PHOSPHORUS IN FLORIDA'S ECOSYSTEMS: ANALYSIS OF CURRENT ISSUES

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ABSTRACT

Phosphorus is often one of the major nutrients limiting the productivity of terrestrial, wetland and aquatic ecosystems. In Florida, anthropogenic P loads from urban and agricultural activities in terrestrial ecosystems have increased the trophic state of associated aquatic systems to the eutrophic or hypereutrophic condition. Because many of these systems are hydrologically linked, the extent of eutrophication has increased through time as impacts have been transferred downstream. Although many of the fate processes within a specific ecosystem have been determined, the transfer of P across landscapes and between ecosystems is poorly understood. In this paper we present some key issues related to the sources, transformations, and transport of P within and across ecosystems, as related to surface water quality. Selected examples from various hydrologic units of Florida will be presented.

Phosphorus is imported into terrestrial ecosystems through fertilizers, organic wastes, wastewater, and animal feeds. Some of this P is exported out of the system as part of the product produced, some is exported to adjacent aquatic systems, but the majority remains within the system. Long histories of P rich material application have built up soil P levels in many
ecosystems, and the residual P may be sufficient to meet all the P requirements of the crops grown on these soils. Application of organic wastes based on crop nitrogen needs usually result in application of P in excess of crop needs. Soils contaminated with P as a result of intensive use or abandoned agricultural lands with long history of fertilization can continue to release P even after the farming activities on these lands are stopped.

Wetlands can function as sources or sinks for P, depending on the type of wetland and the loading rates. Most of the P added is retained within the system, resulting in accumulation of large reserves of P. Wetlands created on agricultural lands can function as sources of P for some period unless the bioavailable P is stabilized. Similarly, lakes can function as sinks for P and store large amounts of P in sediments, but their sediments can become a net source of P after a reduction in external P loading.

The major issues that need to be addressed are as follows: 1) What are the major sources of excess P?, 2) How can the controllable sources of P be reduced? 3) What processes transport excess P from one ecosystem to another?, 4) What are the transformations of excess P and how do they affect P bioavailability?, 5) What are the thresholds of P concentration and P load at which unacceptable ecological changes occur?, and 6) What will be the response of P-enriched ecosystems to a reduction in the P load? A brief discussion is provided on each of these issues using information available for some of Florida's major ecosystems.
Introduction

Phosphorus is one of the major nutrients limiting the productivity of terrestrial (upland), wetland and aquatic ecosystems. Florida's ecosystems are sensitive to anthropogenic nutrient loads and many of its aquatic systems are now eutrophic or hypereutrophic. The quality of water leaving one ecosystem can significantly impact the water quality of another ecosystem (Fig. 1). Accordingly, urban, agricultural, and environmental management practices implemented in one system have the potential to impact adjacent systems. For example, drainage water resulting from agricultural practices in the Everglades Agricultural Area (EAA) have created eutrophic conditions in the Water Conservation Areas (WCAs) of the Everglades. Similarly, nutrient loads from the dairy industry have moved Lake Okeechobee toward hypereutrophic conditions. Holistic management practices should consider such indirect impacts when developing water quality improvement plans.

The objectives of this paper are to (1) present an overview of key issues related to P management in Florida's ecosystems as related to surface water quality, (2) provide examples of the experimental data developed to address these issues, and (3) identify key research and monitoring needs for integrated P management.

Wetlands and aquatic systems are important natural resources because they provide habitat for diverse flora and fauna, and water for many agricultural, domestic, and industrial activities. The water quality of these ecosystems is affected by land use, basin hydrology, geology, and water management practices. Although many point sources of P discharges to wetlands and aquatic systems have been controlled or reduced, nonpoint sources through surface and sub-surface flow pose a greater danger in increasing loads to adjacent water bodies (Sharpley, et al., 1994). The USEPA has identified agricultural operations as one of the major sources of P to lakes, rivers, and
estuaries (Parry, 1998). To abate this problem, many state and federal agencies are in the process of developing watershed management strategies to reduce P loads into the water bodies. For example, the International Joint Commission between the U.S. and Canada effectively implemented several P management strategies to reduce loads to the Great Lakes (Rohlich and O'Connor, 1980). Similarly, the State of Florida has implemented several management strategies to reduce P loads to Lake Okeechobee (Bottcher, et al., 1998). The USEPA is in the process of developing comprehensive national strategy to control nutrients from nonpoint sources (Parry, 1998).

Florida encompasses a surface area of approximately 149,900 km² with 36% occupied by forested land, 9% rangeland, 12% cropland (row crops, vegetables, citrus, sugarcane, and others), 15% pasture, 8% urban, and 27% wetlands and aquatic systems (Bottcher et al., 1998). Based on river basins, Florida can be divided into five regions: the Northwest region (covers area of the Panhandle of Florida), the Suwannee River Region, the Southwest region, the St. John's River Region, and the Kissimmee - Okeechobee - Everglades Region. Five Water Management Districts (WMD's) have been established by the State of Florida to manage the water quantity and quality of these regions (Fig. 2). Prominent examples of the State's water resources within these regions are the Kissimmee-Okeechobee-Everglades system, the Florida Keys, the Indian River Lagoon, Wakulla Springs, the Green Swamp, the Apalachicola River and Bay, the Suwannee River, and the Floridian and Biscayne aquifers.

Florida's water resources are considerable. Due to its flat topography and high rainfall, 27% of its surface area is wetlands, lakes, and streams (Bottcher et al., 1998). It ranks third in the United States in precipitation with an annual average rainfall of 1350 mm/yr and has more available groundwater, proportionally, than any other state. Florida has 27 first-magnitude springs, more than any other state, more than 1,700 streams and rivers, and about 7,800 freshwater lakes
(Fernald and Patton, 1984). With their associated uplands, these water resources support productive and economically significant ecosystems. For example, sport fishing alone is valued at $500 million annually in the Indian River Lagoon (Florida Sea Grant, 1993). Florida’s biodiversity is nationally recognized. It ranks among the top three states in the continental U.S. in the total number of species of plants and animals (Florida Biodiversity Task Force, 1993). This biological wealth attracts many nature enthusiasts, and the combined economic value of consumptive and non-consumptive water uses has been estimated at $5.2 billion statewide (Cox et al., 1994).

Cultural eutrophication is a major threat to Florida’s valuable water resources. As a result of P enrichment, many of Florida’s lakes are moving from oligotrophic to either mesotrophic or eutrophic conditions (Canfield, 1981). Eutrophication of natural wetlands is also a serious concern in Florida. Phosphorus enrichment also converted many oligotrophic wetlands into eutrophic wetlands, resulting in alteration of plant and microbial communities, increased productivity, and nutrient accumulation. For example, the effect of P loading on wetland eutrophication is clearly evident in the Everglades Water Conservation Areas (SWIM, 1992; Davis, 1994) and in wetlands of the Upper St. Johns River Basin (SJRWMD, unpublished). Other prominent and valuable Florida ecosystems threatened by, or already damaged by increases in the supply of P and other nutrients include the Florida Everglades, Lake Okeechobee, Lake Apopka and the Harris Chain of Lakes, Tampa Bay, Florida Bay, and Apalachicola Bay.

Florida’s five water management districts (Fig. 2) were given the primary responsibility to resolve the ecological problems of several major aquatic ecosystems by the 1987 Surface Water Improvement and Management Act, the Lake Apopka Restoration Acts of 1985 and 1996, and the 1994 Everglades Forever Act. The requirements of these laws have stimulated research by the water management districts, state and federal agencies, universities, and private industry to develop
an understanding of P dynamics in Florida's ecosystems and to develop methods to reduce P loads. These research projects have studied how physical, biological, geological and chemical factors interact to regulate the fate and transport of P in soils/sediments and water components of terrestrial, wetland and aquatic ecosystems (e.g., St. Johns River Basin, Oklawaha River Basin, Lake Apopka, Lake Okeechobee Basin; Lake Okeechobee, the Everglades, and Florida Bay).

