Phosphorus Loss in Agricultural Drainage: Historical Perspective and Current Research

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ABSTRACT

The importance of P originating from agricultural sources to the nonpoint source pollution of surface waters has been an environmental issue for decades because of the well-known role of P in eutrophication. Most previous research and nonpoint source control efforts have emphasized P losses by surface erosion and runoff because of the relative immobility of P in soils. Consequently, P leaching and losses of P via subsurface runoff have rarely been considered important pathways for the movement of agricultural P to surface waters. However, there are situations where environmentally significant export of P in agricultural drainage has occurred (e.g., deep sandy soils, high organic matter soils, or soils with high soil P concentrations from long-term overfertilization and/or excessive use of organic wastes). In this paper we review research on P leaching and export in subsurface runoff and present overviews of ongoing research in the Atlantic Coastal Plain of the USA (Delaware), the midwestern USA (Indiana), and eastern Canada (Quebec). Our objectives are to illustrate the importance of agricultural drainage to nonpoint source pollution of surface waters and to emphasize the need for soil and water conservation practices that can minimize P losses in subsurface runoff.

THE TRANSPORT of P from agricultural soils to surface waters sensitive to eutrophication has been a worldwide environmental concern for more than 30 yr. Almost 25 yr ago, Ryden et al. (1973) reviewed the body of research on soil and hydrologic factors controlling the loss of P from agricultural and forested watersheds and in urban runoff, and stated: "Increasing evidence suggests that P in surface waters is a primary factor controlling the eutrophication of surface water supplies" and that "Assessment of the relative contribution of the different sources of P to surface waters is of critical importance to prevent or reverse P-induced eutrophication." Later reviews have provided critical analyses of the processes involved in the release, transport, and biological availability of soil P (Logan, 1982; Sonzogni et al., 1982), the specific impacts agricultural P can have on surface water bodies (Correl, 1998; Sharpley and Menzel, 1987; Sharpley et al., 1987; Taylor and Kilmer, 1980), and, perhaps most importantly, of the management practices needed to minimize the eutrophication of surface waters by P from agricultural watersheds (Daniel et al., 1994; Flaig and Reddy, 1995; Sharpley et al., 1994).

Despite the extensive body of knowledge on this subject, we still, today, face significant problems with the nonpoint source pollution of surface waters by agricultural P (Foy and Withers, 1995; USEPA, 1994). As stated by Sharpley et al. (1994), "...one of the main issues facing the establishment of effective nonpoint source management controls (for surface water quality) is the development of economically and environmentally sound P management systems and the balancing of productivity with environmental values." Not surprisingly, the problems are most severe in areas where water movement from soil to surface water is greatest and where soil P levels are highest. Areas where soils high in P predominate can be rapidly identified by a geographic analysis of soil test P data from public or private soil testing programs (Fig. 1), although this provides no information on the hydrologic or management factors important to P transport to surface waters. To identify agricultural fields most vulnerable to P loss, Lemunyon and Gilbert (1993) proposed a rather simple, field-scale analysis, the Phosphorus Index, which integrates soil test data, soil erosion and runoff potentials and P fertilizer or organic waste application rate, method, and timing. This index was later validated by Sharpley (1995) for watersheds in Oklahoma and Texas. More sophisticated mod-

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Abbreviations: BAP, biologically available P; DIP, dissolved inorganic P; DPS, degree of P saturation; DRP, dissolved reactive P; EPC_0, equilibrium P concentration at zero sorption; Ortho P, orthophosphate; MRP, molybdate reactive P; PSC, P sorption capacity; TP, total P; TDP, total dissolved P; TPP, total particulate P.

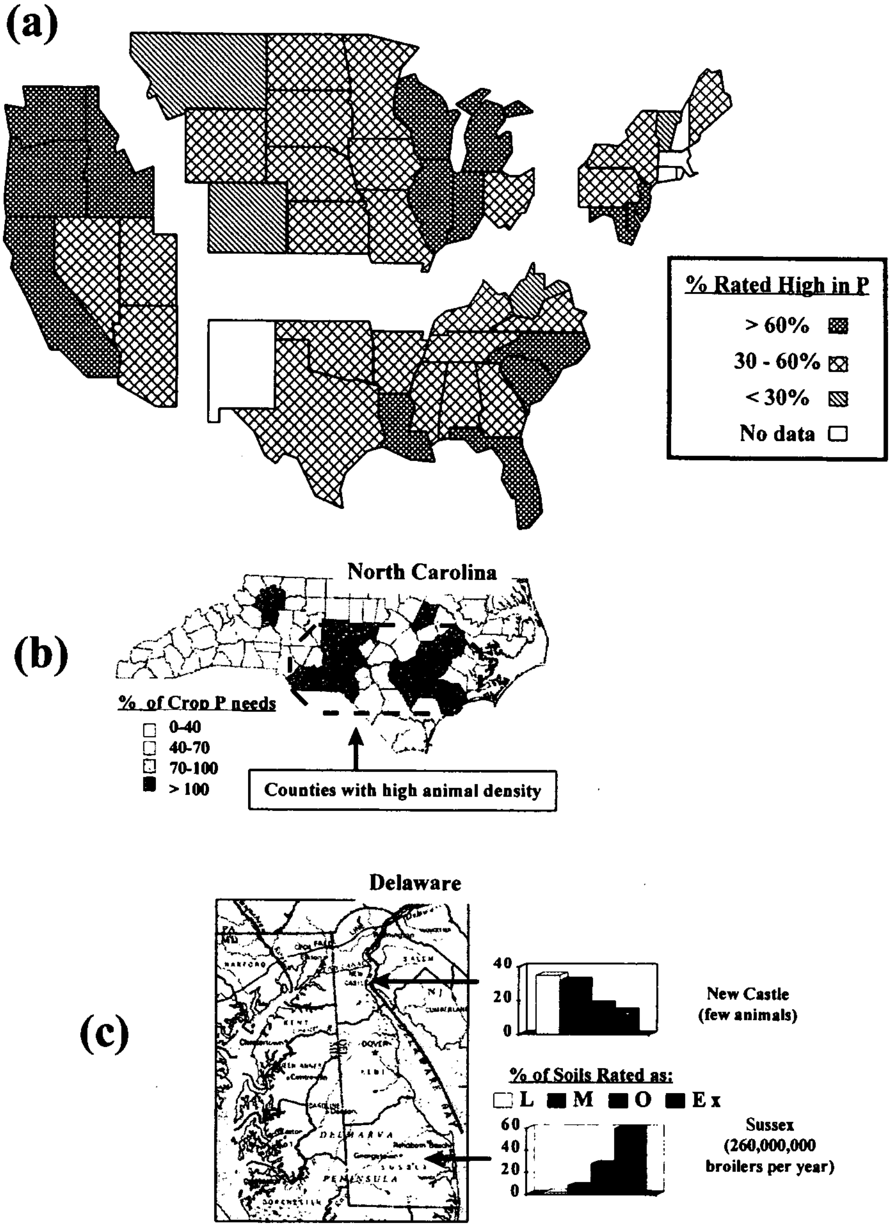

Fig. 1. Illustration of the use of soil testing summaries to characterize the geographic distribution of high P soils in (*a*) the United States, by region (PPI, 1994); (*b*) North Carolina counties with high animal densities (S. Hodges, Dep. of Soil Science, North Carolina State Univ., 1996, personal communication); and (*c*) Delaware for a county with low animal density (New Castle) and a highly concentrated poultry industry (Sussex) (Sims, 1997).

els (e.g., EPIC and AGNPS) have also been used to identify areas with greater potential for P loss and to simulate the effects of changing soil and crop management and land uses on nonpoint source pollution of surface waters by P (Sugiharto et al., 1994).

The emphasis of most of the research reviewed by these authors, and subsequently integrated into soil and water conservation practices and watershed scale indexes or models, has been on P transport by surface erosion (detachment and movement of soil by water; Brady and Weil, 1996) and surface runoff (that part of rainwater or snowmelt that flows over the land surface to stream channels; Ryden et al., 1973). We distinguish between these two processes to emphasize that it is not only particulate P loss from soils that is important for nonpoint source pollution, but dissolved P as well. Focusing on erosion and runoff as important transport processes for P is certainly justified as many studies have shown that P from fertilizers, animal wastes, and other soil amendments (e.g., municipal biosolids) accumulates in agricultural topsoils where it is bound to the most erodible soil components (clays, organic matter, and oxides of Fe and Al). Consequently, P leaching and subsequent losses of P via subsurface runoff (that part of precipitation that infiltrates the surface soil and moves toward streams as ephemeral, shallow, perched groundwater and that in many agricultural areas...may be intercepted by artificial drainage systems, for example, tiles, drains, accelerating its movement to streams; Ryden et al., 1973) has, until recently, rarely been considered a significant transport process for agricultural P to surface waters.

In this paper we review research, primarily in the USA and Canada, on the loss of P from agricultural soils by subsurface runoff, with particular emphasis on P export in artificial drainage systems (tiles and ditches). We also present summaries, case studies of a sort, from three distinct hydrologic settings: the Atlantic Coastal Plain of the USA (Delaware), the midwestern USA (Indiana), and eastern Canada (Quebec). These studies provide specific examples of the importance of artificial land drainage to nonpoint source pollution of surface waters by agricultural P and point to the need for soil and water conservation practices to minimize P export by these processes.

PHOSPHORUS LEACHING IN AGRICULTURAL SOILS

Environmentally significant export of anthropogenic P from agricultural soils by subsurface runoff begins with downward movement of P, either by slow leaching through the soil profile or preferential flow thorough macropores (e.g., soil cracks, root channels, earthworm [*Lumbricus terrestris*] borings). Phosphorus leaching has normally been considered to be inconsequential in most soils but a review of the literature finds that there are combinations of agricultural management practices, soil properties, and climatic conditions that can result in significant P accumulations in subsoils. Whether or not P that leaches into subsurface horizons is later transported from subsoils to surface waters depends on the depth of P leaching and the hydrology of the watershed; however, as noted by Ryden et al. (1973): "...losses of P to subsurface and ground water runoff, although of little significance from an agricultural standpoint, may under certain conditions constitute a significant loss of P from agricultural watersheds in terms of the P enrichment of surface waters" and that "...losses of P in subsurface runoff can be similar or even greater than those in surface runoff." For instance, Ryden et al. (1973) cited eight studies where dissolved inorganic P (DIP) concentrations in subsurface runoff in artificial drainage systems were higher than values usually associated with eutrophication of surface waters (0.01–0.05 mg/L; Foy and Withers, 1995; Sawyer, 1947; Sharpley et al., 1996; Vollenweider, 1968; USEPA, 1986). Mean DIP concentrations in these studies were from 0.012 to 0.44 mg/L. They also cited studies where P export (kg/ha per yr) in subsurface flow equaled or exceeded values typically found in surface runoff (Bolton et al., 1970; Sylvester, 1961).

Investigations of P leaching in soils amended with commercial fertilizers and/or organic wastes have been conducted intermittently for more than 50 yr and more intensively in the last 15 yr as interest has grown in reducing nonpoint source pollution of surface waters by P. Previous research has generally shown that in most mineral soils if fertilizer P is applied at agronomically acceptable rates based on soil testing recommendations, it is unusual for significant P leaching to occur. However, P leaching can occur in deep sandy soils, in high organic matter soils, and in soils where overfertilization and/or excessive use of organic wastes have increased soil P values well above those required by crops. Environmentally significant quantities of P may then be exported if subsurface runoff occurs naturally and moreso if it is enhanced by artificial drainage systems.

Several studies from Florida, a U.S. state with an intensive, high-input agriculture, an extremely humid climate with frequent, heavy rainfall events, and widespread use of irrigation and drainage, illustrate the extent of P leaching that can occur in deep, sandy soils. In one of the earliest studies, Bryan (1933) reported P leaching to depths of at least 90 cm in heavily fertilized (1680–3360 kg superphosphate/ha per year) Florida citrus groves of varying ages, all located on deep, sandy soils. Available P (0.002 *N* H_2SO_4-extractable) at the 0 to 30, 30 to 60, and 60 to 90 cm depths was 173, 76, and 48 mg/kg in old citrus groves (45–65 yr in age) compared to 23, 9, and 7 mg/kg in young groves (2–6 yr in age), and 10, 14, and 6 mg/kg in nearby virgin soils. In another early Florida study, Neller (1946) used field lysimeters and natural rainfall to evaluate P leaching from the A horizon of an acidic (pH 4.0–5.0) Leon fine sand (sandy, siliceous, thermic Aeric Alaquods) amended with several different sources of P. The percentage of added P leached ranged from 62 to 99% when the soil was amended with superphosphate and calcined phosphate and from 1.2 to 10.5% for ground rock phosphate. Related studies with five acidic (pH 4.2–5.8) fine sandy loams, however, resulted in leaching losses of <7% of P added as superphosphate suggesting that the presence of even small quantities of clay and/or Al or Fe oxides can retard P leaching. Spencer (1957) reported "adsorbed" P (NH_4F-extractable) values in a Florida citrus grove located on a Lakeland fine sand at the 45 to 60, 60 to 90, and 90 to 120 cm depths were approximately 100, 60, and 40 mg/kg in limed soils and 150, 100, and 60 mg/kg in unlimed soils; adsorbed P in a nearby unphosphated soil ranged from 20 to 25 mg/kg at these same depths. Humphreys and Pritchett (1971) examined the distribution of P in the profiles of six soil series in northern Florida 6 to 10 yr after applying superphosphate for the production of slash pine (*Pinus elliottii*). Based on total P distributions in the soil profile, they reported that all fertilizer P had leached below a depth of 50 cm in two very sandy (>93% sand) soils (Pomello [sandy, siliceous, hyperthermic Oxyaquic Alorthods] and Myakka [sandy, siliceous, hyperthermic Aeric Haplaquods] sands) with extremely low Al contents (<0.10 cmol/kg) and that extensive P leaching and subsequent accumulation in a spodic horizon occurred in a third deep sandy soil (Leon fine sand). Phosphorus leaching was markedly reduced in finer-textured soils or in those with higher concentrations of Al.

