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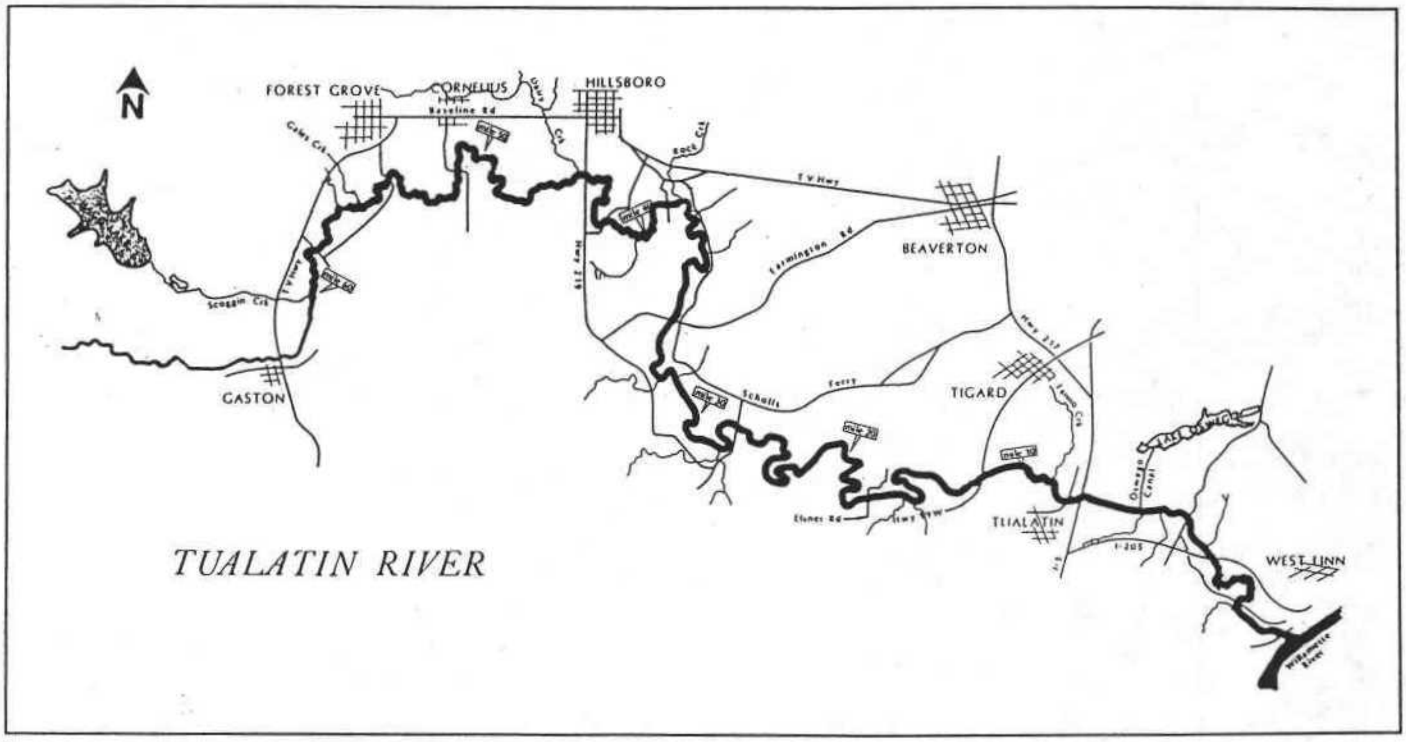
A Literature Review

Land Use and Nonpoint Phosphorus Pollution in the Tualatin Basin, Oregon



Oregon State University Extension Service
Oregon Water Resources Research Institute

TUALATIN RIVER BASIN SPECIAL REPORTS

The Tualatin River Basin in Washington County, Oregon, is a complex area with highly developed agricultural, forestry, industrial, commercial, and residential activities. Population has grown in the past thirty years from fifty to over 270 thousand. Accompanying this population growth have been the associated increases in transportation, construction, and recreational activities. Major improvements have occurred in treatment of wastewater discharges from communities and industries in the area. A surface water runoff management plan is in operation. Agricultural and forestry operations have adopted practices designed to reduce water quality impacts. In spite of efforts to-date, the standards required to protect appropriate beneficial uses of water have not been met in the slow-moving river.

The Oregon Department of Environmental Quality awarded a grant in 1992 to the Oregon Water Resources Research Institute (WRRI) at Oregon State University to review existing information on the Tualatin, organize that information so that it can be readily evaluated, develop a method to examine effectiveness, costs and benefits of alternative pollution abatement strategies, and allow for the evaluation of various scenarios proposed for water management in the Tualatin Basin. Faculty members from eight departments at Oregon State University and Portland State University are contributing to the project. Many local interests groups, industry, state and federal agencies are contributing to the understanding of water quality issues in the basin. This WRRI project is based on all these research, planning and management studies.

This publication is one in a series designed to make the results of this project available to interested persons and to promote useful discussions on issues and solutions. You are invited to share your insights and comments on these publications and on the process in which we are engaged. This will aid us in moving towards a better understanding of the complex relationships between people's needs, the natural environment in which they and their children will live, and the decisions that will be made on resource management.

INTRODUCTION

Background

Water quality is of increasing concern to society. Supplies of water are for all practical purposes static while the demand for high quality water for industry, agriculture, population and recreation steadily increases. Currently Oregon's Tualatin River is the center of a controversy relating to this concern for water that is clean and useable for a variety of purposes.

The Tualatin River originates in the Oregon Coast Range and runs east to join the Willamette River 40 miles away. Along its way it meanders through roughly 86 miles of channel and drains 711 square miles of land with varied topography and use (Carter, 1975). For most of its last 40 miles, the channel has little drop in elevation, giving it a slow moving, almost lake-like, character during the summer low flow period. During the summer this stretch may experience periods of eutrophication (Castle, 1991).

Although the quality of water in the Tualatin River has been of concern to some for a long time, concern was focused in 1986 when the Northwest Environmental Defense Center filed suit against the Environmental Protection Agency (EPA) (Castle, 1991; Cleland, 1991). The suit sought to force the adoption and enforcement of pollutant limits for Oregon streams in general and the Tualatin specifically. It was decided that these limits, called Total Maximum Daily Loads or TMDL, should have more local input than the federal government could provide, so the task was passed to the Oregon Department of Environmental Quality (DEQ) (Castle, 1991).

In 1987 DEQ conducted a statewide assessment of nonpoint source (NPS) pollution problems. As a result of this assessment, the Tualatin River Basin became the DEQ's priority surface water concern. The Tualatin was defined as "water quality limited," a designation that has specific meaning in relation to practices required to reduce pollutants (Cleland, 1991; Soil Conservation Service, 1990). For the Tualatin, studies showed that both ammonia and phosphorus (**P**) were factors limiting water quality, but **P** was considered to be the key limiting factor and a stringent TMDL, 0.07 mg/l (70ug/l) of total **P**, was set for the Tualatin River Basin. It is estimated that of the total **P** contributed to the Tualatin on a yearly basis, 85% is from point sources such as sewage treatment plants and 15% is from nonpoint sources (Castle, 1991). As a water quality limited basin, all sources that contribute to the problem are responsible for bringing the problem under control.

Of the subbasins comprising the Tualatin drainage, the Dairy-McKay subbasin (or hydrologic unit) was identified as a major contributor to the water quality problems of the Tualatin River. This hydrologic unit contains only about one-third of the total area (256 square miles) of the Tualatin Basin, but about half of the forested land and half of the agricultural land. These land uses contribute sediments and sediment-related nutrients to surface waters, with about 60% of agricultural lands eroding at three times the rate considered acceptable by the Soil Conservation Service (Soil Conservation Service, 1990). On Dairy and McKay creeks the TMDL for **P** is frequently violated upstream of any known point sources, indicating that a portion of the problem is from nonpoint sources (Soil Conservation Service, 1990). This has focused attention on all land management activities in the basin.

Objectives

This paper has two purposes:

1. A review of relevant literature to explore possible connections between land use and the sources and transport of NPS phosphorus in the Tualatin Basin.
2. If these connections exist, identify in the literature how these practices might be modified to reduce contributions of NPS phosphorus to the problem.

LITERATURE REVIEW

The Nature of Nonpoint Source Pollution

Point source pollutants are those contaminants that enter the environment from an identifiable source (Chesters and Schierow, 1985). Point sources are often responsible for pollutants that are concentrated at the point of entry but disperse as they move farther away from the source.

Nonpoint source (NPS) pollutants are those contaminants that enter the environment from diffuse sources (Chesters and Schierow, 1985). In contrast to point source pollutants, which lessen in concentration after they enter the environment, nonpoint pollutants may not be recognized as a problem until they become more concentrated by wind or water patterns.

While these definitions may seem concise, in practice the boundaries between point and nonpoint pollution are not always clear. The smokestack of a factory is an identifiable point source pollutant of air. Yet as pollutants from the smoke disperse, settle to the ground, and are carried into watercourses, they become nonpoint source pollutants of water. The identification of pollutants as point or nonpoint source includes not only the source, but the way pollutants are delivered to the affected resource.

The nature of NPS pollutants makes them difficult to identify and control. These pollutants may be from airborne or land-based sources (Chesters and Schierow, 1985; Loehr, 1974). Airborne pollutants may originate from point sources, such as the exhaust pipe of a vehicle, but are spread and deposited over wide areas (Chesters and Schierow, 1985).