The major issues confronting water managers are: 1) What are the major sources of excess P?, 2) How can the controllable sources of P be reduced?, 3) What processes transport excess P from one ecosystem to another?, 4) What are the transformations of excess P and how do they affect P bioavailability?, 5) What are the thresholds of P concentration and P load at which unacceptable ecological changes occur?, and 6) What will be the response of P-enriched ecosystems to a reduction in the P load? In this paper, we will consider each of these issues using information available for some of Florida's major ecosystems.

1. What are the major sources of excess P?

Most of the P that enters aquatic ecosystems stems from activities in the upland portion of the drainage basin. Phosphorus is added to uplands in fertilizers, organic solids (sewage sludge, animal wastes, composts, and crop residues), wastewater, and feeds (Table 1). Some of this P is exported out of the drainage basin as a part of the product produced, but the majority remains within the basin and can contribute to the eutrophication of streams, lakes, and estuaries, (Boggess et al., 1995; Sharpley et al., 1996)

Approximately 42,660 Mg of fertilizer P was used during 1996 in Florida. Fifty percent of this was applied within the boundaries of SFWMD, 20 % in the SJRWMD, 17 % in the SWFWMD, 8% in the NFWMD, and 5% in the SRWMD (FLDACS, 1996). Fertilizer P is
primarily in inorganic form, which is bioavailable and can be a major source of P for many ecosystems. For example, fertilizer P accounted for 51% of P imports to the Okeechobee Basin (Boggess et al., 1995). Unfortunately, similar estimates are not available for other drainage basins, despite the potential importance of this input.

Feed supplements are another significant source of P. Although statewide estimates of the magnitude of their use are not available, they can be a major contributor of P. For example, in the Lake Okeechobee drainage basin 49% of the P load to the lake stemmed from feeds used by the dairy industry (Boggess, et al., 1995).

Organic solids are in the form of biosolids (commonly known as 'sewage sludge'), animal wastes, composts, and crop residues. Florida produces about 253,000 dry tons of biosolids. About two-thirds of the supply is applied to land, and the remainder is disposed in land fills and incinerators. The P content of biosolids is in the range of 0.4% to 5.3% (mean=1.8% dry weight basis) (Obreza, 1997). In biosolids, only 10% to 30% of the total P is present as organic P (Wolf and Baker, 1985). Assuming about two-thirds of biosolids produced are applied on land, the estimated P added from this source to Florida's ecosystem is approximately 3,040 Mg/P/yr.

In 1995, about 22 million Mg of MSW was produced in Florida; 55% of this waste can be composted (Hinklely and Goven, 1996). In 1992 about 0.5 million Mg of animal manure and 2.8 million Mg of yard trimmings were available for composting (Cooperative Extension Office, IFAS, University of Florida). Phosphorus available in animal manures obtained from various livestock operations in Florida was estimated at 4,700 Mg/yr (Landers et al., 1998). The MSW, animal manure, and yard waste can potentially produce approximately 5.4 million Mg of usable compost. Phosphorus in organic solids occurs as both inorganic and organic forms. The proportion of these forms in organic solids is important to P availability for plant uptake and for transport in leaching
and surface runoff. Data obtained for different types of manures indicate that the ratio of soluble P to total P is in the range of 0.24 to 0.70, suggesting that a significant portion can be associated with manure particles (Reddy, et al., 1978).

The volume of municipal wastewater discharges has been about 6.76 million m³/day, with about 16% applied to land, and 17% discharged through deepwell injection. Assuming an effluent total P concentration of 2 mg/L, the estimated P load is about 13,520 kg P/day (4,930 Mg P/yr). Approximately 50% are discharged into bays, rivers, and wetlands, while the remainder goes to the ocean.

Phosphorus inputs through atmospheric deposition are estimated to contribute about 20-80 mg P/m² yr into Florida (Redfield, 1998). At this rate, annual load to Florida’s ecosystem is estimated to be approximately 3,000 to 12,000 Mg P/yr. Rates below 30 mg P/m² yr are found near coastal and remote areas, 30 to 50 mg P/m² yr are found in mixed land uses, and >50 mg P/m² yr are associated with urban and agricultural areas (Redfield, 1998). Because of the uncertainties associated with these estimates, we used a more conservative value of 40 mg P/m² yr for P input through atmospheric deposition. Phosphorus added through this source is significant and is higher than P added through waste materials (Table 1). Under natural conditions, background levels of P can also be contributed by parent material through weathering and dissolution of rocks and minerals of variable solubilities. For example, high background levels of P in surface waters in many areas of the southwest region of Florida is due to dissolution of P from native phosphatic minerals. In order to assess the P impacts, we need to know the relative contribution of these natural sources as compared to P imports through fertilizers, organic solids, and feeds. This should be given serious consideration, when determining P budgets and in developing management regulation and strategies to restore ecosystems.
2) How can the controllable sources of excess P be reduced?

The above analysis shows that most of the major sources of P are controllable. How, then, can P use be reduced while still accomplishing its intended purposes? For fertilizers, there is good evidence that application rates are often higher than crop requirements for optimum productivity. Long histories of fertilizer and other P-rich material application has built up soil P levels in many ecosystems, and the residual P may be sufficient to meet some or all the P requirements of the crops grown on these soils. In many areas of Florida, soil test P values exceed crop requirements. For example, organic soils used for vegetable production in the Lake Apopka Basin have sufficient available P, and crops grown on these soils do not respond to application of P fertilizers. Several best management practices (BMPs) are now recommended to improve the P use efficiency by crops and to reduce P in runoff and drainage (Bottcher et al., 1998; Izuno and Whalen, 1998). One BMP is to use soil tests to determine P fertilizer needs, rather than relying on historic application rates.

Agricultural scientists, and some farmers, use plant and soil analyses to determine the P status and requirements of crops. The most common approach is to use soil test procedures, which involve extraction of soils with selected chemicals. The amount of P extracted is related to crop yields to determine the P fertility of soils. These relationships have been developed for various crops and soil types. For mineral soils (Entisols, Spodosols, and Ultisols), soils are extracted with Mehlich I \((0.025 \ M \ H_2 \ SO_4 + 0.05 \ M \ HCl)\) reagents for 5 min. (soil to solution ratio: 1:4), and filtered solutions are analyzed for soluble P. For organic soils (Histosols), soils are extracted with water (soil to solution ratio of 1: 100), and filtered solutions are analyzed for soluble P. Such tests, however, are not directly based on the capacity of soils to retain or release P, and its ultimate effect on the surrounding environment.
Extensive soil testing on mineral soils has shown that many crops do not respond to additional P fertilization, if Mehlich I-P levels are >31 mg P/kg (Kidder et al., 1997). At Mehlich I-P levels of <10 mg P/kg P fertilization is recommended at a level needed to optimize crop yields, and at P levels between 10-30 mg P/kg, lower rates of fertilizer P are recommended (Kidder et al., 1997). For pastures (bahiagrass) grown on flatwood soils, decreasing fertilizer P application rates from 48 to 24 kg P/ha yr did not decrease forage yields (Fig. 3) (Rechcigl and Bottcher, 1995). Although these soils contained Mehlich I-P in the range of 9-13 mg P/kg soil, they did not respond to added fertilizer P, suggesting that further validation is needed between soil test values and crop response. Despite the poor responses of crops to additional P applications, and recommendations to reduce P fertilization, many farmers continue to apply P at rates beyond crop needs (Kidder et al., 1991). Over 50% of the fertilizer P purchased in 1996 is used in the South Florida Water Management District Region, where aquatic systems are sensitive to P loading.

