface drainage network. Foster et al. (1990) estimate 700 kg ha^{-1} yr^{-1} of sediment flux due to cattle disturbing stream banks in grasslands. In the LaPlatte River watershed, we also developed another measure of agricultural practice, a "cows in stream" variable. A high r^2

Table 7. R^2's for regression models using agriculture and different wetland expressions as independent variables to predict phosphorus flux (kg ha^{-1} yr^{-1}) from 8 watersheds in the Lake Champlain Basin (1991 P flux, 1973 land use).

Explanatory variables	R^2
Agricultural land use used singly (ha)	0.39**
Wetland expressions used in conjunction with agricultural land	
wetlands, lakes, and ponds	0.70***
forested and scrub-shrub wetlands	0.57*
emergent and open water wetlands	0.40
mixed emergent and scrub-shrub wetlands	0.57*
non-agricultural wetlands, lakes and ponds	0.69***
lakes, temporarily flooded wetlands and seasonally flooded wetlands	0.62**
lakes, ponds, and wetlands isolated or along first order streams	0.61**
wetlands along second or third order streams	0.40
poorly and very poorly drained soils excluding agriculture	0.77***

* significant at $p<0.10$ ** significant at $p<0.05$ *** significant at $p<0.01$

value for land use variable set 4 suggests that the presence of cows in the streams is an important determinant of P export (TP, see Table 6, set 4 or 5). Clearly, agricultural practice can be an important determinant of P fluxes. Unfortunately in our Vermont work, this was the only data on agricultural practices we had. This was not the case for our other studies in the Chablais in France.

In the Chablais study, using the same variables in the Vermont study (cropland, pasture, wetland), 58% of the variation in TP concentration is explained (r^2 in Table 8). However, additional residual variance is explained by using detail available through low altitude air photography and/or field surveys to classify cultivated cropland and pasture. For TSS, good relationships were found with variables expressing high altitude conditions and steep slopes (alpage or mountain pastures). For TP concentration additional variables also improve the fit (Table 8).

Because there were 17 available descriptions of land cover and cultivation practices, the number of possible combinations of dependent and independent variables was enormous. To simplify the analyses, step-wise multiple regression was used to select possible variable combinations, with a limitation of three independent variables. In addition, some models were chosen to evaluate initial notions of what might be determining P flux. The results of many regressions, some of which are shown in Table 8, suggests that a great many land cover and agricultural practice variables have a relationship to phosphorus concentrations in the drainage waters. While no clear pattern emerges from the regressions from the various dates, for the average TP concentration in the waters, the percentage of land in cultivated crops, regardless of crop, seems to be important. This crop-related P emission variable was often in combination with one other variable, grain^3, which is the area of exposed soil after wheat or cereal crop harvest with no cultivation or herbicide application. This set of practices results in good post-harvest field coverage due to regrowth of cereal crops and the new growth of weeds. Grain^3 has a negative coefficient suggesting that it reduces P emission. Together with the area in cultivated crops, the model explains almost 90% of the variation in TP concentration.

Thus, the amount of phosphorus in runoff is generally related to cultivated areas, but is more directly controlled by the condition of the soil and its susceptibility to superficial runoff, which is mediated by agricultural practices. Certain practices on cultivated fields (e.g., preparing a seed bed by cultivating the soil until it achieves a small, level, particle-size distribution) facilitate "crusting" of the soil (Vansteelant et al., 1997). This crusting was observed for the studied watersheds and, along with one other independent variable (grain^3) also explains almost 90% of the variation in P concentration. Alone, this variables explains over half of the variation in average P concentration (r^2=0.57), which is similar to the result for cultivated cropland (r^2=0.49). Different cultivation practices on cropland can lead to different soil conditions, and thus potentially this measure of crusted soil could lead to a differentiation of cropland by cultivation practices. However, in our study area, the amount of cultivated cropland and the amount of soil showing crusting correlated quite well (r = 0.98), thus preventing inferences about which explained patterns of P in the landscape better.

Additional information on the potential role of agricultural practices in controlling P fluxes requires a different approach than used above. All of the 17 independent variables can not be included in a regression model because in our case we had only 13 degrees of freedom. Thus some grouping of variables or other method of including the information on the independent variables is needed. One way we used to combine the land practice data was to calculate a simple additive index, Rsum, of the estimated risk of runoff from each type of field (14 agricultural field types). Individual components of risk were based on field observation of runoff due to the different combinations of cultural practices and included information on frequency of surface runoff events on each field type during a one year survey (Vansteelant et al., 1997). This variable explained just under half of the variation in P samples (Table 8). Unfortunately, in this attempt to consolidate many independant variables , Rsum correlated quite closely with both the amount of cultivated cropland (r = 0.86) and the amount of crusted soil (r = 0.92). This correlation makes it difficult to infer which one of these aspects of land practice is more related to P emission.