Land-based pollutants are generally from sources that are diffuse. The assortment of sediments, nutrients, and chemicals carried with urban runoff; sediments from agriculture, silviculture, construction, or mining; toxic metals from mining; nutrients from croplands or silviculture; and animal wastes from livestock are examples of diffuse land-based pollutants (Chesters and Schierow, 1985; Loehr, 1974; Myers et al., 1985).

The majority of NPS pollutants come from either agricultural, mining, urban, construction, or silvicultural sources (Myers et al., 1985). These nonpoint sources are estimated to be responsible for half or more of all nitrogen, coliform bacteria, iron, phosphorus, oil, zinc, lead, chromium, and copper contributed to surface waters of the United States annually (Chesters and Schierow, 1985). Sediments from nonpoint sources are responsible for an estimated \$6 billion in damage per year (Clark, 1985). NPS phosphorus contributions are large compared to other sources. Agriculture alone is responsible for five times as much **P** that enters surface waters as is contributed by all point sources (Duda and Johnson, 1985). Although NPS phosphorus is contributed by many sources including construction, urban runoff, and silviculture, an estimated 60 percent of the total annual contribution is from agriculture (Chesters and Schierow, 1985).

Phosphorus and Aquatic Environments

Much of the literature on **P** from agricultural, soil science and geological sources is concerned with the scarcity of **P** and the problems of delivering it in the amounts necessary for vigorous plant growth rather than with **P** as a pollutant. Generally, terrestrial concentrations of **P** are low, and where abundant it is a benefit not a detriment. Aquatic researchers generally considered concentrations to be too insignificant to be of interest until the early 1960s (Griffith, 1973).

Much as a weed is a plant in the wrong place, **P** as a pollutant is a nutrient in the wrong place. Phosphorus is an element essential for life that occurs naturally in all living cells and every ecosystem (Griffith, 1973; Hooper, 1973; Kilmer and Taylor, 1980; McKelvey, 1973).

The problems **P** poses are not from its presence, but from the relative amounts in which it is present (Kilmer and Taylor, 1980; Sharpley and Menzel, 1987). Phosphorus and nitrogen are the two elements that are the most critical to the growth of plants (Brady, 1974). Although limiting to plants in general, **P** is the single most limiting factor for aquatic plant growth (Nelson and Logan, 1983; Sharpley and Menzel, 1987; Syers et al., 1973). Large inputs of other nutrients have relatively little effect on algal growth unless **P** is available (Nelson and Logan, 1983). Amounts lost from soils through erosion or leaching that are insignificant to the total soil content, or to terrestrial plants, may cause eutrophication when added to aquatic systems because of the low **P** levels those systems evolved with (Kilmer and Taylor, 1980; Sharpley and Menzel, 1987).

While increased plant growth might seem a desirable goal, in aquatic environments the effects of that growth can be far from desirable. Although slightly increased water fertility can have beneficial effects on fisheries, most aquatic ecosystems evolved with low nutrient levels (Smith, 1959; Verduin, 1970). A shift to higher nutrient levels, especially phosphorus, affects primary productivity and causes aquatic plants, in general, and algae in particular, to reproduce and grow at high rates (Forster et al., 1985; Nelson and Logan, 1983; MacKenthun, 1973a; Odum, 1975; Sawyer, 1973; Syers et al., 1973; Thomas, 1973; Verduin, 1970). These rapid growth conditions can have extensive effects on aquatic systems. An increase of surface plants reduces light penetration which in turn reduces or eliminates plant growth at greater depths. This, combined with competition for other essential needs and the relatively short life span of many of the individual plants, produces a large amount of dead organic material which increases the biochemical oxygen demand (BOD) for decay processes (MacKenthun, 1973a; Odum, 1975; Sawyer, 1973; Verduin, 1970).

This demand for oxygen, along with the rotting vegetation, can result in bad tastes in drinking water even after passing through water purification systems, and foul smells in water used for recreation or drinking (Odum, 1975; Verduin, 1970). During periods of high photosynthetic production, the withdrawal of CO_2 for plant respiration may cause the water to become more alkaline (Boers, 1991; MacKenthun, 1973a). The change in pH and oxygen depletion from increased BOD may result in the death of other aquatic organisms or a change in the species composition of the aquatic community (O'Kelley, 1973; Sawyer, 1973; Verduin, 1970).

This process, known as eutrophication, may affect the potential uses of water. Eutrophic waters may be unsuitable for boating, fishing, swimming, drinking, irrigation, or other uses (Armstrong and Rohlich, 1973; MacKenthun and Ingram, 1967; Odum, 1975). Although other nutrients may play a role in eutrophication, it is **P** that is considered to be the nutrient most amenable to control, because it is generally the most limiting nutrient and because measures to control **P** are more likely to succeed than for other nutrients (Cahill et al., 1974; Syers et al., 1973; Thomas, 1973). In general, lakes and streams that are relatively undisturbed by human activities have relatively low levels of **P** that are insufficient for the rapid algal growth characteristic of eutrophication (Thomas, 1973; Verduin, 1970).

The other element that is commonly limiting to aquatic systems, nitrogen, may be delivered to aquatic systems in large quantities by rainwater, or by the ability of bacteria and blue-green algae to fix atmospheric nitrogen (Thomas, 1973). In contrast to nitrogen, **P** does not have a gaseous phase in its biogeochemical cycle and once present, will be stored in aquatic systems unless physically removed (Keup, 1968; Thomas, 1973).

In surface waters the majority of **P** is held in a variety of organic storage pools (Hooper, 1973; MacKenthun, 1973a). These storage pools include the living aquatic community of plants and animals that are either free-swimming or associated with the bottom, organic particulate material, dissolved organic compounds, and the **P** contained in bottom sediments (Hooper, 1973; MacKenthun, 1973a; MacKenthun and Ingram, 1967). In relatively undisturbed lakes the portion contained in organic particulate material, living and dead, may be greater than 60% of all **P** in the surface water ecosystem (Hooper, 1973). The amount contained in each of these storage pools changes rapidly in response to alterations in physical conditions or nutrient inputs (Hooper, 1973). This rapid cycling between storage pools gives **P** in aquatic systems a transient character, with availability varying in response to the interactions of biological, chemical, and physical factors (Hooper, 1973; MacKenthun, 1973a). Uptake or release of phosphorus from one storage pool may greatly affect the availability of **P** to others, such as the estimate that on a daily basis, the waste products and dead cells of lake zooplankton may provide 40% to 70% of the phosphorus needed by phytoplankton in a noneutrophic lake (Golterman, 1973).

As the rapid growth conditions that accompany eutrophication begin to change the patterns of **P** cycling, changes in species composition take place (O'Kelley, 1973; Sawyer, 1973; Verduin, 1970). Some species of algae and bacteria have a greater positive response to increased phosphorus availability and their growth is greatly favored by these conditions (O'Kelley, 1973). When **P** is present in quantities greater than that needed for the immediate growth needs, many algae and bacteria may take up more than is needed for current growth (luxury uptake), excreting that above their storage capacity in a highly bioavailable form, or holding it until supplies are deficient or death releases it to another storage pool (Boberg and Persson, 1988; Hooper, 1973; O'Kelley, 1973; Sawyer, 1973).

In addition to the large amounts of **P**, relative to phytoplanton needs, that may be released by the decay of algae produced during a "bloom," the withdrawal of CO_2 for plant respiration may cause the pH of the water to rise to values of 10 or more (Boers, 1991; MacKenthun, 1973a). The availability of **P** from sediments is significantly increased for pH values above 9.5 indicating that eutrophic conditions may increase the availability of **P** for further plant growth (Boers, 1991).

The harmful effects of **P** are only those indirect effects associated with increased water fertility. Phosphorus as a nutrient is not toxic in any way (Griffith, 1973; Kilmer and Taylor, 1980; Sharpley and Menzel, 1987).

Sources of Phosphorus

The quantity of **P** in streams and lakes is a product of complex relationships involving geologic processes, atmospheric inputs, biological and chemical processes, hydrology, topography, vegetation, land use and management (Ellis et al. 1989; Keup, 1968; Loehr, 1974). A discussion of those sources most responsible follows.

Geological Sources

Phosphorus is the eleventh most abundant element in the earth's crust (Holtan et al., 1988; McKelvey, 1973). It is widely distributed, being a constituent of most rocks, but is classified as a trace element since it comprises only about 0.1% of the rocks of the earth's crust (McKelvey, 1973). It occurs naturally in more than 200 phosphate compounds, although most, perhaps 95%, occurs as fluorapatite (Boberg and Persson, 1988; Fisher, 1973; Holtan et al., 1988; McKelvey, 1973).

Igneous rocks have highly differing phosphate contents, even when considered in terms of major groups (McKelvey, 1973). The content of apatite in igneous and metamorphic rocks can be as high as 18%, but is generally less than 12% (Boberg and Persson, 1988; Holtan et al., 1988). Sedimentary rocks are derived from the weathering products of igneous rocks, and because of the diversity of combinations of elements, including the remnants of biological processes, have even more diverse **P** contents (McKelvey, 1973). The phosphate content of rocks may be high in a local area, but generally the content is relatively low, less than 0.2% (McKelvey, 1973).

The local availability of **P** from geologic sources is largely influenced by three factors. First, the phosphate content of the rocks in the local area. Rocks with a content much higher than average may occur over large areas and still be a negligible part of the total content of the earth's crust (McKelvey, 1973). This is especially significant to basaltic volcanic rocks (McKelvey, 1973). Secondly, the forms of phosphate found in rocks vary widely in their availability to plants (Dillon and Kirchner, 1975; McKelvey, 1973). Lastly, the environment of the local area--its climate, background pH, and the presence of other minerals which may tend to fix or liberate **P**--will affect the availability of **P** from geologic sources (Golterman et al., 1983; McKelvey, 1973).