Acceptable land application rates of organic wastes are usually calculated by consideration of the fate and transport of N in the given soil-water-plant system. In Florida, because of the P-sensitivity of many aquatic systems, serious consideration should be given to application of organic solids based on the fate and transport of P. In many cases, applications of organic solids based on crop N needs result in application of P in excess of agricultural requirements, resulting in adverse impacts on surface and groundwater (McCoy et al., 1986). For example, the average N content of biosolids is about 3% of the dry weight (range = 0.6-7.5%), and the average P content is about 1.8% (range=0.4 - 5.3%). This low N/P ratio means that when the land application rates of biosolids are based on the N content, there is a potential for creating water quality problems in land areas with soils possessing poor P retention capacity. On the other hand, if the application rates are determined based on P needs, rates would be much lower and supplemental N may need to be
applied as inorganic fertilizers to meet the N demand. Organic waste loading rates should be based on site soil characteristics, the bioavailability of N and P, and the hydrologic characteristics of the site.

Another critical issue in application of P-rich substances is to what extent the added P is retained by soils. Unlike C and N, much of the P added through fertilizers and organic solids is retained within the system. In many of Florida's upland ecosystems, the long-term use of the land by agricultural and animal industries has increased soil P levels. The soil P content is higher in surface soil than subsoil. Soil total P content can be highly variable. For mineral soils, (soil bulk density of about 1.5 g/cm\(^3\)) total P content can range from 20 to 2000 mg P/kg, and in organic soils, (soil bulk density of 0.1-0.3 g/cm\(^3\)) total P content can vary from 300 - 1500 mg P/kg. For example, in the Okeechobee Basin, the total P content of surface soil (A horizon) was 1900 mg/kg in intensive areas and holding areas of cattle (Graetz and Nair, 1995). In the same soil profile, subsoil total P concentrations were in the range of 150-180 mg P/kg. In natural areas and lands used for pasture, soil total P in surface horizons was in the range of 30-150 mg P/kg (Graetz and Nair, 1995).

The amount of P accumulation depends on the capacities of soils to retain P through adsorption and precipitation reactions. In mineral soils (Spodosols, Entisols, and Ultisols) inorganic P typically constitutes 50-90% of total P, whereas in organic soils (Histosols), inorganic P typically constitutes only 10-50% of total P. Inorganic P forms are associated with amorphous, and poorly crystalline Fe and Al in acid soils, and with Ca compounds in alkaline soils. Organic P forms are associated with sugar phosphates, nucleic acids, adenosine phosphates, phospholipids, inositols, and fulvic and humic acids.
Long-term application of P can decrease a soil's capacity to retain P. Soils have finite capacity to retain P through sorption and precipitation reactions (Berkheiser et al., 1980; Rhue and Harris, 1998). Phosphorus sorption processes have been extensively studied using various soil types and soil minerals. General conclusions from these studies are that P sorption in acid mineral soils can be correlated to amorphous and poorly crystalline forms of Fe and Al (oxalate extractable Fe and Al) and crystalline forms of Fe and Al (citrate-dithionate-bicarbonate (CDB) extractable Fe and Al). In alkaline-calcareous soils, P retention is associated with Ca-minerals. Several studies have been conducted on Florida's soils to determine P retention capacities (Yuan and Lucas, 1982; Burgoa, 1989; Harris et al., 1996; Nair et al., 1998). Using various soils, Harris et al. (1996) developed an index of relative P adsorption (RPA) capacities. The RPA is defined as the ratio between the absolute amount of P sorbed to the maximum amount of P that can be retained. This ratio has been used to characterize selected Florida soils for their P retention capacities (Table 2). Surface horizons of Spodosols and Entisols (clean sands) have RPAs in the range of 0.05-0.26, as compared to Utisols and Entisols (with coated sands), with RPAs of 0.74 and 0.48, respectively. Most of the Spodosols are in south and central Florida where soils are intensively used for agriculture, and many of these areas are considered to be potential sources of P contamination. For example, in the Okeechobee Basin, Spodosols in areas intensively used for dairies and beef pastures showed very little or no P retention. Low P sorption capacities were reported for A and E horizon of Spodosols and relatively high capacities in subsurface Bh horizon (Nair et al., 1998).

Although soils can accumulate P as a result of continual loading, their capacity to buffer dissolved P in the soil porewater may decrease with time, resulting in elevated P levels in the drainage water. This effect can be evaluated by determining the EPCo (equilibrium P concentration, mg/L), for a soil (Fig. 4). A soil retains P only if the influent water P concentrations
is > EPCo for that soil. If the water moving through the soil has a P concentration of < EPCo, the soil releases P. Earlier studies with soils loaded with manures showed that increased P loading decreased soil P retention capacity, and increased EPCo (Reddy et al., 1978). Soil test procedures (Olsen, Bray –1, and Mehlich I) provided reasonably good estimates of labile P, EPCo, and algal-available P, for a diverse group of noncalcareous agricultural soils (Wolf et al., 1985). For Spodosols in the Okeechobee Basin, Nair et al. (1998) showed high EPCo for soils impacted by P loading (Table 3).

Soils in upland ecosystems are not adequately characterized with respect to their long-term capacity for P retention or release. Results available at this time cannot be spatially extrapolated to watershed or basin scales unless relationships between P retention characteristics and easily measurable soil properties are developed. Data on easily measurable soil properties such as organic matter content, and extractable P, Fe, Al, and Ca, can be obtained in a cost-effective manner for a large number of sites within the ecosystem.

In addition to chemical processes, P retention by soils is also regulated by several biological processes including uptake and release by higher plants and microorganisms. In P limited soils, a significant portion of P can be tied up in organic pools, and the turnover of this pool through the activities of extracellular enzymes can play a significant role in regulating the bioavailable P. Forms of organic P, and their breakdown in upland soils are discussed in detail by Newman and Robinson (1998) and Wetzel (1998). Phosphorus retention in organic pools, especially as inositol phosphate and P bound to fulvic and humic acids, represents a stable, long-term P sink. Biological processes can, thus, be very important in regulating mobility of P especially in organic soils, and surface horizons of mineral soils.
Upland systems can be managed for maximum P retention especially when soils are used for land application of organic wastes. Some of these waste materials contain metallic cations that can increase the overall P retention capacity of the soil. Wastes low in metallic cations can be amended with P binding chemicals (such as alum, ferric chloride or lime) before their application on the land (Moore and Miller, 1994). Additional research is needed to determine the utility of chemical amendments in reducing the bioavailability of pools of P in organic wastes. As discussed earlier, long term application of fertilizers and wastes increases the total and bioavailable P of soils. The stability of this stored P depends on soil physico-chemical properties, management practices, and hydrology. Limited data available for selected sites is not sufficient to extrapolate to landscape level. Studies suggest that a large proportion of P stored in upland soils is in stable forms (Graetz and Nair, 1995). However, it is critical to determine the fraction of P that is mobile or released into water.

3) What processes transport excess P from one ecosystem to another?

Uplands:

The majority of Florida's P load is applied to upland ecosystems. Phosphorus from mineral upland soils can be transported via surface and subsurface flow. In poorly drained flatwood soils, both surface and subsurface runoff (lateral flows) can be important, especially during heavy rainfall events. Where the spodic horizon is shallow, there is greater potential for P transport through surface runoff (Campbell et al., 1995). Areas farmed on organic soils (Lake Apopka Basin, the Everglades Agricultural Area) are typically artificially drained through a network of drainage canals. Phosphorus transport in these soils primarily occurs through subsurface flow. Although much of the P added to an upland ecosystem is retained within the soil, the amount of P that leaves
the system through surface and subsurface flow is sufficient to affect the water quality of adjacent wetlands and aquatic ecosystems. The amount of P leaving an upland system is affected by a number of factors including, soil type, hydrology, land use, and management practices. Measurement of P outputs is expensive, and often P discharges of only a few of the watersheds are adequately characterized. For example, P discharges from the EAA and vegetable farms adjacent to Lake Apopka have been quantified. Simulation models have been used to estimate P transport from the Okeechobee Basin (Campbell et al., 1995). Our understanding of P transport has improved through the development of database and simulation models, especially for upland ecosystems in the St. Johns River Region and Kissimmee-Okeechobee-Everglades Region. However, such a database is not available for other regions of Florida. Phosphorus transport can be quantified and predicted through the use of simulation models, but only after thorough testing and validation with experimental data.