Phosphorus leaching has also been reported to occur in organic soils or coarse-textured mineral soils with high organic matter contents, presumably because of the low concentrations of the soil constituents primarily responsible for P retention (clays, oxides of Fe and Al, and carbonates). Fox and Kamprath (1971) evaluated P leaching from an acid, organic sand (90% sand and 10% organic matter) and an acid, organic muck soil in a laboratory column study. All added fertilizer P was rapidly leached from the organic sand and from 85 to 100% of the added P from the muck soil. In another column study Larsen et al. (1958) reported that P leaching from five muck soils depended upon sesquioxide content. About 80% of applied fertilizer P (200 kg/ha) was leached from a virgin muck soil low in sesquioxides (850 mg total sesquioxides/kg) relative to ~20% from a cultivated Gumz muck soil with 4100 mg/kg sesquioxides; fertilizer P moved <4 cm in a mineral soil. Cogger and Duxbury (1984) noted that the P retention capacity of organic soils was a function of total mineral content, length of period of cultivation, liming, and the content of Ca, Al, and Fe. They estimated that as much as 44% of fertilizer and mineralized P could be lost via leaching from a deep, cultivated Carlisle muck (euic, mesic Typic Medisaprists) low in total Al+Fe (0.84%), relative to <2% from a shallower Palms muck (loamy, mixed, euic, mesic Terric Medisaprists) higher in total Al+Fe (1.9%) and suggested that sesquioxide content (Al+Fe) was the best predictor of P leaching from organic soils in New York.

Continuous, long-term fertilization in excess of crop requirements has also been shown to cause P leaching in fine-textured soils. Kao and Blanchar (1973) reported the effects of 82 yr of almost continuous manuring (1888–1927) and P fertilization (1927–1970) on the forms and availability of P in the profile of a Mexico silt loam (a fine, montmorillonitic, mesic Mollic Endoaqualf) used for agronomic crop production in Missouri. Significant P leaching occurred to a depth of about 1.0 to 1.4 m; for example, total P and available P (0.1 N HCl + 0.03 N NH_4F) values were 700 and 67 mg/kg in the C1 horizon (86–107 cm) of a continuously fertilized plot relative to 388 and 31 mg/kg in the C1 horizon (107–137 cm) of a nearby unfertilized soil. Schwab and Kulyingyong (1989) evaluated changes in soil test P (Bray P_1) in a Smolan silt loam (a fine, montmorillonitic, mesic Pachic Argiustoll) after 40 yr of continuous fertilization with superphosphate (40 kg P/ha) and cropping with smooth brome grass (*Bromus inermis* Leysser) in Kansas. Continuously fertilized soils had Bray P_1 values in the surface horizon of 350 mg/kg vs. 13 mg/kg in unfertilized soils (a Bray P_1 value of $\sim$30 mg/kg is considered optimum for most crops; Knudsen and Beegle, 1996); soil test P values at the 30, 60, and 90 cm depths were 56, 10, and 3 mg/kg for continuous fertilization vs. 3, 2, and 2 mg/kg when no P was added. It is important to note that while these two studies indicate that P leaching can occur in fine-textured soils they are somewhat unrealistic because they represent situations where fertilizer P had been applied continuously, independent of soil test P, a management practice that is not recommended, is not economical, and thus would be unlikely to occur for extended periods of time.

While the previous studies clearly show that there are some situations where the use of inorganic P fertilizers can result in P leaching (i.e., deep sandy soils, organic soils, or long-term overfertilization with P), the most common agricultural situation associated with significant downward movement of P has been the accumulation of P to "very high" or "excessive" levels in soils from continuous applications of organic wastes (manures, litters, and municipal or industrial wastes and wastewaters). As will be discussed in more detail below, this is an extremely problematic environmental issue for, unlike fertilizer P, land application programs for organic wastes often cannot avoid annual applications of P, even if soil test P is in an optimum or excessive range, because of the continual need to dispose of these wastes. Several recent studies of the effects of animal wastes on P leaching are compared in Fig. 2 and described below.

Sharpley et al. (1984) determined the distribution of total, inorganic, organic, and available P in the profile of a Pullman clay loam (a fine, mixed, thermic Torrertic Paleustoll) used for sorghum [*Sorghum bicolor* (L.) Moench] production in Texas after long-term fertilization and manuring with cattle feedlot waste (FLW). Total P in the soil profile of a nearby virgin prairie soil, a cropped control plot, and a NPK fertilizer treatment (56 kg P/ha per year for 8 yr) was 328, 312, and 350 mg/kg, respectively. In contrast, where FLW was applied at 22 and 67 Mg/ha per year for 8 yr or 269 Mg/ha per year for 5 yr, total P was 538, 996, and 1278 mg/kg. Most of the P accumulated in the upper 45 cm of the soil profile, and particularly in the top 30 cm where soil test P (Bray P_1) values were 16, 15, 56, 63, 230, and 370 mg/kg for the virgin prairie, control, NPK, and FLW at 22, 67, and 269 Mg/ha per year treatments, respectively (Fig. 2a). Kingery et al. (1994) reported P leaching to a depth of $\sim$60 cm in tall fescue (*Festuca arundinacea* Schreb.) pastures in the Sand Mountain region of northern Alabama that had received long-term applications (15–28 yr) of poultry litter (Fig. 2b). Again, most P accumulated in the upper 45 cm of the soil and soil test P (Mehlich 1) values in topsoils were extremely high ($\sim$230 mg/kg) relative to optimum values for crop production in this region (25 mg/kg; Cope et al., 1981). In a similar study in a more arid region, Sharpley et al. (1993) reported that long-term (12–35 yr) poultry litter application (4.5–9.0 Mg/ha per year, dry weight basis) to bermuda grass [*Cynodon dactylon* (L.) Pers.] pastures in Oklahoma caused marked increases in soil test P and some P leaching, but only to a depth of 30 cm (Fig. 2b). Average soil test P (Bray P_1) values for 12 litter amended sites at the 0 to 5 and 0 to 30 cm depths were 188 and 290 mg/kg compared to 9 and 22 mg/kg in untreated soils from nearby fields. They estimated that from 59 to 92% (avg. = 72%) of the litter P applied accumulated in the upper 30 cm, primarily as inorganic P. Litter applications also reduced P sorption capacity at the 0 to 5 cm soil depth as indicated by a decrease in the average P sorption index (Bache and Williams, 1971) of the 12 soils from 354 to 174. Decreases in P sorption capacity with manuring, also reported by Mozaffari and Sims (1996), increase the likelihood of P loss to surface waters via erosion and runoff as well as the potential for long-term P leaching to subsoils.

Simard et al. (1995) reported P leaching occurred in agricultural soils from a watershed with high livestock (hog, *Porcus* sp.) density in eastern Quebec. Total P values in the A, B, and C horizons of forest soils in this watershed were $\sim$725, 625, and 575 mg/kg, respectively. In comparison, total P for the A, B, and C horizons of dairy farms were 1010, 675, and 650 mg/kg and for surplus N farms (those with intensive hog production and an average surplus of 230 kg manure N/ha per year) were 1150, 700, and 725 mg/kg, respectively. Increases in water-soluble and soil test P (Mehlich 3; Fig. 2c) paralleled changes in total P. Water-soluble P in the A, B, and C horizons of forest soils were <2.5 mg/kg; for the A, B, and C horizons in soils from dairy farms and surplus N farms water-soluble P was 7.8, 3.5, and 3.5 mg/kg and 7.3, 3.0, and 4.0 mg/kg, respectively.

Eghball et al. (1996) measured soil test P (Olsen P: 0.5 M $NaHCO_3$) in the profile of a Tripp very fine sandy loam (a coarse-silty, mixed, mesic Aridic Haplustoll) that had received long-term ($>$50 yr) applications of cattle feedlot manure and/or fertilizer P (Fig. 2d). Crops grown included sugarbeet (*Beta vulgaris* L.), potato (*Solanum tuberosum* L.), and corn (*Zea mays* L.). Increases in soil test P showed P leaching had occurred to $\sim$75 cm with fertilizer P (superphosphate) and to $\sim$1.0 m for manure or manure+fertilizer P. Phosphorus leached from the superphosphate treatment remained above a subsoil layer with a high P sorption capacity (due to a high $CaCO_3$ content), but P in manured soils moved through this layer, thereby suggesting a greater downward mobility for organic forms of P.

Kuo and Baker (1982) evaluated the leaching of P in five pastureland soils in eastern Washington, ranging in drainage class from well-drained to poorly drained, that had received long-term (20+ yr) applications of beef manure slurry (Fig. 2d). Total and soil test (Olsen P) P were increased to a depth of $\sim$75 cm in manured soils, relative to a nearby control soil and P leaching was generally greater in the more well-drained soils.

Mozaffari and Sims (1994) measured soil test P (Mehlich 1) values with depth in cultivated and wooded soils on farms in a coastal plain watershed dominated by intensive poultry production and frequent applications of poultry litter (Fig. 2e). Phosphorus leaching to depths of $\sim$60 to 75 cm was commonly observed in agricultural fields, as were very high soil test P values in topsoils relative to those considered optimum for most agronomic crops (25 mg/kg; Sims and Gartley, 1996).

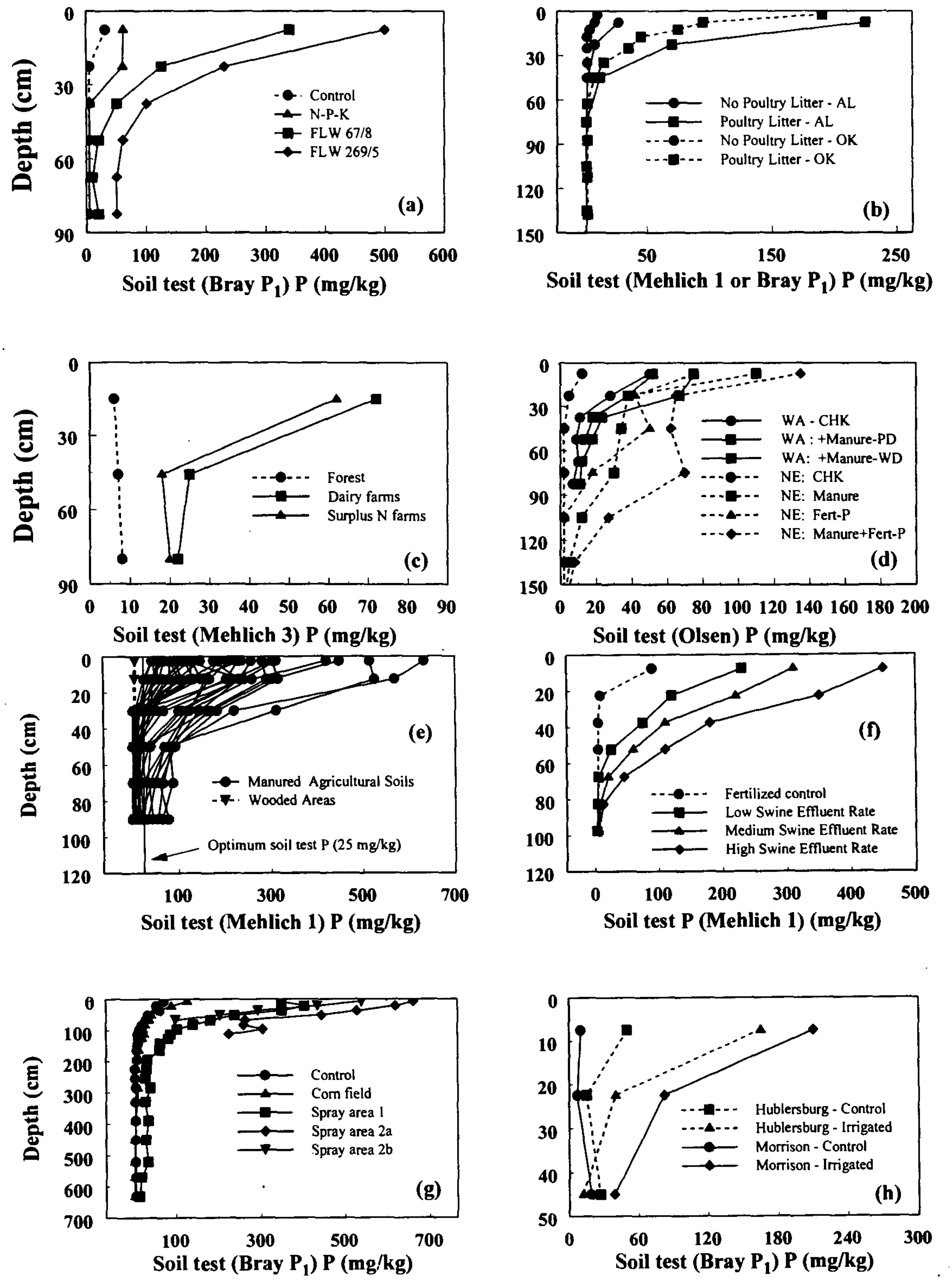

Fig. 2. Summary of field research evaluating the influence of organic wastes (animal manures and wastewater irrigation) on P leaching in soils. Results shown are discussed in text and are adapted from (*a*) Sharpley et al., 1984; (*b*) Kingery et al., 1994; (*c*) Simard et al., 1995; (*d*) Kuo and Baker, 1982; (*e*) Mozaffari and Sims, 1994; (*f*) King et al., 1990; (*g*) Adriano et al., 1975; and (*h*) Sommers et al., 1979. Abbreviations used in figures are: N–P–K = inorganic fertilizer treatment providing N, P, and K; FLW = cattle feedlot waste; CHK = check plot; PD = poorly drained soil; WD = well-drained soil; Fert-P = fertilizer P treatment.

