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PHOSPHORUS CYCLES IN TERRESTRIAL AND AQUATIC ECOSYSTEMS

REGIONAL WORKSHOP 3:
SOUTH AND CENTRAL AMERICA

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Edited by
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PHOSPHORUS CYCLES
IN TERRESTRIAL AND AQUATIC ECOSYSTEMS
REGIONAL WORKSHOP 3: South and Central America

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INTRODUCTION

Phosphorus is a key limiting element in the highly weathered soils of the Amazon basin and adjacent peneplains. Our knowledge of the natural P cycles in these regions is extremely limited. At the same time, the stability of many of the region's ecosystems is threatened by deforestation and agricultural development. Scientific information is increasingly needed to guide the use of P in the development of these regions. This requires an understanding not only of the fate of P in areas under human use, but also of the natural P transformations in different parts of these vast and divers ecosystems.

To this end, information on the phosphorus cycle needs to be summarised and integrated across landscapes, and combined with the knowledge of other nutrient elements and their interactions. Phosphorus significantly affects the cycles of carbon, nitrogen and sulphur and thereby determines ecosystem productivity and stability in P limited regions.

The need for adequate P supplies in agriculture, forestry and extractive systems (such as rubber, oil or Brazil nut collection) requires an understanding of the sources and sinks of P in natural and anthropogenic ecosystems, and their interaction with added fertilizer P. New methods, developed in temperate regions, have helped quantify levels of biologically active P and relatively inert physically and chemically occluded forms in ecosystems. Key processes of phosphorus interactions with carbon, nitrogen and sulphur have been identified and incorporated into computer models to guide interpretation of phosphorus data. These models need to be extended to a wider range of ecosystems so that, in addition to the data base, they will provide a mechanism for the evaluation of the short- and long-term impact of man's manipulation of phosphorus in the biosphere. A realistic understanding of elemental cycles is not possible in isolation from other nutrients or from the effects of landscapes. A holistic approach to global biogeochemistry is needed.

SCOPE project in general

In response to the above noted concerns, SCOPE organised a workshop on the biogeochemical P cycles in Venezuela. This is part of a major new study aimed at better understanding the nature, sources and fluxes of phosphorus in terrestrial and aquatic ecosystems and into the global environmental effects of P through interactions with cycles of other elements.

The objectives of the global project are:

1) To critically assess knowledge of the nature, sources, sinks and fluxes of phosphorus in the biosphere.

2) To identify mechanisms of supply of biologically active phosphorus in terrestrial and aquatic ecosystems.

3) To provide the information required to more effectively meet worldwide phosphorus requirements for sustainable food and fiber production while minimizing adverse effects on the environment.

4) To evaluate the environmental effects of current and projected use of phosphorus in relation to the biogeochemical cycles of carbon, nitrogen, sulphur and metals.

5) To evaluate the transfers of phosphorus from terrestrial to aquatic and marine environments and the relationship to the cycling of other elements.

6) To assess current and likely future economic trends in phosphorus use.

The primary focus of this project will be to integrate and synthesize information on phosphorus in diverse environments with emphasis on the flows of phosphorus between terrestrial systems, ground waters, rivers, lakes, estuaries, and oceans. Both natural and anthropogenic fluxes in the phosphorus cycle will be assessed in a study of the biogeochemical processes. Particular attention will be paid to phosphorus interactions with other elements (carbon, nitrogen, sulphur and metals).

The project comprises three levels of activity:

1) Organization of regional workshops in Africa, Asia, Europe and South America; these workshops will synthesize data on the phosphorus cycle in major ecosystems of each area.

2) Development of conceptual and simulation models of phosphorus cycling in major ecosystems of the world; these models will identify gaps in knowledge, and areas where careful management of phosphorus resources will be required to optimize food and fiber production and minimize hazards in various environments.

3) Integration of the information from the four regional workshops and presentation of the results at a final international workshop to be held in India. Every attempt will be made at this workshop to link these findings on a global scale including oceanic fluxes; it is expected that this will provide a global perspective on the economic and environmental consequences of the use of phosphorus resources.

This information is required to develop management strategies and policies for an important and potentially limiting resource. In addition, it will provide understanding of the interrelationship of nutrient elements in a changing world environment. It will develop insights into the mechanisms and processes involved in phosphorus cycling, in addition to addressing major agricultural and environmental issues. It is anticipated that the project will stimulate further investigations by UNEP, SCOPE, and other scientific organizations.

Specific objectives of this meeting

This third meeting is an attempt to integrate and synthesize information on phosphorus in South and Central America with special emphasis on the Amazon. The region is characterised by low soil phosphorus reserves in old, stable ecosystems which are drained by the largest rivers system on earth. Large annual fluctuations in water levels result in thousands of square kilometers of flood plains which play a major part in the regions nutrient cycles. The uplands to the south and north of the Amazon Basin are dominated by savannahs or grasslands with low nutrient availabilities. These areas are increasingly used for pasture and arable agriculture, creating the need for extensive fertilisation and management of phosphorus fixation capacities. In this context, the meeting specifically addresses the following subjects

1) Processes of P cycling in natural rain forests. How do natural ecosystems adapt to low nutrient availabilities?

2) The exchange of P between land and water in the extensive flood plains. How are P levels regulated by annual flooding cycles in lakes, steams and oxbows?

3) The forms and quantities of P drained in surface waters. Can these flows be quantified despite large seasonal fluctuations? How are these flows affected by different anthropogenic impacts on watersheds?

4) Phosphorus cycles under low-input management systems like shifting cultivation and agroforestry. Do these systems require additional P inputs? Are they "conservative" in their nutrient uses and losses?

5) Phosphorus requirements under pasture and arable agriculture. How can phosphorus limitations be overcome, and what fertilizer resources are available?

An impressive amount of information addressing these topics has been brought together by the contributors to the workshop. At the same time some weaknesses have been pointed out in our understanding of P cycles:

1) Very little is known about the P budgets in soils in the Amazon rainforests. What forms of P? how much? to what depth? with what mobility and availability? This is unlike the situation in many temperate regions, where extensive knowledge of soil P pools and their properties has permitted the application of models for the cycling of P in conjunction with other elements. In these regions, such models have progressed to considerable predictive capabilities. Much work is still needed before the rainforests, wet savannahs and derived anthropogenic systems can be understood at a level adequate for predictive nutrient models.

2) The distribution and movement of P across landscapes is barely understood. Few data are available on the variability and distribution of different P forms across landscapes. As in the two previous workshops in Europe and Southeast Asia, large gaps existed in the understanding of how P in different parts of landscapes is related. Erosional fluxes, water-sediment exchanges in flood plains, and transformations between different P forms (e.g. organic-inorganic, "fixed"-labile) as dependent on landscape position and hydrology need much further thought and study.

A synthesis of the knowledge presented has been attempted at the workshop, but the problems discussed and gaps in understanding were so divers that it has been impossible for the editors to synthesise the information. It will be apparent in several contributions that the concepts and understanding of different authors differ. These different views, and the recommendations of the working groups will be available to a later meeting whose objective it is to construct a framework of conceptual and mathematical models that will permit a better integration of the wealth of information presented, and at the same time provide guidance for future research towards understanding of P cycles in diverse ecosystems.

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BIOGEOCHEMICAL CYCLES OF PHOSPHATE IN LAKES

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It has been known since about the mid 1940's, that phosphate plays an important role in controlling primary production in lakes (mainly the algal growth). Overwhelming evidence was produced by a large communal effort in the OECD (OECD, 1982) that there is a strong correlation between phosphate input and chlorophyll a which serves as a measure of algal standing crop, or even algal growth. There are however, many exceptions to this rule: there are several lakes where so much phosphate has been added, that nitrogen or even light may now be the factor controlling algal growth. This has often happened in shallow lakes (Golterman, 1983).

The OECD correlation tells us little or nothing of the processes involved in the relationship between the disposal of phosphate into a lake and the response of its populations. The purpose of this paper is to show the importance of some of these processes and to point out the need for determining their rate constants, which are essential for a quantitative understanding of the phenomena involved, and of course, for any serious attempt at modelling.

A flow chart of processes in P transformations is given in Figure 1. We can distinguish between chemical and biological processes. The former control relationship between loading and the subsequent concentration not only by dilution, but also by the adsorption onto suspended matter, followed by its sedimentation. The latter encompass biological uptake, and mineralization, though autolysis, bacterial decomposition or digestion by zooplankton. While the biological processes take place mainly in the water and are relatively rapid, the chemical processes are of a longer duration and take place to a large extent in the sediments.

Relation between loading and concentration

The relation between loading and concentration for a non-conservative element can be expressed by the simple mathematical formulation:

$$(o\text{-}P)_w = L/z(\sigma + \rho)$$

where: $(o\text{-}P)_w$ = ortho phosphate conc. in the water (mg m^{-3})
L = Loading (g m^{-2} y^{-1})
z = mean depth (m)
ρ = out-wash coefficient (t^{-1})
σ = sedimentation coefficient (t^{-1})

The most difficult factor to be estimated is the influence of the sediments, which may bind considerable amounts of phosphate that may be released later if measures are taken to decrease the phosphate input. It has been shown (Golterman 1980, 1984) that the adsorption of phosphate by sediments follows the adsorption mechanics:

$(o\text{-}P)_{sed} = A^* \; ((o\text{-}P)_w)^B$

The constant "B" is often found to be about 0.3. Therefore, an increase of $(o\text{-}P)_w$ by a factor 10 is only causing an increase in the sediment phosphate concentration by a factor 2-2.5. Müller (1977) and Wagner (1976) have shown that in Lake Constance the $(o\text{-}P)_{sed}$ increased by a factor 2 only, while the (o-P)w concentration increased by about a factor 10 (For a more detailed discussion, see Golterman, 1980). The constant "A" depends on the amount of sediments entering a lake and their adsorption characteristics. The first variable is rarely measured, but a reasonable estimate can be obtained by estimating the sedimentation rate for which several methods are available. The adsorption characteristics "B" can be measured with sediments obtained from the lake itself without technical difficulties, but the interpretation of the results should be done with great care.

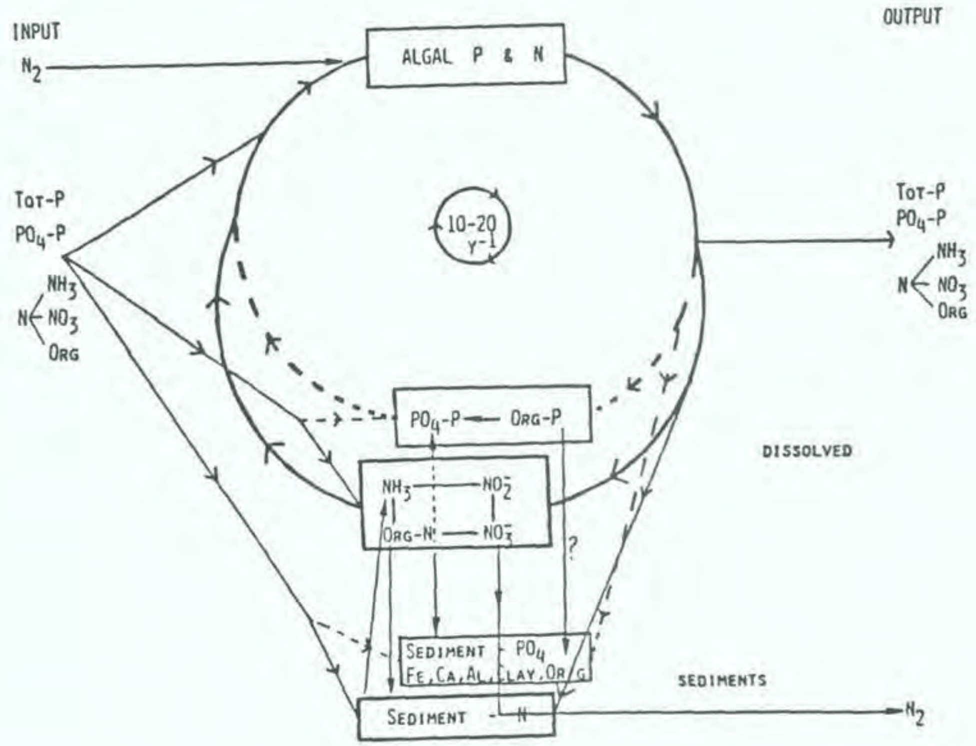

Figure 1. Flow chart of phosphate fluxes in fresh water ecosystems. From Golterman, 1984.

The possible release of phosphate from sediments is a very important process for lake management (see several papers in Sly, 1986). It is often calculated by measuring, *in situ*, the increase of the phosphate concentration of the water overlying the sediments or, *in vitro*, by measuring the phosphate concentration in isolated cores. Release rates found in these ways range between 0 and 100 mg m^{-2} y^{-1} (Bostrom et al, 1982 and Golterman and de Oude, 1990). A review by French (1983), gives range of release rates between -10 and 50 mg m^{-2} d^{-1} for aerobic and 0 - 150 mg m^{-2} d^{-1} for anaerobic conditions. The higher release under anaerobic conditions is probably caused by mineralization of organic matter containing phosphate and not by reduction of Fe(OOH)≈P, as is usually believed in the literature. Both the in situ and in vitro measurements are, however, fraught with problems, and our own work is therefore more directed towards a knowledge of the chemical equilibria between phosphate in the sediments and in the water.

The nature of the chemical binding in the adsorption process must be understood. Phosphate may be adsorbed onto Fe(OOH) and/or $CaCO_3$ but specific extractions of Fe(OOH)≈P and $CaCO_3$≈P are difficult. A standardized procedure is in use, applying "functional extractions" such as NaOH to extract Fe(OOH)≈P and H_2SO_4 to extract the $CaCO_3$≈P (for example Hieltjes and Lijklema, 1980). Golterman and Booman (1988) and De Groot and Golterman (1988) applied specific chelators to extract the inorganic part of the sediment phosphate: Fe(OOH)≈P is extracted with a Ca-NTA/dithionite solution, followed by an extraction of $CaCO_3$≈P with Na-EDTA. Results are so far satisfactory; the difference between the functional and chelating extractions is shown in Figure 2.

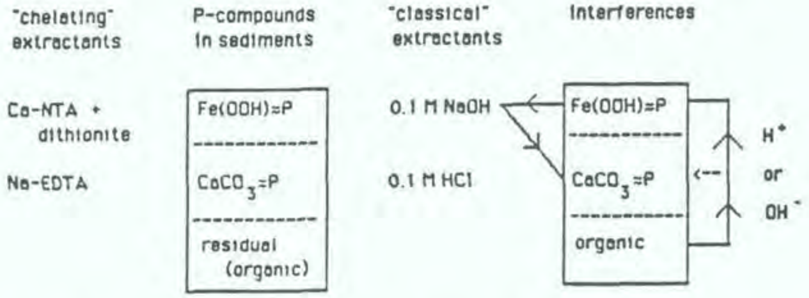

Figure 2. Chelating and 'classical' extractions of sediment phosphates with possible interferences in the classical extractions. From de Groot & Golterman, 1990.

These specific extractions have made it possible to study the distribution of phosphate between the different pools. It appears that even in a sediment with about 20% of $CaCO_3$ most of the phosphate was adsorbed onto the Fe(OOH), of which the concentration was only 0.5% (Golterman and Booman, 1988). Therefore a solubility diagram (see Figure 3) has been developed (Golterman, 1988) which describes the distribution of phosphate between these pools.

The exchange of inorganic P between sediments and water is controlled by chemical equilibria, based on adsorption and sedimentation mechanisms. It is therefore controlled by the phosphate concentration in water and in the sediments, the pH, the temperature, etc. The exchange processes of the organic pool are based on non-reversible reactions: the input is the sedimentation of dead organic matter sinking to the bottom of the lake, or the uptake by organisms occurring in the sediments. The release of phosphate is mainly caused by the mineralization of the organic matter but not all phosphate mineralized will escape from the sediments. This mineralization is probably the origin of the influence of a low redox-potential on P release by sediments, wrongly supposed to be a reduction of Fe(OOH)≈P.

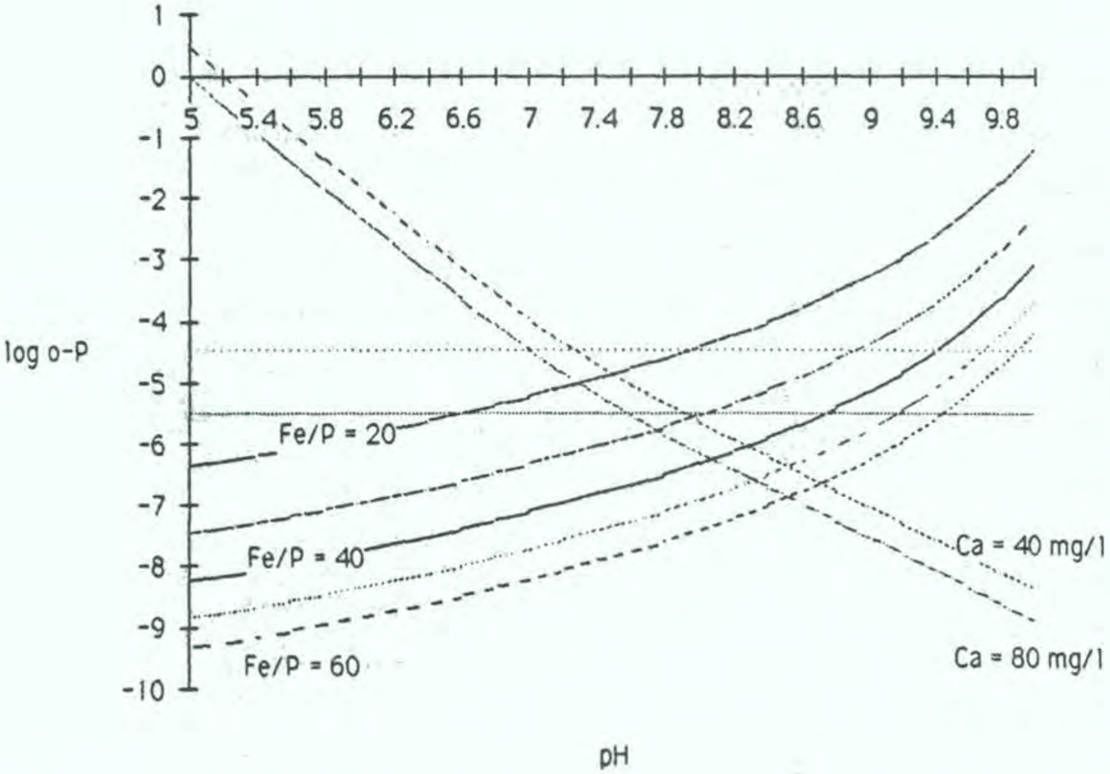

Figure 3. Solubility diagram of iron and calcium bound phosphate. From Golterman, 1988.

The specific extractions help us to understand the bio-availability of sediment phosphate. Thus Grobbelaar (1983) has shown that NTA - extractable phosphate correlated well with bio-availability. De Graaf and Golterman (1989, in press) have shown that in about 10 Dutch lakes a good correlation existed between Fe(OOH)≈P plus $CaCO_3$≈P and the bio-availability for algal cultures (and not between Fe(OOH)≈P alone and the bio-availability, as was expected) (Figure 4).

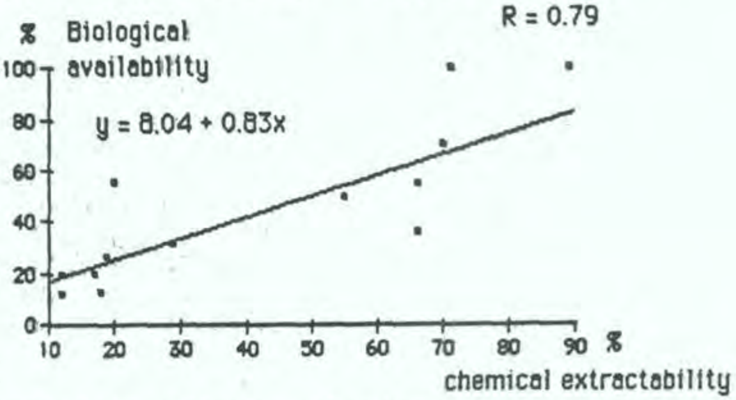

Figure 4. Comparison between chemical extractability and biological availability in 14 lake sediments in the Netherlands. From De Graaf Bierbrauwer-Würtz & Golterman, 1989.

The largest part of primary production in lakes is produced by phytoplankton, i.e. unicellular, microscopic algae. The role of orthophosphate (o-P) is often twofold:

1) Algal biomass is limited by the availability of the o-P (yield limitation).

2) The growth rate of the algae may be limited by the o-P concentration (rate limitation).

1) The o-P concentration of dry algal biomass falls usually between 0.5 and 1%. Before the 1950's, lake o-P concentrations were usually low enough to maintain low biomass. At present, there is still a strong correlation between the amount of phosphate and that of chlorophyll a (chl a is often around 1-1.5% of the dry weight and can therefore be used as a measure of algal biomass) indicating some P limitation. Because of the large variability of ecosystems and the weakness of the measured data, the correlation cannot be used for the management of a particular lake.

2) The growth rate of algae may depend on the concentration of one of several nutrients. In lakes this may often be the o-P concentration. The relation can be described by the well known Monod equation:

$$\mu = \mu_{max} \, . \, (o\text{-}P)/\{Ks + (o\text{-}P)\}$$

where μ = growth rate (t^{-1})
K_S = a constant (mg l^{-1})

K_S is a physiological factor, describing the "efficiency" of phosphate utilization; it has the dimension of a phosphate concentration (Figure 5). K_S is often measured in cultures (Healy, 1982) and there are very few "in lake" observations. K_S probably depends on environmental factors, such as pH, temperature etc. These relations are still unknown.

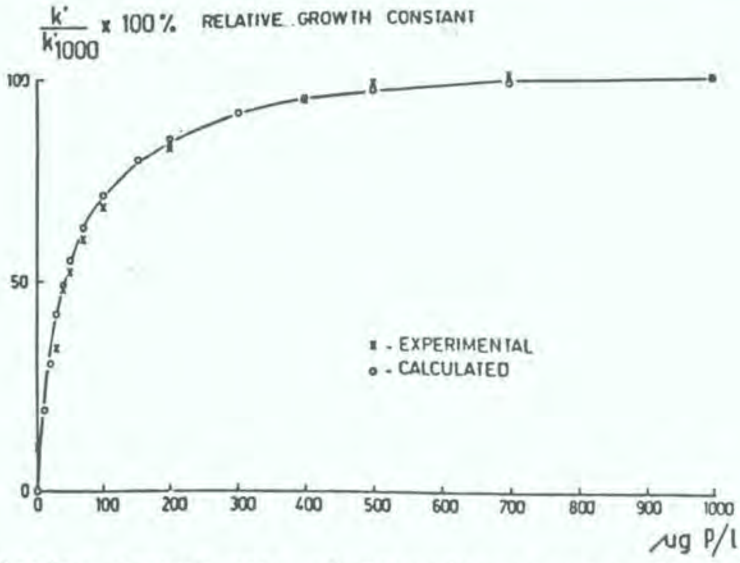

Figure 5. Relative growth rate of Scenedesmus spec. in culture solutions. Ks is the phosphate concentration where the relative growth rate is 50%. From Golterman et al. 1969.

In cultures yield and rate limitations can occur simultaneously (Figure 6). In lakes this is not necessarily the case. The growth limitation of the diatom *Asterionella formosa* e.g. in Lake Windermere may be described based on the work of Lund (1960), as follows:

$$dN/dt = M_{max} \,.\, N.(o\text{-}P)/\{K_S + (o\text{-}P)\}.(N_{Si} - N)/N_{Si}$$

where N = the number of diatoms

N_{Si} = the maximum number of diatoms (carrying capacity as limited by SiO_2)

In the beginning of the growth season the o-P concentration is small and the SiO_2 concentration is large; the o-P controls therefore the growth rate and N_{Si} has hardly any influence. Later when N approaches N_{Si}, the growth rate becomes 0 (yield limitation). Thus, the yield is controlled by the availability of SiO_2, but the growth rate is controlled by the o-P concentration.

There are only a few lakes where these growth limitations are so well studied. Because of the present way of financing research, there is only a small possibility that this situation will improve. Better understanding of the rate constants of the phosphate cycle will help us to quantify the relations between phosphate and algal growth. Thus, it has been possible in a theoretical study (Golterman, 1989) to demonstrate that nitrogen limitation of the mineralization process can change the phosphate/chlorophyll a relationships, improving the simple linear relationships which have too long dominated the thinking of water managers.

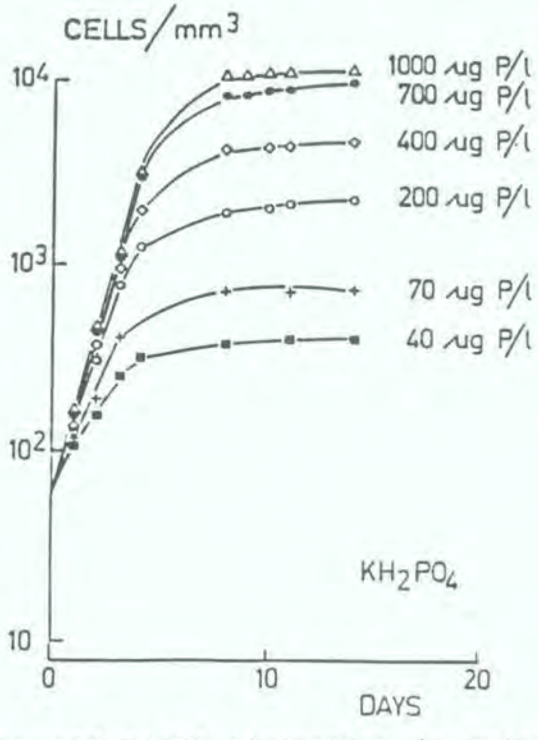

Figure 6. Growth rate and yield limitation in cultures of Scenedesmus spec. From Golterman et al. 1969.

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PHOSPHORUS IN THE AMAZON RIVER MAINSTEM: CONCENTRATIONS, FORMS AND TRANSPORT TO THE OCEAN

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Introduction

The global biogeochemical P cycle begins with weathering and mobilization of primary rock phosphate and ends with permanent burial in marine sediments. Along the way P is cycled through the biosphere and involved in various inorganic reactions primarily in soils. Unlike the other bioactive elements - carbon, nitrogen and sulfur - P has no significant gaseous forms in the natural environment, therefore; the major conduit for the transport from the continents to the world's oceans is as dissolved and particulate phases in rivers. Although the contemporary riverine flux to the oceans is not well known, it has been variously estimated to be about 20 to 25 Tg/y ($Tg=10^{12}$ g) and the pre-agricultural riverine flux is thought to be about half that (Meybeck, 1982; Froelich et al., 1982, Berner and Berner, 1987). For comparison, Richey (1983) and Graham and Duce (1979) have estimated the aeolian flux at about 5% of the riverine transport. The only other input of potential consequence is glacial scour but it is also small (Kempe, 1979).

The forms of P in transport in a river are generally classified according to the method used to determine them rather than by any strict chemical or physical definition. Thus, the total dissolved P (TDP) pool is frequently divided into soluble reactive P (SRP), which is normally assumed to be primarily PO_4, and dissolved organic P (DOP). Although there are many different classifications of particulate P based on the numerous extraction methods, the most ecologically useful are probably particulate inorganic P and particulate organic P; the most unambiguous is total particulate P (TPP). The distinction between particulate and dissolved is defined by the nominal pore size of the filter used to separate them, usually 0.45 µm. (Thorough discussions of the different analytical methods for determination of the various P forms are given by Froelich (1988), and Strickland and Parsons (1967.)

The average TDP level in the world's unpolluted rivers is about 1 µM and that of TPP is 15 µM (Froelich 1988, Meybeck, 1982). However, in rivers draining highly industrialized or agriculturlized areas P contents can increase dramatically, e.g. in the Thames (Great Britain) SRP values of 80 µM have been reported (Meybeck 1982).

Combining the annual river discharge of 3.2×10^6 m^3 s^{-1} (Berner and Berner, 1987) with average river concentrations yields the estimate of total, pre-agricultural P transport to the oceans of about 14 Tg/y. The difference between this number and that given above is the anthropogenic component. However, for several reasons, there are large uncertainties in these numbers: (1) the lack of information on the particulate forms

(Meybeck, 1982), (2) the difficulty of representatively sampling river particulate distributions that are not uniform either laterally or with depth (Curtis et al., 1979; Richey et al. 1986), (3) differences in analytical methodologies, especially for TPP, and (4) lack of good seasonal coverage of many rivers. This is especially true for the large tropical rivers which contribute a major fraction of the water and sediment discharge to the ocean.

In this report we present the results of a study undertaken to improve our understanding of temporal and spatial distributions of the different P forms, concentrations and fluxes in the Amazon river, the world's largest river. The study is part of the CAMREX (Carbon in the AMazon River EXperiment) project, which also investigated carbon and nitrogen. Results of the carbon and nitrogen investigations have been reported previously (Hedges, et al.. 1986; Forsberg et al.,1988, Richey et al., in press). The total length of the Amazon river is about 3500 km and its drainage basin area is nearly 6 million km^2. The Amazon main channel receives input from major and minor tributaries. During its annual 10 meter rise and fall, the river also exchanges water and chemicals with its seasonally inundated floodplain or "varzea". The varzea acts as a large reservoir (100,000 km^2, Junk, 1985) for flood discharge and precipitation and, at times, as much as 30% of the water in transit in the river has passed through this reservoir (Richey et al., in press). With an average water discharge of about 200,000 m^2s^{-1}, the Amazon accounts for fully 20% to the total annual water discharge to the oceans (Richey et al. 1986). Also, its sediment load of about 10^9 Mg y^{-1} constitutes slightly less that 10% of the total riverine transport (Milliman and Meade, 1983). Thus, a thorough knowledge of P transport in this mighty river is crucial to quantifying the global P budget.

Methods

Measurements were made over a 1700 km reach of the river during 7 cruises of the Brazilian research vessel, R/V Amanai, between Vargem Grande (3°16'S, 67°50'W) and Obidos (1°55'S, 55°29' W). On each cruise measurements were made at 9 mainstem stations plus stations on the 7 principal tributaries in the reach. All measurements were made between 1982 and 1984, and covered the different portions of the river hydrograph (Figure 1).

In order to derive a quantitative estimate of the amounts of dissolved and particulate P transported from the continent by the Amazon river it is first necessary to have a good estimate of the water discharge. Additionally, the cross sectional and depth distributions of both dissolved (Richey et al. 1988) and particulate phases (Meade, 1985) are not uniform so these must be determined also. To determine water discharges and cross-sectional distributions samples were taken with the integrating sampling array described by Richey et al. (1986). Briefly, the array consisted of a Price AA current meter that was lowered to the bottom and retrieved at a constant rate. Also attached to the array was a collapsible bag sampler that filled in proportion to the in situ current velocity. Thus, when lowered to the bottom and retrieved the sampler obtained a depth-integrated, flow weighted sample (profile) from one point in the river. At each mainstem station 18 such profiles were collected at equally spaced intervals across the river and tributaries were sampled using 12 profiles. The water discharge at the section was then calculated by integrating the individual velocity profiles across the river channel and a depth-integrated, flow weighted water sample for chemical analysis was obtained by compositing the individual profile samples.

Prior to chemical analysis the composite water sample was passed through a 63 µm mesh screen to separate the coarse particulate fraction which was dried and weighed to determine coarse suspended sediment (CSS) concentration. A small portion of the sample (~300 ml) was then filtered through pre-weighed, 0.45 µm filters and stored for later, gravimetric determination of fine suspended sediment (FSS) concentration (Meade et al. 1979). The filtrate was saved for analysis of the dissolved chemical species (see below). The remainder of the composite sample (~30 l) was passed through a Sharples centrifuge operated continuously at 15,000 RPM to obtain the fine suspended sediment fraction for chemical analysis (Hedges et al., 1986).

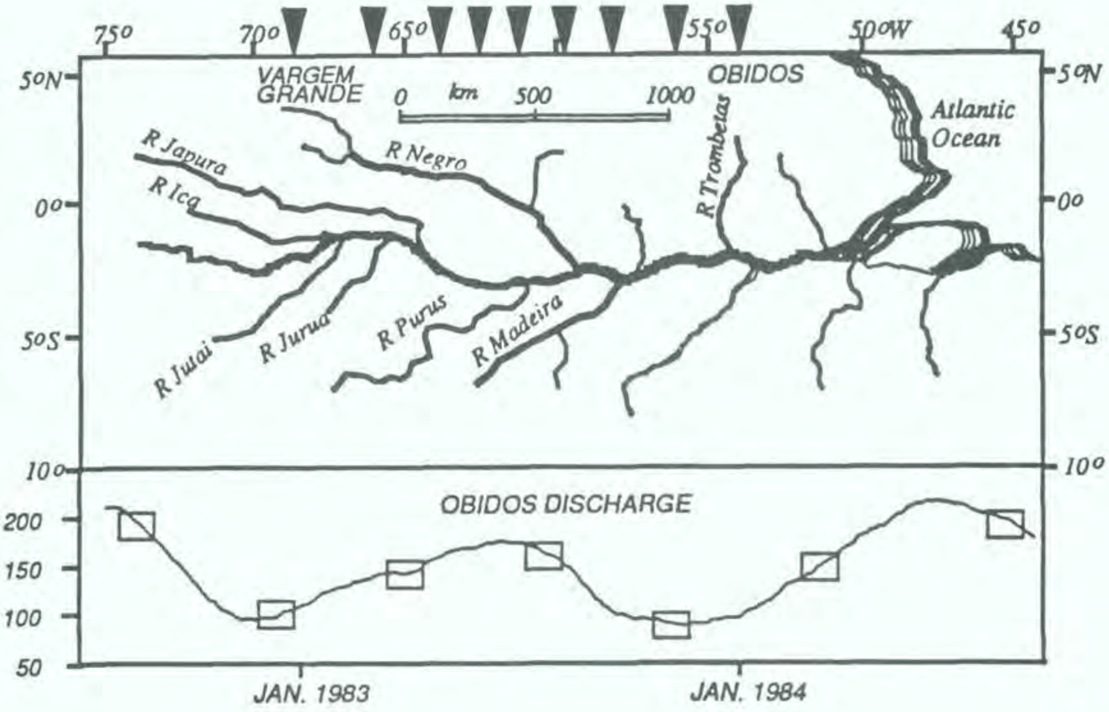

Figure 1. Upper - The Amazon mainstem and its principal tributaries. The locations of the CAMREX sampling stations are marked by filled triangles at the top. Lower - Water discharge at Obidos (10^3 m^3s^{-1}) vs. time. Boxes indicate timing of the seven cruises: cruise 2 (Aug.-Sept. 1982), cruise 3 (Nov.-Dec. 1982), cruise 4 (Mar.-Apr. 1983), cruise 5 (Jun.-Jul. 1983), cruise 6 (Oct.-Nov. 1983), cruise 7 (Feb.-Mar. 1984) and cruise 8 (Jul.-Aug. 1984).

Fine and coarse particulate organic carbon (FPOC, CPOC) and particulate nitrogen (FPN, CPN) were determined in duplicate with a Carlo-Erba elemental analyzer (model 1106) using the method of Hedges and Stern (1984). Fine and coarse total particulate P (FPP and CPP) were determined for sediment leachates that were first combusted at high temperature (550°C) and subsequently extracted with strong HCl for 24 h (Aspila et al.

1976). In addition to the total P analysis of bulk river sediments, the P content of both the woody and leafy tissue of the 20 most common floodplain plants was also determined. These analyses were made on samples previously analyzed for organic carbon and organic nitrogen (Hedges et al. 1986).

Total dissolved P and nitrogen concentrations were determined on the filtered samples using the persulfate digestion procedure of Valderama (1981). Soluble species -- phosphate (SRP), nitrate, and nitrite -- were determined with methods outlined in Strickland and Parsons (1972). Dissolved organic P was calculated as the difference between TDP and SRP. Although ammonium was routinely analyzed, it was never detected in significant amounts. Dissolved organic carbon was determined on acidified, nitrogen sparged, filtered samples by UV-enhanced persulfate oxidation (Ertell et al., 1986).

Results and discussion

Bulk P composition. Before discussion of the patterns and transport of P in the Amazon river, it is instructive to look at the overall averages of the eight cruise (Figure 2). For both P and nitrogen the highest total levels were found in the mainstem, 9.2 μM for P and 51.8 μM for nitrogen. The average tributary total P value was only about half that of the mainstem, 4.1 μM, and concentrations were halved yet again, to 2.2 μM, in the varzea waters. Of the individual forms of P, TPP was by far the dominant form accounting for up to at least 75% of the total. The remainder was about equally split between DOP and PO_4. Individually, both particulate P and PO_4 concentrations also showed decreases from the mainstem to tributaries to the varzea. Although the lowest levels of DOP were also observed in the varzea, the tributaries actually showed slightly higher DOP concentrations than the mainstem. A similar pattern was noted for total nitrogen except that the decrease in overall concentration from the mainstem (51.8 μM) to the tributaries (35.5 μM) to the varzea (27.0 μM) was not as great. The nitrogen distributions differed from those of P, however, in that the dissolved forms of nitrogen, DON and NO_3, comprised the bulk of the nitrogen, >60%.

Although no attempt was made to determine the organic and inorganic components separately, indirect evidence indicates that much of the TPP is probably inorganic in nature. Hedges et al. (1986) plotted weight percent carbon vs. weight percent nitrogen for samples obtained on the first four cruises. There was a good linear relationship between the two for both the fine and coarse sediment fractions. Importantly, although the slopes, i.e. the C:N ratios, were different for the two fractions (C:N ratios by atoms for the fine and coarse fractions were 11 and 24 respectively) in both cases the intercepts were zero indicating that all the particulate nitrogen was organic. When a similar plot was constructed for P (Figure 3a) there was no significant relationship. Rather, for any given sample (mainstem or tributary) FPP and CPP were relatively constant. On the other hand, when FPP and CPP were plotted vs suspended sediment (Figure 3b) there was a significant relationship, both for the fine fraction (slope = 0.026, intercept=.19, R^2=0.94) and the coarse fraction (slope=0.013, intercept=0.06, R^2=0.82). Also in both cases, the intercepts of the regressions were not statistically different from zero. This lack of correlation with carbon but high correlation with total suspended sediment indicates that much of the suspended particulate P and, consequently, the majority of P in transport in the Amazon river is probably inorganic.

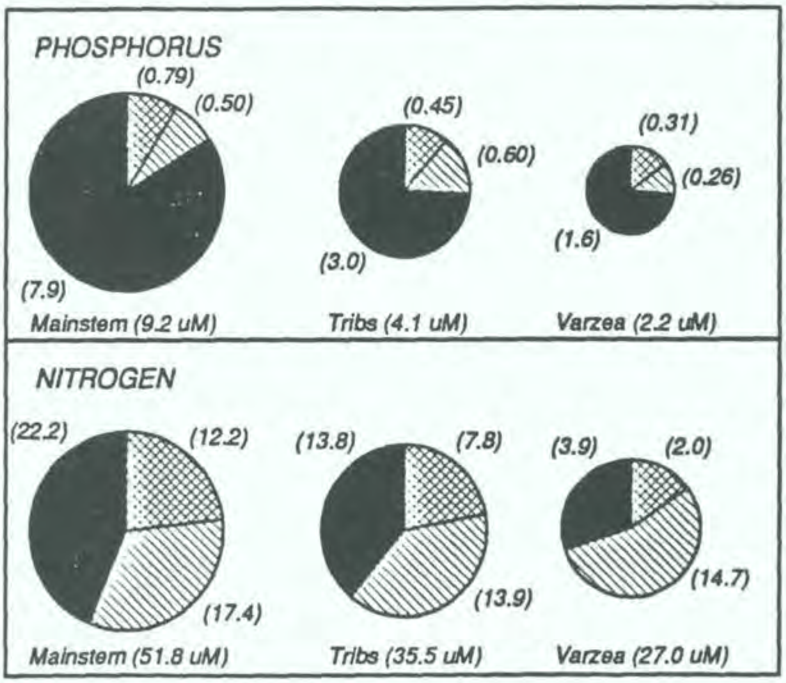

Figure 2. Concentrations of TPP (CPP plus FPP) and PON (solid), POP and DON (hatched), and PO_4 and NO_3 (cross hatched). Numbers near pie section give the actual concentration (μM) while total concentrations are given below. Phosphorus and nitrogen data from varzea waters are from Forsberg et al. 1988.

Downstream and seasonal patterns. Within any river, P, both particulate and dissolved, is transported by the water flow. Also, as tributaries contribute to the mainstem water flow they will either increase or decrease the P concentrations depending on their concentrations. Thus, in order to interpret the downstream and seasonal phosphors patterns, it is important to know the river hydrology. The downstream patterns of water discharge for the Amazon river are shown in Figure 4. In this figure, and subsequent figures, the cruises are grouped according to flood stage; thus cruises 2,5, and 8 were early falling water cruises, 3 and 6 were low water cruises and 4 and 7 were rising water cruises (see Figure 1).

Over the entire reach between Vargem Grande and Obidos, discharge showed the greatest increase, from about 40,000 m^3s^{-1} to about 180,000 m^3s^{-1}, on the early falling water cruises, 2,5, and 8. During this period, tributary discharges were still high and flood water stored on the varzea was presumably draining back into the river. Slightly smaller increases in water discharge occurred on the mid-rising water cruises, 4 and 7; 60,000 m^3s^{-1} to 160,000 m^3s^{-1}. This is a period during which tributary flows were increasing but some water was being stored on the floodplain. The smallest increases in discharge, and also the smallest discharges, were observed on the two low water cruises, 3 and 6. Thus, as tributary and varzea concentrations of the different P species were significantly different than those in the mainstem (Figure 2), tributary influences on mainstem concentration should be greatest on the early falling water cruises and least during the low water periods.

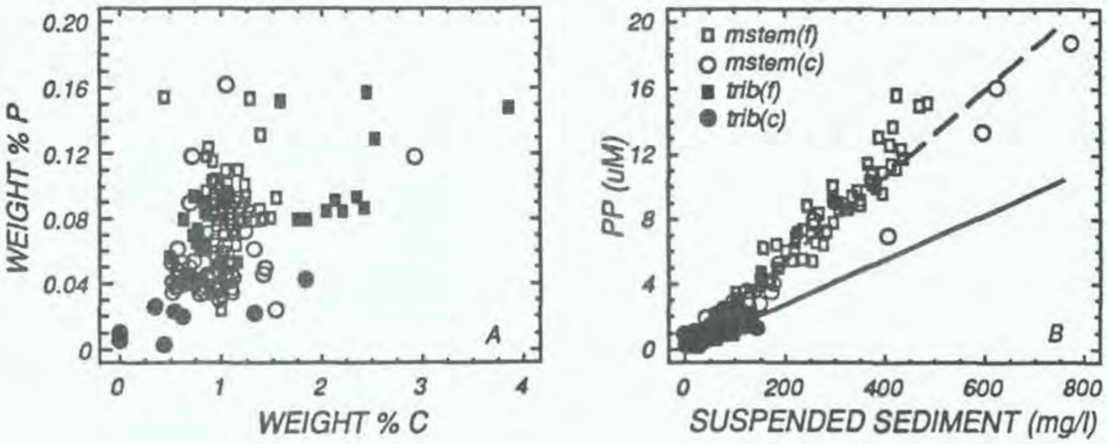

Figure 3. (A) Weight percent P vs. weight % carbon for Amazon river particulate material. (B) Particulate P concentration, FPP or CPP, vs. suspended sediment concentration in the corresponding size fraction. The different symbols identified on (B) also apply to (A).

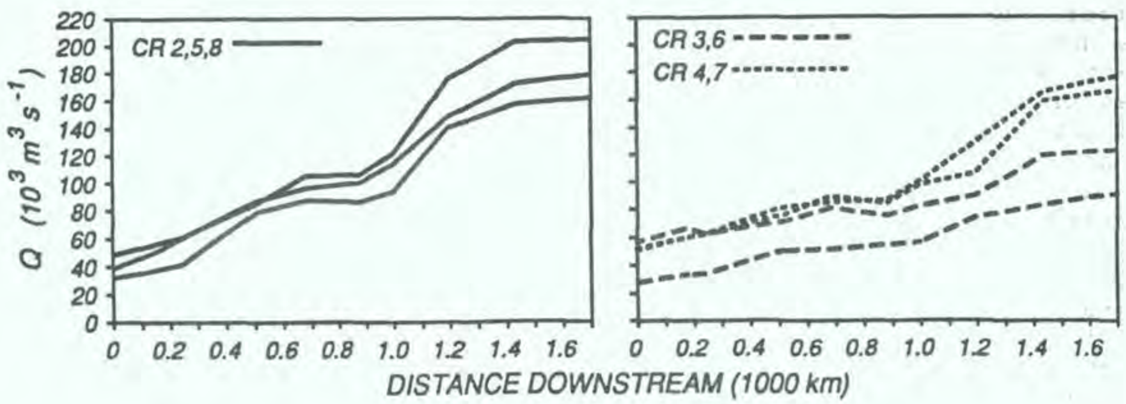

Figure 4. Water discharge vs. distance downstream from Vargem Grande.

The distributions of PO_4, DOP and TPP with respect to distance downstream are shown in Figure 5. The highest concentrations of PO_4 were observed on the two low water cruises, 3 and 6. On both cruises concentrations were high upriver, about 1.1 µM and remained so with distance downstream, decreasing only slightly near Obidos. In contrast, during the three early falling water cruises, 2,5, and 8, PO_4 concentrations were also high upriver, about 1 µM, but they then decreased rather regularly to values at Obidos of about 0.6 µM at Obidos. On the two rising water cruises, upriver concentrations were somewhat lower than during the other cruises, between 0.4 µM on cruise 4 and 0.8 µM on cruise 7. Whereas concentrations remained about constant on cruise 4, they decreased significantly on cruise 7 to about 0.5 µM at Obidos.

To understand the downstream and seasonal changes of PO_4, it must be realized that most of the inorganic solutes in the Amazon river are derived from sources in the Andes mountains (Gibbs, 1967, Stallard and Edmond, 1983). Consequently, waters originating in the Andes have much higher concentrations of inorganic nutrients than rivers draining primarily lowland and shield areas (Forsberg et al. 1988). This, for the most part, explains why the average mainstem PO_4 concentration is higher than that of the average tributary, which, in turn, is higher than the varzea (Figure 2). Thus, the two low water cruises had the highest PO_4 values because, at this time, at Vargem Grande the river contained relatively more Andean water and the amount of dilution by subsequent tributary inputs was small compared with other periods. During the early falling water cruises the PO_4 levels at Vargem Grande were also relatively high. However, draining varzea waters and large tributary inputs between Vargem Grande and Obidos resulted in the largest increases in water discharge and, concomitantly, dilution of PO_4. An intermediate situation was observed on the two rising water cruises, with somewhat lower PO_4 concentrations upstream and also somewhat smaller increases in discharge and dilution downstream to Obidos. Thus, much of the pattern of seasonal and downstream changes in the PO_4 can be explained by dilution of mainstem water with PO_4-poor tributary and varzea waters.

Although on any given cruise there were large fluctuations in the DOP concentration downstream, in general, there was no overall trend of either increase or decrease (Figure 5). There were however, some interesting differences between the concentrations of DOP on the different cruises. The highest DOP concentrations, about 1 µM were observed on the early rising water cruises, 4 and 7, while the lowest values, about 0.3 µM, were seen on the low water cruises, 3 and 6. The early falling water cruises had intermediate values of about 0.6 µM. The lack of a distinct downstream trend is explained by the fact that there are apparently no systematic differences between mainstem and tributary concentrations of DOP. Instead, differences are relatively random (probably dependent on local geology, vegetation and climate) and such that the resultant addition of tributary water to the mainstem, while producing fluctuations, does not change the overall DOP concentration. The differences between cruises are more difficult to explain. As pointed out above, the low water cruises had the least dilution by lowland tributary input. This would indicate that water of Andean origin should have low values of DOP (average low water DOP=0.22 µM). The highest DOP concentrations were encountered on the rising water cruises (average DOP=0.97 µM). Why rising water concentrations were higher than early falling water concentrations (average DOP=0.47 µM) is not clear at present.

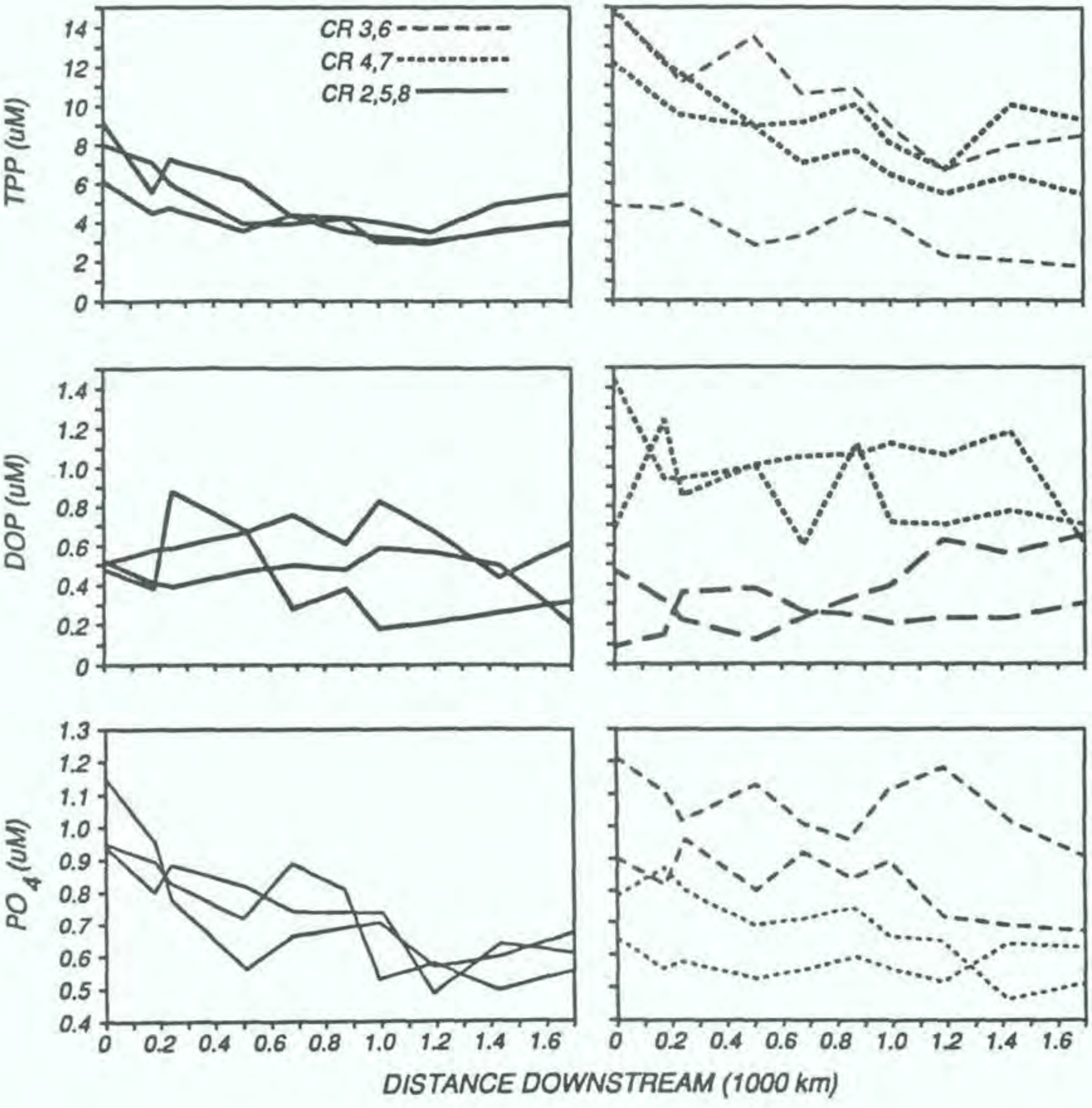

Figure 5. Concentration vs. distance downstream (in thousands of km) for particulate P, TPP, dissolved organic P, DOP, and PO_4. Cruise number and line type given in upper left panel apply to all pannels.

In all cases TPP concentrations were greatest at Vargem Grande and decreased down river. Concentrations and downstream patterns were very similar on all three early falling water cruises. Concentrations were greatest upstream, 6 µM to 10 µM, and decreased to about 4 µM over the first 1000 km, beyond which they remained relatively constant. In contrast, the two rising water cruises had much higher concentrations upstream, 12 µM to 15 µM, and these concentrations decreased more or less continuously to between 10 and 6 µM at Obidos. The two low water cruises were distinctly different with cruise 3 showing a pattern similar to the rising water cruises and cruise 6 resembling the early falling water cruises. Due to the close correlation between TPP and TSS, the downstream distributions and seasonal variations in TPP will parallel those of the bulk suspended sediment, and the hydrology of deposition and resuspension will control the distributions. The TPP is discussed further below.

The downstream patterns presented above present 7 "snapshots" of the P distributions in the Amazon over a two year period. They probably set a reasonable range for concentrations expected at other times of the year and they are of great value for the calculation of weathering rate and riverine transport of P to the ocean. However, the explanations of the seasonal and downstream patterns are tentative and more data on seasonality, especially in the tributaries, is necessary to complete the analysis.

Phosphorus mass balance of the river. As seen above, it is possible to qualitatively explain much of the seasonal and downstream pattern of the dissolved components in the Amazon river simply by mixing of mainstem waters and tributary waters. However, mixing is not the only process affecting concentrations, in situ reaction could also be important. For example, PO_4 participates in many inorganic reactions (House and Casey, 1989) and is returned to solution during the decomposition of organic matter. The importance and nature of such in situ processes in the Amazon River can be investigated through the use of mass balances similar to those constructed for C (Richey, et al. in press) and O_2 (Devol et al. 1987).

Mass balances were done over the entire study reach. The mass flux into the reach at Vargem Grande, F_i, and the mass flux out of the reach at Obidos, F_o, were calculated from the product of the water discharge (m^3s^{-1}) and concentration (mass m^{-3}). Individual tributary inputs to the mainstem were calculated in a similar manner, and summed to arrive at the total tributary flux, F_t. In addition to the major tributaries, water also entered the mainstem via minor tributaries and varzea drainage. Although these water amounts were not directly measured, they could be estimated by taking the difference between the water outflow at Obidos and the sum of the measured water inflows, i.e. Vargem Grande plus tributaries. The mass fluxes of these diffuse inputs, F_v, could then estimated from the average varzea concentrations (Figure 2). The mass fluxes were then used to calculate a balance parameter β:

$$\beta = \frac{\textbf{observed}}{\textbf{expected}} -1 = \frac{F_i + F_t + F_v}{F_o} -1$$

Thus, if the sum of the inputs equals the flux out at Obidos β=0. This indicates a balance and no in situ production or consumption is required. However, positive values of β require an in situ source, while negative values require a sink.

The mass balance parameter, β, for the fine particulate phases of total suspended matter, carbon and P are shown in Figure 6. Some interesting patterns are evident. (The same patterns hold for fine fraction PON and all the coarse fraction components). First, as would be expected from their rather constant ratios with total particulate material (Figure 3), both P and carbon mirror the trends shown by FSS, even though they are only small components of the FSS (C ~1.0 wt% and P ~0.01 wt%). Secondly, the data again tend to differentiate according to flood stage. On all three early falling water cruises, mass balances are positive and near 1.0. Essentially, this means that on these cruises about twice as much material was leaving the river reach at Obidos as was entering. This material must have been supplied either by resuspension of sediments stored within the river channel itself, or by erosion of the river banks.

Conversely, on all other cruises the mass balance was either negative or zero. During these cruises net deposition within the channel is required to balance the excess of input over output. Thus, as noted above, the hydrology of sediment deposition and resuspension control both the downstream and seasonal pattern of particulate concentration in the Amazon River.

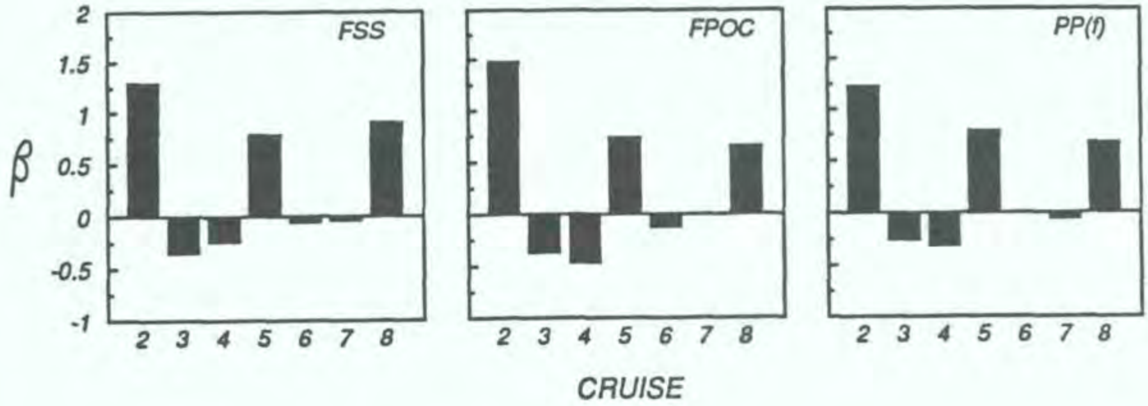

Figure 6. Mass balances, ß, for fine suspended sediment (FSS), fine particulate organic carbon (FPOC) and fine particulate P (FPP). Positive values of ß indicate a within river source, while negative values indicate a sink.

The mass balances for the dissolved P (Figure 7) forms differed significantly from those seen for the particulate phases (Again, although not shown, dissolved nitrogen species mirror those of P.) Even though the absolute magnitude of DOP changed from cruise to cruise, the mass balance parameter, ß, for DOP was near zero on all cruises. This good overall balance suggests that DOP in the amazon river is conservative and is not produced or consumed in any significant amount. This conclusion is in agreement with that of Richey et al. (in press) who suggest that the majority of DOC in transport is also refractory.

In contrast to DOP, the opposite situation was observed for PO_4. On all cruises there was an excess of PO_4 over what could be explained by the sum of the inputs. Because the mass balance accounts for fluvial inputs, this excess PO_4 probably comes from in situ reaction; either through: 1) dissolution of primary and secondary minerals or, 2) desorption of PO_4 associated with clay minerals and metal oxides or, 3) liberation of organically bound PO_4 during organic matter oxidation.

In addition to positive values of ß for PO_4, there were also positive mass balances for NO_3 and, after accounting for gas exchange, for total CO_2 (Richey et al. in press). If the excess CO_2, NO_3 and PO_4 are due to decomposition of organic matter then the ratio of the excess fluxes (Moles $time^{-1}$) should give the elemental ratios of the organic matter being oxidized. However, if the excess PO_4 results primarily from dissolution or desorption of mineral phases then the C:N ratio of the dissolved species will remain constant and the C:P ratio will decrease. The apparent C:N, N:P and C:N:P ratios of the flux excess along with the corresponding ratios of the potential source materials are given in Table 1.

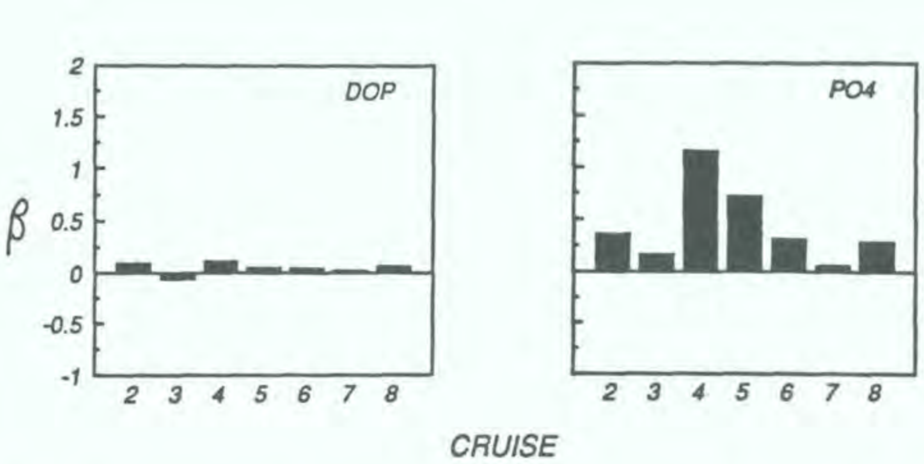

Figure 7. Mass balance anomalies for dissolved organic P (DOP) and dissolved inorganic P (PO_4).

The C:N and N:P ratios of the apparent flux excess were 54:1 and 34:1, respectively. In contrast, the apparent C:N and N:P ratios of the bulk suspended sediment were 10:1 and 4:1, respectively. Hedges et al. (1986) and Richey et al. (in press) have shown that there are virtually no inorganic forms of carbon or nitrogen contained within the bulk suspended sediment -- in other words, all the C and N is organic. Thus, oxidation of the bulk organic mater in suspension would result in a C:N flux excess of the dissolved species that is much lower than that observed. Similarly, although we do not know the relative contributions of organic and inorganic P to the overall bulk suspended P, stoichiometric conversion to dissolved forms would result in low apparent flux excess N:P ratios also. Again, these were not observed. Thus, it is unlikely that the observed flux excess results from stoichiometric decomposition of the bulk suspended matter only. Indeed, the observed excess C:N and N:P ratios are more similar to those of the potential organic substrates, which strongly suggests that the majority of the flux excess is derived from oxidation of organic substrates such as wood, leaf and macrophyte tissue as well DOC. Furthermore, although we can not rule out dissolution and desorption of inorganic P phases as a significant source of P, we feel that it is unlikely because of the similarity of the N:P ratios of the flux excess and the potential substrates. Dissolution and desorption of mineral phases would result in lowered C:P and N:P ratios relative to the known organic substrates. For dissolution and desorption to be important contributors to the flux excess and still produce the observed C:N:P flux excess ratios requires either a yet unidentified organic substrate rich in C and N relative to P, or preferential liberation C and N relative to P during oxidation of the known organic forms. Thus, we conclude that the excess PO_4 in the Amazon river is generated primarily by organic matter decomposition and that dissolution and desorption of inorganic P is probably of only minor importance. However, identification of the actual organic matter undergoing decomposition is not possible from the apparent flux excess because the C:N ratio does not match any of the identified organic matter pools. The excess may arise from mineralization of a mixture of the pools, from the mineralization of a subcomponent of one of the pools or a combination of these processes.

Table. 1. The apparent elemental ratios (by atoms) calculated from the average flux imbalances. The total CO_2 balance included respiration and was taken from Richey et al. (in press). Also listed are the elemental ratios of common organic tissues found on the floodplain as well as the riverine DOM and TSS

	C:N	N:P	C:N:P
Apparent flux excess ratio	54:1	34:1	1840:34:1
Average varzea wood tissue	156:1	38:1	5940:38:1
Average varzea leaf tissue	28:1	36:1	1000:36:1
Average macrophyte tissue	32:1	36:1	1160:36:1
Mainstem TSS	10:1	4:1	40:4:1
Mainstem DOM (DOC:DON:DOP)	23:1	27:1	607:27:

Table 2. Concentrations and transports of nitrogen and P species of the Amazon river and their global significance. World average rivers are from Meybeck (1982) and are for natural rivers

	Concentration (μM)		Transport (Tg y^{-1})		
	Amazon	World	Amazon	World	% Total
TPP	6.2	17.0	1.23	19.0	7%
DOP	0.5	0.9	0.10	1.0	10%
PO_4	0.8	0.4	0.13	0.4	30%
PON	19.9		--	1.62	--
DON	13.7	27.0	0.62	14.7	6%
NO_3	12.0	7.0	0.83	3.7	23%

Nitrogen and P transport from the continent to the ocean. From the P and nitrogen concentrations determined at the downstream station, Obidos, during the seven CAMREX cruises it is possible to calculate average concentrations that can then be compared to "world average" river values reported by Meybeck (1982). Furthermore, when combined with Obidos water discharge measurements it is possible to quantitatively evaluate the transport of these elements from the continents to the oceans by the world's largest river. The results of both calculations are shown in Table 2. (The average concentrations given in Table 2 are slightly different from those of Figure 2 because the values in Figure 2 include all mainstem stations, not just Obidos.)

Comparison of the concentrations of dissolved and particulate phases of nitrogen and P in the Amazon river with those of the "world average" river reveals that although they may vary by a factor of 2, the Amazon is chemically quite ordinary. For both elements, particulate and organic concentrations are lower than the world average while the inorganic forms are higher. However, because the water discharge of the Amazon is so large, the overall transport of nitrogen and P from the continent to the ocean is a significant fraction of the total riverine transport. The Amazon river alone is responsible for about 7% of the particulate P transport, about 10% of the DOP transport and about one third of the total natural PO_4 transport. The river is also highly significant in global nitrogen transport. Thus, in terms of overall transport, chemically the Amazon river is quite typical; nevertheless its great water discharge makes it significant in the global cycles of nitrogen and P.

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TRANSPORT OF PHOSPHORUS BY VENEZUELAN RIVERS

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Introduction

Phosphorus as an essential nutrient is tied to primary productivity and to the biogeochemical cycles of other major elements. Richey (1983), Froelich (1984) and Froelich et al. (1982) evaluated the global P cycle and showed that the major P reservoirs are marine sediments and soils and that the major P input to the ocean is by rivers. Meybeck (1982) has estimated the fluvial flux of P to be 10^9 kg y^{-1} for total dissolved phosphorus (TDP) and 20 x 10^9 kg y^{-1} for total particulate phosphorus (TPP). Most of the data on fluvial fluxes of TDP and TPP have been generated in the last three decades when many disturbances in the natural environments have been occurring due to the increase in the world human population. Humans have intervened in the P cycle by increasing erosion; by mining and shipping phosphate rocks; by using fertilizers; by the production of industrial, human and animal wastes, and of detergents (Maki et al., 1984).

This paper presents the results of hydrogeochemical investigations carried out in Venezuelan Rivers with the purpose of quantifying the total P input into the Venezuelan Caribbean Sea and distinguishing between its sources (natural and anthropogenic) and its dissolved (orthophosphate, polyphosphates and organic) and particulate (inorganic and organic) forms.

Figure 1. Map of the Venezuelan hydrographic basins

Description of the study area

The Venezuelan fluvial systems can be divided into three groups, contributories to the Atlantic Ocean, the Caribbean Sea and to Lake Valencia (Zinck, 1982). The Atlantic Ocean basin consists of the Orinoco, San Juan-Guanipa, Cuyuni and Negro drainages. The basin draining into the Caribbean Sea is subdivided into those rivers draining directly into Sea and those draining through Lake Maracaibo (Lake Maracaibo basin) (Figure 1). The Caribbean Sea, Lake Maracaibo and Lake Valencia basins cover 18% of the Venezuelan territory, receive about 3% of the total water discharge, and contain more than 70% of the total Venezuelan population.