Soils with high levels of P as a result of intensive use, such as abandoned agricultural lands with long histories of fertilization, can continue to release P even after the farming activities on these lands stop. For example, land use activities around dairies and beef ranches in the Lake Okeechobee Basin have resulted in P enrichment of surface soils, especially in soils of intensive areas and holding areas surrounding milking barns of dairies. Several of these dairies in the Lake Okeechobee Basin have been bought by the State of Florida, in order to reduce P inputs to the lake. Agricultural lands in central and south Florida have also been acquired by state agencies to protect adjacent aquatic systems. Some examples are vegetable farms in the Oklawaha River Basin, Lake Apopka Basin, Upper St. Johns River basin and the Everglades. Some of these acquired lands have been converted into wetlands for habitat restoration and water quality improvement. However, long
term farming in these areas created P enriched soils that can release P into drainage water for long periods of time after farming has ceased.

Wetlands:

At the landscape level, wetlands often form a critical interface between uplands and adjacent water bodies, as all of these ecosystems are hydrologically linked. Consequently, a thorough understanding of P dynamics in wetlands is critical to understanding the processes which transport P from uplands to lakes and streams. Water and associated contaminants (such as P) are transported from uplands by either subsurface or surface flow. Surface flow can include first-, second-, and third-order streams as well as the associated riparian floodplains, marshes and swamps. In low gradient systems, streams are largely composed of interconnected marshes and swamps. To improve drainage in agricultural areas, ditches are often cut to connect isolated wetlands. The resulting flow of water in the drainage basin follows a complex path through wetlands, ditches and streams. Thus, P loading to the receiving aquatic system depends on the retention capacity of several components of the basin.

Wetlands can function as sources or sinks for P, depending on the type of wetland and the loading rates. Most of the P added is retained within the system, resulting in accumulation of large reserves of P. Thus, retention is defined as the capacity of wetlands to remove water column P through physical, chemical and biological processes, and retain it in a form not readily released under normal conditions. Retention of P by wetlands decreases the load to downstream aquatic systems (Reddy, et al., 1996). Wetlands not only store P, but also transform P from biologically available forms into non-available forms and vice-versa. Thus, it is important to include the contribution of wetlands in retaining P when developing best management practices for a drainage
basin. Land use practices in uplands, along with processes occurring in wetlands, should all be considered in nutrient management options for a water body.

Wetlands, as low lying areas in the landscape, receive P inputs from all adjacent uplands (agricultural and urban activities). If the integrity of an upland area is compromised, it is likely that it will soon be reflected in the integrity of associated wetlands. However, if the integrity of the wetland is compromised the effect may not be immediately reflected in the condition of the upland, since materials transfer is largely towards the wetlands. The response of a wetland to P inputs varies and depends on the wetland type (e.g., forested, marsh with emergent macrophytes), as compared to lakes and streams which respond to P inputs more rapidly due to more mixing. For many natural wetlands, external P loads (especially nonpoint sources) have not been characterized. Phosphorus inputs from the EAA and other nonpoint sources into adjacent natural wetlands (Water Conservation Areas 1, 2a and 3a) have been estimated to be about 347 Mg P/yr, and rainfall inputs to these areas have been about 272 Mg P/yr (SWIM, 1992). On an areal basis, loading rates from the EAA and rainfall were: 0.23, 0.25 and 0.13 g P/m² yr for WCA-1, WCA-2a, and WCA-3a, respectively (SWIM, 1992). Phosphorus inputs to wetlands located in the lower Kissimmee River and Taylor Creek/Nubbin slough watersheds were estimated to be in the range of 2.8 - 4.0 g/m² yr (Reddy, et al. 1996).

Natural and constructed wetlands intentionally used as buffers to retain P are usually managed to improve their overall performance, and to maintain wetland integrity. The extent of management required depends upon the P retention capacity of the wetlands and the desired effluent quality. Management scenarios can vary, depending on the the type of wetland. For example, wetlands used for municipal wastewater treatment are usually small. They can be
managed efficiently by altering the hydraulic loading rate or integrating them with conventional treatment systems. Large-scale systems can be managed by controlling P loads.

A wide range of P loads (0.1 - 1000 g P/m² yr) is used in treatment wetlands (Kadlec and Knight, 1996). Analyzing water quality data from several treatment wetlands, Kadlec and Knight (1996) observed that an increase in the P loading rate increased the effluent P concentration. Richardson et al. (1997) in their analysis of water quality data from treatment wetlands, showed that at P loading rates of approximately 1 g P/m² yr, the effluent P concentration was approximately 40 μg/L. However, the data presented by Richardson et al., (1997) shows a high degree of variability between loading rates 0.1 to 10 g P/m²m yr, suggesting that it is too simplistic to establish one loading rate to determine the lower limit of wetlands to remove P. Clearly wetlands can not reduce P levels below background levels. Although simple input/output analysis found in the literature provides some general guidance relative to P loading rates and effluent P concentrations, they can not be directly applied to a particular wetland, without due consideration of site-specific conditions.

Phosphorus added to wetlands is retained through biotic processes such as assimilation by vegetation, periphyton, and microorganisms, and through abiotic processes such as sedimentation, sorption and precipitation. Steady P loading increases the P content of all components of wetlands (vegetation, soil, and periphyton), and results in distinct horizontal and vertical gradients in soils and water columns (Koch and Reddy, 1992, DeBusk et al., 1994). Phosphorus loading also increases bioavailable P pools in soil, and decreases soil capacity to buffer porewater P at the original level. Addition of P also increases microbially mediated processes, resulting in short-term storage of P in microbial cells as polyphosphates. In P enriched wetlands, a significant amount of stored P is in labile pools, as low C/P ratios of detrital tissue favor rapid decomposition and release.
The lability and stability of detrital tissue is an important factor regulating wetland P dynamics as cycling of detrital material can maintain eutrophic conditions in a wetland, even after external loads are curtailed. These conditions can increase the EPCw (equilibrium P concentration between soil and water column) of the soil, which determines the direction of P flux between soil and overlying water column (Fig. 5). If the inflow concentrations are decreased below the EPCw (as a result of reduction in P loads), wetland soils could function as a steady source of P to the water column until a new equilibrium is attained.

When agricultural lands are converted into wetlands, the residual fertilizer P stored in these soils can be rapidly released upon flooding. For example, vegetable farms in the Lake Apopka Basin converted into wetlands released P into the overlying water column, even 32 months after flooding (Table 4). In soils of the Lake Griffin flow-way, up to 30% of the total P was present in bioavailable pool that can be potentially released into the water column (Reddy, et al, 1997). Even after 30 weeks after flooding, soils from the flow-way maintained an average EPCw value of 0.22 mg P/L, suggesting that these soils cannot reduce effluent P levels below 0.22 mg P/L (Reddy et al., 1997). However, filtration of particulate P can cause removal of total P below this value if the sedimentation rate exceeds the rate of P release (SJRWMID, unpublished).

Bioavailable P in agricultural lands converted to wetlands can be stabilized through application of phosphate binding chemicals such as alum, $\text{Al}_2(\text{SO}_4)_3$ ferric chloride (FeCl$_3$), and lime (Ca(OH)$_2$) (Ann, 1996). In addition, establishment of macrophyte communities may stabilize soil porewater P, and reduce overall P flux to overlying water column.