King et al. (1990) examined the effect of 11 yr of swine lagoon effluent applications on P distribution within the profile of a Paleudult soil used for coastal bermuda grass pasture in North Carolina (Fig. 2f). As in the above studies topsoil values for soil test P (Mehlich 1) were much greater than required for crop production (from 225–450 mg/kg, as a function of effluent rate, vs. an optimum soil test value of ~20–25 mg/kg) and leaching of P occurred to ~75 cm. Soil test P at the 15 to 30, 30 to 45, 45 to 60, and 60 to 75 cm depths was <5 mg/kg in a nearby unfertilized pasture. However at the same depths, soil test P was about 120, 75, 25, and 5 mg/kg at the lowest effluent rate (335 kg N/ha per year) and 350, 175, 125, and 50 mg/kg at the highest effluent rate (1340 kg N/ha per year).

Many other studies have reported P leaching in manured soils (Campbell and Racz, 1975; Campbell et al., 1986; Chang et al., 1991; James et al., 1996; Meek et al., 1979, 1982; Reddy et al., 1980; Vitosh et al., 1973) but research is more limited for agricultural soils amended with other organic wastes. However, P leaching has been observed in soils irrigated with municipal or industrial wastewaters. Adriano et al. (1975) reported that long-term spray irrigation with food processing wastewaters caused P leaching to as deep as 6.6 m in two sandy, Michigan Haplorthods (Fig. 2g). Ortho-phosphate concentrations in subsurface waters at the two sites averaged 0.92 and 0.58 mg P/L, consistently greater than surface discharge limits (0.05 mg P/L) in Michigan. Annual P losses at the two sites were 26 and 11 kg/ha, equivalent to 27 and 2% of the soluble P input.

Sommers et al. (1979) evaluated the effect of 11 yr of irrigation with municipal wastewaters on P leaching at two Pennsylvania sites located on a Hublersburg clay loam (a clayey, illitic, mesic Typic Hapludult) or a Morrison sandy loam (a fine-loamy, mixed, mesic Ultic Hapludalf). Little P leaching was observed with the Hublersburg clay loam, but irrigation significantly increased total P (from 74–262 mg/kg) and available P (Bray P_1; from 19–39 mg/kg) at the 30 to 60 cm depth of the Morrison sandy loam (Fig. 2h). Irrigation also caused marked increases in EPC_0 (solution P concentration at which neither sorption nor desorption of P occurs; Taylor and Kunishi, 1971), from 1.2 to 73.0 μg/L, and decreases in P sorption capacity at the 30 to 60 cm depth of the Morrison sandy loam, suggesting that subsoils at this site were becoming saturated with P and were maintaining soluble P concentrations above values normally associated with eutrophication of surface waters. Increases in EPC_0 values were also noted at the 15 to 30 cm depth of the Hublersburg soil (from <0.1–6.5 μg/L for non-irrigated soils and from 5.4–19.0 μg/L for irrigated soils) but not at the 30 to 60 cm depth.

The documented accumulations and leaching of P in soils described above have begun to raise questions, particularly in areas dominated by concentrated animal production, as to whether the P sorption capacity of some soil profiles is becoming saturated to the extent that continuous, environmentally significant leaching of P will occur. If so, subsurface runoff could be expected to be an important source of P export from agricultural fields to surface waters. Breeuswma et al. (1995) reviewed the concept of P saturation of soils, citing data from numerous studies in the Netherlands. In one example, more than 80% of the soils in a watershed with intensive livestock production were sufficiently saturated with P during winter months, when groundwater discharge to surface waters was highest, to cause concentrations of total P in the shallow groundwater to exceed surface water quality standards (0.15 mg /L of total P). More than 40% of the areas sampled had total P concentrations in groundwater of >1.0 mg/L. The surpluses of P associated with intensive animal production (dairy, pigs, and poultry) and the resulting high P levels in manured agricultural soils are clearly the major contributing factor to P leaching and P enrichment of shallow groundwaters in the Netherlands. However, Breeuswma et al. (1995) also reported that the agriculturally inevitable P losses that result if soil fertility is maintained in an amply sufficient range strictly by the use of commercial fertilizers may also result in P saturation of soils and P losses that are not environmentally acceptable in the long term. They estimated for soils in the Netherlands that leaching losses of P > 0.43 kg/ha per year would cause groundwater P to exceed the recommended surface water standard (0.15 mg total P/L), and concluded "...P saturation cannot always be prevented by following fertilizer recommendations. Soils with a low natural fertility level require a substantial surplus of P to obtain the desired soil-P level. In addition, a small surplus is needed to control this level. This compensates for leaching losses, which are nevertheless substantially higher than what is acceptable environmentally. Therefore, in sensitive areas, P concentrations in ground water and surface waters may increase (on a long term) even under good agricultural practices."

In response to the fact that the P sorption capacity of soils is not unlimited, Dutch environmental policies have incorporated the concept of a critical degree of P saturation (DPS), defined as the "...saturation percentage that should not be exceeded to prevent adverse effects on ground water quality" with the specific goal that "the phosphate concentration in the ground water should not exceed 0.10 mg/L of ortho P at the level of the Mean High Water table" (Breeuswma et al., 1995). A critical DPS value of 25% has been used in the Netherlands to determine the surplus of P that can be applied to varying soil types before P saturation, and thus significant P export in subsurface runoff, can be expected to occur. Operationally, DPS is defined as oxalate-extractable P divided by the phosphate sorption capacity of the soil that is estimated from equations including oxalate-extractable Fe and Al (Breeuwsma et al., 1995).

Assessment of the effect of land use (e.g., cropping systems and livestock density) on the degree of soil P saturation and hydrologic parameters on the potential for P loss via subsurface runoff has been conducted recently in the Netherlands (Breeuswma et al., 1995) and Belgium (Lookman et al., 1996) at watershed and national scales. The aim of this research has been to develop watershed-scale maps of P saturation to identify areas to implement improved management practices or impose regulations for the more environmentally efficient use of fertilizer and manure P. Lookman et al. (1995) applied the DPS concept to 700 km^2 area in northern Belgium where the typical annual input of manure P (65–150 kg/ha) greatly exceeds P outputs in harvested crops (~30 kg/ha). The area is primarily grassland and is the site of intensified animal agriculture. Based on a critical DPS value of 25%, they estimated that >75% of the soils were considered to be saturated with P to a depth equal to the highest average groundwater table. Average DPS values for arable land (primarily used for corn production) were 43% and for grassland were 35%. In a subsequent study (Lookman et al., 1996) with the same soils they showed that DPS at the 0 to 30 cm depth was highly correlated with soluble P in these soils ($r^2 = 0.78^{***}$) and thus to the potential for P loss in runoff or leaching. However, Beauchemin et al. (1996) cautioned that a single parameter alone (e.g., water soluble P, soil test P, or DPS) cannot be expected to accurately predict the potential for P leaching, stating that "...a measure of soil P capacity or intensity of P accumulation (e.g., DPS) must be accompanied by a measure of desorbability (e.g., water soluble or soil test P) to fully assess the risk of contamination of drainage waters by P leaching." They also pointed out that the critical DPS value will vary with land use, soil physical and chemical properties, and surface water quality standards. For instance, from their study of a watershed with high animal density in Quebec, where the surface water limit for P (0.03 mg/L) is lower than the Netherlands, a DPS value of 9% in A horizons of agricultural soils might represent sufficient P saturation to be of environmental concern.

PHOSPHORUS LOSSES IN AGRICULTURAL DRAINAGE

Accumulations of total, extractable, or water-soluble P in subsoils and/or high DPS values provide evidence that P leaching has occurred or is likely to occur but, without water quality and hydrologic data, do not provide direct evidence of P export from agricultural fields via subsurface runoff. However, several studies of agricultural drainage systems (tiles and ditches) have determined P concentrations and export (kg/ha per yr) and related these values to soil properties and soil management practices. Several of these studies are reviewed below to illustrate key points about P loss via subsurface runoff; others are summarized in Table 1.

As mentioned above the traditional dogma with regard to P losses has been that surface erosion and runoff are of greater concern than P leaching and subsurface runoff. Hanway and Laflen (1974) verified this in a study that measured total and soluble P losses via surface runoff and tile drainage for 3 yr at four Iowa farms. Total P losses, averaged over all sites and years, were much greater from surface runoff than tile drainage (0.72 vs. 0.015 kg/ha per year). Average total and soluble P concentrations were also higher in surface runoff (2.08 and 0.02 mg/L) than in tile drainage (0.06 and 0.01 mg/L). This study supports the contention that P losses in subsurface runoff from fine-textured soils with moderate to optimum soil test P values (Bray P_1 = 7, 17, 18, and 42 mg/kg) are unlikely to be of serious environmental concern, relative to losses in erosion and runoff.

Studies have also shown that, in addition to increasing soil productivity, artificial drainage can decrease total P losses. Bengston et al. (1988) evaluated the effects of tile drainage on P losses from a Commerce clay loam (a fine-silty, mixed, nonacid, thermic Aeric Fluvaquent) used for corn production in Louisiana for 5 yr. Total P losses from undrained soils averaged 7.8 kg/ha per year, all via surface runoff. When soils were drained, total P losses decreased to 5.0 kg/ha per year and only 6% of the P loss occurred via tile drainage. Lower total P losses in drained soils were primarily due to reductions in soil loss by surface erosion, from an annual average of 4986 kg/ha for undrained soils to 3482 kg/ha for drained soils.

While these studies and others cited in Table 1 generally support the contention that P losses via subsurface runoff are likely to have little environmental impact, other research provides contrasting evidence. Calvert (1975) measured P losses in tile drainage from an Oldsmar sand (a sandy, siliceous, hyperthermic Arenic Alaquod) used for citrus production in Florida, as affected by surface tillage (ST), deep tillage (DT) to a depth of 105 cm to incorporate clay and organic matter from an underlying spodic horizon into the sandy soil above the tile lines, and by deep tillage plus liming (DTL). Average soluble P concentrations in tile drainage waters from ST plots exceeded values associated with eutrophication of surface waters, averaging 0.63 and 0.42 mg/L, for spring to early fall and fall to spring, respectively. Deep tillage or tillage plus liming reduced P losses significantly relative to surface tillage alone. Average soluble P concentrations in tile drainage waters from spring to early fall were 0.27 and 0.17 mg/L for DT and DTL, and from fall to the following spring were 0.07 and 0.007 mg/L. Estimated losses of applied fertilizer P were 14, 3, and 2% for ST, DT, and DTL, respectively. This study and the research cited above on P leaching in sandy soils (e.g., Bryan, 1933; Mozaffari and Sims, 1994; Neller, 1946) illustrate the potential for environmentally significant losses of P from artificially drained, sandy soils.

Agricultural cropping of organic soils is also commonly associated with P losses in subsurface runoff. Miller (1979) compared P leaching losses via subsurface drainage from intensively cropped mineral (Humaquepts) and organic soils (Terric Borosaprist) in Ontario. Dissolved ortho-P concentrations and losses from the organic soils were much higher than in any of the studies discussed earlier, ranging from 1.14 to 18.2 mg P/L and 1.6 to 36.8 kg/ha per year, respectively and were considerably higher than losses from mineral soils in this study (dissolved ortho-P concentrations of 0.015–0.072 mg/L and losses of 0.03–0.24 kg/ha per year). The authors also noted that, for one organic soil, ortho-P concentrations in drainage fluctuated in a similar manner as NO_3–N and suggested that P was behaving as a nonreactive solute. In other organic soils, P sorption and thus potential leachability was primarily controlled by the total Fe+Al content. High rates of fertilizer P previously used at these sites for >30 yr (~100 kg P/ha per year) were also identified as a cause of the very high P concentrations in subsurface drainage. In similar studies, Izuno et al. (1991) quantified total and dissolved P concentrations in surface drainage waters (ditches and canals) from sugarcane (*Saccharum officinarum* L.), vegetable, and fallow fields in the Everglades Agricultural Area (EAA) on muck soils (Lithic Medisaprists). Average total P concentrations in field drainage waters ranged from 0.25 to 1.03 mg/L, relative to concentrations of 0.07 mg/L in rainfall and 0.16 mg/L in main farm canals. Total dissolved P concentrations ranged from 0.17 to 0.51 mg/L vs. 0.08 mg/L in the main farm canals. Total P loading rates ranged from 0.59 to 4.70 kg/ha per year, much of which was as dissolved P (0.34–1.48 kg/ha per year); inputs of P from precipitation were estimated at 0.72 kg/ha per year. Heavily fertilized vegetable crops and fields subjected to regular flooding and draining cycles had the highest P concentrations in drainage waters and the highest P losses. In a related study, Coale et al. (1994b) evaluated the effect of drainage management on P losses from sugarcane fields in the EAA. Slow drainage practices resulted in significantly higher total P and total dissolved P concentrations in field drainage waters (0.13–0.38 and 0.04– 0.27 mg/L) than fast drainage (0.08– 0.22 and 0.04–0.14 mg/L). Off-field total P loadings, however, were greater for fast (0.03–0.24 kg/ha) than slow drainage (<0.01–0.12 kg/ha) because of a greater volume of water discharged in the fast drainage treatment (770 vs. 297 m^3/ha).