The Caribbean Sea basin (Figure 2) has an area of 85.700 km^2 and has been divided into eight subbasins: 1) Maticora-Mitare; 2) Ricoa-Hueque; 3) Tucurere-Tocuyo; 4) Aroa-Yaracuy; 5) Litoral Central; 6) Curiepe-Uchire; 7) Unare; and 8) Neverí-Cariaco. The Curiepe-Uchire drainages present the highest population density (530 person km^{-2}) and the lowest annual per capita water discharge (820 m^3 $person^{-1}$ y^{-1}); therefore, the Curiepe-Uchire watershed has been the most extensively studied and was used to estimate the total anthropogenic input of P as a function of the number of inhabitants in the area.

The Maticora-Mitare, Ricoa-Hueque, Tucurere-Tocuyo and Aroa-Yaracuy watersheds integrate the western Caribbean Sea subbasin, whereas the Litoral Central, Curiepe-Uchire, Unare and Neverí-Cariaco drainages constitute the eastern part of the Caribbean Sea basin (Figure 2).

Material and methods

Water samples have been collected since March of 1987 in the rivers draining directly into the Caribbean Sea (Figure 2), except the Tuy River which has been monitored since April of 1979. The water samples were filtered through a 0.45 μm membrane filter. The filtrate was acidified with 8 M HNO_3 and stored in a 1 liter polyethylene bottle. The membrane filter was dried (40-45°C) in the laboratory and weighted on an analytical balance. Total dissolved (TDP) and dissolved reactive (DRP) phosphorus were determined in the filtered water samples by colorimetry (Skougstad et al., 1979). The total particulate phosphorus (TPP) was determined by plasma emission spectrometry after digestion of the sample. Particulate organic phosphorus (POP) was estimated, after extracting most of the inorganic P with 1 N HCL (Aspila et al., 1976), by oxidation of the HCl-residue with 30% H_2O_2 (Filipek and Owen, 1981) and extraction with 1 N HCl.

Results and discussion

Natural background. The results obtained in this study for each drainage basin were flow-weighted averaged and are summarized in Table 1. It is observed that the concentrations of DRP as well as TDP are essentially constant for each watershed and that DRP constitutes more than 75% of the average TDP for the western and eastern parts of the Caribbean Sea basin. These results indicate that both DRP and TDP concentration values seem to be independent of the lithological, hydrological and geomorphological characteristics of each watershed. However, the TPP concentration values are lower for the western sub-basins. This may be attributed to lithological differences between the western and eastern sub-basins (Colina et al. 1989).

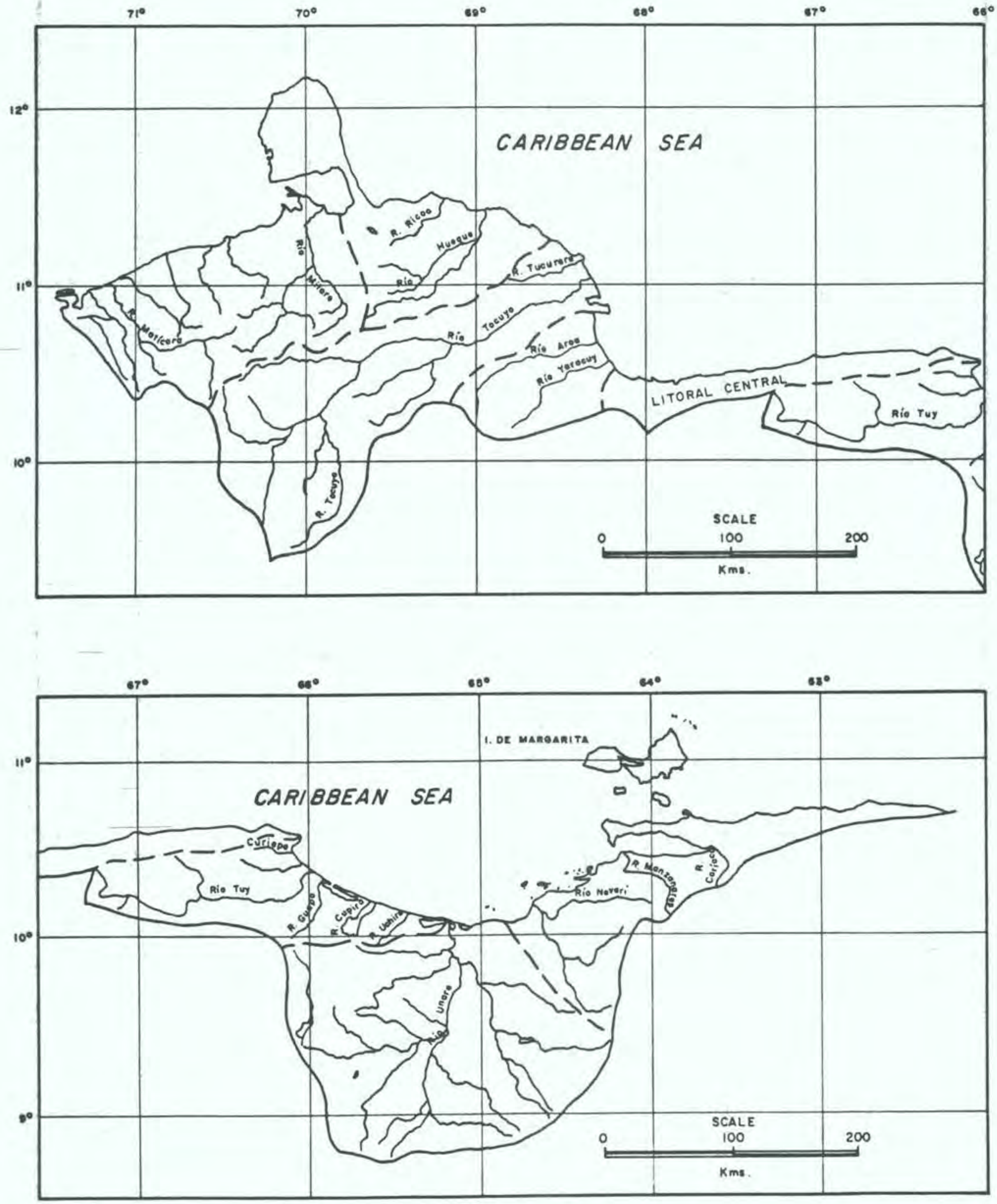

Figure 2. Detailed map of the Caribbean Sea basin

Table 1. Natural background concentrations and annual yields of dissolved reactive (DRP), total dissolved (TDP) and total particulate (TPP) phosphorus for the Caribbean Sea basin

Basin	Area (km^2)	Runoff (mm)	DRP ($\mu g\ l^{-1}$)	TDP ($\mu g\ l^{-1}$)	TPP ($mg\ kg^{-1}$)	TDP ($kg\ km^{-2}$)	TPP ($kg\ km^{-2}$)
(1) Maticora-Mitare	12000	110	16	22	400	2.4	1140
(2) Ricoa-Hueque	10000	130	13	20	480	2.6	427
(3) Tucurere-Tocuyo	18000	92	14	16	510	1.5	1120
(4) Aroa-Yaracuy	5200	203	16	20	470	4.1	306
Western Subbasin	45200	118	15	19	461	2.2	878
(5) Litoral Central	3300	218	13	16	670	3.5	288
(6) Curiepe-Uchire	8400	410	12	16	710	6.6	540
(7) Unare	22300	65	16	17	490	1.1	74
(8) Neveri-Cariaco	6500	240	16	18	650	4.3	169
Eastern Subbasin	40500	177	14	17	640	3.0	203
Whole Basin	85700	146	14	18	485	2.6	559

The TDP and TPP concentration values are within the range reported in the literature (Berner and Berner, 1987; Lewis and Saunders, 1989; Meybeck, 1982; Probst, 1985) for pristine fluvial systems. The dissolved organic phosphorus (DOP) concentration estimated as the difference between TDP and DRP, for the whole Caribbean sea basin is equal to 4 μg L^{-1}. This DOP value is considered low when compared with the value of 15 μg L^{-1} reported by Meybeck (1982) as the world average and the value of 11 μg L^{-1} reported by Lewis and Saunders (1989) for the Río Orinoco.

The low DOP concentration values found in this study are a reflection of the low dissolved organic carbon (DOC) concentrations of only about 2 mg L^{-1} (Colina et al. 1989). The DOC/DOP ratio of 500 obtained from these values for the whole Caribbean Sea basin can be considered essentially the same, within the error of those estimations, to the DOC/DOP weight ratios of 367 and 404 calculated from Meybeck's (1982) and Lewis and Saunders (1989) data, respectively.

The variabilities observed (Table 1) for the TDP and TPP total yield data reflect the runoff and sediment yield differences among the sub-basins respectively. The Curiepe-Uchire and Unare subbasins, having the highest and lowest runoff values, present the highest and lowest TDP yield values, respectively. The highest TPP yield values occur in the Maticora-Mitare and Tucurere-Tocuyo watershed while the lowest TPP yield value is found in the Unare watershed. The ratio between the highest and lowest TPP yield values is equal to 15, whereas the ratio for the highest and lowest TDP values is 6. This shows a higher variability for the TPP yield values which is reflected in the TDP and TPP average values obtained for the western and eastern subbasins (Table 1). The TPP/TDP yield ratios for the western and eastern subbasins are 400 and 68, respectively and are indicative of a higher sediment yield in the western part of the basin where the particulate P constitutes more than 98% of the total P yield.

The western subbasin contributes 45% of the TDP and 83% of the TPP exported from the Caribbean Sea basin, and therefore accounts for most of the annual fluvial transport of particulate P into the Caribbean Sea even though the average P concentration in the particulates is lower than in the eastern terrain.

Anthropogenic input. Table 2 summarizes the P export rates estimated in this study for the Caribbean Sea basin and those reported for the Lake Maracaibo (Parra Pardi et al., 1979) and Río Orinoco (Lewis and Saunders, 1989) basins. The Curiepe-Uchire watershed has been used to estimate the per capita P input into the Caribbean Sea (Ramírez et al., 1988) by quantifying the total (anthropogenic + natural) and the natural annual transport of the different forms of P. The anthropogenic input was calculated by difference between the total and the natural transport and then divided by the total number of persons in the watershed. The results for DRP, TDP, TPP and TP were 0.17, 0.47, 1.63 and 2.10 kg $person^{-1}$ y^{-1}, respectively. The per capita TP value of 2.10 kg $person^{-1}$ y^{-1} exceeds the TP value of 1.3 kg $person^{-1}$ y^{-1} reported for USA domestic wastewater (Tchobanoglous and Schroeder, 1985) and reflects the additional industrial, fertilizer and animal waste inputs of P into the fluvial system. The DRP value of 0.17 kg $person^{-1}$ y^{-1} is very close to the DRP value of 0.14 kg $person^{-1}$ y^{-1} estimated from Meybeck's (1982) demographic index data for Venezuela. The fact that both TP and DRP per capita values are greater than those reported in the literature may be attributed to the lack of wastewater treatments in the study area. The annual per capita inputs of

DRP, TDP, TPP and TP together with the population densities for 1989 were used to estimate the anthropogenic P yield data (Table 2) for the Curiepe-Uchire subbasin (530 persons km^{-2}) and for the whole Caribbean Sea basin (105 persons km^{-2}).

The P input from anthropogenic sources in the Curiepe-Uchire watershed exceeds the natural contribution, especially for the dissolved forms of DRP and TDP. This is indicative of the degree of pollution that exists in this North-Central part of Venezuela. The DRP from anthropogenic origin constitutes 37% of the TDP and the difference between TDP and DRP is essentially all polyphosphates because the dissolved organic carbon concentration is low (Ramírez et al., 1988; Colina et al., 1989) for all the polluted rivers studied. This means that the larger difference observed between the natural and anthropogenically derived P is in the polyphosphate form (159 kg km^{-2} y^{-1}) which suggests a high contribution of dissolved P from detergents (Maki et al., 1984). This anthropogenic input of pholyphosphates exceeds the total P applied as fertilizers (115 kg km^{-2} y^{-1}) in 1988 (Rodríguez, 1989), despite the possible hydrolysis during the fluvial transport under tropical climate conditions (Zinder et al., 1984).

Table 2. Dissolved reactive (DRP), total dissolved (TDP), particulate organic (POP), total particulate (TPP) and total (TP) phosphorus yields for four major basins

Basins	Source	DRP	TDP	TPP	POP	TP
		kg km^{-2} y^{-1}				
Curiepe-Uchire[1]						
	Background	4.9	6.6	540	37	547
	Anthropogenic	90	249	864	217	1113
	Total	**95**	**256**	**1404**	**254**	**1660**
Caribbean Sea[1]						
	Background	2.0	2.6	559	56	562
	Anthropogenic	18	50	170	43	220
	Total	**20**	**53**	**729**	**99**	**782**
Lake Maracaibo[2]	**Total**	**145**	**406**	**459**	-	**865**
Rio Orinoco[3]	**Total**	**11**	**24**	**48**	-	**72**

1. This study.
2. Parra Pardi et al. (1979).
3. Lewis and Saunders (1989).

The total anthropogenic input of P in the Curiepe-Uchire watershed corresponds to 1110 kg km^{-2} y^{-1} and is about ten times greater than the total P value of 115 kg km^{-2} y^{-1} (Rodríguez, 1989) applied as fertilizers in 1988. This suggests that the contribution of P from the use of fertilizers to the total P export rate derived from anthropogenic sources is very low. The total P yield (1660 kg km^{-2} y^{-1}) for the Curiepe-Uchire subbasin is very high but the TP export rate for the Lake Valencia (3310 kg km^{-2} y^{-1}) is even higher (Lewis and Weibezahn, 1983). Lake Valencia is the Venezuelan hydrographic basin with the highest population and industrial densities.

The DRP and TDP values for Lake Maracaibo clearly exceed those of the Caribbean Sea basin (Table 2). This indicates that most of the P from anthropogenic sources in the Lake Maracaibo is classified as dissolved. On this basis, a TP background value of 400 kg km^{-2} y^{-1} has been estimated for the Lake Maracaibo according with the natural background established for the Caribbean Sea. The Río Orinoco shows a TDP value four times higher than the TDP background value of the Curiepe-Uchire watershed but a TPP value which is much lower than any other (Table 2).

Particulate organic phosphorus. Data for organic P has been obtained for the suspended solids and bottom sediments of the Curiepe-Uchire watershed. The particulate organic carbon (POC)/particulate organic P (POP) weight ratios obtained in this study were 107 and 204 for heavily polluted and non-polluted sediments, respectively. These results show that the particulate organic matter (POM) from anthropogenic sources is about twice richer in P than the POM from natural sources. In addition, these results plus the natural background concentration of POC in fluvial particulates (10 mg g^{-1}) and the anthropogenic input of POC (44 kg $person^{-1}$ y^{-1}) reported by Ramírez et al. (1988) permit to estimate a natural background concentration of organic P in particulates of 49 mg kg^{-1}, and an annual per capita input of POP of 0.41 kg $person^{-1}$ y^{-1}. The POP data for the Curiepe-Uchire and whole Caribbean Sea basins are summarized in Table 2.

The POP data (Table 2) show that the particulate inorganic P (PIP)/POP ratio is equal to 3.0 for the anthropogenic sources and equal to 8.9 for the whole Caribbean Sea basin. This indicates that TPP from anthropogenic sources contains a greater proportion of organic P which could participate in the marine biogeochemical P cycle than TPP from natural sources.

Total phosphorus transport. Table 3 summarizes the annual transport of P from Venezuelan Land to the sea. It is observed that the Río Orinoco contributes 98% of the TDP but only 35% of the TPP fluxes from natural sources. This high contribution in TDP is attributed to the high runoff (1150 mm) of the Río Orinoco while the low contribution in TPP is mainly due to a lower sediment yield than the fluvial world-wide average (Milliman and Meade, 1983). The anthropogenic sources contribute 63% of the TDP and 13% of the TPP fluxes. It is clear that the most biogeochemical active form of P, TDP, has strong anthropogenic components.

The fact that the Caribbean Sea and Lake Maracaibo basins supply 68% of the total P transported to the sea from Venezuelan Land indicate the importance of high population and industrial densities in young geological terrains when estimating global P fluxes.

Table 3. Natural and anthropogenic exports of total dissolved (TDP), total particulate (TPP) and total (TP) P from Venezuelan basins in 10^6 Kg y^{-1}

Basin	Source	TDP	TPP	TP
Caribbean Sea[1]				
	Natural	0.22	47.9	48.1
	Antropogenic	4.29	14.6	18.9
Lake Maracaibo[2]				
	Natural	0.3	35.6	35.9
	Antropogenic	36.1	5.3	41.4
Rio Orinoco[3]				
	Natural	23	45	68
	Anthropogenic	0	0	0
Total Exported				
	Natural	23.5	128.5	152.0
	Anthropogenic	40.4	19.9	60.3

1. This study.
2. Parra Pardi et al. (1979).
3. Lewis and Saunders (1989).

The total P flux of 213 x 10^6 kg y^{-1} (Table 3) is larger than the total P (100 x 10^6 kg y^{-1}) used as fertilizers in Venezuela in 1988 (Rodríguez, 1989). This suggests that the P derived from anthropogenic sources should be recycled, especially when considering that most of the P fertilizers used in Venezuela are imported (Elizalde, 1989; Rodríguez, 1989).

Conclusions

The natural background values of P obtained in this study for the Caribbean Sea watershed were 2.0, 0.58, 503 and 56 kg km^{-2} y^{-1} for dissolved inorganic (DIP), dissolved organic (DOP), particulate inorganic (PIP) and particulate organic (POP) phosphorus, respectively. The total phosphorus (TP) yield was 562 kg km^{-2} y^{-1} and is 2.67 times greater than the world average of 210 kg km^{-2} y^{-1} (Meybeck, 1982), although the world runoff average (374 mm) is 2.56 times larger than the runoff for the Caribbean Sea watershed (146 mm).

The anthropogenically-derived P yields in the study area were 49, 1.0, 127, and 43 kg km^{-2} y^{-1} for DIP, DOP, PIP and POP respectively. These values illustrate a very high anthropogenic influence in the dissolved P and in the POP when compared with the natural background. The P from anthropogenic sources transported into the sea is contributing with a very large fraction of the two forms of P having a high bioavailability, TDP and POP.

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INPUT, RECYCLING, AND EXPORT OF N AND P ON THE AMAZON FLOODPLAIN AT LAKE CALADO

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Introduction

Lowland floodplains of large rivers such as the Amazon are important components of the biogeochemistry and ecology of riverine systems. Watersheds with wetlands export more organic carbon per unit area than drainages without wetlands (Schlesinger and Melack 1981), and destruction of existing wetlands contributes significant amounts of stored carbon to the atmosphere (de la Cruz 1986). In the Amazon, aquatic wetland grasses contribute ca. 5% to the total organic matter transported downstream (Hedges et al. 1986) but occupy only ca. 0.5% of the basin. The mosaic of flooded forest, lakes, and floating macrophytes in the central Amazon floodplain make a significant contribution to tropospheric methane (Crill et al. 1988, Bartlett et al. 1988, Devol et al. 1988), and the fishery potential of large rivers is closely tied to the area of floodplain (Welcome 1979, Lesack 1986). The majority of fishes harvested in the Amazon basin obtain nutrition in flooded forests (Goulding 1980) or from organic matter derived from floodplain phytoplankton (Araujo-Lima et al. 1986), although the trophic dynamics are not clear (Bayley 1989).

A key characteristic of floodplains that leads to high productivity and carbon export may be nutrient utilization and recycling. Longer storage times, more circuitous routing for water, and adsorption and immobilization by abiotic processes and biotic uptake may enhance the efficiency of nutrient utilization on floodplains. Important processes are biological uptake and sedimentation of N and P (Fisher et al. 1988, Smith-Morrill 1987), adsorption of dissolved inorganic N and P by sinking particulates (Smith and Fisher, submitted), and long-term storage in the sediments of organic C, N, and P (Smith et al, submitted). The utilization and retention of nutrients on the floodplain by these processes is analogous to the nutrient spiralling concept (Newbold et al. 1982) that has been applied to small streams.

Floodplains bordering large rivers differ in several important spects from the riparian areas of small streams. Along the Amazon River, he floodplain contains interconnected lakes and seasonally inundated lands ith channels to the river, allowing slow circulation of water on the floodplain parallel to that within the river. Because of the scale of floodplains associated with large rivers, these are significant features which provide important seasonal habitat for plants and animals with a large and predictable hydrograph (Junk et al. 1989). In the case of the Amazon, the annual rise and fall is approximately 10 m, with low water during Oct.-Dec. and high water in June. In contrast to a small stream, where the flooded area might be on the order of 1-10 m and is inundated irregularly by storm events, the Amazon's floodplain is on the order of 1-10 km and is regularly and predictably inundated.

Junk et al. (1989) have termed the seasonally flooded land on the Amazon floodplain the Aquatic Terrestrial Transition Zone or ATTZ. This zone is alternatively exploited by biota as an aquatic and terrestrial environment. Because of the large scale, large range of the hydrograph, and relatively flat topography, the ATTZ is large in area and is functionally equivalent to the littoral zone of a shallow lake.

There are two important types of ATTZ's on the Amazon River floodplain. The seasonally flooded lands near the river and near the terrestrial margins of the floodplain are morphologically distinct and have different plant communities. Each area is characterized by horizontal and vertical gradients of conductivity, turbidity, and nutrients. The river - lake interface is the region near the river where mixing of river and lake water occurs. It is usually located in the channel to the river or in a broad open area near the river channel (see Figure 1) with a characteristic plant cover of annual grasses and forbs which begin as rooted terrestrial plants and develop into a floating, tangled mat of emergent macrophytes known as the "floating meadows" (Junk 1984).

The stream - lake interface is an area on the upland side of the floodplain where gradients indicate mixing of upland stream and lake waters. Stream waters are lower in conductivity and turbidity, but contain higher concentrations of nitrate than lake waters. This interface usually occurs in narrow stream valleys adjacent to unflooded (upland) areas (see Figure 1) with a special forest of reduced species abundance adapted to partial or complete inundation for months at a time ("flooded forest", Prance 1979).

Each interface is characterized by gradients of conductivity, turbidity, and nutrients. Conductivity and turbidity are low in the streams, increase rapidly across the stream - lake interface, are relatively constant through the open areas of the permanent lake, and increase rapidly again across the river - lake interface to the turbid, high conductivity river waters. In contrast, nutrients are highest both in streams and in the river due to biological uptake on the floodplain by plankton in the pelagic zone, and by plants, epiphytes, and sediments in both interfaces of the ATTZ (Fisher and Parsley 1979, Alves and Fisher 1987, Engle and Melack 1990). Movement of water onto the floodplain across either interface results in nutrient utilization.

This document is a review of a 10 year research project on nutrient inputs, recycling, and losses in Lake Calado, a permanent lake on the Amazon floodplain. The project is a joint effort between the University of Maryland - CEES, the University of California - Santa Barbara, and the National Institute of Amazon Research (INPA) in Manaus, Brazil. The research has been funded by the Ecosystems Studies Program of NSF and the Brazilian CNPq since 1979, and the project is officially recognized by NSF and CNPq as a joint US - Brazilian research effort.

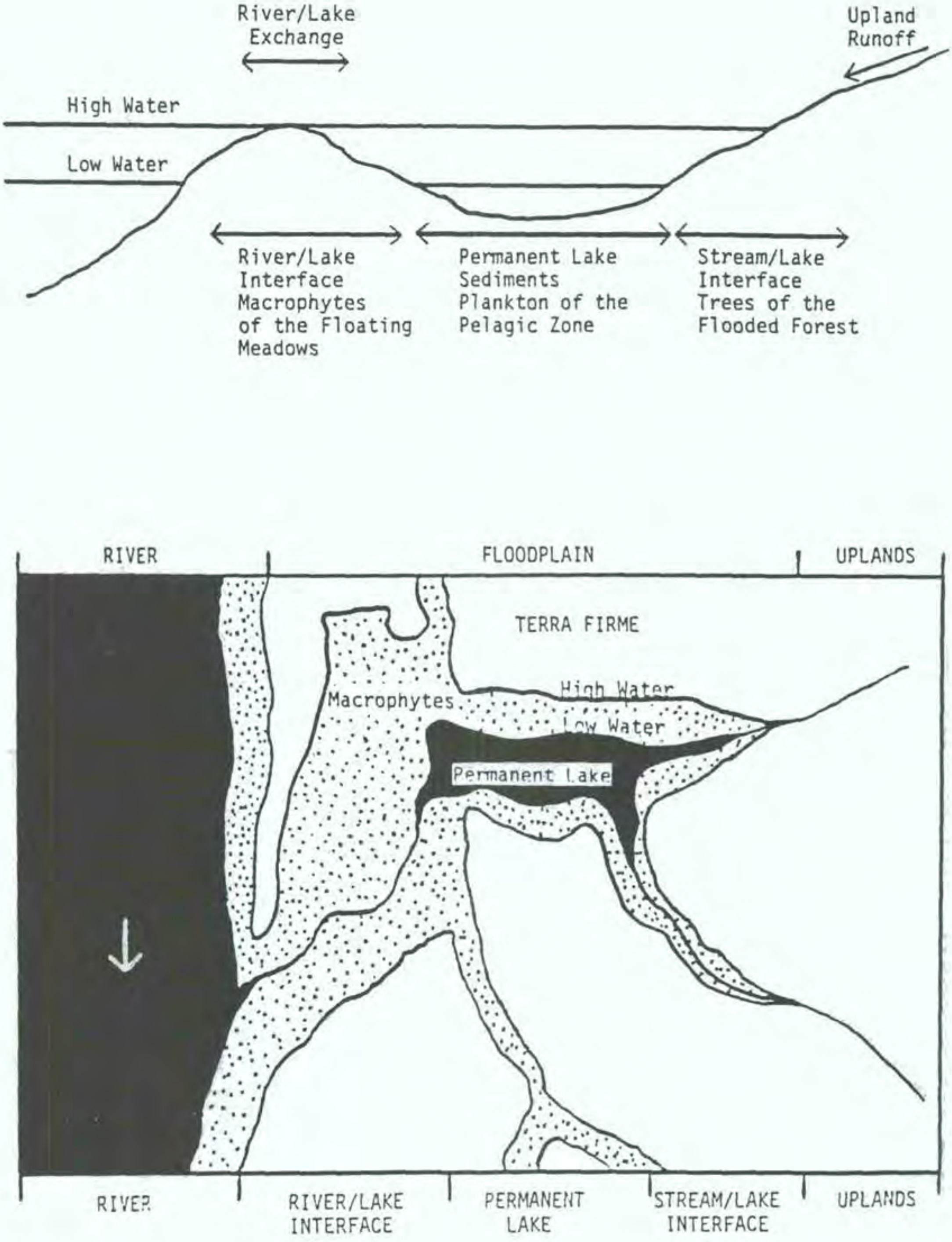

Figure 1. Schematic diagram of the Amazon River floodplain showing the Aquatic Terrestrial Transition Zone (ATTZ) with its two interfaces and characteristic vegetative cover.

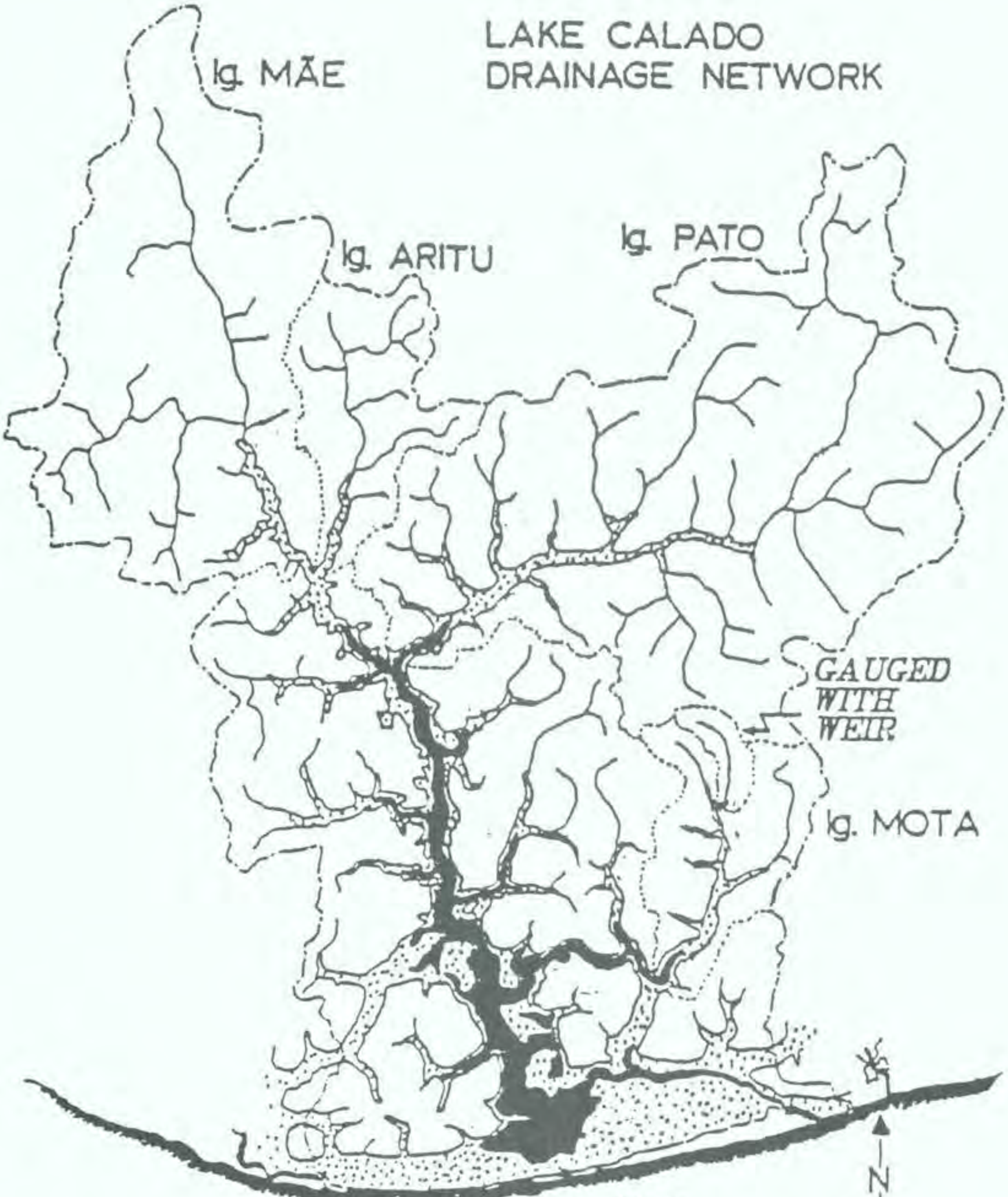

Figure 2. Drainage basin, permanent lake area, and ATTZ of Lake Calado on the Amazon Floodplain (after Lesack 1988). Solimoes River is the Brazilian name for the Amazon mainstem above the junction with the Rio Negro.

Study site description

Lake Calado (3°15' S, 60°34' W) is a permanent body of water occupying a small dendritic portion of the Amazon's floodplain. The lake is located about 80 km upstream from the where the junction of the rivers Negro and Solimoes form the Amazon proper in central Amazonia. The lake is permanently connected to the Solimoes River via a channel ("parana"), and this connection links the stage of the lake to that of the river (Figure 2). Water levels in the lake fluctuate by ~10 m annually due to seasonal rainfall and are controlled by river stage and local basin runoff. The lake varies in depth from 1-2 m in Oct.-Dec. to 10-12 m in June. Due to the relatively flat topography, the 10 m change in depth expands the aquatic zone from the black area in Figure 2 at low water into the stippled zone (the ATTZ), increasing the aquatic area of the lake from 2 to 8 km^2 and the volume from 1 to 57 x 10^6 m^3.

Water accumulates in Lake Calado during the rising water phase (Jan. - June) from several sources. Rapid increases in river stage result in flooding of turbid, nutrient - rich river water onto the floodplain (Fisher and Parsley 1979, Fisher 1978, Lesack 1988). However, the rising river stage also acts as a hydraulic dam, blocking the discharge of some of the local water sources (direct rainfall, groundwater, surface runoff), which are of lower turbidity, conductivity, and nutrient content in comparison with river water. The result is an accumulation of water on the floodplain from several sources, with intermediate turbidity, conductivity, and nutrient content ("lake water"). The major horizontal gradients of turbidity, nutrients, and conductivity are confined to the SLI and RLI due to wind mixing in the more open floodplain areas (Alves and Fisher 1987; MacIntyre and Melack 1984, 1988; Engle and Sarnelle, in press).

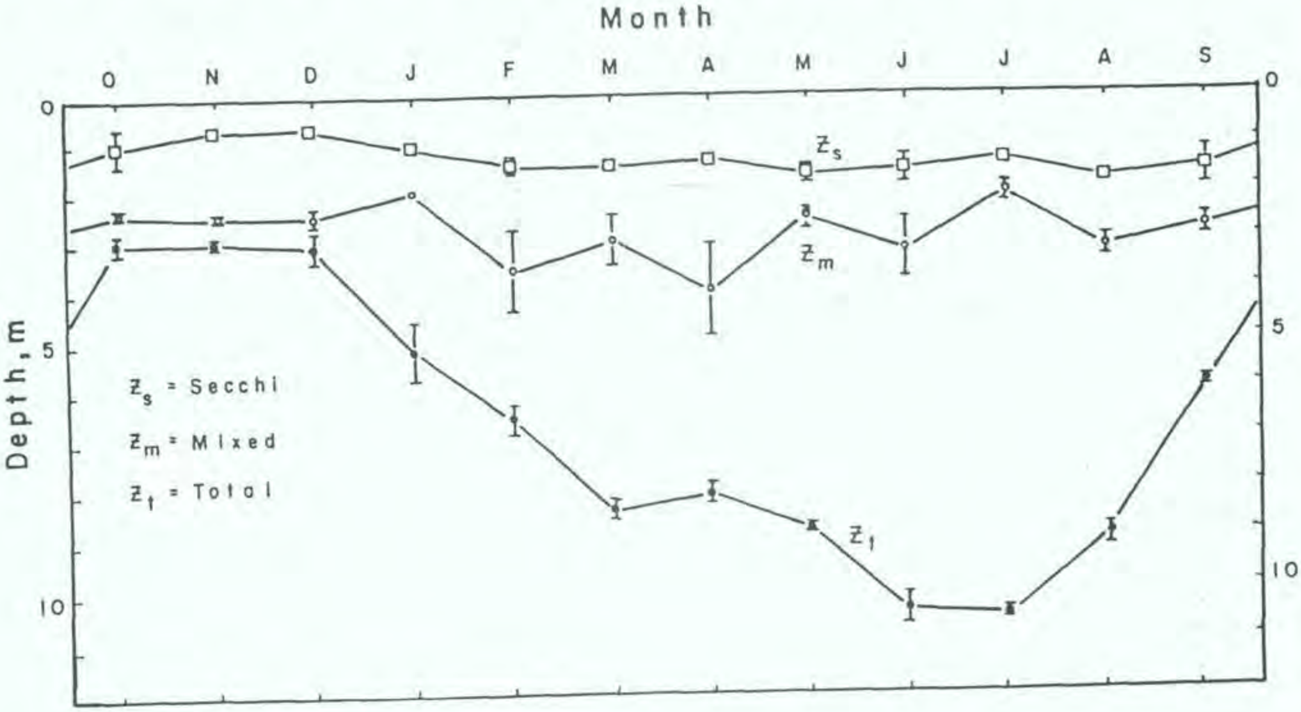

Figure 3. Seasonal variation in depths of the water column, surface mixed layer, and Secchi transparency in Lake Calado. (after Smith and Fisher submitted).

The pelagic zone of Lake Calado develops thermal stratification on seasonal and diel time scales. When water depths exceed 5 m (usually by Feb.), a seasonal thermocline develops at ca. 4-5 m depth which persists until September (Figure 3), punctuated only occasionally by deep mixing events during overcast, windy conditions. The hypolimnion becomes anoxic within days, and the epilimnion undergoes diurnal thermal stratification and noctural convective mixing in the top 3-4 m (Fisher et al. 1983, MacIntyre and Melack 1984, 1988, Smith and Fisher submitted).

Planktonic biomass, photosynthesis, and respiration in the surface mixed layer of Lake Calado are modest. Algal biomass (as chlorophyll a) is typically 5-10 $\mu g\ l^{-1}$, bacterial numbers are 2-5 x 10^9 cells l^{-1}, heterotrophic nanoplankton density is 10^3 - 10^4 individuals l^{-1}, macrozooplankton biomass is 100-500 animals l^{-1}, and total planktonic biomass in the surface mixed layer is typically 1-4 g C m^{-2} (Fisher et al. 1983, Setaro and Melack 1984, Lenz et al. 1986, Smith-Morrill 1987, Peele et al. 1988, Smith and Fisher submitted, E. Lessard et al. unpub.). Planktonic photosynthesis and respiration are both 0.5-1 g C m^{-2} d^{-1}, and respiration typically exceeds photosynthesis due to inputs of organic matter from the macrophytes and periphyton of the ATTZ (Melack and Fisher 1983, Smith and Fisher submitted). This results in chronic undersaturation of oxygen in the open waters of the permanent lake, and there is a net transport of atmospheric oxygen into the surface mixed layer (Melack and Fisher 1983).

No consistent patterns were found for N or P limitation of phytoplankton growth or biomass. Impoverishment of phytoplankton with respect to both N and P was demonstrated throughout the seasonal cycle. Nutrient limitation was continuous, but changed weekly or seasonally between N and P, implying nearly balanced supplies with no consistent excess of N or P (Setaro 1983, Setaro and Melack 1984, Pinheiro 1985).

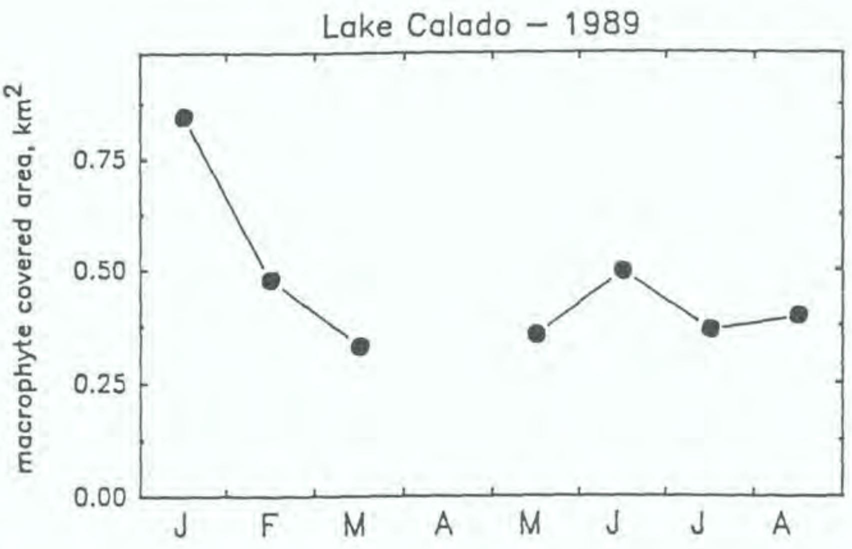

Figure 4. Seasonal variations in the macrophyte covered area in the southern portion of Lake Calado in 1989. Data were obtained by monthly serial overflights and image analysis of digitized video imagery (Fisher, unpub.). There was no overflight in April, 1989.

There is a large seasonal accumulation of macrophyte and periphyton biomass within the ATTZ. In the river - lake interface, grasses and forbs of the floating meadows are typically 30-800 g C m^{-2}, and epiphytic periphyton are 0.4-30 gC m^{-2} (Fisher et al. 1988, Engle and Melack 1990, Fisher et al. unpub.). The macrophytes in this community undergo a seasonal sucession of species, and the macrophyte covered area changes seasonally as the water rises and spreads across the ATTZ (Figure 4). In the stream - lake interface, tree biomass is unmeasured but probably higher than the grasses and forbs of the floating meadows, although periphyton biomass is less (0.01-0.1 g C m^{-2}, Alves and Fisher 1987).

Inputs of N and P

Lake Calado receives inputs from the Solimoes River, local rain, runoff, groundwater, and connections with neighboring lakes. The isotopic composition of water sources ($\delta^{18}O$ and δD) varies seasonally (Fisher et al. 1986), and, with the exception of rain during the Mar-May period, there are significant differences between the isotopic composition of rain, river, and terrestrial waters (stream + groundwater, Figure 5). The contribution of each of these has been estimated using direct monitoring of lake level, channel currents, rainfall, stream discharge, groundwater levels and seepage, and meteorological parameters to calculate evaporation by the energy budget method. Direct monitoring of water inputs and chemical composition have enabled computation of water, N, and P budgets for Lake Calado. Rain, upland runoff, groundwater, and inflows from neighboring lakes were measured directly, whereas river exchange was estimated by difference (Lesack 1988).

Rain. Central Amazon rain is dilute (conductivity 10-20 %mho cm^{-1}) and weakly acidic (pH=5). Nitrogen is abundant, but phosphorus concentrations are low. For water year 1984, rainfall was maximal in February - March, minimal in July - August, and the annual total was 2.6 m y^{-1}. Rain inputs of total N and P were 6.7 kg N and 0.39 kg P ha^{-1} y^{-1} (Lesack 1988).

Upland runoff. A gauged watershed of 23.4 ha in upland forest was the primary study area, and other streams in the Calado basin were also sampled. Water from the gauged catchment is dilute (conductivity ca. 10 umho cm^{-1}) and weakly acidic (pH = 5-6); it contains little phosphorus, but substantial quantities of N. Runoff peaks in May, with a minimum in December; for water year 1984, 1.6 m was discharged from the watershed. The area yield coefficients for total N and P were 4.3 kg N and 0.082 kg P ha^{-1} y^{-1} (Lesack 1988). These values are typical of export coefficients from undisturbed tropical catchments (1-10 kg of N and 0.08-0.46 kg of P ha^{-1} y^{-1}, Lesack et al. 1984, Lewis 1986, Saunders and Lewis 1988).

Groundwater. Wells, piezometers, and seepage meters were used to examine groundwater levels and chemistry. Groundwater levels generally lagged behind lake levels with the result that, during rising water, groundwater levels were lower than lake water levels, and groundwater storage occurred. During falling water, groundwater was elevated with respect to the lake level, and discharge occurred. Groundwater is elevated in nitrogen and phosphorus due to interactions with sediments and soils (Lesack 1988).

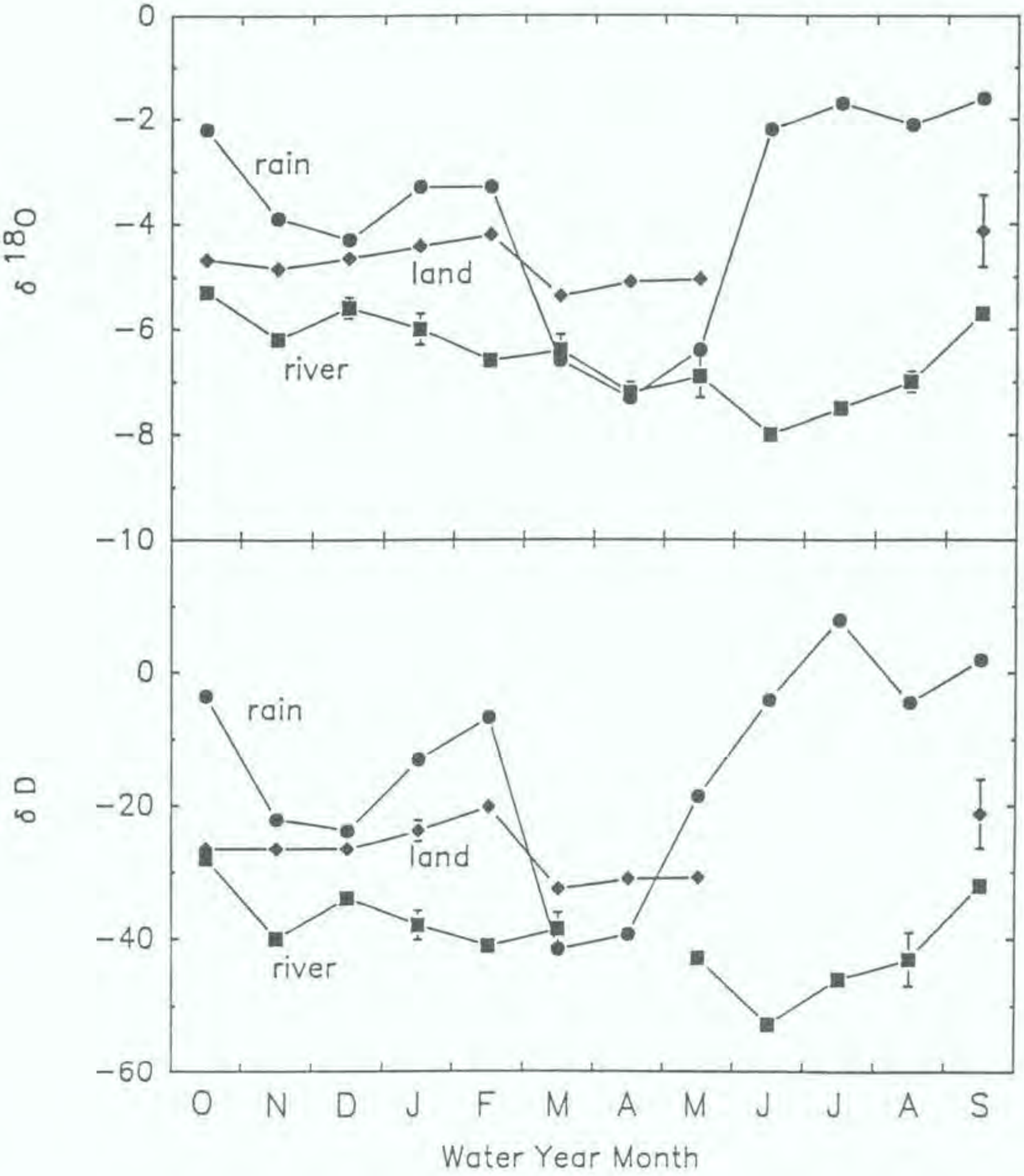

Figure 5. Seasonal variations in isotopic composition of water from the land (stream and groundwater), rain, and the Solimoes River near Lake Calado (Fisher et al. 1986).

Adjacent lakes. There are connections with two lakes adjacent to Calado. However, exchange is significant only with Lake Miriti (immediately west of Calado), and occurs only during high water unidirectionally into Calado. Velocity, cross - sectional area, and water chemistry were measured during high water in order to estimate inputs. The chemical composition of the water flowing from Miriti to Calado is similar to river water because of river flooding into Lake Miriti near the connection to Calado (Lesack 1988).

River flooding. The movement of river water onto the floodplain results in nutrient inputs which stimulate phytoplankton biomass and primary productivity (Fisher and Parsley 1979, Fisher 1978, Engle and Sarnelle in press). Both N and P are present in high concentrations. However, river flooding events are difficult to quantify directly, since they occur irregularly with low velocities, and may occur at depth and not be visible from the surface of the lake (Figure 6).

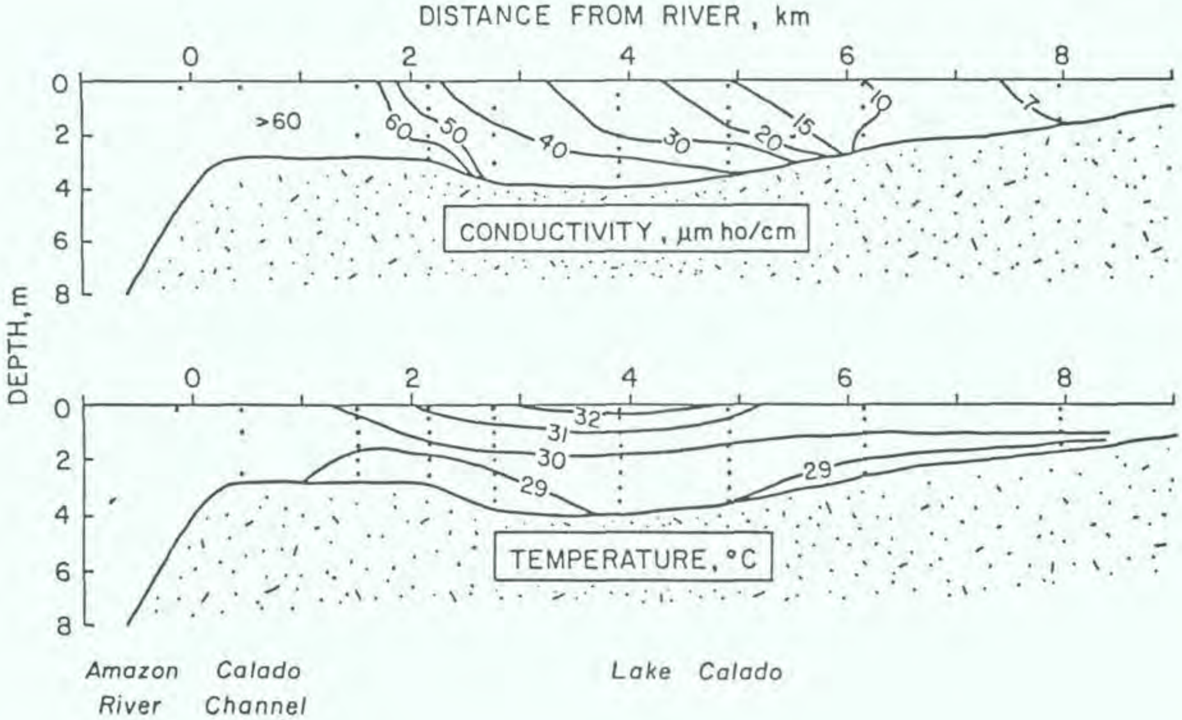

Figure 6. River flooding of Lake Calado at low water (9 Nov. 80), showing river water entering at and below the thermocline.

N and P input summary. Surface runoff and river inflows are the two dominant sources of N and P to the lake (Table 1). The river is the most important source of both N and P during rising water, but on an annual basis, the inputs of N from the local water sources dominate (66 %). However, due to its high P content, the river is the dominant source of P on an annual basis (51 %), and groundwater efflux is the second largest P source (19 %).

For comparison with internal processing of N and P (below), the inputs of total N and P (10^6 moles y^{-1}) have been rescaled. The following simplifying assumptions have been made: (1) nutrient inputs are uniformly distributed over an average lake area of 5.85 km^2 (average depth of 6 m) and (2) inputs are uniformly distributed in time. Clearly there are important spatial and temporal variations, but for comparison with N and P recycling these have been ignored. The rates given at the bottom of Table 1 (1.4 mmol TN and 0.045 mmol TP m^{-2} d^{-1}) represent values for N and P inputs averaged spatially over the lake surface and temporally over the year. The rates are equivalent to 74 kg N and 5.1 kg P ha^{-1} y^{-1}. The atomic ratio of the inputs is 32 N : 1 P, twice the compositional ratio of phytoplankton (16:1), indicating a surplus of N relative to P in the inputs. In the absence of other processes (e.g., denitrification, differential nutrient cycling), these inputs would lead to P limitation of plankton populations.

Table 1. Inputs of total nitrogen (TN) and total phosphorus (TP) to Lake Calado on the floodplain of the Amazon River. Units in the upper table are 10^6 moles N or P y^{-1}, and % is the percentage of the total contributed by each source. The original data on inputs are from Lesack (1988)

source	TN	%	TP	%
direct rainfall	0.24	8	0.006	6
surface runoff	1.29	42	0.011	11
groundwater	0.26	8	0.018	19
adjacent lakes	0.25	8	0.013	13
Amazon River	1.03	34	0.049	51
all sources	3.07	100	0.097	100

ave. areal inputs to the aquatic system, assuming uniform loading over space and time, and an average aquatic area of 5.85 km^2 at an average depth of 6 m. Units below are mmol N or P m^{-2} d^{-1}.

source	TN	%	TP	%
all sources	1.44		0.045	

Internal recycling of N and P

Nutrient inputs to Lake Calado undergo considerable biological processing. In the open waters of the lake away from the ATTZ, nutrients are consumed and recycled by a variety of mechanisms. In this section we emphasize planktonic processes in the permanent lake only because equivalent data are not yet available for macrophytes and periphyton in the ATTZ. The recycling processes are water column regeneration via heterotrophic activity, sedimentation, and sediment diagenesis.

Macrozooplankton. Excretion by cladocera, adult copepods, and copepodites (> 53 %m) was estimated as accumulation of ammonium and phosphate under starved and fed conditions. These animals had an abundance of 0.1-3 x 10^5 individuals m^{-2}, and showed seasonal variations in abundance and nutrient excretion. Excretion rates ranged from 0.2-8.3 µmol PO_4 and 2-77 µmol NH_4 m^{-2} h^{-1}, increasing in magnitude away from the river - lake interface (Lenz et al. 1986).

Regeneration. Measurements of heterotrophic regeneration of ammonium and phosphate by plankton were made using ^{15}N and ^{32}P isotope dilution with size fractionated water samples. Ammonium and nitrate pool sizes were measured colorometrically whereas biologically available phosphate was estimated with Rigler bioassays. Uptake and regeneration of ammonium and phosphate by plankton were approximately balanced in the water column on time scales of minutes to hours, and nitrate uptake was only important within the river - lake and stream - lake interfaces. Uptake of N and P was primarily controlled by nutrient availability, and was dominated by bacteria sized organisms < 3 %m. In contrast, regeneration was approximately equally divided between organisms > and < 3 %m. Recycling in the water column results in turnover times of the small inorganic pools of minutes to hours; the larger particulate pools turn over on longer time scales (hours to days). Uptake and regeneration rates ranged from 100-900 nmol NH_4 and 10-100 nmol PO_4 l^{-1} h^{-1} (Fisher and Morrissey 1985, Morrissey and Fisher 1988, Fisher et al. 1988a, b, and Alves 1990).

Sedimentation. Sinking of epilimnetic particulates was measured with cylindrical traps moored below the seasonal thermocline for 2-5 day intervals. Particulates sedimented from the surface mixed layer at an average rate of 6 g dry m^{-2} d^{-1}, but rates were about 50 % higher under the floating macrophyte beds of the ATTZ and declined away from the river. Composition of the sedimenting material was similar to that of seston and periphyton, and was equivalent to ca. 75 % of the gross primary productivity of these two communities (Smith-Morrill 1987, Smith and Fisher submitted).

Diagenesis. Nutrient fluxes from sediments were estimated in situ using bottom chambers and interstitial water samplers. There are large gradients of ammonium and phosphate in interstitial waters which drive fluxes from the sediments to the water column. Rates were larger under anoxic conditions in permanent lake sediments than under oxygenated conditions in seasonally flooded soils. Ammonium fluxes were highest in permanent lake sediments near the terrestrial border of the floodplain, whereas phosphate rates were highest near the river. The fluxes of ammonium and phosphate across the sediment water interface ranged from 200-500 µmol NH_4 and 8-50 µmol PO_4 m^{-2} h^{-1} and slightly exceeded, but were not significantly different from the independently measured sedimentation fluxes shown in Table 2 (Smith and Fisher 1985, Figuereido 1986, Smith-Morrill 1987).

Recycling summary. Regeneration of ammonium and phosphate per m^{-2} of lake surface is dominated by heterotrophic water column activity (Table 2). Macrozooplankton excretion is a minor percentage, indicating that heterotrophs < 53 µm are responsible for most of the water column excretion. The sediments are a smaller but important mechanism of nutrient cycling, and N and P sedimentation and diagenesis result in a turnover time of days to weeks for particulate N and P in the epilimnion, compared to hours to days for water column uptake and regeneration.

Recycling rates have a significant impact on the ratio of N to P. All of the recycling processes, and in particular the water column regeneration, had low ratios of N:P (4:1 to 9:1) relative to algal biomass (16:1); i.e., P cycled faster than N in the water column relative to the compositional ratio ("Redfield ratio", Redfield et al. 1963) of phytoplankton. Overall, recycling by all mechanisms resulted in an atomic ratio of N:P = 5:1, which suggests that recycling will tend to make P available in surplus relative to N requirements by phytoplankton. This contrasts with the external input ratio (N:P = 32:1, Table 1), which supplies N in excess. Thus, external inputs supply N and P with excess N, but internal recycling regenerates P faster than N.

Table 2. Summary of N and P regeneration rates in Lake Calado. Water column and zooplankton data are integrated over a 3 m epilimnion, assuming little activity in the anoxic hypolimnion. Sedimentation fluxes are those measured beneath the thermocline, and nutrient fluxes are measured at the sediment water interface using bottom chambers. Numbers in parentheses are percentages of total recycling rate represented by each process. N/P is the atomic ratio of the rates. Macrozooplankton excretion was probably underrepresented in the small volume isotope dilution measurements ("water column regeneration") and have been added to the recycling total in the bottom line

	<-------- $mmol/m^2 \cdot d$	--------->	
source	N	P	N/P
zooplankton > 53 %m (as NH_4 and PO_4)	1.0 ± 0.2 (5%)	0.11 ± 0.02 (2%)	9 ± 2
water column regen. (as NH_4 and PO_4)	13.8 ± 1.2 (65%)	3.36 ± 0.48 (78%)	4 ± 1
sedimentation fluxes (as PN and PP)	6.0 ± 2.4	0.53 ± 0.17	11 ± 6
sediment nutrient flux (as NH_4 and PO_4)	6.5 ± 1.4 (30%)	0.89 ± 0.46 (20%)	7 ± 4
total recycling (lines 1+2+4)	21.3 ± 1.9 (100%)	4.36 ± 0.66 (100%)	5 ± 1

Outputs of N and P

N and P are lost from Lake Calado by groundwater exchange, burial, and outflow to the river. Seasonal groundwater storage occurs during rising water, and represents a temporary removal of N and P. Export of N and P to the Amazon River is most intensive during the falling water levels of July - Sept, whereas burial is a slower process driven by sedimentation and diagenesis (Lesack 1988, Smith et al. submitted).

Groundwater. Lake water seeps into groundwater during the rising water phase due to an increased hydraulic head in the lake relative to groundwater. This storage results in a temporary loss of N and P until falling water levels in the lake result in flow from groundwater to the lake. Overall, about 50 % more groundwater seeps out than seeps into groundwater (Lesack 1988).

Export to the Amazon River. After high water in June, exchange between the lake and river is rapid and unidirectional towards the river. Most of the export to the river occurs during the period of rapidly falling water levels in July - September, although some occurs earlier during the rising water phase. The TN/TP ratio of lake water discharged into the river was 22:1 (Table 3), indicating a differential loss of N relative to P in comparison with the input N:P of 32:1.

Table 3. Losses of total nitrogen (TN) and total phosphorus (TP) from Lake Calado on the floodplain of the Amazon River. Units in the upper table are 10^6 moles N or P y^{-1}, and % is the percentage of the total contributed by each source. The original data on losses are from Lesack (1988)

source	TN	%	TP	%
groundwater	-0.18	4	-0.012	5
burial	-1.36	28	-0.072	31
lake outflow	-3.26	68	-0.152	64
all losses	-4.80	100	-0.236	100
Average areal losses from the aquatic system, assuming uniform distribution over space and time, and an average aquatic area of 5.85 km^2 at an average depth of 6 m. Units below are mmol N or P m^{-2} d^{-1}.				
all losses	-2.25		-0.110	

Burial. Long-term N and P burial rates in sediments were derived from ^{210}Pb data at 3 stations along the main lake axis. Rates averaged 1.86 ± 0.91 mmol N and 0.099 ± 0.043 mmol P m^{-2} d^{-1} and were maximal in the middle of the permanent lake sediments. The N:P of buried material was 19:1, again less than the N:P input ratio of 32:1 and suggestive of differential loss of N or mobilization of P on the floodplain (Smith-Morrill 1987, Melack and Fisher 1988, Smith et al. submitted). The average burial rates in the sediments of the permanent lake area (2 km^2) are equivalent to 95 kg N and 11 kg P ha^{-1} y^{-1}, again much higher than terrestrial export coefficients but equivalent in magnitude to the export coefficients of the aquatic zone of the floodplain.

Loss summary. Lake export to the river was the dominant loss term (60-70 % of total losses in Table 3). Burial in permanent lake sediments was less important (30%), and seasonal groundwater storage was quite small. The dominant term, export of N and P to the river, was about 3 times the annual input of N and P from the river to the lake during rising water (Table 1). Therefore, for water year 1984, this portion of the Amazon floodplain was a net source of N and P to the river (2.23 x 10^6 moles N and

0.103 x 10^6 moles P y^{-1}). However, due to interannual variability in flooding and draining patterns, we anticipate that the net exchange of N and P between the river and lake is highly variable from year to year.

Synthesis

This review of the research at Lake Calado began with the concept that nutrient utilization and recycling were important characteristics of large floodplains. The data presented above permit a detailed examination of the validity of this concept.

Consumption and recycling of the inputs to Lake Calado is intensive. Table 4 summarizes inputs, recycling in the water column and sediments, burial in sediments, and losses of N and P via export. The largest terms are the heterotrophic recycling of N and P in the oxygenated surface mixed layer; inputs and outputs are 10-100 times less. These data are used below to compute the total nutrient supply available to photosynthetic organisms, to explain the observed pattern of nutrient limitation, and to compute the number of cycles which the inputs of N and P undergo within the lake.

Table 4. Summary of the input, recycling, burial, and losses of total N and P in the Lake Calado basin. Outputs and burial exceed inputs by ca. a factor of 2 probably due to interannual variability, scaling differences between the terms, and cumulative errors. Inputs, recycling, and burial + losses all have units of mmol m^{-2} d^{-1}. # of cycles is a unitless ratio computed as recycling/inputs and indicates the number of times the inputs are recycled before burial or loss

	N	#cycles	P	#cycles	N/P
inputs	1.44		0.045		32
water column recycling	13.8		3.47		4
# of water column cycles		10		72	
sediment recycling	6.5		0.89		7
# of sediment cycles		4		20	
burial + losses	-2.25		-0.110		20
total nutrient supply (inputs + recycling terms)	21.7		4.41		5

The total nutrient supply available to photosynthetic organisms is the sum of inputs and recycling. The total supply of N and P may be computed from the first, second, and fourth lines of Table 4 and is shown at the bottom of the table. These rates, 21.7 mmol N and 4.41 mmol P m^{-2}

d^{-1}, are equivalent to supply rates of 1110 kg N and 499 kg P ha^{-1} y^{-1}, and represent the total amount of N and P available per unit area of the water column of Lake Calado. Inputs of N and P are small in comparison with recycling, emphasizing the importance of recycling for the nutrient supply to photosynthetic organisms on the Amazon floodplain.

The dominance of the nutrient supply by recycling has important implications for nutrient limitation in Lake Calado. Phytoplankton nutrient limitation appeared to be continuous, but changed weekly or spatially between N and P (Setaro 1983, Setaro and Melack 1984, Pinheiro 1985). Since the inputs of N and P (N:P = 32:1) contain excess N relative to phytoplankton composition (16:1), nutrient inputs will tend to produce P limitation of algal growth. However, N cycling appears to be slower than that of P (N:P = 4:1 in the water column, 7:1 in the sediments), and the more rapid P cycling can alleviate P limitation resulting from N-rich inputs. The dominance of the total nutrient supply by recycling suggests that, on average, the plankton should be N limited, given the N:P ratio of 5:1 of the total nutrient supply (Table 4). However, spatial and seasonal variations in nutrient loading and recycling can shift the N:P ratio of the nutrient supply between 32:1 (inputs dominating) and 7:1 (sediment recycling dominating), and 4:1 (water column recycling dominating), providing either excess N or P, respectively, for phytoplankton (16:1). This may explain the observed variations in the indices of nutrient limitation.

The data in Table 4 also enable estimates of the number of cycles which inputs of N and P undergo. The ratio of recycling to inputs is a dimensionless number which can be interpreted as the number of times that an element is used within an ecosystem before burial or export (Eppley 1981). The data of Table 4 indicate that the N and P inputs of Lake Calado complete 10-72 cycles in the surface mixed layer before export or burial, and that inputs undergo fewer cycles (4-20) within the sediments before export or burial. The number of input cycles (cycling rate/input rate = # cycles) should not be confused with the turnover rate of epilimnetic N or P pools ([recycling rate d^{-1}]/biomass pool = # pool turnovers d^{-1}). The number of pool turnovers d^{-1} (Smith and Fisher submitted) suggest that particulates in the epilimnion turnover 2-4 times in the water column on a time scale of days for each turnover via sediments on a time scale of weeks. Thus, we conclude that within Lake Calado, on the Amazon's floodplain, nutrients are rapidly and repeatedly recycled, supporting the concept of floodplains as important sites of nutrient utilization and recycling within the river basin.

Nutrient retention on the floodplain may also be an important characteristic of floodplain ecosystems. N and P may be retained on the floodplain by accumulation within permanent lake sediments, by accumulation on soils or plants of the ATTZ, and by accumulation within the plankton of the water column. Accumulation of permanent lake sediments is the only known long - term retention mechanism for N and P on interannual time scales, corresponding to 44 % of the N and 74 % of the P inputs (Tables 1 and 3). While plankton accumulates within the lake seasonally as the open lake area expands (10-30 mmol N and 0.4-1.8 mmol P m^{-2} , Smith and Fisher, submitted), there is no evidence for interannual accumulation (Fisher et al, unpub.). This probably results from the fact than annually ca. 90 % of the volume of Calado drains back into the river. We have little information on soils of the ATTZ, but Smith- Morrill (1987) measured fluxes of nutrients from seasonally submerged soils of the ATTZ into the water column. Although these rates were less than for permanent lake sediments, it is unlikely that the soils are a long term sink for nutrients.

Macrophytes of the floating meadows may represent a long term nutrient storage mechanism, but the time scale is uncertain. Large quantities of N and P accumulate seasonally as macrophytes and periphyton near the river lake interface (typically 2700 mmol N and 170 mmol P m^{-2} at high water, Fisher et al. unpub.). When multiplied by the maximum area covered annually by macrophytes in June (0.55 km^2, Figure 4), this accumulation of macrophytes and periphyton represents a maximum of 1.5 x 10^6 moles N and 0.094 x 10^6 moles P y^{-1}. These values are very similar to the external N inputs of 3.1 x 10^6 moles TN and 0.097 x 10^6 moles TP y^{-1}, indicating that an amount of N and P equivalent to 50-95 % of all external inputs annually accumulates as macrophytes and periphyton. However, the fate of this material during falling water when the macrophytes are stranded on emerging shorelines of the ATTZ is unknown. Much of the macrophyte material appears to decompose on the drying soil surface and may enter the dissolved pools of the lake, to be consumed and/or exported from the lake. The only known interannual accumulation of N and P is within the 2 km^2 area of permanent lake sediments.

The composition of exported and buried material differs from that of inputs. Losses of N and P via burial (N:P = 19:1) and export to the River (N:P = 21:1) are depleted in N relative to the inputs (32:1), and denitrification may contribute to the apparent loss of N during biological utilization and recycling. N fixation and denitrification are known to occur in Lake Calado, but are poorly quantified (Melack and Fisher 1988). Our current research is addressing some of these issues.

This review of our project at Lake Calado has largely supported the concept that intensive nutrient cycling and utilization are important characteristics of the Amazon's floodplain. The data show that nutrients were consumed and repeatedly reused on the floodplain, and that an amount equivalent to 50-95 % of the inputs annually accumulated as macrophyte and periphyton biomass near the River Lake interface. Although 40-70 % of the N and P inputs were retained in the permanent sediments, this section of the Amazon's floodplain was a net source of N and P to the river, principally due to local drainage from uplands on the terrestrial side of the floodplain.