Phosphorus is often one of the major nutrients limiting the productivity of wetland ecosystems. Much of the P added to wetlands is retained within the system, resulting in accumulation of large reserves in detrital tissue and soil, which can serve as a P source for a long
period of time, even after external loads are reduced. Similarly, wetlands created on agricultural lands can function as sources of P for some period until the bioavailable P is stabilized. When considering wetlands for storing P, several issues needed to be addressed: (1) what was the natural P loading rate and resultant trophic state, (2) what is the relative bioavailability of various forms of P in wetland soils and lake sediments, (3) how stable is the stored P and under what conditions will it be released back into the water column, (4) what is the long-term assimilative capacity of these systems, and (5) how long can the stored P maintain eutrophic conditions, once external loads are curtailed.

Aquatic systems:

Aquatic systems, such as lakes, are the final recipients of P discharged from adjacent uplands and wetlands. As P moves through uplands and wetlands, it undergoes various biogeochemical transformations, and most of it remains within these systems. In terms of the P mass balances (of upland ecosystems), the amount of P transferred to lakes may not be significant. However, these relatively small losses may be enough to create eutrophic conditions in adjacent lakes. Addition of P to lakes increases algal productivity and decreases dissolved oxygen and biodiversity, and changes in these parameters are often used as indicators of eutrophication. Since algae can obtain C and N from the atmosphere, P regulates the growth of algal biomass, and ultimately the trophic state of the lake. Phosphorus and other nutrients stored in algal biomass are cycled within the water column during decomposition, and the P associated with recalcitrant dead algal biomass settles and becomes an integral part of the sediments. Although sediments usually function as a major storage reservoir of P, they can at times function as sources of P to the water column.
The P concentration in lakes depends on inputs from tributaries, groundwater and the atmosphere, interactions between the bottom sediments and the overlying water column, exchanges between the vegetated (littoral) and open (limnetic) zones, and internal biogeochemical processes in the sediment and water column. Management strategies for the restoration of lakes must address several issues: (i) what was the natural loading of P to the lake and the resultant trophic state? (ii) what is the relative bioavailability of various forms of P in the sediment and water column? (iii) what is the relative contribution of P loads from internal sources as compared to external sources? and, (iv) what is the time span required for these systems to reach their background condition, after external loads are curtailed.

4) What are the transformations of P and how do they affect P bioavailability?

Phosphorus is discharged into surface waters in both organic and inorganic forms. The relative proportion of each of these forms depends on the P source, the soils in the drainage basin, and the land uses. For example, drainage and runoff water from areas with organic soils can have a greater proportion of total P in organic form than inorganic forms. Conversely, runoff water and drainage from mineral soils contain P mostly in inorganic form, with large amounts associated with suspended sediments. Some of the issues related to P concentrations of water discharged from uplands are (i) what is the ratio of inorganic P to organic P? (ii) what proportion of total P is present in bioavailable form, and (iii) what proportion of total P is stable and resistant to biological breakdown?

Continuous, long-term application of P also increases the proportion of P which is bioavailable. Soil P undergoes various transformations as it cycles through inorganic pools (Fe and Al, Ca and Mg associated minerals) and organic pools (through plants, animals, microbes, and soil
organic matter). Predicting these transformations requires a detailed understanding of the soil characteristics, chemical equilibria, and rates of transformation of organic and inorganic pools, and how P availability is affected by various management strategies. With respect to water quality, the amount of inorganic and organic P present in a bioavailable pool is critical, since these pools of P can be readily mobilized. However, there is no single suitable method to quantify the fraction of total P that is bioavailable P. The bioavailable P is defined as a form of organic and inorganic P that can be readily utilized by biota including higher plants, algae, and microbes. Numerous chemical and biological methods have been used to determine bioavailable P in soils (Sharpley, 1991, 1993), but many of these methods yield results that are 'operationally defined'. Analysis of selected surface water samples from the Upper St. Johns River Basin indicated that about 70% of total P was bioavailable as estimated by the Fe-oxide strip method and the acid hydrolyzable method (Fig. 6; Reddy, K.R. unpublished results).

Floodplain areas, which have long been drained and farmed, are now being acquired by state and federal agencies with the intent to restore lost habitat, and to reduce the pollution of adjacent water bodies. In many cases, the management goal for these areas is to recreate wetlands or, if soil subsidence has been severe, to create lakes. Reflooding of these lands, however, can temporarily create water quality problems as the residual fertilizer P, and the P released from oxidation of organic material, is rapidly released into the water column. With time after flooding, hydrophytic vegetation becomes established. Uptake of P by plants, and microbial processes gradually reduces the bioavailable P. In Florida, agricultural lands currently being considered for conversion into wetlands are dominated by organic soils. For example, farmlands proposed for conversion into treatment wetlands (15,000 ha) to buffer agricultural drainage water from the EAA have organic soils used for sugarcane production. Similarly, several thousand hectares of
agricultural lands with organic and mineral soils are flooded in central Florida to create wetland habitats. The unique soil conditions created by flooding influences the transformation and availability of both soil organic and inorganic P. Earlier studies have shown that flooding and anaerobic decomposition processes increased the rate of soluble P production (Reddy, 1983, D'Angelo and Reddy, 1994a, b), and soluble N and C (Reddy, 1982), thus adversely impacting surface water. In contrast, macrophytes and periphyton can also convert labile inorganic P into organic P through uptake and storage, stabilize the soil porewater P, and reduce the P concentrations of surface water. However, long-term stable storage of P in organic pools depends on the quality of detrital material accreted on the soil surface.

5) What are the thresholds of P concentration and P load at which unacceptable ecological changes occur?

In P-limited wetlands and lakes, the P concentration of surface water is very low, as added P is rapidly assimilated into biota and adsorbed onto suspended particles. The small fraction of bioavailable P undergoes rapid turnover, and is efficiently used by the organism present in the water column. As the system is loaded with P, the surface water P concentration increases only after the P uptake by organisms reaches saturation level. Threshold P concentrations that cause ecological changes in lakes are fairly well known (Correll, 1998), but very limited information is available for wetlands. The threshold P concentration for a lake or a wetland depends on the capacity of the system to assimilate P without causing significant ecological changes. However, many systems are weakly buffered and the biotic communities respond to P loading rapidly. In the Everglades, microbial and algal communities responded rapidly while responses of macrophytes followed later (McCormick and O'Dell, 1996). For the Everglades WCAs, substantial changes in
periphyton communities have been noted at a water column total P concentration of 10-30 µg/L (Vymazal and Richardson, 1995; McCormick and O'Dell, 1996). Similarly, addition of P to an oligotrophic or mesotrophic lake can result in shift of plankton species, with later shifts by other organisms.

The phosphorus concentration of the water column is often used to assess the trophic status of a wetland or a lake. So, how reliable are the established concentration limits in determining threshold for wetlands and aquatic systems? The Vollenweider model, commonly used for lakes, takes mass loadings into consideration (Vollenweider, 1976). If P concentration is used, Correll (1998) points out that total P should be used as criteria rather than dissolved reactive P (DRP), because of its rapid turnover in the water column. In eutrophic lakes such as Lake Apopka, the DRP levels are < 5 µg/L, while total P concentrations range from 100-200 µg/L (Reddy and Graetz, 1990). The low DRP concentration in this lake reflects rapid turnover resulting in high planktonic activity in the water column.

Surface water total P concentration of 165 Florida lakes surveyed ranged from 3 to 834 µg/L (Canfield, 1981), with total P concentration of 75% of the lakes in excess of 10 µg/L. Consequently, the majority of these lakes are classified as either mesotrophic or eutrophic. There is evidence, however, that the trophic state of subtropical lakes is lower than that of temperate lakes for any given level of P (Salas and Martino, 1991).