Agricultural management practices, such as tillage method, may also affect the loss of P via subsurface runoff. Gaynor and Findlay (1995) compared P loss in surface runoff and tile drainage from a Brookston clay loam (a fine-loamy, mixed, mesic Typic Argiaquoll) used for corn production, as affected by tillage practice (zero till, ridge till, and conventional tillage). Three-year average concentrations of ortho P in tile drainage waters were 0.24, 0.34, and 0.54 mg/L for conventional, ridge, and zero tillage, respectively, relative to 0.29, 0.55, and 1.02 mg/L in surface runoff. Surprisingly, ortho-P losses (grams per hectare per year) were higher in tile drainage than surface runoff for all three tillage systems, with 3-yr averages of 382, 527, and 865 g/ha per year for tile drainage vs. 176, 439, and 654 g/ha per year for surface runoff. Although only measured in 1 yr, sediment-attached P losses (grams per hectare per year) in tile drainage consistently equaled or exceeded losses in surface runoff, particularly with conservation tillage systems. The authors suggested that extensive cracking in the soil at this site may have enhanced preferential flow of soluble and sediment-bound P into tile drains. Averaged over 3 yr the ortho-P losses via drainage represented 3 and 7% of the total fertilizer P added for the conventional and conservation tillage treatments, respectively.

The role of preferential flow in enhancing P export from fine-textured soils via artificial drainage cited by Gaynor and Findlay (1995) has been noted in other studies. Turtola and Jaakkola (1995) compared surface and subsurface runoff losses of P over a 3-yr period from a heavy clay soil in southwest Finland used for barley (*Hordeum vulgare* L.) and grass ley. Two fertilizer rates were used for each crop (barley = 21 and 42 kg/ha; grass ley = 42 and 84 kg/ha). Most of the total P loss from this soil occurred via surface runoff (56–65%), averaging 0.79 kg/ha per year for barley and 1.27 kg/ha per year for grass ley. In contrast, total P losses in drainage waters averaged 0.42 kg/ha per year for barley and 0.33 kg/hg per

Table 1. Summary of various field studies reporting P losses in subsurface runoff from artificially drained soils.

Location and site information	Summary of P concentration and/or export in drainage waters	Reference
California: Tile drainage from four irrigated cropland sites in the San Joaquin Valley.	Flood irrigation increased P losses that were proportional to fertilizer P rate; percentages of total P applied (fertilizer and irrigation water) lost via drainage were 1, 3, 13, and 17% at the four sites. Total P range over all sites and years was 0.05 to 0.23 mg/L and weighted averaged concentration was 0.079 mg/L.	Johnston et al., 1965
Ontario: Tile drainage from Brookston clay cropped to corn, bluegrass, and rotational crops.	Six-year average total soluble P losses with and without fertilizer P were 0.12 and 0.19 kg/ha/yr. Highest P losses occurred with fertilized, continuous corn (six-year average = 0.29 kg/ha per yr). Concentrations of total soluble P in drainage waters, with and without fertilizer P, averaged 0.18 and 0.21 mg P/L.	Bolton et al., 1970
Vermont: Tile drainage from a Cabot silt loam used for corn and alfalfa rotations.	Ortho-P concentrations in subsurface drain samples from 12 plots located on a sloping (6% slope), poorly drained soil were consistently <0.02 mg/L. Surface runoff concentrations of ortho P averaged 0.8, 0.7, and 0.9 mg/L for alfalfa, corn, and hay-pasture, respectively, and were as high as 3.0 mg/L.	Benoit, 1973
New York: Tile drainage from Lima (Glossoboric Hapludalf) and Kendaia (Aeric Haplaquent) soils.	Neither crop management nor fertilizer rate (12–32 kg P/ha) affected ortho-P concentrations in drainage waters (range: 0.004–0.01 mg/L).	Zwerman et al., 1972
Ontario: Tile drainage from cultivated (vegetables) and uncultivated muckland.	Average soluble P concentrations were 0.019 mg/L for uncultivated marsh vs. 0.031 mg/L for cultivated marsh used for vegetable production. Total P losses were also greater for cultivated (1.56 kg/ha per yr) than uncultivated (0.34 kg/ha per yr) marsh. Most P loss occurred during a spring "pumping period" and 90% of the P lost was reported to be in a soluble, reactive form that was readily available to algae and aquatic plants. Causes of increased P losses in the cultivated marsh were reported to be fertilization and increased oxidation of organic matter due to artificial drainage and pumping of the cultivated marsh that enhance mineralization of organic P.	Nicholls and MacCrimmon, 1974
Iowa: Three-yr study with four tile drainage systems in a Clarion-Webster silt loam soil association used for corn production.	For all years and drainage systems, soluble and total P concentrations ranged from 0 to 0.038 mg/L and from 0.007 to 0.182 mg/L, respectively; only 6 of 477 samples had total P concentrations >0.100 mg P/L. Annual total P losses were estimated to be 0.0003 kg/ha.	Baker et al., 1975
New York: Two-year study of tile drainage systems installed at varying depths in organic soils from two cultivated mucklands.	Molybdate reactive P (MRP) export ranged from 0.9 to 30.7 kg/ha per yr and MRP losses and concentrations were much higher in drainage from deep muck soils (5.4–7.8 mg MRP/L) where the tile line was located in organic material than in sites where tiles were installed in calcareous, mineral subsoils (0.2–1.8 mg MRP/L). Sources of P were mineralization of organic matter (~50 kg/ha per yr) and labile P from long-term fertilization (~40 kg/ha per yr).	Duxbury and Peverly, 1978
Connecticut: Tile drainage from soils cropped to corn, hay, tobacco, and vegetables.	Highest soluble P concentrations in tile drainage waters occurred with vegetable production, ranging from 0.052 to 0.124 mg P/L; the range in soluble P for the other crops was from 0.012 to 0.077 mg P.L.	Sawhney, 1978
Ohio: Ten-year study of nutrient and sediment losses in deep and shallow tile drainage from a silty clay soil (Mollic Haplaquept) cropped in rotation to corn, oat, fallow, alfalfa, and soybean.	Total P losses (water and sediment-bound) averaged 0.8 and 1.3 kg/ha per yr in shallow (0.5 m) and deep (1.0 m) pipe drains vs. 2.2 kg/ha per yr in surface drains. An average of 43% of the annual total P loss in deep drains was as sediment–bound P. Concentrations of total P during a 6-yr period of above average precipitation averaged 0.7 mg/L for deep pipe drains and 0.9 mg/L for surface drains. Total P export averaged 1.6 and 1.9 kg/ha per yr for tile and surface drains, much less than rainfall inputs of 4.6 kg/ha per yr.	Schwab et al., 1980
Indiana: Tile drainage measured for 3 yr from a 17-ha area dominated by Hoytville silty clay soils (Mollic Ochraqualfs) and used for the production of a variety of agronomic crops.	Average concentrations of soluble inorganic P, soluble organic P, and sediment-bound P were 0.04, 0.03, and 0.21 mg P/L, equivalent to losses of 0.035, 0.026, and 0.16 kg/ha per yr. Study showed that solid phase P can sometimes be the major source of P export in subsurface tile drainage as fine-sized sediments leach into tile lines.	Bottcher et al., 1981
Ontario: Six-year study of effects of manure and fertilizer on P losses in tile drainage and surface runoff from a Mountain sandy clay loam (Aquic Eutrochrept) cropped to continuous silage corn.	Two-year average ortho-P concentrations in tile drainage from three springtime flow events were 0.01 mg/L for check plots and plots fertilized at 134 kg/P per ha vs. 0.02 to 0.17 mg/L for manured plots. Winter applied manure resulted in the highest ortho-P concentrations in tile effluent. Surface runoff concentrations of ortho P were always higher than in tile drainage, ranging from 0.12 mg/L (check) to 1.95 mg/L (highest manure rate).	Phillips et al., 1981
Ontario: Tile drainage from two eastern Ontario watersheds dominated by Brandon clay loam soils (Typic Haplaquoll).	Average annual ortho-P concentrations varied little in the 6-yr study (0.01–0.07 mg P/L in one watershed and 0.01–0.12 mg/L in the second). Canadian surface water quality limits for ortho P at that time were 0.065 mg/L. Losses of P were <1% of total annual inputs of P to the watersheds.	Phillips et al., 1982
Ontario: Two studies evaluating crop rotation and P fertilizer rate effects on P losses in tile drainage from a Brookston clay loam soil (Haplaquoll).	*Experiment I*: Sediment P concentrations in tile drainage from continuous corn were twice those from grass sod; however, total dissolved P (TDP) concentrations from bluegrass sod (0.57 mg/L) were sixfold those from corn (0.09 mg/L). Fertilization (30 P/ha per yr) increased concentrations of all forms of P in tile drainage (e.g., TDP in unfertilized plots was ~0.05 mg/L vs. 0.15 [corn] to 1.1 mg/L [bluegrass] for fertilized plots). *Experiment II* showed that increasing tile drain depth from 0.6 to 1.0 m decreased export of sediment P (from 0.47 to 0.18 kg/ha per yr) and TDP (from 0.36–0.14 kg/ha per yr).	Culley et al., 1983
Ontario: Watershed-scale study estimating P losses in tile drainage, soil erosion, and surface runoff, from agricultural cropland dominated by poorly drained Brookston clay soils.	Estimates of P losses from drainage were determined from tile drainage data from experimental plots (0.1 ha) located in the watershed (Culley et al., 1983) and precipitation, stream flow, runoff, and cropping information obtained from within the 51 km^2 watershed. Based on these data, the authors estimated that at least 25% of the total P and 50% of ortho-P export from the watershed came from tile drainage. Conservation strategies to reduce sediment and soluble-P losses via tile drainage were recommended for water quality improvement.	Culley and Bolton, 1983
Florida: Drainage from potato crops grown on a sandy Florida soil with high water table using two methods of irrigation/drainage (subsurface irrigation-tile drains vs. furrow irrigation-ditches).	Average, flow-weighted, ortho-P concentrations were greater in ditch drainage (0.30 mg/L), which contained P from lateral subsurface flow and surface runoff, than in tile drainage (0.05 mg/L). Overall ortho-P losses (surface runoff and tile drainage) were about 60% less with subsurface irrigation and tile drains than from water furrow-irrigation and surface ditches.	Campbell et al., 1985

Table 1 cont.

Table 1. Cont.

Location and site information	Summary of P concentration and/or export in drainage waters	Reference
Florida: Ditch drainage from a Lauderhill muck (Terric Medisaprist) in the Everglades Agricultural Area used for sugarcane production.	No significant differences were detected in total dissolved P or total P concentrations or "off-field loading" of P between drainage waters from sugarcane and uncropped fallow fields. Total P concentrations in drainage waters during 12 drainage events ranged from ~0.2 to 1.4 mg/L and off-field loadings from <0.01 to 0.12 kg/ha per event. Authors reported that P mineralization from organic soils used for sugarcane can be as much as 400% of fertilizer P rate, hence land drainage and oxidation of soil organic matter may be a more important source of P to drainage waters than fertilizer P.	Coale et al., 1994a
Finland: Tile drainage from a heavy clay soil cropped to barley or sown as a timothy-fescue-red clover pasture.	Dissolved P and particulate P (PP) averaged 0.04 and 0.20 mg/L from barley and 0.03 and 0.20 mg/L from pasture before drainage improvement practices were implemented. Improved drainage practices (new tiles, wood chip backfill above drains) slightly decreased DP concentrations in tile drainage waters but tended to increase PP concentrations.	Turtola and Paajanen, 1995
Minnesota: Tile drainage from a Webster clay loam fertilized with dairy manure of urea.	Ortho-phosphate detected in only 1 of 35 samples from manured plots; total P detected in 11 and 7% of samples from manured and urea-fertilized plots (average of detects = 0.04 mg/L). Bray P_1 values in manure and urea treatments were 40 and 26 mg/kg, respectively.	Randall et al., 1996, personal communicaton
Northern Ireland: Ten-year study of changes in soluble reactive P concentrations in tile drainage from a small rural grassland watershed dominated by gleyed, medium to heavy textured soils and used for dairy production.	Soil test P (Olsen P) concentrations in tile drainage from the watershed were estimated to increase by 1 mg/kg per yr due to the P surplus occurring on dairy farms (22–28 kg P/ha per yr). Soluble reactive P (SRP) concentrations in tile drainage (for flows < 50 000 ha/L per d) increased in response to the soil P buildup from 0.020 mg SRP/L in 1981 to 1982 to 0.029 mg SRP/L in 1990 to 1991. Loads of SRP increased from 0.1 to 0.2 g/ha per d over the same time interval. The authors estimated that each 1.0 mg/kg per yr increase in Olsen P resulted in a 0.001 mg/L increase in SRP concentrations in tile drainage.	Smith et al., 1995
England: Three-year study of P losses in surface lateral runoff and field drainage from permanent, grazed grasslands on a Hallsworth clay (stagno-Dystric Gleysol)	Molybdate reactive P (MRP) concentrations in tile drains installed at 85 cm ranged from below detection to 0.27 mg/L. Mean total MRP losses in field drains were from 0.05 to 0.10 kg/ha per yr, compared to 0.11 to 0.22 kg/ha per yr in surface lateral flow from undrained plots. MRP losses were unaffected by N management or grazing strategies but did increase with P fertilizer rate.	Hawkins and Scholefield, 1996
Switzerland: Tile drainage from manured grassland in a watershed dominated by low permeability, gley and brown earth soils, and intensive swine production.	Molybdate reactive P (MRP) concentrations in tile drainage during baseflow and peak flow periods were from 0.050 to 0.150 mg MRP/L and from 0.600 to 1.50 mg MRP/L, respectively. MRP concentrations as high as 4.80 mg/L occurred shortly after manure applications. Losses of MRP ranged from 600 to 1900 g MRP/ha per yr during April to October, much higher than accepted critical losses (400 g/ha per yr). Studies with artificial dyes indicated that preferential flow was an important mechanism for MRP export in tile drainage.	Stamm et al., 1997

year for grass ley, much of which was lost as particulate P (from 63–82% for barley and 38–55% for grass ley). Dissolved ortho-P losses via drainage were 0.11 and 0.17 kg/ha per year. The authors also noted that, for the more heavily fertilized grass ley, ortho-P concentrations in drainage waters increased shortly after P fertilization and speculated that rapid P leaching to tile drains through fissures and macropores may have occurred. For instance, dissolved ortho-P concentrations increased from about 0.1 mg/L to >0.4 mg/L in 1981 and to >0.8 mg/L in 1982 within 1 mo after fertilizer P was applied, gradually declining back to ~0.1 to 0.2 mg P/L within a few months after fertilization.