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WATER BIOGEOCHEMISTRY IN MOGI-GUAÇU RIVER FLOODPLAIN OXBOW LAKES (Jataí Ecological Station, São Paulo, Brazil)

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Introduction

Despite their importance to a complete understanding of global biogeochemical cycles, and their large extent mainly in tropical regions, the role of floodplain ecosystems - with their fluvial, lacustrine, wetlands and local watershed components - has only recently been recognized by earth scientists.

The hydrological regime of these floodplain lakes in tropical regions possesses a unique characteristic if compared to high elevation or temperate lakes. Rivers such the Amazon, during its annual hydrological cycle inundates an area in the Central Valley of about 10^5 Km^2 (Melack, 1984) as a result of a 10 m fluctuation in the river's level. On the South American continent 3.2×10^5 Km^2 of land is annually inundated by peak flood (Welcomme, 1979). The floodplain lakes are then seasonally enriched with river nutrients and their water chemistry transformed. Intense biological activity (Fisher and Parsley, 1979; Fosberg, 1984), sedimentation/resuspension of particles (Schmidt, 1973), and sediment release (Figueiredo, 1984; Smith-Morill, 1987), in addition to changes induced by differences in the timing and elemental composition of river and local inputs (Schmidt, 1973; Furch, 1982; Fosberg et al., 1988) all occur.

It is known that the interactions existing between a river and its floodplain are extremely important for fish production (Welcomme, 1979; 1985). However, as demonstrated for the Amazon river system, it is not yet clear whether floodplains and their lakes are a net source of organic matter and nutrients to the mainstem (Sioli, 1975a, b; Devol et al., 1984; Junk, 1980; Hedges et al., 1986; Fosberg et al., 1988). In the Orinoco river floodplain, Lewis (1988) has shown that these ecosystems, although productive, export little of the carbon internally produced to the river. However, he did not considered as part of the floodplain areas, what he called, "near-channel stagnant or slow-flowing areas".

In general, as pointed out by Richey (1987), little is known about biogeochemical processes that link mainstem, floodplain and off-channel environments. This knowledge is vital for the complete understanding of these systems. Recently Fosberg et al. (1988) have made a significant contribution to the understanding of these complex linkages. Various lakes were studied in Central Amazônia and it was documented that the main mechanism controlling their water quality is the variable mixing of waters of river and local origin.

The aim of the research presented here is to investigate the origin of different water masses and some of the factors that exert control on the water quality of oxbow lakes of the Mogi-Guaçu river floodplain (upper Paraná river basin). Due to a similarity existing among the ecosystems at the Amazonian rivers and the Mogi-Guaçu river there exists an opportunity to make a comparison of both systems in order to understand the global biogeochemistry of elements in this type of environment. This research project is one aspect of an ongoing multidisciplinary program with the overall goal of developing an environmental evaluation and understanding of the ecological role of the Mogi-Guaçu river oxbow lakes.

Methods

Study area. Four oxbow lakes, three streams and the mainstem, Mogi-Guaçu river, were sampled at a variable time scale for each sub-system during the period of 1987 to 1989. The floodplain area is located within the Jataí Ecological Station (4,532 ha; Lat. 21° 33'/21° 37'S and Long. 47° 45'/47° 51' W) in the Mogi-Guaçu river basin (17,400 Km^2). This basin is in the upper part of the Paraná river basin, the second largest basin in South America (2.8×10^6 Km^2), and is located in the Northwestern region of São Paulo state, Brazil (see Figure 1). The timing of field trips and their position in relation to the Mogi-Guaçu river hydrograph at Porto Pulador (about 40 Km up stream) can be seen using figures and tables describing data and the hydrograph itself. Particular information on the hydrologic setting of each lake is found in the text (section of results and discussion).

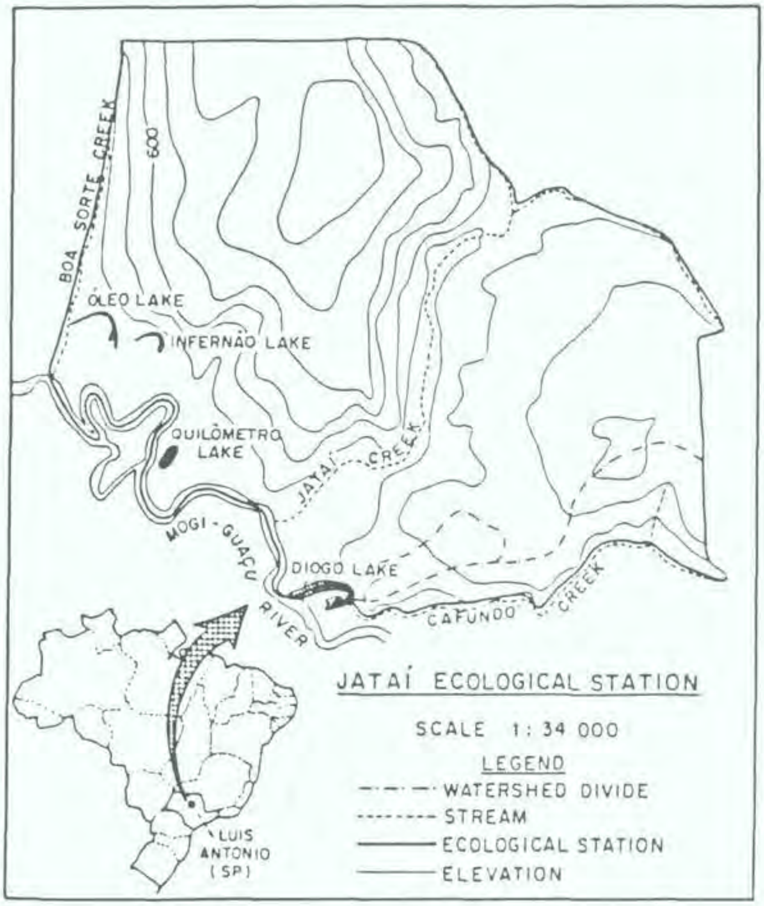


Figure 1. Study area map showing lakes, streams and the Mogi-Guaçu river sampling sites at the Jataí Ecological Station, Luiz Antonio, SP, Brazil.

Lakes are in general very shallow at all times. The mean depth varies from <5 m to <1 m. Except for one of the lakes which is nearly round, all the other have a "U" shape (old meander). There are about 15 oxbows at the Jataí Ecological Station, but permanent water is found in only 4 of them. The others have been filled through time by sediments and developed a dense vegetation cover (mixture of some aquatic and terrestrial plants).

Sampling and methods of chemical and isotopic analysis. Surface waters were sampled directly with pre-washed polyethylene bottles. Samples at depth were collected with a Niskin bottle. Water temperature, pH and conductivity were measured in the field. Major ions and dissolved inorganic nutrient analyses were performed on filtered samples (cellulose acetate membrane with nominal pore size of 0.45 µm) within 24 to 48 h after sample collection. Total nutrient analyses were done on unfiltered and non-preserved samples within 12 h. Total alkalinity (total acid neutralizing capacity) was determined within 12 h on unfiltered samples by the Gran method (Gran, 1952; Edmond, 1970).

Calcium and magnesium were analysed titrimetrically with EDTA (standard deviation of 1%), while sodium and potassium were analysed by flame photometry (SD 1%) (Basset et al., 1981). All other constituents were analysed spectrophotometrically by flow injection analysis (FIA). Chloride was determined by the mercury thyocianate/iron (III) method (SD 1.5%) (Ruzicka et al., 1976), while sulphate was done turbidimetrically (Krug et al., 1977). Reactive silica was determined by the molybdenum blue method (SD 1%) (Ferreira et al., 1987). Nitrate and nitrite (SD 1%) were determined with the method outlined by Giné et al. (1980). Ammonium (SD 0.5%) was performed by the Berthelot modified reagent described by Krug et al. (1983). Total nitrogen (SD 1%) and total phosphorus (SD 0.5%) were done respectively as nitrite and orthosphosphate (after persulphate digestion described by Ebina et al., 1983) by the method of Giné et al. (1980) and Nóbrega et al. (1988). Soluble reactive phosphorus (or orthophosphate) (SD 0.5%) was performed by the method of Nóbrega et al. (1988).

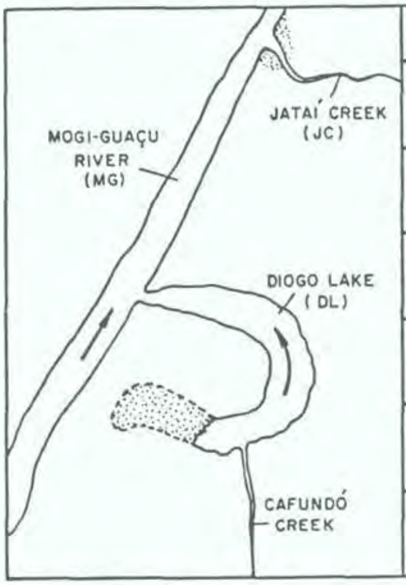
Oxygen-18 and deuterium-2 in water were analysed by methods described by Matsui (1980) and Matsui et al. (1971) respectively. Carbon-13 contents in plants and sediments were determined with the methodology of Craig (1954; 1957). δ values are expressed as an isotopic relation between sample(s) and standard (std):

$$\delta \ \% = (R_s - R_{std})/R_{std} \times 10^3$$

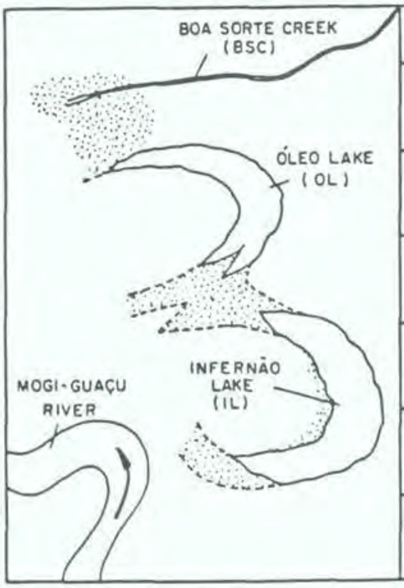
where R are the isotopic ratios ($^{16}O/^{18}O$; $^{1}H/^{2}H$; $^{12}C/^{13}C$) in the sample and the standard (SMOW for ^{18}O and ^{2}H in the water and PDB for ^{13}C in the organic matter). The overall analytical errors in these analyses are respectively 0.3 %, 3% and 0.3%.

Results and discussion

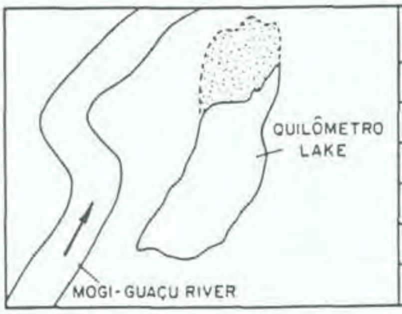
This paper is based on two different sets of data. One was generated during two field trips undertaken specifically during high (19-21/03/1988) and low (16-18/10/1987) water seasons in which samples were collected in four oxbow lakes, three streams and the Mogi-Guaçu river. Figure 2 shows average data for these sites.



$\mu.eq.L^{-1}$		LOW WATER	HIGH WATER
Na^+	DL	26.8	54.6
	MG	400.2	165.3
	JC	26.1	26.1
K^+	DL	25.1	29.8
	MG	66.5	53.7
	JC	23.0	17.9
Ca^{2+}	DL	44.9	95.5
	MG	254.5	185.6
	JC	50.0	48.4
Mg^{2+}	DL	41.1	59.8
	MG	115.2	101.2
	JC	8.3	44.4
Cl^-	DL	8.9	31.3
	MG	220.0	95.9
	JC	5.6	8.5
TZ^+	DL	137.9	239.7
	MG	836.4	505.8
	JC	107.4	136.8



$\mu.eq.L^{-1}$		LOW WATER	HIGH WATER
Na^+	OL	15.2	120.7
	IL	26.8	54.6
	BSC	8.7	13.1
K^+	OL	7.7	51.8
	IL	25.1	29.8
	BSC	5.1	10.2
Ca^{2+}	OL	44.9	148.2
	IL	44.9	95.5
	BSC	2.0	2.0
Mg^{2+}	OL	27.8	102.8
	IL	41.1	59.8
	BSC	2.5	2.5
Cl^-	OL	9.9	78.3
	IL	8.9	31.3
	BSC	<	8.5
TZ^+	OL	95.6	423.5
	IL	137.9	239.7
	BSC	18.3	27.8



$\mu.eq.L$	LOW WATER	HIGH WATER
Na^+	168.6	121.8
K^+	49.9	50.3
Ca^{2+}	91.1	183.6
Mg^{2+}	63.7	108.0
Cl^-	91.7	78.0
TZ^+	373.3	465.8

Figure 2. Average concentrations of major ions ($\mu eq.l^{-1}$) in the Mogi-Guaçu river, oxbow lakes and streams sampled in this study. TZ+ is the total cation concentration, i.e., $\mu eq.l^{-1}$ of Ca + Mg + Na + K (Stallard and Edmond, 1983). Maps are not to scale. Dots approximately represent wet land areas.

A second set of data has been generated since April/1987. The field trips were undertaken nearly on a monthly basis. Samples were collected in Diogo lake, Cafundó creek and Mogi-Guaçu river. This data set is presented here as a case-study (Krusche, 1989; Krusche and Mozeto, ms. in prep.) (see below) for the period of April/88 to August/89. A complete data set is available on request. The timing of these field trips is given in Figure 3 in relation to the hydrograph of the Mogi-Guaçu river at Porto Pulador.

It should be mentioned that the high water field trip (19-21/03/88) does not correspond exactly to the peak flood (21-22/03/88) for Diogo Lake and Cafundó creek which were sampled on 19/03/88. All the other systems were sampled during the peak flood. The low water field trip corresponds to the minimum river stage observed at the Jataí Ecological Station. The data of the case-study presented here refer to high, low and intermediate river stages.

Study site hydrology. The hydrograph of the Mogi-Guaçu river (Figure 3) basically displays an unimodal pattern with a variation in water level close to 4 m for the last two seasonal floods. It was observed in the field that the discharge peak only lasted a few days with rapid recession of the waters to the main channel. These characteristics are very different from the hydrograph of a much larger river such as the Amazonas (Fosberg et al., 1988) for which the recession period is longer owing to a larger basin with peculiar geomorphological features.

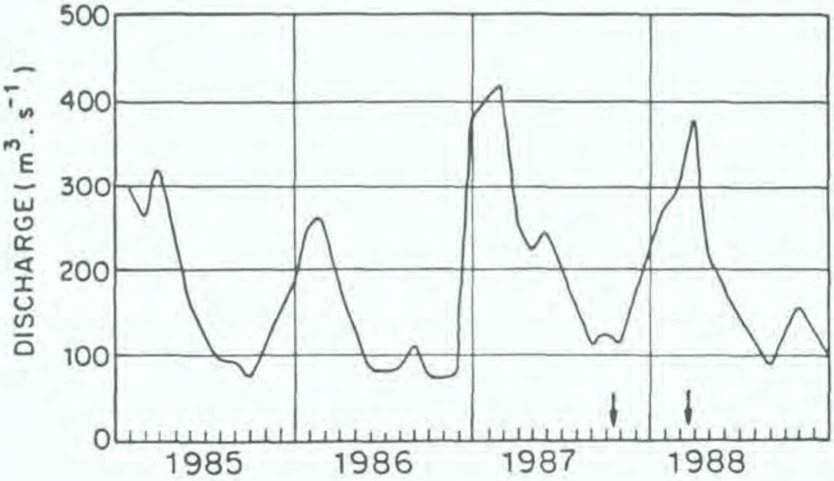


Figure 3. Hydrograph of the Mogi-Guaçu river at Porto Pulador. The timing of low and high water river stage surveys are marked by arrows.

Several short flooding events occurred during a specific high water season at the Jataí Ecological Station. During these seasonable floods, river waters enter the oxbow lakes and inundate a large surface area of low lands. This phenomenon has been indicated as influencing the nutrient chemistry in lakes of the central floodplain of the Amazonas river by Fosberg et al. (1988), who used alkalinity as a conservative tracer (of hydrological nature) to quantify the relative mix of river and local waters.

The data presented here also show that changes in river stage and the resulting inundation of low lands where the lakes are, is the main driving force which determines the water biogeochemistry in these ecosystems. Based on the characteristically different chemical composition of water in these ecosystems for low and high water periods (Figure 4) it can be anticipated that different patterns of ions and nutrient dynamics will occur.

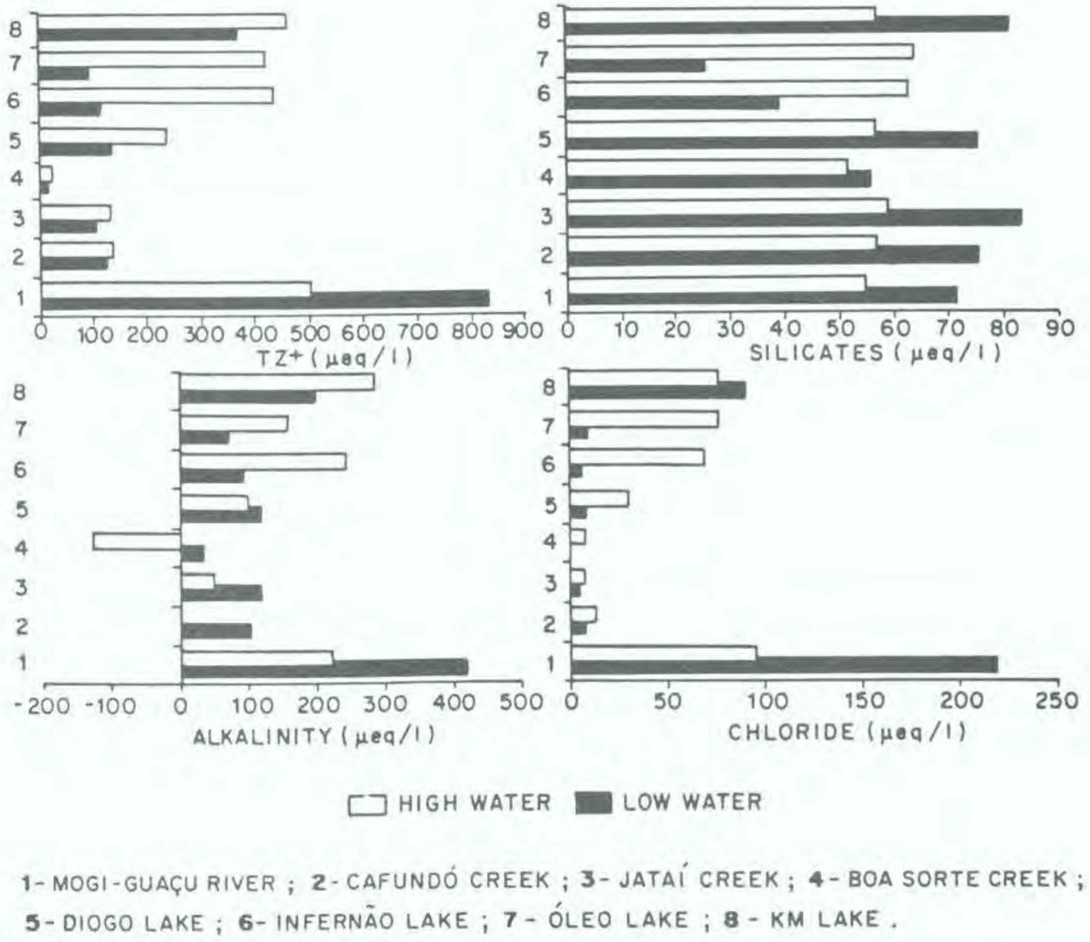


Figure 4. Comparisons of average total cation concentration (TZ+), reactive silica, alkalinity and chloride concentrations for the Mogi-Guaçu river, oxbow lakes and streams sampled at the Jataí Ecological Station at low (16-18/10/87) and high (19-21/03/88) water river stage.

During low water period TZ+ (μeq of Ca+Mg+Na+K; Stallard and Edmond, 1983) (Figure 4a) is much larger in the river (836 μeq) than in the lakes (average of 180 μeq) and in the streams (average of 85 μeq). The increased concentrations of major cations in the river, possibly due to evaporation (see below), is diluted with basin runoff plus direct rain during the high water period. As river waters enter the lakes (Infernão, Oleo and Quilômetro) their water quality becomes similar to that found in the river. Diogo lake, a lake which is connected to the river at all times, is affected by the river water to a lesser extent since it is fed by the

Cafundó creek with a relatively large volume of local runoff (average base flow discharge of 850 l s^{-1} at low water period). This local runoff is greatly enhanced during the rainy season and apparently the system Cafundó creek-Diogo lake always drains into the river (in spite of the damming caused by the river). Average water quality in Diogo lake therefore does not reach river standards (Figure 4a), and the lake may be classified as a drainage lake. Water residence time in this system during the dry season is only about 20 h (Krusche, 1989).

The chosen sampling points in the streams were not affected by river waters during the time of sample collection. We only noted the action of damming at their mouths caused by elevated river waters during the rainy period. Ion load in streams increased significantly during this period owing to local surface and ground water runoff. As their TZ+ are relatively low in the dry season, this contribution promoted a large difference between dry and rainy period TZ+ values.

Infernão and Oleo lakes possess a similar water quality at low or high water period. These two lakes are only about 200-250 m apart and are surrounded by a richly vegetated wet land. They become completely connected during high river stage. These two lakes and Quilômetro lake are directly fed during low water season by ground water seepage.These observations are confirmed by data for reactive silica (b), alkalinity (c) and chloride (d) (Figure 3) which are also good hydrological tracers of interactions occurring between river water and local water existing in the floodplain.

Stable isotope data show that waters are significantly depleted in ^{18}O and ^{2}H during the high water season compared to the low water season (Figure 5). Ground water, surface runoff and rain water are very similar to waters found in all lakes and creeks which demonstrates that there is only one water mass in the floodplain at the high river stage.

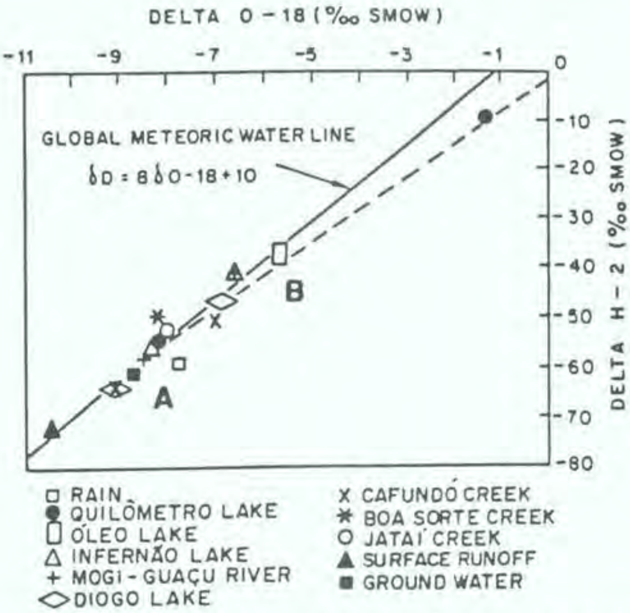


Figure 5. Relationships between average δ0-18 and average δH-2 relative to SMOW in meteoric waters samples. Most composition falls on the global meteoric water line δH-2 = 8 δ0-18 + 10 (Craig, 1961). See text for details.

The same systems, however contain waters significantly richer in ^{18}O and deuterium during the dry period. Evaporation under equilibrium conditions is invoked to explain these trends. Most of the system waters plot at the global meteoric water line ($\delta D = 8\ \delta^{18}O + 10$; Craig, 1961) for high (group A) and low water (group B) river stage while some of them plot at a line with a different slope ($\delta D = 6.9\ \delta^{18}O - 0.6$) typical of water which suffered strong evaporation effects in a fast evaporation process. This is specifically the case of Quilômetro lake, a typical closed basin lake, which shows high heavy isotope contents.

Bottom lake sediments of these lakes show, as in the case of Infernão lake (average ^{13}C content of -30.6 per mil), that although this lake is nearly completely covered with aquatic macrophytes (all C3 plants; average ^{13}C composition of -27.0 per mil) during the entire year, lake sediments show a strong contribution (as much as 67% in the Infernão lake) of terrestrial carbon (-29.0 per mil). More data is however, necessary to verify this observation.

Mogi-Guaçu river bottom sediments have a more positive $\delta^{13}C$ value (-24.2 per mil) compared to the other water bodies. Such a value may well represent the contribution of sugar-cane plants (a C4 plant; preferentially concentrates ^{13}C in its tissues) which are cultivated throughout its watershed. Similar explanation could be invoked in the analysis of the average $\delta^{13}C$ value for Diogo lake sediments (-24.8 per mil). An extra source of heavier carbon must be considered in this case that is represented by an abundant gramminae stand found on lake banks (average $\delta^{13}C$ of -25.9 per mil).

Surface water geochemistry (lakes, streams and river). To identify the origin of the surface waters found in the study site the major features of the relationship between water chemistry and substrate lithologies is examined, focusing on the geological features that characterize the area and the relation of these to the chemistry of the waters.

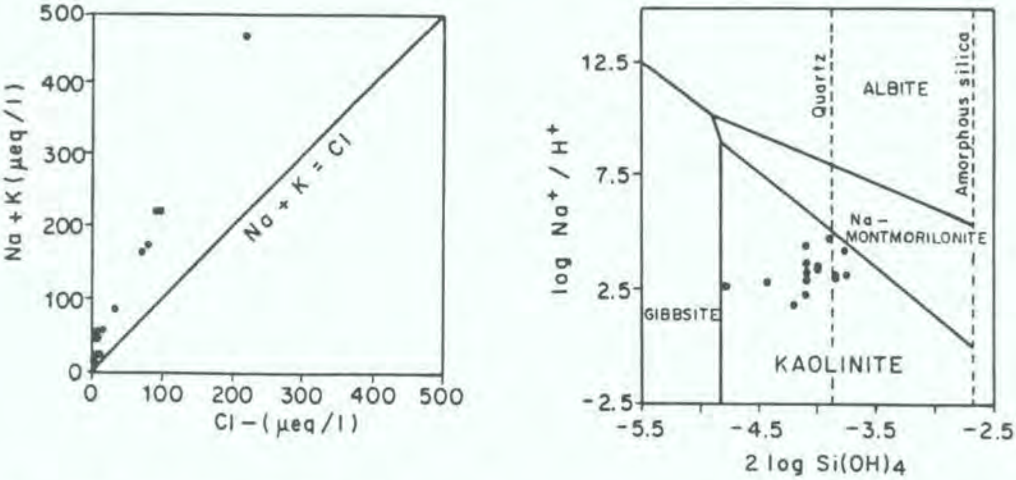


Figure 6. Graphs of $Na^+ + K^+$ versus Cl^- (μeq l^{-1}) showing the effects of chloride mineral weathering. (b) Stability diagram of some silicate minerals. See text for explanations.

All water samples collected at both high and low water river stage from lakes, streams and river have a relationship of concentration of [Na+K] versus Cl^- indicating that their chemistry is controlled by weathering of Na and K aluminosilicates (Figure 6a) as they plot above the 1:1 line. A graph (not shown in this paper) of concentrations of 2[Ca+Mg] versus [alkalinity + 2 SO_4] corroborates this, as the waters plot below the 1:1 line indicating that part of the anions are possibly utilized balancing the cations produced in the weathering of the silicates. These relationships were originally proposed by Stallard and Edmond (1983) for Amazonian rivers.

The relatively low ionic load in local waters at the study site is in agreement with highly weathered rocks, sandstones with basalt intrusions, found in the area (Botucatu and Pirambóia Formations) (Sinelli, 1979). The incongruent dissolution of the plagioclases and feldspars will establish an equilibrium with kaolinite in the water. All samples plot in the stability field of that mineral (Figure 6b) which was found in all soil samples studied in the watershed of Jataí and Cafundó creek (Lorandi, ms. in prep.)

Diogo lake: A case-study of seasonal variations. The main features of biogeochemical seasonal variations in the Cafundó creek-Diogo lake system and the adjacent Mogi-Guaçu river are shown in Figures 7a (Na, K and Cl^-), 7b (Ca, Mg and alkalinity) and 8a (total N, total P and total suspended solids) and 8b (nitrate, ammonium and reactive silica) (Krusche, 1989; Krusche and Mozeto, ms. in prep.).

Seasonal variations in the concentrations of major ions, and the difference in the chemical load between river (upper curve) and lake-stream system (lower curve) is evident. The river always has a higher ion concentration than the lake and the stream, and almost invariably, its waters are progressively enriched by evaporation through the dry season. Sodium and chloride are the best tracers of this phenomenon (Figure 7).

At the beginning of the rainy season river water is diluted by rain (apparently except for magnesium) but element concentrations are still higher than in the local waters. These, with the advent of the inundation of the low lands, undergo an enrichment at a level which is proportional to the process of mixing of different water masses. Since the hydrograph recedes rapidly and the residence time in this particular floodplain watershed is very short, water in the Diogo lake reaches a quality comparable to the low water river stage. Such a period, as shown by Figure 7, is characterized by a higher chemical stability in the systems in which local waters are chemically very poor (low ionic strength). The behaviour is opposit to the one found at high river stage in which waters in the floodplain have a low chemical stability with high ionic strength.

These field data (major ion distributions) permit one to define functional units valid for the ecosystems at the Jataí Ecological Station: the local watershed for low river stage period in which the discharge of ground water is significant in determining a high chemical stability in the systems (constant water quality associated with a low ionic strength), and the regional watershed for the high river stage period in which the inundation pulses is the main cause for disturbance, establishing a low chemical stability (variable water quality with a high ionic strength) in the systems.

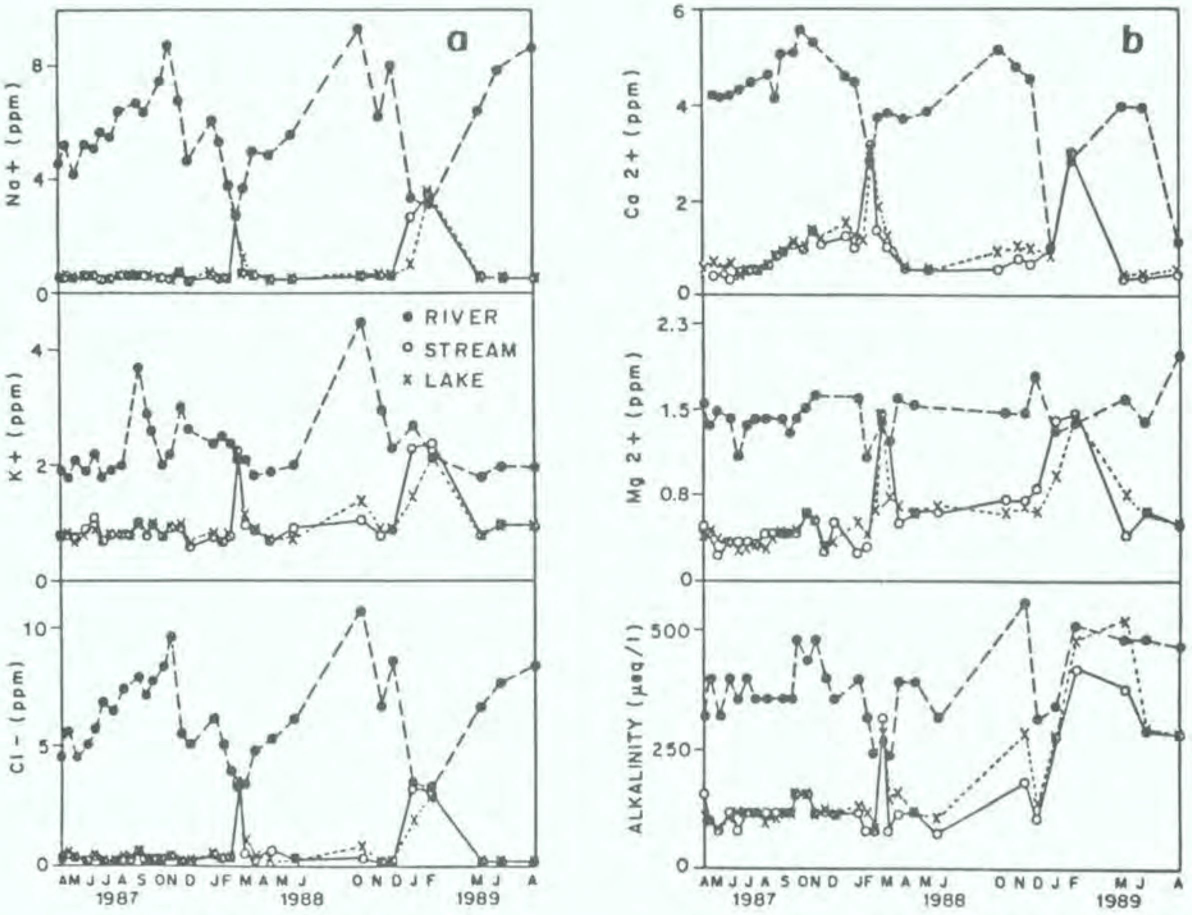


Figure 7. Seasonal variations in concentrations of sodium, potassium and chloride (a); calcium, magnesium and alkalinity (b) at high, low and intermediate water river stage.

Seasonal variations in nutrient concentrations are not as discernible as they are for the major ions. Mogi-Guaçu river carries larger concentrations of total N, total P and total suspended solids (TSS) in its waters compared to those existing in Diogo lake and Cafundó creek waters (Figure 8a). High water river stage are periods during which total N, P and TSS increase significantly, mainly due to the increase in surface runoff in the regional and local watersheds. The low water season is characterized by lower, but still rather variable, contents is these nutrients. These seasonal variations in different nutrients are somewhat related, but no statistically significant correlation exists among them. The data set is thought not to be extensive enough for statistical significance. It would be expected, though, that most of the total P. is associated with the particulate fraction in the water, TSS (Schmidt, 1972; Fosberg, 1984). The same trend would be expected for total N, as most N in the study site is also present in the particulate form, a trend which is opposed by that found in the Amazon river and its lakes (Schmidt, 1972; Fosberg, 1984).

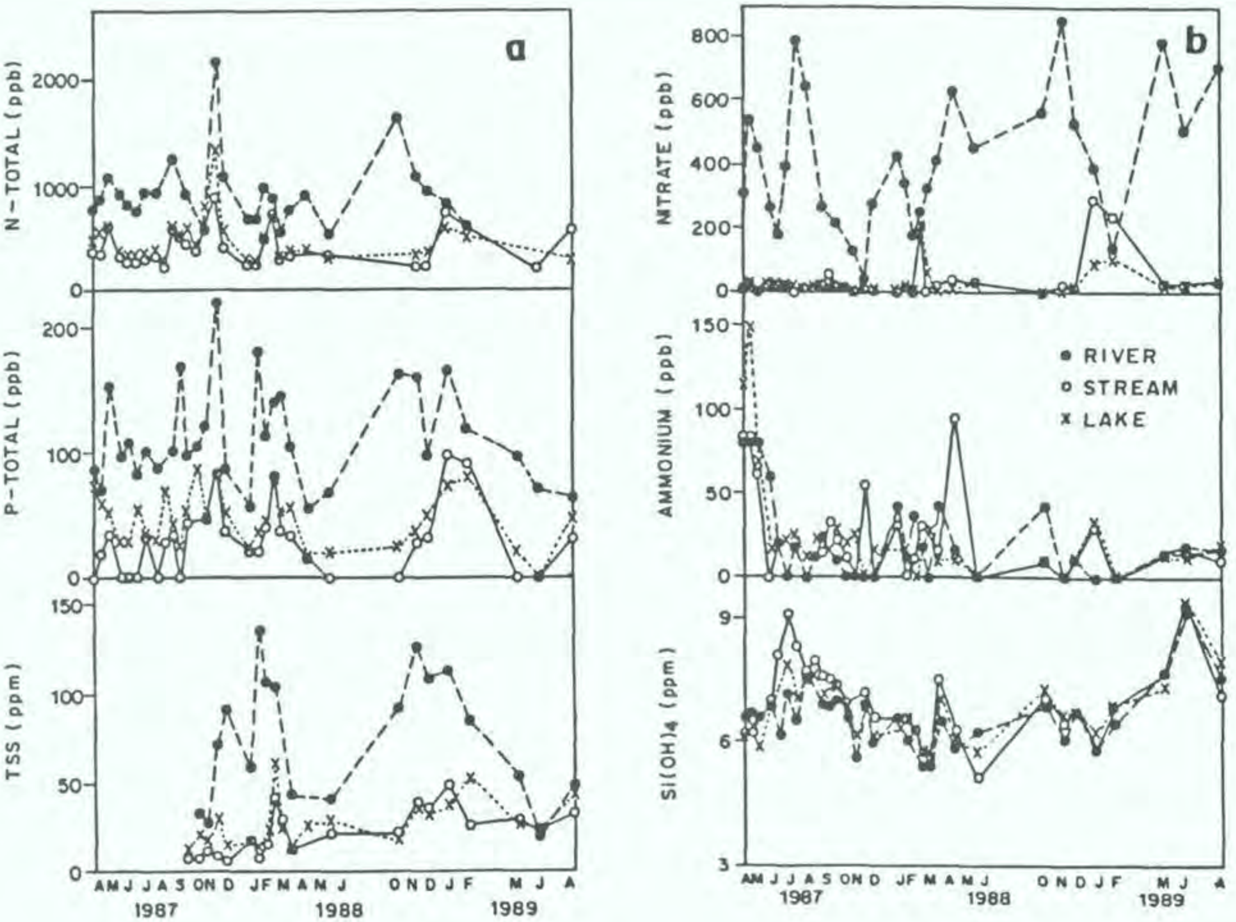


Figure 8. Seasonal variations in concentrations of total nitrogen (N-tot.), total phosphorus (P-tot.) and total suspended solids (TSS) (a); nitrate, ammonium and reactive silica (b) at high, low and intermediate water river stage.

Nitrate and ammonium concentrations (Figure 8b) (nitrite is below detection level of 10 $\mu g\ l^{-1}$) are quite variable for the river throughout the hydrologic year and make a significant fraction of the total N (see Table 1). Dissolved inorganic nitrogen is present in lower concentrations in the lake and many times lower in the stream waters than in the river waters.

The changing nutrient composition in water of low to high water periods can also be attributed to storage effects. These effects would obviously have different importance for a drainage lake (Diogo) (water residence time is of one day) as opposed to a ground water seepage Lake (Infernão). Lake Infernão is connected with the river for a very short period of time during high water stage, but, since it is not drained, the residence time of river-borne materials must be high. These materials would not be exported back to the river until next year's inundation.

These patterns should define a very distinct storage effect in nutrient concentrations among these lakes. Unfortunately, the present set of data does not permit any further interpretation in this respect.

Table 1. Average concentrations of soluble reactive phosphorus (SRP), total phosphorus (TP), dissolved inorganic nitrogen (DIN) and total nitrogen (TN) in $\mu g.L^{-1}$ of lakes and rivers in the Amazonian ecosystems (Fosberg et al. 1988) and in the Mogi-Guaçu river ecosystems (this work)

	Amazônia			Mogi-Guaçu system*			
	Floodplain	Rivers		Lakes		Rivers	
	lakes	Andean waters	Black/Clear waters	H	L	H	L
SRP	8.4	21.4	5.0	<10	<10	<10	<10
TP	27.3	61.4	19.2	64.4	60.4	145.0	98.2
DIN	25.9	156.9	46.2	113.8	70.5	333.0	232.3
TN	413.2	372.6	358.6	352.0	650.3	547.0	932.4

H = high water river stage
L = low water river stage
*(i) Various river and lake samples recently collected were analysed for SRP through a new technique (flow injection analysis with prior sample enrichment in an ion exchange column; Mozeto et al., ms. in prep.) and the concentration level was below detection limit of 0.5 µg.L-1. (ii) DIN in the river and lakes for high and low water periods are respectively 100 and 95.6% nitrate, 82% nitrate and 75.8% ammonium. This high concentration of nitrate in the river water is probably of anthropic origin.

Concentrations of nutrients (soluble and total, P and N) of ecosystems at the Jataí Ecological Station are compared to the Amazonian rivers and floodplain lakes (see Table 1). As for the Mogi-Guaçu river and oxbow lakes, Amazonian floodplain lakes and rivers have concentrations of SRP (soluble reactive phosphorus) below 10 μgL^{-1} (except for Andean water rivers). Mogi-Guaçu systems have, in general, higher concentrations of total P than the Amazonian systems which are, however, comparable to the Andean water rivers. Dissolved inorganic N concentrations are in general higher at the Mogi-Guaçu river and lakes than the Amazonian rivers and floodplain lakes while total N levels are comparable for lakes of both systems but higher at the Mogi-Guaçu river in comparison to the Amazonian rivers (see Table 1).

Implications for ecological impact. One important question regarding the ecology of floodplain lakes is the role that the factors controlling lake nutrient chemistry play in sustaining the fisheries productivity in these systems (Fosberg et al., 1988). At the Jataí Ecological Station lakes is not yet known whether N and P are growth-limiting nutrients for phytoplankton. Rates of N immobilisation in the water column and the sediments at the Infernão lake are very low and would indicate that N is not a limiting nutrient for phytoplankton growth

(Ballester, 1989). In addition to this, other studies have shown that the supply of nutrients by the river to the lakes is the controlling process of aquatic macrophytes growth and hence the growth of periphyton (Nogueira, 1989). As dissolved inorganic forms of N and mainly soluble reactive phosphorus are present in lake waters at relatively low concentration levels or below detection limits (10 µgL-1), particulate N and P brought by the river together with the portion generated in the systems (not yet known) would be the key forms of nutrients available for producers. These particulates are a major portion of the total N and P in these systems (see Figure 8 and Table 1).

Productivity in these systems may therefore be limited by the supply of particulate N and P by the river as well as the amount released by the decomposition of aquatic plants and from the sediments. These two reservoirs stock the majority of the N and P at Infernão lake (Nogueira, 1989). However, it is still unknown how fast and how much bioavailable N and P these reservoirs provide to the water column. Ongoing studies planned to evaluate the sediment release should define these rates.

In the Amazonian basin (Araujo-Lima et al., 1986) floodplain lakes dominate productivity of characiform detritivore fish which depend on growth of planktonic algae. The Mogi-Guaçu oxbow lakes may represent a similar case as fish is the most important fauna in these systems (Galetti et al., 1989). Therefore, changes in the way in which these lakes interact with the river and with terrestrial systems will determine future changes in fisheries productivity.

Nutrient and ion chemistries of the different lakes and streams studied are in general quite distinct from one to the other, despite the relatively small surface area of the floodplain site. The explanation for this observation may be that these lakes are presently in different ecological stages. These stages would determine different internal levels of material metabolism.

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SPATIAL VARIATION IN PHOSPHATE CONCENTRATIONS OF SMALL-ORDER STREAMS DRAINING VOLCANIC LANDSCAPES IN COSTA RICA: SOURCES AND IMPLICATIONS FOR NUTRIENT CYCLING

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Introduction

Relative to temperate streams, little is known about the nutrient chemistry of tropical streams (Meybeck 1982) and how it reflects regional biogeochemistry. In North America, generalizations have often been made regarding chemical properties of water based on regional geology. For example, water draining glacial deposits has been classified into composition categories corresponding to the type of glacial materials drained (e.g. Freeze and Cherry 1979). Regional generalizations have also been made regarding nitrogen versus P limitation of primary production based on ratios of biologically important nutrients in stream waters. While streams of the eastern US exhibit relatively high molar N:P ratios, suggesting P limitation (Omernik 1977), streams draining desert/semi-desert areas of the southwestern US and Pacific Northwest tend toward nitrogen-limitation (Thut and Haydu 1971, Sommerfeld et al. 1974, Grimm et. al. 1981, Triska et al. 1989).

While our understanding of tropical aquatic systems in the New World has been greatly expanded by studies in the Amazon basin (e.g. Fittkau 1967, Sioli 1975, Melack and Fisher 1983, Tundisi and Curi 1984, Lesack 1987), these studies have few parallels in Central America. Basic data on nutrient concentrations of Central American streams are scarce (McColl 1970); consequently there is a lack of fundamental knowledge of how regional geochemistry is reflected by water chemistry and aquatic biota. Because of the recent geologic history of volcanic activity in Central America, many watersheds are underlain by volcanic basalt and are characterized by young soils that contain more P relative to older soils of the Amazon (Vitousek 1984). Correspondingly, relatively high P levels have been measured in many streams that drain younger volcanic parent materials in Costa Rica (e.g. Pringle and Triska 1986, Pringle et al. 1986, Paaby- Hansen 1988, Pringle et al in press).

This paper examines the variability in P levels of small-order streams over three spatial scales: (1) variation between watersheds draining volcanoes at three different stages of volcanic activity (dormant, moderately active, explosively active); (2) variation in ambient stream nutrient levels within the watershed of a lowland swampforest stream; and (3) bankside variation in groundwater nutrient chemistry on a scale of meters across the riparian zone of a lowland stream.

Study Sites and Methods

Nutrient chemistry was measured in streams that drain three volcanoes (Figure 1) representing different stages of volcanic evolution: (1) Arenal (1633 m), an explosively active volcano located in the Cordillera de Tilaran; (2) Poas (2708 m), a moderately active volcano located 60 km to the southeast of Arenal within the Cordillera Central; and (3) Barva (2906 m), a dormant volcano located adjacent to Poas in the same mountain range to the southeast. In addition to streams that drain recent to Pleistocene lavas, several streams were sampled that drain alluvium and older Pliocene lavas, located to the northeast and northwest of Volcan Barva, respectively (Figure 2).

Spatial variability in stream nutrient chemistry was also determined at the watershed scale along a lowland swampforest stream, the Quebrada Salto, that drains La Selva Biological Reserve (10°26'N, 84°58'W), located in the lowland foothills of Volcan Barva's Atlantic slope. The Salto is an undisturbed third-order stream that spans elevations between 300 and 36 m above sea level (Pringle et al. 1984). Water samples were collected at the mouths of all major tributaries entering the Salto and its tributary, the Pantano, which lie within the Reserve.

Bankside variation in groundwater nutrient chemistry across a swampforest riparian zone of the Salto watershed was also measured. Two transects of groundwater wells were established perpendicular to the Salto River within the swampforest on the west and east banks (Figure 3), after a design modified from Peterjohn and Correll (1984). The west Salto well transect consisted of three clusters of wells located perpendicular to the stream channel at 3, 10 and 16 m from the channel. The east transect consisted of two well clusters, also located perpendicular to the stream channel at 3 and 10 m from the channel (Figure 5). Each well cluster consisted of 3-4 wells located approximately 3 m apart. Wells were 3-4 m deep, consisting of 10 cm diameter, polyvinyl pipes capped with .006 slot well points (Geotech®). One day prior to sampling, wells were pumped free of water and allowed to refill with fresh groundwater. Water samples were collected using a battery-operated peristaltic pump (Geopump®) equipped with a filtration apparatus (0.45 µm membrane filters).

Two replicate filtered (0.45 µm Millipore®) and two replicate unfiltered samples were collected in clean polyethylene bottles at each sampling site. Filtered water samples were analyzed for nitrogen (NO_2+NO_3-N) and soluble reactive phosphorus (SRP) using the hydrazine reduction method (Kamphake et al. 1967) and molybdenum blue technique (Strickland and Parsons 1972, APHA 1985), respectively. Unfiltered samples were used to measure total phosphorus (TP) using the molybdenum blue technique, preceded by acid hydrolysis. Conductivity was measured with a conductivity meter. Cl and SO_4 were measured with an ion chromatograph and Mg, Na, and Ca were measured with atomic absorption spectrophotometry. The relationship between SRP, major solutes and conductivity was determined via Pearson correlation analysis (Sokal and Rohlf 1981).

Results

Streams draining Pleistocene lavas within all three volcanic study areas displayed a high variation in SRP, TP , chloride and conductivity (Table 1). Arenal, the youngest and most active of the three volcanoes, displayed highest mean levels of all four chemical parameters. Mean levels of SRP ranged from 55.7 $\mu g\ L^{-1}$ (Poas) to 88.4 $\mu g\ L^{-1}$ (Arenal); (Table 1). In contrast, all waters draining the older and more highly weathered Pliocene lavas located to the northwest of Volcan Barva had SRP levels below detection (<5.0 $\mu g\ L^{-1}$). Streams draining alluvium were relatively high in P . However, the small sample size (n=3) and the fact that all streams sampled on this landscape unit were highly disturbed (draining pastureland that has been almost completely deforested), precludes any generalizations.

A high spatial variability in P levels occurred within watersheds in the lower foothills of Volcan Barva. Levels of P in spring seeps and tributaries within the Salto watershed (Figure 3) range from less than 5.0 to 250 μg L^{-1}. The mainstem of the Salto receives numerous P-rich inputs, exhibiting high ambient P levels (50-200 μg L^{-1}). In contrast, the Pantano, a closely adjacent tributary of the Salto (Figure 3), receives no P-rich inputs from seepages and tributaries and ambient stream P levels are consequently relatively low (typically < 15 μg L^{-1}). A survey of forty first- and second-order streams draining the lower foothills of Volcan Barva indicated that P levels were highly correlated with conductivity (r=0.93), alkalinity (r=0.95), Cl (r=0.89), Na (r=0.90), Ca (r=0.84), Mg (r=0.85) and SO_4 (r=0.73) but not NO_3 (r=-0.09) (Data from Pringle et al. in press).

Figure 1. Map of Costa Rica showing location of the Cordillera Central, Cordillera de Tilaran and respective locations of the three volcanic study sites: (1) Arenal, (2) Poas, and (3) Barva. (Figure from Pringle et al. ms submitted.)

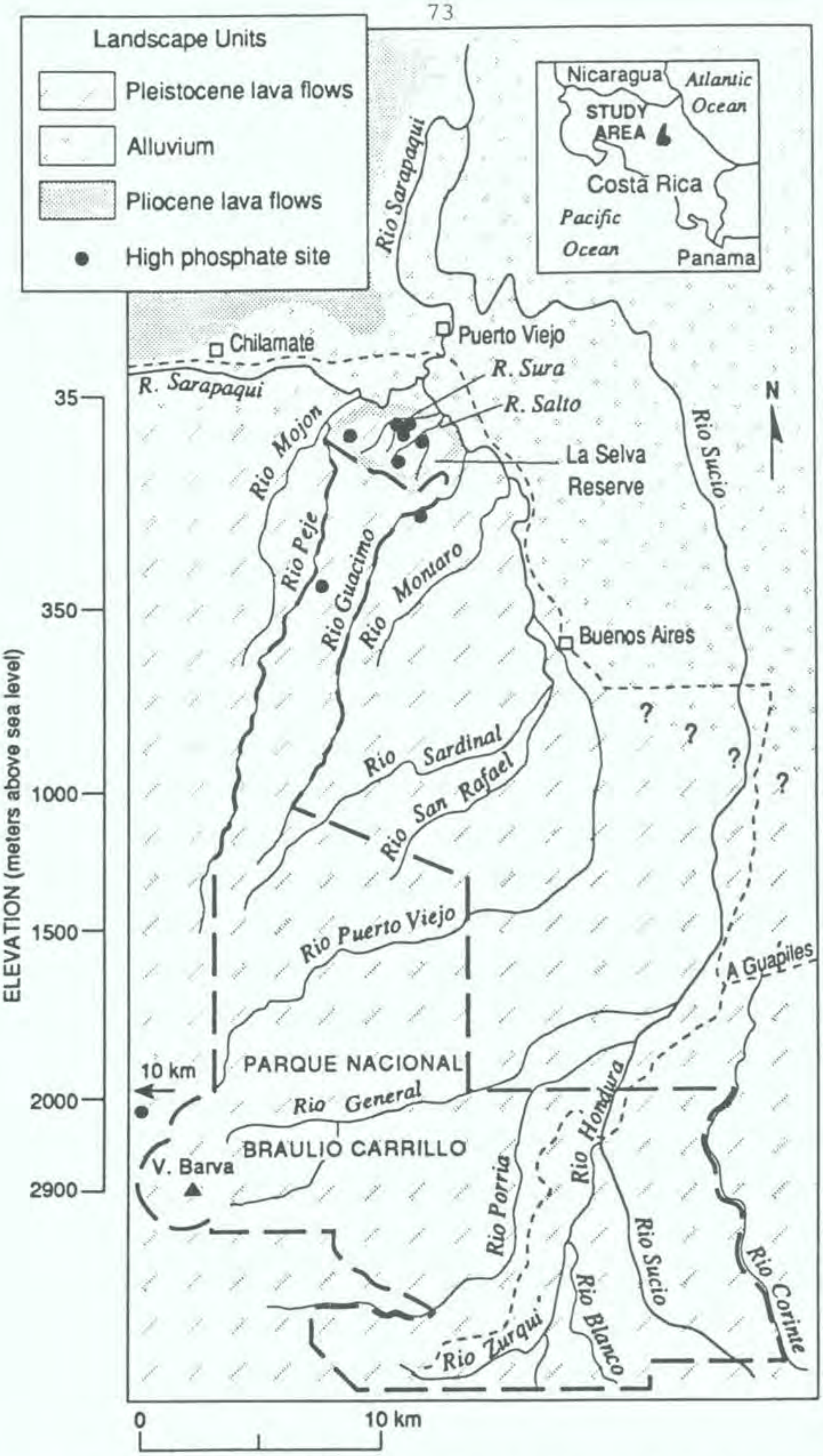

Figure 2. Location of P-rich streams (≥ 50 µg SRP L^{-1}) with respect to the location of Pleistocene and Pliocene lava flows and the elevational continuum. The location of the La Selva-Braulio Carrillo Park land transect on Costa Rica's Caribbean Slope is indicated in the inset. In some locations the exact boundary between alluvium and Pleistocene lavas is not known (?). (Figure from Pringle et al. ms submitted).

Table 1. Levels of SRP ($\mu g\ L^{-1}$), TP ($\mu g\ L^{-1}$), chloride ($mg\ L^{-1}$) and conductivity ($\mu S\ cm^{-1}$) in first- to third- order streams draining different landscape units: (1) Pliocene lavas; (2) Pleistocene lavas; and (3) alluvium. Volcan Barva (dormant), Volcan Poas (moderately active) and Volcan Arenal (explosively active) represent different stages of volcanic activity within the Pleistocene landscape unit. (Data from Pringle et al. in press and Pringle et al. ms. submitted.)

		Landscape Unit				
		Pliocene Lavas	Pleistocene Lavas			Alluvium
			Barva	Poas*	Arenal*	
SRP	x	<5.0	75	56	88	129
	S.D.	---	95	121	77	28
	min.	<5.0	<5	10	8	97
	max.	---	301	471	234	150
	n	9	27	14	10	3
TP	x	5.2	89	57	102	216
	S.D.	5.2	110	123	85	112
	min.	<5.0	<5	11	9	92
	max.	5.9	405	481	279	309
	n	9	28	14	10	3
Cl	x	2.8	5.8	11.4	44.3	2.2
	S.D.	1.3	6.6	21.2	53.3	0.2
	min.	2.2	1.8	1.4	0.3	2.0
	max.	6.1	28.5	81.6	132	2.4
	n	9	28	14	10	3.0
cond.	x	31	94	516	1034	142
	S.D.	9	117	1494	1166	45
	min.	20	11	47	75	90
	max.	43	425	5700	3000	175
	n	9	27	14	10	3

*Also recent lavas

Spatial variation in P , nitrogen and chloride concentration was also observed locally within the east and west Salto well transects (Figure 4). Solute concentrations in groundwater sampled from wells near the stream channel generally decreased with increasing distance from the channel. Phosphorus (Figure 4a) was highest in spring seepage areas in the east and west swamp transects and in the east swamp wells. Phosphorus levels were lower in the stream channel than spring seeps. Patterns of chloride concentration (Figure 4c) were similar to P , however, chloride was also relatively high in the west swamp wells located closest to the stream channel (3m), diminishing in concentration with increasing distance from the stream channel (10m, 16m). Nitrate levels were highest in the stream channel, slightly lower in the spring seeps and very low in all groundwater samples.

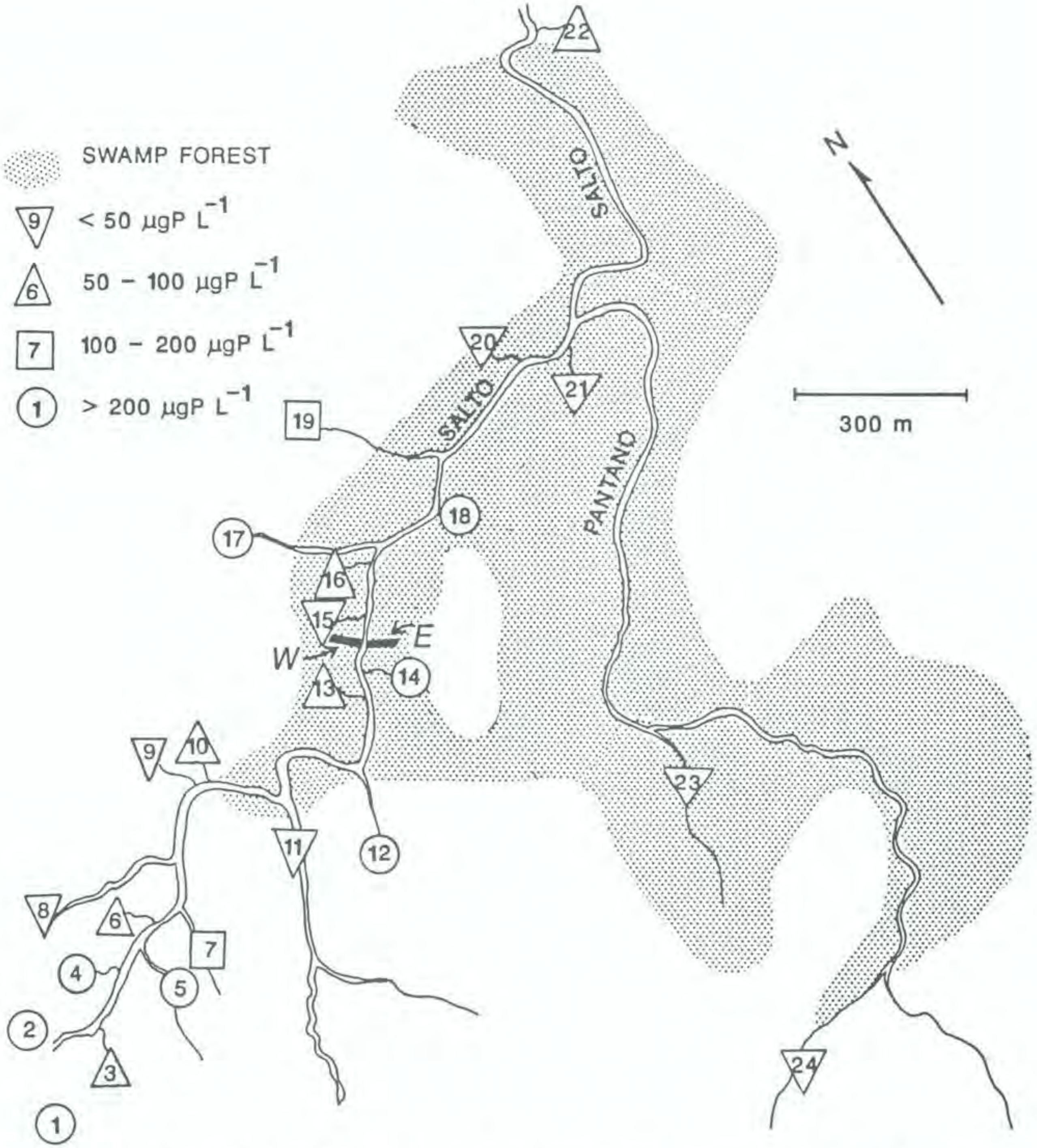

Figure 3. Spatial variability in phosphorus (SRP) concentrations within the lower Salto watershed, La Selva Biological Station, Costa Rica. The location of the east (E) and west (W) swamp well transects are indicated. (Figure modified from Pringle et al. ms submitted).

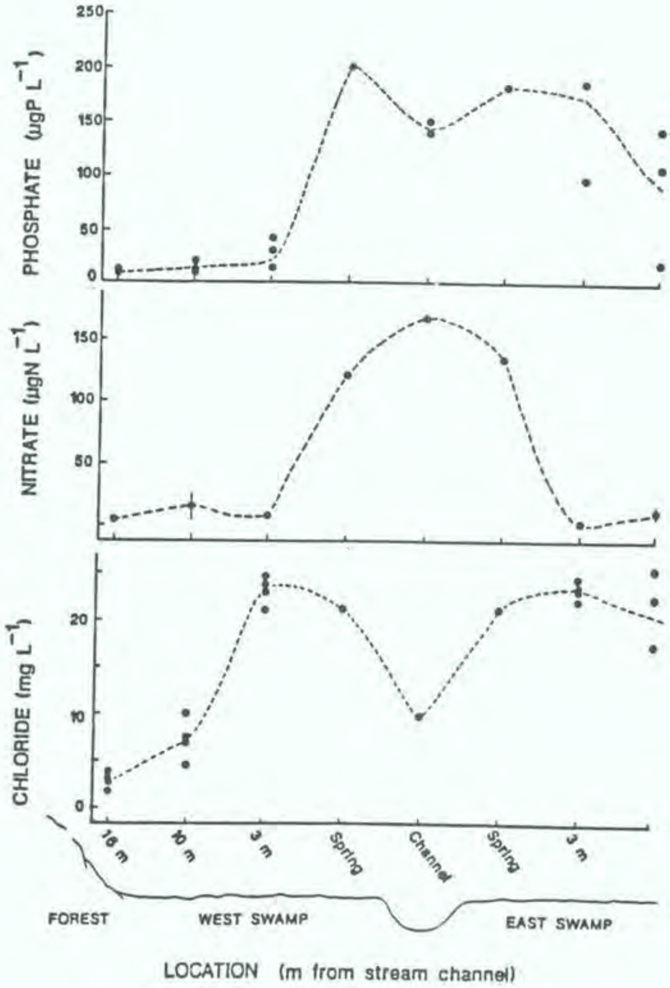

Figure 4. Levels of: (a) orthophosphate (μg L^{-1}); (b) nitrate + nitrite - N (μg L^{-1}); and (c) chloride (mg L^{-1}) in the stream channel, spring seep and groundwater at various distances (3 m, 10 m, 16 m) from the stream channel in the East and West swamp transects. Water samples were collected at the beginning of the rainy season on 29 April 1987.

Discussion

In contrast to older, highly leached soils of the Amazon Basin, volcanic landscapes of Central America were formed by geomorphic processes (e.g. lava flows, lahars and ashfall) that have resulted in high variability in geochemistry both within and between landscape patches in a relatively small geographic area. Stream nutrient chemistry not only reflects patchiness of the parent material within the drainage but also the weathering potential of different types of volcanic waters within the drainage (e.g. thermal, acidic, etc.). The leaching potential of thermal waters and gases emitted from magmatic processes, results in the weathering and breakdown of volcanic rock and waters highly charged with mineral salts (Paniagua and VanDerbilt 1979) including biologically important elements found in volcanic rock such as P .

Of the streams draining Volcan Poas, highest SRP levels (471 $\mu g\ L^{-1}$; Table 1) were measured in the Rio Agrio which arises directly from the flanks of the active crater as springs with low pH (pH=2.2) and high salinity (80 mg Cl L^{-1}) (Brantley et al. 1987, Pringle et al. ms. submitted). A hot acid lake that was situated in the active Poas crater disappeared in March 1988 to become stored as groundwater, some of which re-issues within the Rio Agrio drainage. The lake has since re-formed. Before the lake's disappearance in 1988, studies conducted by Brantley et al. (1987) indicated that high lake water temperatures were a result of the passage of fumaroles through bottom sediments and that the extremely low pH of the water was a result of the continual condensation of fumarolic acid vapors. Brine seepage was supported by the fact that, despite continual inputs, no outlets from the lake were observed for dissolved species other than Si, S, and Ca.

One fifth of the forty active volcanoes in Central America contain crater lakes and evidence exists that crater lakes commonly evolve along the same chemical lines (e.g. Brantley et al. 1987). In addition, many volcanoes once contained lakes that acted as condensers for fumarolic inputs. Groundwater may still be affected by subsurface seepage that took place in the past history of the volcano. Furthermore, brine formation can take place where fumarolic vapors condense in the absence of hot crater lakes (White 1957). Thus, it is a reasonable hypothesis that similar phenomena are likewise affecting ground and surface waters draining volcanic landscapes in other parts of Central America.

Volcan Arenal has the youngest volcanic terrain of the three volcanic study sites and provides another example of the weathering influence of volcanic waters in a geochemically patchy terrain. All high P streams sampled arose on the north side of the volcano which is covered with very recent lava flows and tephras deposited between 1968 and present. Three of the ten streams sampled were thermal (32-36°C), displaying high levels of Cl (69-108 mg L^{-1}), SO_4 (86-221 mg L^{-1}), Mg (42-110 mg L^{-1}), Ca (91-143 mg L^{-1}), Na (142-240 mg L^{-1}), and SRP (136.8-234.1 $\mu g\ L^{-1}$; Pringle et al. ms. submitted).

On the Atlantic slope of dormant Volcan Barva, an investigation of streams draining elevations from near sea level to 2,900 m above sea level (Pringle and Triska 1986, Pringle et al. in press) indicated that highest variability in ambient P levels was exhibited by streams in a localized area at low elevations (35-60 m), where the foothills of the central mountain range merge with the coastal plain (Figure 2). Extreme variability in P levels in streams of this localized region is due to deep, solute-rich, groundwater inputs that have localized discharge points at the terminus of Pleistocene lava flows (Figure 6; Pringle et al. in press). Given that the weathering and breakdown of phosphatized volcanic rock yields clay and free ions such as phosphate, silicate, iron and aluminum hydroxides in addition to other mineral salts, the high correlation between P and Cl, Na, Ca, Mg and SO_4 and the high levels of Si and Fe measured in P-rich seeps (Pringle and Triska, in press) is consistent with our hypothesis that solute-rich groundwater is influenced by volcanic brines and/or has been in contact with volcanic rock for a longer time period relative to more surficial waters.

It is difficult to make regional generalizations regarding chemical properties of water in volcanic landscapes such as Costa Rica. Ongoing volcanic processes occurring within active volcanoes (e.g. Volcan Poas and

Arenal) and processes that operated in the volcanoes' past geologic history (e.g. Barva) appear to be largely responsible for the anomalous patterns in nutrient chemistry displayed by streams draining respective volcanic landscapes (Table 1). In contrast, studies that have examined the nutrient chemistry of streams draining the Amazon (e.g. Sioli 1975, Margaleff 1983) indicate that stream chemistry does reflect the regional geochemistry of old and highly weathered soils. While some regions of Amazonia are geochemically richer than others (e.g. Fittkau et al. 1975), ambient P levels are relatively low compared to levels measured in volcanic areas in Africa (e.g. Talling and Talling 1965, Golterman 1973, 1975, Viner 1973, Lesack et al. 1984) or Central America (Pringle et al. in press), where phosphate is weathered from volcanic rocks.