In any case, in contrast to many other nutrient cycles, the lack of a gaseous component in the interactions in the phosphorus cycle provides **P** with a one-way trip from rocks, through waterways to the sea (Holtan et al., 1988, Keup, 1968; McKelvey, 1973). It will be millions of years before geologic processes expose the deposits of phosphorus currently contained in ocean sediments to the weathering processes that can begin the cycle again (Holtan et al., 1988).

<u>Soils</u>

As a product of geologic parent material soils naturally contain **P** compounds although the amounts vary widely (Brady, 1974). Those soils formed from igneous rock generally have the highest **P** content (Bailey, 1968). Generally soils that are well-drained also have higher **P** levels (Bailey, 1968). However, a large portion of this phosphorus total, especially in soils high in clays, calcium, aluminum, or iron is chemically bound (adsorbed) to soil particles and unavailable, at least in the short-term, to terrestrial plants (Brady, 1974; Holtan et al., 1988; Kilmer and Taylor, 1980; Nelson and Logan, 1983; Sharpley and Menzel, 1987).

The capacity of soils to adsorb **P** also varies widely (Table 1) (Barrow, 1980; Brady, 1974; McAllister and Logan, 1978; Oloya and Logan, 1980). Those with high

Table 1: Phosphorus Content, Availability, and Absorption Capacity for Soils and Sediments in the Maumee River Basin, Ohio (from McAllister and Logan, 1978)

	Total P (ug/g)	Available P (ug/g)	Adsorption maximum (ug/g)
Soils			
Roselms I	1,018	26.8	287
Broughton	568	2.7	209
Roselms II	554	15.8	249
Lenawee	976	46.4	216
Blount	450	13.7	244
Paulding	780	8.6	199
Hoytville	816	21.7	258
Bottom sediments			
Independence (12/1/1975)	476	36.7	222
Auglaize (12/1/1975)	1,260	28.6	4,870
Tiffin (12/1/1975)	753	24.2	1,930
Independence (3/24/1976)	949	19.0	3,580
Auglaize (3/24/1976)	1,150	13.9	4,550

capacities tend to become phosphorus enriched over time as phosphorus is made available for adsorption to the soil particles from weathering, organic materials, or fertilizers (Table 2). As an example of this high adsorption capacity and enrichment, a study cited by Kao and Blanchar (1973) of an Indiana soil after 82 years of phosphate fertilization showed the **P** content of the soil had nearly doubled while leaving the adsorption capacity nearly unchanged.

The generally high adsorption capacity of soils prevents little phosphorus from escaping the soil profile, although small amounts are present in subsurface runoff (water traveling through soil below the surface) (Ellis et al., 1989; Freeze, 1972; Nelson and Logan, 1983). However, the largest amounts of **P** carried in runoff are not leachates from water percolating through the soil profile, but **P** carried with sediments detached from the soil surface (Ahl, 1988; Cahill, 1977; Maas et al., 1987; Reddy et al., 1978; Sharpley and Syers, 1979).

When soil particles are moved to watercourses by erosional processes the adsorbed **P** is carried with them. In aquatic systems, any phosphorus compound may be transformed to a bioavailable form given the right conditions and enough time (Van Wazer, 1973). Waterborne sediments with adsorbed **P** are surrounded by water and all surfaces are subject to the enzymatic processes of algae. These processes can transform phosphorus compounds to bioavailable forms at rates tens of thousands of times more rapid than chemical hydrolysis reactions at normal temperatures (Van Wazer, 1973). Eroded sediments are a rich source of **P** for aquatic systems.

Atmospheric Inputs

The phosphorus cycle does not have a gaseous component, but significant atmospheric contributions can be made by wind borne particles. These contributions may include dust carried by wind or rain, and wind borne seeds and pollen (Ahl, 1988; Holtan et al., 1988; Sober and Bates, 1979). Pollen is very rich in phosphorus, making it a potentially rich source for streams with a dense canopy of vegetation (Griffith, 1973).

The **P** content of precipitation is highly variable both regionally and temporally (Holtan et al., 1988; Loehr, 1974; Sober and Bates, 1979). While rain-carried dust may come from many sources, activities that increase the amount of dust, and as activities,

Table 2: Total Phosphorus Content of Soils: Examples from Four States (from Brady, 1974; Reddy et al., 1978)

Soils	Total P (ug/g)	Organic Fraction (%)	Reference
Western Oregon soils			Bertramson and Stephenson, 1942
Hill soils	357	65.9	
Old valley-filling soils	1,479	29.4	
Recent valley soils	848	25.6	
Iowa soils			Pearson and Simonson, 1939
Prairie soils	613	41.6	
Gray-brown podzolic soils	574	37.3	
Planosols	495	52.7	
Arizona soils			Fuller and McGeorge, 1951
Surface soils	703	36.0	
Subsurface soils	125	34.0	
Ohio soils			Reddy et al., 1978
Silty clay	715	44.9	
Silt loam	679	49.3	
Sandy loam	398	43.2	

such as fertilization, enrich soils with **P**, the dust is enriched and the contribution is increased (Ahl, 1988). Generally atmospheric contributions are highest during summer and near industrial or agricultural areas and lowest near remote areas and during the season of highest precipitation (Holtan, et al., 1988).

Phosphorus from atmospheric sources can be a significant nonpoint source (Table 3), but the relative contribution varies widely, from 1.2% to 80%, and is generally greatest where other sources are limited (Barica and Armstrong, 1971; Loehr, 1974; Sober and Bates, 1979).

Anthropogenic Inputs

Point sources may contribute **P** directly to water in readily available forms from industrial discharges, sewage effluent (often enriched by **P** from detergents), and concentrated animal waste from confined livestock operations (Ellis et al., 1989; Holtan et al., 1970; Maas et al., 1987). Similarly septic tank drain fields, dispersed animal wastes, crop fertilizers, irrigation return, tile drainage, and nutrients contained in urban runoff may be significant sources of nonpoint source enrichment if improperly managed (Holtan et al., 1988; Loehr, 1974; Maas et al., 1987; Miller et al., 1978).

Groundwater

For P-enriched surface water to recharge groundwater aquifers it must first percolate through the overlying layers of soil and other materials (Nelson and Logan, 1983). The concentration of **P** in groundwater is determined by the **P** adsorption characteristics of those materials (Nelson and Logan, 1983).

Generally, the concentrations of **P** in groundwater are low, due to the high P adsorption capacity of most soils, although extremely porous or cracked soils may not allow sufficient time for complete adsorption to take place (Bailey, 1968; Keup, 1968; Maas et al., 1987; Nelson and Logan, 1983). In a review of studies in Wisconsin and Maine, Keup (1968) found over 80% of groundwater had concentrations less than 0.02 mg/l (20 ug/l). The highest concentration was a single spring in Wisconsin with a

Table 3: Atmospheric Phosphorus Loading (from Ahl, 1988; Barica and Armstrong, 1971; Holtan et al., 1988; Loehr, 1974; Sober, 1979)

Location	P Load (kg/yr/ha)	Reference
United Kingdom		
Upland	0.27	Owens, 1970
Northern	0.2-1.0	Owens, 1970
Scotland	0.45-0.7	Crisp, 1966
Finland	0.14	Happala, 1977
Denmark	0.10-0.40	Harremoes, 1977
Sweden	0.20	Ahl and Oden, 1975
Norway	0.34	Berge et al., 1979
USSR	0.11-0.15	Evdokimova et al., 1976
United States		
Cincinnati, Ohio	0.6	Weibel, 1969
Northern Mississippi	0.41	Duffy et al., 1978
Lake Mendota, Wisc.	1.02	Sonzogni and Lee, 1974
Ithaca, New York	0.05	Likens, 1972
Hubbard Brook, New Hampshire	0.10	Hobbie and Likens, 1973
Lake Carl Blackwell, Oklahoma	0.605	Sober and Bates, 1979
Canada		
Northern	0.046	Schindler et al., 1974
Central	0.24-0.53	Schindler et al., 1976
Eastern	0.30	Schindler and Nighswander 1970
Ontario	0.77	Dillon, 1975
New Zealand	0.41	Bargh, 1977

concentration of .192 mg/l (192 ug/l). Concentrations of phosphorus have rarely been of concern as an impairment of groundwater (Maas et al., 1987).

Transport of Phosphorus

During the transport of nonpoint source phosphorus from one site to another it is capable of undergoing a number of transformations. Unfortunately, the chemistry of the interactions of **P** as it is carried by soil, sediment, and water are extremely complex and not completely understood (Bostrom et al., 1988b; Nelson and Logan, 1983). Since the chemical forms **P** takes during transport vary widely and rapidly, its transport may be better understood by considering the physical forms of transport, particulate and soluble (or dissolved).

Soluble Phosphorus

Background levels of **P** are difficult to determine since **P** transport is very sensitive to man's activities, and virtually no basins are completely undisturbed (Ahl, 1988). However, soluble **P** levels of 0.007 mg/l (7 ug/l) have been commonly reported for forest streams and lakes without sources of urban or agricultural runoff and is considered to be a common value for background levels of soluble **P** in many surface waters (Verduin, 1970). This does not mean that all systems have similar background levels. A summary of eight studies from relatively undisturbed forested watersheds by Loehr (1974) found values as high as 0.008 mg/l (8 ug/l) soluble **P** for the Tieton River and 0.009 mg/l (9 ug/l) soluble **P** for the Yakima River. Waterways that drain organic soils, such as muck soils, may have background concentrations that are higher by ten times or more (Table 4) (Duxbury and Peverly, 1978). Despite exceptions, it is assumed that most aquatic ecosystems developed with concentrations close to the .007 mg/l (7 ug/l) level of dissolved P, but by the early 1970s all major streams in the United States had levels five to thirty times greater (Verduin, 1970).