Additional recent evidence of the effects of long-term fertilization and preferential flow on P loss from fine-textured soils was provided by Heckrath et al. (1995). They measured the concentrations of P in drainage waters from a Batcome silty clay loam (a fine, mixed, mesic Mollic Ochraqualf) at the Broadbalk Continuous Wheat Experiment at Rothamsted Experiment Station in England. Soil test P (Olsen P) concentrations in the plots at this site varied widely (from ~5–110 mg/kg; optimum Olsen P range = 16–46 mg/kg) due to the long-term, differential fertilization and manuring that has occurred in this experiment. Drainage water samples collected in 1992 and 1993 ranged in total P from 0.03 to 0.23 mg/L in a plot that had never received fertilizer or manure P to from 0.55 to 2.75 mg/L in two fertilized plots with very high (>90 mg/kg) Olsen P concentrations. After the installation of new drain lines on these plots in late 1993, total P concentrations in drainage from fertilized plots were much higher, consistently exceeding 1.0 mg/L. The authors speculated that installation of the new drains disturbed the soil and created rapid flowpaths for P movement into drainage waters. Most of the total P in the drainage water from the fertilized plots (78–86%) was dissolved reactive P (DRP). Total P and DRP concentrations were closely related to Olsen P, and the authors identified a change point of ~60 mg Olsen P/kg beyond which P concentrations in drainage waters increased in a near-linear manner with soil test P.

As noted above the situation most commonly associated with extensive P leaching, and thus the increased potential for P loss via subsurface runoff, has been the long-term use of animal manures. Hergert et al. (1981) illustrated how animal waste management can affect P loss by measuring the effects of dairy manure applications on P losses via tile drainage from a Glossoboric Hapludalf in New York cropped to corn, wheat (*Triticum* spp.), and soybean [*Glycine max* (L.) Merr.]. Dairy manure was applied at 35 and 200 wet Mg/ha annually in factorial combination with three application timings (winter-spring plow down, spring-spring plow down, and summer top-dress). The recommended dairy manure rate for silage corn at that time in New York was 40 wet Mg/ha. Before manuring total dissolved P (TDP = ortho-P plus dissolved organic P) in tile drainage was always <0.05 mg/L and usually <0.02 mg/L. In the 2 mo after a spring manure application TDP concentrations increased to ~0.05 to 0.08 mg/L at the 35 Mg/ha rate and to 0.05 to 2.20 mg/L at the 200 Mg/ha rate. Dissolved P concentrations were highest immediately after manure application, declined gradually for several weeks of low tile flow and increased during the next major drainage event. When all data were combined, frequency distributions showed that before and after the application of 35 Mg/ha dairy manure 95 and 82% of the tile effluent samples had TDP concentrations <0.03 mg/L, respectively. At the 200 Mg/ha rate, 93% of the samples were <0.03 mg/L before manuring and 51% after manuring; 16% of the post-manuring samples at this rate had TDP values >0.100 mg/L. Three-year average, flow weighted dissolved ortho-P concentrations in tile effluents were 0.011, 0.014, and 0.218 mg/L for the 0, 35, and 200 Mg/ha dairy manure rates. The authors calculated, based on all application rates and timings, that the probability that dissolved ortho-P would exceed 0.03 mg/L before manuring was <0.06; after the application of 35 and 200 Mg/ha dairy manure the combined probabilities that this would occur were 0.08 and 0.47, respectively. Summer applications had the greatest probability (0.17) to exceed 0.03 mg/L at the 35 Mg/ha rate, while timing of manure application had no effect on this probability (0.45–0.50) at the 200 Mg/ha rate. One important point to note about this study was that the initial soil test P values were in the medium range relative to crop requirements (11 mg/kg Olsen P). After 3 yr of manure applications soil test P was unchanged at the 35 Mg/ha manure rate but increased to 35 mg/kg at the 200 Mg/ha rate. Hence this study illustrates that, in the near-term (3 yr), application of dairy manure at recommended rates will have minor effects on soil and drainage water P relative to excessive manuring. However, the long-term effects of continuous application of dairy manure at recommended, N-based rates, which has been shown in many other studies to markedly increase soil test P in the upper portion of the soil profile (Fig. 2), on P losses in tile drainage remain unresolved.

CURRENT RESEARCH ON PHOSPHORUS LEACHING AND LOSS IN SUBSURFACE RUNOFF IN DELAWARE, INDIANA, AND QUEBEC

Characterizing the impact of P in subsurface runoff on surface water quality and developing management strategies to reduce this impact are challenging tasks. This is clearly illustrated by the studies reviewed above and by ongoing research in Delaware, Indiana, and Quebec, discussed below, which has in common the long-term effects of intensive, animal-based agriculture and artificial drainage on the sorption, mobility, and transport of P to surface waters.

In Delaware, the poultry industry has grown almost fivefold since 1960 while, in the same time period the cropland available to receive the animal wastes from this industry has declined from 325 000 to 230 000 ha. Today, ~270 000 000 broiler chickens (*Gallus domesticus*) are produced annually, with more than 90% of the poultry farms located in Sussex County, Delaware, which has ~100 000 ha of cropland. At currently recommended, N-based application rates (~7 Mg/ha), poultry litter adds 135 kg P/ha per year relative to typical crop removal values for corn of 25 kg/ha per year. Further, estimated annual surpluses of P for a typical, modern Delaware poultry operation (three, 20 000 bird capacity broiler houses that produce five flocks each per year for a total of 300 000 birds per year) located on 100 ha of cropland are ~10 Mg/ha per year[1]. Given this, it is not surprising that most of the agricultural soils in Sussex County, as in other areas of intensive animal production (e.g., North Carolina, Fig. 1b) are now rated as optimum (29%) or excessive (53%) in P (Fig. 1c) and that the potential environmental impact of these high P soils is of great public interest.

The focus of concerns about P losses via subsurface runoff in Delaware has been in the Inland Bays watershed, site of a National Estuary (Rehoboth, Indian River, and Little Assawoman Bays). Phosphorus loss via agricultural drainage is an emerging issue in this watershed not only because of the intensive, animal-based agriculture now practiced there but also because much of the cropland is dominated by poorly drained or very poorly drained soils that are farmed only because of the presence of an extensive network of open drainage ditches. Some of the ditches eventually drain into natural waterways that lead to the Inland Bays or streams and ponds in the watershed, whereas others simply end in the field. The ditches were installed beginning in the 1930s to collect surface runoff and lower water tables that limited agricultural productivity and today form a fairly direct pathway for P transport from agricultural fields to surface waters such as the Inland Bays.

Research in Delaware has focused on quantifying total and dissolved P concentrations in drainage waters and on characterizing the P status and the sorption-desorption of P from soils in agricultural and wooded fields adjacent to drainage ditches. Related studies have evaluated the influence of reduction and re-oxidation on P sorption and desorption from soil horizons with differing physical and chemical properties (Vadas, 1996) and on the release of P from drainage ditch sediments. (Sallade and Sims, 1997a,b). Studies now underway seek to quantify relationships that exist between the degree of P saturation of soil horizons and readily desorbable P (Sims et al., 1996).

Sallade and Sims (1997a) reported that total and dissolved P concentrations in water samples from 17 drainage ditches located on six farms in the Inland Bays watershed ranged from 0.04 to 6.14 mg/L and <0.01 to 0.82 mg/L, respectively. Total P concentrations in all samples and dissolved P concentrations in >40% of the water samples exceeded the eutrophication standard of 0.01 mg/L established for the Inland Bays as did the EPC_0 values for the upper (0–5 cm) sediment layer (EPC_0 = 0.02–0.28 mg P/L). Biologically available P (BAP) in the sediments could be accurately predicted ($R^2 = 0.80^{***}$) from soil test (Mehlich 1) P and organic matter content and sediment P sorption capacity from an equation including the P sorption index (PSI; Mozaffari and Sims, 1994) and sediment organic matter content. In a subsequent laboratory study, Sallade and Sims (1997b) reported soluble P in equilibrium with ditch sediments increased markedly following incubation under anoxic conditions and was highly correlated with sediment DPS ($r = 0.75^{***}$; where DPS = BAP ÷ PSI) and sediment Fe-oxide content ($r = 0.66^{***}$). The authors suggested that P was released from Fe-P forms upon sediment reduction and that a sediment DPS value of 40% could be used to identify sediments with a higher potential to release P into overlying drainage waters.

Vadas and Sims (1998) conducted laboratory studies on the influence of reducing conditions, which can occur in this watershed during spring conditions when water tables are at or near the soil surface, on P solubility and P sorption-desorption in the A, B, and C horizons of two cultivated, poorly drained soils (Fallsington [a fine-loamy, mixed, mesic Typic Ochraquult] and Pocomoke [a coarse-loamy, siliceous, thermic Typic Umbraquult] sandy loams). Anoxic incubation for 28 d created moderately reducing conditions, increased soluble P in all horizons, and decreased P sorption capacity (PSC) from initial values of 488, 508, and 548 mg/kg (Fallsington) and 508, 537, 555 mg/kg (Pocomoke) to, after the 28 d incubation, 279, 359, and 424 mg/kg (Fallsington) and 359, 429, 449 mg/kg (Pocomoke). Together these data suggest that saturated soil conditions, such as commonly occur during the winter-spring recharge period, will increase P solubility and decrease PSC. This can then be expected to increase the potential for

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[1] Annual surpluses estimated by subtracting P outputs from the farm (e.g., harvested crops, animal products) from P inputs (e.g., feed and fertilizer) using a computer spreadsheet program developed at the University of Delaware (J.T. Sims, 1996, unpublished data).

P transport to surface waters as water drains from agricultural cropland via the network of surface ditches in this watershed.

A more comprehensive study integrating the factors controlling P release from these soils via drainage with field-scale hydrologic and water quality investigations is now underway in this watershed (Sims et al., 1996). One goal has been to determine if the DPS approach used in the Netherlands can help identify soils in need of more intensive management to protect surface water quality. Initial results have been positive. For example, studies with 32 surface and subsoil horizons from this watershed found that DPS determined from a single, rapid, oxalate extraction for Al, Fe, and P [DPS-Ox = P-Ox ÷ (Al-Ox + Fe-Ox)] was highly correlated ($r = 0.97^{***}$) with DPS determined from P-Ox divided by PSC estimated from Langmuir isotherms. Further, increases in readily desorbed P, cumulative water soluble P, and Fe-oxide strip P were related to DPS-Ox (Fig. 3a) and, as in the Netherlands, horizons with DPS >25% tended to have higher desorbable P values.

The situation in Indiana, which has been a leader in livestock production in the USA for decades, is similar in many respects to Delaware. Indiana has recently experienced an intensification in swine and poultry production as these industries, mirroring most other regions in the USA, have become more vertically integrated. For example, since the 1970s more than 4.3 million hogs have been in inventory in Indiana, but the number of hog operations has steadily decreased, from 27 000 in 1976 to only 9600 in 1995. Indiana's hog industry has remained competitive because most producers have expanded by gradually building larger, total confinement facilities that have significantly increased pork production efficiency. However, the resultant increases in animal density have also presented new challenges in the collection, storage, and land application of vast quantities of manure. As in Delaware, the intensification of animal production (swine and poultry) has created areas where nutrients produced in animal wastes greatly exceed crop production requirements. Further, to reduce handling costs, many producers have consistently applied manure to fields closest to their manure storage facilities. Even if swine manures are applied at rates that match crop N needs, P applications usually exceed crop P removal by 300 to 500%. For example, a 9400 kg/ha corn crop requires ~200 kg N/ha, which can be provided by 65 450 L of swine manure/ha (assuming a 15% loss of N). This rate of swine manure also provides ~90 kg P/ha of which only 20 kg P/ha will be removed in harvested grain, leaving a residual of ~73 kg P/ha per year. This residual P would raise soil Bray P_1 levels by approximately 8 mg P/kg per year (Vitosh et al., 1973) under an almost ideal N-based manure management scenario. The cumulative amount of P applied to many Indiana soils with long histories of livestock and poultry production has exceeded the P retention capacity of soil surface horizons and increased concerns about P leaching to lower horizons in the soil profile. Because the majority of soils in Indiana are naturally poorly or somewhat poorly drained, most fields have been extensively tile drained. Tile drains have effectively limited the profile depth available for P sorption to the tile depth (approximately 90 cm). Routine soil tests are generally only performed on the surface 0 to 20 cm, so the potential P retention capacity of the soil profile above a tile drain or seasonal high water table is seldom even measured.

Research in Indiana consequently has focused on establishing quantitative relationships between soil test P throughout the soil profile, P sorption capacity, and threshold concentrations of water-soluble P that may be of environmental concern for both tile drainage and surface runoff. One goal has been to determine if soil test P (Bray P_1) can effectively predict the continued release of water-soluble P from soils of widely differing properties. As seen in Fig. 3b and Fig. 3c, Provin et al. (1995) found cumulative water-soluble P in soils and subsoils was well correlated with Bray P_1, suggesting that routine soil tests may be useful tools in predicting the potential for P loss in leaching and drainage waters. Later studies by these authors have shown that incorporating other routinely measured soil properties (e.g., exchangeable Ca) into regression equations with soil test P can improve the ability to predict cumulative water-soluble P from multiple, consecutive extractions.