In volcanic landscapes of Costa Rica, stream nutrient chemistry can vary significantly, not only from watershed to watershed but between streams in close proximity within the same watershed such as the Salto and Pantano. Phosphorus-rich waters are almost always first-order streams and/or spring seeps that become progressively diluted by larger streams, accounting for the range in P levels exhibited within stream drainages at lower elevations at the base of Barva (Pringle et al. in press). The Salto River (Figure 3) illustrates the high spatial variation in P concentration that can occur within a watershed. Some spring seeps and tributaries are solute-poor, dominated locally by surficial weathering processes, while others are dominated by deep and possibly distant groundwater inputs. Solute rich spring inputs occur along the mainstem of the Salto (Figure 3) which marks the eastern boundary of the youngest lava flow (deposited between early and late Pleistocene) of the two known lava flows of La Selva Biological Reserve (Sancho and Mata 1987). The Pantano tributary, which receives no solute-rich inputs, drains the older lava flow which was deposited in the early Pleistocene (Alvarado 1985).

Spring seeps surfacing near the east and west Salto swamp well transects illustrate the influence of deep groundwater inputs (Figures 4, 5). Groundwater in the east and west Salto spring seeps and deep well clusters 1, 4 and 5 (Figures 2, 5) had high chloride levels and presumably a different source of water than other wells farther from the stream or higher in the watershed (Pringle and Triska, ms. submitted).

Chloride and SRP were highly positively correlated in a survey of forty streams. However, the correlation between Cl and P levels was diminished in the 3 m and 10 m well clusters of the west Salto swamp transect (Figure 4a,c). Since Cl is conservative, low P relative to Cl in these locations may indicate P sorption to bankside sediments which have a high clay content (Pringle and Triska, ms. submitted). Based on Cl, results indicate that wells 1a-c, 4a-c and 5a-c in the east and west Salto transects are dominated by the same water source as the east and west Salto seeps (Figure 5).

Deep groundwater intrusions can have profound effects on nutrient cycling and biotic production. Nutrient limitation and its role in autochthonous stream production and biogeochemical cycling can vary significantly between streams in close proximity in the same watershed. *In situ* nutrient bioassays of algal growth have indicated that levels of major nutrients (N and P) are not limiting algal growth (as measured by chlorophyll a accrual rates) in the P-rich Salto, however, P amendments stimulate algal growth in the Pantano where light is not limiting (Pringle and Triska ms. submitted.). When P was experimentally added to the Pantano it was effectively retained by the channel. Dilution corrected retention was approximately 17% and eventually all phosphate was retained in a 700 m reach. Potential mechanisms of retention included both biotic uptake and sediment sorption (Pringle and Triska, ms. submitted). Other bioassay studies conducted in a drainage directly adjacent to the Salto (the Sura), that also receives P-rich groundwater intrusions, likewise indicate that ambient nitrate and phosphate levels are saturating to algal growth (Paaby-Hansen 1988). To our knowledge, virtually no studies have directly assessed nutrient limitation of algal growth in lotic systems of Amazonia so general comparisons are not possible. However, studies conducted in a floodplain lake (Setaro and Melack 1984) and reservoir (Tundisi and Curi 1984) indicated N limitation.

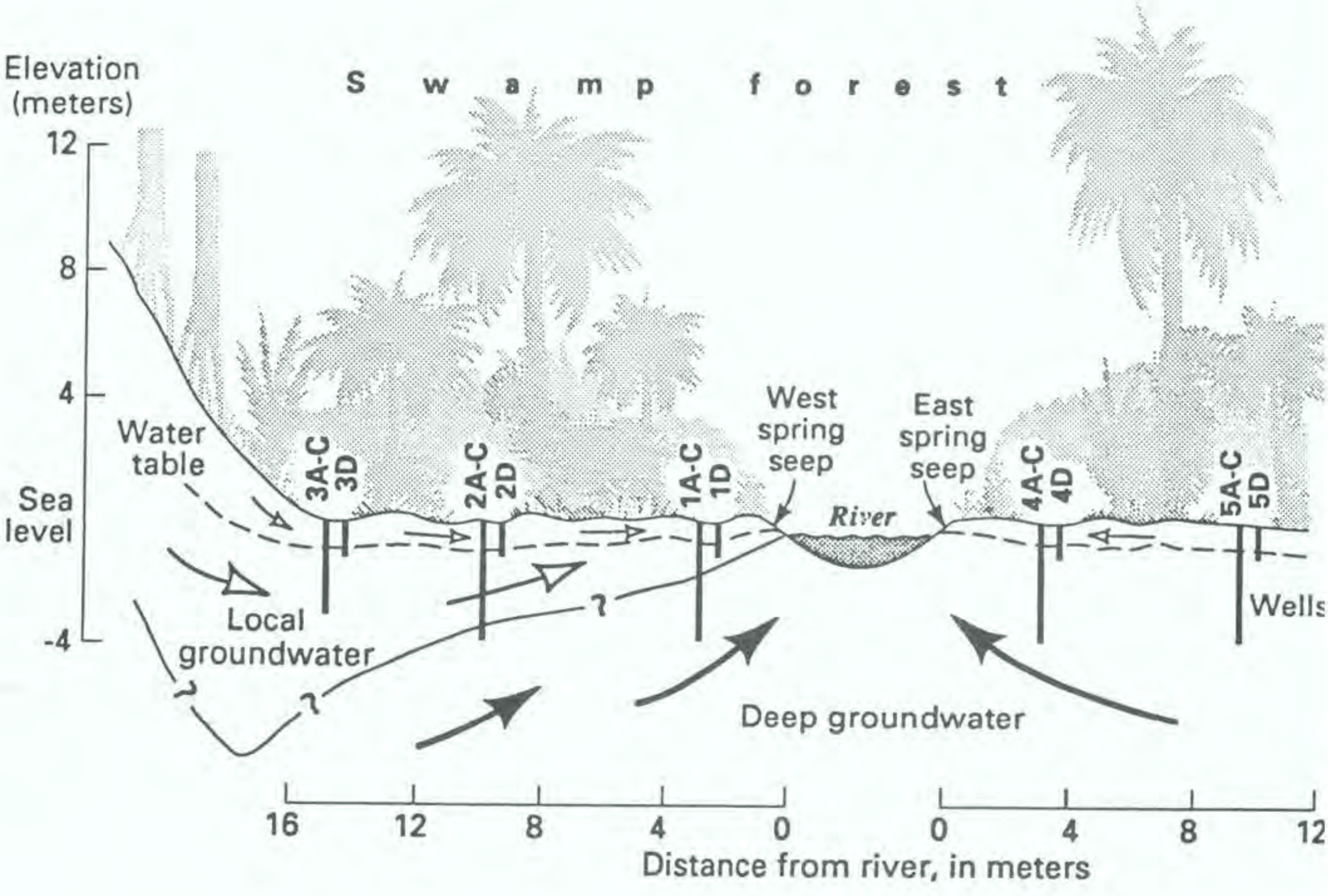

Figure 5. Conceptual model explaining spatial variability in water chemistry in the east and west Salto swamp well transects. Dark arrows indicate P inputs and soil saturation via deep groundwater sources while clear arrows indicate nitrogen inputs and soil saturation via deep groundwater sources while clear arrows indicate nitrogen inputs via surficial runoff. Depth of well penetration is indicated. Composition of waters at depths beneath the wells (>4 m) is unknown (?). (Figure modified from Pringle and Triska, ms. submitted).

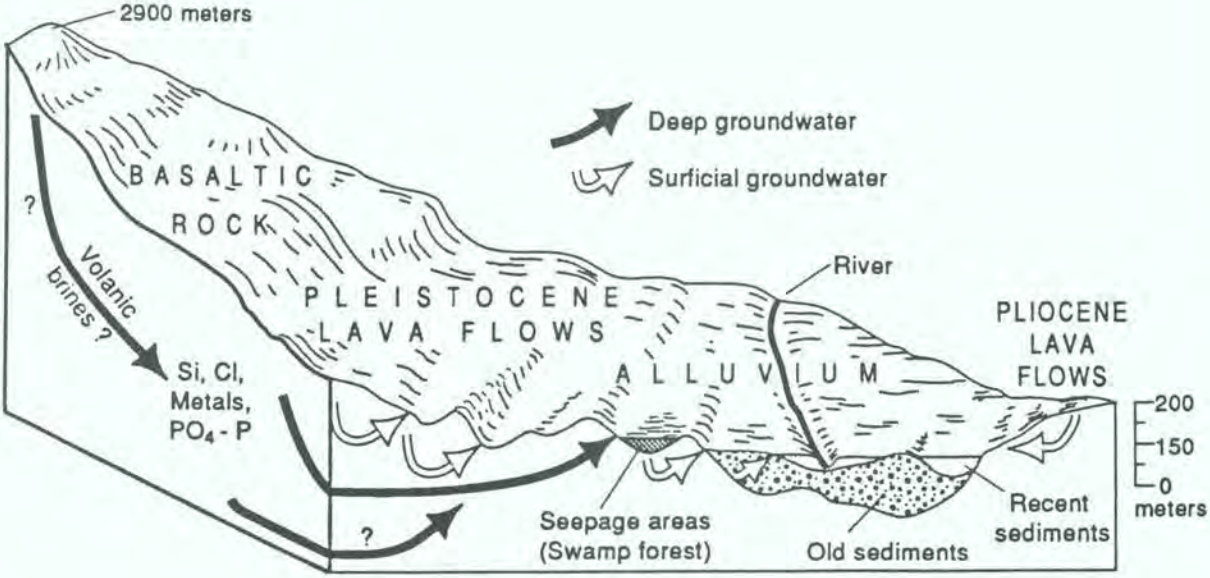

Figure 6. Simple conceptual model of groundwater transport in the La Selva-Braulio Carrillo elevational continuum on Costa Rica's Atlantic slope. Heavy precipitation results in soil saturation and downward percolation of deep, solute-rich groundwater (dark arrows) that weathers basaltic rock. Solute-rich waters are either old volcanic brines (?) stored in the water table that have subsequently become diluted by mixing with meteoric waters or they may be waters that have had a relatively long contact time with younger parent material of more recent Pleistocene lava flows. Solute-rich groundwater arises where lava flows merge with the coastal plan, sometimes coinciding with the location of lowland swamp forests. Shallow, solute-poor groundwater (light arrows) arises at various locations along the elevational continuum. (Figure modified from Pringle et al. in press.)

In conclusion, high regional and local variability exists in ambient P levels in small-order streams draining volcanic landscapes in Costa Rica. Phosphorus concentrations vary significantly and often dramatically between watersheds of different volcanic peaks, adjacent watersheds of the same volcanic peak, closely adjacent stream tributaries within the same watershed, and even across swampforest riparian zones on a scale of meters. Phosphate is weathered from volcanic rock and in some instances, enters streams via deep groundwater intrusions that are spatially localized within the landscape. Groundwater intrusions represent inputs of biotically essential nutrients that can influence nutrient limitation and its role in authochthonous stream production and basic nutrient flux within the stream ecosystem.

Acknowledgements

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PHOSPHORUS/NITROGEN INTERACTIONS IN ADJACENT AMAZON FORESTS WITH CONTRASTING SOILS AND WATER AVAILABILITY

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Introduction

Early ecological analyses of tropical rain forests emphasized the critical role of nutrients for maintaining forest productivity. Variations in soil characteristics such as texture and aeration, pH of soil solution, nutrient availability and presence of toxic elements exert a strong influence on species behavior and distribution. Adaptation to acid soils require highly efficient uptake and/or utilization of nutrients, especially P, calcium and magnesium (Marschner 1986). Most tropical lowland forests appear to be limited by P (Vitousek 1984, Vitousek & Sanford 1986), while their plant species seem to be tolerant to high levels of mobile aluminum (Sprick 1979, Sobrado and Medina 1980). On the other hand, in several tropical forests N has been shown to be the mineral nutrient recycled in larger quantities through litterfall than in any other terrestrial ecosystems (Rodin & Bazilevich 1967).

Forest ecosystems found in the northern portion of the Amazon basin, mainly within the Río Negro basin, are characterized by acid, highly leached soils with very low nutrient availability (Fittkau et al. 1975). Concretional oxisols and ultisols are characterized by low availability of cations and P, while spodosols are relatively impoverished in N (Medina and Cuevas 1989). Nutrient fluxes in litterfall reflect these differences (Cuevas and Medina 1986): smaller amounts of P, and larger amounts of N, are cycled in Mixed Forest of Tierra Firme on concretional oxisols than in Tall Amazon Caatinga forest on spodosols.

Phosphorus and N contents of leaves of higher plants are usually correlated (Medina 1984), although their respective internal mobilization and allocation patterns differ markedly (Marschner 1986). Phosphorus content on a molar basis is around one order of magnitude lower than that of N. Phosphorus deficiency impairs N utilization, due to its role as a universal molecule for energy transfer in organisms (Mengel and Kirby 1982). There is a differential rate of transport of N and P within the plant, P being translocated faster than N into tissues with meristematic activity (Grubb 1977). Nitrogen/phosphorus ratios in storages and fluxes within ecosystems can also be used to estimate vegetation response to varying conditions of soil water and nutrient availability. Here we present a comparative analysis of N/P ratios in two adjacent forest types with contrasting soils and water availability, Mixed Forest on Tierra Firme oxisols and Tall Amazon Caatinga on spodosols, in the Upper Río Negro region in the northern part of the Amazon basin in Venezuela. These forests grow on soils which appear to have extremely low nutrient availability (Herrera et al. 1978). N/P ratios in litter fall and decomposing litter will be related to leaf properties, rainfall regime and degree of soil water saturation.

Climate, soil types and soil water along topographical gradients in San Carlos de Río Negro

Climate. San Carlos de Río Negro has a tropical rain forest climate, with monthly rainfall generally higher than pan A evaporation, an average annual temperature of 26°C, and annual rainfall of 3565 mm. Median monthly rates calculated for the period between 1977-1985 show that maximum rainfall occurs around June, with lower rainfall periods during December and January (Fig. 1). In spite of high monthly rainfall values, there is a clear seasonality in the phenological behavior of the dominant tree species in the forest types considered here (Cuevas and Medina 1986). Annual variations in temperature and photoperiod duration are almost negligible at this latitude, therefore the vegetation appears to be responding to annual variation in the number of dry days (rainfall< evaporation, Medina et al. 1978). Due to the generally low water retention capacity of the soils, substantial water stress may develop during a series of even few dry days (Medina et al. 1978).

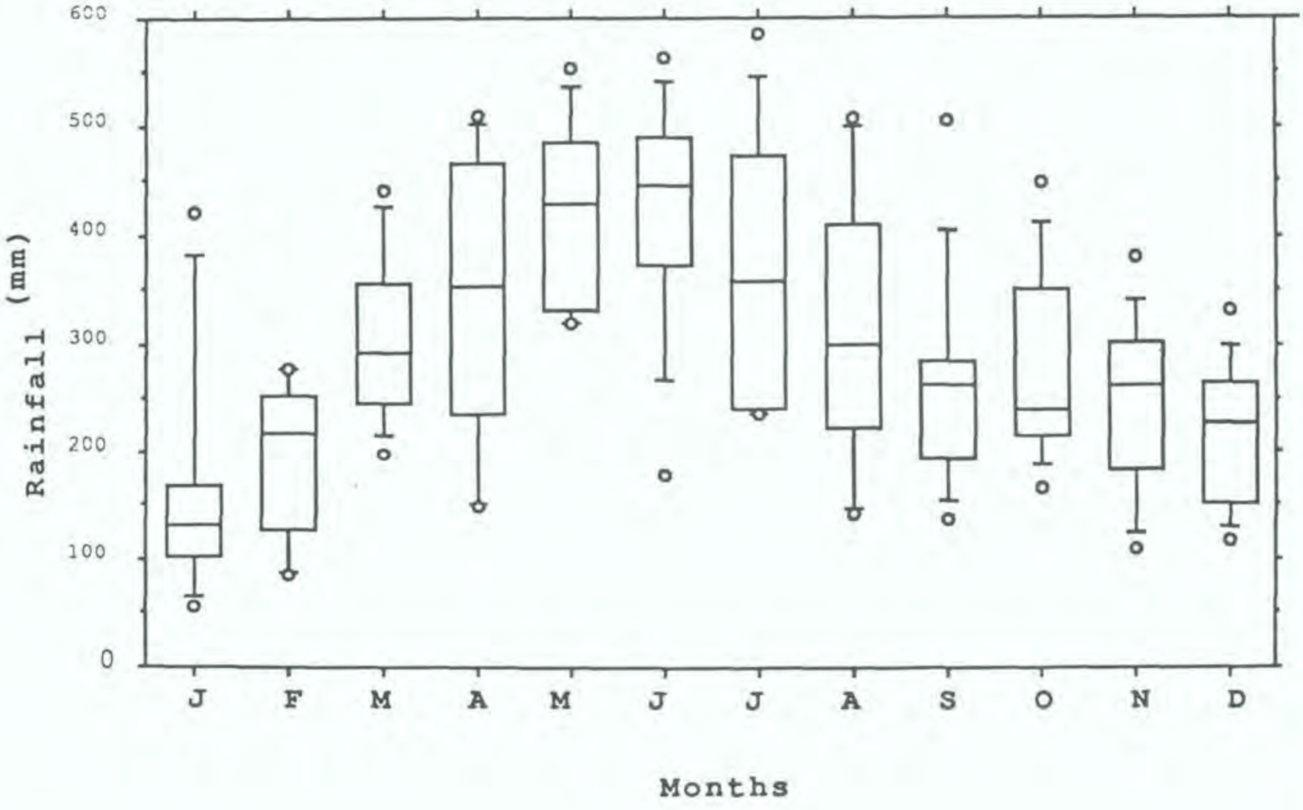

Figure 1. Median monthly rates of rainfall for the period between 1977-1985 for San Carlos de Río Negro, Territorio Federal Amazonas, Venezuela. Horizontal line in each box plot is the median of values, boxes indicate where 50% of the values fall, vertical lines are 95% confidence intervals for each month, and open circles are outliers.

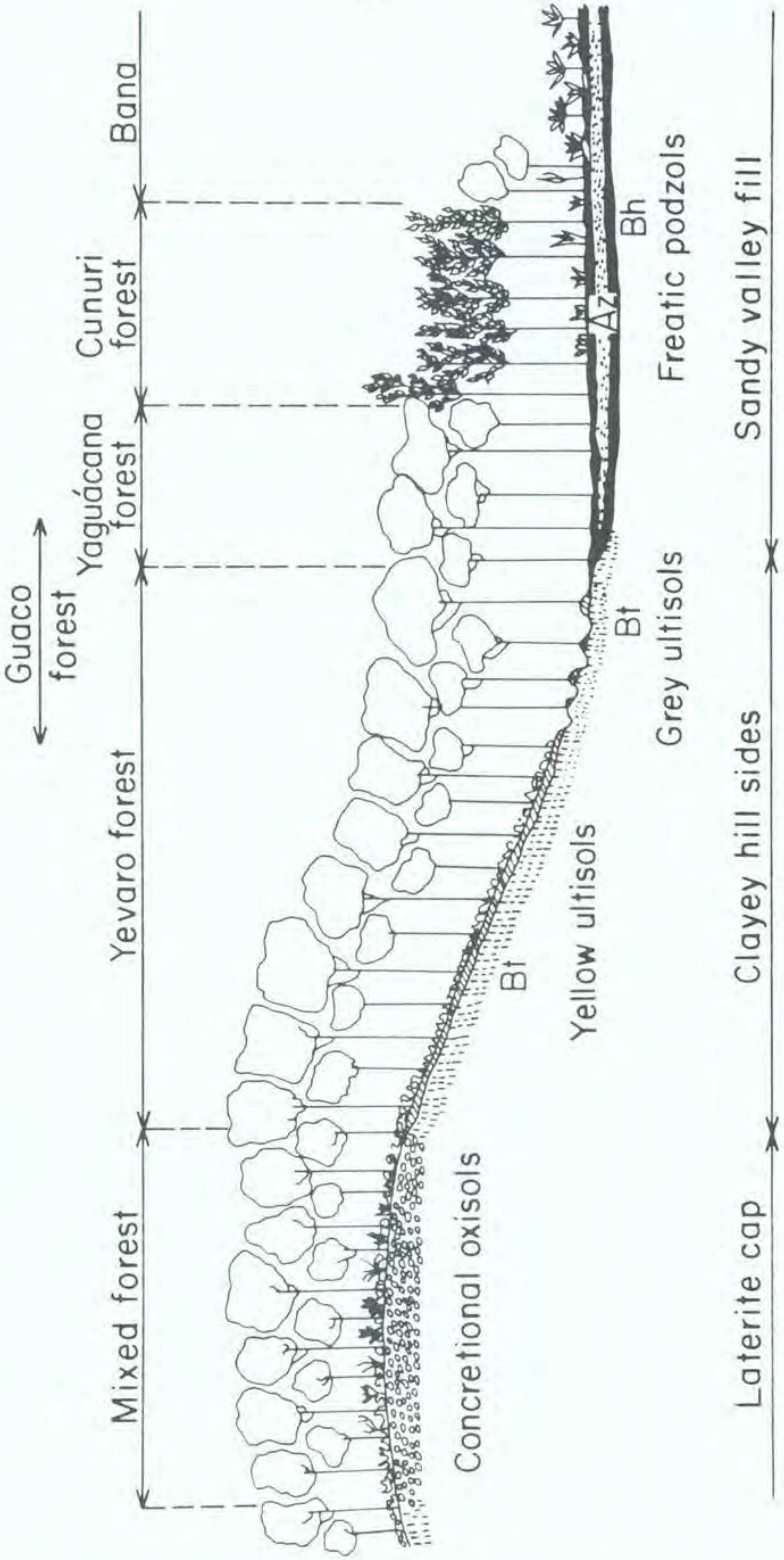

Figure 2. Schematic representation of the different soils, vegetation types and their respective topographical position in the study area, San Carlos de Río Negro, Territorio Federal Amazonas, Venezuela (modified from

Soils and vegetation types in the study area. The different soils found in the area are associated with distinct topographical positions and vegetation types (Fig. 2, modified from Breimer 1985). Basically there are three soil types (Breimer 1985):

a) concretionary oxisols (ferralsols) located on tops of rolling hills covered with mixed forest.

b) Yellow ultisols (acrisols, Paleudults or Tropodults) located on the hill-sides and characterized by a grey sandy topsoil over a yellow clay horizon caused by clay illuviation. These soils support the tallest forests in the area with a species composition dominated by legumes.

c) Ground water spodosols (Tropaquods) found between the rolling hills and constituted by sand probably originated in situ from weathering of local granite (Schnütgen and Bremer 1985). The Amazon Caatinga forest complex occurs on these sandy soils (Klinge et al. 1978).

Mixed forests on oxisols and Yévaro forests on ultisols are never flooded, hence the name of Tierra Firme forests.

There is a strong accumulation of fine roots near the or above the soil surface in the San Carlos forests. Sanford (1985) showed that in the upper 10 cm of soil, including root mat, the amount of fine roots was 18 Mg ha^{-1} in Mixed Forest on Tierra Firme (30% of total root biomass) and 23 Mg ha^{-1} in Tall Amazon Caatinga (38% of total root biomass).

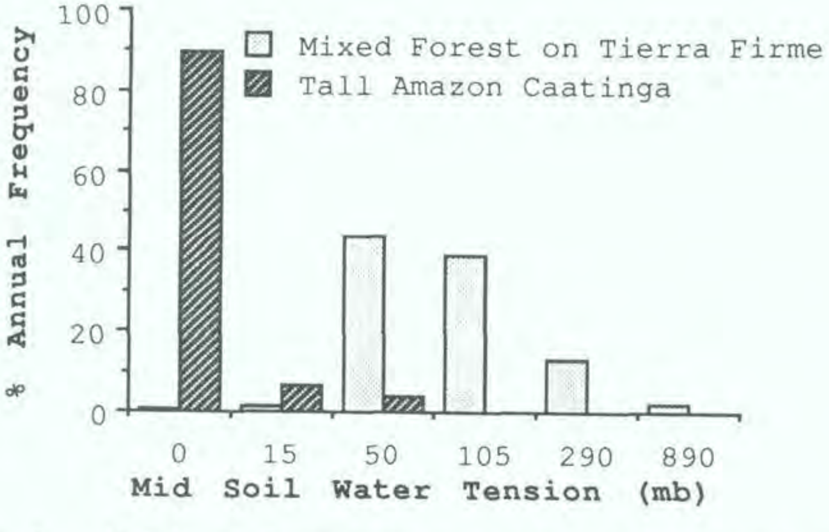

Figure 3. Weighted frequency of soil water tension (mb) under Mixed Forest on Tierra Firme (concretionary oxisol) and Tall Amazon Caatinga (spodosol) in the top 0-75 cm of soil.

The two soil types compared in this paper, concretionary oxisols and spodosols, present contrasting textural and chemical characteristics (Herrera 1977). Concretional oxisols have higher clay content than the spodosols. They are very acidic, specially in the topmost layer, have lower organic carbon and higher N concentrations, and considerably lower available cation concentrations than spodosols. Total phosphorus concentration in the concretionary oxisol is higher than in the spodosols.

Soil Water. Detailed tensiometer studies conducted along the soil-vegetation sequence described above (Dezzeo and Franco, unplublished; Medina and Cuevas 1989) show that the occurrence of different levels of soil saturation is related to topographical position (Fig. 3). Average soil water tension was higher in the concretionary oxisol and decreased towards the spodosols which support the Amazon Caatinga complex. Soils in the Amazon Caatinga were saturated more than 80% of the time. The forest sequence observed in the study area is a product of the interacting soil physico-chemical properties and predominant water regime. Areas in lower topographical positions are less likely to be affected by dry periods and will also receive nutrient enriched leachates from the surrounding areas.

Nitrogen and phosphorus in Mixed Forest on Tierra Firme and Tall Amazon Caatinga

Nitrogen and phosphorus in total biomass. Nutrient inventories in the forests studied show similar N but different P levels (Medina and Cuevas 1989). Nitrogen storage is similar in both forests (1485 and 1145 kg ha^{-1} for Mixed Forest on Tierra Firme and Tall Amazon Caatinga, respectively), while P storage is considerably lower in Mixed Forest on Tierra Firme (48 kg ha^{-1}) than in Tall Amazon Caatinga (101 kg ha^{-1}) in spite of similar total biomass. The relative impoverishment of P in Mixed Forest on Tierra Firme is shown in the N/P ratio of total biomass of 32 as compared to 11 for the Tall Amazon Caatinga. This difference in nutrient inventories is probably a consequence of the relative topographical positions and soil types of the forest series. The chemical characteristics of the organic layer in the soil (0-10 cm depth) reflect these differences (Table 1). Nitrogen concentration under Mixed Forest on Tierra Firme is almost twice that under tall Amazon Caatinga. On the other hand, available P is higher in the spodosol than in the oxisol. In wetland flooded soils, such as the spodosols, enhanced Fe reduction increases phosphate solubility and availability if FeIII(PO$_4$) is present in sufficient amounts (Marschner 1986). This could partially explain the greater amount of P in the living vegetation despite the low total soil P levels. On the other hand, Fe and Al-complexed P, coupled with relatively high soil water tensions in the concretional oxisols, reduce P availability in the Tierra Firme forests (Jordan 1982).

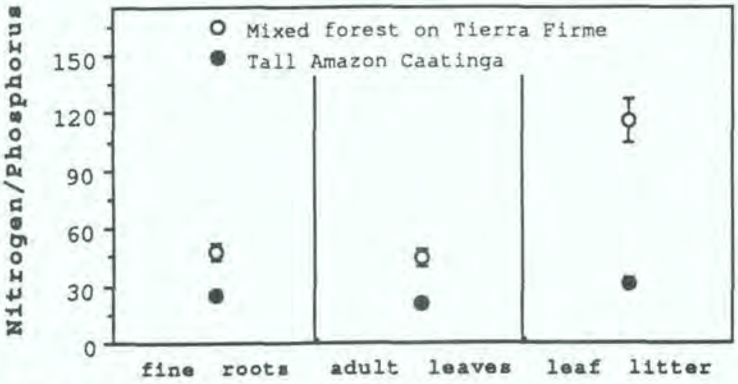

Figure 4. N to P ratios in metabolically active tissues, fine roots and adult leaves, and leaf litter of Mixed Forest on Tierra Firme and Tall Amazon Caatinga. Ratios based on mmol kg^{-1} values.

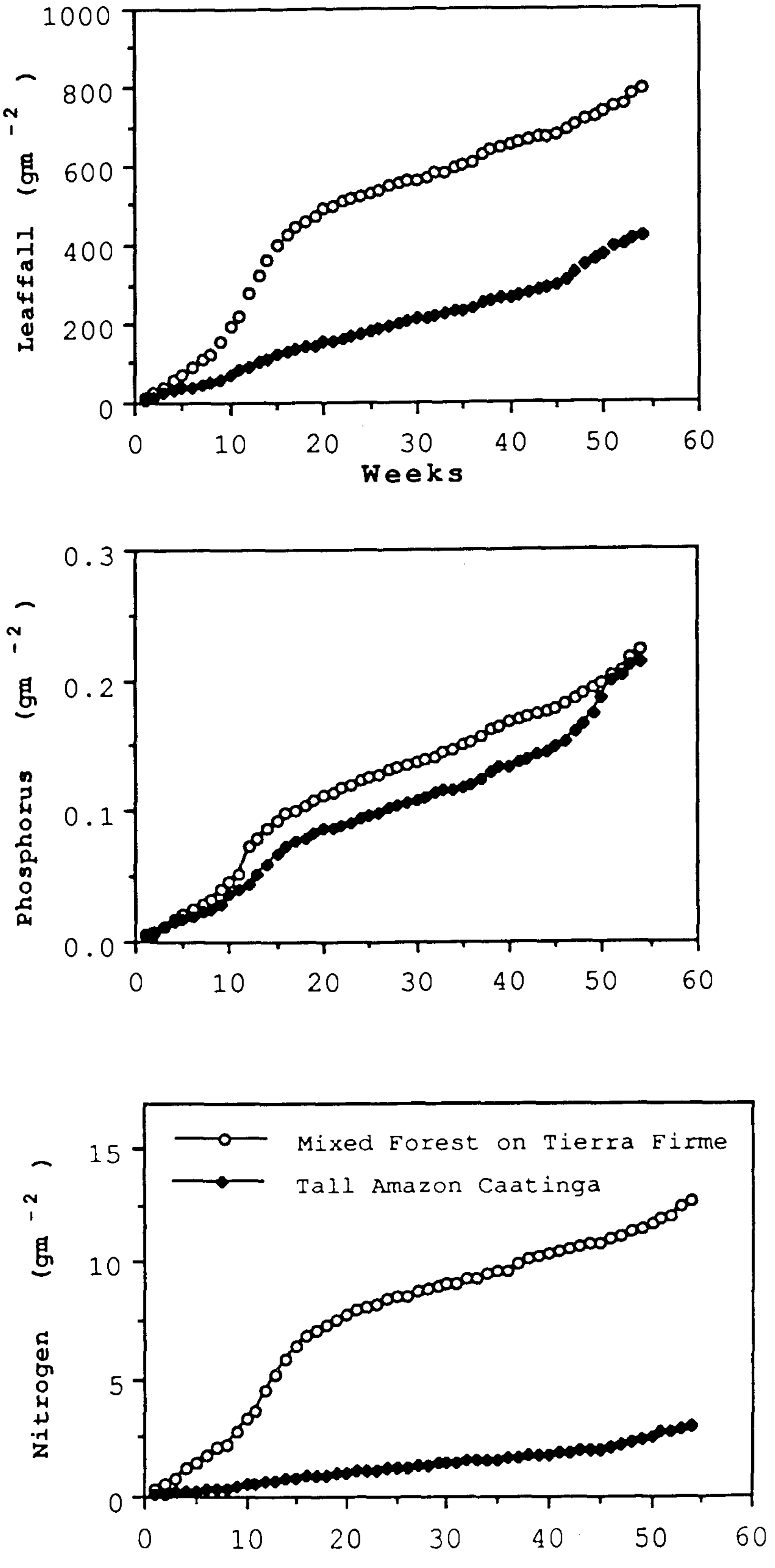

Figure 5. Cummulative nitrogen, phosphorus and mass fluxes in litter fall of Mixed Forest on Tierra Firme and Tall Amazon Caatinga.

N and P concentrations in living tissue and litterfall. In metabolically active tissues such as leaves and fine roots, but also in leaf litter, N/P ratios in Mixed Forest on Tierra Firme are always higher than in the Tall Amazon Caatinga forest (Fig. 4). N/P ratios of fine litterfall increase relative to mature leaves in Mixed Forest on Tierra Firme but remain similar in Tall Amazon Caatinga. There seems to be preferential retranslocation of P in the Mixed Forest on Tierra Firme, while in the Tall Amazon Caatinga, N and P appear to be retranslocated in similar proportions. Sprick (1979), working in the same area, found that 15% of N and 67% of P was retranslocated in leaves of the Mixed Forest on Tierra Firme, while 58% of N and 65% of P was retranslocated from leaves in the Tall Amazon Caatinga.

There are significant differences in N and P fluxes in leaf fall between the two forests (Fig. 5). Nitrogen additions to the forest floor are constantly and considerably higher in the Mixed Forest on Tierra Firme. On the other hand, P fluxes are similar for both forests, although organic mass flux is twice as high in the Mixed Forest on Tierra Firme. When an index of nutrient use efficiency (g of leaf litter produced/ g of nutrient in leaf litter, Vitousek 1984) is estimated (Fig. 6), contrasting patterns are observed. Tall Amazon Caatinga has a higher N use efficiency than Mixed Forest on Tierra Firme, whereas the inverse holds true for P. Again, this reflects the relative impoverishment in available P and N enrichment in the soil and vegetation in the Mixed Forest on Tierra Firme.

Table 1. Chemical characteristics of the organic layer of the soil profile from Tierra Firme forest on oxisol and Tall Amazon Caatinga on spodosol (from Cuevas 1983)

Forest type	pH	organic C	total N	avail. P	K	Ca	Mg
	(1:5 H_2O)	g kg^{-1}	mg kg^{-1}				
Tierra Firme	3.75	120	13600	24.5	145	4	30
Tall Amazon Caatinga	3.50	106	7800	36.8	82	410	182

Nitrogen and phosphorus relationships in decomposing leaves. Rates of organic matter decomposition on the forest floor have been shown to be relatively slow in the San Carlos forests, with turnover times of dominant species being greater than a year (Cuevas and Medina 1988). In general leaves of the Mixed Forest on Tierra Firme decompose faster than leaves from Tall Amazon Caatinga (Table 2), even though the N/P ratios of fine litterfall (Fig. 4) in the latter are more favourable for decomposition (Vogt et al. 1986). Slow decomposition is associated with the predominantly sclerophyllous character of the leaves of these forests (Sobrado and Medina 1980, Medina 1984) and the high lignin/nitrogen ratios (Cuevas and Medina 1988). Waterlogging in Tall Amazon Caatinga also contributes to slower rates of decomposition.

Table 2. Per cent organic matter remaining in decomposition bags at 242 days of leaves from dominant species of Mixed Forest on Tierra Firme and Tall Amazon Caatinga. Values in parenthesis are standard errors of the mean. Different letters indicate significant differences at $p = 0.1$; Gabriel's test (Sokal and Rohlf 1969). Data from Cuevas (1983)

Forest type	n	% mass remaining
Mixed Forest on Tierra Firme	9	28.6 (9.8) a
Tall Amazon Caatinga	9	49.1 (2.3) b

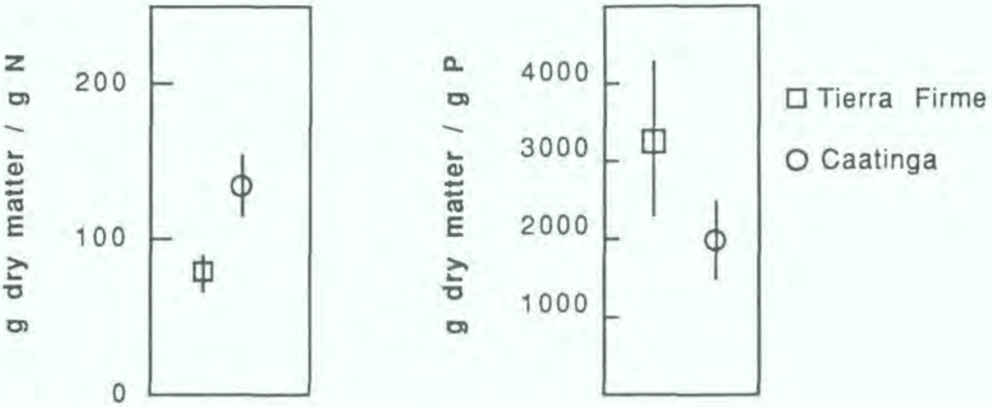

Figure 6. Estimated nutrient use efficiency (g dry matter produced per g nutrient) for leaf ltter from Mixed Forest and Tall Amazon Caatinga

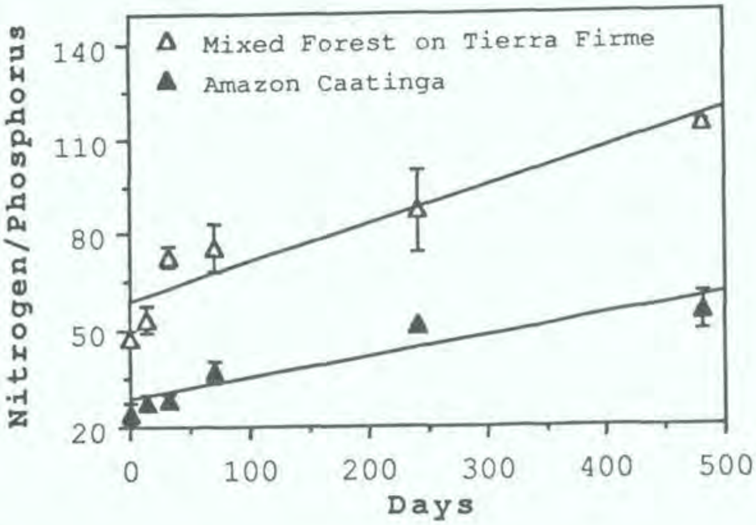

Figure 7. Nitrogen and P dynamics in litter from decomposition bags. Ratios based on mmol kg^{-1} values.

Nitrogen and P dynamics in decomposing litter present a distinct pattern reflected in the N/P ratios of the remaining material (Fig. 7). In both forests there is a strong positive linear correlation with time. Although in both cases N is enriched over time, P tends to be released faster in the Mixed Forest on Tierra Firme, litter than in Caatinga. These relationships reflect different immobilization patterns related to differences in nutrient availability. In Mixed Forest on Tierra Firme organic matter added to the soil is relatively low in P but high in N, whereas in the Tall Amazon Caatinga the opposite holds true. Available nutrients in the soil profile reflect this condition (Table 1). In local fertilization experiments using fine root growth as an index of nutrient deficiency, Cuevas and Medina (1988) found that the root mat and fine roots in the superficial mineral soil layer in the Mixed Forest on Tierra Firme were more responsive to P additions, while in Tall Amazon Caatinga root growth was more responsive to N. In another experiment where fine root adherence was either allowed or prevented there was faster P release from decomposing leaves of the Mixed Forest on Tierra Firme after fine root adherence (Medina and Cuevas 1989). These results strongly suggest that rhizosphere processes might play an important role in coping with limited P availability in the soil. These experiments point again to the different limiting nutritional factors operating in the forest sequence, and corroborate the results obtained in the patterns of nutrient allocation in the two forests.

Conclusions

There is a clear interaction between soil types and topography in determining gradients of soil nutrient availability and water logging regimes. Patterns of nutrient allocation in the vegetation depend on soil nutrient supply and the flooding regime which in turn affect nutrient uptake ability. Different limiting nutritional factors determine the contrasting oligotrophic character of the two forests. Mixed Forest on Tierra Firme is P limited, while Tall Amazon Caatinga is limited by N. Total N concentration in the organic layer of soil under Mixed Forest on Tierra Firme is higher than under Tall Amazon Caatinga. Frequent waterlogging conditions in the Tall Amazon Caatinga might impair microbial activity, thus maintaining the N in an immobilized stage, and reducing mineralization rates. On the other hand, the relatively drier conditions of the soils under Mixed Forest on Tierra Firme could keep P in immobilized as Fe and Al complexes.

Acknowledgements

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PHOSPHORUS DYNAMICS IN A FLOODED SAVANNA

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Introduction

Soils located in tropical and subtropical areas are generally characterized by a strong profile development and a low natural fertility particularly in areas dominated by forest or savanna vegetation (Jordan, 1987, San José and Medina, 1975). Soils covered by savanna ecosystems in Venezuela are characteristically phosphorus deficient, the only exception perhaps is a very small area with nutrient rich Mollisols located in the Western Central Llanos. Landscapes in Higher-Central and Eastern Llanos are mainly covered by well weathered Ultisols and Oxisols (Matheus, 1986; Berroteran, 1988), whereas less developed, although also nutrient limited, Entisols, Alfisols and Vertisols are dominant in Southwestern Lower Llanos (Comerma and Luque, 1971; Schargel and Gonzalez, 1974).

The Southwestern Lower Llanos are seasonally flooded due to the convergence of many river courses, the presence of abundant Vertisols or related badly drained soils, and the extremely flat landscape. An extensive network of dikes have been built in the zone to alleviate the floods and to retain water through the dry period (López-Hernández, 1986). The dikes have allowed an increase in primary production (Garcia, Luis personal communication). The environment for the native flora and fauna, however has been altered (Morales and Pacheco, 1986).

Information concerning phosphorus relationships in Venezuelan savannas is scant and mainly related to fertility studies. For flooded savannas, however, more detailed research is available as a consequence of the convergence of the interest of different ecological research teams. In this review, information will be presented on the reserves, distribution and the main outputs and inputs of phosphorus in a seasonally flooded savanna near Mantecal, Estado Apure.

The area under study is an alluvial flooded savanna between the rivers Arauca and Apure (7°8'N and 68°45'W) affected by the seasonality of precipitation. During the dry season (November to April), the lack of water restricts cattle production which is the main agronomic activity of the region, whereas, continuous rain fall during the wet season (May to October) is responsible for catastrophic floods. In order to alleviate that situation a network of dikes (called modules) were constructed. Each dike regulates through a floodgate, the water level in a small area (3000 ha). Diked savannas maintain a water layer until the end of the dry season. The better hydrological regime has produced a change in the original plant community, and modulated areas are dominated by *Himenachne amplexicaulis*, a species with a higher productivity than those of natural savanna (Bulla et al. 1980).

With the data presently available, it is not possible to produce a complete P cycle, but details of the distribution and main annual transfers in the ecosystem can be provided (Figure 1). The savanna ecosystem studied contains *Leersia hexandra* and *Himenachne amplexicaulis* as the dominant graminoid component, together with *Paratheria*, *Reimarochloa*, *Panicum* and *Luziola* as minor components. In the upper part of the landscape (not flooded) the graminae *Paspalum*, *Axonopus*, *Andropogon*, *Eragrostis*, *Setaria* and *Sporobolus* are common, while *Calopogonium Stylosanthes*, *Alysicarpus*, *Indigofera*, *Macroptilium*, *Centrosema* and *Teramnus* are the main leguminoseae (Ramia, 1972; Tejos, 1987). Soils in the area are acid, with high P-sorption capacities (Schargel and Gonzalez, 1974; López-Hernández et al. 1977).

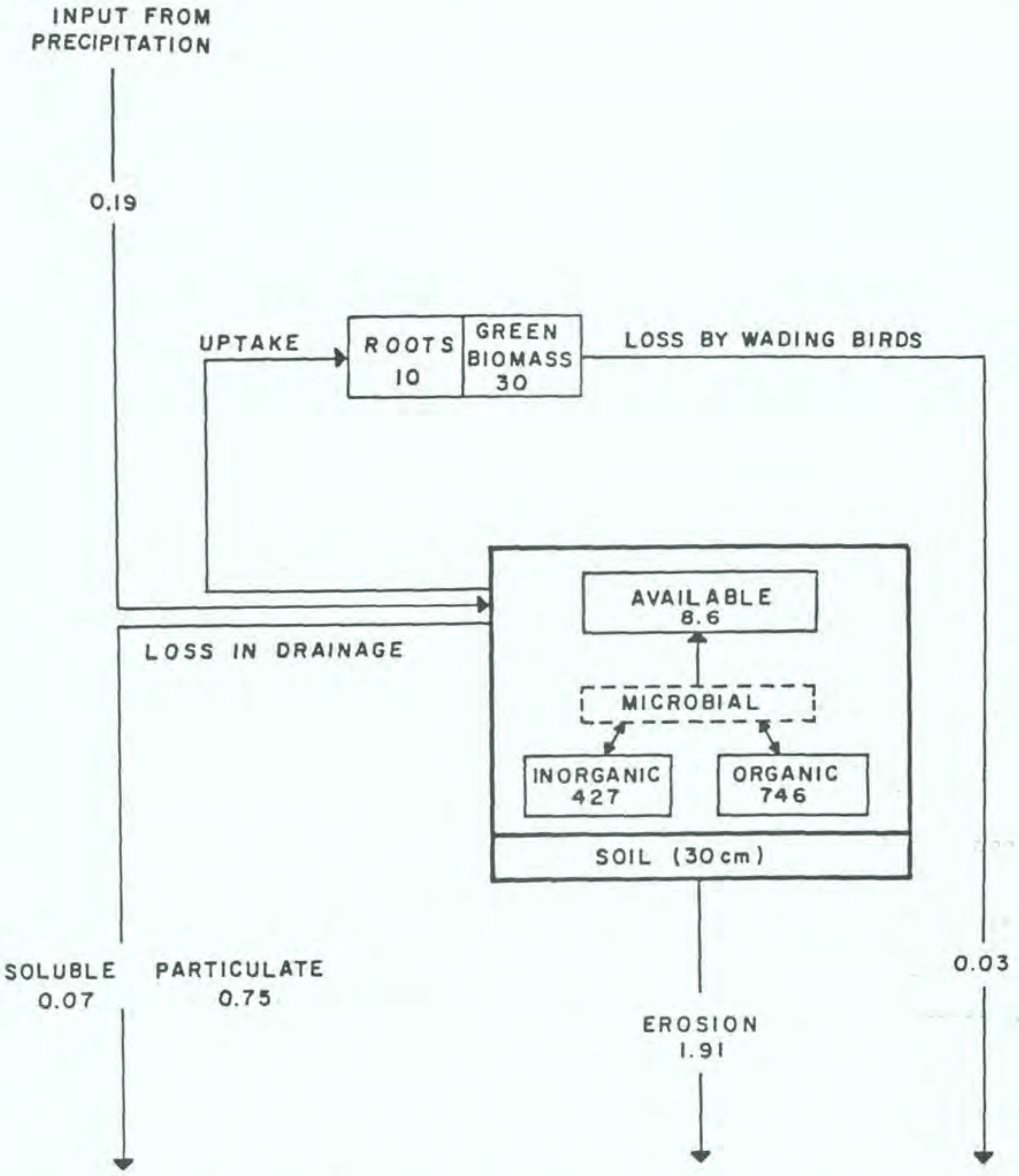

Figure 1. Phosphorus cycle in a flooded savanna (Mantecal, Edo.Apure, Venezuela): contents in Kg ha^{-1} and transfers in Kg ha^{-1} yr^{-1}.

P reserves in soils and vegetation

Soils. Because of the high variability in soil P (López-Hernández, 1987) the total P reserves of Western Venezuelan savannas are not precisely defined. Total soil P for silty loams present in the area range from 165 to 374 µg g^{-1} (Table 1); however López-Hernández et al. (1977) reported a wider range (194 to 1063 µg g^{-1}) for a series of samples which include also some clay soils rich in total P (Vertisols). Soil P reserves to a depth of 30 cm (the zone of highest root activity for tropical grasses) account for 97% of the total P reserve of the ecosystem (Figure 1).

Vegetation. Biomass production in Mantecal savannas has been extensively studied (Bulla et al. 1980; Gonzalez, 1982; Escobar and Gonzalez Jimenez, 1979). Bulla et al. 1980 reported a net primary production of 22 and 12 Mg ha^{-1} for green and root biomass, respectively. An independent measurement done by Tejos (1987) gives an above ground yield of 16 and 10 Mg ha^{-1} for *Leersia hexandra* and *Himenachne amplexicaulis*, the most important grasses in the study area. Average phosphorus content in leaf tissues for both species is around 2.0 mg g^{-1} (Gonzalez, 1982; Tejos, 1987; López-Hernández, et al. 1977), therefore the accumulation of P in the green biomass amounts to between 20 and 40 Kg ha^{-1}. Quantities of P in roots are more difficult to measure, although a conservative value might be approximately 10 Kg ha^{-1} (Figure 1). The large P reserve contained in the vegetation as compared with other savanna ecosystems (Table 2), is related to the high biomass production of this flooded environment.

Table 1. Distribution of various inorganic phosphorus forms extracted by Peterson and Corey (1966) and organic phosphorus by Saunders and Williams (1955) methods in two selected Mantecal profiles (adapted from Espinoza, 1985)

		Inorganic P							
Soil	Depth	Soluble	Al+Fe	Ca	Reduct.	Residual	Sum	Org. P	Total P
	cm	mg kg^{-1}							
Banco	0-10	1	40	3	7	12	63	193	256
	10-45	1	26	2	0	89	119	31	150
Estero	0-10	1	56	4	0	64	125	237	362
	10-25	1	71	4	0	49	125	173	295
	25-60	1	53	7	0	11	72	152	224

The graminoid species present in the area accumulate significant amounts of phosphorus (Gonzalez, 1982; Briceño, 1983; Tejos, 1987). There is little information on phosphorus requirements for the aquatic grasses *L. hexandra* and *H. amplexicaulis* under field conditions. However Briceño (1983) concluded from laboratory experiments that *H. amplexicaulis* might have a critical internal P level above 2.0 mg g^{-1}, and *L. hexandra* around

1.8 mg g^{-1}. The interpretation of those results was complicated because of the difficulties to obtain genetically homogeneous material. Compared with the data obtained for P tissue levels in some other natural and cultivated plants, *L. hexandra* and *H. amplexicaulis* seem to have relatively high P requirements (Table 2). Briceño and López-Hernández (1980) have hypothesized that this could be a consequence of a better P status in the flooded savanna due to the anoxic condition induced by the accumulated water layer. Phosphorus concentrations in the surface water layer as high as 0.32 µg ml^{-1} have been recorded (Briceño, unpublished results 1983), this amount of P in solution is considered enough to supply P requirements in most cultivated species (Fox, 1981; López-Hernández et al. 1987).

Table 2. Above-ground net organic matter production, and phosphorus content in flooded and non-flooded (Cerrado) savannas

Savanna type	Production	P content	Reference(*)
	Mg $ha^{-1}yr^{-1}$	mg g^{-1}	
Flooded Savannas			
Banco	4.3	1.7	1
Bajio	5.5	1.9	1
Estero	9.1	2.1	1
Leersia hexandra dominated	10.0	2.1	2
H. amplexicaulis dominated	15.6	2.5	2
Non flooded- *Trachypogon* savannas			
- cut	2.4	0.6	3
- burned	3.3	0.7	3

(*)[1]Escobar and González Jiménez, 1975
[2]Tejos, 1987
[3]Medina, Mendoza and Montes, 1978

Forms of the P in flooded savanna soils

The ecosystem studied represents a soil catena where the summit is locally called Banco, the middle portion Bajio, and the lower part is known as Estero. Esteros and Bajios are exposed to seasonal floods, while Bancos, are generally never flooded. In the short distances between Banco and Esteros (sometimes less than 200 m), a drastic change in physiography, soil type and vegetation can occur (Morales and Pacheco, 1980; López-Hernández et al. 1986).

The form of P in the soil can also be strongly affected by landscape position. Total P is higher in Esteros than in Bancos (López-Hernández et al. 1977; Espinoza, 1985) (Table 1). Espinoza (1985) presented a Chang and Jackson (1957) fractionation for two soils of Estero and Banco, concluding that the P-Fe and P-Al are the most important P fractions. As expected in this highly weathered soil, soluble-P and P-Ca contents are insignificant. Most of the P is in an unknown non-extractable, non-identified form, as found in other ecosystems by Syers et al. (1967); and Udo and Ogunwale, (1977).

About 70% of total P are present as Po in the surface soils. For many years, researchers have claimed the potential role of organic P in defining P fertility in tropical soils (Bornemisza, 1966, López-Hernández 1977), however there is a lack of experimental evidence.

In the soil from Banco, Pi increased with increasing depth, whereas in the heavy textured Estero soil, Pi is less affected by depth (Table 1). In both soils soluble Pi amounts to a very small fraction of the total P.

Main inputs and outputs in savannas

Inputs of P. Mantecal savannas are located far from important industrial and agricultural activities, therefore rain fall is the main input of P to the ecosystem. Yanes (1983) and Garcia (1985) have measured less than 0.5 Kg P ha^{-1} yr^{-1} precipitation inputs (Table 3) during the rainy season. Montes and San José (1987) have reported similar results for Calabozo savannas located 200 Km from Mantecal. Important loads of P from atmospheric deposition can occur in agroecosystems where burning is practiced (Sequera et al., this volume).

Bird droppings may also be an important input to Mantecal savannas because the dikes serve as a natural refuge for abundant savanna birds, especially during the dry season. Birds, like other consumers, act as accelerators of nutrient cycling through food consumption and faeces deposition within the ecosystem. Morales and Pacheco (1986) have studied the flow of some nutrients through wading birds in Mantecal, and presented information on internal cycling and nutrient export, emphasizing the probable role of the avian fauna in the overall nutrient-impoverishment of the savannas enclosed by dikes; more detail about this particular subject shall be discussed when analyzing phosphorus losses.

The release of P from parent material by weathering is difficult to ascertain. Sediments in the area are previously weathered (Berroteran et al., 1982), and the release of P from preweathered sediments containing secondary Al and Fe phosphate minerals is likely to be low.

Losses of P. Lateral drainage of soluble forms of P was recorded by Sosa (1982) and Garcia (1985) but P losses can be considered negligible (Table 3), the result is not surprising because of the high P sorption capacity of the soil under study (López-Hernández et al., 1977, 1979). Losses of soluble Po were higher than Pi losses (Table 3). Phosphate as well other mineral losses through internal drainage are likely to be minimal in flooded savannas because of the abundance of Vertisols in the area. Hydrological budgets in Mantecal savannas indicate that water percolation in the area is insignificant. Although the flooded savanna presents an almost flat topography, major losses of phosphorus in particulate material occurred (Table 4). Guzman in the same area (unpublished results) has found a similar result for Zn and Cu. Consequently, although the drainage waters are poor in the soluble forms of those elements, particulate materials in suspension can carry important quantities (Figure 1).

Table 3. Monthly phosphorus input by precipitation and losses by lateral drainage (kg ha^{-1}) in the experimental site, Mantecal

	Input	Losses	
		Soluble	Particulate (organic)
May	0.031	0.01	0.01
June	0.042	0.01	0.42
July	0.040	0.01	0.14
August	0.014	0.01	0.08
September	0.017	0.01	0.09
October	0.022	0.02	0.01
November	0.027	-	-
Total	0.193	0.07	0.75

Morales and Pacheco (1986) found that important amounts of N and P can be exported from modules to nesting sites by wading birds. They found an export of 0.03 Kg P ha^{-1} yr^{-1}, which is an important flow, if we take into account the minor amounts of P which enter and leave the system (Table 3).

Table 4. Budgets for P in a flooded savanna. Data in Kg ha^{-1} yr^{-1}

Input		0.193
Output		
	soluble	-0.07
	export by wading birds	-0.03
	organic particulate	-0.72
Total input-output		-0.627
Loss by erosion		-1.91

Phosphorus budget. Gains and losses of P in different ecosystems tend to have very similar values of below 1 Kg ha^{-1} yr^{-1}. Consequently the budget is near zero, or in some cases, there is a small positive budget indicating P accumulation within the system (Likens et al. 1977, Harrison, 1988). Table 4 presents the balance of P for the Mantecal experimental area. A budget based only on soluble P data in precipitation and drainage waters is positive. Corrected values using total P (Pi + Po), and P losses by wading birds generate a negative budget of -0.63 Kg ha^{-1} yr^{-1}. Important losses of P occurring as particulate material (69% of total) give a significant negative budget. Likens et al. 1977 found erosion losses amounting to 63% of totals for a temperate forest ecosystem. The effects of erosion losses on P concentration in an ecosystem cannot be easily evaluated since total soil material is exported.

Internal P cycling in flooded savannas

The information on this subject is far from complete, nonetheless some cycling processes of P in flooded savannas are presented here.

P uptake. Phosphorus uptake by savanna grasses can approximately be calculated from biomass production and P tissue contents. Data is given for *H. amplexicaulis* and *L. hexandra* the dominant grasses in diked savannas. Phosphorus content in tissues does not vary significantly within the wet or dry seasons (Tejos, 1987), but is lower in the dry season than during the flooded period (Tejos, 1987). This may be explained with a higher P availability when the soil is wet (Ponnamperuma 1972). The relatively large amount of P immobilized by Mantecal's natural grasses (Table 2) is a consequence of the higher biomass production in flooded than in Cerrado savannas (Medina, 1980). Tejos (1987), reported a response of native pastures to increasing level of P fertilizer, those results clearly indicate: i) the existence of low available P in the soil system and ii) that the P requirements of the natural species surpass the P level of the soil solution.

Decomposition and P release. P recycling in flooded savannas is mainly governed by the P forms in foliar litter. The high percentage of P in both dead and green foliar tissues indicate that it should be the main source for P mineralization processes. A high rate of annual P uptake, and an important annual biomass turnover in savannas suggest that the P retained in biomass is quickly released to the ecosystem. The rate of decomposition of litter is a function of i) climate, ii) soil microbial activity and iii) litter quality (Swift et al. 1977; Harrison, 1989). The high temperature of the zone (mean annual temperature above 25°C) allows for a high rate of decomposition of the accumulated litter with the first rains (Bastardo, 1981). Bastardo, (personal communication) found that organic P forms in *H. amplexicaulis* can disappear within 6 months (95% loss), whereas Po from *L. hexandra* is mineralized at a lower rate (95% P loss after 9 months). No changes in decomposition processes have been recorded due to the altered water regime of diked areas. Bulla et al. (1980) when studying the production and decomposition of biomass in Mantecal savannas, have not seen significant accumulations of litter within the dikes.

P recirculation by termites

In Venezuelan savannas, termites are important in controlling both the amount and distribution of phosphorus in soils (López-Hernández et al., 1989 a,b). Mounds of *Nasutitermes ephratae*, a common termite species abundant in seasonally flooded and dry savannas, show both higher contents of available forms of P, especially water-soluble P, and a high organic matter content (Table 5). Mounts however did not display higher phosphatase activity. The elevated organic carbon content, and the high level of soluble P, together with very high C/Po ratios found in mounds probably inhibited phosphatase activity (López-Hernández et al., 1989 a). Phosphorus sorption measured by isotopic dilution in mounds was only one tenth of that in surrounding soils (López-Hernández et al. 1989 b).

Table 5. Organic carbon, C/Po, phosphatase activity (measured with paranitro phenol), phosphorus forms, P remaining in solution in mounds and adjacent soils (Adapted from López-Hernández et al. 1989b)

	C	C/Po	Phosphatase Activity	Total P	Pi	Po
	mg g^{-1}		µmol g^{-1}h^{-1}	µg g^{-1}		
Mound	93.3	742	1.18	397	224	172
Soil	21.3	212	1.13	162	49	113

Wood et al., (1983) related the increase in available P in mounds of humus-eating African termites to the effect of the high pH in the anterior hind-gut of soil-feeders on ingested soil organic matter. López-Hernández et al. (1989 b) have attributed the significantly greater amounts of both total and available phosphorus in mounds than in adjacent soils to a direct competition between organic molecules (abundant in the *Nasutitermes* mounds) and orthophophate for sorbing sites. They concluded that the abundant *Nasutitermes* could play an important role in controlling P availability and P redistribution in tropical South American savannas, which are typically P-deficient.

Conclusions

P input with rain fall in Venezuelan savannas is low, consequently it does not play a significant role in P cycling. Because of the strong P sorption capacities of savanna soils, insignificant amounts of soluble P forms are leached throughout the ecosystem. Significant amounts of P, however are lost as particulate inorganic forms. Additional P losses are due to wading birds nesting outside the area. There is a lack of information concerning internal P cycling in Venezuelan savannas. P uptake by grasses is high because of the high rate of biomass production (20 µg $^{-1}$ ha^{-1} yr^{-1}) together with a relatively high P content in foliar tissues (2 - 3 µg g^{-1}). The P accumulated within the plants should be mineralized in order to provide enough Pi for the next growth cycle. We now know that an important amount of P in the ecosystem is the organic soil P reserve, the processes of mineralization and microbial biomass turnover need to be studied.

Acknowledgements

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NOTE ON THE DYNAMICS OF P IN DECOMPOSITION OF TROPICAL GRASSES

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This contribution examines the fate of grass biomass and its P content during the rainy season of the flooded savannas described by Lopez-Hernandez (this volume). Given the soils of relatively low fertility, the main source of nutrients for primary production is the decomposition of grasses. Part of the plant biomass is utilized by herbivores, but the majority of it is deposited on the soil as plant litter. Among the grasses of the flooded savannas, there are three main species contributing to the nutrient budget: *Panicum laxum,* Swartz (contributing about 325 g m^{-2}); *Hymenachne amplexicaulis*, Rudge (1400 g m^{-2}) and *Leersia hexandra*, Swartz (375 g m^{-2}). Eighty nine percent of all herbaceous litter was due to these three grasses, with at least 50 % by *H. amplexicaulis* (Bastardo 1981).

The disappearance of P during the decomposition of *P. laxum; H. amplexicaulis;* and *L. hexandra* was analyzed. Plant material was collected from an area where each grass was predominant during the rainy season, except *L. hexandra* which was collected only at the end of the rainy period. Between 40 and 50 g of fresh plant litter was placed in decomposition net bags of 0.1 cm mesh size. Twenty bags for each grass species and physiographic zone were used and 5 replicates were taken for analysis after 16, 32, 64 and 128 days. Dry weight and total P content were determined.

The disappearance of dry weight and total P was fitted to a negative exponential equation.

$$Wt = Wo \, . \, e^{-kt}$$

Where Wt= dry weight at time t; Wo= initial dry weight; k= rate constant per day; t= time. This model assumes the fractional rate of decay to be constant. In all cases the field data approximated the exponential model. Therefore, the slopes of the regression lines (k) define the relative decomposition rates.

Decomposition was highest during the beginning and the end of the rainy season. At the beginning of the rainy season, the plant material is left from a very dry period during which it remained undecomposed as plant litter. Upon the onset of decomposition with the first rains, a rapid release of organic and inorganic soluble compounds occurs (Bastardo and Rivera 1986). Towards the end of rainy season grasses begin to senesce with concommittant decomposition of litter and a release of nutrients.

The analysis of *H. amplexicaulis* shows that during the beginning of the rainy season the disappearance of P is much lower in the bajío bajo than in the estero, whereas at the end of the rains P release is very similar in both physiographic zones (Table 1). Phosphorus release from *P. laxum* is very rapid during the beginning of rains in the *bajío alto;* with a 92 % loss after 128 days. On the drier *banco* 17 % of P were lost after 128 days. *L. hexandra* was analyzed only at the end of the rainy season. The contribution of P from the litter to the soil was higher in the *estero* than in the *bajío bajo* during the experimental period, with a P loss of 38 % in 16 days, while in the estero 55 % were lost.

H. amplexicaulis releases P most rapidly during both periods and in all physiographic zones. The rate of disappearance is followed by *L. hexandra* and *P. laxum* (Table 1). *H. amplexicaulis* has the most rapid rate of P loss in the *estero* during both periods. In the case of *P. laxum* P disappears rapidly only at the beginning of the rainy season in the *bajío alto*.

Studies by Kjoller and Struwe (1982) and Bastardo and Rivera (1986) showed that the quality of the substrate and physico-chemical environment conditions play an important role in the growth of fungi and bacterial populations during the colonization of substrate in natural conditions. According to the results obtained in the present studies we can suggest that in the savannas examined, it is very important to understand how management of the ecological factors can control nutrient availability in the production processes.

Table 1. Statistical parameters for disappearance of P from three tropical grasses. k is the rate constant per day; r is the correlation coefficient of the regression (significant level at p=0.05).

Species	Location	Beginning of rains				End of rains			
		k ($days^{-1}$)	r	50% loss (days)	95% loss (days)	k ($days^{-1}$)	r	50% loss (days)	95% loss (days)
P.laxum	banco	-0.0110	0.83	32	241	-0.0072	0.88	65	384
	bajío alto	-0.0182	0.98	27	154	-0.0075	0.92	66	372
H.amplex.	bajío bajo	-0.0075	0.84	58	364	-0.0124	0.81	7	193
	estero	-0.0160	0.87	5	149	-0.135	0.81	3	167
L.hexandra	bajío bajo	-	-	-	-	-0.0100	0.92	39	269
	estero	-	-	-	-	-0.0144	0.77	8	168

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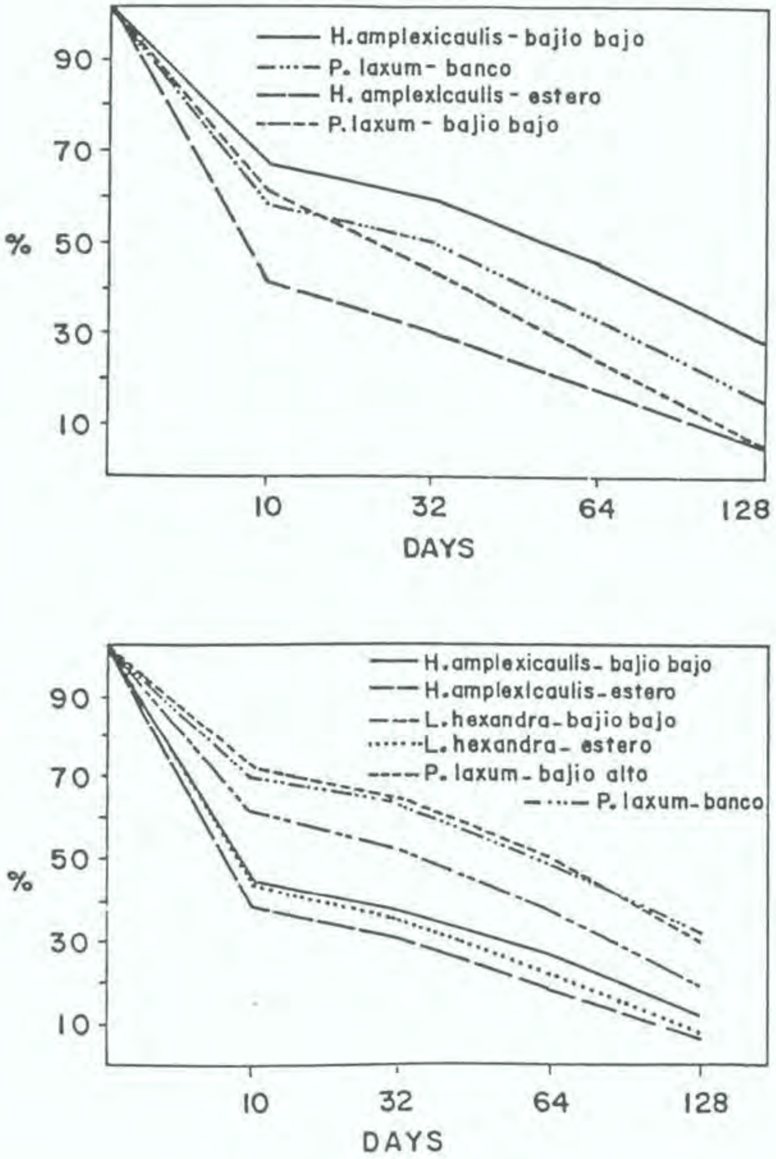

Figure 1. Disappearance of P during the decomposition of three species of tropical grasses at the beginning (top) and end (bottom) of the rainy season.