Table 4: Phosphorus Concentration in Drainage Water from Mucklands (Histosols)
(from Duxbury and Peverly, 1978)

Location	Soluble P concentration (ug/l)	Reference
New York		
Shallow soils	3,850	Duxbury and Peverly, 1978
Deep soils	9,100	Duxbury and Peverly, 1978
Florida		
	30,000	Hortenstine and Forbes, 1972

By definition, soluble **P** is a form that will easily dissolve in water and is readily available for plant growth (Bostrom et al., 1988a; Nelson and Logan, 1983; Sharpley and Menzel, 1987; Syers et al., 1973). It is this dissolved form of **P**, rather than total **P** (including adsorbed forms), that is primarily responsible for eutrophication and other water quality problems (Nelson and Logan, 1983; Sharpley and Menzel, 1987; Syers et al., 1973). Unfortunately, the separation of dissolved **P** in field samples is an arbitrary process, being defined as the portion that will pass through a 0.45 micron filter which may include some fine colloidal materials with adsorbed **P** (Nelson and Logan, 1983). It may be difficult to differentiate between materials that are truly dissolved and those associated with fine clays and silts (Walling, 1977).

Concentrations of soluble **P** are the product of a variety of watershed processes and generally independent of stream discharge rates (Cahill, 1977; Nelson and Logan, 1983; Prairie and Kalff, 1988a). No modeling equations have been found which adequately predict the complexities of soluble phosphorus equilibrium reactions under varying conditions (Nelson and Logan, 1983).

Soluble **P** may be contributed directly or indirectly to a watercourse in groundwater, and by fertilizers or animal wastes (Armstrong and Rohlich, 1973; Boreham et al., 1987; Brady, 1974; Collin, 1975; Duda and Johnson, 1985; Hall, 1986; Kilmer and Taylor, 1980; Nelson and Logan, 1983; Nielsen, 1987; OECD, 1986; Phillips, 1986; Phillips, 1987; Sharpley and Menzel, 1987; Sommers and Sutton, 1980;

Thornley and Bos, 1985, Unwin, 1987; Verduin, 1970; Young et al., 1985). Soluble P may also be a product of P equilibrium reactions that take place during rainfall and runoff events.

One such reaction occurs when water drops from rainfall or sprinkler irrigation leach small amounts of P from the leaves of plants as they pass over the leaf surface (Sharpley, 1981; Sharpley et al., 1981b; Sharpley and Menzel, 1987). A similar reaction takes place when water running over the surface of the soil causes the desorption of P from the thin surface layer of soil with which it is in contact (Sharpley et al., 1981a; Sharpley et al., 1981b; Sharpley and Menzel, 1987). The losses from the plant canopy, soil surface, and leaching from dead plant material are the primary sources of soluble P added to overland runoff (Sharpley, 1981; Sharpley et al., 1981b; Sharpley and Menzel, 1987).

Particulate Phosphorus

Particulate P includes organic materials and minerals containing P and soil particles with P bound to them (adsorbed P) (Boberg and Persson, 1988). Particulate P in streams includes P bound to sediments from surface and streambank erosion, and contained in the remains of living and dead aquatic organisms (Boberg and Persson, 1988; Hooper, 1973; Sharpley and Syers, 1979; Walling, 1977).

The major portion of P carried in runoff is generally sediment bound (Sharpley and Syers, 1979). As long as this P is bound to the particles it is not available for plant growth (Kilmer and Taylor, 1980; Nelson and Logan, 1983; Sharpley and Menzel, 1987). The amount of particulate P transformed to the dissolved form and available to plants as these materials break down varies, but can be significant, especially from organic material (Nelson and Logan, 1983; Oloya and Logan, 1980; Sharpley and Menzel, 1987; Sharpley and Syers, 1979). While it is known that the transformation from particulate to dissolved form takes place, the combinations of conditions that cause this are not fully understood (Bostrom et al., 1988b).

Fine-textured soils, such as clays and silts, have the greatest affinity for P (Day et al., 1987; McAllister and Logan, 1978; Miller, 1977; Nelson and Logan, 1983; Sharpley and Menzel, 1987). Soil erosion processes from overland flow are selective, with fine sediments being more likely to be carried in runoff (Kilmer and Taylor, 1980; Miller, 1977; Sharpley and Menzel, 1987). This preference increases the nutrient concentration of sediments by selecting those particles most likely to be carrying adsorbed **P** (Miller, 1977; Nelson and Logan, 1983; Sharpley and Menzel, 1987). The ratio between the **P** concentration in runoff sediment and the **P** concentration of the soil it was derived from is termed the enrichment ratio (Nelson and Logan, 1983; Sharpley and Menzel, 1987). Enrichment ratios of 1:2 to 1:6 are not uncommon for large watersheds and are one of the elements of predicting total sediment **P** loads (Nelson and Logan, 1983). Total sediment **P** loads are considered to be a product of soil loss, the **P** content of surface soil, sediment enrichment ratio, and sediment delivery rate (Nelson and Logan, 1983). Sediment enrichment ratios have an inverse relationship with soil loss. As the sediment delivery ratio decreases, it is the finest, most enriched particles that will continue to be carried in runoff (Nelson and Logan, 1983).

Most of the soil eroded from the land surface in any given year is moved during large storm events (Knox, 1977; Sullivan, 1985). The movement of soil as sediment depends on the detachment and transport of soil particles by such forces as flowing water or the impact of raindrops (Brady, 1974). Vegetative cover lessens both the force of raindrop impact for detachment, and the velocity of the overland flow for transport (Brady, 1974). Vegetation can also decrease the amounts of sediment delivered to a stream by reducing the velocity of overland flow, decreasing its capacity to carry sediments (Cooper et al., 1987; Lowrance, 1990; Lowrance et al., 1984; Lowrance et al., 1985; Omernik et al., 1981; Schlosser and Karr, 1981).

One of the transformations between particulate **P** and soluble **P** takes place as soil particles suspended in overland flow interact with rainwater in sorption-desorption reactions during detachment and transport (Nelson and Logan, 1983). The extent and direction of these exchanges depends on the **P** equilibria of the soil carried and the soluble **P** concentration of the rainwater (Nelson and Logan, 1983).

Other transformations from particulate to soluble form take place after sediments are delivered to the watercourse. Suspended sediments act as a buffering agent for phosphorus in streams, adsorbing **P** when soluble concentrations are high, and desorbing **P** when soluble concentrations are low (Black, 1970; Golterman, 1973; Hill, 1982). As particles settle, in slack water areas or lakes, the buffering is reduced in both speed and capacity (Hill, 1982; Sharpley and Menzel, 1987). This is moderated in part by the tendency of the smallest particles, which have a greater capacity for sorption-desorption reactions, to stay suspended for the greatest time (Hesse, 1973; Oloya and Logan, 1980).

In turbulent streams or rivers there is generally enough energy to re-suspend or keep suspended both large, heavy particles and lighter, finer particles. As velocity decreases, such as during periods of low flow, the heavier particles will settle out, leaving the clays and silts in suspension (Boberg and Persson, 1988). This selectivity for particle size is similar to that of overland erosion and has similar effects. During low flow periods, the total suspended material will decrease, but the total **P** in the water column will stay proportionally higher.

The pH of water surrounding particles also has an affect on transformations. In soils, phosphorus is most available for plants in a range near neutral pH. At lower pH levels it tends to form less soluble compounds with readily available iron, aluminum or manganese, and at higher pH levels, with calcium (Brady, 1974). In water, these elements are not as available, and **P** from sediments is increasingly released into solution at pH levels below 5 or above 9.5 (Boers, 1991; DeLaune et al., 1981).

The **P** bound to sediments is not all immediately available to plants, but is available as transformations take place over time (Nelson and Logan, 1983; Sharpley and Menzel, 1987; Van Wazer, 1973). Sediments rich in iron or aluminum may have more immediate effects on algal growth since phosphorus bound to these elements is most readily available to algae (Dorich et al., 1980). As algae reduce the soluble **P** concentration in water below that of the equilibrium **P** concentration of the sediment, sediment-bound **P** will be desorbed and become available for plant growth (Golterman, 1973; McAllister and Logan, 1978). Rooted aquatic plants are able to extract **P** from sediments that have been deposited and are less available to sorption-desorption reactions (MacKenthun, 1973a). Material from dead aquatic plants, rooted or free-floating, is a ready source of **P** for other aquatic plants (MacKenthun, 1968; Nelson and Logan, 1983; Sharpley and Menzel, 1987). In this organic mass, large quantities of **P** can be moved gradually downstream in a form that is not routinely analyzed (Keup, 1968). These aquatic plants have been found to make up a substantial portion of particulate matter in streams (Rigler, 1979). Individual P atoms are taken up by plants, released in organic compounds, perhaps settle to the bottom as sediment for a time, are taken up again by plants, and continue moving downstream in this spiral fashion (Meyer et al., 1988; Mulholland et al., 1983; Newbold et al., 1983). The rate and distance P moves with each spiral varies with seasonal changes in light, temperature, and other environmental factors (Mulholland et al., 1983). When conditions are favorable biological cycling can be very rapid, especially in shallow waters in which the entire water column is exposed to light, such as is common in streams without a closed overhead canopy (Holtan et al., 1988). At any one time, much of the total phosphorus of the system may be tied up in one of these temporary storage pools. An accurate analysis of the total P in the system and predictions of its movement and transformations, even if fully understood, would be exceedingly complex.