Table 8. Regression results using a step-wise and an intuitive approach for TP and TSS concentrations for watersheds in the Chablais during a high flow period in winter 1995. (-) before a variable indicates a negative relationship.

TP					TSS			
Observation	r^2	independent variables			r^2	independent variables		
by simple	land	cover						
Average	0.12	agriculture	forest		0.16	agriculture	forest	
Average	0.51**	cropland***	pasture		0.14	cropland	pasture	
Average	0.58**	cropland***	pasture	(-)wetland	0.14	cropland	pasture	(-)wetland
by	stepwise							
19 Jan 95	0.73***	roads***	(-)grain^3***	pasture**	0.81***	alpage**	(-)grain^3***	roads***
23 Jan 95	0.51*	pasture***	grain^1	soil P conc*	0.45**	forest*	soil P conc.	
30 Jan 95	0.92***	(-)grain^3***	fallow***	connected cropland*	0.69***	cereal**	(-)grain^3***	soil P conc.
6 Feb 95	0.75***	grain corn***	plowed	roads***	0.22	grain corn	silage corn	grain^1
16 Feb 95	0.83***	connected cropland***	(-)grain^3***	cereal**	0.42	pasture	grain^1	(-)grain^3
24 Feb 95	0.74***	(-)grain^2***	silage corn**	wetland**	0.53*	(-)grain^3**	grain^2	soil P conc.
6 Mar 95	0.83***	grain corn***	wetland***	road***	0.50*	forest	alpage*	(-)grain^3
Average	0.90***	connected cropland***	(-)grain^3***	forest	0.89***	alpage***	cereal***	(-)grain^3***
by	intuition							
Average	0.89***	cropland***	(-)grain^3***		0.54**	cropland	alpage***	
Average	0.87***	crusted soil***	(-)grain^3***		0.58***	crusted soil*	alpage***	
Average	0.45***	Rsum***			0.0	Rsum		

* $p<0.10$ ** $p<0.05$ *** $p<0.01$

alpage – high elevation grasslands on mountain slopes and meadows; grain^1 - grain crop (wheat, cereals) with surface plowing after harvest; grain^2 - grain crop with no plowing and herbicide spraying after harvest; grain^3 - grain crop - soil covered with vegetation during the study period; fallow – would be covered by grass and weeds during study period; connected cropland - cropland area connected to the hydrologic network, often by drainage ditches; Rsum is derived from indices of risk of P in runoff, see text for details; alpage is high altitude semi-natural pasture used in summer (not actively used during the study period).

Our landscape observations appear to provide evidence that agricultural practice, as opposed to general land cover, may provide more information about sources of P emission in the landscape. We infer that the P emission gradient: forest - grassland - cropland is not a discrete series of surface types, but may be better represented as a series of overlapping distributions where land practice determines where in the distribution a particular parcel of land may be. For example a cultivated field with good year-round vegetative cover and a mulched surface can export lower amounts of P than an overgrazed pasture. Our approach, using widely dispersed surface water samples, allows a general evaluation of this comparison of practice vs. land cover.

3.4. Is Landscape Position An Important Modifier Of Function?

Information describing the relationship of elements within the landscape can provide important structural information describing processes affecting the transfer of phosphorus at the scale of the landscape. Perhaps most important among the possible structural descriptions is the location of the active hydrologic network and the relationship of individual sources and sinks of phosphorus to this network.

Observations in study 2 (Chablais) included information on the relationship of cultivated fields to the network of ditches and streams draining the landscape. These observations were made based on the idea that phosphorus moving over the surface of a field during a rain event only becomes a eutrophication threat when it enters the surface water network (Jordan et al., 1998). For each watershed, the percentage of fields adjacent to actively flowing ditches or streams (connection) was evaluated at the time of water sampling. In most cases, this surface water network consisted of man-made drainage ditches of various styles and constructions. If more than 50% of the fields were connected, we considered a critical threshold to be reached, and assumed connection was not a limiting factor (connection = 100%). This evaluation of connection to the hydrologic network was multiplied by the area of cultivated cropland to estimate the area of connected, cropland surface for each watershed. The correlation of TP concentration as a function of cropland alone yields an r^2 of 0.49, while use of the connected, cultivated cropland improves the r^2 to 0.73. This clear improvement suggests that the manner and degree to which farmers connect landscape elements to the hydrologic network plays a role in determining P flux to surface waters.