MECHANISMS CONTROLLING PHOSPHORUS RETENTION IN THE LITTER MAT OF ATLANTIC COASTAL FORESTS

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Introduction

The main natural ecosystem in most of the coastal region of Brazil is the Atlantic Coastal Forest - a tropical rain forest. Most of it has been deforested and replaced by sugar cane plantations. However, some areas of the original forests, with varying degrees of perturbation, still remain. Extremely limited information related to nutrient cycling in this ecosystem is available (Sampaio et al., 1988).

A characteristic of tropical rain forest soils is the presence of a thick organic (O2) horizon permeated by roots (Klinge, 1973) that are often heavily infected by vesicular-arbuscular mycorrhizae (VAM) (Went and Stark, 1968). This root-litter mat is purported to be important for nutrient retention (Stark and Jordan, 1978). In a well conserved Atlantic Coastal forest site, near Recife-PE, Brazil, Sampaio et al. (1988) measured 15.7 $kg.ha^{-1}$ of P held in root-litter mat while annual inputs of P through litterfall were 3.4 $kg\ ha^{-1}\ yr^{-1}$. A tight P cycling has been proposed for forest ecosystems (Vitousek, 1984). Thus, annual P uptake should be similar to reported annual inputs. Double acid (Mehlich) extractable-P in the mineral soil at the Recife forest is extremely low (less than 0.5 $mg.kg^{-1}$ of soil, unpublished results), suggesting that decomposing litter supplies most of the P demand by the vegetation.

It has been hypothesized that P is supplied by direct root uptake from decaying leaves (Went and Stark, 1968), through a VAM mediated mechanism (Herrera et al., 1978). The lack of movement of ^{32}P from the humus into the underlying mineral soil of an Oxisol (Stark and Jordan, 1978) further supports the hypothesis of direct P cycling. In this way P is kept out of reach of the soil mineral phase that, through fixation processes, would probably render it less available than in the humus layer.

Stark and Jordan (1978) suggested three mechanisms for P retention in the root-litter mat: mycorrhizal uptake, microbial uptake and sorption onto organic matter surfaces. Although evidence of the role of VAM in P uptake has been obtained (Herrera, et al., 1978), the role of the two remaining mechanisms is yet to be substantiated. Furthermore, the relative importance of these mechanisms in P retention in the forest floor remains unknown. In this work we will summarize results from several research efforts that focus on the main mechanisms that control P retention in the root-litter mat of Atlantic Coastal-Forest areas in Pernambuco.

Preliminary survey of forest floor

The abundance of roots in the Atlantic Coastal forest floor was assessed by sampling the O1+O2 horizons and underlying mineral soil of eleven forest sites (Freitas et al., 1989). These sites were distributed within the coastal fringe of Pernambuco, Oxisols and Ultisols being the predominant soil types. Sample points within transects in each forest were 100 m apart and varied between 12 and 20 points per transect. At each point a metal cylinder (25 cm long and 15 cm diameter) was pounded into the forest floor and mineral soil underneath. The cylinder contents were separated into undecomposed litter, humus+fragments, mineral soil and roots in each layer.

Results

Some of the fractions separated in each cylinder are summarized in Table 1. Total dry matter in the O1+O2 horizons varied from 6.6 to 21.7 $t.ha^{-1}$ among forests and did not correlate significantly with the amounts of roots present in the mat or with total organic C in the mineral soil. The presence of roots intermixed with this material was conspicuous and represented, in some of the forest sites, up to 10% of the total dry matter weight of the forest floor. Cylinder penetration into the mineral soil underneath the forest floor varied from 4.2 to 6.9 cm so root weights in the mineral soil were normalized linearly to an average 5.3 cm of mineral soil depth. The weight ratio of roots in the humus and mineral soil varied from 2 to 10.

Table 1. Weights of roots and litter in the O horizon and mineral soil of eleven tropical Atlantic Coastal Forest sites in Pernambuco-Brazil

	Roots		Litter		
Sites	O horizon	Minimum soil	O horizon	Minimum soil	Total C
	$Mg.ha^{-1}$				%
1	1.48	2.83	11.46	3.88	1.36
2	0.52	1.41	9.64	0.82	1.15
3	0.92	4.39	15.64	2.03	1.11
4	1.58	2.42	20.14	2.87	1.51
5	1.19	3.82	14.01	3.27	3.39
6	0.50	5.76	6.93	1.81	2.45
7	0.45	4.40	10.61	1.80	2.49
8	0.57	2.63	10.38	2.20	2.86
9	0.25	2.11	10.34	3.77	2.60
10	0.28	2.35	6.39	1.31	2.92
11	0.72	2.95	12.95	1.87	1.76

In addition to the roots, considerable amounts of other undecomposed organic materials of plant origin were separated from the mineral soil (Table 1). In some sites, the weight of these undecomposed materials is even larger than the weight of mineral-soil roots. The large value of dry matter in the forest floor of site 4 (Dois Irmaos) substantiates earlier findings of a three-year study at this site (Sampaio et al., 1988).

First field experiment

The objectives of this experiment were to: 1) quantify the degree of VAM infection of roots in the humus and mineral soil; 2) compare P retentivity and availability between humus and mineral soil and, 3) determine the relative importance of root uptake, microbial uptake and sorption reactions in P retention in both layers. We carried out the following experiment (Elliott et al., submitted): 10 metal cylinders (25 cm long by 15 cm diameter) were pounded into the forest soil at site 1 (Table 1) followed by the injection of a 10^{-6} M solution of P, labelled with ^{32}P, at two depths. In five cylinders (serving as blocks) undecomposed leaves were carefully lifted and five one-ml aliquots injected approximately at the midpoint of the O2 layer, within a five-cm diameter plastic ring. In the five remaining cylinders the labelled solution was injected 5 cm below the mineral soil surface, by means of a specially constructed 20 cm-long stainless steel canula. The same injection procedure was repeated 2 m away from the previous ones, but without using cylinders. We designated these as trenched and untrenched treatments, respectively. Only a 5-cm diameter plastic ring was used to guide the needle and left in the field to mark the injection point. The same experiment using only one depth of injection (7 cm into the mineral soil) was repeated in a neighboring sugar-cane field in its first year of cropping after slashing and burning the forest. Two weeks after injection cylinders were aligned with the plastic rings left at the untrenched sites and pounded into the soil to a 20 cm depth. All cylinders were excavated with a shovel. Samples from the soil below all cylinders and from roots in the O2 and A1 horizon in the vicinity of untrenched sites were also taken. During this period rainfall was 203 mm.

The cylinder contents of the forest sites were separated into five components: 1) intact leaves and large fragments (O1 horizon) free of attached roots; b) humus plus small fragments and undifferentiable organic matter (O2 horizon); c) roots in O2; d) roots in the mineral soil and e) mineral soil. From the sugar cane sites, which had no litter layer, only separates d) and e) were obtained. After sample preparation and digestion (for details see Elliott et al., submitted) total ^{32}P and ^{31}P were determined in all separates (by Cerenkov counting and colorimetry, respectively). Roots were analysed for VAM infection (Rose et al., 1987a,b). The humus material was cut with scissors to pieces of about 2-3 mm and mixed. Three ^{32}P and ^{31}P pools were determined in this material and in the mineral soil: a) resin-P (2 g of material: 40 ml water); b) bicarbonate-P (5 g of material: 100 ml of 0.5 M $NaHCO_3$ and digestion of the extract for total P determination); c) microbial biomass-P: one-hour fumigation of 5 g of material with $CHCl_3$ vapors followed by $NaHCO_3$ extraction. The difference in P concentrations between fumigated and non-fumigated bicarbonate extracts is designated as P-flush. This P-quantity is an unknown proportion of total microbial-P since the efficiency of extraction of microbial biomass by the bicarbonate was not determined.

Results

Average concentrations of total P in the litter (O1), humus (O2) and mineral soil were 488, 647 and 83 mg.kg^{-1} of dry matter. The result for humus is expressed on an ash free basis to compensate for variable amounts of sand included in this layer, from the humus-mineral soil interface. Total C and N contents in humus were 28 2% and 0.113% and in the underlying mineral soil 0.34% and 0.019%.

Recovery of ^{32}P added. When the tracer was added to the humus of the trenched sites, 20% was recovered in the humus and 61% in the mineral soil, yielding 81% of ^{32}P recovery per cylinder (not shown). The downward movement of ^{32}P from the humus into the mineral soil contradicts results from Stark and Jordan (1978). They did not detect significant amounts of radioactivity in the water percolating the root-litter mat.

When the tracer was added to the mineral soil, total recoveries were much higher (93% and 101% for the forest and sugar-cane mineral soils, respectively). In a separate one-day experiment with heavy raining, losses of ^{32}P injected in the humus were 37%; when injected in the mineral soils they were 10 and 20% for the forest and sugar-cane, respectively (Elliott et al., submitted). Since we did not detect radioactivity in the mineral soil below the cylinder, no clear explanation can be given for these ^{32}P losses. Additions of tracer in the mineral soil yielded consistent higher recoveries than when injected in the humus. Elliot et al., (submitted) suggested that colloidal organic complexes with P could leach out to deeper soil layers. If uncharged, they would not be sorbed by the mineral soil surfaces.

VAM infection and P concentrations in roots. The humus layer had higher amounts of fine roots than the mineral soil and the VAM % infectivity in the humus roots was twice that in the forest or sugar-cane mineral soils (Table 2). These two factors contribute to a larger absorbing surface and help explain the higher concentration of ^{32}P in humus roots relative to those in the mineral soils (Table 2). However, mineral roots in the forest retained twice as much ^{32}P per cylinder than those in the humus. These results contradict our original hypothesis about relative availability of P in the forest humus and mineral-soil layers.

Effect of trenching. Trenching reduced the concentration of ^{31}P in humus roots by 25% but did not affect the roots in the mineral soils (Table 2). For tracer data trenching had a marked effect only in forest mineral-soil roots. Coincidently, these roots, when sampled from outside the cylinder area in the untrenched treatment, had the highest degree of labelling (Table 2). This suggests that these superficial roots in the mineral soil are mostly horizontally oriented and active in P translocation to the plants. The absence of a trenching effect on tracer data for roots in the humus layer suggests mostly vertically oriented short-range feeder roots growing from the deeper ones in the mineral soil.

Extractable and microbial-P. Trenching had a significant effect only upon resin extractable-P (Elliott et al., submitted). Therefore, to simplify the discussion, in Table 3 we averaged results of trenched and untrenched treatments. Resin extractable ^{31}P was very low in the humus and mineral-soil fractions. Total HCO_3- extractable ^{31}P was more than 10x greater in the humus than in the mineral soils. Differences between humus and mineral soils become even larger (100x) for the effect of fumigation (flush) - the increase in HCO_3^- extractable P after the $CHCl_3$ treatment. In the humus this flush extracted 110.2 mgP kg^{-1} of ash-free humus (an unknown proportion of total biomass-P) constituting almost 25% of total ^{31}P in the humus.

Table 2. Relative proportion of fine roots, mycorrhizal infection and ^{31}P and ^{32}P concentrations in root dry matter. Values shown are from layers where ^{32}P was injected (5 replicates)

	Position		
Variables	Forest humus	Forest mineral	Sugar cane
Roots <1 mm (% by weight)	93	52	80
VAM infection (%)	46	23	20
^{31}P concentration (mg kg^{-1})			
a) Untrenched	936a[1]	410a	360a
b) Trenched	676b	439a	298a
^{32}P per cylinder (cpm*10^{-3})	958a	1758a	354a
a) Untrenched	791a	464a	384
^{32}P in roots from outside the cylinders (min. soil) (cpm*10^{-6} kg^{-1})	11	447	27

[1] Means followed by the same letter within each position are not statistically different at the 5% level of probability (Tukey).

The interpretation of the tracer data is made more difficult by the different weights of humus and mineral fractions (average of 35 and 2,000 g per cylinder, respectively). Since both layers received the same amount of tracer, the concentration of ^{32}P in the humus is 60 times higher than in the mineral soil. This should reflect to some extent upon ^{32}P extracted per unit weight.

As expected, HCO_3^- and $CHCl_3$-flush extractable^{-32}P are about 30x larger in the humus than in the mineral soil. However, resin extractable^{-32}P concentrations in these two soil fractions are very similar (Table 3). The lack of extractability of ^{32}P in the humus by the resin was also observed in a one-day field experiment (Elliott et al., submitted). Thus, we concluded that most of the ^{32}P added to the humus fraction was lost from solution by fast mechanisms that convert it into non-exchangeable forms.

The high ^{32}P concentrations in the $CHCl_3$-flush (Table 3) indicate that one of the mechanisms is uptake by microbial biomass. The other mechanism is related to the HCO_3-extractable-P pool, that also contained large amounts of ^{32}P. The nature of this pool is still unclear. Our hypothesis is that HCO_3- extractable-P is related to Fe-Al-organic matter complexes. These metals bridge P and fluvic acids but bonds are hydrolyzed by mild alkalis (Schnitzer and Khan, 1972).

Table 3. Concentrations of extractable ^{31}P and ^{32}P averaged over untrenched and trenched treatments. Values are only from layers where 32P was injected (5 replicates)

Extraction	Forest humus	Forest mineral	Sugar cane
		^{31}P (m kg^{-1} D.M.)	
Resin	0.35	0.30	1.43
HCO_3^-	44.0	1.59	3.41
$CHCl_3$-flush	110.2	1.43	1.63
		^{32}P (cpm*10^{-6} kg^{-1} D.M.)	
Resin	2.41	3.01	4.52
HCO_3^-	252.0	7.57	9.11
$CHCl_3$-flush	211.2	9.76	6.58
		^{32}P (cpm*10^{-3} $cylinder^{-1}$)	
Resin	86	6,052	8,981
HCO_3^-	4,547	15,202	18,098
$CHCl_3$-flush	7,571	19,598	13,064

The ^{32}P extracted by HCO_3- and $CHCl_3$-flush on a per cylinder basis is larger in the mineral soils than in the humus (Table 3). The forest mineral soil has 30% more ^{32}P in the biomass than the sugar-cane soil, for comparable biomass-^{31}P sizes. This suggests higher biomass activity in the forest mineral soil, probably due to higher C inputs that leach out from the root-litter mat.

Results discussed so far give preliminary evidence to the importance of biomass activity upon P retention in the root-litter mat, as hypothesized by Stark and Jordan (1978). The role and relative importance of VAM and of sorption reactions is still inconclusive.

Laboratory experiment

In the following laboratory experiment we tried to further substantiate the importance of the microbial biomass in P retention (Salcedo et al., submitted). At the same time we tested a modified methodology to estimate microbial-P, since we thought the combined HCO_3-

$CHCl_3$-flush treatment was inappropriate for this purpose. The HCO_3-extractable-P in unfumigated samples was high and could include an unknown quantity of microbial-P. To avoid the high background values we tried the $CHCl_3$-flush of water extractable-P instead. Low resin-P values (Table 3) indicated that unfumigated samples would yield low values of water extractable-P (Pw), assuring that no microbial-P is included.

We collected a composite humus sample from the same location of the field experiment and immediately took it to the lab. Larger roots were separated and the humus and tissue fragments were cut with scissors and homogenized. Ten g of field moist humus were immediately weighted into 15 plastic centrifuge bottles (250 ml capacity) and divided into three treatments: 1) five bottles received a 1-ml aliquot of a 10^{-7} M solution of NaH_2PO_4 labelled with ^{32}P and were incubated in the laboratory for 24 h. After adding 100 ml of deionized water the bottles were shaken for one hour, centrifuged and the supernatant separated by filtration (membrane with 0.47 µm pore-diameter). The bottles were weighed for carryover corrections and the residue extracted again with 100 ml of 0.5 M $NaHCO_3$ for 16 h. The supernatant was separated through centrifugation and filtration. 2) Five bottles were treated as above. However, after the 24 h incubation period with 32P and before extraction with water, the samples were fumigated for one hour with CHCl3 vapors. 3) Five bottles were fumigated for one hour before tracer addition. The remaining sequence was as described in 1). Separate samples were used to determine the water content of the humus and ash-free weights. Aliquots of the water and HCO_3-extracts were digested and total ^{31}P and ^{32}P determined (colorimetry and Cerenkov counting, respectively).

Results

Water extractable-^{31}P ($^{31}P_w$) in the non-chloroformed samples amounted to 3.3 mgP kg^{-1} of ash free humus (Table 4). The humus contained 245% of water on a dry humus basis and 62.1 ±2% of ash-free material. Thus the humus solution has 0.84 mgP l^{-1} ($2.7*10^{-5}$ M), which is within the range of concentration encountered in mineral soils. Bicarbonate extractable-P ($^{31}P_b$) was 85 mg kg^{-1} and amounts to 26% of the total P content of the humus (332.6 mgP kg^{-1}). This is twice the amount of $^{31}P_b$ extracted in the humus of the field experiment (Table 3) and reflects the large variability within the forest floor.

In column 2 (Table 4), fumigation after tracer addition-incubation yielded 127 mg kg^{-1} of $^{31}P_w$. By subtracting $^{31}P_w$ in the unfumigated samples, the flush yields 123.7 mg kg^{-1} of ^{31}P. This result is similar to that obtained in the field experiment with the $HCO_3^-/CHCl_3$-flush combination (Table 3). The HCO_3- following the water yielded additional 103 mg kg^{-1} and a flush of 18 mg kg^{-1} of P.

Fumigation before tracer addition and incubation (Table 4, column 3) yielded the largest amounts of $^{31}P_w$ (192 mgP kg^{-1} of humus). Differences in $^{31}P_w$ between columns 2 and 3 are justified by different fumigation periods: when samples were fumigated prior to tracer addition (treatment 3), residual $CHCl_3$ vapors remained during the 24-hour incubation period after adding ^{32}P. These vapors not only prevented biomass regrowth but probably continued to lyse more unaccessible biomass. The increased P extraction efficiency of water in column 3 yielded smaller amounts of $^{31}P_b$

relative to column 2 which suggests that $^{31}P_b$ in column 2 might include microbial P.

Table 4. Effect of fumigation and timing of ^{32}P addition on water and 0.5 M $NaHCO_3$-extractable ^{31}P and ^{32}P

Extractant	Non-chloroformed (Treatment 1)	Chloroformed: After ^{32}P addition & incubation (Treatment 2)	Chloroformed: Before ^{32}P addition & incubation (Treatment 3)
		^{31}P (mg kg^{-1} humus)	
Water (P_w)	3.33c[1]	127b	193a
NaHCO3 (P_b)	85.0ab	103a	68.9b
		^{32}P (cpm*10-8 kg^{-1} humus)	
Water	0.031(0.2)[2]c	8.4(66)b	12.4(97)a
$NaHCO_3$	3.22(25)a	2.7(21)a	0.6(5)b

[1] Averages in the same line followed by the same letter do not differ significantly ($P<0.05$) by the Tukey test.

[2] Numbers between brackets indicate the percent recovery of total ^{32}P added.

Tracer recovery by water in the non-fumigated samples was near zero (0.2%) while fumigation after tracer addition yielded 66% of ^{32}P in the water (Table 4). When the biomass was killed prior to ^{32}P addition (column 3) 97% of the tracer was recovered in the water. These three results give clear evidence that microbial biomass activity controls, to a large extent, solution-P concentrations in humus. However, $^{32}P_b$ results also indicate that P-sorption reactions on the humus material are also occurring: in treatment 3 large amounts of ^{31}P were released by fumigation prior to tracer addition. We can reasonably assume that this release occupied much of the sorption capacity of the humus, yielding a small value of $^{32}P_b$. On the other hand, in treatments 1 and 2, where sites were not saturated with ^{31}P before tracer addition, about 23% of ^{32}P was HCO_3-extractable. Thus, this set of results strongly suggest that HCO_3-extractable-P in unfumigated or fumigated samples is probably a mixture of chemically bound-P and of microbial-P.

Second field experiment

Results from the previous laboratory experiment confirmed the role of microbial biomass and sorption reactions in the control of solution-P. However, due to the high variability of the forest floor, we were unable to devise treatments in which these variables could be experimentally manipulated in the field. Thus, we decided to use the natural forest floor variability and correlation analysis to test the hypothesis that retention of added tracer P is related to microbial biomass and adsorption reactions.

The experiment was conducted at site 1 (Table 1). Changes in methodologies with respect to the previous field experiment were as follows: we used 13 sites along a transect each containing one trenched and one untrenched treatment. The sugar-cane field was not used in this experiment. One day was allowed between trenching and tracer addition to allow for root die-back. The ^{32}P was applied on the surface of the humus layer (O2 horizon) rather than injecting it into the layer. No tracer was added in the mineral soil. The cylinders were recovered five days after tracer addition. Rainfall during this period amounted to 26 mm. Cylinder processing was as described above, except that the mineral layer was separated into two layers: 0-5 cm and 5 to approximately 15 cm. Roots in these two layers were sampled separately and analyzed for VAM infection. Determinations of microbial-P and HCO_3- extractable-P were as described in the laboratory experiment: a series of samples were sequentially extracted with water followed by HCO_3-; another series was first fumigated with $CHCl_3$ vapors and then extracted as the unfumigated ones. The humus samples were analyzed for pyrophosphate extractable-Fe and Al.

Results

The basic hypothesis of this second field experiment was that microbial biomass activity and P reactions with Fe-Al-OM complexes, and not roots, are the main sinks for P entering the soil solution and, hence, responsible for its retention in the humus layer. The flush in water-extractable ^{31}P in $CHCl_3$- fumigated samples, interpreted as microbial-P, was significantly correlated (r=0.74**) with the amounts of tracer retained in the humus (Figure 1), thus supporting part of the hypothesis (Table 5).

There is a significant correlation between ^{32}P retained in the humus and pyrophosphate-extractable Fe (Table 5). Total ^{31}P in the humus also correlates significantly with Fe (r=0.68**) and Al (r=0.56**) (not shown). Both results point to the importance of sorption and/or complex formation reactions in P retention and support the remaining part of the hypothesis. The smaller correlation between Al and total ^{31}P might justify the lack of correlation between Al and ^{32}P, but we should keep in mind that only five days of equilibration between tracer and humus were allowed.

Results from the prior laboratory experiment led to the hypothesis that P_b in non-fumigated and fumigated humus samples was related to microbial biomass and Fe-Al-OM complexes. In the present field experiment, the significant correlation coefficients of P_b and P_b after fumigation with $P_w$flush (microbial-P) and also with Fe and Al substantiate this hypothesis (Table 5).

Roots-VAM association. Root data confirmed results of the first field experiment: significantly higher P concentrations and VAM% infection in humus roots, and a significant trenching effect upon ^{32}P content per cylinder in the roots from the top 5 cm of the mineral soil. Recoveries of ^{32}P in untrenched roots amounted to 1% or less of total tracer added, and significant amounts of ^{32}P were translocated to roots outside the cylinder area in the untrenched treatments.

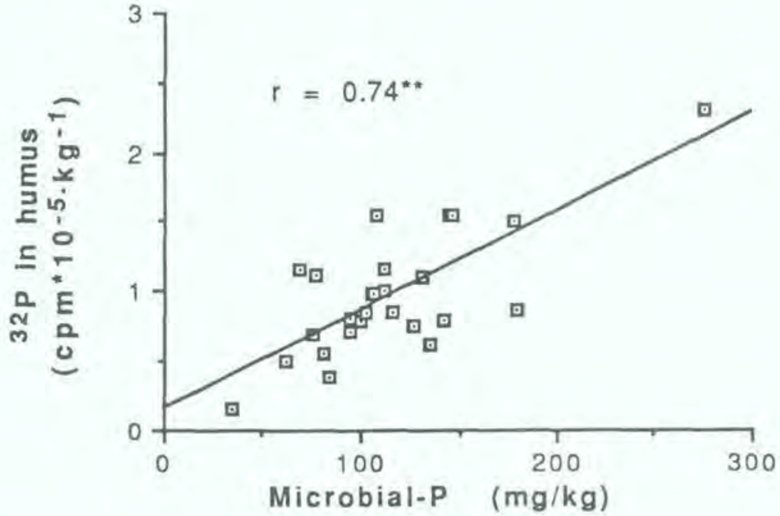

Figure 1. Relationship between tracer concentration and microbial-P content in the humus layer

The concentration of ^{32}P in the untrenched humus roots did not correlate with 32P recovered in the humus layer but was significantly correlated with VAM% infection (r=0.83**) (Figure 2). This substantiates, at field level, findings by Herrera et al. (1978) that P absorption by roots in the humus layer is a VAM mediated mechanism.

^{32}P recoveries and downward movement of P. Trenching did not have a significant effect upon tracer recoveries, that ranged from 44.7 to 86.5% and averaged 67.1% of total tracer added per cylinder in both treatments. Trenching neither affected the average proportion of tracer recovered in the humus (46.1%) nor in the top (0-5 cm) mineral soil (16.9%). The incomplete tracer recovery in the trenched treatment plus the presence of tracer in the top mineral soil contradict previous results by Stark and Jordan (1978), who did not detect tracer movement from the humus layer into the mineral soil. Further evidence of P movement into the mineral soil is given by the significantly larger total ^{31}P concentration in the top mineral soil layer (104 mg kg^{-1}) relatively to the 5-15 cm depth (85 $mg.kg^{-1}$).

No radioactivity was detected in the 5-15 cm layer of the mineral soil. This result is in contradiction with the low total ^{32}P-recovery per cylinder. Since trenching effect was not significant, the low recovery cannot be attributed to lateral diffusion or action of roots. Furthermore, low recoveries in the present experiment confirm similar results obtained in the first field experiment and in a separate one-day experiment (see text and Elliot et al., submitted, respectively). Further studies are needed to explain these low tracer recoveries together with the lack of radioactivity in the second depth of mineral soil sampled.

Table 5. Interrelationships among selected variables[1] (n=26)

Humus	P_w flush	P_b	P_b fumigation	P_b flush	Fe	Al	^{32}P
P_b	0.91	--					
P_b fumigation	0.92	0.90	--				
P_b flush	0.59	n.s.	0.77	--			
Fe^2	0.58*	0.69	0.82	0.88	--		
Al	n.s.	0.49	0.52	n.s.	0.59*	--	
^{32}P humus	0.74	0.68	0.81	0.70	0.63*	n.s.	--

[1] All coefficients significant at $P<0.01$ except at $P<0.05$ for Fe when indicated with *; n.s. = non significant.

[2] Sample number reduce to 13 since cylinders that remained in the field (trenched treatment) contaminated soil with Fe.

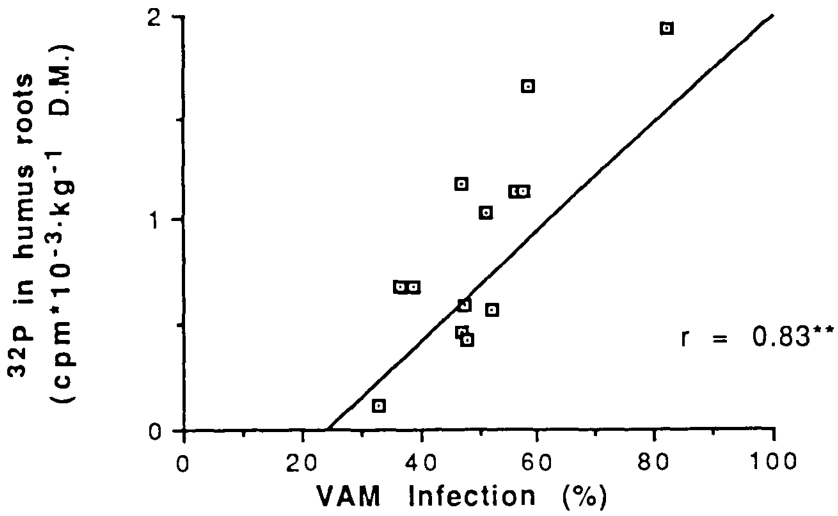

Figure 2. Relationship between tracer concentration and VAM infection in roots from the humus layer.

Conclusions

Large amounts of roots are present in the floor of the Atlantic-Coastal forest ecosystem. At the study site, P in the humus is cycled back into the vegetation by a VAM mediated mechanism. This mechanism is probably a response to a highly competitive demand for P by saprophytic organisms decomposing the litter and by chemical reactions with by-products of this decomposition (Al and Fe-organic matter complexes). Microbial biomass activity and chemical reactions are the main mechanisms controlling instantaneous P concentrations in solution. The proposed laboratory methodology to evaluate these P pools needs to be better understood and tested in various kinds of forest floor materials. More research focussing on the chemical composition of the O2 horizon material and in the interrelationships between organic and inorganic components in this layer is needed.

When tracer is added to the humus, it moves down into the top mineral soil and a significant enrichment in total ^{31}P of the top mineral soil layer was evident. Roots in this top mineral soil were also radioactive, probably through absorption of ^{32}P moving down from the humus layer plus translocation from humus roots. Trenching effects suggested that roots in the humus are short-range feeder roots from deeper ones in the mineral soil. Tracer losses from the cylinder, as determined by mass balance, were larger when injected in the humus than in the mineral soil. The quantities and forms of P lost this way should be further studied, since they contradict the accepted idea of a close P-cycle in tropical forest ecosystems.

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PHOSPHORUS CYCLING IN HUMID TROPICAL SUCCESSIONAL FORESTS

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Introduction

The purpose of this paper is to review what is known of phosphorus cycling in humid tropical successional forests. In order to understand phosphorus cycling, estimates are needed of the size of various vegetation and soil P pools, of fluxes into and out of the forest ecosystem and among the different pools, and of how controls on these processes operate. Unfortunately, such information is incomplete for even the most well-characterized ecosystem - agroecosystems found on base-rich soils in the temperate zones. The data base for humid tropical forests, especially successional ones, is much less complete.

A further complication is that the concept of successional forests includes the notions of forest disturbance and subsequent regrowth. Disturbances can differ in form, aereal extent, intensity, frequency, and duration. These differences, in turn, can affect the quantity, chemical forms, and availability of P as well as the nature and rate of forest regrowth, and so add complexity to the study of such systems.

Windthrows, fire, lightning strikes, landslides, and insect or disease-induced mortality are examples of disturbances which occur in humid tropical forests, but perhaps the most wide-spread and most studied disturbance, and that currently causing the greatest concern, is land-clearing followed by agricultural production. Consequently, much of the following discussion is devoted to considering how this type of disturbance affects phosphorus pools, transformations, and transfers, although the effects of other types of disturbance are also considered when possible.

Land-clearing

The effects of land-clearing on soil and vegetation phosphorus pools can vary greatly depending on the method of vegetation removal and the extent of soil disturbance. Total and available soil P pools, and important determinants of organic P levels in soil such as soil organic matter contents and soil aggregation, generally decrease as disturbance intensity increases (Uhl et al., 1982; Alegre and Cassel, 1986; Alegre et al., 1988). Mechanical clearing is clearly the most deleterious land-clearing practice since most of the vegetation and topsoil, and the P stored in these compartments (a high proportion of total P in weathered soils) is often removed. In general, the use of other, less intense land-clearing methods, such as manual clearing or manual clearing and burning, result in little immediate change in total or available P levels (Lal and

Cummings, 1979; Ramakrishnan and Toky, 1981; Uhl et al., 1982; Smyth and Bastos, 1984; Stromgaard, 1984; Alegre et al., 1988), although some researchers have reported losses of Olsen-available P from the top 3 cm of a Dystropept due to forest felling alone (Ewel et al., 1981).

The size of the cleared site affects changes in soil P availability. There is little change in extractable soil P when small (<300 m^2) areas of forest are felled, presumably because there is little change in microclimate, hence in organic matter mineralization and P release. Any P mineralized is rapidly adsorbed onto variable charge soil minerals, or taken up by roots of standing vegetation (Vitousek and Denslow, 1987). These generalizations, however, may not hold for larger clearings (Denslow, 1987). Further studies of P availability and loss from a range of different-sized clearings, including those representative of shifting agriculture, are needed.

In cases where felled vegetation is burned on-site, available P levels in soil usually increase after burning (Zinke et al., 1978; Ramakrishnan and Toky, 1981; Kyuma and Pairintra, 1983; Sanchez et al., 1983; Smyth and Bastos, 1984; Stromgarrd, 1984; Werner, 1984; Lambert and Arnason, 1986; Alegre et al., 1988; however, see Ewel et al., 1981). The magnitude and timing of the peak in P availability vary greatly among studies, probably due to differences in the quantity of biomass burned and the amount of P contained in the ashes, interactions of organic and inorganic forms of P with inorganic soil constituents, fluctuations in soil microclimate, rates of mineralization of organic P, and microbial immobilization processes. Fassbender (1975) and Sertsu and Sanchez (1978) have noted changes in P availability in soil due to heating.

It is unclear whether "losses" of P associated with forest felling and burning, such as those reported by Ewel et al. (1981), actually occur as a result of aeolian transport of ashes and particulate matter or increased erosion, runoff, and leaching, or whether phosphorus is simply converted to other forms that are not extractable by the methods used. There have been few studies which have attempted to investigate the forms in which P is lost and the routes by which these losses occur. The loss of P in ash, especially, is not well quantified. Erosion and runoff are likely to be the main routes of loss of P in organic or particulate form from exposed soil in high rainfall regions (Vitousek, 1983). Relatively large quantities of soil and P may be lost during relatively short periods of time when the soil is exposed (Kyuma and Pairintra, 1983).

It has been suggested that P leaching, at least in temperate forest ecosystems, may occur via production of low molecular weight, narrow C/P ratio organic acids (fulvic acids) which are more mobile than inorganic phosphate anions and may be transported downwards through the soil profile or in surface run-off (Rolston et al., 1975; Schoenau and Bettany, 1987; Frossard et al., 1989). Some data suggest that leaching may also occur in areas clearead of tropical forest (Stromgaard, 1984). While it is clear that high soil temperature, greater fluctuations in soil moisture, and the destruction of soil aggregates as a result of clearing of tropical forest may result in rapid rates of decomposition and the production of such organic P complexes, other factors such as the pattern of water movement through the soil profile and the interaction of these complexes with charged soil minerals are likely to determine the importance of leaching as a mechanism of P loss. Given that these processes are complex and are likely to differ among permanent and variable charge soils (Oades, et al., 1989) much work remains to be done in order to assess the importance of leaching for P loss from disturbed tropical sites.

Crop production

Crop production following land-clearing decreases total P levels in soil due to P removal in harvested products. Reductions in the size of the soil organic P pool also occur as a result of increased mineralization (Agboola and Oko, 1976; Adepetu and Corey, 1977; Mueller-Harvey et al., 1985; Mueller-Harvey and Wild, 1986). Limited data show that the relative amounts of organic P mineralized can vary widely even on similar soil types (Nye and Bertheux, 1957; Acquaye, 1963; Brams, 1971; Igue et al., 1971; Adepetu and Corey, 1977; Mueller Harvey et al., 1985). These differences may be due to differences in the quantity and type of soil minerals present, limitation of mineralization by the availability of other nutrients, wetting and drying cycles, soil temperature, and organic matter quality (Duxbury, et al., 1989). During organic matter mineralization, soil C/P ratios often narrow, suggesting that P limits soil microbial populations and is immobilized by them or that interactions of P, especially inositol phosphates, with charged soil minerals result in relatively greater retention of P than C (Acquaye, 1963; Adepetu and Corey, 1976; 1977; Mueller-Harvey et al., 1985; Stewart and Tiessen, 1987). Since it has been suggested that P availability limits forest productivity on weathered tropical soils (Vitousek, 1984) and that organic forms of phosphorus are highly correlated with crop production on such soils (Acquaye, 1963; Adepetu and Corey, 1976; 1977; Agboola and Oko, 1976; Tiessen et al., 1984;), a much more detailed understanding of organic P transformations is needed, including the development of methods for measuring organic P mineralization.

Changes in inorganic soil P pools occur concomitant with changes in organic P. Some organic P mineralized is taken up by crops or remains available to plants, but other portions become associated with less-available inorganic Fe-, Al-, and occluded-P fractions. The amount of P in these fractions usually increases in proportion to their relative amount initially present (Igue et al., 1971; Adepetu and Corey, 1976, 1977; Mueller-Harvey et al., 1985; Mueller-Harvey and Wild, 1986). Quantities of P mineralized may depend on past management. In one study, P mineralization was greater in soil taken from a recently cleared secondary forest compared to soil from a previously cropped field; relative quantities of organic P mineralized, however, were similar (Table 1). With continued cropping, available P levels may be determined by buffering reactions with organic or inorganic soil P pools (Stewart and Tiessen, 1987).

In temperate ecosystems, the absolute and relative sizes of the inorganic (eg. labile $NaHCO_3$-P, Fe- and Al-P, Ca-P, and occluded P) and organic (eg. $NaHCO_3$ and NaOH-P) soil P fractions and interactions between them vary with weathering of parent material, mineralogy, topographic position, soil texture, and management (Smeck, 1973, 1985; Tiessen et al., 1984; Roberts et al., 1985; O'Halloran et al., 1985, 1987a, 1987b).

Similar studies of the interrelationships among inorganic and organic P fractions are need for tropical soils where mineralogy and microclimate are often different from those in temperate zones.

Table 1. Changes in phosphorus fractions (mg kg^{-1}) after two crops on previously fallowed or cropped sites. (From Adepetu and Corey, 1977).

	Fallowed site				Cropped site			
Fraction	Before Crop	After Crop	Change	%	Before Crop	After Crop	Change	%
Organic	213	158	-55	26	136	98	-38	28
Al-P	64	82	+18	28	45	5	+11	24
Fe-P	103	130	+27	26	82	106	+24	29
Occluded	75	77	+ 2	3	53	50	- 3	6
Ca-P	11	14	+ 3	27	10	11	+ 1	10
TOTAL	466	461	- 5		326	321	- 5	

P dynamics during forest regrowth

Additions and losses. Additions of P in successional humid tropical forests occur via atmospheric deposition and mineral weathering. Data (Vitousek and Sanford, 1986) from undisturbed humid tropical forests can serve as rough estimates for similar processes in successional forests, since data for the latter are lacking. In general, inputs via rainfall and dust are expected to be relatively small in magnitude, around 0.1 to 0.5 kg $ha^{-1}y^{-1}$ and additions of P to the rhizosphere from mineral weathering on deep, highly weathered soils are also apt to be very small. On younger, more calcareous soils, additions from weathering may be greater, but such minerals are lost relatively rapidly due to rapid weathering under humid tropical conditions.

As previously noted, hydrological factors determining leaching, sediment, and runoff losses are complex and methodological difficulties in quantifying such fluxes result in a scarcity of data. It is clear that most losses, if present, will occur early in succession when the vegetation cover is poorly developed, decomposition is rapid, and plant demand for P is low. Despite these conditions, losses of P may nevertheless be slight from soils which have a high P-fixation capacity (Andisols, Oxisols, humic sub-groups of Inceptisols and Ultisols), and/or high rates of water flow through macropores (eg. Psamments, Spodosols, and well-aggregated Andisols and Oxisols). Moreover, P losses from older secondary forests are apt to be similar to those in undisturbed forest, on the order of less than 1 kg $ha^{-1}y^{-1}$ (Vitousek and Sanford, 1986).

P accumulation in organic matter. It has been suggested that P availability may limit biomass and P accumulation in undisturbed tropical forests growing on Ultisols, Oxisols, Andisols, and Psamments (Vitousek, 1984; Vitousek and Sanford, 1986), but not on moderately fertile soils such as Alfisols, some Inceptisols, and Entisols. This pattern also appears to be generally true for tropical secondary forests (Table 2). Secondary vegetation growing on moderately fertile soils appears to accumulate about 2 times more aboveground biomass and P, over similar periods of time, than vegetation found on infertile soils. Departures from this generalization may be due to a number of factors: 1) Available P levels in soil and rates of mineralization of organic P can be relatively high in the period following field abandonment even on relatively P-infertile sites. 2) Differences in rates of biomass and P accumulation on similar soils may be due solely to difference in age of the vegetation. Rates of biomass and P accumulation are usually greater in younger, as compared to older, vegetation, hence only forests of similar age should be compared. 3) Soils may be misclassified (in some cases, the soil classification was not mentioned or appeared to be doubtful based on cited properties). 4) Previous managment can affect P availability and accumulation of P and biomass. For example, the accumulation of biomass and P in a 3-year-old secondary forest on land cleared by bulldozer were 50 to 100 times less than in a secondary forest growing in an area that had been cut but not burned (Uhl et al., 1982).

Belowground biomass production and rates of P accumulation and their relationship to soil type are difficult to evaluate due to lack of data. Limited data suggest that rates of accumulation of P in roots are much lower than those measured aboveground and are lower on infertile compared to fertile soils (Table 2).

P transfer in litter. Rates of foliage production, aboveground litter production, and the quantity of P cycled in litter in young secondary forest increase rapidly up until canopy closure (approximately 10 years), then level off or decline with time (Miller, 1984; 1986). It has been suggested that the quantities of litterfall are higher and litterfall mass/nutrient ratios are lower on more fertile sites (Vitousek, 1984, Cuevas and Medina, this volume). Unfortunately, few data are available to examine this hypothesis for tropical successional forests. There appears to be no consistent pattern with regard to the relationship between soil fertility and litter production; the amounts of phosphorus returned in litterfall, however, appear to be lower on infertile as compared to fertile soils (Table 2). Again, caution is required in interpreting these patterns due to the considerations mentioned above.

Patterns of P cycling via belowground litter production and exudates in successional forests and their relation to soil fertility are unknown. Few data exist for root litter production or the quantities of P contained in root litter, P retranslocation from roots before their death, or whether root P status changes with vegetation age or species composition. Such studies are greatly needed.

Rates of P transfer from litter to soil depend on rates of mineralization from litter. The latter may be affected by organic residue quality, placement, climate, and soil texture. As a result, rates of litter decomposition in tropical forests are variable (Anderson and Swift, 1983) and difficult to relate to soil type (Vitousek and Sanford, 1986). In tropical succesional forests, rates are also apt to vary with successional stage due to changes in chemical composition of litter as a result of changes in plant species or nutritional status of a given species and changes in nutrient availability. Rates of mineralization might be expected to be more rapid in early as compared to later successional stages due to the production of more nutrient-rich litter, greater nutrient availability, higher soil temperature, and greater fluctuations in moisture contents of soil and litter in early successional ecosystems.

Table 2. Accumulation of aboveground and belowground biomass (Mg ha^{-1}) and phosphorus (kg ha^{-1}), and dry mass (Mg ha^{-1}) and phosphorus (kg ha^{-1}) content of litterfall in humid tropical successional forests.

Site	Age (y) Ref.	Aboveground		Belowground		Litterfall		
		Mass	P	Mass	P	Mass	P	
Moderately fertile soils								
Zaire, ferralitic (Alfisol?)	2	17.9	16.9	6.9	4.9	1.9	0.4	1
	5	76.7	21.0	25.7	7.7	9.8	3.2	
	8	121.6	24.9	22.7	9.1	8.0	4.1	
	18	121.1	69.7	31.2	34.2	22.8	4.1	
Colombia, (Inceptisol?)	2	19.0	16.0					2
	5	68.0	22.0					
	16	203.0	55.0					
Panama, (Inceptisol?)	2	15.6	31.0					3
	2	28.5	48.0					
	4	42.5	89.0					
	6	56.5	91.0					
India, (Oxisol?)*	1	5.0	5.0			1.2	1.0	4
	5	23.3	20.0			4.8	2.0	
	10	57.5	25.2			7.1	3.0	
	15	103.7	44.1			7.7	4.0	
	20	147.6	64.1			9.7	5.0	
India, (Oxisol?)*	5	19.8	31.0				17.0	5
	10	50.9	78.0				14.0	
	15	96.8	116.0				26.0	

Table 2. Continued.

Site	Age (y) Ref.	Aboveground		Belowground		Litterfall		
		Mass	P	Mass	P	Mass	P	
Moderately fertile soils (Cont'd)								
India, (Oxisol?)*	1	6.3	7.8			1.1	0.6	6
	2	10.1	13.4			3.3	2.1	
	4	20.1	25.1			5.7	4.7	
	8	50.0	26.7			6.7	5.6	
	12	83.7	46.3			7.3	6.0	
Infertile soils								
Ivory Coast, ferralitic	1.2	8.8	4.6					7
	2.2	14.1	6.9					
	4.0	21.6	10.5					
	6.5	38.4	14.4					
	15.0	77.6	20.2					
Sarawak, Ultisol	12-15	32.1	6.2	6.3	4.0	4.0		8
Venezuela Oxisol	5	40.0	8.4	8.4	2.7	8.2	1.6	9
Venezuela Spodosol	3	0.8	0.4	(bulldozer cleared)				10
	3	12.9	15.7	(cut)				
	3	8.7	17.4	(cut and burned)				
Peru, Ultisol*	3	39.7	18.2		0.6			11
	10	75.0	22.7		5.5			
	25-30	204.0	54.5		15.9			

* Soil classification in doubt based on soil properties.
References: 1 = Bartholomew et al., 1953; 2 = Folster et al., 1976; 3 = Golley et al., 1975; 4 = Toky and Ramakrishnan, 1983; 5 = Rao and Ramakrishnan, 1989; 6 = Swamy and Ramakrishnan, 1987a, b; 7 = Jaffe, 1985; 8 = Andriesse and Schelhaas, 1987; 9 = Uhl and Jordan, 1984; 10 = Uhl et al., 1982; 11 = Scott, 1977.

P transformations in soil. Transformations among various inorganic and organic forms of soil P, differing in availability to plants and soil microorganisms, occur during succession. Although progressively greater amounts of P are accumulated in vegetation during secondary succession, various studies have noted that available forms of P in soil decrease during the first 4-8 years but increase thereafter (Zinke et al., 1978; Aweto, 1981; Ramakrishnan and Toky, 1981; Szott and Palm, unpublished). These changes in available P are usually associated with similar patterns in soil C, suggesting that soil organic matter plays an important role in buffering available P. The time required for levels of available P to reach levels measured at abandonment vary, perhaps due to differences in soil mineralogy, the quantity and quality of litter added to the soil, and in factors affecting P mineralization/immobilization processes. The importance of these factors deserves further research.

In shifting agriculture fallows, the fallow rotation period, and the number of previous crop cycles, appear to affect the status of available soil P (Table 3). Although there was little difference in organic C in soil among different lengths of fallows, levels of available soil P were much higher with increasing length of the fallow rotation. Presumably, more frequent harvests and greater removal of P in the products harvested in systems with short fallow rotations deplete more rapidly available inorganic P pools and other soil P fractions which buffer this pool. Further work is needed to define the interactions between organic and inorganic soil P pools and their relationship with fallow period.

Table 3. Organic carbon and available phosphorus contents of soil (0-40 cm) prior to clearing. The fields differed in length of the fallow cycle.

Fallow cycle	Organic C	Available P
y	Mg ha^{-1}	kg ha^{-1}
5	8.2	8.3
10	11.7	18.7
30	10.0	24.9

(calculated from Ramakrishnan and Toky, 1981).

Conclusions

The majority of work related to phosphorus cycling in humid tropical successional forest has concentrated on quantifying stocks in aboveground biomass and determining changes of available phosphorus in soil. In order to achieve an understanding of phosphorus cycles, however, much more research having a different focus is needed. Major gaps in knowledge include the dynamic relationships between different inorganic and organic phosphorus pools in soil, controls on organic phosphorus mineralization/immobilization processes, and how these relationships change in response to different types of disturbances. Advances in such research will depend on the development and refinement of appropriate methods.

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AGROFORESTRY SYSTEMS IN LOWLAND TROPICS: IS PHOSPHORUS IMPORTANT?

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INTRODUCTION

Agroforestry is often recommended as the preferred agricultural system for the humid tropics. Reasons given generally pertain to the role of trees in maintaining soil fertility by recycling nutrients via litterfall and prunings, thereby mimicing the natural forest ecosystem. In addition, trees have deeper rooting systems and can tap and recycle a source of nutrients initially unavailable to most annual food crops. There is however, little data to support or refute such claims.

The high productivity of tropical forest ecosystems on infertile soils is attributed to efficient nutrient cycling with relatively small nutrient losses. Nutrient cycling processes and pathways in agroforestry systems are similar to those of natural forest ecosystems. There are, however, important differences in the relative sizes or fluxes, of certain nutrient cycling pathways between the two types of systems and even among different agroforestry systems. The major flux difference between natural forest ecosystems and agroforestry systems is that of nutrient losses. Whereas nutrient losses via leaching, erosion, and gaseous emissions are probably similar for the two systems agroforestry systems have an additional loss pathway through product harvest that does not exist in natural systems. These differences will result in different nutrient budgets that will determine the productivity and sustainability of various agroforestry systems.

Most nutrient cycling research in agroforestry to date has dealt with nitrogen cycles and budgets. The use of nitrogen-fixing, leguminous trees in most agroforestry systems provides a source of N that can partially, or in some cases completely offset the loss of N through crop or product exports. Little attention has been given to the cycles and budgets of other nutrients. Whereas trees can recycle these other nutrients they cannot provide a new input of these nutrients to the system as they can for N. It therefore becomes crucial to understand the nutrient cycling processes and budgets of these nutrients in agroforestry systems in order to ascertain the necessary management practice for continued productivity of the system. Among these other nutrients, P is particularly important considering the extent of P fixing soils in the tropics and the large proportion of P extracted by plants that is exported in crop harvest (Sanchez et al., this volume).

The objectives of this paper are to look at P cycling and budgets and soil P changes in several different agroforestry systems, to evaluate the long term consequences of agroforestry systems on P availability, and to consider the role of trees in the P cycle of agroforestry systems. Much of the data referred to comes from research conducted in Yurimaguas, Peru by the Tropical Soils Research Program of North Carolina State University

(Caudle and McCants, 1987; Caudle, 1989). The soils are acid (pH 4.5), infertile Ultisols with relatively low P-fixing capacity. The remainder of the data comes from research on a variety of tropical soils of varying fertility and P-fixing capacities.

Phosphorus cycling and budgets, and soil phosphorus in various agroforestry systems

Agroforestry systems vary considerably in the amount of P cycled in litterfall or prunings and the amount of P exported in products on a yearly basis. These differences have important consequences to long-term productivity and sustainability. Several types of agroforestry systems will be compared that differ in the intensity of management and subsequent nutrient export.

Shifting Cultivation and Improved Fallows. Shifting cultivation, the traditional farming system of the tropics, could be considered the original simplest, and least intensive agroforestry system (Raintree, 1987). It is a temporal agroforestry system in that crops are grown for a year or two and replaced by trees as the natural fallow vegetation regrows. In this system, as with all agroforestry systems, nutrient losses occur with forest clearing and burning and during the establishment phase. Further P losses via crop harvest are generally small, less than 10 kg ha^{-1} over a two year period (Sanchez et al., this volume). These losses are partially recuperated during a 10 to 15 fallow phase from atmospheric P inputs of 0.1-1 kg ha^{-1} yr^{-1}. One of the purposes of the fallow is to restore soil fertility by accumulating nutrients in the vegetative biomass. This process represents a transfer of nutrients from the soil to the vegetation and should not be considered a net addition to the system. The nutrients stored in the biomass become a source of ash fertilizer when the vegetation is cut and burned.

Improved or managed fallows are different from natural fallows of shifting cultivation in that specific trees, shrubs, or cover crops are intentionally planted following crop abandonment rather than allowing the natural vegetation to recover. Improved fallows are similar to shifting cultivation in that recycling of nutrients by the vegetation is maximized and nutrient exports are generally low. One of the purposes of improved fallows is to restore nutrient stocks in the vegetation more rapidly than natural fallows. By accumulating nutrients more rapidly, improved fallows should provide for higher production by allowing more frequent cropping. More frequent cropping, however, also has implications to long-term productivity.

A study in Yuirimaguas comparing a natural fallow (BF) to six leguminous fallow treatments, *Centrosema macrocarpum* (Cm), *Stylosanthes guinanensis* (Sg), *Pueraria phaseloides* (Pp), *Cajanus cajan* (Cc), *Desmodium ovalifolium* (Do), and *Inga edulis* (Ie) showed that the fallows stored P from a low of 15 kg ha^{-1} (sg) to a high of 35 kg ha^{-1} (Do) during the first 29 months (Figure 1; Szott, 1987). Only the *Desmodium* and *Inga* treatments stored more P than the natural fallow. Those two treatments also had the highest total and tree biomass (Table 1). The improved fallows without trees, *Stylosanthes* and *Pueraria*, stored less P and less biomass than the natural fallow in 29 months. The remaining two treatments were similar to the natural fallow.

Table 1. Above-ground biomass in six managed fallows and a natural fallow at 29 months (Szott, 1987)

Fallow treatment	Biomass (Mg ha^{-1})
Centrosema macrocarpum	9.8
Stylosanthes guianensis	8.6
Pueraria phaseoloides	7.7
Cajanius cajan	12.4
Desmodium ovalifolium	21.9
Inga edulis	24.3
Natural fallow	16.4

Available soil P averaged 15 kg ha^{-1} and did not differ among the treatments at 29 months. There was however, a decline of about 10 kg in available P during the 29 months in all treatments (Figure 1). This decline was obviously due to the transfer of P from the soil to the vegetation. It is interesting to note that there was an overall increase in the sum of the P in the vegetation and that available in the soil (Figure 1). This increase indicates a transfer of P from some of the unavailable forms in the soil to more available forms. In fact, as much as 20 kg ha^{-1} would have to come from this unavailable soil P to satisfy the amount of P found in the vegetation of some of the treatments. The differences in the P stored in the various fallows may in part reflect differences in the P uptake kinetics of different species and/or their ability to tap more unavailable forms of soil P.

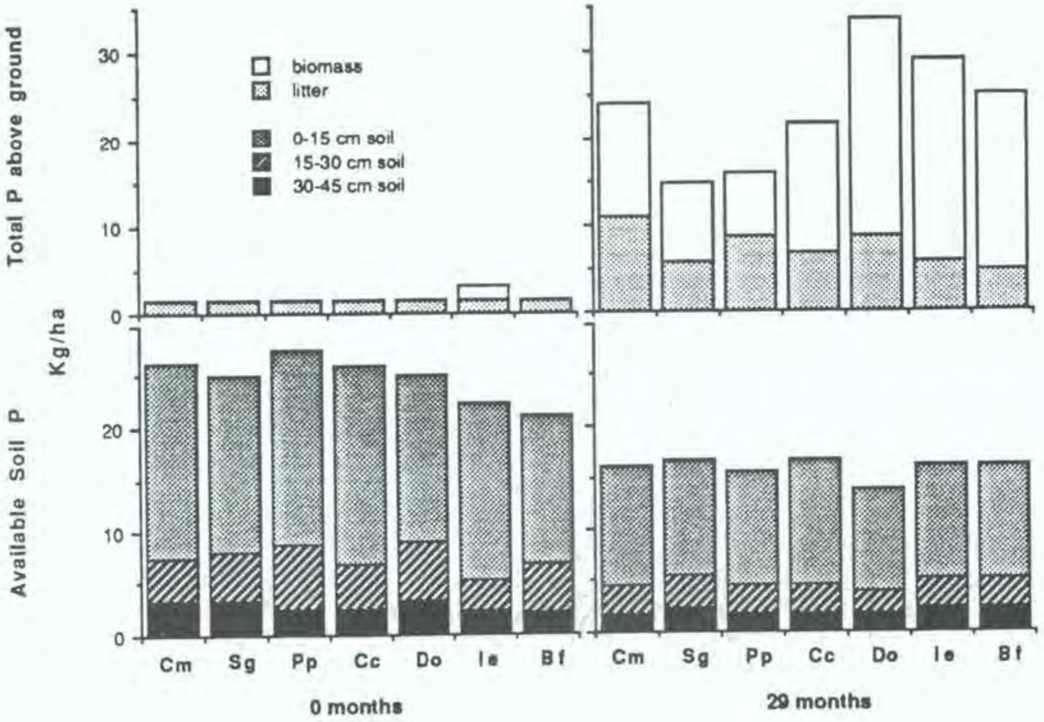

Figure 1. Changes in total P stored in the biomass and in available P in six improved and a natural fallow in 29 months (Szott, 1987).

It appears that certain improved fallows have the potential to store P in the vegetation more quickly than the natural fallow but this depends on the biomass accumulation capacity of the specific vegetation and therefore probably on the presence of trees. The ability of the various fallow species to take up soil P may also be crucial. The relevance of these differences in rates of P accumulation to improved crop yields and shorter crop-fallow cycles is currently under investigation.

The fact that fallow vegetation draws on total soil P and that P is exported in crops has implications to the sustainability of these systems. In the case cited above, the fallow vegetation drew down the total soil P by 10 kg ha^{-1}, or less than 5% of the original soil P. The question arises as to how many crop-fallow cycles are possible if P is not added to the system. Perhaps some of the P stored in the vegetation becomes available upon burning but some of it may also be fixed in the soil and become unavailable again. Understanding the P transfers between soil and vegetation and the transformations in the soil during fallow recovery, clearing, and cropping will aid in the management of these systems.

Alley cropping. Alley cropping is an agroforestry system in which arable crops are grown in the alleys between hedgerows of trees or shrubs. The hedgerow trees, preferably legumes, are periodically pruned to prevent shading and to supply mulch to the crop. Alley cropping is similar to fallow systems in that part of the land is planted to trees and is permanently in fallow. The purpose of the trees is to provide nutrients for the arable crop by fixing nitrogen, recycling nutrients, and extracting nutrients from the subsoil. Alley cropping is more intensive than shifting cultivation or improved fallows because two or three crops are grown per year, and total annual nutrient exports in crops are higher.

The extent to which trees actually provide nutrients for the crops grown in the alleys has not been studied in sufficient detail. Data from alley cropping experiments on both Alfisols (Kang and Wilson, 1987) and Ultisols (Szott, 1987) indicates that tree prunings may provide sufficient nitrogen, potassium, and calcium to meet crop demands, while P demands are not met (Table 2). The data are often confounded because P, calcium, and potassium fertilizers are added to the crops because nitrogen has been the nutrient of interest. The data in Table 2 stress that P will be the primary limiting nutrient. Prunings of *Leucaena leucocephala* and *Inga edulis* do not contain enough P to produce one corn or rice crop. Again, the soil P reserves are being drawn down to produce the crops and there is a net loss of P from the system. The productivity of alley cropping systems will decrease much faster than shifting cultivation because of more frequent cropping.

Decreases in extractable soil P were also found in an alley cropping system on an Alfisol after four years of maize production with *Leucaena leucocephala* as the hedgerow species (Kang et al., 1981). These decreases were found even with annual additions of 20 kg ha^{-1} of P. Similar decreases were found after 32 months of alley cropping on an unfertilized Ultisol with *Inga edulis* and *Cajanus cajan* as the hedgerow species (Szott, 1987).

Phosphorus is obviously the nutrient of most concern to the success of continuous crop production in alley cropping systems. Research in many areas is necessary to improve this situation. Selection and management of trees with higher leaf biomass production and higher P concentrations in the leaves may help by providing more P in prunings. Phosphorus availability from the prunings may also depend on the quality of the leaf material and the resultant decomposition and mineralization processes. Pruning additions may actually increase P availability by converting more recalcitrant inorganic forms of soil P taken up by the trees to organic forms supplied in the prunings. Some trees may have deeper rooting systems and be less competitive for nutrients with arable crops than trees with shallow, spreading root systems. Any improvement in the efficiency of P cycling in alley cropping systems will help maintain crop production over the short term but it is obvious from simple input-output budgets that additional inputs of P will be necessary for long term production.

Table 2. Nutrients added in prunings and removed in crop harvest in two alley cropping systems in Nigeria.(Adapted from Kang and Wilson, 1987; Sanchez, 1976) and Peru (Adapted from Szott 1987; Sanchez 1976).

	Biomass	N	P	K	Ca	Mg
	Mg ha^{-1} per crop	kg ha^{-1} per crop				
Ibadan, Nigeria (Alfisol)						
Leucaena prunings	3.0	98	8	74	40	6
Corn*						
Grain	3.0	75	18	45	9	6
Straw	4.5	45	9	54	13.5	9
Yurimaguas, Peru (Ultisols)						
Inga edulis prunings	2.5	62	5	24	15	4
Rice**						
Grain	1.5	35	7	10	1.4	0.3
Straw	1.5	7	1	18	2.6	2.2

*Fertilized with 20 kg P, 20 kg K each crop
**No fertilizer

Tree-Crop Production Systems. Perennial tree crop production systems range from monoculture tree crops such as oil palm and rubber plantations to multispecies, multistrata systems such as coffee plantations with interplantings of *Erythrina*, for shade and prunings, and *Cordia alliodora*, for timber. Tree crop systems are perhaps the most successful and well known agroforestry systems. They also depend on trees for recycling and extracting nutrients from the subsoil. When compared to shifting cultivation systems they are more intensively managed and have much larger, annual nutrient exports in the form of product harvests. Phosphorus removal by harvest can be as high as 25 kg ha^{-1} yr^{-1} as for oil palm plantations to a low of around 4 kg ha^{-1} yr^{-1} for rubber plantations (Table 3). To maintain crop production in these systems, P must clearly be added.

Table 3. Phosphorus removed in crop harvests from various tree crop plantation systems (Turner and Gillbanks, 1974)

Crop	Yield	P
		kg ha^{-1} yr^{-1}
Oil Palm	25 t fruit bunches	11.0
Coconut	2400 kg copra	6.8
Cocoa	1125 kg dried beans	5.0
Coffee	1125 kg made coffee	7.3
Tea	1350 kg dried tea leaves	4.5
Rubber	1928 kg dry rubber	3.8

One explanation for the continued production in agroforestry systems is that nutrient cycling via litterfall provides enough P to balance exports. An example is shown for a cacao plantation in Costa Rica in which 4 kg P ha^{-1} yr^{-1} are extracted from the system in cacao fruits but litterfall adds as much as 14 kg ha^{-1} yr^{-1} to the soil surface (Table 4, Heuveldop et al., 1988). It is necessary to keep in mind that P additions to the soil surface from litterfall are only an internal transfer of P between the soil and vegetation pools in the system, not a new source. As with alley cropping or any other agroforestry system, trees may extract P from recalcitrant pools or from the subsoil and make it available for annual crops or other trees via decomposition of litterfall. Crop and litterfall production should be expected to decrease with time if external sources of P are not added. In fact, P fertilizers were added to these plantations at an annual rate of 29 kg ha^{-1} (Fassbender et al., 1988). Crop and leaf production were maintained by an external input of P; the system was not self-sustaining.

Table 4. Phosphorus removed in fruit harvest and P in litterfall in cacao plantations under *Cordia alliodora* or *Erythrina poeppigiana* (Heuveldop et al., 1988)

Turrialba, Costa Rica (Dystropept)	
Nutrient extractions in cacao fruits	kg P ha^{-1} yr^{-1}
T. cacao/*C. alliodora*	4.0
T. cacao/*E. poeppigiana*	4.3
Nutrients in litterfall	
T. cacao/*C. alliodora*	13.9
T. cacao/*E. peoppigiana*	8.8

*29.3 kg $ha^{-1}yr^{-1}$ phosphate application

Data from a low-input based (no fertilizer additions) multistrata system in Yurimaguas shows again the high level of P extraction in product harvest (Table 5). Crops were interplanted with the trees of *Inga edulis*, *Bactris gasepaes*, and *Cedrelinga caeteniformis* during the first two years. After the second year, the trees shaded out the crops and a shade-tolerant cover crop, *Centrosema macrocarpum*, was planted. In the fourth year, the *Inga edulis* trees were pruned for fuelwood. A total of 26 kg ha^{-1} of P was removed from the system in 4 years compared to much smaller additions from atmospheric inputs. Despite the high rates of P extraction from the system and no additions of P fertilizer, the trees continue to grow rapidly (Alegre et al., 1989) at soil P levels below that for crop production. Once again recalcitrant forms of soil P are probably being used by the trees.

Table 5. Phosphorus extracted from a low-input, multistrata tree production system during the first four years of establishment (Alegre, Szott, and Palm unpublished data)

Product harvested	Year	Yield t ha^{-1}	P kg ha^{-1}
Rice	1	1.7	8
Rice	1	1.2	6
Cowpea	1	0.9	4
Rice	2	0.6	3
Fuelwood	4	13.0	5
		TOTAL	**26**

Summary and Research Imperatives

In conclusion, the large amount of P extracted in product harvest coupled with the low P availability of most acid tropical soils makes P an important and potentially limiting nutrient in most agroforestry systems. Many agroforestry systems remain productive only because of large annual inputs of P fertilizers, but some agroforestry systems remain productive even without P fertilizers, indicating that trees play a special role in P cycling, transformations, and availability.

These observations raise two important questions relevant to all agroforestry systems: How long can tree-based systems maintain production without external inputs of nutrients, particularly P? And, what are the processes and mechanisms by which trees can maintain high rates of growth and production under conditions of extremely low available soil P?

Several questions address these research imperatives: 1) What are the relative extraction rates and uptake kinetics of P by trees at different depths in the soil as compared to annual crops? 2) What are the sources or pools of soil P tapped by trees and what are the mechanisms? And, 3) What are the P transformations in the soil as a result of litterfall and decomposition and how does that affect P availability?

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PHOSPHORUS DYNAMICS IN SHIFTING CULTIVATION SYSTEMS IN THE AMAZON

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Introduction

Phosphorus is a key limiting nutrient in the Amazon as well as in most other humid tropical forest ecosystems where shifting cultivation is the most widespread form of human intervention. In spite of the extensive research conducted on this element, the available information on P cycling in shifting cultivation systems is limited. This is in contrast with the more extensive knowledge about P cycling in natural ecosystems (Jordan, 1985; Vitousek and Sanford, 1986) as well as about the response of crops and pastures to P fertilization in the Amazon (Le Mare, 1982; Valverde and Bandy, 1982; Toledo and Serrão 1982; Smyth and Bastos, 1985; Lopes et al, 1987; Smyth and Cravo 1990ab). Low-input systems such as shifting cultivation, crop/managed fallow sequences, legume-based pastures and agroforestry occupy an intermediate position between the extremes represented by the natural system on one hand, and intensively managed agricultural systems on the other. The purpose of this paper is to review information on P cycling in shifting cultivation in the Amazon and identify knowledge gaps.