In addition to sediment from the land surface, another major source of sediment is streambank erosion (Sharpley and Syers, 1979; Walling, 1977). Increased flow and velocity, which may re-suspend instream sediments, also affect the amount of sediment removed from streambanks (Sharpley and Syers, 1979; Walling, 1977). Although the subsoils from streambanks tend to be less phosphorus enriched than field topsoils, streambanks may contribute as much as 75% of all sediments for some streams (Kilmer and Taylor, 1980; Sharpley and Menzel, 1987; Sharpley and Syers, 1979). Streambank erosion is most severe for wet, unprotected banks that are subjected to rising flows with their accompanying increased velocities (Walling, 1977).

Sediments in flowing water systems move on a seasonal scale. Small particles and organic material will tend to stay suspended for longer periods and during lower flows than larger soil particles (Boberg and Persson, 1988; Keup, 1968; Walling, 1977). During high flows, sediments deposited during periods of low flow will be re-suspended and moved downstream (Boberg and Persson, 1988; Cahill, 1977; Cahill et al., 1974; Hill, 1982; Keup, 1968; Meyer et al., 1988; Rigler, 1979). The total P stored in stream or river sediments can be very large (Hill, 1982; Keup, 1968; Rigler, 1979). Most of this sediment is moved on the rising limb of the storm hydrograph, especially during high flow events early in the runoff season (Paustian and Beschta, 1979; Meyer et al., 1988; Rigler, 1979; Sullivan, 1985). The concentrations of sediment during stormflow can

Seasonal Variations

Both load and concentration of phosphorus vary over time. Loads are highly affected by the seasonal variation in suspended sediment load discussed above. Concentrations also vary seasonally, but the influences are not as clear. Three explanations, runoff effect, dilution effect, and temperature effect, have been offered, and the answer probably lies in combinations of these effects.

The runoff effect describes an increase in concentration with an increase in streamflow (Cahill et al., 1974). This may be explained by an suspended sediment load, both those entering the stream from the land surface and those reentering the water column from the bottom as increasing flow increases turbulence (Cahill, 1974). The magnitude of this effect will be influenced by the **P** content of the sediments.

The effect of dilution may be used to describe two related conditions. One is a rise in **P** concentrations as streamflow decreases, and the other is a drop in **P** concentrations as flow increases (Cahill et al., 1974). In streams with a groundwater baseflow that is relatively enriched, summer low flows are not diluted by low concentration stormwater and concentrations will be high (Cahill et al., 1974; Prairie and Kalff, 1988a). As water from a storm enters such a stream concentrations may drop as baseflow is diluted. However, if the storm event is a large one and flow rises rapidly, then the runoff effect may be the predominant influence.

Temperature may directly affect **P** concentration dynamics. In soil, high temperatures will yield solutions with high **P** concentrations (Barrow, 1974; Barrow and Shaw, 1975; Stuanes, 1982). For water similar relationships have been shown. When water temperatures rise above 15°C (60°F) the rate at which **P** is released from suspended solids and sediments is substantially increased (Karr and Schlosser, 1978).

Temperature may also affect **P** concentrations indirectly. Warmer temperatures may promote the growth of aquatic plants, which in turn may remove **P** from the water for growth. Over time, as the plants die, this **P** will then reenter the stream and be recycled in the spiral fashion previously described.

Other seasonal events that may affect phosphorus concentration and load include the addition of nutrient-rich leaves in autumn and increased aquatic plant growth with increased light (Mulholland et al., 1985; Waller and Hart, 1986). In many heavily shaded streams maximum plant growth may occur in late fall after streamside plants have shed their leaves and the canopy has opened (Mulholland et al, 1985).

Management Implications for the Tualatin Basin

No matter the form, virtually all **P** that makes its way into surface waters is carried there by water, primarily overland flow (Fig. 1). Particulate **P** contributed to a stream is primarily carried in runoff sediments (Ellis, et al. 1989; Kilmer and Taylor, 1980; Nelson and Logan, 1983; Sharpley and Menzel, 1987). Dissolved **P** that infiltrates the soil surface will very likely be adsorbed, and losses from subsurface drainage are slight, although amounts are higher for very rapid subsurface runoff since the time available for adsorption is reduced (Sharpley and Menzel, 1987; Sharpley and Syers, 1979). Simply stated, to control NPS phosphorus, prevent overland flow and soil erosion by keeping soil and water on site.

The TMDL placed on the Tualatin basin is based on concentrations of total **P**, particulate and dissolved loads combined, rather than the more bioavailable dissolved form. As with any other sediments, particulate **P** may settle out of the water column and be incorporated into bottom deposits (Hill, 1982; Keup, 1968; Leopold et al., 1964; Meyer and Likens, 1979). Sediment **P** deposited in the photic zone may be available to rooted plants (Meyer and Likens, 1979). Organic materials from these plants, a source of bioavailable **P**, may then reenter the water column for transport downstream (Nelson and Logan, 1983; Oloya and Logan, 1980; Sharpley and Menzel, 1987). Sediment deposits, including organic material, may be re-suspended and moved downstream when flows increase, especially by the moderate frequency and magnitude flows which are responsible for most sediment transport (Keup, 1968; Leopold et al., 1964).

Eventually these **P** enriched sediments may be deposited in the main stem of the Tualatin. River reaches with low gradient and water velocity are generally the most likely places for these sediments to be deposited and for eutrophication to occur (Leopold, et al., 1964). It is likely that **P** in sediments in this portion of the river act similarly to sediment **P** in lakes. Research on **P** in lakes has shown that portions of this total sediment load can be desorbed over time (Armstrong et al., 1987; Bostrom et al., 1988b; Oloya and Logan, 1980; Syers et al., 1973). The retention time of **P** in production. Generally, poor planning and maintenance cause even greater problems (Brown, 1980). Surface erosion associated with roads and skid trails is proportional to road and skid trail density (Brown, 1980). In addition, roads along streams have the greatest effect on sediment production (Anderson, 1974). In basins that are **P** sensitive, road construction should be planned to minimize the length of roads necessary, continued maintenance should be performed, and in both planning and maintenance, every technique available to reduce sediment production should be used. Roads, skid trails, and log landing sites, should also be placed some distance from streams to decrease the likelihood of sediments reaching stream channels (Anderson, 1974; Lynch et al., 1985; Lynch and Corbett, 1990). Alternative yarding methods, that reduce soil disturbance associated with contact between the log and the ground and that have a reduced reliance on roads, such as helicopter or balloon logging, should also be seriously considered in **P** sensitive basins.

ii. Logging. Generally, tree removal by itself has little effect on sediment production. An exception would be especially steep sites or those with unstable soils, where the removal of forest vegetation may cause increased sediment production by removing the stabilizing influence of root systems (Brooks et al., 1991; Coats and Miller, 1981; Nikolayenko, 1974; Swanston, 1971). In **P** sensitive basins these areas must be managed thoughtfully. Harvest of these areas should be avoided, or performed selectively to avoid destabilizing large areas in a short time period.

iii. Site Preparation. Studies of the effects of slash burning on nutrients have somewhat variable results. Fredriksen (1971) reported a doubling of dissolved **P** after slash burning in the western Cascades of Oregon, while a study in the Oregon Coast Range by Brown and others (1973) and another by Harr and Fredriksen (1988) in the western Cascades of Oregon found concentrations of dissolved **P** to be relatively unchanged after slash burning. Although none of the studies reported total **P** or particulate **P** concentrations, all reported large increases of sediment following slash burning, which would be consistent with large losses of particulate P. In the study reported by Harr and Fredriksen (1988) suspended sediment levels remained elevated for nine years following slash burning. Slash burning would not be a recommended management practice in **P** sensitive basins.

Slash burial, although more expensive, has been considered as an alternative means of slash disposal (Larson and Wooldridge, 1980). Studies of nutrients in water flowing through buried slash piles indicate that buried slash may not be a significant source of nutrients (Larson and Wooldridge, 1980). Generally, the **P** found in organic materials becomes bioavailable as materials decay, but is readily adsorbed by soil particles if transported in subsurface flow. The disposal of slash by burial would be preferable to burning, but careful consideration would need to be given to the amount of soil disturbance necessary for burial.