A more geographically extensive study (study 3) using eight watersheds ranging in size from 3,058 to 21,005 ha permitted another test of the idea that position in the landscape has important consequences for phosphorus export to surface waters. Taking agricultural and forest land cover as a phosphorus source area, and one expression of wetland presence as a sink variable, regressions on the dependent variable of annual phosphorus flux revealed a pattern suggesting that riparian wetlands near 1st- through 4th-order streams related best to the observed flux of P from the watersheds (r^2=0.88, significant at $p<0.05$, Table 9).

Table 9. Adjusted R values and associated probability of greater F from linear regression. These regressions related watershed characteristics to measured phosphorus loads from each watershed. Agricultural and forested area comprised the initial two independent variables. Wetland variables were added individually to the regression.

MODEL	Adjusted R^2	P>F
Agricultural and forested area alone	0.63	
Agricultural and forested areas with wetland variables:		
-Wetland quantity		
wetland area (Ha)	0.57	0.64
wetland (N)	0.70	0.23
total perimeter (Km)	0.62	0.67
-Wetland type		
deciduous forest (Ha)	0.65	0.32
coniferous forest (Ha)	0.58	0.59
mixed forest (Ha)	0.83	0.06
scrub-scrub/emergent(Ha)	0.57	0.61
Land use in buffer zones around wetlands:		
forest (ha)	0.59	0.51
agriculture (ha)	0.59	0.52
Wetland and streams:		
wetlands near first-order streams	0.76	0.13
wetland near first and second-order streams	0.79	0.10
wetlands near first, second, third-order steams	0.85	0.05
wetlands near first, second, third ,fourth-order steams	0.88	0.03
wetlands near first, second, third, fourth, fifth-order steams	0.78	0.10
total area of riparian wetlands	0.84	0.05

Here again, we used the strength of the regression (r^2 and significance level) to infer that wetlands in a particular spatial position are more important for P attenuation than wetlands located in other positions. While the numerical difference between using the 1st-through 4th-order riparian wetlands (r^2 =0.88) and all riparian wetlands (r^2=0.84) is unimportant and thus probably not justifiable as a basis for differentiating the two, the pattern of difference moving toward just lower-order streams or other measures of wetland presence suggests that riparian wetlands have some special status with regard to diffuse phosphorus pollution. Note this conclusion differs from that obtained from a more geographically restricted analysis of one of the watersheds (LaPlatte watershed reported above).

Finally, we inferred from the coefficients of the regression model (Pflux = 0.86 Agriculture + 0.64 Forest - 30 riparian wetland) that the sink power of riparian wetland areas (30) was many times greater than the source export of P from comparable-sized agricultural (0.86) or forested land (0.64). Depending on which wetland variable was used, the coefficient reflecting the strength of the P sink varied from –0.1 for mixed forest wetlands to –47 for wetlands proximal to 1st and 2nd order streams. This suggests that quantitative reliance on the coefficient as a measure of the per unit area annual flux of P is not justified without further corroboration. However, the consistency of the coefficient as a negative term that was considerably larger than the agricultural coefficient more strongly suggests that wetlands, and particularly riparian wetlands, are an important landscape element attenuating diffuse phosphorus pollution.

Our approach seems to allow a reasonable inference that landscape position is important for both sources and sinks of diffuse phosphorus. In both cases that we investigated, the important aspect of position is the relationship to the surface water network.

4. THE UTILITY OF THIS LANDSCAPE APPROACH.

The observations and analyses referred to above lead us to a variety of inferences about spatial and temporal patterns of diffuse P in the landscape. These inferences lead to 1) targeting land cover and agricultural practices most responsible for diffuse P pollution, 2) assessing at what times and under what conditions the risk is highest, and 3) refining our concept of diffuse P dynamics in the landscape to including structural components such as position with respect to the hydrologic network. Before concluding these inferences are worthy of being used as a framework for developing management plans, the rigor of these inferences must be considered.