A phosphorus cycle model for shifting cultivation

Phosphorus cycling between plants and soils is more complicated than that of other plant nutrients. Organic (P_o) and inorganic (P_i) soil pools are both important in the P cycle, unlike that of nitrogen, which is primarily organic in nature and that of potassium which is primarily inorganic. A cycling model is presented for the two main portions of the shifting cultivation process: the cropping period and the fallow period (Nye and Greenland, 1960).

The cropping period. A conceptual P cycle for shifting cultivation during the cropping phase is shown in Figure 1. There is no primary organic P source. All P originates in the mineral fraction of the soil. Vegetative growth creates organic pools. Aboveground forest biomass P, including standing biomass, litter and the root mat are transformed into three compartments upon slash and burn: ash, unburned biomass (partially charred leaves, branches, trunks and stumps) and P lost to the atmosphere as particulate matter. The ash (P_i) then follows a basically inorganic pathway incorporating itself rapidly into the available soil P_i pool, which can be defined by a standard soil test extraction. The unburned biomass P as well as the forest root biomass P decompose slowly into the soil available P_i pool and the microbial P_o pool which in turn flow to the slow and passive P_o pools with turnover rates of tens or hundreds of years (Parton et al., 1988).

The microbial P_o pool upon mineralization produces $H_2PO_4^-$ ions which join the available P_i pool. The available P_i pool is largely regulated by the soil P sorption capacity which fixes and releases available P_i. The two primary inorganic P_i pools represent the P contained in weatherable minerals or in occluded forms that are slowly released to the available P_i pool. In Oxisols and Ultisols these two inorganic pools are thought to be very recalcitrant and of low solubility; therefore the fixed P_i and the microbial P_o pools are the ones that probably control available P_i. In contrast the more extensively studied Mollisols and Vertisols of the temperate region have large quantities of reactive P minerals, making the weatherable P_i pool a major control of available P_i.

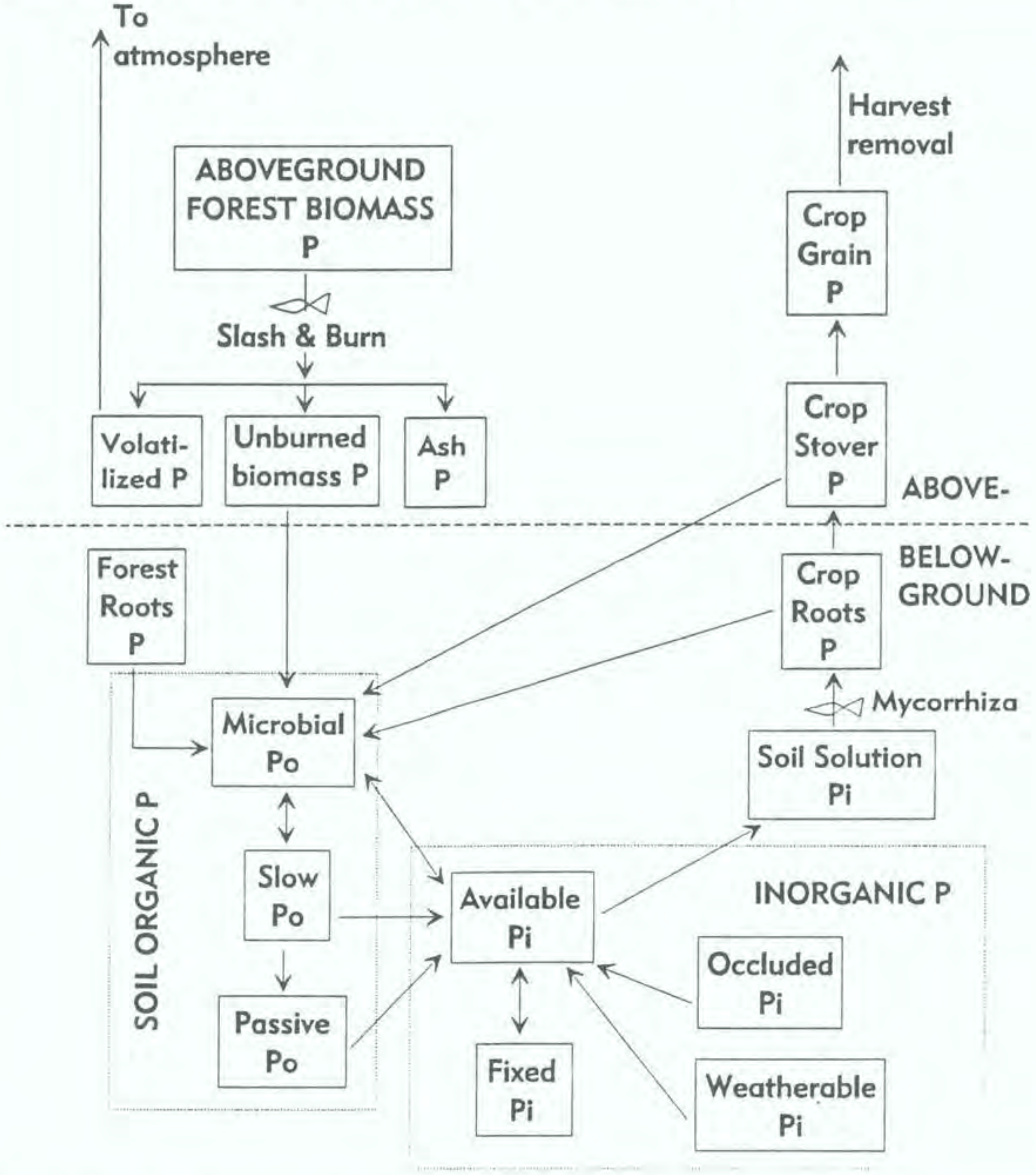

Figure 1. The P cycle during the cropping phase of shifting cultivation

The available P_i pool controls soil solution P_i composed of $H_2PO_4^-$ and $HPO_4^=$ ions which are the only forms of P that plants can utilize. Since P in the soil solution moves only by diffusion, increases in soil solution volume accessible to roots can be enhanced by mycorrhizal associations. Most crops grown immediately after slash and burn in the Amazon are highly mycorrhizal (Ruíz et al., 1989).

In cereals and grain legume crops P accumulates mainly in the grain, which is removed from the system at harvest. Rice, maize and sorghum concentrate about 70 percent of their P in the grain. If all the straw or stover is returned to the soil, only 30% of the P (Sanchez et al., 1989) is recycled.

Leaching losses are negligible, except in extremely sandy soils. Erosion losses are unlikely to be important in traditional shifting cultivation, but may be important when newcomers to the system practice shifting cultivation on steep slopes (Sanchez, 1976). Crop extraction, therefore, is the main P loss mechanism from the system.

The Fallow Period. After the cropping period, the field is abandoned and either a natural or managed fallow is allowed to grow. The P cycle during the fallow period is shown in Figure 2. The main P sources are probably the slowly decomposing, unburned above-ground biomass, large remaining forest roots, plus recycled crop residues. Ash input is not relevant. There are no large losses from the system in terms of crop harvest removal since only occasional fruit harvesting may take place. The below-ground reactions involving the various organic and inorganic P pools continue, with mycorrhizae facilitating P uptake by the fallow vegetation, perhaps more so than before, since key species of leguminous fallows have been found to have a higher percentage of mycorrhizal infection than annual crops (Ruíz et al., 1989). Phosphorus then accumulates in the fallow above-ground biomass and some of it is recycled back to the soil as above-ground litter. Below-ground root turnover also recycles P back to the soil.

Soil phosphorus pools

Data limitations. There is no data set available in the literature that provides quantitative information on the size of the pools and the magnitude of fluxes shown in the model described in Figures 1 and 2. The following section provides examples of some of the data available, drawn largely from the authors' research in Ultisols near Yurimaguas, Peru and Oxisols near Manaus, Brazil.

Available P_i. Topsoil available P determined by standard soil test procedures is the most common pool measured. Cochrane and Sanchez (1982) estimate that 90 percent of the soils in the Amazon have available topsoil P_i values lower than 7 mg kg^{-1} by the Bray II method (Table 1). This value is below the critical level, 8 to 15 mg kg^{-1}, associated with most crops (Marín 1977; Smyth and Cravo 1990b).

Available P_i in shifting cultivation systems has been estimated at different soil depths, ranging from as little as 3 cm (Ewel et al., 1981) to as deep as 100 cm (Russell, 1983). For P cycling, soil P pools should be measured as deep as the majority of fine roots penetrate, which is on the order of 50 cm in Oxisols and Ultisols of the Amazon (Szott 1987, Scholes and Salazar 1989). Ignoring subsoil available P grossly underestimates the size of this pool. For example Russell (1983) measured 35 kg P ha^{-1} in the top 20 cm of an Ultisol from Jari, Brazil but 103 kg P ha^{-1} in the top 100 cm. Likewise, we measured 6 kg ha^{-1} of available P_i in the top 15 cm of an Ultisol near Yurimaguas but a total of 13 kg ha^{-1} in the top 45 cm where most of the root development is concentrated.

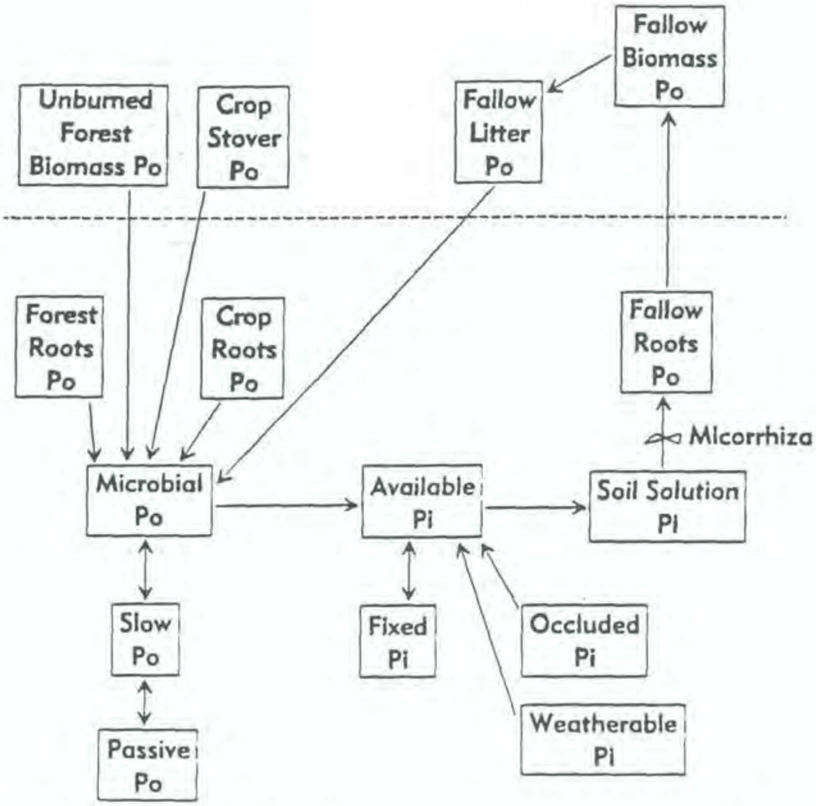

Figure 2. The P cycle during the fallow phase of shifting cultivation

Table 1. Phosphorus status of Amazonian soils (Cochrane and Sanchez 1982, Cochrane et al.1985)

Topsoil data (0-20 cm)	Million hectares	% of Amazon
Available P (Bray II):		
<-3 mg kg^{-1}	277	57
3-7 mg kg^{-1}	159	33
> 7 mg kg^{-1}	48	10
P Fixation (FCC):		
High	77	16
Low	406	84

A further important aspect of available P_i measurements is the quantitative difference in values among extraction procedures. Although such differences are not of orders of magnitude, they are relevant. Table 2 compares the values obtained on the same Oxisol by Mehlich 1 extraction, (also called double-acid or North Carolina extraction), the modified Olsen, and Bray I methods. Mehlich 1 is the routine procedure used in most soil test laboratories in Brazil; modified Olsen is used in Peru and many other countries while the Bray I method is used in Colombia. In Oxisols and Ultisol, the Mehlich 1 method gives consistently lower values than the modified Olsen, while Bray I gives the highest values. All methods are sensitive to the increase in available P_i with P fertilization (Table 2). It is also interesting to observe that the differences between methods decreased with continuous cultivation, (Table 2). If a choice of methods is possible at a given research site, we feel the modified Olsen method is preferable for budgeting P pools in Oxisols and Ultisols. Bray I tends to give very high values, while Mehlich 1 values are known not to be realistic when phosphate rock is applied (Sfredo et al., 1979; van Raij, 1978). Furthermore, the modified Olsen extraction can be used for simultaneous determinations of potassium and micronutrients and is also involved in the proposed sequential method of P fractionation of Hedley and co-workers (1982) which they call "bicarbonate-extractable P". Unfortunately few conversion factors from one method to the other are available (and are unlikely to exist across different soils, ed.). Some examples are shown in Smyth and Cravo (1990b) for Oxisols of Manaus.

Table 2. Available P_i values determined by different extraction methods in a clayey Oxisol near Manaus as a fraction of superphosphate applied shortly after burning (1 month) and its residual effect after 4 years (8 crops). Soil depth 0-15 cm, bulk density 1.04 Mg m^{-3}. Source: Smyth and Cravo (unpublished).

		Available Soil P measured by					
P Fertilizer applied	Crop No.	Mehlich 1	Modified Olsen	Bray I	Mehlich 1	Modified Olsen	Bray I
kg ha^{-1}		mg kg^{-1}			kg ha^{-1}		
0	1	3.0	7.4	13.2	5	12	21
	8	4.2	5.1	5.5	7	8	9
176	1	45.2	55.4	111.2	71	87	174
	8	16.5	12.5	18.7	26	20	29

Soil test values are concentration estimates and must be corrected for bulk density in order to express them in kg ha^{-1}, the unit used in the model. Bulk density with depth is seldom measured; furthermore topsoil bulk density changes during the course of shifting cultivation (Alegre et al., 1986). Estimates from the literature might be used as a first approximation, but actual bulk density determinations must be done at specific sties. Bulk density is highly correlated with soil texture, and this property has very high spatial variability in the humid tropics. Conversions from concentration values determined in the laboratory to mass values, therefore should be done with on-site bulk density determinations.

Soil solution P_i. Only orthophosphate is known to be taken up by plant roots. Soil solution measurements are unreliable because of large microspatial and temporal variability and the analytical precision required to detect very low concentrations.

Fixed P_i. Unlike high base status soils of the temperate region where most P cycling studies have been conducted, P fixation by iron and aluminum hydroxide surfaces is a major process controlling P cycling in most Oxisols and Ultisols. Although some degree of P fixation occurs in all such soils, the magnitude of this process is highly correlated with topsoil clay content in Oxisols, Ultisols and oxidic families of Alfisols and Inceptisols because iron and aluminum hydroxides are located in the clay fraction (Lopes and Cox, 1979). Phosphorus fixation is commonly estimated by P sorption isotherms as the amount of inorganic P added to reach a specific level of soil solution P_i (Fox and Kamprath, 1970). An example of the range found in soils representative of extensive areas in the Amazon is shown in Figure 3, where the P sorption isotherm of a clayey Oxisol near Manaus (82% clay) is compared with that of a sandy loam Ultisol near Yurimaguas (10% clay). In order to reach 0.1 mg kg^{-1} soil solution P, a level considered sufficient to most plants, about 420 kg P ha^{-1} need to be added to the high P-fixing Manaus Oxisol while only about 34 kg ha^{-1} will suffice for the Yurimaguas Ultisol.

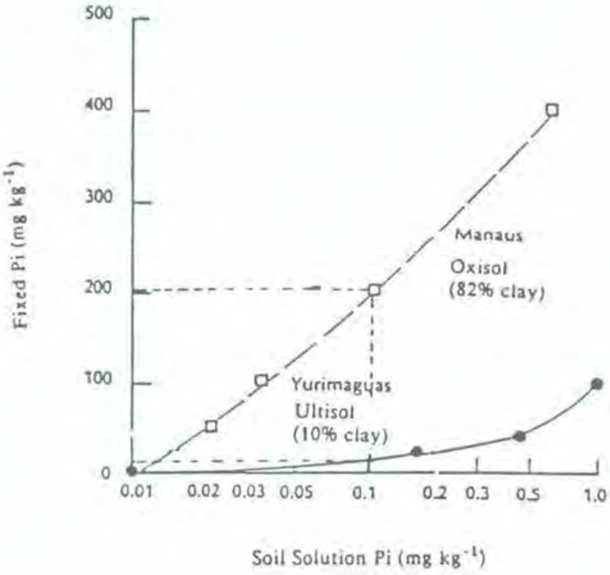

Figure 3. Contrasting P fixation capacities of two Amazonian soils determined by the Fox and Kamprath (1970) method. Source: Lopes et al. 1987

The relative abundance of soils with high and low P fixation in the Amazon is shown in Table 1, taken form the land resource study by Cochrane et al., (1985). This table is based on the fertility capability classification system criteria for high P fixation: topsoil clay content higher than 35% and a sesquioxide/clay ratio greater than 15% (Buol et al., 1975; Sanchez et al., 1982). Fortunately most of the Amazon soils do not suffer from high P fixation; therefore fixed P_i levels may be more commonly represented by the Yurimaguas curve in Figure 3 than by the Manaus curve. Nevertheless, 77 million hectares of the Amazon definitely has high P fixation capacity represented by the Manaus curve.

Phosphorus mineralized from the microbial P_o pool or released from other pools goes into the available P_i pool and some is quickly fixed by the sesquioxide surfaces. Fixed P is slowly released back to the available P_i pool and thus becomes available to plants. The fixed P_i pool therefore is not only a sink, but a major regulator of P dynamics in Oxisols and Ultisols. The relative size of this pool among different soils can be estimated by the Fox and Kamprath (1970) method at a given level of soil solution P_i.

Weatherable P_i and occluded P_i. The amount of P-bearing primary minerals, apatite, variscite, strengite, among others, is usually very low in Amazon soils, since most were developed from pre-weathered sediments. Occluded P_i consists of inorganic P compounds wrapped in sesquioxide coats that make them inaccessible to dissolution unless low oxidation-reduction potentials promote reduction of iron to the ferrous form. Localized anaerobic microsites may make temporary reduction possible in well drained soils. Although the rate of dissolution of P from these pools is very low, weatherable and occluded P_i may make important contributions to P cycling in the long run. No direct measurements of these two compartments are available from Amazon soils. These pools, however, may constitute the ultimate reservoir of P in these soils. Gross estimates of their size could be arrived at by subtracting total P_o, fixed P_i and available P_i from total P determinations.

Organic P soil pools. Organic pools are believed to account for 60 to 80 percent of the total soil P in Ultisols and Oxisols (Sanchez, 1976). This is in contrast with 20 to 50 percent found in glaciated soils from the northern temperate region. Direct measurements of P_o in Amazon soils are lacking.

Soil P_o is believed to consist of several functional pools, microbial P_o, slow P_o and passive P_o; but the estimation of the size of such pools is fraught with methodological difficulties. These three pools release P into the available P_i pool through mineralization but the major contribution is believed to come from the microbial P_o pool.

The microbial P_o pool may be the main source of available P_i but it is also capable of immobilizing P_i to meet microbial requirements. The slow P_o pool may exist in the organic matter stabilizing macroaggregates while the passive P_o may be found inside microaggregates (Elliott, 1986). Since iron and aluminum hydroxide coats largely control soil aggregation in Oxisols and Ultisols, there is likely to be a strong interaction between these P-fixing agents and the two less labile organic P pools.

Methodological advances need to be made before organic P pools can be estimated with a reasonable degree of reliability. Microbial P estimations by fumigation-extraction techniques in Oxisols and Ultisols are presently unreliable because of the P-fixing characteristics. Estimation of the size of the other two pools, could perhaps be done with the method of Hedley and co-workers (1982).

Total soil phosphorus. Total P contents in soils generally decrease with increasing stage of weathering (Westin and de Brito, 1969). Topsoil total P values average 6000 kg ha^{-1} in the United States Midwest and 1000 kg ha^{-1} in Ultisols of Southeastern United States (Olsen and Englestad, 1972). Mean total P contents in a samples of Amazon topsoil of 1268 Kg ha^{-1} were in the range of those found in the southeastern United States (Table 3). Subsoils contain even larger P stocks in spite of low concentrations due to their greater depth (Table 3).

Table 3. Total P content in Amazon soil samples, collected by Marbut and Manifold (1926) assuming a mean bulk density of 1.15 g cm^{-3}

Layer	No. of profile	Depth (cm) mean ± sd	kg P/ha mean ± sd	kg P/ha range
Topsoils	12	27 ± 8	1268 ± 1183	456 - 2283
Subsoils	14	100 ± 31	4708 ± 4307	304 - 11413

Extrapolations from Table 3 suggest a range of total P in Amazonian soils of the order of 700 to 11000 kg P ha^{-1} for the top 50 cm. The lower values may represent the most weathered soils and the higher value alluvial soils. A sandy loam Ultisol from Yurimaguas has the following total P values: 173 kg P ha^{-1} for the 0-15 cm layer, and a total of 786 kg ha^{-1} for the top 60 cm (Szott 1987).

It is hypothesized that the bulk of total P in the soil is in the passive P_o pool as well as in the occluded and weatherable P_i pools. This assertion needs quantification in order to understand the nature of these ultimate reserves of P in soils of the Amazon.

Biomass phosphorus pools

Above-ground forest biomass P levels have been extensively measured and range from 2 to 290 kg ha^{-1} in humid tropical forested ecosystems (Vitousek and Sanford, 1986). Above-ground biomass P levels in Oxisols and Ultisols fall within a narrower range: 21 to 101 kg ha^{-1} for primary forests or secondary forest fallows older than 10 years (Table 4).

Root P determinations are rare in the literature. Values for primary forests range from 5 to 69 kg P ha^{-1} (Table 4). The high value is for Spodosols where root P content is twice that held in above-ground biomass in these extremely infertile, sandy soils. In Oxisols, most of the biomass P is above-ground (Table 4). The same relationship was found in an 11-year old forest fallow on an Ultisol of Yurimaguas which contained 46 kg P ha^{-1} above-ground and 7 kg P ha^{-1} below-ground. Total soil P stocks, therefore, far exceed the biomass P stocks. This calls to question the common belief that most of the P in humid tropical ecosystems is held in the vegetation.

Such assertions are usually based on comparing topsoil available P_i, not total soil P at rooting depth, with total biomass P_o.

Table 4. Forest biomass P levels in several humid tropical forest ecosystems.

			Above-ground		Below-ground	
Location and (references)	Soil	Forest Age	Biomass	P	Biomass	P
			Mg ha^{-1}	kg ha^{-1}	Mg ha^{-1}	kg ha^{-1}
Manaus, Brazil (1)	Oxisol	Primary	406	59	32	5
San Carlos, Venezuela (2,3)	Oxisol	Primary	264	31	33	20
	Spodosol	Primary	185	32	132	69
Carare, Colombia (4)	Oxisol	Primary	184	27	nd	nd
		16 yr	203	55	nd	nd
		5 yr	68	22	nd	nd
		2 yr	19	16	nd	nd
Yurimaguas, Peru (5)	Ultisol	11 yr	70	46	6	7
		2.4 yr	16	20	nd	nd
		1.4 yr	12	12	nd	nd
		0.7 yr	5	4	nd	nd
Yangambi, Zaire (6)	Ultisol	18 yr	142	101	33	nd
		8 yr	133	46	21	nd
		5 yr	96	31	22	nd
		2 yr	12	20	8	nd
Tai Forest Côte d'Ivoire (7)	Ultisol	15 yr	78	21	nd	nd
		6.5 yr	38	14	nd	nd
		4 yr	22	10	nd	nd
		2 yr	14	7	nd	nd
		1 yr	9	5	nd	nd

1. Klinge et al., 1975; **2**. Jordan 1989; **3**. Henem 1979 from Vitousek and Sanford 1986; **4**. De las Salas 1978; **5**. Szott 1987; Smyth, Palm and Alegre - unpublished; **6**. Bartholemew et al., 1953; **7**. Van Reuten and Jansen 1987.

Fluxes upon clearing and burning

The fate of biomass P upon slashing and burning is illustrated with data from an 11-year old secondary forest from Yurimaguas (Table 5). Upon slashing and burning the above-ground biomass P content of 46 kg ha^{-1}, 10 Kg (22%) of the P was recovered in the ash. An estimated 22 Kg (48%) remained as unburned above-ground biomass. The available Pi content of the topsoil increased 15 kg ha^{-1}, more than that provided by the ash suggesting that roots and fractions of organic debris may have burned and contributed to this increase. Of course inaccuracies in the methodologies can also contribute to this discrepancy.

Table 5. Phosphorus transfers upon burning an 11 year old fallow in Yurimaguas (Smyth, Palm and Alegre, unpublished).

	P content in pool	
	kg ha^{-1}	%
Before burning:		
Total above-ground biomass	46	100
Combustible biomass[a]	24	52
After burning:		
Non-combustible biomass[b]	22	48
Ash from burning	10	22
Increase in topsoil (0-15 cm)	15*	33
Unaccounted for	9-14	20-30

* Modified Olsen extraction

[a] Calculated from P contained in leaves, small branches, and forest floor litter.

[b] Amount of P contained in trunks and large branches, the majority of which did not burn.

Assuming this last increase represents the true contribution of the ash, and other quick release processes then about 20 percent of the biomass P prior to burning is unaccounted for. This fraction is probably swept away from the site as particulate matter in the fire column since there are no obvious mechanisms for volatilization of P. Although this may not constitute a net loss to the ecosystem because the particulate matter will be eventually deposited elsewhere, it is a loss to the site. Ewel et al., (1981) reported a loss of 51% of the above-ground biomass P (11 kg P ha^{-1}) right after burning in Costa Rica.

Ash P_i. Ash production and its rapid incorporation into the topsoil by rains is the first transfer of P from the vegetation to the soil upon slash and burn. Ash production levels vary with moisture content of the vegetation and with the thoroughness of the burn. Phosphorus inputs in the ash vary with location but change little with the type or age of fallow at one location. This is shown in Table 6 where ash P contents are considerably lower on Oxisols near Manaus than Ultisols of Yurimaguas. Ash production was similar after burning a primary forest near Manaus and a 12-year old forest fallow near Yurimaguas. Little differences were observed at Yurimaguas among fallows of different age, except in one case where the burn was very poor because of rain during the burn. *Pueraria phaseoloides* (kudzu) managed fallows produced similar ash contents to that of secondary forest fallows at Yurimaguas (Table 6).

Unburned above-ground biomass. Given the incomplete nature of many burns in udic environments with a weak dry season, much of the biomass remains unburned. This material is believed to mineralize slowly but in 2 to 3 years most shifting cultivation fields are essentially devoid of forest remains except for a few hardwood stumps. In cases where selected logging prior to burning or removal of unburned logs and branches for firewood after the burn takes place, there is a net removal of P from the system.

Table 6. Phosphorus inputs in the ash (Smyth and Bastos 1984; Sanchez 1987; Smyth, Palm and Alegre, unpublished).

Vegetation burned	kg P ha^{-1}
Manaus (Typic Acrorthox):	
Primary forest	6
12-yr old forest	8
4-yr old kudzu	3
Yurimaguas (Typic Paleudult):	
25-yr old forest	17
17-yr old forest	6*
11-yr old forest	10
1-yr old kudzu	17
5-yr old kudzu/guinean grass	12

* very poor burn due to wet conditions

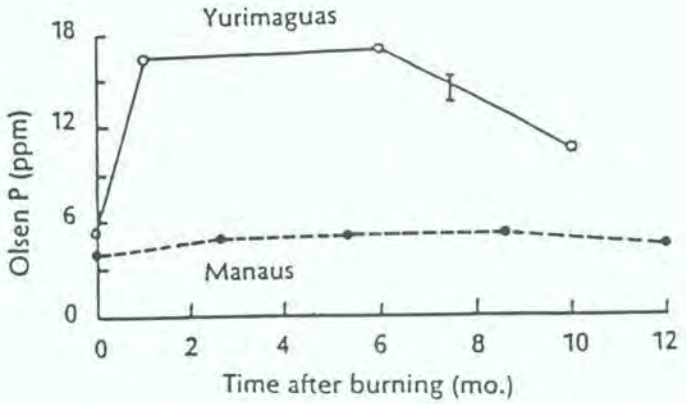

Figure 4. Topsoil available Pi dynamics during the first year after clearing and cropping without fertilization in Yurimaguas and Manaus. (Lopes et al. 1987)

Changes in Available P_i. Available P is the only pool for which there is sufficient data available about dynamics. Topsoil available P generally increases right after burning in response to the ash input and decreases during the cropping period (Seubert et al., 1977; Sanchez et al., 1983; 1985; Falesi 1976; Silva, 1981; Russell, 1983; Adedeji, 1984; Ramakrishnan, 1989; Andriesse and Koopmans, 1984; Stromgaard 1984). A definite increase in available P was evident in Yurimaguas, exceeding critical levels of 15 ppm for 6 months before a decrease started. In contrast small changes were observed in Manaus, with available P levels never rising above critical levels (Figure 4). Two factors account for this difference: 1) higher ash P_i inputs in Yurimaguas (Table 6), and 2) lower P fixing capacity of the Yurimaguas Ultisol as compared with the Manaus Oxisol.

Cropping removes considerable quantities of P from the system, since most of the P accumulated by grain crops is the grain. Sanchez and Benites (1987) report that 20 kg P ha^{-1} was removed in rice and cowpea grain in 7-crops during 3 years without fertilization. Topsoil available P levels therefore decline with time reaching values way below the critical level for such crops. Phosphorus fertilization is then needed for continuous cultivation, in order to replace what was removed or fixed. The decline pattern in available P_i and its reversal with fertilization is shown in Table 7 with data from another experiment from Yurimaguas.

Table 7. Available P dynamics after burning in a rice-cowpea low-input system in Yurimaguas (Smyth, Palm and Alegre, unpublished)

Sampling time	Available P (Olsen) mg kg^{-1}
Before burning	3.7 ± 1.0
After burning	10.6 ± 3.0
After 1st crop (rice)	4.7 ± 0.8
After 2nd crop (rice)	3.8 ± 0.9
After 3rd crop (cowpea)	4.7 ± 3.0
Fertilized with 22 kg P ha^{-1}	
After 4th crop (rice)	18.1 ± 2.7

The depletion process. A preliminary calculation using data from the experiment shown in Tables 5, 6, 7 and upland rice - cowpea crop uptake data is presented in Table 8 as an indication of the P depletion process during the cropping phase of shifting cultivation under a low-input system. During the first year, 3 crops accumulated 15.5 kg P ha^{-1} but returned 40% of that amount back to the soil as straw and roots. During the second year, the next 3 crops yielded less, accumulating 12.2 kg P ha^{-1} and recycling about the same proportion back to the soil.

Topsoil available P_i levels, determined by the Olsen extraction fell below 10 mg kg^{-1} (20 kg P ha^{-1} in this soil with a bulk density of 1.3 g cm^{-3}) after the first crop. Phosphorus fertilizer was added after the third crop but no significant yield responses were obtained.

Rough estimates are made in Table 9 on the rates of biomass P mineralization, soil P release from the less labile organic and inorganic forms, and rates of P fixation of newly available P. We assume that 10% of the remaining above and below-ground biomass P was mineralized per year, that only 0.01% of the total topsoil P was released every year, that 30% of the newly available P was fixed every year. Subsoil P was ignored. There is no experimental basis for any of these assumptions.

Table 8. Dry matter production and phosphorus uptake and % recycling of an upland rice-cowpea rotation in Yurimaguas.

	Dry matter production				P accumulated				P recycled	
Crop Sequence	Grain	Straw[1]	Roots[2]	Total	Grain	Straw	Roots	Total		
	Mg ha^{-1}				kg ha^{-1}					%
Year 1										
1. Upland rice	1.9	2.3	0.8	5.0	4.4	1.6	0.7	6.7	2.3	34
2. Upland rice	1.2	1.4	0.5	3.1	2.8	1.0	0.5	4.3	1.5	35
3. Cowpea	0.6	1.2	0.7	2.5	2.1	1.5	0.9	4.5	2.1	53
Total	3.7	4.9	2.0	10.6	9.3	4.1	2.1	15.5	6.2	40
Year 2										
4. Upland rice	1.6	1.9	0.7	4.2	3.7	1.3	0.6	5.6	1.9	34
5. Upland rice	1.2	1.4	0.5	3.1	2.8	1.0	0.6	4.4	1.6	36
6. Cowpea	0.3	0.6	0.4	1.3	1.1	0.7	0.4	2.2	1.1	50
Total	3.1	3.9	1.6	8.6	7.6	3.0	1.6	12.2	4.6	38

[1] Based on grain/straw ratios of 0.82 and 0.52 for rice and cowpea respectively. Cowpea pods ignored (from Benites and Sanchez 1989).

[2] Based on grain/fine root biomass ratio at anthesis of 2.27 for rice and 0.83 for cowpeas (Scholes and Salazar, 1989).

[3] Calculated from mean P contents of 0.23, 0.07 and 0.09% P for rice grain, straw and roots and 0.35, 0.13, 0.12% for cowpea grain, stover and roots, respectively (Benites and Sanchez, 1989).

Hypothetical calculations shown in Table 9 suggest a net input of 11.7 and 5.8 kg P ha^{-1} for the first and second year, respectively. The initial available P_i pools for year 1 and year 2 were 7.4 and 11.0 kg ha^{-1} respectively. Grain removal left a balance of 11 and 9.3 kg P ha^{-1}, respectively which closely coincides to measured available P data (Table 9). Such a close match may be a coincidence, but illustrates a depletion pattern of annual P additions to the system, which in the second year are less than removal. It also shows why no responses to P fertilization occurred in the second year; inputs from other sources appear sufficient.

No firm conclusions can be drawn from such calculations, except to suggest that these different P pools and their fluxes should be determined. In other instances where higher yields were produced, there is a clear evidence of P depletion and response (Gichuru and Sanchez, 1988).

The fallow period. Contrary to commonly held beliefs, the fallow period of the shifting cultivation system does not improve the (inorganic, ed.) fertility status of the soil. Available P_i levels in the topsoil are almost always lower than those found under a mature forest (Ramakrishnam and Toky, 1981, Szott and Palm, 1986; Sanchez et al., 1985). In fact, there is a marked decline in available P levels during the first few years of fallowing. Golley et al., (1974) attributed this effect to the transfer of P from the soil to the rapidly growing vegetation. Evidence of such buildup in above-ground biomass P is shown in Table 4 with examples from Colombia, Peru, Zaire and Côte d'Ivoire.

Table 9. Hypothetical calculations of topsoil (0-15 cm) available P_i inputs and outputs for the first and second year of the cropping phase of shifting cultivation with an upland rice-cowpea rotation without external nutrient inputs after slashing and burning an 11-year old secondary forest fallow in Yurimaguas, Peru. (Smyth, Palm and Alegre, unpublished)

	Year 1			Year 2		
Source of P	Pool size kg ha^{-1}	Turnover %/year	Flux kg ha^{-1}	Pool size kg ha^{-1}	Turnover %/year	Flux kg ha^{-1}
Ash P_i	10	100	10.0	0	-	0
Unburned biomass	22	10	2.2	19.8	10	1.9
Forest roots P_o	7	10	0.7	6.3	10	0.6
Total Soil P	173	0.01	0.02	173	0.01	0.02
Crop recycled						
straw	2.6	100	2.6	3.8	100	3.8
roots	1.2	100	1.2	2.0	100	2.0

	Year 1	Year 2
Subtotal inputs	16.7	8.3
Less net 30% P fixation of inputs	5.0	2.5
Net inputs	11.7	5.8
Initial available P_i[2]	7.4	11.0
Total Input	19.1	16.8
Grain harvest removal[3]	9.3	7.6
Balance (calculated)	9.8	9.2
Actual Available P[4]	9.4	10.0

[1] Crop 1 and 2 for year 1; crops 3, 4 & 5 for year 2
[2] Pre-burn for year 1; after 3rd crop for year 2
[3] Crop 1-3 for year 1; crop 4 and 6 for year 2
[4] After 3rd crop for year 1; after 5 crop for year 2

Where is this P coming from? It is hard to deplete topsoil available P much below 10 kg P ha^{-1} (5 $\mu g\ g^{-1}$) in soils such as those of Yurimaguas at the levels of cropping intensity reported. Szott's (1987) study of several fallow species suggest that biomass P is accumulating at the expense of a decrease in available P_i levels in layers as deep as 45 cm, where roots are active.

Which P pools contribute to this effect is also an interesting question. Rates of P uptake by fallow vegetation, suggested in Table 4, appear slower than P uptake by crops shown in Table 8. Time and a slower rate of P uptake by trees during fallow vs. short-term crops, plus nutrient cycling through the establishment of an effective litter layer in fallows may explain these differences. Furthermore, little if any P is extracted from a forest fallow as opposed to crop harvest removals. Consequently P, extracted from soil pools at a rate too slow to support crop growth, is accumulated in the forest fallow biomass as a savings account ready to be tapped by the shifting cultivator in the next cycle.

The problem of paramount concern is many areas of the humid tropics is that the length of the fallow period is rapidly shortening due to population pressures. Farmers, therefore are tapping this nutrient savings account before it can build up much capital in forest fallow.

Conclusions

This review outlines the cycle of P under shifting cultivation in the Amazon, and raises questions concerning rates and magnitudes of P transfers between soil and vegetation pools during the shifting cultivation cycle.

Phosphorus accumulated in the biomass of primary forest or tall secondary forest fallows in Oxisols and Ultisols of the humid tropics is about two orders of magnitude lower than the P in the soil to an average rooting depth of 50 cm. Due to the incomplete burning common in humid tropical areas with a short dry season, only about 20% of the above-ground biomass P is converted to ash, another 50% remains unburned and the remainder is apparently lost to the atmosphere (as fly ash). The remaining biomass decomposes slowly into available and unavailable inorganic P, microbial P, and slow and passive organic soil P pools. Phosphorus fixation by soil minerals is an important control on P availability in Oxisols and Ultisols. Weatherable and occluded inorganic P pools, although very slowly reactive, may play a major role in the long term. Quantitative determinations of these pools are fraught with methodological difficulties. Estimates of the various soil P pools vary with the extraction method used, depth interval considered and the bulk density conversion factor used.

Within this framework, several questions need to be answered to provide a basis for improved management of a degrading agricultural system. What is the long-term role of the inorganic P pools? What are the interactions between inorganic and organic pools? How does nutrient depletion during the cropping cycle proceed and from where does the P accumulated in the biomass during the fallow period come? Regardless of the answers to these questions, crop plants do extract more P from the system than are recycled back and P fertilization is likely to be needed in most Oxisols and Ultisols in the Amazon in order to sustain crop production under current land pressures.

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SPATIAL VARIABILITY OF EXTRACTABLE PHOSPHORUS IN AN AMAZON FOREST

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Introduction

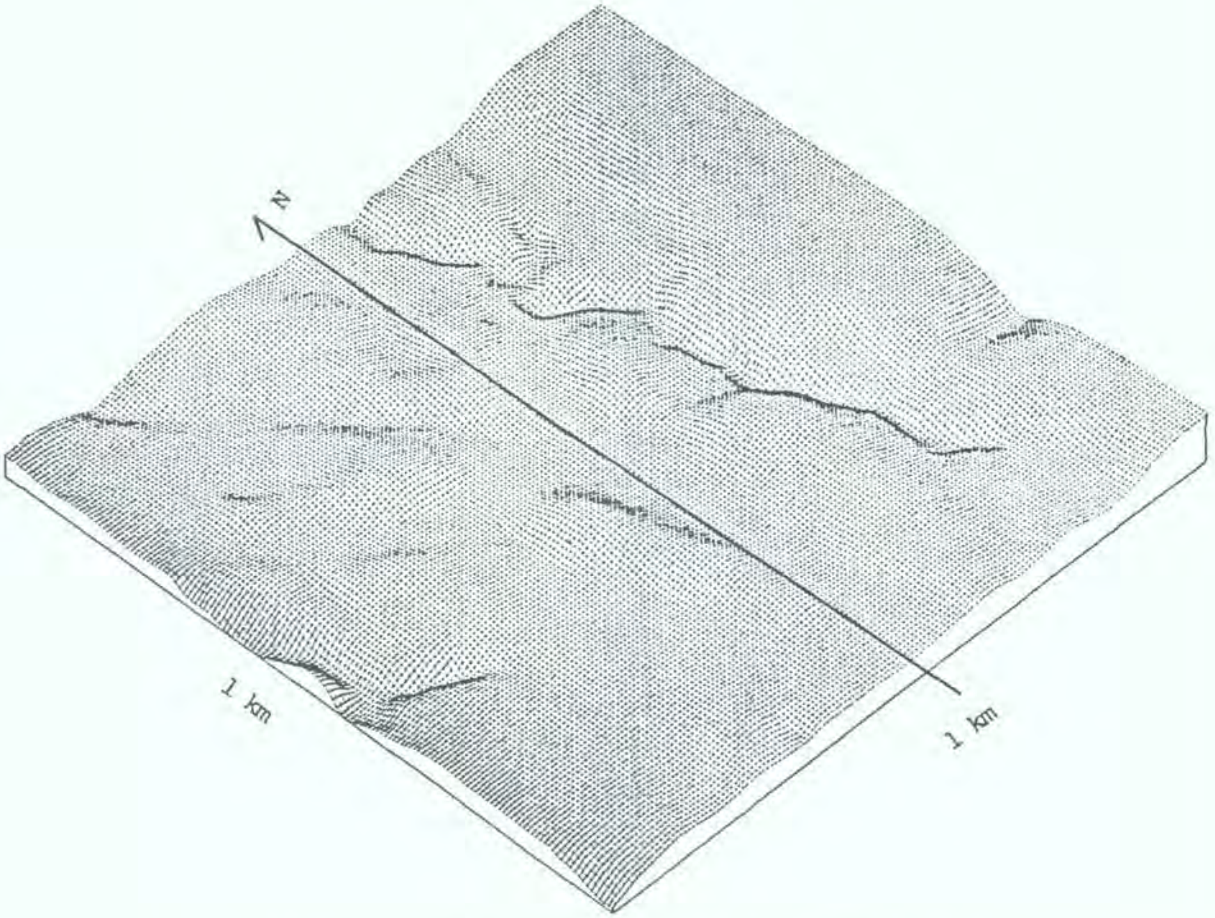
A multidisciplinary study is under way at a site within an Amazon rain forest near Manaus, Brazil to identify topographic, edaphic, and biological factors that may contribute to the distribution pattern of individual species of trees belonging to the Brazil nut family (Lecythidaceae). Since the Lecythidaceae family represents an important structural component of the Amazon forest (Mori and Prance 1987), a study of its basic ecology and biology should afford essential information for conservation management and prediction of possible consequences due to continued disturbance of the Amazon ecosystem.

An early component of this study involved the sampling of topsoil on a defined grid to examine the spatial variability of certain chemical and physical properties that may affect plant community structure. Because P is a major nutrient deficient in most Amazonian soils (Cochrane and Sanchez 1982), and because studies in other tropical forests have shown a correlation between soil P and floral distribution patterns (Gartlan et al. 1986, Newbery et al. 1986), there was considerable interest in determining the quantity of extractable P and its distribution across the study site. Although other soil properties were determined, this paper only emphasizes results obtained from the initial survey of extractable P.

Materials and methods

Located within a large forest reserve about 90 km north of Manaus, a 100-ha plot of terra firme forest (Prance 1978) has been set aside for studies on the ecology and biology of trees belonging to the Lecythidaceae family (Fig. 1). Soils of the region are generally classified as yellow, clayey latosols (Chauvel et al. 1987). The whole study plot has been partitioned into 20 x 20 m quadrats, with the corners of each quadrat marked with PVC posts numbered on a Cartesian coordinate system.

An initial survey of surface soil properties was performed over the western half of the plot (50-ha, 500 x 1000 m) by collecting topsoil (0 to 0.2 m) near each PVC post staked out in a 20 x 20 m grid (1326 locations). An additional 322 topsoil samples were taken at 2.5 m intervals on four 200 m transects that crossed major topographic features (Fig. 2). All soil samples were taken over a two month period in mid-1988; 26 one-km lines running north- south were defined with each line sampled randomly over time to minimize any temporal variability. The four transects were sampled after completing all 26 lines. The sampling routine at each location was as follows. After removing surface litter, a Dutch auger was used to obtain three cores within 0.3 m of each other. The three cores were mixed well before taking a composite soil sample for analysis. Before analysis, the soil was dried in a solar drier, crushed with a rolling pin, then passed through a 2 mm sieve. All analyses were conducted by the EMBRAPA-UEPAE agricultural research station in Manaus. Soil P was measured by the Murphy and Riley (1962) colorimetric method after extracting P with a mixed acid solution (0.05 M HCl + 0.05 M H_2SO_4, 1:10 soil to solution, 5 min shaking). Other soil properties measured were texture (pipet method), pH (both water and KCl), total N (micro- Kjeldahl), organic C (Walkley-Black), and exchangeable acidity, Ca, Mg, and K. For each property, an analysis of spatial structure was performed using semi-variograms (Trangmar et al. 1985) computed with geostatistical software from the University of Hawaii (Yost et al. 1988).



LECYTHIDACEAE PLOT - 45 deg view from SW

Figure 1. Three-dimensional relief of the 100-ha Lecythidaceae plot viewed from the southwest. Surface soil was studied on the half nearest the viewer.

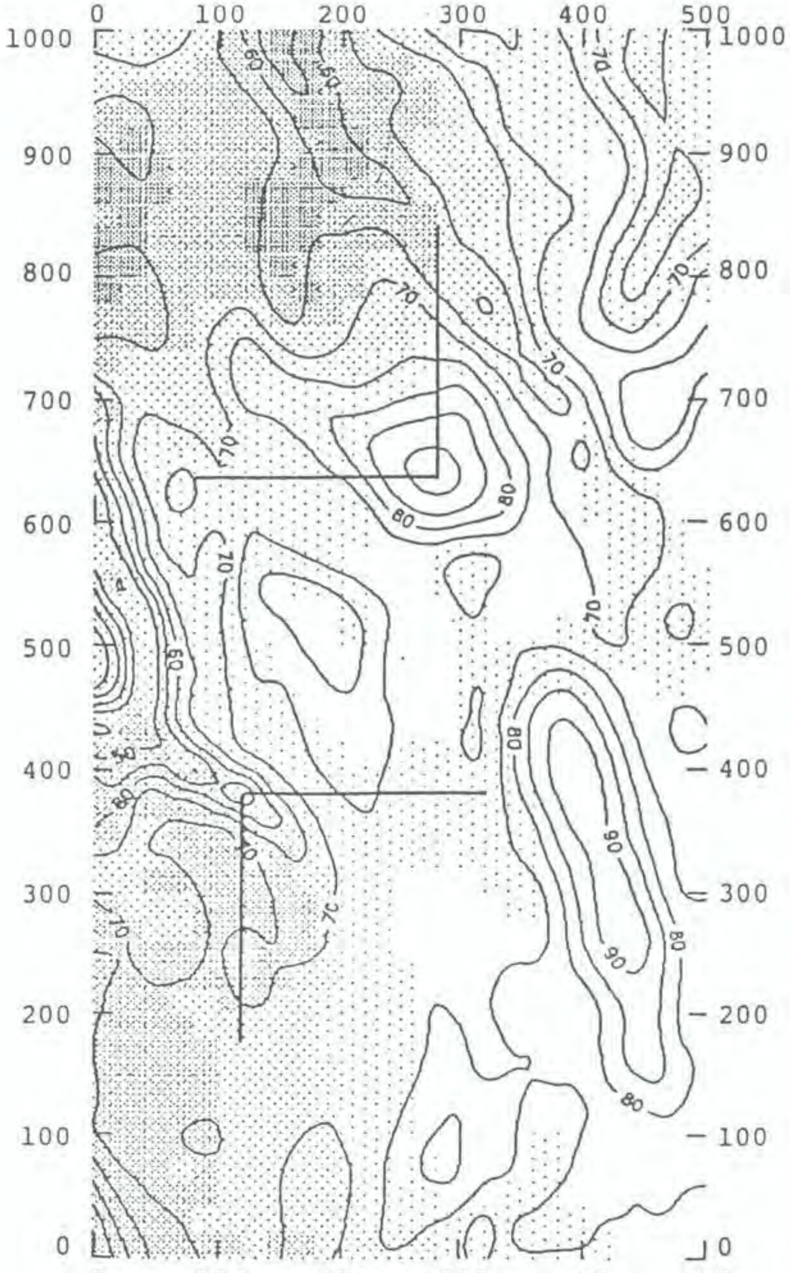


Figure 2. A topographic map of the 50-ha study plot with 5-cm contour levels. The two sets of orthogonal lines show the supplementary transects on which samples were collected at 2.5 m intervals. The dots show kriged estimates of percent sand; in order of increasing density, the shading corresponds to 6-20, 30- 39, 40-59, 60-79, and 80-94 % sand.

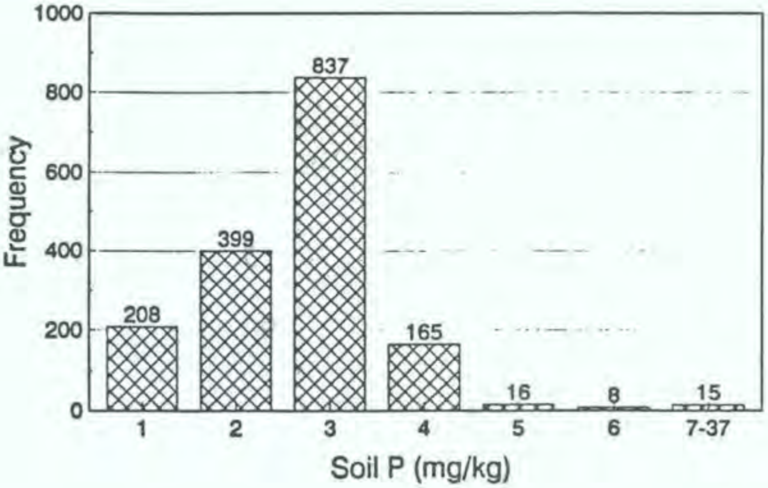


Figure 3. Frequency distribution of extractable soil P measured across the 50-ha plot.

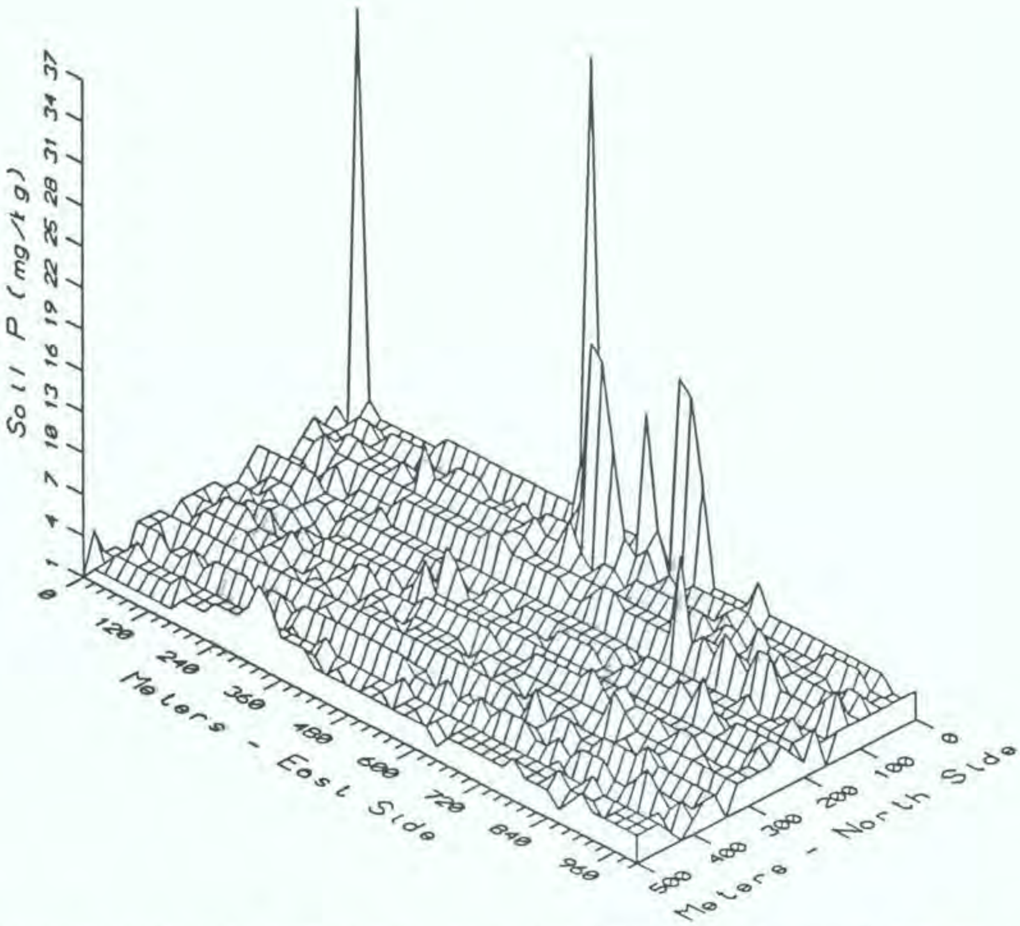


Figure 4. Three-dimensional diagram of the 50-ha plot and extractable soil P values in the surface 0.2 m, viewed from the northeast.

Table 1. Properties of soil samples (0 - 0.2 m) taken across the 50-ha forest site (n = 1648).

Soil property	Mean	Standard deviation	CV (%)	Range
P (mg kg^{-1})	2.7	1.68	61	1 - 37
K (mg kg^{-1})	20.3	9.36	46	2 - 126
Ca (cmol kg^{-1})	0.07	0.039	54	0.01 - 0.94
Mg (cmol kg^{-1})	0.06	0.027	44	0.02 - 0.52
Total N (g kg^{-1})	1.47	0.56	38	0.3 - 11.2
Organic C (g kg^{-1})	21.9	11.3	52	0.4 - 191
pH (KCL)	3.9	0.17	4	2.9 - 4.4
pH (water)	4.5	0.23	5	3.6 - 5.3
Sand (%)	36	22.4	62	6 - 94
Silt (%)	16	5.1	32	0 - 39
Clay (%)	48	18.9	40	4 - 82

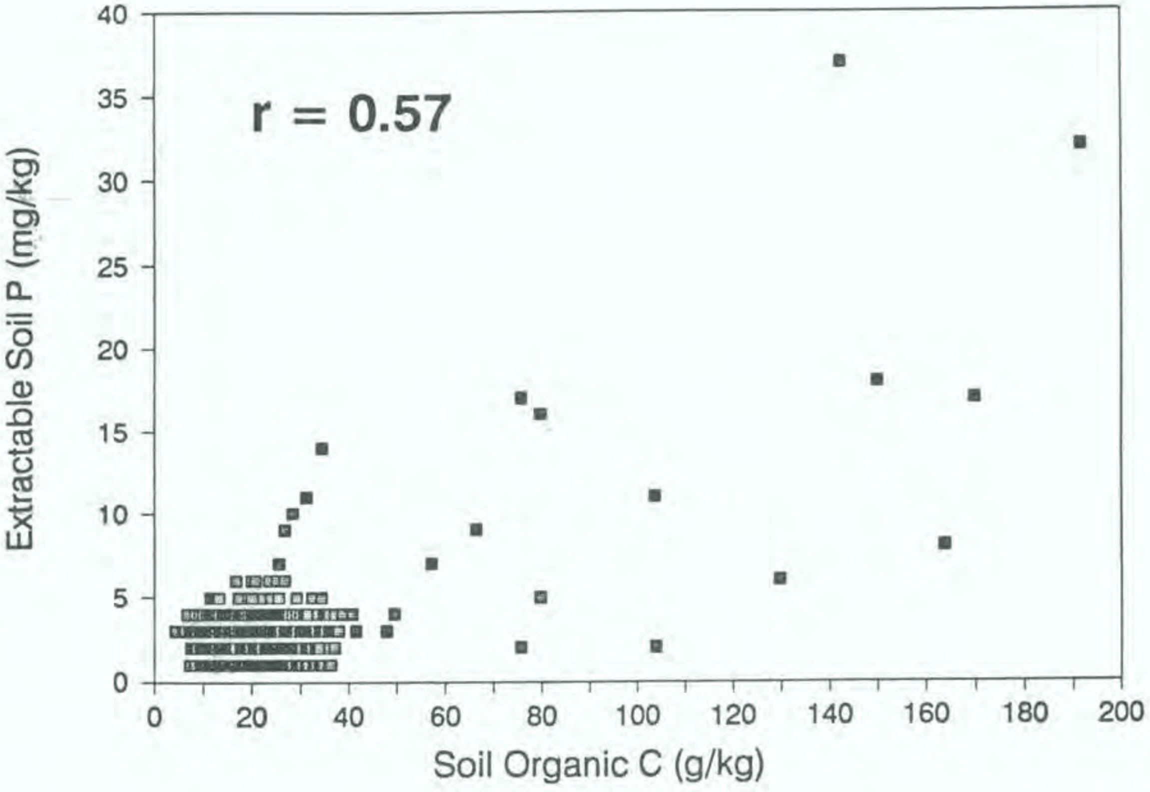


Figure 5. Relationship between extractable soil P and soil organic C for all locations.

Results and discussion

An intensive sampling of topsoil across the 50-ha plot showed quite conclusively the predominance of very low levels of extractable P (Fig. 3). Although the range in values was 1-37 mg kg^{-1}, only 2.4 % of the 1648 samples had values above 4 mg kg^{-1}. A three-dimensional perspective of P levels and distribution, viewed from the northeast, is shown in Figure 4. The few locations with values greater than 7 mg kg^{-1} were predominantly low lying wet areas; specifically low wet areas on the mid-west side of the plot and in the southwest corner. These poorly drained locations contained greater amounts of organic C which was correlated with extractable P (Fig. 5), an indication that the higher levels of P were not due to differences in parent material or soil forming processes but rather differences in degree of organic matter decomposition affected by drainage.

Clearly the higher soil P values found in poorly drained areas on the west side of the plot were not indicative of variation across the rest of the plot. After omitting values from the two wettest areas by dropping data for the first three lines from the west, the variability and range in soil P decreased substantially; the mean remained at 2.7 mg kg^{-1} but the standard deviation decreased from 1.68 to 0.97 and the range decreased from 1-37 to 1- 11 (n = 1495). Semivariogram analysis of the reduced number of grid samples as well as the transect samples revealed no quantifiable spatial relationship for soil P. The range in values was small, and most likely there was no real difference in a value of 1 versus 3 mg kg^{-1}.

Statistical results for all soil properties measured are given in Table 1. Other nutrients in addition to P were found to be quite low. Once again, most of the variation in nutrient levels, particularly Ca and Mg, could be traced to higher levels found in the poorly drained areas with a greater C content. The greatest degree of structured spatial variation was found for soil texture. For example, Figure 2 shows that the western corners of the plot were substantially more sandy than the rest of the plot (sand and clay were highly correlated, $r = -0.98$). There was no relationship between texture and extractable P.

Results from the intensive measurement of soil P across this small area of Amazon forest agreed well with estimates made for the whole Amazon basin by Cochrane and Sanchez (1982). After collating data from various locations in the Amazon, these authors determined that 90 % of the Amazon topsoils had extractable P values of 7 mg kg^{-1} or less and 57 % contained less than 3 mg kg^{-1}.

A working hypothesis guiding this 50-ha survey of topsoil properties was that the availability of limiting nutrients, P in this case, might affect the distribution of Lecythidaceae trees. Given, however, the low levels of P found throughout the site, it is unlikely that variation in P availability has affected plant structure to any significant extent, certainly not to the extent that Gartlan et al. (1986) and Newbery et al. (1986) demonstrated for forest sites in Cameroun where soils could be grouped into wider ranges of P availability.

Acknowledgment

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EFFECTS OF SOIL MICROSTRUCTURE ON PHOSPHORUS SORPTION IN SOILS OF THE HUMID TROPICS

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The cycling of P, more than that of any other major nutrient element, is controlled by inorganic chemical reactions. What are these reactions? What are the important controls? This paper reviews recent studies suggesting that the physical characteristics of soil constituents, as opposed to their chemical characteristics, may be more important than previously thought in determining patterns of P availability in acidic soils of the humid tropics.

Some understanding of the inorganic chemical reactions that affect P availability in acidic soils is prerequisite to discussion of the physical factors controlling them. Precipitation reactions have been touted as important but in fact the evidence suggests otherwise. Microprobe studies fail to reveal variscite or strengite as coherent grains in soils (Norris and Rosser 1983), and equilibrium and kinetic considerations suggest that these minerals would be unstable in soils except below pH 3.1 and 1.4, respectively (Bache 1963, Syers and Curtin 1989).

Instead, ligand exchange appears to be the only inorganic reaction of any significance in the acid soils of the humid tropics. The reaction consists of replacement of surface hydroxyls by the phosphate ligand. Three factors affect the rate of ligand exchange (per unit soil volume) and thus the P adsorption capacity of soils: (1) the amount of hydroxyl groups per unit surface area, (2) the extent to which the hydroxyl groups enter into ligand exchange, and (3) the total amount of reactive surface area in the soil. Reactive surface area is provided mainly by layer-silicate clays, by oxides and hydrous oxides of Al and Fe, and by highly disordered aluminosilicates such as allophane. All of these soil constituents are clays with regard to their physical properties and are referred to as "clays" hereafter.

The first factor, amount of hydroxyls per unit surface area, depends on the types of clays present in the soil, that is to say, their mineralogy. Fig. 1 shows P adsorption curves for the clays present commonly in soils of the humid tropics. Of these, allophane and ferrihydrite adsorb most strongly, largely because of their extremely large surface area. Goethite, an Fe oxide, adsorbs at an intermediate level, followed by gibbsite and hematite. Kaolinite, a 1:1 layer-silicate clay, adsorbs relatively little P because its specific surface area is low and because surface hydroxyls are present only along broken edges. The 2:1 layer-silicate clays, such as montmorillonite, adsorb hardly at all.

Methods can affect P adsorption values. Most researchers have used NaCl or KCl as the background electrolyte. Use of $CaCl_2$, however, may give very different results (Barrow 1989). The correct choice of electrolyte may be soil specific. Gillman and Bell (1978), for example, extracted solutions from six North Queensland soils (0-10 cm depth). Na was dominant in one of the six, K in two, Mg in two, and Ca in one. Length of reaction time is also problematic. As discussed later, adsorption is generally monitored for at most a few days. In fact, P continues to be taken up from solution for long periods, although at reduced rates. Nonetheless, the patterns in Fig. 1 indicate the large range in P adsorption capacity of the clays present commonly in soils of the humid tropics.

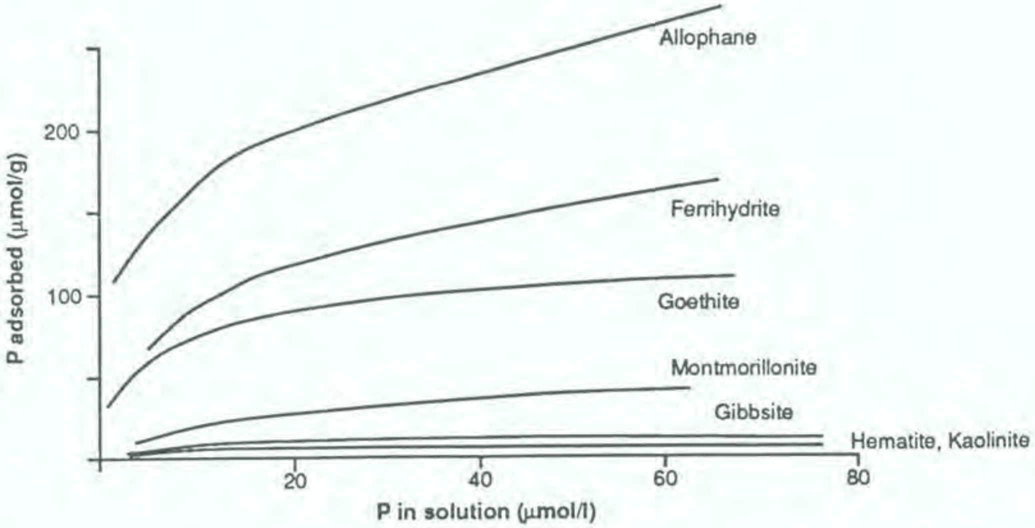

Figure 1. Adsorption curves for phosphate on various minerals (per g of mineral). Except for gibbsite and montmorillonite, all data are for 25 C, 16-h shaking, and 0.002 M $CaCl_2$ (Parfitt 1989). Data for gibbsite are for 0.1 M NaCl (Parfitt et al. 1977). Data for kaolinite and montmorillonite are for 0.02 M KCl and 2-day equilibration (Bar-Yosef et al. 1988).

The second factor, in effect the equilbrium constant for the ligand-exchange reaction, involves the nature of the material in which the hydroxyl groups occur. Most studies with infra-red spectroscopy and other techniques (e.g., Parfitt et al. 1975, Martin and Smart 1987, Parfitt 1989) suggest that, at least for the Fe oxides, P is adsorbed only as a binuclear complex (Fig. 2), the formation of which requires a certain minimum spacing between adjacent OHs. Closely paired OHs occur on only some of the various faces that each clay presents. Thus variations in the proportion of the total hydroxylated surface on which OHs are paired closely enough to permit formation of a binuclear complex might explain some of the variation in P adsorption capacity among minerals and between different preparations of the same mineral.

Figure 2. Formation of binuclear phosphate surface complex (after Mott et al. 1981).

Barrón et al. (1988) used differences in spacing of surface hydroxyls to explain variation in P adsorption in aluminous hematites of different crystal shapes. They were able to explain some of the variability on the basis of estimates of relative surface area accounted for by each crystal face. Possibly, their approach can be extended to other oxide types.

The third factor is total surface area of clays in the soil. Over soil sequences in which mineralogy is fairly constant, this surface area should be approximated closely by clay content.

Role of organic matter

Does organic matter (OM) sorb phosphate? In one sense the answer is "no". Hydroxyl groups attached to C atoms do not enter into ligand exchange; C-O-P linkages do form but the process requires enzymatic mediation (e.g., Stevenson 1986).

Soil OM forms metal complexes readily, especially with transition metals such as Al. Hydroxyl groups attached to the Al in such complexes can then enter into ligand exchange reactions, as shown in Fig. 3. This phenomenon has been little studied, although Gunjigake and Wada (1981) found that P adsorption capacity did increase with levels of pyrophosphate-extractable Al and Fe (thought to represent the humus-bound Al and Fe).

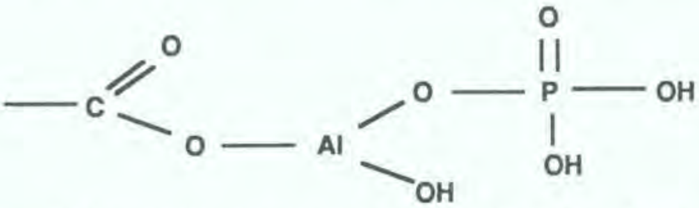

Figure 3. Schematic representation of phosphate surface complex with humus-bound Al.

In fact it is difficult to distinguish between allophane and Al-humus complexes and until recently soils containing such materials were lumped together as Andepts (Soil Taxonomy) or Andosols (FAO). The new Andisol order within Soil Taxonomy separates the soils with Al-humus complexes as Humands, and it will be interesting to compare long-term kinetics of P adsorption in Humands vs. the true allophanic soils.