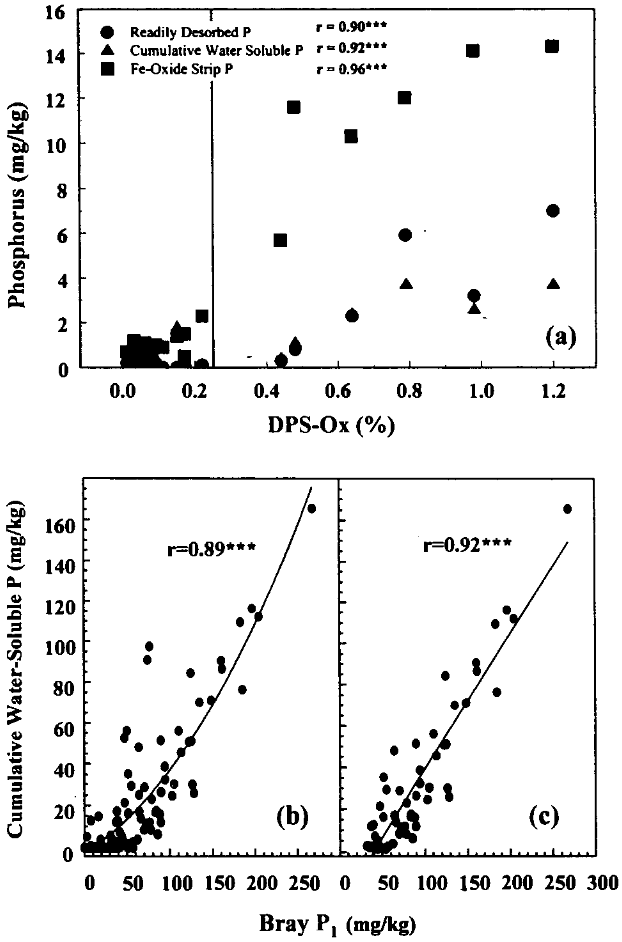

Fig. 3. The relationship between (*a*) degree of P saturation of 32 Delaware soil horizons and several measures of desorbable P (Sims et al., 1996), and (*b*) soil test P (Bray P_1) and cumulative water soluble P for Indiana soil horizons (Provin et al., 1995), and (*c*) soil test P (Bray P_1) and cumulative water soluble P for the same Indiana soil horizons without samples with Bray $P_1 < 4$ mg/kg and pH > 6.8 (Provin et al., 1995).

Quebec has a very diverse combination of land uses, topography, soils, and hydrology and several serious concerns about surface water quality. Eastern Quebec was first settled by Europeans along the waterways and even today most of the population, and most of the concentrated animal-based agriculture, is found along the St. Lawrence River and its tributaries (Patoine, 1995). Similar to Delaware and Indiana, areas in Quebec with intensive animal production have been shown to have soil test P values well above those needed for optimum crop yields (Simard et al., 1995; Tabi et al., 1990). Monitoring of river water quality by the Ministry of Natural Resources in Quebec began in 1967 and results from at least 65 sampling stations per year from 1968 to 1988 indicated that in most rivers total P (TP) concentrations often exceeded the provincial norm of 0.03 mg TP/L (Bobée et al., 1977) and that agriculture was a major cause of the elevated TP levels in river waters (Grimard, 1990). For example, a study of 10 tributaries showed that 16 000 Mg/P per year originated from agricultural sources (Direction de l'Assainissement Agricole. 1988. Contribution des activités agricoles à la pollution de certains tributaires du fleuve Saint-Laurent. Ministère de l'Environnement du Québec, unpublished report). Close correlations between TP and suspended solids in most rivers ($r^2 = 0.77$) suggested that soil erosion was responsible for the elevated TP concentrations. However, more recent studies on fine-textured soils in eastern Canada and elsewhere have suggested that preferential flow through drainage waters contributes a large proportion of the P load to these rivers (Beauchemin et al., 1996; Heckrath et al., 1995; Simoneau, 1996). Preferential flow is believed to play an important role in the movement of dissolved and particulate P to tile drains in the St. Lawrence lowlands because many of the flat, poorly drained soils in this area are also dominated by vermiculitic clays and thus can experience extensive, deep cracking during periods when evapotranspiration exceeds rainfall (Nolin et al., 1991). These cracks may be up to 1 to 2 cm in width and can extend to subsoil horizons where tile drains are located (De Kimpe and Laverdière, 1980; Simard et al., 1989).

Ongoing research in Quebec is evaluating P losses via agricultural drainage from lowland soils and sloping Appalachian soils that are classified as excessively rich in Mehlich 3 P (Simard et al., 1995; Tabi et al., 1990) from long-term fertilization and/or manuring. Many of these studies evaluate the role of preferential flow, through soil cracks and biopores (biological macropores) on P leaching to tile drains. Lowland soils are mainly used for row crop production and, as mentioned above, are vertic in nature; consequently the risk of P leaching to tile drains is particularly high when heavy rains follow extended dry periods. In contrast, forage production dominates the Appalachian region and, since manure is normally spread on the soil surface, losses of P by surface runoff are common. However, because these soils are tilled very infrequently, a well-established network of biopores exists that also increases the risk of P losses via tile drainage.

Recent research on P loss via tile drainage from 27 sites in the St. Lawrence lowlands, dominated by poorly drained clay soils (Humaquepts) confirms these concerns (Beauchemin et al., 1996). Total P concentrations in this 2-yr study (1994–1995) ranged from 0.01 to 1.17 mg/L and exceeded the provincial water quality standard (0.03 mg P/L, Ministère de l'Environnement du Québec, 1993) at 14 sites in 1994 and six sites in 1995. Higher total P concentrations in 1994 than in 1995, and higher percentages of total particulate P (TPP), may have been due to sampling after an extended dry period when soil cracking could have enhanced preferential flow. In 1995, DOP was the dominant form in tile effluents and was correlated with dissolved organic C, consistent with previous research with other soils from this province (Simard et al., 1992). In another Quebec study with lowland soils concentrations of DRP in drainage waters were always larger in ridge tillage than with moldboard plowing and again were much greater in an October sampling that occurred after an extended dry period. Ridge tillage has been reported to enhance the formation of macropores, relative to conventional tillage, and to reduce surface runoff by increasing infiltration rates, both of which could increase P losses by preferential flow (Gaynor and Findlay, 1995). In a third, long-term study with lowland soils, initiated by Bolton et al. (1970) and continued by Culley et al. (1983), greater total dissolved P (TDP) losses were usually found in tile discharge from fertilized soils cropped to forages than to continuous corn (Fig. 4a). Higher TDP concentrations in tile drainage from forages, (e.g., Kentucky bluegrass) are likely due to a better established network of biopores that enhance preferential flow and/or to a greater mineralization of organic P under permanent grass.

Studies on a sloping soil (Coaticook silt loam: a mixed, frigid Typic Humaquept) in the Appalachian region also showed that losses of DRP in tile drainage were greater under forages (0.34 kg/ha per year for timothy clover [*Phleum pratense* L.] and red and white clovers, *Trifolium pratense* L. and

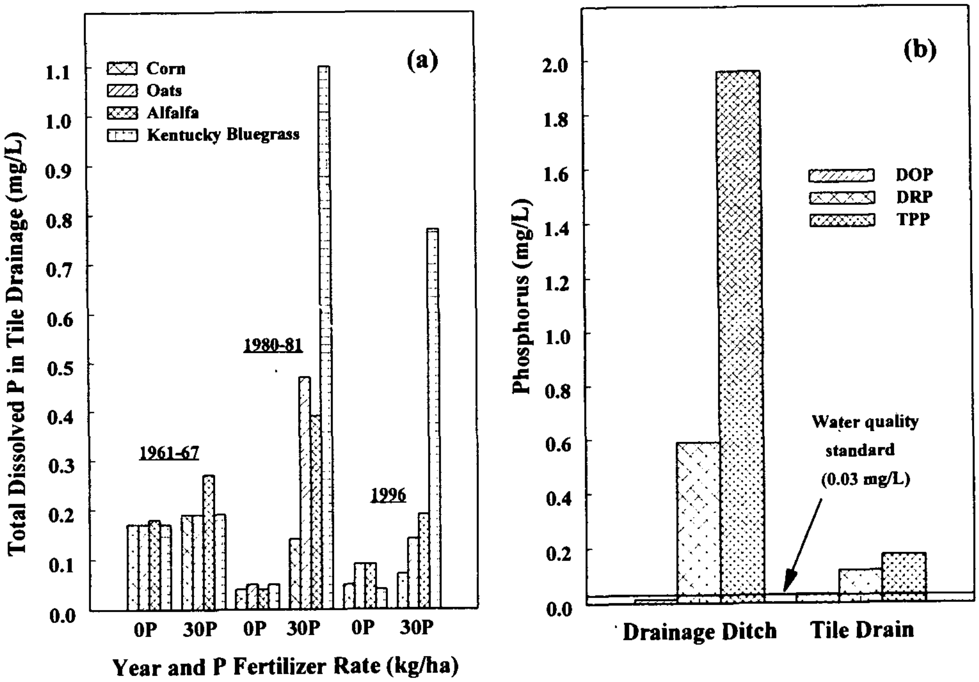

Fig. 4. Canadian studies of (*a*) the influence of crop rotation and P fertilizer rate on total dissolved P concentrations in tile drainage (Culley et al., 1983); and (*b*) forms of P in drainage ditch waters and tile drainage waters from the Boyer River watershed, relative to surface water quality standards (R.R. Simard, 1996, unpublished data).

T. repens L., respectively) than corn (0.27 kg/ha per year). Concentrations of DRP in tile drainage ranged from 0.048 to 0.138 mg/L under corn and from 0.066 to 0.143 mg/L under forages. The similar and very high concentrations of DRP in tile drainage waters under forages, which received lower application rates of P than corn (110 vs. 265 kg/ha per year) tend to support the possible role of macropore flow and/or organic P mineralization in P losses via drainage. In the same study fall applications of liquid hog manure consistently resulted in the greatest losses of TP and DRP. Average losses (over all crops) for inorganic fertilizer, spring manuring, and fall manuring were 0.61, 0.40, and 0.70 kg/ha per year for TP and 0.27, 0.20, and 0.41 kg/ha per year for DRP. A final study in the Appalachian region is underway at 28 sites in the Boyer River watershed where farms with varying animal densities and cropping systems are located on soils with 5 to 15% slopes. Results show that tile drainage and surface runoff both contained very high concentrations of P, averaging 10 and 85 times the current water quality standard (0.03 mg/L), respectively (Fig. 4b). Particulate P was a higher percentage of TP in drainage ditches that also received surface runoff (77%), than in tile drainage (54%).

CONCLUSIONS AND FUTURE DIRECTIONS

Previous and ongoing research clearly indicates that P losses in subsurface runoff can be an important component of the total P export from some agricultural watersheds and should be considered as we develop management strategies to minimize nonpoint source pollution of surface waters. It seems apparent that the most immediate concerns are with those areas where soil P concentrations are already very high (e.g., regions with intensive animal production or heavy P fertilizer use for vegetable crops), soil P sorption capacities are low (sandy soils and high organic matter soils), and subsurface transport is enhanced by artificial drainage systems (tiles and surface ditches). It also seems clear that environmental impacts from P losses in subsurface runoff are not universal problems and that in many situations P leaching and transport to surface waters in tile or ditch drainage will be of little consequence, relative to surface erosion and runoff. Examples include fine-textured soils with low degrees of P saturation that are fertilized in accordance with soil-testing recommendations. In cases such as these, unless soils drain to surface waters highly sensitive to very low P concentrations and loading rates, or extensive preferential flow occurs through soil cracks and biopores, environmentally significant quantities of P are unlikely to be lost in subsurface runoff.

While the above generalizations seem reasonably well supported by past research, many questions remain about the importance of P losses in subsurface runoff to surface water quality. Some relate to fundamental research needs, others to the development of management practices that can reduce P losses via leaching and subsurface flow. We pose the following questions as a starting point for a focused effort that integrates research, management, and policy to protect surface waters from nonpoint source pollution by P in subsurface runoff:

What criteria should be used to identify those fields, farms, or watersheds that are exporting environmentally significant quantities of P via subsurface runoff? Much of the research cited in this paper has compared P concentrations in tile or ditch drainage waters to values commonly associated with eutrophication in streams or lakes or established by governmental policy as surface water quality standards. Given the potential changes that can occur in P concentrations between drain outlets and the surface water of concern (e.g., sorption of P by stream sediments, biological uptake by aquatic plants, dilution of P concentrations by drainage waters from nonagricultural areas in a watershed), there is a need to determine if these are the appropriate standards for agricultural drainage waters. Additionally, many different forms of P have been measured in past research (total P, sediment-P, total particulate P, total dissolved P, dissolved reactive P, dissolved unreactive P, molybdate reactive P, and soluble P) and compared with water quality standards, which themselves are often based on different forms of P (e.g., total P vs. ortho P). For example, should the dissolved P standard of 0.01 mg /L proposed by Vollenweider (1968) for lakes be used as an indication that agricultural drainage waters will impair a nearby surface water? or the USEPA (1986) total P standard of 0.05 mg/L for lakes and 0.100 mg/L for streams? or the groundwater standard of 0.100 mg ortho P/L used in the Netherlands (Breeuswma et al., 1995)? Few of the studies cited in this paper reported dissolved P concentrations in drainage waters <0.01 mg/L and many had total P concentrations in excess of 0.05 mg/L, suggesting that, if standards such as these are used, agricultural drainage waters will often have P concentrations of environmental concern. Clearly there is no single standard for P that is appropriate for all agricultural and hydrologic settings; just as clearly there is a need to develop a more uniform means to develop P-based standards for agricultural drainage waters.