Most natural wetlands are not used for reducing P levels of agricultural or urban runoff water. However, in those cases where they have been, eutrophication has become a major issue, because of adverse impacts on wetland plants and animal communities.

Because most interest in wetlands with respect to P has been in its removal, little attention has been paid to the effects of P on wetland trophic state. As a result, the conceptual basis for
management of wetlands lags well behind that for lakes. Lowe and Keenan (1997) point out that P
effect in wetlands are patterned and localized unlike effects in lakes which are unpatterned and
generalized. They propose that management of eutrophication of wetlands should, therefore, focus
on managing the size of the zone effects rather than the P concentration. In many cases the extent
of impacts resulting from P loading have been difficult to measure, with the notable exception of
the Everglades. In developing P loading strategies, we need to address the following issues: (i)
what are the impacts of P loading on natural wetlands?, (ii) are some wetlands more tolerant (or
have greater assimilatory capacities) than others and why?, (iii) what level or size of impact is
acceptable?, (iv) can we reverse eutrophication effects and restore wetlands to their original
condition?, and (v) to what extent is it legally permissible to load P into wetlands?

At present, state and federal agencies are actively involved in restoration of the Everglades
through various P control strategies (Izuno and Whalen, 1998; Moustafa et al., 1998). The
Everglades restoration program should provide insights that can be applied to other wetland
restoration programs in Florida and other sub-tropical regions of the world.

6) What will be the response of P-enriched ecosystems to a reduction in the P load?

Long-term P loading saturates the biotic (uptake by microbes, algae, and higher plants) and
abiotic (sorption on particulate matter) components of wetland and lakes, resulting in increased P
levels in the water column and underlying soils or sediments. The response of ecosystems to
reduction in external P loads depends on their internal reserve of P and its stability and
bioavailability. Thus, characterization and identification of internal P reserves is essential to
determine the time required for recovery.
Once external loads are curtailed, P enriched wetland soils are exposed to an overlying water column with low P concentration. This creates steep gradients in the soluble P concentration between soil and water column. Phosphorus enriched areas have a larger mass of bioavailable soil P than unimpacted areas. For example, when soil cores from impacted areas of the Everglades Water Conservation Area 2a were flooded with low P (<5 µg P/L) water, rapid P flux into overlying water was observed (Fig. 7). It is estimated that the impacted soil can sustain this rate of P flux for about 5 years. This estimate was based on the assumption that about 25% of the total P in the top 30 cm of soil is mobile and potentially can diffuse into the overlying water column. At present, there are major data gaps exist for determining the factors that regulate the recovery of P-impacted wetlands to their historical condition.

Many of Florida's lakes are eutrophic, and some have reached hypereutrophic status. Based on N/P (mass ratio), Canfield (1981) concluded that N may be limiting in hypereutrophic lakes. Thus, these lakes may not respond to small reductions in P loading. External P loads to many Florida lakes have not been quantified. Two shallow lakes, Lake Apopka (central Florida) and Lake Okeechobee (south Florida) have been extensively studied (see Stites and Reddy, 1998; Steinman et al., 1998). Lake Apopka (surface area = 125 km²) receives P loads of 62 Mg P/yr from vegetable farms (84%), atmospheric deposition (8%), and other sources (8%) (Stites et al., 1997). Lake Okeechobee (surface area = 1,732 km²) receives annual P loads of 518 Mg/yr (Steinman et al., 1998). The average annual total P concentration in Lake Okeechobee increased from near 50 µg/L in the 1970s to the current 90 µg/L, as a result of P loading from the upstream basin (Flaig and Havens, 1995). In Lake Apopka, the P concentration increased from <60 µg/L to more than 200 µg/L in response to farm discharges, which began in the 1940s (Lowe et al., submitted). For
both lakes, intensive efforts are being made to reduce P loads through establishment of loading criteria and implementation of BMPs.

Nonpoint sources of P often dominate loads that lead to the eutrophication of aquatic systems, including lakes. Thus, in many situations, alternative land use management practices have been implemented in an effort to reduce the overall P load to receiving water bodies. For example, historic agricultural management practices north of Lake Okeechobee significantly impacted the water quality of the lake. As a result, this shallow subtropical lake may be moving from a naturally mesotrophic state to a hypereutrophic state (Flaig and Havens, 1995; James et al., 1995). The time needed for lakes to recover after P loads are reduced varies (Sas, 1991). For example, lakes with high sediment P levels may not respond as rapidly to external P load reduction, as lakes with low background P levels (Welch and Cooke, 1995). Shallow lakes apparently respond nonlinearly to P-load reduction with rapid shifts in lake characteristics possible once threshold P concentration are reached and with alternative stable status at intermediate P levels (Moss et al., 1996; Klinge, et al., 1994).

Both biotic and abiotic reactions regulate the dissolved P concentration of the water column. In eutrophic lakes, much of the dissolved reactive P added is rapidly assimilated by algae. Dead algal cells, along with the particulate inorganic and organic solids, accrete on bottom sediments. Sediment bound P accretion rate increases with P loading. For example, in Lake Okeechobee, P accretion rates have increased about four-fold since the 1900s (from about 0.25 g P/m² yr before 1910 to about 1 g P/m² in the 1980s) (Brezonik and Engstrom, 1997). The concentration of total P and its P fractions are generally higher in recent sediments, and decrease with depth, suggesting the influence of increased P loading. Although accretion of sediment bound P suggests that P flux is downward (i.e. from water column to sediments), the dissolved reactive P flux is upwards (i.e. from
sediments to water column) in response to concentration gradients established at the sediment-water interface.

In shallow lakes, P flux across the sediment-water interface occurs in different modes, depending on meteorological and hydrodynamic conditions. During calm days, when vertical turbulent mixing and bottom shear stress is insufficient to resuspend surface sediment, dissolved P moves via passive diffusion and advection. The processes affecting P exchange in this mode include: (i) diffusion and advection due to wind-driven currents, (ii) diffusion and advection due to flow and bioturbation, (iii) processes within the water column (mineralization, sorption by particulate matter, and biotic uptake and release), (iv) diagenetic processes (mineralization, sorption and precipitation/dissolution) in bottom sediments, and (v) redox conditions (O₂ content) at the sediment-water interface. During windy periods, resuspension of P from buried sediments may be an important mode of P transfer to the water column. Because sediment resuspension events are transitory, P flux due to this process may occur at short-time scales at a rapid rate, as compared to diffusive flux. Estimates of P flux due to diffusion only (based on concentration gradients) are shown in Table 5. The relative importance of P transfer due to diffusive flux and resuspension flux must be quantified to accurately estimate annual P flux from internal sources (see Sheng, 1998 for detailed discussion). This internal load (once the result of external load) can extend the time required for ecosystems to reach their historical condition (Sas, 1989; Chapra and Canale, 1991) (Fig. 8). Such lag time for recovery should be considered in developing management strategies for an aquatic system.

Decisions regarding management and restoration of wetlands and aquatic systems are often difficult and controversial as they involve regulating P loads from external sources. Implementation of P reduction goals can have significant costs and economic impacts. Thus, we
need to have a thorough understanding of the dynamics of physical, chemical and biological processes that regulate water quality within these ecosystems. This scientific foundation is needed to develop management models, which can be used as decision-making tools, and to evaluate the responses of wetlands and aquatic systems to reduction in P loads. The key questions often asked are: (i) will the wetlands and aquatic systems respond to P load reduction? (ii) if so, how long will it take for these systems to recover and reach their historical condition?, and (iii) are there any economically feasible management options to speed up the recovery process?