Another suggested alternative has been leaving slash on the soil surface to naturally decompose (Lynch et al., 1985). The study by Harr and Fredriksen (1988) showed that this method worked well to reduce sediment production following harvest. Suspended sediment levels from this unit showed a brief increase associated with road construction and then returned to background levels. This method is not without costs however. Growth of the replanted tree stand was approximately half of that in the burned area, possibly from increased shrub competition.

iv. Streamside Management. Riparian areas have been identified as sites which influence many aspects of water quality in streams, including sediments (Brooks et al., 1991; Brown, 1980; Karr and Dudley, 1981; Karr and Schlosser, 1978; Lowrance et al., 1984; Lowrance et al., 1985; Lynch and Corbett, 1990; Nikolayenko, 1974; Omernik et al., 1981; Schlosser and Karr, 1981). Riparian vegetation helps reduce the velocity of any overland flow, increasing the opportunity for sediments to deposit before reaching watercourses, and promotes infiltration (Gregory et al., 1991; Karr and Schlosser, 1978; Li and Shen, 1973). Plant uptake will also help detain **P** before it reaches the stream (Karr and Dudley, 1981; Lowrance et al., 1984; Lowrance et al., 1985; Omernik et al., 1981; Schlosser and Karr, 1981). Phosphorus that infiltrates is likely to be adsorbed onto soil particles and unlikely to move farther except as particulate P, associated with either sediment or organic materials (Lowrance et al., 1984; Lowrance et al., 1985; Omernik et al., 1981; Sharpley and Menzel, 1987). Over time these riparian nutrient sinks need to be managed with selective forest harvest and minimal soil disturbance to move the **P** retained in them off site rather than into streams as organic material (Lowrance et al., 1984; Lowrance et al., 1985; Lowrance, 1990). Opinions on the required width of this riparian buffer zone necessary to control sediments varies, but generally these strips are wider than those only concerned with vegetation for stream shading and woody debris contributions (Brown, 1980; Lynch and Corbett, 1990; Lowrance, 1990). The costs of this practice are primarily those of removing land from production (Karr and Dudley, 1981).

v. General Guidelines. The land disturbance of silviculture at any one time is on a relatively small scale compared to agricultural land uses. But the affect on the site and affected local area, although short-lived, is high in both impact and pollutant loading (McElroy, 1977).

Where NPS phosphorus is a water quality concern, the impacts of forest activities will depend on whether total **P** or dissolved **P** is used as the basis for regulation. If dissolved **P** is the pollutant of concern, then forest activities (with the possible exception of slash burning) have relatively little effect. However, if total **P** is of concern, then any forest activity that disturbs the soil surface and has the potential of producing sediment is of concern. As an example of the potential effects of forest practices on P, in some areas average particulate **P** concentrations can be predicted from the extent of forest cover (Prairie and Kalff, 1988b).

In areas regulated by total **P** concentrations, methods that are most effective in preventing or reducing sediment production need to be identified and implemented. If effective control is not achieved, then any method that reduces soil erosion, including the reduction or elimination of forest harvest may need to be considered to meet NPS pollutant goals. This should prove a strong incentive for research and change.

Agriculture

Because of the large areas of land used for agricultural purposes in the United States, agriculture's share of land-based NPS pollution is large. The diffuse nature of these pollutants makes direct measurement difficult, but estimates indicate that for the nearly two-thirds of the nation's nonfederal land that is cultivated or used for grazing, agriculture contributes nearly 70 percent (3 million tons annually) of the total load of phosphorus, primarily from sediments, fertilizers and animal wastes (Chesters and Schierow, 1985; Myers, 1986).

Without doubt agriculture has had great effect on the appearance, structure, and function of the land. These changes have created many benefits, but also some unanticipated problems. The development of farmland alters the quantity of nutrients and sediments exported from it, and the quality and flow of water that drains from it (Smith, 1959). The ratio of change from background loads to current load of sediment and nutrients is generally proportional to the extent that land has been changed from its natural state to a disturbed state, generally agriculture (McElroy, 1977; Prairie and Kalff, 1988b). Even the conversion of land from forest to pasture, which is a relatively stable use, has doubled the amount of phosphorus exported from the watershed in some areas (Dickinson and Wall, 1977; Dillon and Kirchner, 1975).

a. Effects of Agricultural Practices on Dissolved Phosphorus

Agriculture may contribute soluble phosphorus to a stream in a variety of ways (see Fig. 4). There is a basic dilemma in attempting to supply the **P** necessary for plant growth. In order for **P** to be available it must be supplied in quantities greater than can be immediately adsorbed by soil and in forms that are readily soluble (Brady, 1974). This also increases quantities of soluble **P** at the soil surface where it is most likely tobe carried off-site with overland flow (Armstong and Rohlich, 1973; Kilmer and Taylor, 1980; Nelson and Logan, 1983; Sharpley and Menzel, 1987; Verduin, 1970).

Dissolved **P** may also be contributed to surface waters in significant amounts when animal wastes are carried in runoff (Boreham et al., 1987; Brady, 1974; Collin,1975; Duda and Johnson, 1985; Hall, 1986; Nielsen, 1987; OECD, 1986; Phillips, 1986; Phillips, 1987; Sommers and Sutton, 1980; Thornley and Bos, 1985, Unwin, 1987; Young et al., 1985).

b. Effects of Agricultural Practices on Particulate Phosphorus

Activities that disturb the land surface may contribute particulate **P** to streams (see Fig. 5) Most of the **P** lost from agricultural lands is moved in particulate form as sediment in stormflow runoff (Cahill, 1977; Reddy et al., 1978). The volume of runoff and amount of sediment lost are related to the amount of rainfall and irrigation, erodibility of soils, the length and steepness of slopes, and extent and nature of

Figure 4: Influences of Agricultural Practices on the Transport of Dissolved P

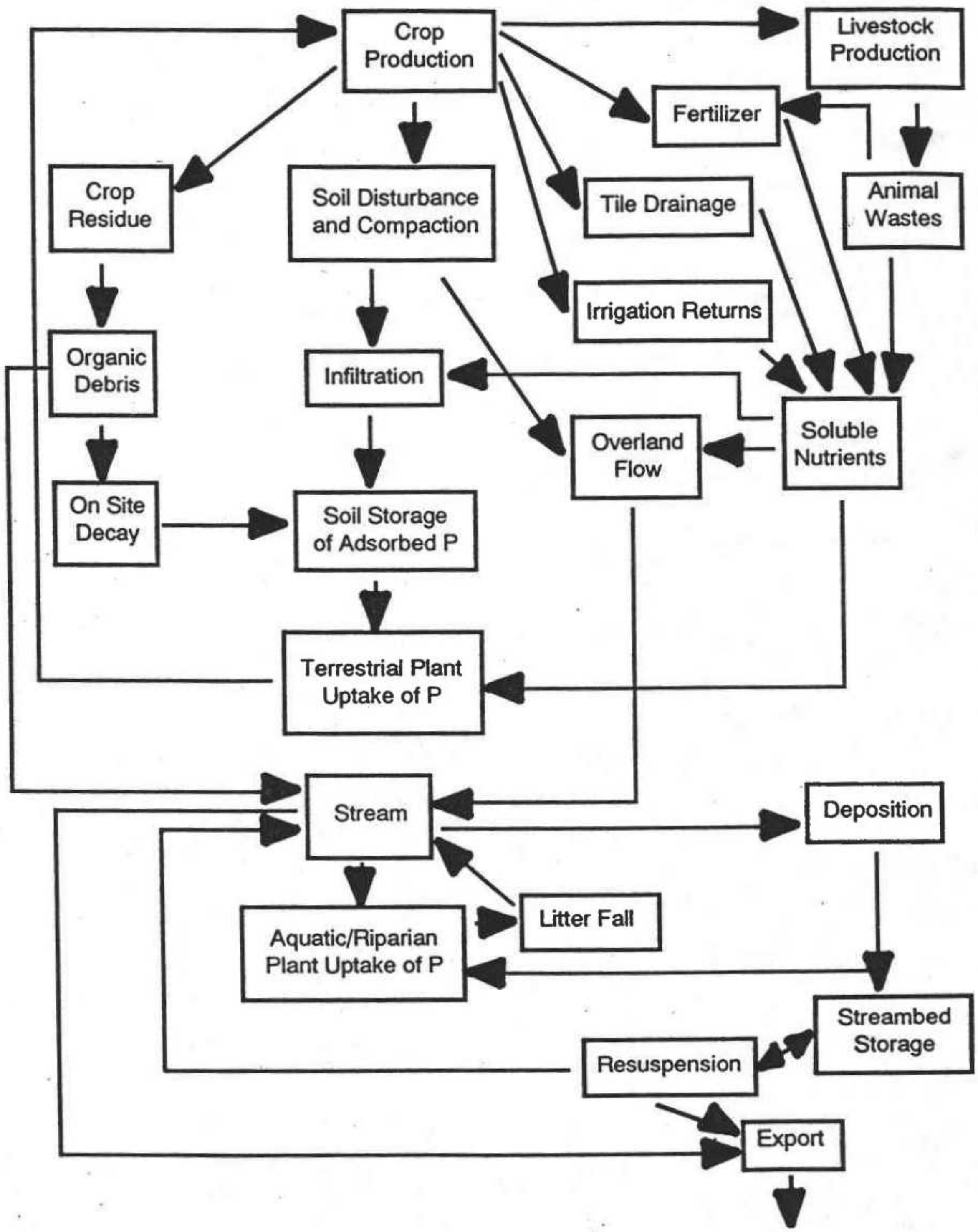
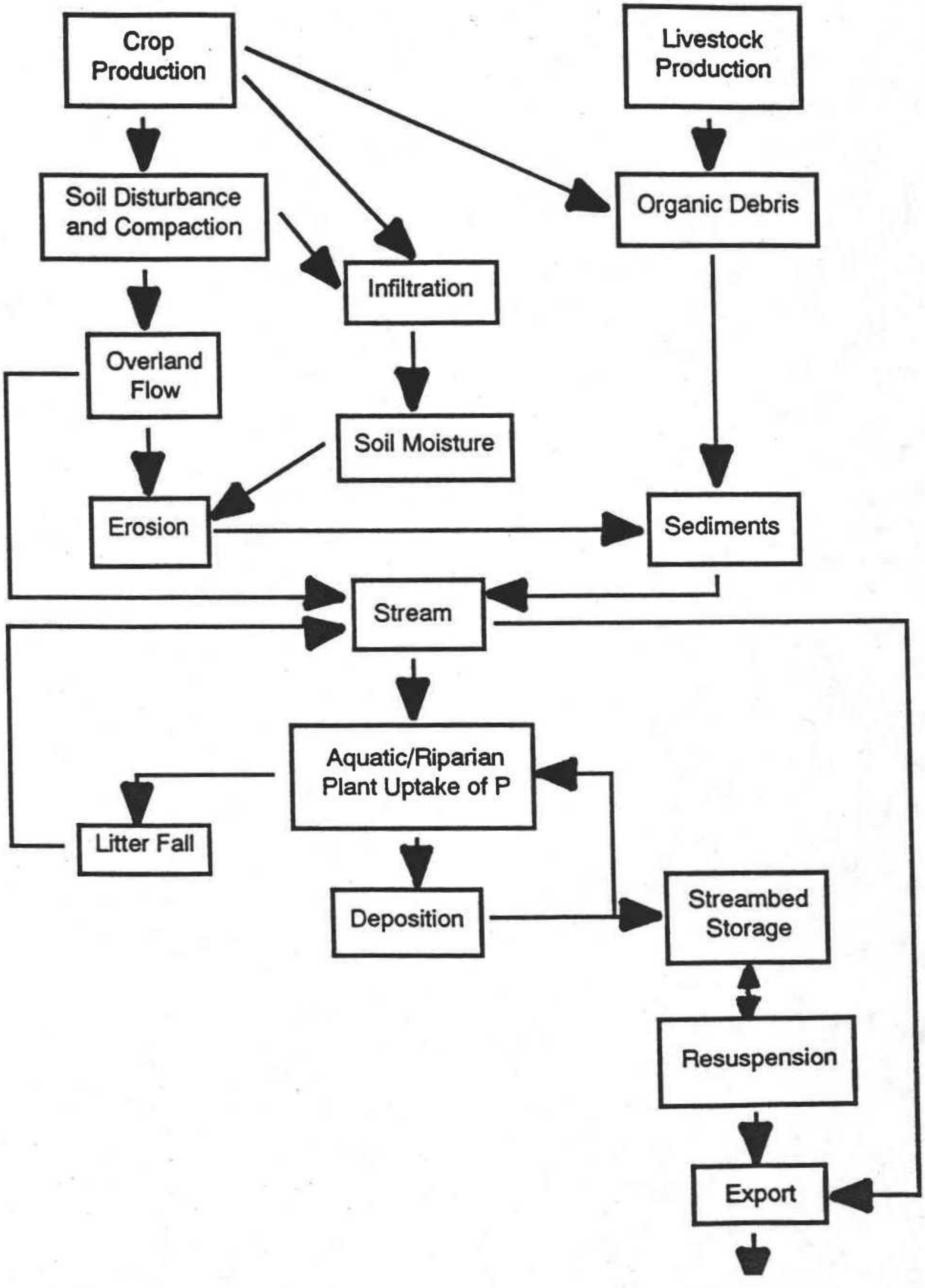