4.1. Statistical Limitations

A tool should be used within the constraints of its assumptions. There are several such assumptions that are important in our use of multiple regression. Regression is a parametric procedure based on assumptions of normality of input variables and error terms. Thus the derived parameters of r^2 and the significance level of the model and its coefficients are very much dependent on the normality of the collected data. We relied heavily on these parameters in making inferences from the regression analyses presented in this paper. The normality of input variables is both easily calculated and often violated. The normality of the error term in the regression model is easily observed by examination of the model residuals. These residuals can be tested (e.g., Shapiro-Wilk W, JMP 3.2, SAS Institute). The problem arises if normality is not apparent and not easily compensated for by transformations of the data (e.g., log, arc sin, etc.). Models that did not meet this requirement could not be quantitatively compared. The homoscedasticity (constancy of variance) of the data is also important for performing hypothesis tests on the regression coefficients (Helsel and Hirsh, 1992). In our work several models required transformations of the data ($\log_{10}$) in order to approximate this requirement. However, once the data are transformed, comparisons between models with transformed and untransformed data are not possible. Much of what we inferred depended on a comparison of r^2's and significance levels in large matrices of regression results. In some cases, necessary transformations were not made to permit construction of the larger comparative tables.

Often variables are not independent and thus care must be taken to use variables only in independent sets. For example cultivated cropland and connected cultivated cropland include some of the same land area and thus should not be used concurrently. Multicollinearity of independent variables presents a different kind of problem in regression, causing instability in the coefficients. Multicollinearity was assessed using the variance inflation factor (VIF, Neter et al., 1989), and values greater than 10 resulted in the exclusion of problem variables. The number of independent variables usable in a multiple regression model is dependent on the degrees of freedom (number of watersheds in our case). In most of our analyses we had about 15 observations and therefore limited the number of independent variables to three because of limitations due to the relatively few degrees of freedom available. While there is no firm rule in this case, we felt using more than three variables lead to unjustifiably high r^2's and do not help with data interpretation.

As noted above, our sets of independent and dependent observations on land and phosphorus permitted many combinations (up to many hundreds) of variables to be included in multiple regression models. The significance levels reported hold true for individual models, but just like for multiple comparisons in ANOVA , the joint significance level for all the possible combinations of independent variables can be very much lower. We did not compensate quantitatively for the effect of multiple trials/analyses on reported significance levels, especially since the number of trials was dynamic. Thus significance levels in our analyses only suggest the strength of relationship (the probability of achieving the same result from random data). Again, strict quantitative comparison of either the r^2 or the significance parameters is probably not warranted and generates a false sense of quantitative rigor.

4.2. The Validity Of Inferences From Regression

Deriving cause and effect relationships from significant correlations or regressions is inappropriate. Nevertheless, social scientists, epidemiologists, ecologists and others and having been using this approach to test ideas and generate hypotheses about the nature and function of their respective subjects of study for many years. Does this approach require additional justification for its use as a research tool in landscape ecology?

Common to all the inferences made above is the idea that a higher r^2 value infers a stronger relationship. While this statement may be statistically correct (the above statistical caveats taken into consideration), this can not be taken to mean there is a parallel, stronger causal relationship. For instance, in both the Vermont and France studies we observed stronger relationships of P concentration or flux with cropland than with general agricultural land. While this makes intuitive sense and is supported by a great deal of research (Dillon and Kirchner, 1975; Omernick, 1976; Belamie, 1986; CIPEL, 1988; Sharpley et al., 1993), we can not rigorously defend that this means cropland exports more P than general agricultural land. Concluding that cropland exports more P than agricultural land employs inductive logic not normally considered to be appropriate at the later stages of the scientific method. Induction stimulates the generation of hypotheses that are then tested using standard, rigorous, scientific methods with proofs and significance levels. This tension between the classical scientific method and conclusions based on inductive reasoning and circumstantial evidence is evident in the literature of diffuse pollution (e.g., Whigham et al., 1988; Beaulac and Reckhow, 1982; Omernik, 1976), but is age-old and not limited to landscape-level investigations. Our purpose in bringing it up here is to provide some context for our approach, which makes causal inferences from correlated information.

As stated in the beginning of this paper, our approach is based on our conceptual model of the system we seek to understand. The dynamics of phosphorus in the landscape is conceived of as a highly interactive, diverse system. It is this conception that leads us to start our research by sampling extensively and to make inferences from statistical relationships rather than seek to define general truths from intensive study of a few elements in the landscape. A more classical approach might take the following steps: 1) study a landscape element, e.g., and farm field under silage corn production on a particular soil-slope situation, 2) develop a model based on observational and/or experimental studies to define the flux of P from the field under a variety of climatic/seasonal conditions, 3) use this model to deduce the phosphorus flux from other fields exposed to some other set of cultural and environmental conditions (for example, see Haygarth and Jarvis, 1997). If these other fields exist within the parameter boundaries of the initial intensive study, then the extrapolation may work and the deduction is within legitimate and accepted scientific practice. On the other hand, if the other field lies outside this boundary (e.g., with respect to slope, soil type, climate, vegetation, cultural practices, etc.), then the extrapolation is perhaps flawed and certainly not part of accepted scientific practice. This field-scale approach also overlooks other elements in the landscape system such as the buffer and attenuation elements, and in-stream processing.