How can we improve our soil testing and characterization methods to prioritize management efforts to reduce the environmental impact of P in subsurface runoff? Soil testing for P has always had an agronomic focus but in recent years there has been an increased emphasis on the development of P soil test methods that can identify soils (or sediments) likely to have an environmental impact on water quality (Sharpley et al., 1996; Sims, 1993). This is an area in need of continued research if we are to develop standardized methods that can relate P quantity and intensity factors (e.g., P sorption capacity and EPC_0, DPS, soil test P, Fe-oxide strip P, readily soluble P) to the desorption and downward movement of P and thus to the potential for P loss in subsurface runoff. Other soil properties that would be useful in this regard include Al and Fe oxides (estimated by oxalate extraction), organic matter and $CaCO_3$ contents, and soil pH. Beyond this, there is a need to incorporate soil chemical analyses with other, perhaps more qualitative parameters, thereby developing simple, field-scale models for the prediction of P losses in subsurface runoff. Examples include soil drainage class (from soil survey manuals), soil texture and soil cracking potential (from soil series and mineralogy), basic hydrologic parameters (depth to tile drains or mean high water table), and cropping system-soil type interactions (e.g., likelihood of biopores as a function of crop rotation and soil type). This follows, but improves and expands upon, the P Index approach described by Lemunyon and Gilbert (1993), which focused exclusively upon erosion and surface runoff as transport pathways for soil P to water.

Is our understanding of the hydrologic pathways operative in subsurface runoff adequate and can these pathways be accurately modeled at the watershed scale? While field-scale assessment tools such as the P Index are valuable first steps in prioritizing management efforts within and between watersheds, there is a pressing need for a better fundamental understanding of the hydrologic pathways of P movement via subsurface runoff. Based on the research cited in this paper it seems likely that at least two hydrologic pathways are operative in the subsurface transport of P from soil to water. The first is the gradual, downward movement of P in percolating waters that interact with the bulk of the soil profile and eventually with tiles or subsurface water flowing laterally and discharging into ditches or streams. The second is bypass flow, the rapid movement of dissolved and particulate P via macropores that extend from the soil surface to tile drains or subsoil horizons where accelerated lateral flow to surface waters occurs. It seems likely that the slow, continual leaching of P by the first process would be of greater concern in soils with extremely high P concentrations from long-term fertilization and manuring (e.g., intensive animal agriculture). Bypass flow of P, however, could occur in any soil where physical or biological macropores exist and would be more or less independent of soil P concentration although it would clearly be of greater environmental significance in high P soils. The two processes are obviously not mutually exclusive and could simultaneously contribute to P loss by subsurface runoff in many situations. Identifying the P transport processes of greatest importance to subsurface runoff in a watershed, and how the relative importance of these processes varies seasonally or in response to changing cultural practices (e.g., tillage, crop rotation, and method/timing of P fertilization/manuring) must be a major research thrust in the future if we are to develop management practices that minimize P losses in agricultural drainage waters.

What remedial measures can be used to reduce P losses via subsurface runoff and how will they impact agriculture and other land uses in a watershed? Should water quality criteria and/or soil P tests and soil hydrologic models indicate that environmentally significant subsurface runoff losses of P are likely, the question of remedial measures must be addressed. This is an area that has received very limited research and one that will be highly problematic for several reasons. Of fundamental importance is the fact that, unlike soil P loss by overland flow (erosion and surface runoff), P exported in subsurface runoff discharges into ditches, canals, or streams from within the soil profile. Further, the nature of water movement in subsurface runoff (timing relative to rainfall events, depth of interaction with the soil, location of discharge into the watercourse) depends upon the type of drainage system that has been installed in a field (e.g., drain depth and spacing, nature of backfill material, number and distribution of feeder and main tile lines). Given this it will be difficult to place barriers between the field and the waterway that can retain sediment bound or soluble P, as has been done with the grassed filter strips used for erosion and runoff control. Some research in the Netherlands has examined the use of Fe-oxide slurry walls installed adjacent to drainage ditches to sorb P from water entering the ditches via shallow lateral flow but, given the extensive and complex nature of most tile systems and drainage ditches in agricultural watershed this seems an unlikely option. Alternatives, then, are (i) preventing P leaching to tile drains to reduce losses via subsurface flow or (ii) removing P from surface ditches after field discharge has occurred. Preventing P leaching will require a combination of practices that maximize crop P uptake, minimize P inputs into the system in excess of crop removal in harvested portions, and the use of soil and crop management practices that minimize preferential flow of P through macropores (e.g., tillage systems that disturb macropores, controlled drainage systems that reduce soil drying). Minimizing P inputs into the system will be difficult in some of the areas with the greatest problems, such as those where intensive animal agriculture is practiced and few, if any, alternatives to land application of animal wastes (and thus P) are currently available. Further, given the high soil P concentrations already present in many soils such as these and the slow rate of P depletion by crop removal (McCollum, 1991) this can, at best, be viewed as a long-term solution. The effectiveness of most soil and crop management practices at reducing P leaching is largely unknown, for the most part untested, and may have other negative effects on P loss. For instance, the conventional tillage systems required to disrupt macropores may increase soil loss by erosion and the reduced conditions associated with controlled drainage practices may solubilize Fe-P compounds increasing dissolved P concentrations in drainage waters. Management practices designed to remove P from drainage ditches, canals, and streams, such as constructed wetlands, removal of sediments or chemical treatment of sediments to fix P in insoluble forms, and "in-stream" biochemical filters that can sorb P have been proposed but rarely tested except in a few, highly impacted areas hence their long-term viability has not been verified (Flaig and Reddy, 1995). Several obstacles exist to engineered treatment systems such as these including: acceptance by the agricultural community and the general public who may have concerns about the effectiveness, reliability, and costs of the systems (i.e., who is responsible for the costs); the need to avoid undesirable effects of the systems on adjacent or downstream ecosystems from changes in hydrology and/or introduction of chemicals into waterways; and the need for a reliable means to assess their long-term effectiveness in water quality improvement. This is an obvious area where multidisciplinary research is needed to develop cost-effective "best management practices" ("BMPs") for the treatment of agricultural drainage waters.

Finally, we must recognize that while soil scientists, hydrologists, and their colleagues in other disciplines have an important role to play in minimizing nonpoint source pollution of surface waters by agricultural P, via subsurface runoff or other pathways, socio-economic and political constraints often prevent or slow the adoption of environmentally oriented agricultural management practices. To avoid this, scientists must work closely with the agricultural community, advisory and regulatory agencies, and public advocacy groups to develop BMPs that are both environmentally effective and feasible in the short and long terms.

REFERENCES

Adriano, D.C., L.T. Novak, A.E. Erickson, A.R. Wolcott, and B.G. Ellis. 1975. Effect of long term land disposal by spray irrigation of food processing wastes on some chemical properties of the soil and subsurface water. J. Environ. Qual. 4:242–248.

Bache, B.W., and E.G. Williams. 1971. A phosphate sorption index for soils. J. Soil Sci. 22:289–301.

Baker, J.L., K.L. Campbell, H.P. Johnson, and J.J. Hanway. 1975. Nitrate, phosphorus, and sulfate in subsurface drainage water. J. Environ. Qual. 4:406–412.

Beauchemin, S., R.R. Simard, and D. Cluis. 1996. Phosphorus sorption-desorption kinetics of soil under contrasting land uses. J. Environ. Qual. 25:1317–1325.

Bengtson, R.L., C.E. Carter, H.F. Morris, and S.A. Bartkiewicz. 1988. The influence of subsurface drainage practices on nitrogen and phosphorus losses in a warm, humid climate. Trans. ASAE 31:729–733.

Benoit, G.R. 1973. Effect of agricultural management of wet sloping soil on nitrate and phosphorus in surface and subsurface water. Am. Geophys. Union 9:1296–1303.

Bobée, B., D. Cluis, M. Goulet, M. Lachance, L. Potvin, and A. Tessier. 1977. Èvaluation du réseau de la qualité des eaux. Analyse et interprétation des données de la période 1967–1975. Service de la qualité des eaux, Ministère des resources naturelles du Québec, Q.E. 20, Québec, Canada.

Bolton, E.F., J.W. Aylesworth, and F.R. Hore. 1970. Nutrient losses through tile drains under three cropping systems and two fertility levels on a Brookston clay soil. J. Soil Sci. 50:275–279.

Bottcher, A.B., E.J. Monke, and L.F. Huggins. 1981. Nutrient and sediment loadings from a subsurface drainage system. Trans. ASAE 24:1221–1226.

Brady, N.C., and R.R. Weil. 1996. Nature and property of soils. 11th ed. MacMillan, New York.

Breeuwsma, A., J.G.A. Reijerink, and O.F. Schoumans. 1995. Impact of manure on accumulation and leaching of phosphate in areas of intensive livestock farming. p. 239–251. *In* K. Steele (ed.) Animal waste and the land-water interface. Lewis Publ.-CRC, New York.

Bryan, O.C. 1933. The accumulation and availability of phosphorus in old citrus grove soils. Soil Sci. 36:245–259.

Calvert, D.V. 1975. Nitrate, phosphate, and potassium movement into drainage lines under three soil management systems. J. Environ. Qual. 4:183–186.

Campbell, C.A., M. Schnitzer, J.W.B. Stewart, V.O. Biederbeck, and F. Selles. 1986. Effect of manure and P fertilizer on properties of a black Chernozem in southern Saskatchewan. Can. J. Soil Sci. 66:601–613.

Campbell, K.L., J.S. Rogers, and D.R. Hensel. 1985. Drainage water quality from potato production. Trans. ASAE 28:1798–1801.

Campbell, L.B., and G.J. Racz. 1975. Organic and inorganic P content, movement, and mineralization of P in soil beneath a feedlot. Can. J. Soil Sci. 55:457–466.

Chang, C., T.G. Sommerfeldt, and T. Entz. 1991. Soil chemistry after eleven annual applications of cattle feedlot manure. J. Environ. Qual. 20:475–480.

Coale, F.J., F.T. Izuno, and A.B. Bottcher. 1994a. Sugarcane production impact on nitrogen and phosphorus in drainage water from an Everglades histosol. J. Environ. Qual. 23:116–120.

Coale, F.J., F.T. Izuno, and A.B. Bottcher. 1994b. Phosphorus in drainage water from sugarcane in the Everglades agricultural area as affected by drainage rate. J. Environ. Qual. 23:121–126.

Cogger, C., and J.M. Duxbury. 1984. Factors affecting phosphorus losses from cultivated organic soils. J. Environ. Qual. 13:111–114.

Cope, J.T., C.E. Evans, and H.C. Williams. 1981. Soil test fertilizer recommendations for Alabama crops. Alabama Agric. Exp. Stn. Circ. 251.

Correl, D.L. 1998. The role of phosphorus in the eutrophication of receiving waters: A review. J. Environ. Qual. 28:261–266 (this issue).

Culley, J.L.B., and E.F. Bolton. 1983. Suspended solids and phosphorus loads from a clay soil. II. Watershed studies. J. Environ. Qual. 12:498–503.

Culley, J.L.B., E.F. Bolton, and V. Bernyk. 1983. Suspended solids and phosphorus loads from a clay soil. I. Plot studies. J. Environ. Qual. 12:493–497.

Daniel, T.C., A.N. Sharpley, D.R. Edwards, R. Wedepohl, and J.L. Lemunyon. 1994. Minimizing surface water eutrophication from agriculture by phosphorus management. J. Soil Water Conserv. 49:30–38.

De Kimpe, C.R., and M.R. Laverdière. 1980. Effet du drainage souterrain sur quelques propriétés de sols argileux du Québec. Can. J. Soil Sci. 60:83–96.

Duxbury, J.M., and J.H. Peverly. 1978. Nitrogen and phosphorus losses from organic soils. J. Environ. Qual. 7:566–570.

Eghball, B., G.D. Binford, and D.D. Baltensperger. 1996. Phosphorus movement and adsorption in a soil receiving long-term manure and fertilizer application. J. Environ. Qual. 25:1339–1343.

Flaig, E.G., and K.R. Reddy. 1995. Fate of phosphorus in the Lake Okeechobee watershed, Florida, USA: Overview and recommendations. Ecol. Eng. 5:127–142.

Fox, R.L., and E.J. Kamprath. 1971. Adsorption and leaching of P in acid organic soils and high organic matter sand. Soil Sci. Soc. Am. Proc. 35:154–156.

Foy, R.H., and P.J.A. Withers. 1995. The contribution of agricultural phosphorus to eutrophication. Proc. no. 365. The Fertilizer Society. Greenhill House, Thorpe Wood, Petersborough, UK.

Gaynor, J.D., and W.I. Findlay. 1995. Soil and phosphorus loss from conservation and conventional tillage in corn production. J. Environ. Qual. 24:734–741.

Grimard, Y. 1990. Qualité générale de l eau au Québec. p. 24–37. *In* Colloque sur la qualité de l eau en milieu agricole. Conseil des Productions Végétales du Québec, Québec, Canada.

Hanway, J.J., and J.M. Laflen. 1974. Plant nutrient losses from tile-outlet terraces. J. Environ. Qual. 3:351–356.

Hawkins, J.M.B., and D. Scholefield. 1996. Molybdate-reactive phosphorus losses in surface and drainage waters from permanent grassland. J. Environ. Qual. 25:727–732.

Heckrath, G., P.C. Brookes, P.R. Poulton, and K.W.T. Goulding. 1995. Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk experiment. J. Environ. Qual. 24:904–910.

Hergert, G.W., S.D. Klausner, D.R. Bouldin, and P.J. Zwerman. 1981. Effects of dairy manure on phosphorus concentrations and losses in tile effluent. J. Environ. Qual. 10:345–349.

Humphreys, F.R., and W.L. Pritchett. 1971. Phosphorus adsorption and movement in some sandy forest soils. Soil Sci. Soc. Am. Proc. 35:495–500.

Izuno, F.T., C.A. Sanchez, F.J. Coale, A.B. Bottcher, and D.B. Jones. 1991. Phosphorus concentrations in drainage water in the Everglades Agricultural Area. J. Environ. Qual. 20:608–619.