Figure 5: Influences of Agricultural Practices on the Transport of Particulate P



vegetation on the land surface (Brady, 1974; Brooks et al., 1991; Carter, 1976; Schlosser and Karr, 1981). The amount of sediment delivered to a stream depends on the factors listed above, and the distance to the watercourse (Maas et al., 1987; Schlosser and Karr, 1981). Of the factors that affect both the amount of sediment lost from a site, and the amount of sediment delivered to a watercourse, the one most readily managed by human activity is vegetation. Without vegetative cover, exposed soils are likely to have surface particles loosened by rainfall impact (Brady, 1974; Brooks et al., 1991). Loosened particles may be carried into the surface pores and reduce the infiltration rate, which in turn increases the volume of runoff (Brady, 1974; Brooks et al., 1991; Mueller et al., 1981). As volume of flow increases so does the capacity of the runoff to loosen and transport sediments (Brady, 1974; Brooks et al., 1991). Vegetative cover can decrease sediment delivery rates and volume of overland flow by reducing the impact of raindrops, and providing stems and organic material to reduce runoff velocity, increase surface storage sites, and retain sediments (Brady, 1974; Brooks et al., 1991; Cooper et al., 1987; Lowrance, 1990; Lowrance et al., 1984; Lowrance et al., 1985; Mueller et al., 1981; Omernik et al., 1981; Schlosser and Karr, 1981).

c. Management Implications: Controlling Nonpoint Source Phosphorus from Agricultural Lands

The method for controlling nonpoint source phosphorus pollution from agricultural land is a prescription that applies to nearly all nonpoint source pollutants. Simply stated: Keep soil and water on the site. If all water infiltrates through soils, and all soils stay on site, losses of **P** will be very low, similar to the background level of undisturbed sites. While the solution may be simply stated, it is not a simply applied solution.

The variability of climate and conditions on a site make total elimination of runoff and erosion unlikely. Even undisturbed natural systems at steady state have slight erosion (Jenny, 1984). Currently the U.S. Department of Agriculture considers, on average, five tons per acre to be an acceptable rate of erosion (Jenny, 1984; Worster, 1984). In watersheds with extensive cultivation and P-enriched topsoils, five tons per acre, while not having a negative impact on crop production, could have great detrimental impacts on the receiving waters.

Agriculture has identified Best Management Practices (BMPs) that describe practices effective for retaining soil and water on site under different conditions. The adoption of appropriate BMPs would be a great step toward reducing overland flow and the sediments it carries (Agricultural Research Service, 1975). Although studies on Vermont watersheds summarized by Clausen and Meals (1989) indicate that while effective at reducing pollutant loads, BMPs were often unable to provide the level of control needed to meet water quality goals, others have shown that the effectiveness of BMPs are greatly enhanced when they are used in combination rather than singly (Maas et al., 1987). Effective control of nonpoint phosphorus in the Tualatin Basin will not come from a single strategy, but from a variety of strategies applied in combinations that fit individual site conditions. The following is a summary of some BMPs that may be applicable to the Tualatin Basin.

i. Conservation Tillage. A variety of methods use little or no tillage to retain crop residues on the soil surface and increase the organic content of the surface layer (Agricultural Research Service, 1975; Baker, 1985; Chesters and Schierow, 1985; Crosson and Ostov, 1990; Mueller et al., 1981). The tillage systems most effective in reducing sediment production are those with minimum soil disturbance and maximum organic material left on site (Razavian, 1990).

While valuable for reducing sediment loss and the particulate **P** associated with it, this practice may increase the loss of dissolved **P** through the leaching of the organic material remaining on the site (Alberts and Spomer, 1985; Forster et al., 1985; Langdale et al., 1985; Maas et al., 1987; Mueller et al., 1981; Razavian, 1990). Effectiveness at reducing total **P** loads would probably be site specific, depending on the relative **P** concentrations carried in sediments and leachate.

ii. Winter Cover Crops. Much soil is lost during the portion of the year when soils are saturated, infiltration is slow, and water flowing over unprotected soils easily moves sediment. Vegetation aids in binding the soil and slowing overland flow (Agricultural Research Service, 1975). The effectiveness of this practice depends on the cover crop planted and the growth stage the crop achieves before the non-growing season (Maas et al., 1987). A detriment of this system is that the cover crop delays the spring warming and drying of soil, potentially delaying plowing and planting (Maas et al., 1987).

iii. Contour Farming. Contour plowing and the alternating of row crops with grass or hay strips can reduce soil loss by up to 50% on moderate slopes (Agricultural Research Service, 1975; Chesters and Schierow, 1985; Maas et al., 1987). This method rapidly decreases in effectiveness on steep slopes and is always proportionally more effective at reducing sediments than nutrients (Maas et al., 1987).

iv. Terracing. This method alters the shape of the land surface to reduce the effective slope length. Reducing slope length decreases runoff velocity and the capacity of runoff the carry sediments (Agricultural Research Service, 1975; Chesters and Schierow, 1985). Terracing may reduce soil loss by 50% to 98%, although nutrient loss reductions are not as high (Maas et al., 1987).

v. Grassed Waterways and Diversions. Vegetated waterways reduce instream erosion, reduce the velocity of flows, and allow deposition of sediments on site (Agricultural Research Service, 1975). High runoff volumes or sediment loads greatly reduce the effectiveness of this practice (Maas et al., 1987). Its effectiveness can be enhanced by using it in conjunction with other methods, such as conservation tillage, strip farming, or terracing (Maas, et al., 1987).

vi. Riparian Filter Strips. Strips of riparian forest or forest and grass along streams provide many water quality benefits, including shading and maintaining instream temperatures, providing habitat and food for aquatic organisms, and act as sediment deposition sites and nutrient sinks (Brooks et al., 1991; Brown, 1980; Cooper et al., 1987; Cooper and Gilliam, 1987; Gregory et al., 1991; Karr and Dudley, 1981; Karr and Schlosser, 1978; Li and Shen, 1973; Lowrance et al., 1984; Lowrance et al., 1985; Lynch and Corbett, 1990; Maas et al., 1987; Meyer and Likens, 1979; Nilolayenko, 1974; Omernik et al., 1981; Schlosser and Karr, 1981). Riparian vegetation may reduce sediment delivery to a stream by more than 90%, although the sediment which reaches the stream will be the portion most highly P-enriched (Cooper et al., 1987; Karr and Schlosser, 1978; Nikolayenko, 1974; Omernik et al., 1981). Used as a nutrient management tool with dairy wastes, Dickey and Vanderholm (1981) found nutrients reduced 80% by concentration and 90% by weight. Although it is likely that dissolved phosphorus in water that infiltrates the soil will be adsorbed and stay on site, research has been inconclusive as to the effectiveness for reducing soluble **P** loads to streams (Cooper and Gilliam, 1987; Green and Kauffman, 1989; Lowrance et al., 1984; Lowrance et al., 1985; Meyer and Likens, 1979; Omernik et al., 1981; Sharpley and Menzel, 1987). The instream alteration of light and temperature conditions associated with these areas may also change the rate of nutrient processing in the stream (Cummins, 1974).