In the milieu of applied science, intuitive decisions based on inferences from whatever information is available are perhaps the norm rather than the exception. The application of landscape ecology to current environmental problems highlights this use of scientific knowledge because it is a theory-poor young science with great responsibilities for guiding critically needed practices and policies for land management. Because we see little prospect of conducting the requisite number of intensive studies on the diversity of landscape elements, we suggest that taking an inferential approach using the "flawed" logic of “correlation suggests cause” is a useful complement to the more classical approaches taken by quantitative science. In fact, we suggest that this evidence obtained via regression analysis is a necessary complement to experimental and process-based investigations that can not be justifiably extrapolated to the broad diversity of circumstances found in our complex landscapes. At this time, with respect to understanding diffuse phosphorus pollution, both approaches provide only evidence rather than proof.

6. CONCLUSIONS

Diffuse phosphorus pollution is a landscape-level phenomenon requiring a spatial-temporal approach to clarifying questions of sources, causes, timing, and sinks. P flux is unevenly distributed in time and space as a result of its origins from a complex interaction of human activity and the biophysical landscape. While our initial regressions support the idea that land cover controls P flux, more detailed land cover data suggest that agricultural practices within the same land cover plays an important part in determining P flux. At the field scale, clearly the management of soil surface structure and/or vegetative cover over the soil during times of high runoff risk is essential. Thus management of diffuse phosphorus pollution needs to consider another aspect of P dynamics in the landscape that is determined by socioeconomic factors (technical knowledge of the farmer, tools available, traditions, number of landowners, community structure, etc.).

Our initial approach using synoptic sampling of surface waters across the landscape supports the idea that cultivated land is the dominant source of diffuse P in our landscapes. Our observations also provide evidence that landscape elements can have a P sink function. While individual wetlands may be sources or sinks for P, natural wetlands as a group, collectively act to reduce the load of phosphorus to surface waters during critical times of year. Managed grassland can also provide similar functions (Haygrath and Jarvis, 1997). More detailed information about activities (as opposed to land cover information) also improves our understanding of diffuse phosphorus. Practices in cultivated fields, especially those that lead to soil “crusting”/compaction or to better connection of parcels to the hydrologic network, are an important determinant of P fluxes. Finally, our observations provide evidence that the position of source and sink elements affects the total amount of P coming from the landscape. In particular, the hydrologic network is a critical structural element when considering position. Keeping agricultural practices with the greatest risk of generating runoff at some distance from artificial or natural drainages reduces P flux to surface waters by lowering the probability of hydrologic connection. Again, the presence and arrangement of both agricultural and nonagricultural lands depends on social factors in the local and regional community. These and other observations lead us to suggest that phosphorus dynamics in the landscape can be envisioned as taking place in an interacting social and biophysical system, and it is this entire system that needs to be considered in the continuing effort to manage diffuse phosphorus pollution.

Thus we suggest that diffuse pollution can be envisioned as a function of:

1) *landscape area* - the area of corn culture, grassland, residences, etc.; usually described as land cover and observable from airplanes or satellites;
2) *attenuators* - the effective elements in a landscape can be an areal, linear or count units; for example, buffer strips, riparian areas, and wetlands reduce P flux from landscapes;
3) *activities/practices* - the landscape is still measured as area, but divided into more components as a function of type of activities taking place on the land; for example the area of corn under no-tillage, under conventional tillage, etc.; this level of detail is usually not observable from satellites or airplanes;
4) *position* - all areas of a land cover type are not equal contributors, spatial position mediates pollution; for example if a corn field is many kilometers away from the active stream network, the diffuse pollution does not reach a potential flux point.

These conclusions are based on inferences drawn from observing the relationship between phosphorus flux and descriptions of the landscape. Because the need for management of diffuse pollution is current and great, we suggest that management decisions based on this interim level of understanding are warranted. Understanding based on more rigorous experimental approaches or intensive observations of better controlled systems will eventually provide a more confident basis for management, but for the immediate future we suggest that developing more detailed descriptions of the landscape (environmental and social surveys, remote-sensing, and GIS) coupled with broad-scale water sampling can continue to provide much needed information on diffuse phosphorus dynamics and the risk of eutrophication of surface waters.

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