Slow and fast uptake kinetics

The generalizations presented above are based almost entirely on short-term studies of P adsorption. In early work, uptake of phosphate from solution was monitored for perhaps 24 h, sometimes for as little as one minute (e.g., Parfitt et al. 1977). Later studies showed, however, that there are two parts to the uptake kinetics. At first the phosphate concentration drops rapidly. After a few minutes, however, the rate of uptake slows, then levels off at a steady but much slower rate (Fig. 4) that can continue for months.

Until about 1982, the so-called "slow reaction" was virtually unstudied. Barrow (1985) called attention to its importance, pointing out that the slow part of the reaction kinetics, not the fast part, was likely to govern availability of P to plants. Fortunately, recent data suggest that short- and long-term rates of uptake correlate well across a range of natural and synthetic minerals (Parfitt 1989). Consequently, the earlier studies are still useful, and the discussion presented above of factors controlling short-term rates of P adsorption can be expected to hold also for the long-term rates of P uptake from solution. In addition, the fast part of the kinetics may be worth studying further as a factor important in controlling the balance between immobilization and adsorption of P. The measurement of microbial P may also be affected. When P is released from microbial cells during $CHCl_3$ fumigation, the amount obtained with an extracting solution may be governed mainly by the fast part of the reaction kinetics.

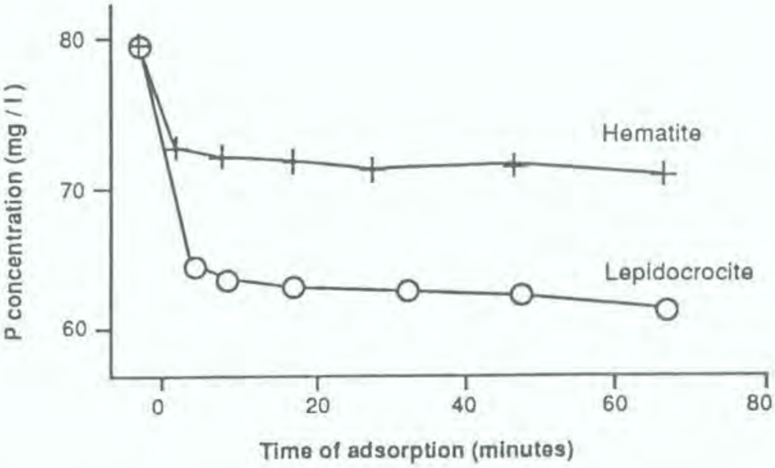

Figure 4. Uptake of phosphate from solution by lepidocrocite and hematite (after Madrid and de Arambarri 1985).

Some recent studies are helping to unravel the mechanisms responsible for the two-part uptake kinetics. Empirical studies of the temperature dependence of P uptake show that, under most conditions, the rate of long-term uptake increases with temperature (see Barrow 1985, Fig. 13). This is unexpected ligand exchange *per se* is an exothermic reaction (Barrow 1985, p. 217), so adding heat, a product of the reaction, should slow it down. In fact, this slowing is seen, but only under very restricted conditions. That the opposite is the case more commonly indicates that ligand exchange is not the only process at work.

There is now consensus that the "slow reaction" is caused by a diffusion process but there is little agreement about its nature. Barrow (1989) noted that the activation energy for the slow part of the P uptake kinetics is large, suggesting that solid- rather than liquid-phase diffusion was involved. Barrow's view is supported by other lines of experimental evidence that point toward solid-phase migration of the phosphate ion into the interior of clay crystals. Such diffusion would be most likely where atoms such as Si have previously migrated into the crystal, replacing the original Fe or Al and creating defects in the crystal structure. For example, Norrish and Rosser (1983) showed that P adsorption in Fe-rich soils correlates with their Al content, not with their Fe content, which suggests that it is not gibbsite but rather the degree of Al substitution in the goethite that controls P uptake.

Parfitt (1989) compared long-term P adsorption rates (30-d) for a synthetic preparation of goethite and for four minerals obtained nearly pure *in situ* (allophane, ferrihydrite, goethite, and hematite). The synthetic goethite was highly crystalline and equilibrated rapidly with phosphate. The natural goethite was more Al-substituted, appeared to be less highly crystalline, and equilibrated more slowly. These results were interpreted as implying solid-phase diffusion through crystal defects. Porosity, however, was not measured and would be expected to be higher in the Al-substituted, natural goethite than in the synthetic material. Thus, the slower equilibration of the natural goethite might also be explained by liquid-phase diffusion through its pore structure.

Other studies point to liquid-phase diffusion of phosphate to adsorption sites as the cause of the slow part of the uptake kinetics. Madrid and de Arambarri (1985) were able to explain the difference in 13-day adsorption by two Fe oxides (lepidocrocite and hematite) simply on the basis of the porosity of the crystals. This suggests that liquid-phase diffusion through fine pores into the interiors of crystals and microaggregates is responsible for the slow uptake kinetics. Likewise, Cabrera et al. (1981) showed that phosphate continued to react with lepidocrocite longer than with goethite. The lepidocrocite formed aggregates of small crystals with pores between them, whereas the goethite was present as single crystals.

The most direct evidence for liquid-phase diffusion comes from studies by Willett et al. (1988) of microaggregates of synthetic ferrihydrite. Autoradiography (Fig. 5) showed that labelling was restricted to the exterior of the microaggregates, but that the depth of penetration of the label increased through time. IR spectra showed no evidence for a change in the nature of the phosphate bond through time, which helps rule out the possibility that P was replacing Si that had previously substituted for Fe or Al in the crystal structure.

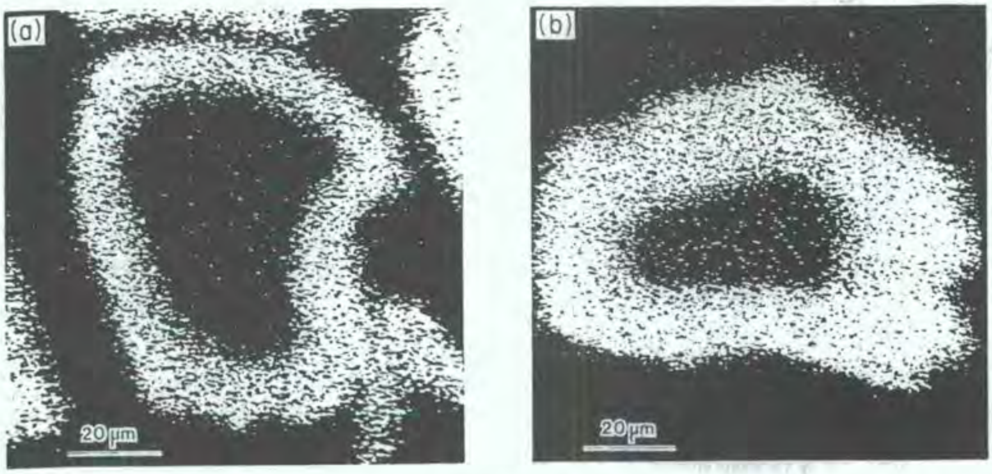

Figure 5. Distribution of phosphate in microaggregates of ferrihydrite (from Willett et al. 1988): (a) after 1 h, (b) after 1 d.

Results for allophane and other poorly ordered clays are hard to interpret. Solid-phase diffusion is unlikely to explain the slow reaction because the spherule walls are only 3-4 atoms thick (Parfitt 1989).

Structural properties of the allophane spherules do seem to affect adsorption. Adsorption seems to occur preferentially in conjunction with defects and/or pores in the allophane spherules (Parfitt 1989), and P adsorption capacity of allophane tends to increase with the degree of substitution of Fe for Ae (Parfitt and Hemni 1980). In any case, allophane spherules flocculate very strongly; so liquid-phase diffusion may well be involved also. Moreover, since allophane is gel-like, allophanic surfaces are constantly being buried and uncovered as the allophane flows and coalesces. Thus the adsorbed phosphate may stay put and the *site* of adsorption may migrate.

The importance of solid- vs. liquid-phase diffusion thus appears to vary with the minerals involved. Dispersion methods, such as ultrasonication, could perhaps be used in conjunction with autoradiography to gain insight into the relative importance of the two modes of diffusion. If P adsorption/desorption are limited by rates of liquid-phase diffusion into the interiors of micro-aggregates, then dispersion of those micro-aggregates should greatly speed desorption. On the other hand, if a solid-state diffusion process limits P adsorption/desorption, then dispersion should have little effect. The experiment could be done first with a defined artificial material, such as the synthetic ferrihydrite aggregates used by Willett et al. (1988). If the ultrasonic dispersion in fact speeds desorption, autoradiography after the ultrasonication could then serve to indicate whether the ultra-sonication had successfully dispersed the labelled exteriors of the aggregates. It is unclear whether the approach would work for highly porous crystals which might be very difficult to break up, but the autoradiography would indicate unequivocally whether dispersion had been achieved. If the overall approach could be shown to work with synthetic materials, it might then be useful for judging which diffusion process is more important in whole soil.

P budgets: inorganic vs. organic reactions

The above papers dealt with well defined, synthetic materials, not whole soil. In recent work with an Oxisol and two Andepts, Parfitt et al. (1989) compared long-term P adsorption (218 d) with P uptake by microflora and clover. Uptake by the microflora ranged from 5-25 mg P kg^{-1}; uptake by the clover ranged from near 0, at low levels of P addition, to 40-60 mg P kg^{-1} of soil at high levels. Phosphorus adsorption was much higher: 50 to 400 mg P kg^{-1} soil depending on the amount of P added. Of the three soils, the Oxisol gave the lowest P adsorption and the highest uptake by microflora. McLaughlin et al. (1988) measured fluxes of P to plants and microflora, along with uptake of P into relatively unavailable organic and inorganic forms. Even though the soil they studied did not adsorb P strongly (a Calci-xerollic Xerochrept), inorganic reactions were estimated to remove about 1/3 of the P added as organic residues, and about 3/4 of the P added as fertilizer. Experiments combining MacLaughlin's and Parfitt's approaches in more strongly P adsorbing soils seem warranted.

Lastly, it should be noted that it may be difficult to rule out microbiological activity as the explanation for the slow part of the uptake kinetics, especially in experiments with whole soil. Some, but not all, investigators mention adding a biocide to retard microbial uptake of P. Even if biocide was added, its effectiveness should not be assumed; plate counts and respiration measurements are easy and would settle the issue. Also, many biocides themselves adsorb strongly on mineral surfaces, a process that might affect P adsorption.

Conclusion

Researchers have concluded that the physical structure of the soil, not just its chemical composition, must be understood in order to predict rates of chemical and biological processes (see recent reviews by Duxbury et al. 1989; Theng et al. 1989). The studies reviewed here suggest that physical arrangement of reactive surfaces in the soil is also important in controlling P transformations and availability. It thus seems likely that techniques based solely on use of chemical extractants will never adequately explain differences in P availability between soils or through time. New approaches are needed that deal with the physical arrangement, not just chemical composition, of soil constituents.

Acknowledgement

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PHOSPHATE IMMOBILIZATION IN ANDEPTS OF COSTA RICA AND PANAMA

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Soils developed on volcanic ash are common in agricultural areas of Costa Rica and Panama, and all major population centers of Costa Rica are located on volcanic soils. Phosphorus immobilization, due to presence of amorphous minerals and organic matter-aluminum complexes, is the principal limiting chemical soil constraint in Andepts (Wada, 1980).

It is believed, that exchangeable Al and Fe can be determined by extraction with KCl, the amorphous fractions and some humus complexes are extractable by acid oxalate, the organically bound fraction by sodium pyrophosphate, and the Al and Fe-humus complexes, non crystalline hydrous oxides of Al and Fe, crystalline oxides of Fe and allophane-like constituents can be characterized by extraction with dithionite-citrate (Shoji and Fujiwara, 1984). "Active aluminum", which is the sum of the citrate-dithionite and ammonium oxalate extractions, is considered to be an important factor in P retention (Inoue, 1986).

The objective of this study was to examine the role of different forms of Al and Fe in P immobilization, and relate them to other properties of soils derived from volcanic ash.

Materials and Methods

All soil samples were obtained from Dystrandepts (Soil Taxonomy 1975) in Costa Rica and Panama, mostly from pastures. Temperature regimes were isomesic or isothermic. With the exception of the Grifo Alto site which has an ustic climate, all sites have an udic moisture regime with most places receiving over 2500 mm of annual rainfall (Table 1).

Soil samples were air dried, passed through a 2 mm sieve. The pH was measured in water (1:2.5) (Diaz-Romeu and Hunter, 1978), and in NaF pH (1:50) (Blakemore et al., 1981). Exchangeable acidity was measured by titration in 1 *M* KCl (Diaz-Romeu and Hunter, 1978). Organic matter was determined by wet combustion (Blakemore et al., 1981) and texture was estimated by the hydrometer method (Forsythe, 1985). For the ammonium oxalate extraction, 0.4 g of air dry soil was agitated for four hours in darkness with 40 ml of 0.2 *M* acid ammonium oxlate (pH 3.0) in 50 mlcentrifuge tubes. Then, four drops of a superfloc solution (0.4%) were added, the suspension again agitated and then centrifuged at 2000 RPM for five minutes as recommended by Blakemore et al. (1981). For the sodium pyrophosphate extraction, 0.4 g of air dry soil was agitated for 16 hours with a 40 ml of a 0.1 *M* sodium pyrophosphate solution in a 50 ml centrifuge tube. Then, five drops of a 0.4% superfloc solution were added and the tubes agitated for 30 more minutes. The suspension was then centrifuged for 30 minutes at 5000 RPM and filtered (Blakemore et al. 1981).

For the dithionite-citrate extraction, 0.4 g samples of soil and 0.4 g of Na dithionite were placed in a 50 ml centrifuge tube and agitated for 16 hours with 20 ml of a 22% Na citrate solution. Subsequently 20 ml of water and five drops of 0.4% superfloc were added, the mixture rapidly agitated, centrifuged and filtered (Blakemore et al., (1981). To obtain information on the Al and Fe in allophane and imogolite, they were extracted with ammonium oxlate, as suggested by Shoji and Fujiwara (1984). The oxlate extractable Al was considerably lower than pyrophosphate extractable Al. Oxlate extractable Al ranged between 0.51 and 6.07%, with a mean of 2.91%.

P retention was characterized by the isotherm method of Fox and Kamprath (1970) and by the New Zealand technique described by Blakemore et al. (1981) which has been developed for soils similar to the Andepts studied here.

Table 1. Characteristics of the sample sites

Location	Classification	Elevation (msnm)	Precipitation (mm)
1- Grifo Alto, Purisal, Costa Rica	Typic Dystrandept	950	2500
2- Fraijanes, Alajuela, Costa Rica	Hydric Dystrandept	1750	3300
3- Bijagua, Upala, Costa Rica	Entic Dystrandept	450	2800
4- San Antonio, Coto Brus, Costa Rica	Hydric Dystrandept	1200	4000
5- Valle Hermoso, Coto Brus, Costa Rica	Typic Dystrandept	1000	3800
6- Zarcero, Alajuela, Costa Rica	Hydric Dystrandept	1800	1800
7- Santa Cruz, Turrialba, Costa Rica	Typic Dystrandept	1600	2300
8- Cervantes, Cartago, Costa Rica	Typic Dystrandept	1400	2400
9- Sereno, Chiriqui, Panama	Hydric Dystrandept	750	3800
10- Caizan, Chiriqui, Panama	Typic Dystrandept	900	3500

Results and Discussion

Most samples show high organic matter levels (Table 2), probably stabilized by Al and partial adsorption on allophane surfaces.

A significant correlation exist between the amount of oxalate extractable Al and the NaF-pH (Table 3), and the regression is linear. The NaF reaction is a measure of Al easily complexed by F including inorganic and organic Al fractions. As expected, NaF-pH showed a high correlation with P retention (Table 3) in a linear relationship. Since P is usually one of the most crop limiting elements in these soils, the understanding of its immobilization is needed for their appropriate use. A NaF-pH of 10.7 corresponds to an Al level of approximately 2%, which corresponds to a P immobilization of 82-85% by the New Zealand method, and to 96% by the isotherm method (Figure 1). This shows the close correlation between this Al fraction and P immobilization, also shown for other volcanic soils of Costa Rica by Canessa et al. (1986). There was a significant correlation between P retention and oxalate extractable Al (Figure 1). This relationship was also reported for similar soils by Kawai (1980).

A significant correlation between oxalate and citrate extractable Al (Table 3), indicated the relationship between allophane like components. Except for one soil (Grifo Alto), oxalate extractable Al was appreciably higher than the dithionite-citrate extractable fraction indicating the relative importance of short-range order minerals and Al-humus complexes in these soils.

The data on alkaline Na pyrophosphate extractable Al, which in accordance to Higashi and Shinegawa (1981) is the fraction associated with organic matter, varied between 0.08 and 1.88%, with an average of 0.74% (Table 2). These values are considered low and it appears that in the soils examined, ash weathering proceeds principally towards amorphous minerals and not organic complexes. The amounts of pyrophosphate soluble Al were higher in the surface than in the subsurface horizons. Evidently, the surface horizons were higher in organic matter but it was not possible to detect a significant correlation between organic matter and pyrophosphate extractable Al (Table 3). This observation does not agree with the experience in Japan (Wada and Higashi 1976, Shoji and Ono 1978, Shoji and Fujiwara 1984), since no significant correlation between the organically complexed Al fraction and P immobilization was found. The level of pyrophosphate extractable Al was higher than that of Fe, probably due to the higher levels of Al which are liberated in the processes of weathering.

Levels of sodium dithionite-citrate extractable Al were between 0.59 and 2.0% with an average of 1.49% (Table 2), and dithionite-citrate, oxalate extractable Al and NaF-pH were significantly correlated (Table 3). Shoji and Ono (1978) have reported a similar correlation. The significant correlation between dithionite-citrate extractable Al and pyrophosphate extractable Al (Table 3) may be explained by the dissolution of much organic bound Al by both pyrophosphate and dithionite-citrate (Wada and Higashi, 1976).

Aluminum associated with allophane as imogolite increased with soil depth, having higher concentrations in the A2 and AB horizons. Similar results were reported by Wada and Gunjicake (1979), and Wada (1980) also found more allophane and imogolite in the sub-surface horizons. These higher Al contents result in slightly higher P immobilization and less fertile subsoils.

P immobilization by the New Zealand method is shown to be closely correlated with Al in allophane like constituents, extracted by dithionite-citrate (Inoue, 1986). The New Zealand method showed better correlation than the isotherm method (Table 3) for the variable studied. This method has been successfully used in Andepts (Blakemore et al., 1981), and has the advantage of being more rapid than the iostherm method.

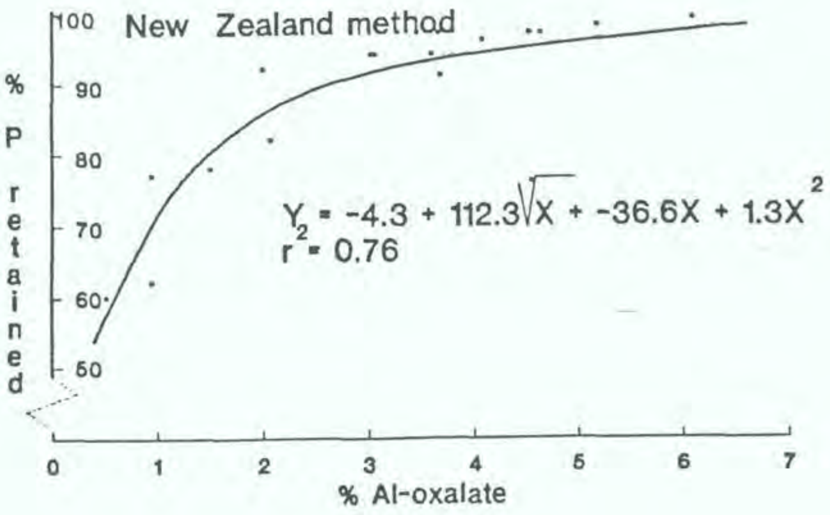
Table 2. Al and Fe fractions and P immobilization in volcanic soils from Costa Rica and Panama

				% Al			% Fe			% P immobilized	
Soil	Hori-zon	O.M. %	pH NaF	oxalate	pyro-phosphate	di-thionite	oxalate	pyro-phosphate	di-thionite	New Zealand	isotherm
1	A1	9.38	10.8	0.94	1.10	1.38	0.90	0.46	4.82	77	98.6
	AB	3.35	9.8	0.51	0.34	0.96	0.59	0.43	5.80	60	90.1
2	A1	12.46	11.5	3.64	1.28	1.96	1.46	0.51	3.02	97	99.8
	A2	10.32	11.2	5.16	0.76	1.72	1.62	0.34	3.18	98	99.8
	Bw1	3.89	11.0	6.07	0.46	1.26	1.75	0.06	3.70	99	99.8
3	A1	11.12	10.5	1.50	0.60	1.11	0.90	0.38	3.95	78	98.3
	A2	5.36	10.6	2.06	0.50	1.32	1.12	0.35	4.48	82	99.3
4	A1	14.07	11.4	3.66	0.76	1.86	0.73	0.24	2.38	91	97.5
	A2	15.41	11.3	4.54	0.70	1.94	1.04	0.16	3.48	76	98.0
5	A1	10.78	11.3	3.00	1.88	2.02	0.87	1.40	2.15	94	95.9
	AB	2.95	11.0	3.44	1.20	1.53	0.92	0.47	4.45	97	99.7
6	Ap	9.08	10.2	0.94	0.26	0.59	1.25	0.10	4.15	62	95.6
7	Ap	19.90	11.1	1.98	0.31	1.54	1.42	0.22	2.80	92	99.6
8	A1	17.87	11.2	4.06	1.56	1.94	1.16	0.12	3.62	96	98.7
9	A1	5.65	11.2	3.58	0.81	1.52	0.81	0.26	2.60	94	97.5
	AB	9.86	11.3	4.51	0.60	1.94	1.08	0.18	3.18	97	98.0
10	A1	14.52	11.2	3.04	0.78	1.56	0.70	0.33	1.65	94	99.3
	AB	10.84	11.3	4.62	0.52	1.88	0.98	0.16	1.65	97	99.8

Table 3. Coefficients of correlation (r) between variables to Al and Fe fraction and phosphate retention

	P retention New Zealand	P retention isotherms	O.M.	pH NaF	Al oxalate	Al dithionite	Al pyrophos-phate	Fe oxalate	Fe dithionite
P retention isotherms	0.70**								
O.M.	0.19	0.27							
pH NaF	0.83**	0.65**	0.48*						
Al oxalate	0.76**	0.55*	0.08	0.73**					
Al dithionite	0.68**	0.37	0.45	0.84**	0.63**				
Al pyrophosphate	0.41	0.14	0.18	0.47	0.16	0.52*			
Fe oxalate	0.34	0.55*	0.10	0.21	0.46	-0.06	-0.13		
Fe dithionite	-0.65**	-0.40	-0.50*	-0.76**	-0.50*	-0.60**	-0.20	-0.02	
Fe pyrophosphate	0.06	-0.24	0.11	0.07	-0.22	0.20	0.66**	-0.28	-0.10

* = significant at $p = 0.05$, $p > 0.47$
** = significant at $p = 0.01$, $p > 0.59$



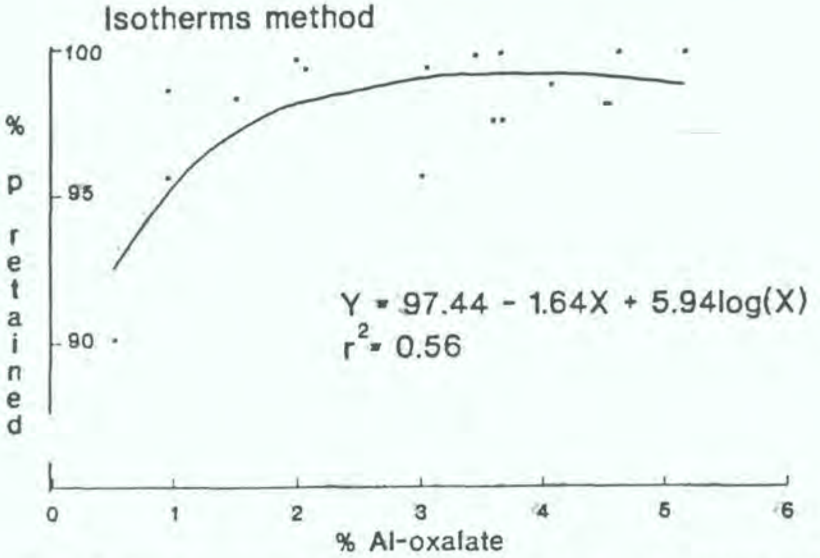


Figure 1. Relationship between phosphate retention by New Zealand and isotherms methods and oxalate extractable aluminum.

Fe-fractions

The largest Fe fraction was the one extracted by dithionite-citrate with an average of 3.65% and a range between 2.15 and 5.80% (Table 3). The oxalate soluble Fe fraction was intermediate with an average of 1.02% and limits between 0.59 and 1.75%, and pyrophosphate soluble Fe was lowest with 0.06 to 1.40% and average of 0.34%. Because of the low pyrophosphate soluble fraction, which is appreciably lower than the corresponding Al fraction, it is concluded that in these soils organic matter combines preferentially with Al, as was observed by Shoji and Fujiwara (1984) in Japan.

The oxalate extractable Fe fraction correlated with the P retained by the isotherm method (Table 3). No correlations were found between the oxlate extractable Fe and the NaF pH values, which could be explained in terms of the slow release of hydroxyls from ferric oxyhydroxides at high pH values (Alvarado and Buol, 1985; Perrot et al., 1976).

The dithionite-citrate extractable Fe presented a negative correlation with the P immobilized (Table 3). This shows that Fe in crystalline oxides and part of the humus-bound Fe which can be extracted by dithionite-citrate, have no influence on P retention in the studied soils. No correlations were found between pyrophosphate extractable Fe and other properties.

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NOTE ON P INPUTS INTO CROPPING SYSTEMS OF ANDOSOLS AND ULTISOLS IN PANAMA

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Both Andosols and Ultisols have strong P fixing capacity, but the chemical processes related to fixation and release of P are different in the Andosols and Ultisols. Laboratory, field and greenhouse studies in Panama, which are reviewed here, have shown that P fixation and release occurs differently in Andosols and Ultisols. Panamanian farmers use 250 Kg ha^{-1} y^{-1} in the Andosols and less than 10 Kg ha^{-1} y^{-1} in the Ultisols. In Panama, Ultisols are found in 40% of the area and Andosols in about 3%. In Central America Andosols are found approximately in 60% of the cultivated land.

Chemically, Ultisols are characterized mainly by a low pH and very low exchangeable base contents. Therefore the interchangeable cationic sites are occupied by extractable acidity (Al^{+3} + H^{+1}); which is a chemical barrier that often limits plant root development. The Ultisols have a loamy clay texture and are well drained. Figure 1 shows a sorption curve of P from a highly P fixing Typic Plintudult, and the corn yield response to P fertilization (broadcast and incorporated). There is a similarity in the relationship between added P and extracted P in the sorption curve and corn yield response and applied P in the field trial. Phosphate fixation in these soils plays an important role in corn yield response to P fertilization. The P fixation of this soil effectively removes low doses of P fertilizer, and produces an almost exponential response curve in the range of application tested.

The response of Ultisols to P differs depending on crops and management used. It was found that P fixation can be reduced by applying fertilizer close to the seed. Quiroz et al. (1987) in a 5 year trial of low input rice farming, found the best rate of P to be 20 Kg ha^{-1}, giving an increase in rice yield from 1,000 to 2,000 Kg ha^{-1}. In high input rice farming system, Quiroz et al. found a positive interaction between lime and P levels. The best response was found at 100 Kg ha^{-1} of P and 3 Mg ha^{-1} of lime. Name et al.(1987) found best economic yields of pineapple in a Ultisol with 5 Mg ha^{-1} of lime and 120 Kg ha^{-1} of P. Aguilar et al. (1987) working with yuca (*Manihot esculenta*, Crantz) was unable to find any response to P in a fine isohypertermic, plintic Tropudult with P levels of 1.7 $\mu g\ ml^{-1}$.

Andosols in Panama are the results of recent volcanic eruptions. The main characteristics of this order are: pH between 4.8 and 5.4, exchangeable base saturations ranging from low to high, high organic matter content and black colour, textures ranging from loam to fine loam, very friable, non-plastic and with good aeration. Very high levels of P application of 400 to 500 Kg ha^{-1} were required in laboratory and greenhouse trials to raise P to the optimum level (three times the critical level). In a subtractive fertilization experiment in the greenhouse, treatments without P yielded only 40% of the complete treatment. The only other nutrient found to be limiting was N (-N was 83% of complete) (Name and Batista 1979). Yields above 50 Mg ha^{-1} were achieved with Potato using 130 Kg of P in dystrandepts (Name et al. 1979).

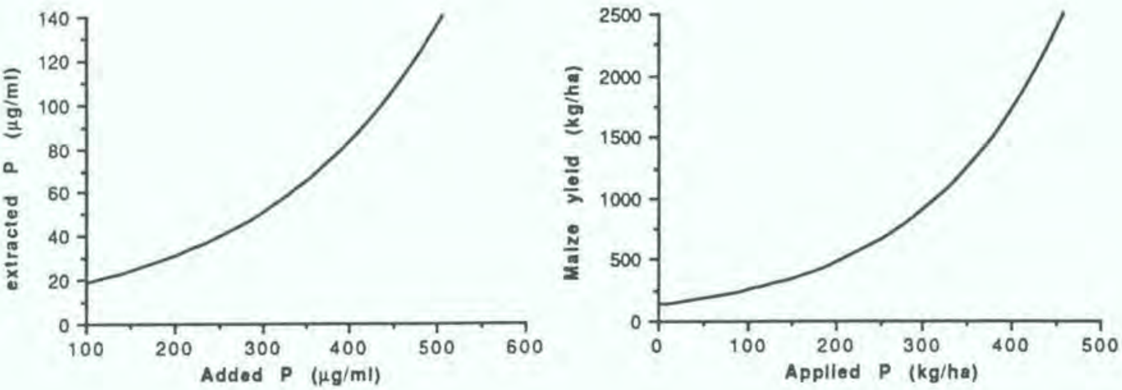

Figure 1. P sorption curve (Name 1979) and response of corn yield to broadcast and incorporated P (Name et al. 1985) on a Typic Plinthudult, Calabacito, Panama.

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PHOSPHORUS RELATED TO PLANT GROWTH AND PLANT-MICROORGANISM ASSOCIATIONS IN AMAZONIAN SOILS

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Introduction

The Amazon Basin extends over about five million Km^2, most of which are covered by a luxurious and diverse rain forest vegetation. Botanical inventories found more than 230 woody plant species per ha (Prance et al. 1976; Gentry 1988). In the world tropics, the number of plant and animals species may surpass three million; almost a million of these may become extinct by the year 2000 as a result of deforestation (Dept. of State Publication 1980). The preservation of this diverse genetic material is necessary to fuel the new biotechnological revolution in the not so distant future.

Deforestation rates in Amazonia have been high in the last 10 years, mostly due to inadequate agricultural practices and policies (Fearnside 1989). Today, although deforestation occurs in about 10% of the area, the 16 million people living in the region are not self-sustained. The challenge for the regional population is to attain self-sustainability in terms of food and related products, and at the same time preserve the primary forest. Soil chemical constraints limit soil use to only 3-5 years, however, inducing farmers to cut and burn new forest areas.

High- and low-input technologies are able to increase plant productivity and duration of land use (Sanchez et al. 1982; 1983; Wade and Sanchez 1984; Sanchez and Benites 1987). High-input technology implies applying lime and fertilizers in order to reduce soil chemical constraints, but few farmers have the economic means for doing so. Phosphorus and nitrogen are two of the most limiting factors in Amazonian soils, 90% being deficient in these elements (Nicholaides et al. 1985). Fortunately, only 16% of Amazonian soils have high P fixation (Cochrane and Sanchez 1982), and the application of only 25 kg ha^{-1} of P may be sufficient for significant yield increases (Smyth and Bastos 1985).

Low-input technology can reach a greater number of farmers, since they rely on better adapted plant species and cultivars for local soil conditions instead of heavy fertilizer applications. To be successful, low-input technology must use plants which use N and P more efficiently. This efficiency may be due to association with soil microorganisms, such as the well known N fixing symbiosis between rhizobia and legumes.

Plant roots also associate with soil bacteria or fungi which are able to improve P supply. Particularly, mycorrhizal associations are formed, which increase the plant's ability to absorb nutrients from the soil. This paper will emphasize biological nitrogen fixation (BNF), vesicular-arbuscular mycorrhiza (VAM), and phosphate solubilizing microorganisms (PSM) associated with the rhizospheres of plant species, because they may be of primary importance in future Amazonian agroecosystems.

It is important to utilize better adapted agricultural systems in Amazonia. Researchers at INPA are studying the use of ecologically adapted plants, especially indigenous species in agroforestry systems, which appear to be more adequate in terms of land use, since they partially mimic the natural forest. These agroforestry systems should be encouraged in already deforested areas, thus helping to preserve the natural forest. Plant species with highly efficient associations with soil microorganisms will be required in order to attain self-sustainability. Therefore, it is important to understand these plant-microorganism associations under Amazonian conditions, specially in relation to P nutrition. Attention must also be given to legumes, because of the potential magnitude of their contribution to agricultural systems in the region (Sanchez 1982).

Biological Nitrogen Fixation

Biological N fixation is important in replacing N fertilizers in different parts of the world (Bothe et al. 1988) and in the Amazon (Oliveira et al. 1979). However, it is impossible to separate BNF under Amazonian soil conditions from P nutrition. Amazonian available P levels are very low (0-10 mg kg^{-1}) and since P is essential for BNF and plant growth (Hernandez and Focht 1985; Collins et al. 1986; Sharpe et al. 1986), any contribution from BNF is dependent upon adequate P supplies. Oliveira and Sylvester-Bradley (1982) emphasize how important P is for nodulation and plant growth of cowpea (*Vigna unguiculata*) in an acid Amazonian Oxisol (Table 1). This experiment was carried out in the greenhouse and showed that the application of P alone increased the number of nodules 7-fold, the nodule weight 100-fold, and the plant weight 5-fold (Table 1). Similar results were found by Oliveira and Bonetti (1983) (Table 2) with an Oxisol collected from another area, indicating that the response to P is consistent in these soils in Amazonia. Studies under field conditions are necessary in order to identify the magnitude of plant response to P. This was done by Oliveira, Smyth and Bonetti (unpubl.) (Table 3) in an Oxisol where P was applied more than two years before planting cowpea. The residual effect of P was consistent and positively affected nodulation and plant growth of cowpea. It is clearly important to apply high Plevels in these soils in order to maintain high crop yields in subsequent years.

Table 1. Effect of fertilizer application on nodulation and dry weight of cowpea in an acid Amazonian Oxisol (Oliveira and Sylvester-Bradley 1982)

Treatment	Nodule number	Nodule dry weight	Plant dry weight
	$plant^{-1}$	mg $plant^{-1}$	g $plant^{-1}$
Control	12 c	3 d	2.9 c
+ P	81 b	359 c	14.7 b
+ PK	126 a	422 bc	14.4 b
+ PKS	138 a	640 a	19.3 a
+ PSK + micronut.	120 a	557 a	20.3 a

Means with different letters are different by Tukey at 5%.

Table 2. Effect of fertilizer application on nodulation, VA mycorrhiza infection and dry weight of cowpea in an acid Amazonian Oxisol (Oliveira and Bonetti 1983)

Treatment	Nodule number	VA mycorrhiza infection	Plant dry weight
	pot^{-1}	%	g pot^{-1}
Ca+Mg	3 b	91 a	2.8 b
Ca+Mg+P	75 a	88 a	14.2 a
Ca+Mg+P+K	138 a	41 b	13.4 a
Ca+Mg+P+K+S	106 a	88 a	18.7 a
Ca+Mg+P+K+S+micronut.	106 a	60 b	20.9 a

Means with different letters are different by Tukey at 5%.

Table 3. Residual effect of P application in a field trial on cowpea in an Amazonian Oxisol (Oliveira, Smyth, and Bonetti, unpubl. results).

kg P ha^{-1}	P in soil	Nodule number	Nodule dry weight	Plant yield
	mg kg^{-1}	$plant^{-1}$	mg $plant^{-1}$	kg ha^{-1}
0	4	82 b	130 bc	15 c
22	4	78 b	80 c	45 c
44	6	258 a	320 bc	139 c
88	14	233 a	510 bc	684 b
176	38	306 a	1070 a	1094 a

Means with different letters are different by Tukey at 5%.

Another aspect to be considered in relation to P availability and BNF is rhizobia survival in the soil. Rhizobia must persist in the soil after nodule degeneration, in order to infect subsequent root growth of the same plant (perennial or semi-perennial species) or subsequent legume crops planted in the same area. Rhizobia populations detected under some forest tree plantations in Amazonia proved to be very low, not surpassing 10 cells per gram of soil (Bonetti et al. 1984). More than 100 cells per gram of soil are necessary for inducing good nodulation. It is important therefore, to select rhizobia strains adapted to the acid nutrient poor Amazonian soils. Rhizobia tolerance to aluminum and soil acidity is important, but P level seems to be equally important (Keyser and Munns 1979; Beck and Munns 1985). Under low P supply, sensitive strains showed deformed shapes (Cassman et al. 1981) which may be deleterious to their survival in the soil. Electron microscopy indicated a tendency for P deficient cells to become elongated, distorted, and packed with lightly staining granules (Cassman et al. 1981). Such studies are necessary in

Amazonia in order to increase the potential of BNF in the region.

Phosphate Solubilizing Microorganisms (PSM)

Plants cause pH variation in their rhizosphere (Smiley 1974; Mugwira and Patel 1977), which may help them to compete for soil P. Under alkaline soil conditions for example, P is bound to calcium and those plants able to acidify their rhizospheres may increase their ability to absorb P. In contrast, under acid soil conditions like those in Amazonia, P is bound to aluminum and iron and plants must raise their rhizosphere pH in order to absorb more P. It has been observed that aluminum tolerant species increase rhizosphere pH, causing a decrease in Al^{+3} ions and a corresponding increase in phosphate solubility, and other anion uptake (Foy et al. 1967; Mugwira and Patel 1977).

The difference of rhizosphere pH among plant species may be related to root exudates and effects of soil microorganisms. These rhizosphere microorganisms may use root exudates, transforming them into alkaline or acid compounds, thus altering rhizosphere pH.

There are many microorganisms in the soil able to solubilize phosphate. Greaves and Webley (1965) showed that 90% of the bacteria isolated from forage grass rhizospheres produced phosphatase. In some countries, such as the USSR, phosphate solubilizing microorganisms are inoculated on plant seeds (Barea 1969). However, it is difficult to identify whether the positive effect of seed inoculation is due to P solubilization. These bacteria may produce phytohormones which may stimulate root growth and increase their ability to absorb P (Brown 1974; Barea et al. 1976). Gonzalez-Eguiarte and Barea (1975) believe that both mechanisms may occur; they recommend more detailed studies to separate P solubilization from root growth effects.

Table 4. Effect of rhizosphere bacteria and carbon sources on pH of growth medium (initial medium pH 6.8-7.0) (Sylvester-Bradley et al. 1982)

Carbon source	Bacterial isolate		
	A	B	C
Glucose	3.1	3.5	3.6
Mannitol	6.1	6.7	6.5
Malate	8.5	8.4	8.7
Succinate	8.5	8.6	8.8
Glycerol	6.8	7.0	6.7
Arabinose	5.5	6.0	5.9

In Amazonia, Sylvester-Bradley et al. (1982) observed the presence of phosphate solubilizing bacteria in laboratory growth medium. They were isolated from grasses (*Paspalum plicatum, Hyparrhenia rufa, Panicum maximum, Brachiaria humidicola, B. decumbens,* and *Andropogon gayanus*), and legumes (*Stylosanthes capitata, S. guianensis, Desmodium ovalifolium,* *Pueraria phaseoloides*, and *Zornia* sp.). In general, there were more of these bacteria in legume than in grass rhizospheres. Interestingly, these authors found that by changing the carbon source in the laboratory medium, different final pHs were obtained (Table 4). When the carbon source was glucose or arabinose, the pH dropped. In contrast, when malate or succinate were used, the pH rose.

Rovira (1969) reports that plant exudates show great variation in their carbon compounds. Sugars (glucose, fructose, arabinose, etc), amino acids (valine, asparagine, proline, etc), organic acids (oxalic, acetic, succinic, etc) were found in wheat rhizospheres, with differences among plant species also reported. Rovira (1969) explains that soil microorganisms may effect root exudations in several ways: a) by modifying root cell permeability, b) by modifying root metabolism, and c) by absorption of certain compounds in root exudates and excretion of other compounds. Thus, it is possible that soil bacteria may modify rhizosphere pH in a magnitude sufficient to alter the ability of plants to absorb P from the soil. More studies with soil bacteria able to solubilize phosphate under Amazonian soil conditions are necessary in order to understand how these associations may help plant adaptation in Amazonia.

Vesicular Arbuscular Mycorrhiza (VAM)

VAM associations may contribute to plant nutrition, especially P, in Amazonian soils by increasing root area in contact with the soil. Increases in plant growth, productivity and P contents are reported in the literature when plants are inoculated with mycorrhizal fungi (Ross and Harper 1970; Mosse et al. 1973; Islam and Ayanaba. 1981; Bonetti 1984), which may affect the interpretation of soil P analysis (Bolan et al. 1984). Ten-fold increases of plant dry weight were observed in siratro (*Macroptilium atropurpureum*) inoculated with VAM under greenhouse conditions in a sterile Amazonian Oxisol (Bonetti 1984). Phosphorus content in the tissues increased from 125 mg kg^{-1} to more than 560 mg kg^{-1} (Bonetti 1984). A large number of Amazonian plant species show VAM infections in the root system (St. John 1980 a,b; St. John and Uhl 1983; Bonetti et al. 1984). Nevertheless, mycorrhizal fungi inoculation on plant roots may further increase plant growth (Bonetti, Oliveira and Donald, unpubl. Table 5). They used a nonsterile soil in order to compare inoculation with selected species of VAM with the natural soil VAM population. Table 5 shows that selected species of VAM were more efficient, since in the control treatment plant nodulation and growth were inferior to the other treatments. These results indicate that VAM inoculation may be essential for an adequate plant growth in Amazonian soils.

The effect of fertilizer application on VAM infection was also studied. VAM infections are reported to be inversely related to P concentration in the soil solution (Mosse et al. 1981). In the study by Oliveira and Bonetti (1983), the application of 40 mg kg^{-1} of P did not decrease VAM infection. A negative effect was observed only when K or micronutrients were also applied (Table 2), perhaps because, in both cases, an increase of elements such as Zn and Cu may have occurred in the soil solution, and inhibited VAM activities, as reported by Hepper (1979).

Because of the wide occurrence of VAM in different plant species, without apparent specificity in terms of infection, scientists have suspected that mycorrhizae may be involved in nutrient transfer between plants. In fact, recent results indicate interspecific and intraspecific transfer of P, C and N (Heap and Newman 1980; Francis and Read 1984; Francis et al., 1986). This transference of nutrients between plants in a crop system may be positive, since deficient plants may absorb nutrients from others, with the maintenance or increase of overall yield. In agroforest systems however, care must be taken in applying VAM, because the nutrient flux may be from a more economically important species to a lesser. Such studies may indicate the most adequate plant species associations for agroforest systems.

Table 5. Effect of VAM innoculation on nodulation, VAM infection and plant dry weight of cowpea in an acid Amazonian Oxisol (Bonetti, Oliveira and Donald unpubl. results)

Treatment	Dry weight of nodules	VA mycorrhiza infection	Plant dry weight
	mg pot^{-1}	%	g pot^{-1}
Acaulospora sp.	131 a	94	8.5 a
Gigaspora heterogama	80 ab	92	6.2 ab
Glomus macrocarpum	82 ab	94	7.5 ab
Control	62 b	97	4.6 b

Means with different letters are different by Tukey at 5%.

Concluding Comments

The challenge in the Amazon Basin is to develop highly efficient agroecosystems in areas already deforested in order to achieve a sustainable production and preserve the primary forest. High- and low-input technologies are applicable in the region, but it is very difficult for the smallholder to obtain fertilizers for use in high-input technology systems. Low-input agroforestry-systems may be one of the best ways to improve regional self-sustainability and preserve forests. However, more plant species better adapted to the acid, nutrient-poor soil conditions are needed for the success of these systems. Care must be taken in selecting species and genotypes efficient in N and P uptake and use. Microbial aspects should be emphasized in selecting these plants, since biological N fixation may supply all or part of the N necessary for adequate plant growth, and rhizosphere microorganisms may be helpful in P and other nutrient absorption. Few studies have been conducted on biological N fixation, VAM associations and P solubilizing microorganisms in Amazonia. Nonetheless, results already obtained suggest that these associations may be of primary importance in increasing the productivity of agricultural systems in the region.

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MODELLING PHOSPHORUS INPUTS FOR CERRADO CROPPING SYSTEMS

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Introduction

The objective of a soil test interpretation is to be able to recommend the optimum rate of fertilizer application. One philosophy of achieving this is to maintain a nutrient so that its extractable concentration is at a level sufficient for maximum yield. In order to compute the optimum rate of a given nutrient, three values are required: (i) existing extractable nutrient concentration in the soil, (ii) the concentration sufficient for a maximum yield, and (iii) a buffer capacity factor indicating the change in soil nutrient concentration during a period per unit of nutrient applied. With the inclusion of economic considerations, the interpretation may be modified to recommend the rate of fertilizer required to maximize net returns.

The levels of extractable P sufficient for maximum yield vary with soil properties (Lins et al., 1985). Extractable P alone, however, may not provide adequate information on the critical concentration of nutrients for predicting fertilizer recommendations. Other factors, for example, clay content have been shown to be correlated with P adsorption capacity (Olsen and Watanabe, 1963) and with P buffering capacity of soils (Ozanne and Shaw, 1968). Phosphorus sorption is also influenced by the type of clay minerals in soils (Karin and Adams, 1984). Lins et al (1987) showed improved prediction of P requirements for soybean when clay content, Mehlich-1, Mehlich-3, Bray-1 and an ion exchange resin extractable P were used.

The objectives of this paper is to present models generated with four different methods of extractable soil P, combined with clay content to predict the optimum P fertilizer rate for soybeans in the Cerrado region of Brazil.

Material and methods

Experimental procedures. The experimental procedures involved in this study, to generate data to feed the developed model, were carried out under greenhouse, laboratory and field conditions. Details of all procedures, were previously reported by Lins and Cox (1989a) and Lins and Cox (1989b). Seven soils were used under both, greenhouse and field conditions. However, data of two out of the seven field sites were used to validate the model.

Inputs necessary to construct the model. The mathematical approach used to generate the present model can be applicable for a particular group of soils located anywhere. However, to build-up this model, the ideal situation would be to have long-term P experiments located within the area of interest. From these experiments, data relative to yield, extractable soil-P, and clay content or other soil properties that reflect soil P buffering capacity are necessary as the input for the model whose objective would be to predict the most economical-P fertilizer rate for crops to be grown in areas where the long P studies were located.

Mathematical approach

A descriptive model to express extractable P level (X) following fertilization (F) with time (T) was developed by Cox et al. (1981), this model was as follows:

$$X = X_{eq} + [(X_0 + b_1F + b_2F^2) - X_{eq}]\ e^{(-KT)} \qquad (1)$$

Where X_{eq} is an equlibrium soil test obtained from the check plots of long term P experiments; X_0 is the extractable soil P before P fertilization in kg ha^{-1}; b_1 and b_2 are linear and quadratic coefficients of the regression equation and express the soil P buffering capacity; K is constant for loss that P fertilizer undergoes with time after its application in the soil. Basically, this model is an integration of two equations. One expresses the change in initial extractable nutrient level (X_{init}) with time which is related to K as follows:

$$X = X_{init}\ e^{(-KT)} \qquad (2)$$

Upon addition of P fertilizer, it is assumed that there are several reactions that occur which differ markedly in rate. We limited our consideration to two groups; an initial, very rapid adsorption expressed by b_1 and b_2 coefficients in Eq. 1, and a slower reaction which follows, probably precipitation, expressed by the loss constant (K) in eq. 1 and 2. Data is often available in which several rates of fertilizer have been applied to establish successively higher soil levels of a nutrient, followed by observations made for a period of years with no further fertilization. With such data, the value of X_{init} and K at each level of fertilization may be preliminarily estimated by Eq. 2.

Preliminary estimates of b_1 and b_2 (Eq. 1) are determined after obtaining the estimated values of X_{init} for each P rate as described above using Eq. 2. The X_{init} values are regressed as a function of F, the rate of fertilization in kg ha^{-1}. If several estimated values of X_{init} are available, it may be determined if the relationship between X_{init} and F is linear or quadratic. The regression coefficients obtained give the preliminary estimates for b_1 and b_2 in Eq. 1. The preliminary estimates of K were obtained by Eq. 2 for each P rate applied to each soil; then they were averaged.

Once the final estimates of b_1, b_2 and K are obtained for a certain soil, solution of Eq. 1 gives the extractable soil P concentration that one could expect to obtain from that particular soil a certain time after a single P fertilizer application.

The descriptive model developed by Cox et al. (1981) allows one to determine the immediate effect of fertilizer rate, F, on extractable soil P concentration for only one initial soil test level X_0. This expression was modified by Lins et al. (1985) in order to allow this model (Eq. 1) to consider other initial soil P concentrations.

Taking into account economic considerations, net income (N) was calculated by computing crop income minus fertilizer cost. Crop income was the yield times the soybean price (Q) and the fertilizer cost was the rate (F) times the fertilizer cost (S).

$$N = QY - SF \tag{3}$$

Combining these approaches gave fertilizer rates for maximum net income for different given soil test levels for sites used in this study. This was done with the data from each site for six initial levels between X_0 and greater values up to where the effect of fertilizer rate became negligible. The predicted fertilizer rates obtained for the five soils were then evaluated by multiple regression so that the fertilizer for maximum net income (R) was a function of initial soil test level (soil P) and other selected soil properties. This expression included the main effect interation term. For example, if soil P and clay are considered, the equation became:

$$R = a+b\ (\text{soil P}) + c\ (\text{clay}) + d\ (\text{soil P})\ (\text{clay}) \tag{4}$$

With this expression, the fertilizer P rate for maximum income becomes a continuous function of initial soil test level and clay content. In some cases, quadratic terms were also used.

Results and discussion

For the purpose of this paper, only data from the field studies will be presented and discussed. Data obtained in greenhouse experiments were previously reported (Lins and Cox 1989a). The soils from the five long term study sites were initially very low in extractable P, ranging from 0.6 to 6.0 g m^{-3} among the four extractants (Table 1). The soil with the least clay content (12%) had the highest native extractable P level. This would be expected from its characteristics of low P adsorption maximum and low surface area (Table 2).

Average maximum yield is estimated by the coefficient "A" in Table 3. The yield response per unit of soil P extracted with M1P, M3P and B1P was greater with increasing clay content, as indicated by the larger value of the coefficient "C" in Table 3. This means that a selected yield, such as 90% of the maximum, would be reached at a much lower soil P level as the clay content increases. The higher requirement on sandy soils is probably due to lower diffusion rate and buffering capacity. The diffusion rate of P is less in sandy than in clayey soils (Olsen and Wantanabe, 1963).

The immediate residual effect of P fertilization estimated as extractable soil P increases as clay content decreases. These effects were similar for M3P and B1P extractants and indicated about 15% immediate recovery for the soils with around 60% clay, increasing to over 50% immediate recovery for the soil with only 12% clay. Here, it becomes important to point out that when Mehlich-1 extractable P was used a quadratic relationship between applied P and soil P recovered just after application and rototilling was observed. However, when the other soil P extractants were used, this relationship was linear. That is the reason why in Table 3 the b_2 term appears only for the Mehlich-1 extractable soil P.

Table 1. Selected chemical properties of the soil prior to treatment

					Extractant			
Clay	pH	Al	Ca + Mg	K	M1P[2]	M3P[3]	B1P[4]	Resin[5]
%	1:2.5	Cmol L^{-1}			g m^{-3}			
68	5.0	0.63	0.32	0.07	0.6	1.5	1.5	2.1
63	4.9	1.12	0.68	0.10	1.7	2.6	2.4	6.0
57	4.8	1.30	0.36	0.07	1.6	1.6	1.9	3.4
27	4.6	1.26	0.95	0.12	3.9	4.7	4.6	5.7
27[1]	5.0	1.10	0.84	0.13	1.1	1.8	1.9	2.3
21[1]	5.7	0.30	0.34	0.07	0.6	1.7	2.3	2.8
12	5.2	0.34	1.04	0.09	5.0	5.9	6.0	5.0

[1] Soils used to validate the proposed model
[2] Mehlich - 1 extractable soil P
[3] Mehlich - 3 extractable soil P
[4] Bray - 1 extractable soil P
[5] Ion exchange resins extractable soil P

Table 2. Surface area, mineralogical composition and P adsorption maximum of soils from the five long - term P studies

Clay	Surface	Gibbsite	Goethite	Kaolinite	Hydroxy Interlayered Vermiculite	Free Iron Oxides	P Adsorption Maximum
%	m^2 g^{-1}	% in soil					mg g^{-1}
68	35.4	44.2	3.4	20.4	0.0	2.7	0.7039
63	32.1	12.6	14.5	28.4	7.6	12.8	0.6643
57	30.8	6.7	7.2	28.8	5.3	6.2	0.6450
27	11.6	0.5	3.0	20.3	3.2	5.1	0.4246
12	6.8	0.1	2.3	7.8	1.8	2.0	0.1462

The amount of fertilizer P recovered immediately by the resin extractant was not the same as that for the other extractants; more P was recovered from the clayey soils and slightly less from the more sandy ones (Table 3). The results from the resin extractant, therefore, should estimate plant uptake better than those from M1P, M3P and B1P extractants. The relationship between rate of decrease of resin P over time expressed by the coefficient "K", was opposite that noted with M1P, M3P and B1P extractants, but the reason for this reversal is not clear.

Table 3. Coefficients or parameters for yield response and residual P equations for four extractants for soybeans grown in the field on five soils differing in clay content

Coefficient or Parameter	Clay (%) 68	63	57	27	12
For M1P					
Xeq	1.04	1.60	2.50	2.96	5.62
Xo	1.20	3.70	3.34	7.80	10.00
b_1	0.0413	0.0996	0.0479	0.2624	0.3136
b_2	0.00079	0.00014	0.00031	0.00028	0.00066
K	0.2493	0.3083	0.2493	0.2596	0.2463
M3P					
Xeq	2.70	2.20	2.60	4.20	6.50
Xo	3.00	5.30	3.20	9.00	11.80
b	0.1312	0.1586	0.1635	0.3466	0.5374
K	0.2118	0.1978	0.2135	0.1193	0.1475
For B1P					
Xeq	2.40	2.60	3.14	6.90	7.50
Xo	3.00	4.80	3.80	9.20	12.00
b	0.1475	0.1924	0.2003	0.4283	0.5328
K	0.2199	0.1954	0.2279	0.2048	0.1444
For Resin					
Xeq	3.42	6.78	3.84	6.22	5.53
Xo	4.20	12.00	6.80	11.40	10.00
b	0.2089	0.2920	0.2598	0.3702	0.4793
K	0.1770	0.2011	0.1984	0.2666	0.2728

Model application

When the optimum P fertilizer rate was predicted with extractable P alone, 66, 67, 57 and 62% of the variation was accounted for with M1P, M3P, B1P and Resin extractants. However, when the clay-squared term was introduced in a multiple regression analysis, 86, 84 and 83% of the variation was accounted for, respectively, with Mehlich-1, Mehlich-3 and

Bray-1 extractants. With the resin, the clay term did not need to be squared and this soil P extractant combined with % clay accounted for 95% of the variation in the optimum P rate. This may be due to the ability of this extractant to extract more labile P than the others.

Besides clay content, surface area, P adsorption maximum and gibbsite content in soils were the three best soil properties to be combined with extractable soil P to predict the optimum P fertilizer rate using this model. Since clay content can be estimated readily it was selected as the factor that should be included when predicting P fertilizer requirement. The relationship found for M1P was:

$$R = 95.6 - 3.14\ (\mathrm{M1P}) + 0.0046\ (\mathrm{clay})^2 - 0.0012\ (\mathrm{M1P})\ (\mathrm{clay})^2$$

where the predicted fertilizer P rate R, and M1P are in kg ha^{-1}.

The relationship found for M3P is similar to that described for M1P and can be described as follows:

$$R = 60.0 - 0.38\ (\mathrm{M3P}) + 0.012\ (\mathrm{clay})^2 - 0.0015\ (\mathrm{M3P})\ (\mathrm{clay})^2$$

Equations to predict optimum P fertilizer rates were also developed for the B1P and resin extractants. They are:

$$R = 62.4 - 0.64\ (\mathrm{B1P}) + 0.011\ (\mathrm{clay})^2 - 0.0011\ (\mathrm{B1P})\ (\mathrm{clay})^2$$

$$R = 31.6 + 0.0742\ (\text{resin-P}) + 1.49\ (\mathrm{clay}) - 0.0624\ (\text{resin-P})\ (\mathrm{clay})$$

The equation for B1P is quite similar to that of M3P, as would be expected from the composition of these extractants. The function for resin-P differs in two respect from the other extractants. First, the coefficient of resin-P is positive, indicating more fertilizer would be recommended as soil P increases. Second, the clay term in the resin-P equation also is a direct effect rather than a squared term as in the other equations.

Model validation

In the two validation studies, maximum soybean grain yields were between 2.600 and 3.100 kg ha^{-1}. The yield response was calculated as a function of fertilizer P and solved at 90% maximum yield. The results showed that 90 and 200 kg ha^{-1} should have been applied on the soils with 27 and 21% clay, respectively. When 27% clay and the P in that soil by Mehlich-1 were used the predicted P rate was 91 kg ha^{-1}. Thus the M1P model was validated for this particular soil. For this same soil, using equations 6, 7 and 8, the P rate is somewhat less, about 70 kg ha^{-1}. This is a fairly reasonable estimate considering that the initial soil P level is extremely low, about 2 g m^{-3}.

For the soil with 21% clay, however, the prediction was only about 1/3 of the required 200 kg P ha^{-1}. This soil had an atypically high amount of gibbsite which may account for the unusually high P rate required. If so, applications of these equations, to soils with low clay contents, should be restricted to the more common, kaolinitic soils of this texture in the region.

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PHOSPHORUS STATUS AND AGRONOMIC EFFICIENCY OF PHOSPHATE FERTILIZERS IN THE CERRADO REGION OF BRAZIL

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Introduction

The great majority of Brazilian soils, mainly those located in the Cerrado, present extremely low levels of total and available P for cultivated crops. This problem is enhanced by the high P "fixation" capacity of these soils which is in general associated with type and percentage of clay components. As a result, high rates of P fertilizers are necessary in order to achieve adequate crop production when these soils are cultivated (Goedert et al. 1986).

The necessity of high rates of P fertilisation, the costs of this practice, and the scarcity of natural phosphate rock deposits in Brazil justify efforts to search for higher agronomic efficiency of P fertilizers.

The search for a higher recuperation of P from Brazilian rock phosphate (RP) deposits needs to involve several steps including mining, industrialization, transportation, storage, and finally the agronomic management techniques to achieve higher absorption efficiency in relation to the rate of applied P.

Several measures to increase agronomic efficiency of P fertilizers have been suggested:

a) reduction in P fixation capacity through adequate liming;
b) combination of most adequate rates and methods of application, for specific crops;
c) rational utilization of P sources available in the region;
d) utilization of plant species more efficient in the utilization of applied P;
e) utilization of more efficient crop production systems including crop rotation, green manure etc.

One serious limitation to the evaluation of agronomic efficiency of phosphate fertilizers in Brazil is the lack of long term field experiments, that would allow to evaluate long term residual effects. Another limitation is that economic evaluation together with agronomic evaluation are not commonly conducted at the same time. In this paper the following concept of relative to agronomic efficiency expressed as AEI % = Agronomic Efficiency Index will be used, according to Goedert et al. (1987):

$$\text{AEI }\% = \frac{\text{(Yield of source under test) - (Test plot yield (-P))}}{\text{(Yield of reference source) - (Test plot yield (-P))}} \times 100$$

where the reference source is single(SSP) or triple superphosphate (TSP).

The objective of this paper is to discuss the agronomic efficiency of P fertilizers for annual crops, pasture, perennial crops and reforestation in the Brazilian Cerrado based upon soil, climate, and crop characteristics and upon types of fertilizers produced and sold in this region.

The Cerrado region

General characteristics. The area under Cerrado vegetation in Central Brazil occupies 2.0 million km^2 or 23% of the country. According to Cochrane et al. (1985), Oxisols cover more than 50% of the area; the remainder consists mostly of Ultisols, Entisols and Inceptisols. Red-Yellow Latosols and Dark Red Latosols (mostly Ustox in U.S. Soil Taxonomy) are preferred for intensive agriculture (Goedert, 1987). Most of these soils are loamy or clayey, but approximately 20% are sandy, deserving special management practices for continuous cropping. The clay fraction is dominated by kaolinite, gibbsite and iron oxides (Hughes, 1980). The predominance of these low-activity clays determines most of the soil properties, especially ion exchange, water holding capacity, and P fixation characteristics (Goedert, 1987). Most of these soils are very acid and have low availability of macro- and micronutrients. They have high aluminum saturation and P fixation capacity.

Phosphorus is the most deficient plant nutrient in these soils. A survey conducted by Lopes (1975) and summarized by Lopes and Cox (1977) indicated that 92% of the topsoil samples contained less than 2 $\mu g\ ml^{-1}$ of Mehlich-1 extractable P. The very low available P is probably one of the most critical limitations for crop growth in these soils. In addition, correction of this problem is more difficult than in other soils due to the high P fixation capacity (Leal and Velloso, 1973). The amount of P required to obtain 0.1 $\mu g\ ml^{-1}$ in equilibrium solution has been reported to vary from 110 to 405 $\mu g\ g^{-1}$ (Le Mare, 1982; Smyth and Sanchez, 1982; Le Mare and Goedert, 1984).

Besides these chemical problems there are other production limitations: a) a normal dry season of 5 to 6 months (April to September); b) dry spells during the rainy season ("veranicos"), generally associated with high evaportranspiration rates; (c) low water-holding capacity; d) limited rooting depth of many crops as a function of aluminum toxicity and/or calcium deficiency in subsurface layers. These points emphasize the need for appropriate management technology to increase the probability of successful crop production, mainly under rain-dependent conditions.

However a series of positive points of this area should also be mentioned. Adamoli et al. (1986) estimate that about 73% of the Cerrado area range from 300 to 900 meters elevation. The average annual temperature ranges from 22°C in the south to 27°C in the north, generally associated with changes in elevation. The average monthly temperature is almost constant throughout the year. Daily solar radiation input is high over the whole area, around 475 to 500 langleys (Adamoli et al., 1986). Temperature and energy characteristics, therefore, are favorable for growth of almost any commercial crop during the whole year (Goedert, 1987). Total annual rainfall varies from 900 to 2,000 mm with distinct wet and dry seasons. A combination of gentle slopes (commonly less than 3%), and deep soils with very good aggregate stability and drainage favour intensive agricultural mechanization.

Approximately 50% of the region is covered by potentially arable lands (Goedert, 1989). A burst in agricultural development has taken place in this area during the last fifteen years, mainly involving grain crops, sugarcane, improved pasture and coffee. The average yield of some of these crops exceeds the national average. The Cerrado contributes more than 20 million metric tons (Mg) of grains, and also a reasonable percentage of Brazil's total production of coffee and sugar cane. It is important to note that until the 1970's this region was considered marginal for crop production. However, the management technology developed for these soils and already practiced by the farmers have completely changed this picture. The estimated potential for the Cerrado region production is 190 million Mg of grains (150 dryland and 40 irrigated), 9 million Mg of beef cattle and 300 million m^3 of forest products (Goedert, 1989).

Rock phosphate deposits. Estimates of worldwide RP reserves indicate a percentage participation of only 2.6% for Brazil (Table 1). It should be stressed that from 1975 to 1987, prospecting indicated a 240% increase in the Brazilian RP deposits, and other areas of potential deposits have been identified (Giulietti et al. 1987).

Table 1. World phosphate rock deposits. (Phosphate Rock Statistics, IFA, Cited by Berger et al. 1987)

Country	Deposits	
	10^6 Mg	%
Morroco	40,000	27.3
USA	35,600	24.3
China	15,000	10.3
USSR	7,125	4.8
Brazil	3,840	2.6
Others	44,742	30.7
World Total	**146,307**	**100.0**

The percent increase in phosphate production in Brazil from 1975 to 1986, was higher (1,207%) than that of other major producers (Morocco 52%; Tunísia 65%; USA 9%; China 188%; Jordan 362% and USSR 37%) (FAO, 1988). This growth was a result of the discovery of new deposits and of the development of adapted technology to process phosphate rocks of magmatic origin, which account for most Brazilian deposits. These magmatic deposits have lower concentrations, and more complex mineralogies (iron, aluminum and magnesium impurities), than marine sedimentary deposits (Lobo, 1986). Approximately 87% of country's capability to produce RP is located in the Cerrado region (ANDA, 1987).

In order to achieve maximum agronomic and economic efficiency of P fertilizers already produced in Brazil (Table 2), it is important not only to consider the fertilizer characteristics themselves but also the soil, climate and crop characteristics in terms of adequate management. This matter is discussed in the next topic.

Table 2. Distribution of P consumption in Brazil (Lobo 1986)

Phosphate fertilizers		98%
Wet route	90%	
..Ammonium phosphates	28%	
..Complex fertilizer	18%	
..Double and Triple superphosphates	18%	
..Single superphosphates	24%	
..Partially acidulated phosphates	2%	
Thermophosphate	2%	
Untreated rock P	6%	
Other uses		2%
Total		100%

Agronomic efficiency of phosphate fertilizers in the Cerrado region

Annual crops. One of the first studies concerning evaluation of agronomic efficiency of phosphate fertilizers in Brazil was carried out by Alvarez et al. (1965) with sugar cane. The results of this study indicated that the natural RPs tested produced considerably lower yields than TSP and magnesium thermophosphate (Figure 1).

During the last decade, much research has been dedicated to the evaluation of new phosphate rock deposits located in the Cerrado region. One of these research programmes was initiated in 1975 at the Cerrado Research Center (EMBRAPA), at Planaltina-DF. One of the main conclusions of this work was the low agronomic efficiency index (AEI) of the Brazilian natural RPs tested for annual crops (Table 3). Even considering five years under annual crops the AEI was below 50% when compared with Gafsa RP, magnesium thermophosphate and TSP. However for andropogon grass, cultivated for three years after the annual crops, the AEI was considerably higher. More recent studies as summarized by Goedert and Dias Filho (1986), have confirmed that Brazilian RPs present low AEI for direct application in annual crops, mainly because these products do not supply P at the adequate rate and intensity required for these crops.