Uptake by plants will also help detain **P** before it reaches the stream (Karr and Dudley, 1981; Lowrance et al., 1984; Lowrance et al., 1985; Omernik et al., 1981; Schlosser and Karr, 1981). The effectiveness of these riparian nutrient sinks can be improved by crop removal or selective forest harvest and minimal soil disturbance to transport assimilated **P** off site rather than into streams as organic material or leachate (Iskandar and Syers, 1980; Lowrance et al., 1984; Lowrance et al., 1985; Lowrance, 1990).

More research is needed to identify the parameters required for effective control in different areas. Effectiveness is controlled by the width of the filter strip, type of vegetation, slope, rate at which runoff is applied, uniformity of runoff application along the length of the filter, concentration and load of pollutant, and management of the filter strip (Maas et al., 1987). Opinions on the width necessary for effective sediment control vary, but are generally wider than those only concerned shading to regulate stream temperatures (Brown, 1980; Lynch and Corbett, 1990; Lowrance, 1990). The costs of this practice are primarily those of removing land from production (Karr and Dudley, 1981).

vii. Low-Input Sustainable Agriculture (LISA) Systems. Traditional agriculture systems relied on the recycling of nutrients present on-site and the maintenance of soil health, while modern agricultural methods have shifted the emphasis from fertility to production (Sawyer, 1973). LISA is based on the idea that agriculture has the ability to renew its own resources and develop economically, culturally, and environmentally sustainable systems with a reduced reliance on purchased inputs (Ikerd, 1990; Rodale, 1990; Schaller, 1990). By emphasizing soil health, erosion reduction, active management of nutrients, and the reduction of synthetic additions such as fertilizers and pesticides, low-input systems have the potential for improving water quality (Crosson and Ostrov, 1990; Flach, 1990; Weinberg, 1990).

viii. Nutrient Management. Fertilizers should be applied at rates consistent with plant needs. This principle applies to both chemical fertilizers and animal wastes used as fertilizers (Agricultural Research Service, 1975; Armstrong and Rohlich, 1973; Brady, 1974; Collin, 1975; Hall, 1986; Hergert et al., 1981a; Kilmer and taylor, 1980; Maas et al., 1987; Nielsen, 1987; OECD, 1986; Phillips, 1986; Phillips, 1987; Reddy et al., 1980; Sharpley and Menzel, 1987; Sommers and Sutton, 1980; Thornley and Bos, 1985; Unwin, 1987; Verduin, 1970, Young et al., 1985). For animal wastes this may require the testing of the nutrient content of manures to insure application at the proper rate. Excessive loading can overload the **P** adsorption capacities of the soil, increasing the amounts of soluble **P** available to move off-site (Iskandar and Syers, 1980; Reddy et al., 1980).

Nutrient applications must be at the proper times. Applying any nutrients (chemical or animal waste) when soils are saturated increases the probability that nutrients will be lost to overland flows or drainage (Agricultural Research Service, 1975; Clausen and Meals, 1989; Hergert et al., 1981b; Iskandar and Syers, 1980; Kilmer and Taylor, 1980; Maas et al., 1987; OECD, 1986). For manures adequate storage allows greater flexibility in timing of applications (Maas et al., 1987; OECD, 1986; Phillips, 1986).

Surface applications should be incorporated into the soil surface. Mixing and distributing fertilizers--chemical, animal waste, or plant residues (cover crop or crop residue)--helps reduce nutrient leaching and aids in water infiltration (Agricultural Research Service, 1975; Clausen and Meals, 1989; Hall, 1986; Maas et al., 1987; OECD, 1986; Phillips, 1986; Phillips, 1987; Reddy et al., 1980; Verduin, 1970).

ix. Animal Waste Management. To control the quality of water flowing through an area with livestock, restricting animal access to streams, and diverting runoff to flow around barnyard or feedlot sites with concentrated wastes are effective practices (Maas et al., 1987).

Animal wastes may also be treated by aerobic or anaerobic digestion to concentrate the slurries and reduce the total volume of effluent to be spread on the land

(Brady, 1974; OECD, 1986; Phillips, 1986). Vegetative filter strips have also been used to effectively treat nutrient-enriched runoff from feedlots (Dickey and Vanderholm, 1981). One innovative recycling proposal suggests growing duckweed, to be used as feed for dairy cattle, in lagoons containing wastes from dairy cattle (Hillman and Culley, 1978).

Urban/Residential

Urban areas contribute a great diversity of NPS pollutants from the many different activities that take place in urban environments. The runoff from urban storm drains may carry large loads of pollutants--antifreeze, oil, various particulates, pesticides, fertilizers--that were originally deposited on city streets, parking lots, lawns, golf courses, and parks (Chesters and Schierow, 1985, Myers et al., 1985).

Urban areas may also contribute significant amounts of phosphorus (see Fig. 6 and Fig. 7). Contributions per unit area can approach one-third of a dairy or be nearly equal to intensively farmed agricultural areas (Kilmer and Taylor, 1980; Sawyer, 1973). Weibel and others (1964), using data from Seattle, reports stormwater runoff concentrations of phosphorus 200 times the accepted background level for streams. Phosphorus carried in runoff from urban areas primarily travels over impervious surfaces and has little opportunity to infiltrate and be adsorbed by soils. Often this runoff is carried directly into watercourses.

Atmospheric sources, lawn and garden fertilizers, animal wastes and **P** carried with sediments all contribute to the total phosphorus load from urban and residential areas, but the single largest source is phosphorus from the leaching of organic trash and debris (Waller and Hart, 1986). For example, leaf litter may lose as much as 30% of its dry weight through leaching within 24 hours of wetting (Cummins et al., 1972; Petersen and Cummins, 1974).

Construction sediments may also be a significant source of P. Disturbed soils on construction sites are not shielded by vegetation from the impact of raindrops, increasing the likelihood of the detachment and transport of soil particles as sediments. Sediments from construction are not retained by vegetation and their sediment delivery ratios may be very high. Depending on the characteristics of rainfall, soils, and topography in a basin the increase in sediment loads with development will vary. Increases associated with development have been reported from Japan by Kinosita and Yamazaki (1974) as 148% of predevelopment levels, from Quebec by Warnock and Lagoke (1974) as 300% to 500%, from Britain by Walling (1974) as 500% to 1000%, and from the Baltimore and Washington D.C. area of the U.S. by Chen (1974) as averaging 6000% for sites without erosion control. An estimated 600 million tons of sediment are lost annually from development sites in the U.S. (Lake, 1991).

a. Management Implications: Controlling Nonpoint Source Phosphorus from Urban/Residential Areas

The wide variety of nonpoint source pollutants from urban areas makes simple effective prescriptions for nonpoint source control impossible. Control of nonpoint source phosphorus depends on combinations of methods which limit sources and retain runoff water through a variety of means.

i. Leaf Collection. Collection of leaves and organic material greatly reduce the amount of leached soluble phosphorus and organic particulate phosphorus delivered to receiving waters (Waller and Hart, 1986).

ii. Sediment Control Measures. Sediment fences and traps used on constructions sites will partially control sediment-bound P. Since smaller particles, those most P-enriched will be more likely to escape, this control measure is a first step, not a method for complete control (Maas et al., 1987).

The construction of sediment retention basins that allow the settling of sediments and P-enriched materials are a second step in the treatment of P-laden runoff (Ellis, 1985; Field, 1985; Lake, 1991; Maas et al., 1987; Waller and Hart, 1986). Since most phosphorus-bearing materials in urban areas are light and easily moved, retaining the first half-inch of runoff has been shown to be effective for P control (Maas et al., 1987).

iii. Runoff Reduction. Impervious surfaces in urban environments prevent water from infiltrating through the soil and being adsorbed to soil particles. One opportunity for increasing infiltration is altering the physiognomy of grassed areas, providing swales for the collection of runoff (Maas, et al., 1987). Another opportunity developed within the last decade is a system of porous asphalt-concrete pavement underlain by a gravel base has been shown to be effective in increasing infiltration and reducing stormflow volume (Ellis, 1985; Field, 1985).

SUMMARY AND CONCLUSIONS

Phosphorus is an element that is not toxic, but which, when present in amounts greater than historic levels, may cause aquatic plants to grow at increased levels. Under these conditions the water quality for many uses declines.

Although the sources of **P** are diverse, amounts present in most systems have been relatively constant and most aquatic systems have developed in concert with these natural levels. Of concern are elevated levels caused by human activities. On a basin scale these include any land or water use that may introduce additional amounts of **P** or disturb the land surface allowing increased amounts to be transported to streams.

Control of eutrophication in systems where **P** is a limiting factor requires the control of **P** entering streams. Point source control is relatively simple, but may be insufficient to reduce **P** to desired levels. In this case the control of nonpoint sources also becomes necessary. For all land; agricultural, forested and urban; this requires controlling runoff and the materials it carries.

The dynamics of phosphorus are complex and it is an element which easily crosses boundaries, between both land uses and traditional disciplines. To come to a workable understanding of its interactions, investigations will also need to cross those boundaries. Phosphorus management requires a more integrated view, not only of its interactions in soil, or water, or plants, or stream biota, but of its interactions between these elements and across the landscape.

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