Although many similarities exist among different wetlands and aquatic ecosystems with respect to physical, chemical, and biological processes, the management strategies required for each ecosystem are site specific. Each wetland or lake system may respond differently to P load reductions, depending on their historical P loading record and the existing hydrology and geology of the site. Historical water quality data and paleoecological information can be used to determine the characteristics of a system's original water quality and ecological conditions. This sets the limit to what can reasonably be achieved. Phosphorus enrichment of sediments and their physicochemical properties can also provide an indicator of the time required for recovery. For example, if Lake Okeechobee sediments are enriched with P to the degree observed in Lake Apopka and other hypereutrophic lakes, then the time required for recovery of Lake Okeechobee would be long. However, P enrichment alone is not an indication of potential recovery time; rather, the proportion of P in the bioavailable pool and the stability of stored P also influence recovery time and impact on water quality. In Lake Apopka, for example, despite many decades of high loading of bioavailable P, the P in sediment is largely (>80%) unavailable. Reduction in P loads through implementation of BMPs are apparently leading to improvements of total P in Lake Apopka and Lake Okeechobee (Flaig and Havens, 1995).
denitrification, and sulfate reduction are some examples, that can provide early indications of impending ecological changes (Reddy and D’Angelo, 1996).

Evaluation of P transfer within and between ecosystems requires knowledge of landscape ecology, hydrology and biogeochemistry. Although, in-situ processes of P and associated nutrients are well understood, the linkage of these processes in P transfer from one unit of an ecosystem to another (e.g. from uplands to wetland) requires further study, especially at the landscape level. At present, P is known to be the key nutrient affecting the productivity of Florida’s ecosystems. However, we should not ignore the interactive effects of P on other elemental cycles (such as C and N), that may have comparable effects on ecosystems. Phosphorus control strategies should be developed in the context of overall ecosystem response. Although biogeochemical processes may be sensitive and reliable indicators of P impacts on an ecosystem, their measurement can be time-consuming and expensive. The concentration of certain chemical substrates, intermediates, and end products of ecologically important biogeochemical processes may provide rapid and inexpensive indicators of the rates of these processes. If these simple measurements made on an indicator ecosystem unit are well correlated with the related processes, the resulting empirical equations can be used to transfer the process level information to landscape level (Fisher, 1997). To evaluate the P impacts and successes of restoration efforts, it is helpful to develop a fundamental understanding of the biogeochemical process regulating the ecosystem function. Risk assessment is only as good as the information/knowledge available at the time. Lack of understanding of the factors that affect the biogeochemical processes regulating the fate and transport of P and associated nutrients decreases the certainty of an assessment.
Future Research Needs:

Effective P control strategies can be strengthened greatly by improving our understanding of the processes controlling the fate and transport of P within and between ecosystems. Research conducted during the past 10 years on selected Florida's ecosystems provided the basis for developing P control strategies for those drainage basins. Only a few ecosystems have been studied in detail, and it is not known how well rates and processes in these systems can be transferred to other ecosystems. In Florida's ecosystems, P movement is rapid, because of unique soil types and their poor P retention capacity, and rapid movement of water through the soil profile. Many wetlands and aquatic systems are located adjacent to upland ecosystems with active urban and agricultural development, and become recipients of P loads.

The following are some of the key research needs to address P issues in Florida's ecosystems.

1. Quantify P budgets within each ecosystem. These budgets should quantify the amount of P imports and exports for each region in Florida. This should be done for each District's boundaries.

2. Determine historical background levels of P for each ecosystem. This requires extensive paleoecological research or monitoring, especially in unimpacted areas of an ecosystem to determine natural sources of P.

3. Quantify P inputs from wet and dry atmospheric sources in each region of Florida.

4. Establish reference sites, representing an upland, wetland, and aquatic ecosystem for each region. These sites can be used to conduct long-term research and monitoring, as well as assessment of impacts to other sites.

5. Standardize methods for selected parameters to be measured in all ecosystems. Methods followed by scientists and managers within each region should be comparable.
6. Develop calibrated soil test procedures of P availability in uplands used for agriculture, which include water quality as one criteria.

7. Land application of wastes (organic solids) should be tailored properly to soil and hydrologic characteristics of the site, and the composition of the waste. The amount of bioavailable P in the waste should be one of the criteria in determining P application rates.

8. Develop soil/sediment criteria for easily measurable biogeochemical properties. This can be used to determine P impacts and long term implications.

9. Determine hydrologic pathways governing P movement within uplands, and transfers to adjacent wetland or aquatic ecosystems.

10. Quantify the influence of P loading on other nutrient cycles, and their feedback in regulating eutrophication of wetlands and aquatic systems.

11. Determine the influence of internal P load on recovery of an ecosystem. For uplands, determine how long P stored in these soils will continue to be released even after agricultural activities and P imports are minimized. For wetlands and aquatic ecosystems, determine how long it takes for these ecosystems to recover after all external loads are curtailed.

12. Develop criteria to use in wetlands and aquatic ecosystems as indicators to determine the success (or lack of success) of P management in adjacent uplands systems.

13. Determine the relationships between biogeochemical processes and easily measurable indicators for different ecosystems. Use the resulting relationships to extrapolate process level information to ecosystem scale.

14. Develop empirical (statistical) and mechanistic models to synthesize experimental data and to aid prediction of impacts and recovery.
15. Integrate monitoring data on biogeochemical indicators into predictive models to assess system behavior at the landscape (which includes multiple ecosystems such as uplands, wetlands, and lakes) level.
References


Table 1. Phosphorus imports from various sources into State of Florida (see text for assumptions made to estimate P inputs).

<table>
<thead>
<tr>
<th>Source</th>
<th>Annual import</th>
<th>% of total</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mg/P/yr</td>
<td></td>
</tr>
<tr>
<td>Fertilizer</td>
<td>42,660</td>
<td>70</td>
</tr>
<tr>
<td>Biosolids</td>
<td>3,040</td>
<td>5</td>
</tr>
<tr>
<td>Wastewater</td>
<td>4,930</td>
<td>8</td>
</tr>
<tr>
<td>Compost</td>
<td>?</td>
<td>?</td>
</tr>
<tr>
<td>Animal manures</td>
<td>4,670</td>
<td>8</td>
</tr>
<tr>
<td>Atmospheric deposition</td>
<td>6,000</td>
<td>10</td>
</tr>
<tr>
<td>Natural weathering of minerals</td>
<td>?</td>
<td>?</td>
</tr>
</tbody>
</table>

Table 2. Relative phosphorus adsorption (RPA) of soils (Harris et al., 1995).

<table>
<thead>
<tr>
<th>Soil</th>
<th>A</th>
<th>E</th>
<th>Bh/C/Bt</th>
</tr>
</thead>
<tbody>
<tr>
<td>Spodosols</td>
<td>0.26</td>
<td>0.08</td>
<td>0.96 (Bh)</td>
</tr>
<tr>
<td>Entisols</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Clean sands</td>
<td>0.05</td>
<td>-</td>
<td>0.01 (C)</td>
</tr>
<tr>
<td>Coated</td>
<td>0.48</td>
<td>-</td>
<td>0.47 (C)</td>
</tr>
<tr>
<td>Ultisols</td>
<td>0.74</td>
<td>0.69</td>
<td>0.96 (Bt)</td>
</tr>
</tbody>
</table>

RPA = absolute amount of P adsorbed/maximum amount of P retained.

Table 3. Influence of land use on total P, relative phosphorus adsorption, (RPA), and equilibrium P concentration (EPCo) of soils (Nair et al., 1997).