James, D.W., J. Kotuby-Amacher, G.L. Anderson, and D.A. Huber. 1996. Phosphorus mobility in calcareous soils under heavy manuring. J. Environ. Qual. 25:770–775.

Johnston, W.R., F. Irrihadieh, R.M. Daum, and A.F. Pillsbury. 1965. Nitrogen and phosphorus in tile drainage effluent. Soil Sci. Soc. Am. Proc. 29:287–289.

Kao, C.W., and R.W. Blanchar. 1973. Distribution and chemistry of phosphorus in an Albaqualf soil after 82 years of phosphate fertilization. J. Environ. Qual. 2:237–240.

King, L.D., J.C. Burns, and P.W. Westerman. 1990. Long-term swine lagoon effluent applications on Coastal bermuda grass: II. Effect on nutrient accumulation in soil. J. Environ. Qual. 19:756–760.

Kingery, W.L., C.W. Wood, D.P. Delaney, J.C. Williams, and G.L. Mullins. 1994. Impact of long-term land application of broiler litter on environmentally related soil properties. J. Environ. Qual. 23: 139–147.

Knudsen, D., and D.B. Beegle. 1996. Recommended phosphorus tests. *In* W.C. Dahnke (ed.) Recommended chemical soil test procedures for the North Central region. Bull. 499 (Rev.) Univ. of Missouri, Columbia.

Kuo, S., and A.S. Baker. 1982. The effect of soil drainage on phosphorus status and availability to corn in long-term manure-amended soils. Soil Sci. Soc. Am. J. 46:744–747.

Larsen, J.E., R. Langston, and G.F. Warren. 1958. Studies on the leaching of applied labeled phosphorus in organic soils. Soil Sci. Soc. Am. Proc. 22:558–560.

Lemunyon, J.L., and R.G. Gilbert. 1993. Concept and need for a phosphorus assessment tool. J. Prod. Agric. 6:483–486.

Logan, T.J. 1982. Mechanisms for release of sediment-bound phosphate to water and the effects of agricultural land management on fluvial transport of particulate and dissolved phosphate. Hydrobiology 92:519–530.

Lookman, R., K. Jansen, R. Merckx, and K. Vlassak. 1996. Relationship between soil properties and phosphate saturation parameters, a transect study in northern Belgium. Geoderma 69:265–274.

Lookman, R., N. Vandeweert, R. Merckx, and K. Vlassak. 1995. Geostatistical assessment of the regional distribution of phosphate sorption capacity parameters (Feox and Alox) in northern Belgium. Geoderma 66:285–296.

McCollum, R.E. 1991. Buildup and decline in soil phosphorus: 30-year trend on a Typic Umbraquult. Agron. J. 83:77–85.

Meek, B.D., L.E. Graham, and T. Donovan. 1982. Long-term effects of manure on soil nitrogen, phosphorus, potassium, sodium, organic matter, and infiltration. Soil Sci. Soc. Am. J. 46:1014–1019.

Meek, B.D, L.E. Graham, T.J. Donovan, and K.S. Mayberry. 1979. Phosphorus availability in a calcareous soil after high loading rates of animal manure. Soil Sci. Soc. Am. J. 43:741–744.

Miller, M.H. 1979. Contribution of nitrogen and phosphorus to subsurface drainage water from intensively cropped mineral and organic soils in Ontario. J. Environ. Qual. 8:42–48.

Ministère de l Environnement du Québec. 1993. Critères de qualité de l eau. Service d évaluation des rejets toxiques et Direction de la qualité de l eau. MENVIQ Envirodoq EN930144, Quebec, Canada.

Mozaffari, M., and J.T. Sims. 1994. Phosphorus availability and sorption in an Atlantic Coastal Plain watershed dominated by animal-based agriculture. Soil Sci. 157:97–107.

Mozaffari, M., and J.T. Sims. 1996. Phosphorus transformations in poultry litter-amended soils of the Atlantic Coastal Plain. J. Environ. Qual. 25:1357–1365.

Neller, J.R. 1946. Mobility of phosphates in sandy soils. Soil Sci. Soc. Am. Proc. 11:227–230.

Nicholls, K.H., and H.R. MacCrimmon. 1974. Nutrients in subsurface and runoff waters of the Holland Marsh, Ontario. J. Environ. Qual. 3:31–35.

Nolin, M.C., L. Lamontagne, and A. Martin. 1991. Esquisse pédologique du Sud-Est de la plaine de Montréal. Agrosol 4 (2):44–57.

Patoine, M. 1995. Perspectives et approche du Québec pour faire face à la problématique environmentale associée à l'intensification et la concentration des élevages. p. 45–62. *In* Siting livestock and poultry operations in the 21st century symposium, Ottawa, Canada. 13–14 July, 1995.

Phillips, P.A., J.L.B. Culley, F.R. Hore, and N.K. Patni. 1981. Pollution potential and corn yields from selected rates and timing of liquid manure applications. Trans. ASAE 24:139–144.

Phillips, P.A., J.L.B. Culley, F.R. Hore, and N.K. Patni. 1982. Dissolved inorganic nitrogen and phosphate concentrations in discharge from two agricultural catchments in eastern Ontario. Agric. Waste Manage. 5:29–40.

Potash and Phosphate Institute. 1994. Soil test summaries: Phosphorus, potassium, and pH. Better Crops Plant Food 78:14–17.

Provin, T.L., B.C. Joern, D.P. Franzmeir, and A.L. Sutton. 1995. Phosphorus retention in selected Indiana soils using short-term sorption isotherms and long-term aerobic incubations. p. 35–42. *In* K. Steele (ed.) Animal waste and the land-water interface. Lewis Publ.-CRC, New York.

Reddy, K.R., M.R. Overcash, R. Khaleel, and P.W. Westerman. 1980. Phosphorus adsorption-desorption characteristics of soils used for disposal of animal manures. J. Environ. Qual. 9:86–92.

Ryden, J.C., J.K. Syers, and R.F. Harris. 1973. Phosphorus in runoff and streams. Adv. Agron. 25:1–45.

Sallade, Y.E., and J.T. Sims. 1997a. Phosphorus transformations in the sediments of Delaware's agricultural drainageways: I. Phosphorus forms and sorption. J. Environ. Qual. 26:1571–1579.

Sallade, Y.E., and J.T. Sims. 1997b. Phosphorus transformations in the sediments of Delaware's agricultural drainageways: II. Effect of reducing conditions on phosphorus release. J. Environ. Qual. 26:1579–1588.

Sawhney, B.L. 1978. Leaching of phosphate from agricultural soils to groundwater. Water Air Soil Pollut. 9:499–505.

Sawyer, C.N. 1947. Fertilization of lakes by agricultural and urban drainage. New Engl. Water Works Assoc. J. 61:109–127.

Schwab, A.P., and S. Kulyingyong. 1989. Changes in phosphate activities and availability indexes with depth after 40 years of fertilization. Soil Sci. 147:179–186.

Schwab, G.O., N.R. Fausey, and D.E. Kopcak. 1980. Sediment and chemical content of agricultural drainage water. Trans. ASAE 23: 1446–1449.

Sharpley, A.N. 1995. Identifying sites vulnerable to phosphorus loss on agricultural runoff. J. Environ. Qual. 24:947–951.

Sharpley, A.N., S.C. Chapra, R. Wedepohl, J.T. Sims, T.C. Daniel, and K.R. Reddy. 1994. Managing agricultural phosphorus for protection of surface waters: Issues and options. J. Environ. Qual. 23:437–451.

Sharpley, A.N., T.C. Daniel, J.T. Sims, and D.H. Pote. 1996. Determining environmentally sound soil phosphorus levels. J. Soil Water Conserv. 51:160–166.

Sharpley, A.N., and R.G. Menzel. 1987. The impact of soil and fertilizer phosphorus on the environment. Adv. Agron. 41:297–324.

Sharpley, A.N., S.J. Smith, and W.R. Bain. 1993. Nitrogen and phosphorus fate from long-term poultry litter applications to Oklahoma soils. Soil Sci. Soc. Am. J. 57:1131–1137.

Sharpley, A.N., S.J. Smith, and J.W. Naney. 1987. Environmental impact of agricultural nitrogen and phosphorus use. J. Agric. Food Chem. 35:812–817.

Sharpley, A.N., S.J. Smith, B.A. Stewart, and A.C. Mathers. 1984. Forms of phosphorus in soil receiving cattle feedlot waste. J. Environ. Qual. 13:211–215.

Simard, R.R., D. Cluis, G. Gangbazo, and S. Beauchemin. 1995. Phosphorus status of forest and agricultural soils from a watershed of high animal density. J. Environ. Qual. 24:1010–1017.

Simard, R.R., C.R. De Kimpe, and J. Zizka. 1989. The kinetics of non-exchangeable potassium and magnesium release from Québec soils. Can. J. Soil Sci. 69:663–675.

Simard, R.R., M.R. Laverdière, D.A. Angers, and N. Bissonnette. 1992. The influence of cropping and tillage practices on P movement in a clay soil profile. p. 728–729. *In* A. Cottenie (ed.) Phosphorus, life and environment. Proc. 4th Int. IMPHOS Conf., Ghent, Belgium. 8–11 Sept. 1992.

Simoneau, M. 1996. Qualité des eaux du bassin de la rivière Châteauguay, 1979 à 1994. Direction des écosystèmes aquatiques. Ministère de l Environnement et de la Faune du Québec.

Sims, J.T. 1993. Environmental soil testing for phosphorus. J. Prod. Agric. 6:501–507.

Sims, J.T. 1997. Animal waste management: Agricultural and environmental issues in the United States and western Europe. p. 7–27. *In* C. Chang (ed.) Animal waste management: National and international perspectives. Can. Soc. Soil Sci., Lethbridge, AB, Canada.

Sims, J.T., A.S. Andres, J.M. Denver, and W.J. Gangloff. 1996. Assessing the impact of agricultural drainage on ground and surface water quality in Delaware. Final Project Report. Delaware Dep. Nat. Res. Environ. Control, Dover.

Sims, J.T., and K.L. Gartley. 1996. Nutrient management handbook for Delaware. Coop. Bull. 59. Univ. of Delaware, Newark.

Smith, R.V., S.D. Lennox, C. Jordan, R.H. Foy, and E. McHale. 1995. Increase in soluble phosphorus transported in drainflow from a grassland catchment in response to soil phosphorus accumulation. Soil Use Manage. 11:204–209.

Sommers, L.E., D.W. Nelson, and L.B. Owens. 1979. Status of inorganic phosphorus in soils irrigated with municipal wastewater. Soil Sci. 127:340–350.

Sonzogni, W.C., S.C. Chapra, D.E. Armstrong, and T.J. Logan. 1982. Bio-availability of phosphorus inputs to lakes. J. Environ. Qual. 11:555–563.

Spencer, W.F. 1957. Distribution and availability of phosphates added to a Lakeland fine sand. Soil Sci. Soc. Am. Proc. 21:141–144.

Stamm, C., R. Gachter, H. Fluhler, J. Leuenberger, and H. Wunderli. 1997. Phosphorus input to a brook through tile-drains under grassland. p. 372–373. *In* H. Tunney et al. (ed.) Phosphorus losses from soil to water. CAB Int., Wallingford, UK.

Sugiharto, T., T.H. McIntosh, R.C. Uhrig, and J.J. Lardinois. 1994. Modeling alternatives to reduce dairy farm and watershed nonpoint source pollution. J. Environ. Qual. 23:18–24.

Sylvester, R.O. 1961. U.S. Public Health Serv. Publ. SEC-TR-W61-3. p. 80–87.

Tabi, M., L. Tardif, D. Carrier, G. Laflamme, and M. Rompré. 1990. Inventaire des problèmes de dégradation des sols agricoles du Québec. Rapport synthèse. MAPAQ and Agriculture Canada, Québec, Canada.

Taylor, A.W., and V.J. Kilmer. 1980. Agricultural phosphorus in the environment. p. 543–557. *In* F.E. Khawsaneh et al. (ed.) The role of phosphorus in agriculture. ASA, CSSA, and SSSA, Madison, WI.

Taylor, A.W., and H.M. Kunishi. 1971. Phosphate equilibria on stream sediment and soil in a watershed draining an agricultural region. J. Agric. Food Chem. 19:827–831.

Turtola, E., and A. Jaakkola. 1995. Loss of phosphorus by surface runoff and leaching from a heavy clay soil under barley and grass ley in Finland. Soil Plant Sci. 45:159–165.

Turtola, E., and A. Paajanen. 1995. Influence of improved subsurface drainage on phosphorus losses and nitrogen leaching from a heavy clay soil. Agric. Water Manage. 28:295–310.

U.S. Environmental Protection Agency. 1986. Quality criteria for water. Office of Water Regulation and Standards. USEPA Rep. 440/5-86-001. U.S. Gov. Print. Office, Washington, DC.

U.S. Environmental Protection Agency. 1994. 1992 Rep. to Congress. Office of Water, National Water Quality Inventory. USEPA Rep. 841R94001. U.S. Gov. Print. Office, Washington, DC.

Vadas, P.A., and J.T. Sims. 1998. Effect of reduction, re-oxidation, and poultry litter on phosphorus solubility in Atlantic Coastal Plain Soils. Soil Sci. Soc. Am. J. 62:(in press).

Vitosh, M.L., J.F. Davis, and B.D. Knezek. 1973. Long-term effects of manure, fertilizer, and plow depth on chemical properties of soils and nutrient movement in a monoculture corn system. J. Environ. Qual. 2:296–299.

Vollenweider, R.A. 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. Pub. No. DAS/SAI/68.27. Organ. Econ. Coop. Dev., Paris, France.

Zwerman, P.J., T. Greweling, S.D. Klausner, and D.J. Lathwell. 1972. Nitrogen and phosphorus content of water from tile drains at two levels of management and fertilization. Soil Sci. Soc. Am. Proc. 36:134–137.