<table>
<thead>
<tr>
<th>Landuse</th>
<th>A</th>
<th>E</th>
<th>Bh</th>
<th>Total P</th>
<th>EPCo</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>mg/kg</td>
<td>mg/L</td>
</tr>
<tr>
<td>Intensive</td>
<td>N/A</td>
<td>0.03</td>
<td>0.38</td>
<td>2,330</td>
<td>5.0</td>
</tr>
<tr>
<td>Holding</td>
<td>0.03</td>
<td>0.20</td>
<td>0.42</td>
<td>181</td>
<td>1.4</td>
</tr>
<tr>
<td>Pasture</td>
<td>0.05</td>
<td>0.08</td>
<td>0.47</td>
<td>31</td>
<td>0.1</td>
</tr>
<tr>
<td>Beef</td>
<td>0.07</td>
<td>N/A</td>
<td>0.37</td>
<td>31</td>
<td>0.1</td>
</tr>
<tr>
<td>Forage</td>
<td>0.07</td>
<td>0.02</td>
<td>0.34</td>
<td>23</td>
<td>0.2</td>
</tr>
<tr>
<td>Native</td>
<td>0.47</td>
<td>0.21</td>
<td>0.36</td>
<td>18</td>
<td>0.1</td>
</tr>
</tbody>
</table>
Table 4. Diffusive flux of soluble P from soil to overlying floodwater in selected wetlands created on agricultural lands (D'Angelo and Reddy, 1998).

<table>
<thead>
<tr>
<th>Site</th>
<th>Time after flooding (months)</th>
<th>Soluble P flux potential (mg/m² day)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lake Apopka</td>
<td>3</td>
<td>0.3</td>
</tr>
<tr>
<td>(50 m from inflow)</td>
<td>8</td>
<td>0.3</td>
</tr>
<tr>
<td></td>
<td>13</td>
<td>0.1</td>
</tr>
<tr>
<td></td>
<td>32*</td>
<td>0.6</td>
</tr>
<tr>
<td>Lake Apopka</td>
<td>3</td>
<td>5.5</td>
</tr>
<tr>
<td>3000 m from inflow</td>
<td>8</td>
<td>3.1</td>
</tr>
<tr>
<td></td>
<td>13</td>
<td>2.4</td>
</tr>
<tr>
<td></td>
<td>21</td>
<td>1.1</td>
</tr>
<tr>
<td>Sunnyhill Farm</td>
<td>48</td>
<td>2.0</td>
</tr>
<tr>
<td></td>
<td>60</td>
<td>0.7</td>
</tr>
<tr>
<td>Emeralda</td>
<td>12</td>
<td>0.9-1.3</td>
</tr>
<tr>
<td>Knights (ENR)</td>
<td>10</td>
<td>0.3-9.2</td>
</tr>
<tr>
<td></td>
<td>16</td>
<td>0.6-1.0</td>
</tr>
</tbody>
</table>

Table 5. Diffusive flux of soluble P from sediments to overlying water in selected aquatic systems.

<table>
<thead>
<tr>
<th>Aquatic System</th>
<th>Diffusive Flux mg P/m² day</th>
<th>Ref</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lake Apopka</td>
<td>1-5.3</td>
<td>Reddy et al. (1996)</td>
</tr>
<tr>
<td>Lake Okeechobee</td>
<td></td>
<td>Moore et al. (1998)</td>
</tr>
<tr>
<td>Mud Zone</td>
<td>0.1-1.9</td>
<td></td>
</tr>
<tr>
<td>Peat Zone</td>
<td>0.2-2.2</td>
<td></td>
</tr>
<tr>
<td>Sand Zone</td>
<td>0.1-0.5</td>
<td></td>
</tr>
<tr>
<td>Littoral Zone</td>
<td>0.6-1.5</td>
<td></td>
</tr>
<tr>
<td>Lake Barco</td>
<td>0.02-0.05</td>
<td>Reddy and Fisher (Unpublished</td>
</tr>
<tr>
<td>Tampa Bay</td>
<td>1.9-6.3</td>
<td>Results)</td>
</tr>
<tr>
<td>Indian River Lagoon</td>
<td>0.7-1.7</td>
<td></td>
</tr>
</tbody>
</table>
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Phosphorus Transfer

Fertilizers
Animal wastes
Biosolids
Wastewater

Atmospheric deposition

Uplands [Sink/Source]

Wetlands [Sink/Source]

Aquatic Systems [Sink]
NWFWMD
[Apalachicola, Choctawhatchee, and Escambia river basins]

SRWMD
[Suwanee River]

SFWMD
[Hillsborough, Peace, and Withlacoochee river basins; the Green Swamp; Tampa Bay]

SJRWMD
[St. Johns River Basin; Oklawaha chain of lakes; Indian River Lagoon]

SFWMD
[Kissimmee River Basin; Lake Okeechobee; the Everglades; Big Cypress Swamp, and Florida Bay]
Fig. 3
Fig. 4

Phosphorus in Soil Porewater

[A] Water

[Desorption] Adsorption

EPCo Smax Precipitation

slope = KD

P desorption under ambient conditions
Fig. 5

[A] Water

\[ P_{ad} \leftrightarrow P_s \]

Soil

\[ P_{w} \leftrightarrow P_{ret} \]

[B]

P retention

\[ EPC_w \]

slope = A

P release

P release under ambient conditions

Phosphorus in Water Column
General Discussion

The Florida Department of Environmental Protection (DEP) has developed the following working definition of ecosystem management: "Ecosystem management is an integrated, flexible approach to management of Florida's biological and physical environments - conducted through the use of tools such as planning, land acquisition, environmental education, regulation, and pollution prevention - designed to maintain, protect, and improve the state's natural, managed, and human communities" (FLDEP, 1994). This definition is adapted from the dominant ecosystem management themes presented by Grumbine (1994). One aspect of ecosystem management involves water quality, more specifically, P management within the ecosystem, to reduce eutrophication of water resources. The effectiveness of P management within the ecosystem can be measured by knowing (i) the baseline P concentration of surface water, sediments, and biota, (ii) the mass P imports and export (budgets) within and between ecosystems, (iii) the status of biogeochemical indicators of water quality, (iv) the P transfer mechanisms within and between ecosystems, (v) the extent to which P fate and input can be predicted by easily measured water quality parameters (water, soil and sediment, vegetation, and periphyton and plankton) and environmental variables, and (vi) the criteria that will be used to evaluate the ecosystem recovery after restoration plans are implemented.

Although state agencies are investing vast amounts of resources for water quality monitoring, many ecosystems remain inadequately monitored to evaluate the adverse impacts of P loading. Many of these monitoring efforts are limited in scope and cannot be readily extrapolated to the whole ecosystem. Very few monitoring programs capture spatial and temporal variations in P concentration in surface water, soils, and sediments. Since sampling analysis is often performed
by several agencies, standardization in parameters to be measured, in methods, and in quality assurance is needed.

For effective P management, P budgets need to be developed for the ecosystem. For example, P budgets for Lake Apopka, Lake Okeechobee, the Lake Okeechobee Basin and the Everglades are available (see example of P budget for the Okeechobee Basin) (Fig. 9). However, similar budgets are not available for other ecosystems, within each of the five WMDs. This type of information is essential for large-scale understanding of the relative importance of sources and sinks at an ecosystem scale, and for developing P control strategies.

In addition to the P concentrations in water and soil; several biogeochemical characteristics of water, soils and sediments; vegetation and periphyton and plankton can be used as indicators of P impacts on an ecosystem. Selection of these indicators should be based on their ease of measurement, and their sensitivity to changes in P concentration of the water column or P loading.

In large-scale ecosystem management and restoration, it is useful to determine what portions of the landscape best indicate the integrity of the entire ecosystem including uplands, wetlands and aquatic habitats. For example, certain patches of an ecosystem (e.g., wetlands or lakes) may contain the best record of overall ecosystem integrity, because of their position in the landscape. Once it is decided on the indicator ecosystem unit or patch, it is necessary to determine what biogeochemical characteristics of the selected area provide the most efficient indicator of pollutant impacts. For example, there is evidence that the rates of certain biogeochemical processes in wetlands reflect changes in materials budget long before such changes are reflected in population of higher organisms. Thus, indicators of biogeochemical processes, such as microbial respiration, organic matter turnover, organic N and P mineralization, phosphate adsorption and desorption,