This behavior is often complicated because the great majority of these soils is extremely acid, with high aluminum toxicity, and liming is the first management practice to achieve adequate yields of non-tolerant crops. Several papers have shown an even lower AEI for Brazilian RPs after adequate liming (Braga, 1969; Tanaka, 1978; EMBRAPA, 1980; Silva et al. 1983).

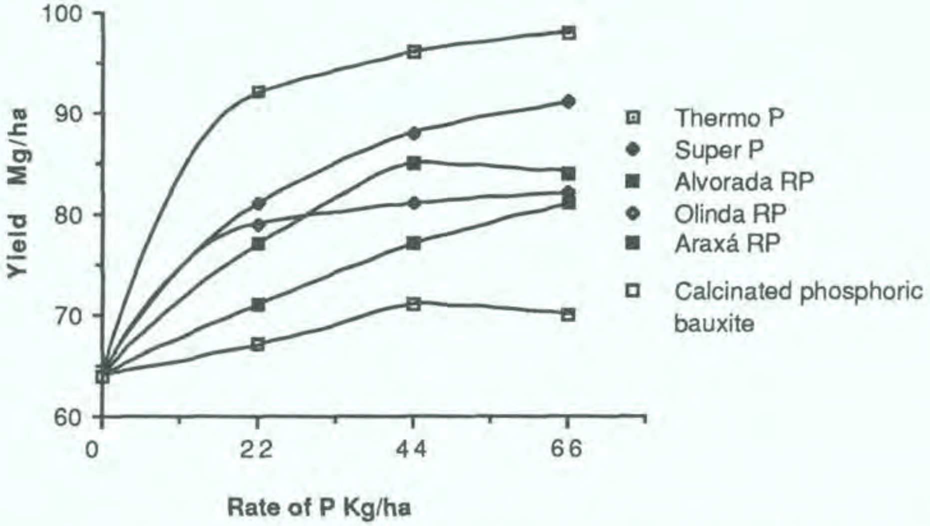

Figure 1. Responses of sugar cane to sources and rates of phosphates fertilizers (Alvarez et al. 1965).

Table 3. Agronomic efficiency of phosphates (90 kg.ha^{-1} of P broadcast in the first year) in a clayey Dark Red Latosol calculated from P absorption data during five years with annual crops followed by three years with Andropogon pasture (Goedert and Lobato 1984).

Phosphate	Agronomic Efficiency Index (AEI)		
	Annual crops	Andropogon	Total
	%		
Patos	45	81	59
Araxá	27	69	41
Abaeté	21	86	43
Catalão	8	36	17
Gafsa	93	110	104
Thermophosphate	92	142	113
Triple Superphosphate	100	100	100

Taking into account the mineralogical and chemical characteristics and the agronomic behavior of Brazilian natural RP, four alternative product groups for improving the low AEI of the simple ground product.have been studied:

a) Traditional soluble phosphates;
b) Phosphates partially acidulated with sulphuric acid;
c) Phosphates solubilized by other chemical process;
d) Thermophosphates.

Most of the P fertilizers produced in Brazil are superphosphates (SSP and TSP) and ammonium phosphates (MAP and DAP). These water soluble P sources are considered excellent in terms of agronomic efficiency but require a relatively complex technology, addition of sulphur, and concentrated RPs with low levels of contaminants (R_2O_3). The last points are extremely important in Brazil, since more than 80% of sulphur consumption is imported, and approximately 40% of the P mined is lost during ore concentration and elimination of impurities (Lobo and Silva, 1984). The need to reduce sulphur consumption and to use RPs with high degrees of impurities, mainly iron and aluminum oxides, has resulted in the production of partially acidulated phosphate at industrial scale during the last 10 years. The technology for industrial production of partially acidulated phosphates is simpler than that for superphosphates. Research to evaluate the agronomic efficiency of these sources is relatively recent. A literature review conducted by Goedert et al. (1986) indicated that with 50% acidulation, the AEI was approximately 60% in relation to the AEI of TSP.

A broad research programme initiated in 1983 evaluated six diverse RPs (Anitápolis, SC; Araxá, MG; Catalão, GO; Olinda, PE; Patos de Minas, MG; and Tapira, MG) at five levels of acidulation. Initially, these thirty products were evaluated under greenhouse conditions using corn as the test crop (Goedert and Souza, 1986). The main conclusion of this study was that the agronomic efficiency of the P sources was directly related to the percentage of water-soluble, citric acid or ammonium citrate extractable P. The correlation between AEI and the proportion of total P extracted by citric acid was highly significant ($r^2=0.88$), and linear with a gradient close to unity, i.e. the AEI is proportional to citric acid extractable P, irrespective to the origin of the RP. Similar results were obtained with ammonium citrate extractions.

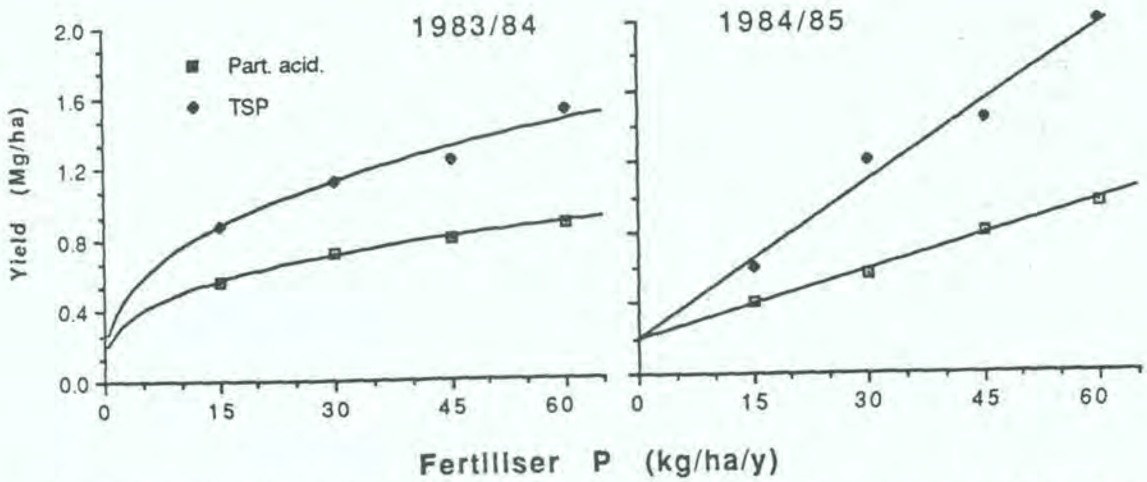

Figure 2. Soybean yield in a Dark Red Latosol under Cerrado as a function of annually sidebanded phosphate fertilizers (TSP and partially acidulated Araxá phosphate) (Goedert et al., 1986)

Some of these products have been evaluated under field conditions with several crops. The results generally confirm the greenhouse data: the agronomic efficiency was proportional to the level of acidulation of the RP and was directly related to the percentage of total P that was soluble in ammonium citrate. Results of two field experiments involving soybeans and corn are shown in Figures 2 and 3. In both locations the AEI for partially accidulated phosphates was considerably lower than that obtained for TSP. The efficiency for the partially acidulated phosphate did not increase with time, suggesting that the insoluble P fraction remained so and was not absorbed by the crop roots. The economic viability of partially acidulated RP for annual crops should be considered as a function of the price of citric acid- or ammonium citrate soluble P.

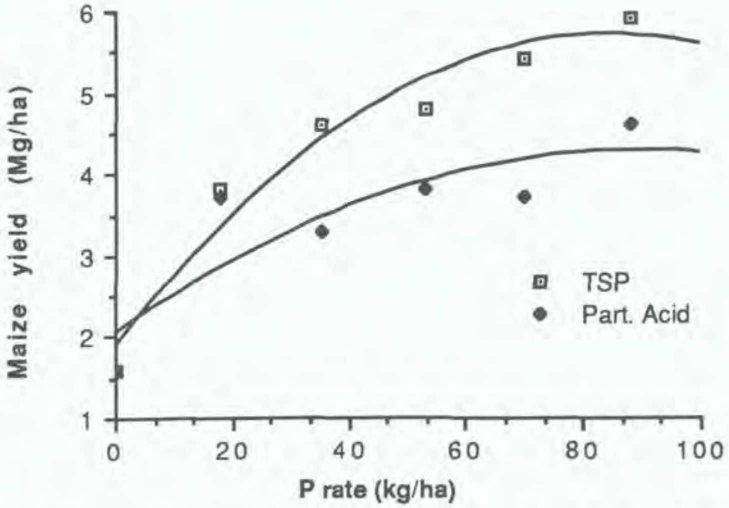

Figure 3. Corn response curve to rates of sidebanded phosphate fertilizers (TSP and partially acidulated phosphate) (Raij and Feitosa, 1986).

Several acid materials other than sulphuric acid have also been evaluated for the solubilisation of Brazilian RPs: phosphoric acid, nitric acid, urea nitrate, urea phosphate, and ammonium bi-sulphate. Rock phosphates (Araxá, Catalão, and Patos de Minas), partially acidulated with phosphoric acid, were studied by Barreto (1977), Franco (1977) and Ferreira and Kaminski (1979). The efficiency of these products was also found to be proportional to the quantity of P solubilized by the acid. Nitrophosphates are highly unstable and hygroscopic. According to Malavolta (1981), nitrophosphates do not present advantages, compared to ammonium phosphates, in agronomic aspects. Goedert and Souza (1986) concluded that Patos RP acidulated with nitric acid presented similar AEI to RP acidulated to the same level (molar ration H^+/P_2O_5) with sulphuric acid. Mixtures of RP with urea nitrate or urea phosphate have been of low efficiency in P solubilization. Several products obtained with ammonium bi-sulphate, mixtures of ammonium sulphate (AS) and sulphuric acid (SA) in various ratios of H^+/P_2O_5 and AS/SA, have received the name RNC (Not Conventional Route). Agronomic studies indicated the potential of this route to solubilize the phosphate rocks with high percentages of iron and aluminum impurities, such as the Patos deposit (Table 4). Phospho-nitro-sulpho-calcium is an alternative product in the ammonium phosphate production route, having the same ratio N/P_2O_5 as ammonium diphosphate (DAP). This product, which is less water-soluble than ammonium phosphate, has shown similar efficiency to TSP.

Table 4. Summary of agronomic efficiency of several phosphate fertilizers for annual crops. (Goedert et al., 1986).

Source of P	Agronomic efficiency (%)
Magnesium thermophosphate	108
Superphosphates and ammonium phosphates	100
Urea phosphates	100
Phospho-nitro-sulpho-calcium	85
RNC	77
Partially acidulated (phosphoric)	75
Partially acidulated (sulphuric)	60
Partially acidulated urea "adutos"	45
Brazilian RP	25

Research involving thermophosphates in Brazil started twenty years ago with sugar cane (Alvarez et al., 1965). Although thermophosphates do not contain water soluble P, when applied as powder (75% smaller than 0.15 mm sieve), the AEI for annual crops has been equivalent or even superior to superphosphates (Tables 3 and 4). An increased soil pH, high percentage of calcium, magnesium, silica and also some micronutrients can explain the observed AEI above 100%. Magnesium thermophosphates are excellent P sources for acid soils of the Cerrado region of Brazil. The main limitations are the comparatively high price, the low percentage of total P, and the requirement of application as a powder.

Pasture species. A literature review of the application of RPs to pasture can be summarized as follows (Sanzonowicz and Goedert, 1986):

a) the efficiency of the phosphates is dependent upon soil type, mainly on texture and acidity;

b) the level of AEI is dependent upon the rate of application, and is very low for low rates, leading to problems of stand establishment;

c) the great majority of field experiments is of short duration, not allowing evaluation of residual effects;

d) the average values for AEI, calculated from several greenhouse and field experiments, was around 30%, confirming that Brazilian RPs are of low and slow solubilization in these soils.

Additional insight in this subject can be obtained by data presented in Figure 4 and Table 5 (Sanzonowicz et al., 1987). A positive response of *Brachiaria decumbens* dry matter yield up to the highest P rate was observed, irrespective to the phosphate fertilizer used. The effect of P sources tested on total dry matter yield were: SSP = magnesium thermophosphate > Gafsa RP > Araxá RP. At the lowest rate of applied P (39 kg ha^{-1}), Araxá RP had an extremely low AEI (1%) compared to Gafsa RP (66%), and SSP (100%), mainly due to the low initial solubilization and poor stand establishment. However at higher rates of 150 and 610 kg ha^{-1}, the AEI of the Brazilian RPs reached 61% and 88%, respectively. These results are important for any final economic evaluation of the use of this low solubility RP in the Cerrado region of Brazil. Other experiments involving *Brachiaria humidicola* and *Andropogon gayanus* conducted in these soils, have shown good results upon application of these natural RPs, as indicated in Table 3 and Figure 5.

In conclusion, the direct application of low solubility Brazilian RPs for pasture production in the Cerrado region is frequently a viable practice if one considers the following aspects:

a) The pasture species must have high P absorption capacity and acid tolerance. Most of the cultivated brachiaria and andropogon species are included in this category.
b) Low solubility RPs have higher efficiency in acid, clayey and high P fixing soils;
c) The efficiency is dependent upon P application rate. In clayey soils with low available P, rates below 40 kg ha^{-1} do not seem viable.
d) To increase the solubilization velocity, the product must be finely ground (85% smaller than 0.075 mm sieve), broadcasted, and mixed with a great soil volume.
e) The slow establishment of pasture stands RP fertilisation, can be improved with initial small applications of water soluble P.
f) While the economic viability is specific for each situation, the price per unit of total P of the RPs should be less than half of the price per unit of total P of a soluble source (e.g. superphosphate).

Table 5. Cumulative dry matter yield of Brachiaria decumbens during ten years (twenty three harvests), average for three lime levels, as response to sources and rates of phosphate fertilizers applied in the first year. Clayey Dark Red Latosol under Cerrado (Sanzonowicz et al. 1987)

Phosphate	Rate of P	Cumulative Yield	Agronomic Efficiency Index (AEI)
	kg ha^{-1}	Mg ha^{-1}	%
Test plot (-P)	0	17.7	-
Araxá phosphate	40	17.8	1
Araxá phosphate	150	47.6	61
Araxá phosphate	610	92.4	88
Gafsa phosphate	40	24.7	66
Gafsa phosphate	150	63.3	93
Gafsa phosphate	610	99.9	97
Thermophosphate	40	25.6	75
Thermophosphate	150	64.8	97
Thermosphosphate	610	109.4	108
Single superphosphate	40	28.3	100
Single superphosphate	150	66.5	100
Single superphosphate	610	102.8	100

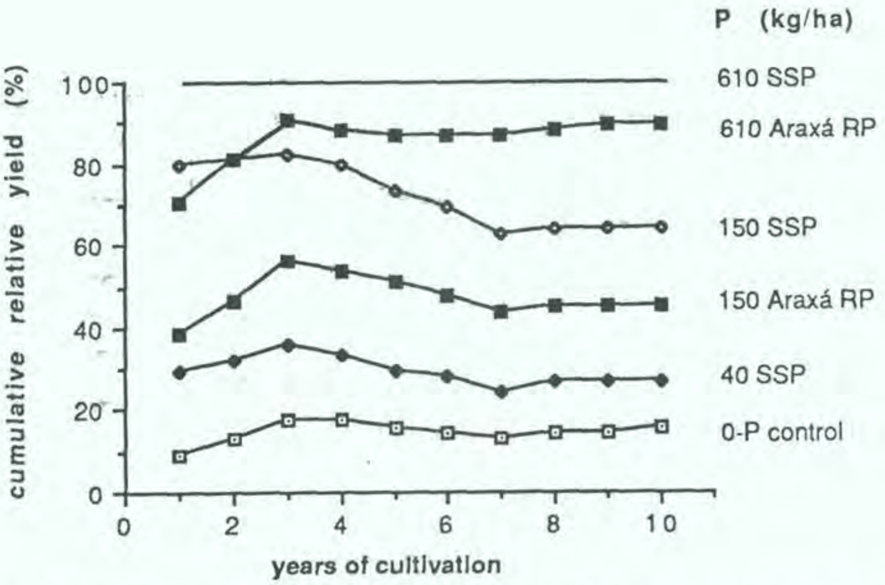

Figure 4. Cumulative relative yield of *Brachiaria decumbes* during ten years, as a function of sources and rates of phosphate fertilizers applied in the first year in a Dark Red Latosol under Cerrado. Sanzonowicz et al. (1987).

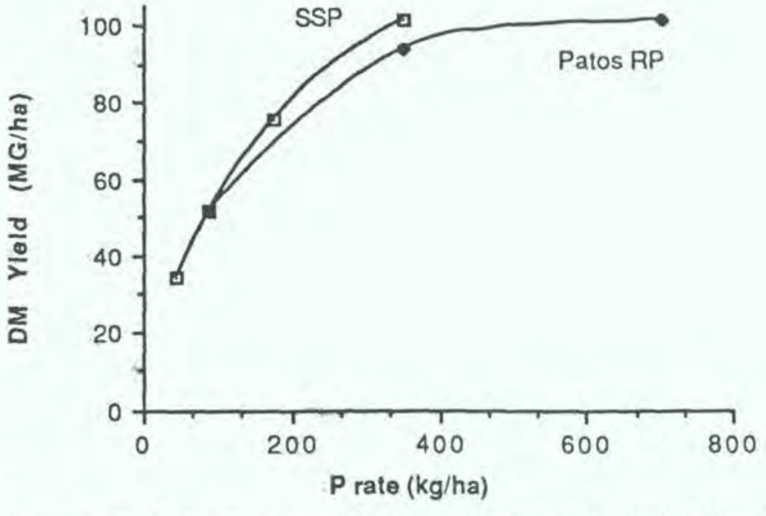

Figure 5. Cumulative dry matter yield of *Brachiaria humidicola* during seven years, as a function of rates of phosphate fertilizers (SSP and Patos Phosphate). Fertilizers were applied two years before pasture formation in a clayey Red-yellow Latosol under Cerrado (Sousa et al., 1987).

Perennial crops and reforestation. Agronomic evaluations of P sources for perennial crops (fruit crops, coffee, reforestation etc.), have been fewer than for annual crops and pastures, and in general, these studies mainly refer to initial growth and development parameters. Few data exist on yield and product quality.

Braga et al. (1977) evaluated five P sources and five rates of P to produce seedlings of *Eucalyptus camaldulensis*. Data presented in Table 6 indicated the possibility of replacement of soluble sources by the Araxá RP, and the low efficiency of Patos phosphate. In another study involving *Eucalyptus saligna* with 2.5 g of Araxá phosphate per seedling, the seedlings reached 29 cm after 90 days, as compared to 15, 16, 21 and 15 cm for the same rate of Patos, Catalão, Tapira and Tufito RPs, respectively. However, field evaluation of height after 790 days for the same species, showed that the 250 g SSP rate per plant were slightly better than 500 g of these natural RPs per plant (Braga and Rocha, 1979).

Table 6. Average height of *Eucalyptus camaldulensis* seedlings, 115 days after application of different phosphate sources and rates (Braga et al. 1971).

Rate applied	Araxá phosphate	Single super-phosphate	Triple super-phosphate	DAP	Patos "Tufito" phosphate
g pot^{-1}			cm		
0.35	1.95	1.80	4.10	2.23	2.55
0.70	7.13	3.83	3.93	3.90	5.05
1.05	5.55	4.00	7.63	4.83	4.28
1.40	7.60	4.00	8.38	6.03	4.03
ACI	1.10	1.00	1.53	0.66	0.38

A more recent study has proven the possibility of RP use in reforestation, mainly when associated with N-P-K (10-28-6) + B + Zn at two different locations in the cerrado of Minas Gerais State. Patos phosphate was not as efficient in increasing the yield of *Eucalyptus grandis*, as Araxá phosphate (Table 7). However, broadcasting of 2 Mg ha^{-1} of Patos phosphate, complemented with 150 g NPK + B + Zn per plant, led to 90% more yield than the mixed fertilizer alone. Broadcasting of 2 Mg ha^{-1} of Araxá phosphate gave yields similar to the mixed fertilizer, but the yield was doubled when this rate of Araxá phosphate was associated with 75 g mixed fertilizer per plant.

Recent partial conclusions on the use of low solubility Brazilian RPs in reforestation with eucalyptus are reported for seven locations in Minas Gerais State (UFV, 1985). These data suggest that less soluble sources of P, particularly Araxá phosphate, can substitute for more soluble sources, if the eucalyptus growth period is at least three years. The yields, however, can be improved by a combination of broadcast low solubility RPs, with more soluble sources at planting. For sandy soils, soluble P sources are more efficient.

Table 7. Eucalyptus grandis yield (at 26 and 24 months), as a function of various fertilizer treatments at Bom Despacho and Carbonita, Minas Gerais. (Adapted from Rezende et al. 1982).

Treatment	Bom Despacho (Patos Phosphate)	Carbonita (Araxá Phosphate)
	$m^3\ ha^{-1}$	
A) 2Mg ha^{-1}PR + 150g·$plant^{-1}$NPK+B+Zn	86.8a	92.3ab
B) 2Mg ha^{-1}PR + 75g·$plant^{-1}$NPK+B+Zn	68.8b	100.8a
C) 2Mg ha^{-1}PR	13.2d	49.3cd
D) 1Mg ha^{-1}PR + 150g·$plant^{-1}$NPK+B+Zn	59.2bc	62.2bc
E) 1Mg ha^{-1}PR	17.2d	29.0de
F) 150g $plant^{-1}$NPK+B+Zn	45.7c	49.7cd
G) Test plot (no fertilizers)	8.4d	2.2e
CV (%)	14.7	25.6

Values in columns followed by same letters are not different at the 5% level.

While coffee is not considered responsive to P, there are data of remarkable positive effects of P fertilisation during initial stages of this crop on Cerrado soils (Hara et al., 1974; Souza and Caixeta, 1974). In this last study, five years coffee production data showed a linear response up to 50 g y^{-1} per plant as SSP. Also Pereira et al. (1979), observed increases in coffee yield ranging from 50 to 148%, due to 13 g y^{-1} per plant in soils from the Zona da Mata of Minas Gerais State. A more recent study indicated that P is the most important plant nutrient for coffee during the first six months in a Dark Red Latosol of Minas Gerais (Santinato et al., 1980). Coffee production response or agronomic efficiency of P are dependent upon soil type (Guimarães 1986). In comparisons between sources of P for coffee , the great majority of field experiments has demonstrated the possibility of low solubility Brazilian RPs to substitute for water soluble P sources. The AEIs have been comparable when twice as much total P as RP was used than as SSP, and four times the total P as RP in relation to TSP (Silva et al., 1983; Santinato et al., 1983; Viana et al., 1985; Santinato et al., 1985).

In fruit crops, mainly citrus, Souza (1976) concluded that P was the most important nutrient during the first three years in Cerrado soils. Maximum aerial plant development was achieved with 190 g total P per plant. Souza et al. (1978), compared different fertilization regimes for "Pera Rio" citrus beds on Cerrado soils of Minas Gerais. Data for two harvests gave the following yield increases relative to the test plot (no P): 900 g

SSP (169%); 2,670 g Araxá phosphate (223%); 640 g bone meal (165%); 20 liter cow manure (290%). The best treatment was a combination of 1,330 g Araxá phosphate + 400 g SSP + 20 liter cow manure (308%).

For both fruit crops and coffee plants, available research data suggest that combination of natural RPs with more soluble P fertilizers is a good approach for plant bed preparation. For bed preparation of coffee and fruit crops in Minas Gerais and Goias States it has been recommended that one half of P should be as Brazilian RPs and the other half as more soluble sources (CFSEMG, 1978; CFSEMG, 1989; CFSG, 1988). For maintenance fertilization, P fertilizers should be in more soluble forms since the common use of lime reduces the agronomic efficiency of Brazilian RPs.

Conclusions

Based upon the data presented and discussed in this literature review the following conclusions can be drawn:

a) The agronomic efficiency of phosphate fertilizers produced in Brazil for annual crops is, in general, proportional to the percentage of soluble P.

b) Brazilian RPs generally have low solubility and low agronomic efficiency for annual crops. Their use is justified only for the establishment of acid tolerant pasture, and for certain perennial crops.

c) Even if the positive industrial aspects of partially acidulated RPs are taken into account, these products will only be viable for annual crops if the costs for the farmer are established as a function of the soluble P fraction. Partially acidulated RPs, however, can be adequate for pastures and perennial crops.

d) Thermophosphates represent an excellent agronomic option for most acid soils of the Brazilian Cerrado, justifying increases in production in Brazil.

e) The traditional soluble sources (superphosphates and ammonium phosphates) are highly efficient and justify their production. The myth that being originally developed for climate and soils from the temperate regions they are not adequate for Brazilian conditions is not justified.

f) Long term field experiments (10 or more years) have shown cumulative absorption of applied soluble P in order the of 60 to 70%. This indicates that P "fixation" problems are minimized with cropping time.

g) Considering the diversity of soils, climates and crops in Brazilian agriculture, production and commercializing several types of phosphate fertilizers in Brazil is justified.

h) The agronomic efficiency of phosphate fertilizers produced and marketed in Brazil is much more a function of adequate soil and crops management than change in industrial technology. In the specific situation of water-soluble P fertilizers, the use of low rates of lime is one serious limiting factor to increases in the agronomic efficiency of P.

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PHOSPHORUS DYNAMICS IN CROPPING SYSTEMS UNDER HIGH AND LOW INPUT TECHNOLOGY IN VENEZUELA

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Introduction

The loss of plant nutrients in eroded sediments has been reported in many countries. In the United States, Massey, et al. (1958), studied runoff samples from experimental plots at four locations in Wisconsin, and they found, at the average values of erosiveness encountered in their study, that the eroded soil material contained 2.1 times as much organic matter, 2.7 times as much N, 3.4 times as much available P, and 19.3 times as much exchangeable K as the residual soil.

Barrows and Kilmer (1963), reviewed data relating to water erosion losses of organic matter and plant nutrients and concluded that significant losses of organic matter, with concomitant removal of N and P, occurs. Large amounts of potassium were removed, but only a small percentage of this was in an exchangeable or plant available form. Thomas et al. (1968) studied, nutrient losses from 18 runoff plots on a 3 percent slope in Georgia and found that the amounts in of P, K and Ca in the washoff were about 0.03, 0.13 and 1.5 kg ha^{-1} respectively, for every Mg ha^{-1} soil loss.

Munn, et al. (1973), in an effort to obtain quantitative information on P and soil movement, treated three soils with P in bare and cropped microplots and subjected them to simulated rainfall. Fractionation of sediments showed the dominant P fractions and their increases with rate and P application to be directly related to extractable Iron (Fe) and Aluminum (Al) content, pH, and percent base saturation of the original soils. White and Williamson (1973) found that losses of plant nutrients in soil eroded from cultivated land may be similar to average losses that would occur naturally if the area was in prairie that was periodically subjected to fire. Romkens, et al. (1973) studied the effect of tillage methods on the N and P composition in runoff water and runoff sediments from corn plots by applying simulated rain-storms. They found that conventional tillage, in which fertilizer was plowed under, had the highest losses of soil and water but small losses of N and P.

Monke, et al. (1977) described the runoff, erosion, and nutrient movement from inter-rill areas. They found that, although N and P concentrations in the soils with excellent tilth were quite high, losses of these nutrients in sediment runoff were low because of increased infiltration.

In Western Nigeria, Lal (1976a, 1976b) reported nutrient losses in eroded sediments on different slopes and under different soil and crop management treatments. He found that the nutrient losses in eroded soil materials from the mulched and no-till treatments were negligible. From the plowed treatments, greatest losses were of organic matter and total N. The enrichment ratios were 2.4 times for organic carbon, 1.6 times for N, and 5.8 times for available P.

In Belgium, Honotiaux (1980), determined the runoff erosion and nutrient losses on loess soils. His data showed a loss of almost a ton of organic matter and the average ratio of C/N was 8.4 indicating that the mobile matter was well humified.

In Venezuela, Gasperi (1982), has reported estimates of nutrient and organic matter losses in eroded sediments from Quibor, Lara State. He concluded that a erosion washed away 700 Mg ha^{-1} of soil with 17.500, 13, 105 and 55 kg ha^{-1} of organic matter, N, P and K, respectively. When the erosion was severe the amount of soil eroded was 2100 Mg ha^{-1} and the loss of organic matter and nutrients increased to 52.500, 39, 365 and 164 kg ha^{-1}, respectively.

Lobo (1988) working on water, soil and nutrient losses on an Alfisol under different covers concluded that asphalt emulsion and residue covers reduced water, soil and nutrient losses compared with the control. These treatments reduced the water loss by runoff and soil loss up to 50 and 60% and 94 and 98%, respectively. The sorghum crop by itself reduced the water loss by runoff and soil loss up to 9.4% and 42.7%, respectively. Under intensive rain the amounts of available P loss for every treatment were: 10 (control), 9.1 (uncover soil + sorghum), 4.7 (asphalt emulsion + sorghum) and 4.5 kg ha^{-1} (residue cover + sorghum).

The only reference to the Amazon Basin of Venezuela where water and soil loss by runoff is mentioned is Jordan (1984). This paper dealt with shifting cultivation, nutrient cycling and productivity near San Carlos de Río Negro. Soil erosion was not measured but the sharp drop in nutrients in the soil could not be explained by leaching losses,so that surface erosion may have been the cause for this loss.

Many authors have reported that soil management treatments like fertilization obscure the effect of soil erosion and loss of nutrients on soil productivity. The use of high and low input technology in Venezuelan agriculture has shown some problems in nutrient cycling and in the environment. This communication describes nutrient losses in eroded sediments under different soil management in two areas of Venezuela where low and high input technology is used.
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Materials and methods

Nutrient losses in eroded sediments were determined using field plots established on a: (1) Cumulic Haplustoll located at the Experimental field of the Facultad de Agronomía, UCV., Aragua State, (2) Typic Haplustalf at a farm in Chaguaramas, Guárico State. The main soil properties of these two soils are shown in Table 1.

Each runoff plot was surrounded by a metal border and equipped with a separation and collection system for sediments and runoff. In the sediment trap coarser material was collected, in collector 1 the fine material and finally in collector 2 water and some soil particles. The treatments imposed in each location are described in Table 2. All the plots were tilled with a Rototiller down the slope except the forest and minimum tillage treatments. Weed control was with "hierbatox" in the bare plots and with paraquat for the minimum tillage treatment.

The soil and crop management treatments (Table 2) were selected according to the most important crops of each location and with suitable management. The forest treatment was used for comparisons. Composite samples of eroded sediments from the sediment trap and collector 1 and 2 from each of the treatments were collected after every rain. Subsamples were dried, ground and passed through a 2 mm sieve and analyzed for organic carbon by wet combustion, total nitrogen by the Kjeldahl method, available P by the Mehlich I method, and exchangeable Ca^{2+}, Mg^{2+}, and K^+ by ammonium acetate (1N) extraction.

Table 1. Properties of the soils

Property	Cumulic Haplustoll Maracay (0 - 10 cm)	Typic Haplustalf Chaguaramas (0 - 20 cm)
Sand (%)	48.8	29.4
Silt (%)	30.5	26.4
Clay (%)	20.7	44.2
pH (1:1)	7.10	6.65
Organic C (mg g^{-1})	35	13
Total N (mg g^{-1})	4.0	1.6
C/N	8.8	8.1
EC (mmhos cm^{-1} x 10^{-5})	25	90
CEC (meq 100 g^{-1})	31.50	29
Exchange Ca (meq 100 g^{-1})	26.55	12.9
Exchange Mg (meq 100 g^{-1})	4.62	10.41
Exchange Na (meq 100 g^{-1})	0.12	4.34
Exchange K (meq 100 g^{-1})	0.61	0.17
Base Saturation (%)	100	96
$CaCO_3$ Equiv. (%)	1.50	1.0
P soluble in citric acid (μg g^{-1})	253	30

Results and discussion

Nutrient Loss in Eroded Soil Materials

Chaguaramas. Total nutrient losses in eroded sediments were significantly influenced by soil management treatment (Table 3). Collector 1 received the highest amount of nutrient and collector 2 the lowest (Figure 1). A similar result was found in Maracay. In the 6%-slope bare soil treatment, the loss of organic matter was 3.3, 25.3, 243.2 times higher than in the treatments: sorghum tilled, sorghum minimum tillage, sorghum-strip, respectively.

Table 4 shows the same comparison for total N, available P, and exchangeable Ca, Mg, K and it is concluded that the loss of organic matter and plant nutrients showed the following tendency: bare soil > sorghum tillage > sorghum minimum tillage > sorghum in strip cropping. In this regard, the beneficial effect of the strip cropping with negligible nutrient losses, is particularly significant.

Table 2. Soil management treatments and estimated soil erosion.

Treatment	Description	Soil Erosion Mg ha^{-1}
Bare Soil with Gravel (Maracay)	The gravel cover (30-40%) was left over the soil surface. No crop planted.	103.4
Bare Soil without Gravel (Maracay)	The gravel cover was reduced to 15% or less. No crop planted.	248.9
Simple Band (Maracay)	The band was formed with Buffel Grass (Cenchrus ciliaris) and was 0.5 m wide located at the middle of the plot.	62.4
Double Band (Maracay)	Same as simple band but an additional band was located immediately before the sediment trap.	38.7
Corn (Maracay)	Corn was seeded at a plant density of 55.555 plants ha^{-1} and common agronomic practices were applied.	72.0
Multiple Cropping (Maracay)	Weeds were hand-controlled. Corn was seeded in triple rows and squash was planted between them. Bananas were planted at the contour of the plot.	10.0
Rotation-Strip (Maracay)	The center of the plot was seeded to Buffel Grass (Cenchrus ciliaris) and the upper and lower part planted with Corn. Common agronomic practices were applied.	1.12
Forest (Maracay)	The plot was located in a natural forest.	0.12
Bare Soil (Chaguaramas)	The plot was tilled up and down the slope. Weeds were controlled with herbicide. No crop planted.	73.8
Sorghum (Chaguaramas)	The plot was tilled up and down the slope. Sorghum was broadcast at 300.000 plants ha^{-1}.	17.33
Sorghum-Minimum Tillage (Chaguaramas)	Seeds were hand-planted in rows. The soil was untilled between rows and residues left on the soil. Common agronomic practices were applied.	2.13
Strip - Cropping (Maracay) (Chaguaramas)	The lower half of the plot was fallow and the upper half was planted with sorghum broadcasted in Chaguaramas and contour-planted under conventional tillage in Maracay.	1.48 (M) 12.0 (C)

Maracay. The loss of organic matter and plant nutrient elements in eroded soil materials from the nine different soil management treatments are shown in Table 5. Using collector 1 as reference and comparing some of the treatments, the losses from the bare soil without gravel were at least two fold higher than the loss produced in the bare soil with gravel. However, both treatments showed high soil erosion (Table 2) and high losses of organic matter and nutrients. The corn treatment followed the bare soil treatments in losses of organic matter and nutrients. Planting down a 15%-slope using conventional tillage was one of the reason of those losses. It was evident that the eroded soil materials (Table 2) were reduced when a simple or double band of buffel Grass was planted. Treatments with any form of plant cover (strip, multiple, rotation, forest) decreased the losses to negligible amounts.

Table 3. Nutrient loss in eroded soil materials in Chaguaramas, Venezuela, as affected by soil management treatments

Treatment	Collector	Organic Matter	Total N	Available P	Exchangeable Ca	Exchangeable Mg	Exchangeable K
		Kg ha^{-1}					
Bare Soil	*S.T.	184.0	17.4	0.70	21.7	4.24	1.00
	1	1313.0	141.4	2.39	180.6	23.41	14.37
	2	147.6	10.8	0.20	8.70	1.23	0.66
Sorghum Tilled	*S.T.	1.90	2.10	0.10	2.46	0.42	1.23
	1	392.9	36.0	1.39	49.86	8.18	2.46
Sorghum Minimum Tillage	1	52.0	4.7	0.19	5.28	6.00	0.29
Sorghum Strip	*S.T.	2.4	0.3	0.01	0.29	0.06	0.02
	1	5.4	4.5	0.11	3.35	0.61	0.50

*S.T. = Sediment Trap

Table 4. Bare soil treatment compared to other treatments in the loss of nutrient in eroded sediments from collector 1 in Chaguaramas. Figures indicate how many times higher the loss from bare soil is relative to the treatment indicated.

	Treatments		
	Sorghum Tilled	Sorghum Minimum Tillaged	Sorghum - Strip
Organic Matter	3.3	25.3	243.2
Total N	3.9	30.1	31.4
Available P	1.7	12.6	21.7
Exchangeable Ca	3.6	34.2	53.9
Exchangeable Mg	2.9	3.9	38.4
Exchangeable K	5.8	49.6	28.7

Table 5. Nutrient loss in eroded soil materials in Maracay, Venezuela, as affected by soil management treatments

Treatment	Collector	Organic Matter	Total N	Available P	Exchangeable		
					Ca	Mg	K
		Kg ha^{-1}					
Bare soil Without gravel	*S.T.	1108	123	27.8	128	10.9	18.9
	1	6042	614	110.6	574	52.9	96.3
	2	563	53	7.1	54	5.1	8.6
Bare soil With gravel	*S.T.	1036	110	26.2	100	9.1	13.0
	1	3022	185	47.5	272	25.5	34.5
	2	258	26	3.6	25	2.5	4.0
Corn	*S.T.	1033	104	13.8	93	6.8	14.5
	1	204	192	16.0	212	13.5	24.5
	2	197	18	1.6	20	1.3	2.1
Simple band	*S.T.	806	91	20.6	85	7.5	10.5
	1	1074	62	18.9	104	8.7	12.5
	2	269	27	3.1	29	2.2	3.2
Double band	*S.T.	499	54	9.7	54	3.8	6.6
	1	856	59	12.7	90	6.7	11.0
	2	77	8	0.9	8	0.6	1.0
Strip cropping	1	450	38	4.3	1	2.9	4.9
	2	91	9	0.6	9	0.6	0.9
Multiple cropping	*S.T.	17	2	0.2	1	0.2	0.2
	1	50	49	6.5	48	3.8	5.7
	2	91	9	0.8	9	0.6	1.0
Rotation-strip	*S.T.	7.4	0.7	0.1	1	0.1	0.1
	1	71	7	0.3	10	0.4	0.7
	2	1.8	0.2	0.1	0.2	0.1	0.1
Forest	1	10	0.9	0.1	1	0.1	0.1

*S.T. = Sediment Trap

Conclusions

The results presented in this paper are in accordance with Lal (1979b) that the organic matter and nutrient element loss in the form of eroded sediments is quite high and this may be one of the major causes of fertility depletion of tropical soils where adequate erosion control measures are not adopted.

The results also showed in both locations that using high-slope bare soils produced the highest loss of eroded soil material and organic matter and nutrients with it. It was evident that these losses decreased from medium to negligible amounts when any form of plant cover was used.

Soil erosion of 10 Mg ha^{-1} reported in the multiple cropping system in this paper similar to the conuco treatment reported by Jordan (1984) in which assumptions of soil erosion were discussed. These factors will be important reasons to explain decline in crop productivity under shifting cultivation in the Amazon.

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PHOSPHORUS CYCLING IN TROPICAL PASTURES

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Introduction

More than one third of Tropical America is occupied by Oxisols and Ultisols, characterized by high acidity and low nutrient status, but often with excellent soil structures (Sanchez and Isbell, 1979; Sanchez and Salinas, 1980) the remaining two thirds, are Inceptisols, Entisols and Alfisols, usually with an acid A horizon. Large areas of Oxisols and Ultisols are covered by forest. In the natural forest ecosystem plant growth depends almost totally on recycled P from organic matter. Microorganisms are involved to a large extent in the recycling process (Salinas and Aif, 1989; Spain and Salinas, 1984). Among the soil microorganisms, vesicular arbuscular (VA) mycorrhizal fungi are of outstanding importance for P uptake from the soil solution and for P transport to plant roots (Abbott and Robson, 1982; Jehne, 1980).

When forests are cut to introduce ranching, the main initial activities are wood extraction, burning of residues, and pasture establishment for beef production (Myers, 1980). However, beef production is often limited by low productivity of unimproved pastures. One of the most limiting nutrients for forage establishment and production in Oxisols and Ultisols is phosphorus due to low P availability in those soils. In many cases this is due to P fixation associated with high contents of Al and Fe oxides of these soils (Sanchez and Salinas 1980), but many of the Ultisols in the Amazon Basin have a relatively low capacity of P fixation (Sanchez and Benitez, 1983).

Possible strategies for the improved utilization of pastures include P fertilizer applications to replace all P extracted from the soil (Sanchez and Salinas, 1980). However, phosphorus fertilization practices, with few exceptions, are restricted by the poor infrastructure existing in the forest regions. Consequently, one approach to better sustained production of tropical pastures is thought to be an efficient management of the organic P recycling combined with a strategical use of small amounts of P fertilizer.

Phosphorus cycling in a soil-plant-animal system

The cycling of P is influenced by its low solubility and low mobility in the soil (Wilkinson and Lowrey, 1973). Hence, total amounts of P in an ecosystem tend to be relatively constant throughout long periods of time, but the quantity of P in different pools may vary from year to year (Jones and Woodmansee, 1979).

Phosphorus removal by harvesting crops or cutting grasses creates an external P requirement, while the P return through the plant and animal residues to the pasture systems under grazing, results in a relatively closed P cycle (Mays et al, 1980). In cattle raising systems there are three P stores (Figure 1): biomass (plants and animals), plant and animal residues, and soil (including ash from recently burned forest). Between them there is a natural mechanism of P recycling.

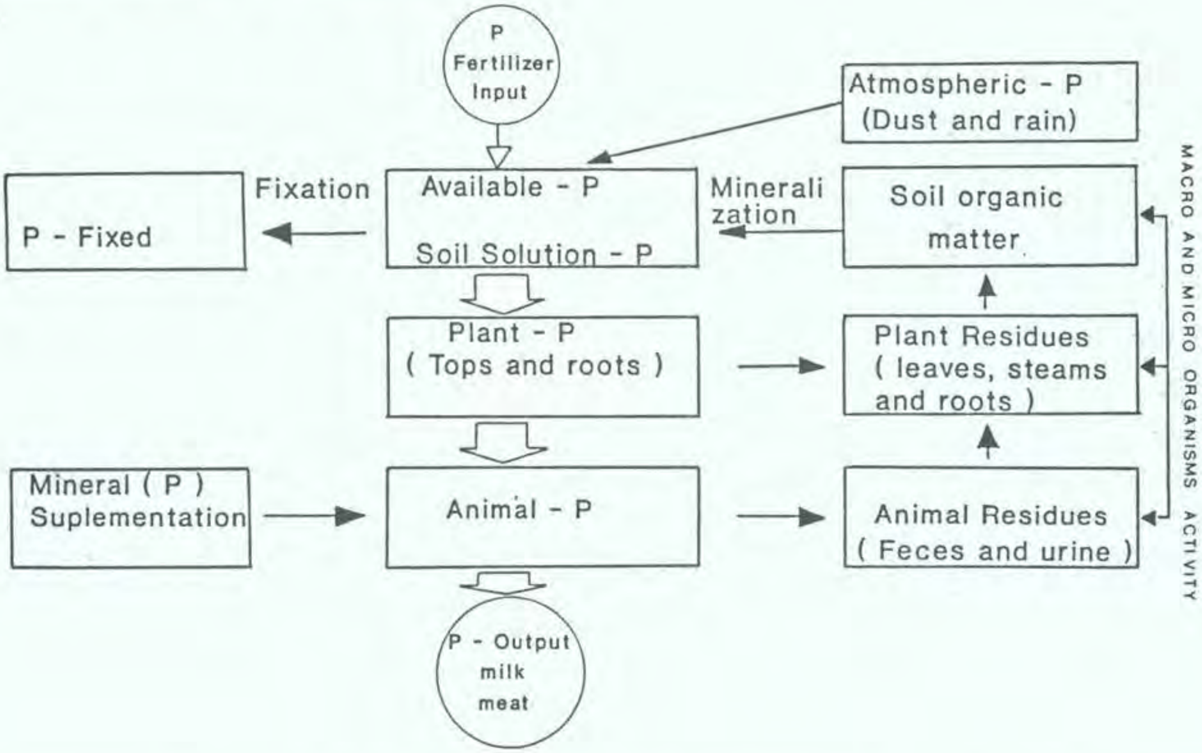

Figure 1. Diagram of P cycling in a soil - Plant - Animal system (adapted from Blair et al., 1976)

Phosphorus inputs. Most available P comes from the ash produced when the forest is burned (Nye and Greenland, 1960), and from mineralization of organic matter. Under acid soil conditions, Seubert et al. (1977) in Perú and Prado and Alvarado (1989) in Bolivia found P additions of 19 and 9 $kg \cdot ha^{-1}$ after burning a primary and a secondary forest, respectively. The input of atmosphere P from dust and/or rain is below 1 $kg \cdot ha^{-1} y^{-1}$ (Katznelson, 1977, Duvigneaud and Denalyer De Smet, 1970 referenced in Wilkinson and Lowrey, 1973). Contributions of P to pasture from weathering of primary soil minerals are of very little significance in P recycling (Cole et al., 1977; Katznelson, 1977), particularly when the parent material is low in P.

Great efforts have been undertaken to identify the organic components of P that can be found in soil (Jones and Woodmansee, 1979), and it is possible that the contribution of P from this source to plants in tropical areas may be significant. Factors such as initial content of organic P, total P, biomass P content, temperature, and time determine whether or not a net mineralization of P from plant and animal residues takes place (Mays et al., 1980). It has been suggested that net mineralization is unlikely to occur when organic P contents are less than 0.09% during the first weeks of decomposition (Floate, 1970). Under temperate conditions, when P content of the residues is less than 0.03% there is a net reduction of labile soil P (Singh and Jones, 1976). Strong interactions between mineralization and P adsorption processes have also been reported. Overall, soil organic P is considered important for pasture production (Blair et al., 1976).

When animals consume forage, some of the P is stored in their organism and may be exported from the system as milk and or meat, while the rest returns to the soil via excreta. Phosphorus is mainly excreted as feces, and only traces can be found in urine (Barrow and Lambourne, 1962). Distribution in space and time of residues from plants and animals is different and depends considerably upon pasture management, rate of consumption and other factors. The redistribution of P in space is a problem since the animal eats from the entire grazing land but deposits feces on a limited percentage of the total area. Because of the uneven distribution and low mobility, P from manure will significantly affect only long-term P fertilizer requirements (Wilkinson and Lowrey, 1973). In addition, although the availability of P in feces mixed with soil is high, when the feces lay on the ground (without mixing) most of the P is inaccessible to the grass roots. Manure black beetles, earthworms and other soil organisms increase the speed of mineralization of feces by burying (Bromfield, 1961; Bornemissza and Williams, 1970).

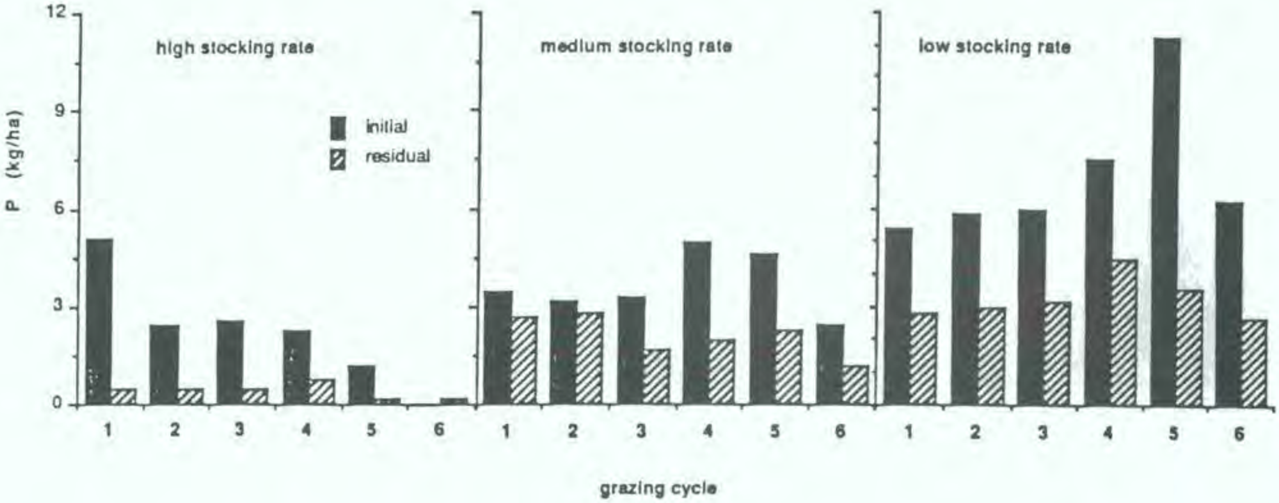

Figure 2. changes over time of P in initial and residual forage under three stocking rates.

During pasture trampling, the animal breaks the aerial part of the pasture, and these organic materials, together with dead roots from the plants' reaction to defoliation by grazing, are mineralized, and part of the P is taken up again by the plant. About 77% of P in litter, and 79% in dead roots become available for the next year (Jones and Woodmansee, 1973). About 60 to 80% of the total P in the aerial part of the plant is soluble in water, and most of it is inorganic P (Bromfield and Jones, 1970).

Different grazing pressures affect the dynamics of P by changing the relative nutrient accumulation, stability, or losses in the pasture system (Figure 2). As the grazing pressure increased, a significant reduction in P was observed after each grazing cycle, primarily associated with the reduction in available forage. Under medium pressure available P in forage remained unchanged after each grazing cycle, while under low pressure there was a tendency to accumulate P in forage. This suggests that the medium pressure system would be adequate to maintain the pasture stable over some time. These results indicate that pasture management is one of the most important factors to guarantee the cyclic return of P to the system and thus the ability to establish a pasture with the minimum requirements of maintenance fertilization over time (CIAT, 1987).

Phosphorus availability in highly weathered soils, depends on the clay and sesquioxides content. Alvarado et al. (1989) showed that aluminum saturation increased along with the clay content reducing available P and increasing P sorption in three Bolivian soils (Table 1).

Table 1. Availability and sorption of P as a function of clay content and Al saturation in three different acid soils in the Chapare region of Bolivia.Adapted from Alvarado et al.(1989)

Location	Clay Content (%)	Al Saturation (%)	Maximum P.Sorption (ppm)	Available P (ppm)
Paractito	28	96	945	9
Mariposas	27	89	958	5
Villa Tunari	8	76	451	21

Phosphorus output. In cattle production systems, P is lost from the soil when complete plants are taken off (hay), when meat or milk are exported, or by leaching or erosion. Losses are greater for dairy cattle for beef production. Katznelson (1977) reported P losses in animal products from 1 to 10 kg $ha^{-1}y^{-1}$, depending on pasture production and stocking rate. On the basis of live weight, the animal approximately contains 0.75% of P. With a gain in live weight of 400 kg $ha^{-1}y^{-1}$, the extractions from the grazing field would be 3.0 kg P (Table 2).

In well drained agricultural soils, P is not subject to significant losses by leaching (Black, 1968; May et al., 1980). Wilkinson and Lowrey (1973), using a hypothetical system of pastures with cows and calfs, determined that P losses of the system by means of leaching were of the order of 0.3 kg ha^{-1}. Under conditions of high rainfall (more than 4.500 mm) and low soil fertility, available P increased after burning the forest but the increase was leached or sorbed after a year (Figure 3), affecting the establishment of the grass. Bromfield (1961) reported that 69 to 80% of total P in the plant can be washed out of vegetation when it is in a dormant state or is dead. Intensity and duration of rain, and the interval between dormancy or senescence of the tissue and the first precipitation affect the amount of P returned to soil or lost by run-off. During short periods of time, the losses by leaching and erosion seem not to be significant in annual pastures, but evidence for that is partially indirect (Jones and Woodmansee, 1979), and the combined losses by leaching and by P fixation, may make it necessary to apply P (CIAT, 1983). According to Alexander (1977), there is no evidence that P is lost in gaseous form, even under anaerobic conditions.

Table 2. Concentration of some nutrients in the animal body, and extraction from a Pasture system, based on a production of 400 Kg LW $ha^{-1}y^{-1}$

	Concentration g Kg^{-1}	Extraction Kg ha^{-1} y^{-1}
N	24.0	9.5
P	7.5	3.0
K	2.0	0.8
Ca	13.5	5.4
Mg	0.4	0.2
S	0.1	0.4

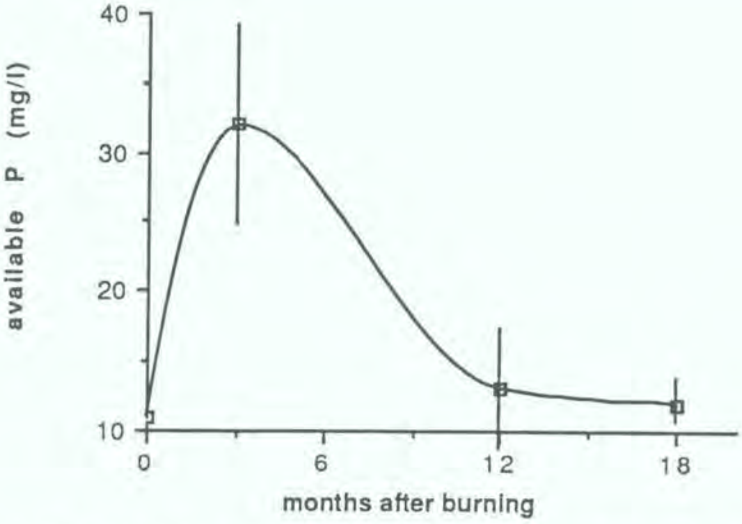

Figure 3. Fertility decline after burning a secondary forest. (The bars are the standard deviation of the mean)

Strategies for an efficient P cycle in tropical pastures

Tropical forage species offer potential advantages to allow a more efficient P cycle. This potential depends mainly on the selection of species adapted to the environment, a combination of stable, persistent and potentially productive associations, an intelligent management of the pastures, and strategic use of fertilizers.

Selection of forage species

Tolerance to soil acidity. One of the most important factors for adaptation of species to the humid tropics are their tolerance to strongly acid soils, with pH lower than 5.0, and to high aluminium levels. Selection of germplasm tolerant to acidity is not difficult (Spain, 1979). There are thousands of introductions of forages grasses and legumes in several institutions, collected from humid tropical regions, with poor and acid soils. Plants have evolved during millenia in an environment where adaptation to acid soil conditions was required.

Tolerance to acidity does not mean that plants are immune to Ca and Mg deficiencies which almost always accompany acid soil. But they are required as nutrients and not as soil amendment, and therefore the applications required are measured in kilograms, not in tonnes.

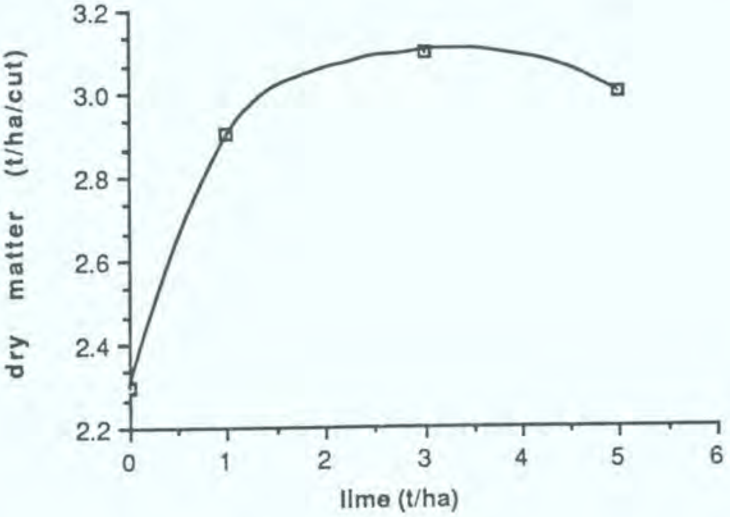

Figure 4. Response of B. decumbens to liming in a degraded grassland. (Adapted from Saavedra, 1981).

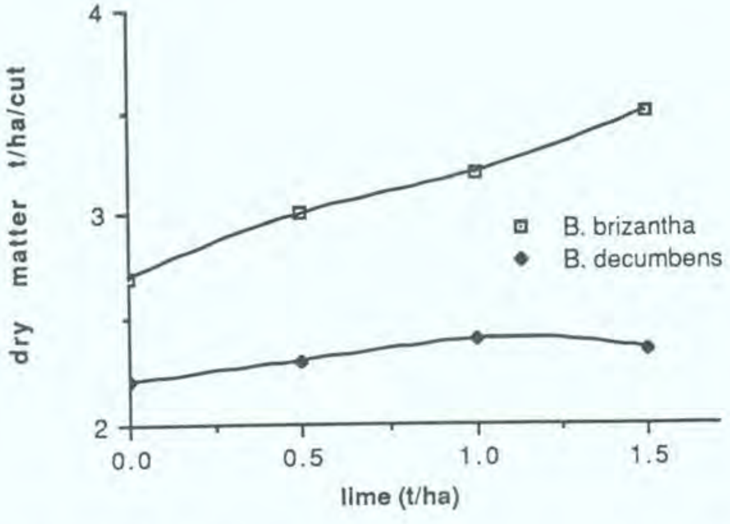

Figure 5. Response of B. brizantha and B. decumbens to liming (with the addition of 20 Kg P ha^{-1}) on a recently cleared area. (Adapted from Vallejos and Saavedra, 1986).

Liming. In many cases lime is expensive because sources are far and/or there is a lack of adequate transport. Depending on the soil texture and the amount of rainfall of the area, liming is effective only up to the depth of tillage, and generally, the subsoil is still very acid. Therefore, if species are not tolerant, they will not be able to take advantage of subsoil humidity during drought periods, neither will they be able to absorb nutrients that leach through the soil profile.

Brachiaria spp. responded to liming differently, depending on the age of the grassland. When the grassland was already degraded, B. decumbens responded to liming dramatically (Saavedra, 1981), but not on recently cleared areas (Vallejos and Saavedra, 1986). However, B. brizantha responded even in recently established stands (Figures 4 and 5).

Tolerance to low soil fertility. Adaptation of plants to poor soils is one of the most important characteristics that influence the efficiency of nutrient recycling in a given environment. Plants can compensate for low nutrient availability by means of a very slow growing rate. Some plants grow slowly throughout their life, and therefore produce little biomass. The most interesting forage plants, are those, which have the capacity of growing slowly during a stage of nutrient accumulation in the biomass, but once enough minerals have been accumulated, it grows much faster, and therefore attains a good production potential. Many species can attain high a biomass production due to efficient nutrient retranslocation within the biomass itself, and the recovery of small amounts of nutrients that were being lost to the soil by means of an efficient mycorrhizal root system. Production is one of the key components of efficiency, along with conservation of the minerals in the systems. A plant or a vegetation complex may conserve nutrients very well, but it cannot be considered efficient if production is too low.

Tolerance to poor drainage. Many areas of the humid tropics are poorly drained; some are permanently, others seasonally waterlogged. Soils under these conditions are periodically flooded and then enriched with sediments that may contain P. Therefore soil fertility status is not poor, but the grasses must tolerate water stress conditions for different periods. Some species like Brachiaria humidicola, B. mutica and Hermarthria altissima are known for their ability to survive and yield well in these systems.

Tolerance to biotic factors. In practice, it has been much more difficult to find germplasm tolerant to diseases and pests than to edaphic and climate factors. The problem of diseases is considerable in a forest ecosystem. The importance of tolerance to biotic factors is obvious because the plant has to grow vigorously (when there is enough humidity) to fulfill its role in recycling and conservation of resources. A diseased plant or a plant exhausted by insect attacks will not be efficient enough. The use of chemical products to control pests and diseases is not generally economical in tropical pastures. Hence, selecting resistant or tolerant ecotypes to a given disease or pest seems to be the best approach to follow.

Grass-legume mixtures. The grass-legume association is one of the strategies used to lower the input levels in pasture systems. In a good grass-legume association, the legume can provide N to the grass, and also contribute with a high percentage of total protein offered to beef cattle. There is a considerable number of references in the literature to the capacities of grasses and legumes to compete for K and P. The work of Gray and co-workers (1953) established that legumes are more competitive than grasses under situations of low soil P availability. In an "improved pasture" of grass and legume, P fertilization would be required only after the fourth or fifth year, once there is a lowering on pasture production along with a reduction in the rate of available P in the soil. Periodic P fertilization along with introduction of a legume and good pasture management will improve the mechanism of P recycling in the soil-plant-animal system (Figure 6), and it is possible that its productivity remains satisfactory for some decades, if the pasture is submitted to optimum conditions of grazing pressure.

In the highlands, tropical grasslands are associated with forest species, particularly along fence lines. In the lowlands, tropical tree legumes such as Inga spp, Erythrina spp, and Gliricidia spp., in addition to the N-fixation benefit, also explore deeper soil horizons, favoring a more complete nutrient cycle when planted as living fences.

Soil fertility management

Relative soil fertility and critical levels. Soil fertility can only be defined relative to the plant which is cultivated (Spain, 1981). An absolutely inadequate soil for corn or soybean production, may be excellent for perennial forage species with high production potential (Figure 7).

Critical P levels for biomass production vary with the development stage of the plant. A newly germinated plant requires a relatively high P concentration in the soil solution, because it has a small root system which has not yet developed symbioses with mycorrhiza and other microorganisms. As the plant grows, the root system and symbiotic association develop, and its P requirement is lowered. Later on, at the fast growing stage, most vigorous species have high P requirements for rapid biomass accumulation. When soil fertility is below the optimum level at this stage growth is slowed. During the grazing state P requirements in terms of concentration in soil solution are relatively low because the plant has already accumulated nutrients in the biomass, symbioses are functioning and P recycling under a good management is efficient (Spain and Salinas, 1985).

When phosphorus concentrations in soil solution are optimum for maximum plant growth, there is a potential for leaching losses. This potential would be proportional to the concentration of P in the soil solution. The P concentrations required for adapted forage species are lower, assuring minimal P loss by leaching. Low P concentration requirements of adapted species are due to their efficient P recycling. The pasture is being consumed the whole year, and plant residues along with animal feces are returned to the soil continuously. Hence, when enough phosphorus has been accumulated in plant's biomass, in organic matter and in fresh residues, the role of the soil solution becomes small.

Maintenance requirements. It seems to be a paradoxical situation, but nutrient losses from pastures reach their lowest rate when productivity is highest. Several factors contribute to this effect: a) When the pasture is productive, a complete ground cover is maintained which assures high water infiltration rate, protection against erosion, little run-off. b) Vigorous plants have efficient root systems with efficient symbioses for fixing nitrogen and efficient absorption of P and other nutrients, along with a minimal loss by leaching, even of the more mobile nutrients (potassium, nitrates). c) The system losses reach the lowest rate when nutrients taken out in products represent the main component of loss, and losses by leaching, run-off and erosion are minimal. d) At highest production levels, higher concentrations in soil solution are required, and at these levels it is probable that leaching losses may be somewhat increased.

Less soluble and low cost sources of P. Edaphic characteristics of humid tropical environments are very favorable for the use of less soluble and low cost nutrient sources. These sources are ideal for providing the needs of the plant for many years. High acidity along with very low concentrations of P in soil solution assure an adequate solubility of rock phosphates. In addition, adapted forage species have lower rates of demand and more time for accumulating the required P than annual plants. However, the response to rock phosphate of B. decumbens under these conditions was not significant (Espinoza, 1987, Ferrufino, 1984) unless accompanied by N and K applications (Vallejos, 1986).

Role of vesicular-arbuscular mycorrhiza on growth and nutrient absorption. Vesicular-arbuscular (VA) mycorrhiza result from the colonization of fine, absorbent, plant roots by soil fungi that belong to the Endogonaceae family. Vesicular-arbuscular mycorrhiza generally increase plant growth by enhancing nutrient absorption. Fungal hyphae that extend into the soil serve as an extension of the root system and are more effective in absorbing nutrients than the roots by themselves. The typical symptom of mycorrhiza deficiency is a nutrient, especially P, deficiency in the host plant. Plant nutritional experiments using isotopes confirm that N, P, K, Ca, S, Zn, and Cu absorbed by mycorrhiza from the soil are transported to the host plant (Cooper and Tinker, 1978; Timmer and Leyden, 1978; Ames et al., 1983). Minerals at more than 4 cm from the root can be absorbed by hyphae and translocated to the root. Mycorrhizae are not only structurally efficient to extract nutrients from soil exchange sites, but can also produce exogenous enzymes such as phosphatases (Gianinnazi-Pearson and Gianinnazi, 1978; Ho and Trappe, 1975).

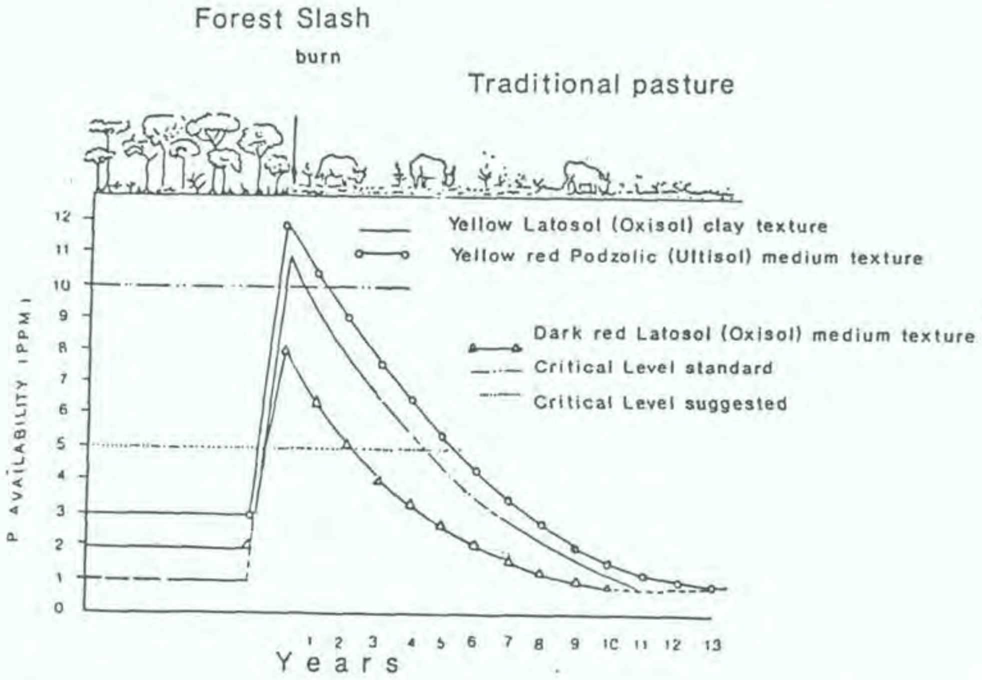

Figure 6. Model of the P dynamic in the soil-plant-animal system in an amazonic forest. (Serao et al., 1979).

Most tropical plants are mycorrhizal by nature. It seems to be a symbiosis fundamental for plant survival and persistence in the environment. However, it has not been possible so far to manipulate or manage VAM in a practical way, mainly because inoculants could not be produced in absence of the host in an artificial medium. But as VAM association is so important for tropical forage species, a better understanding of the effects of the different management aspects of soil-plant-animal system on its effectiveness is required (Salinas and Saif, 1989).

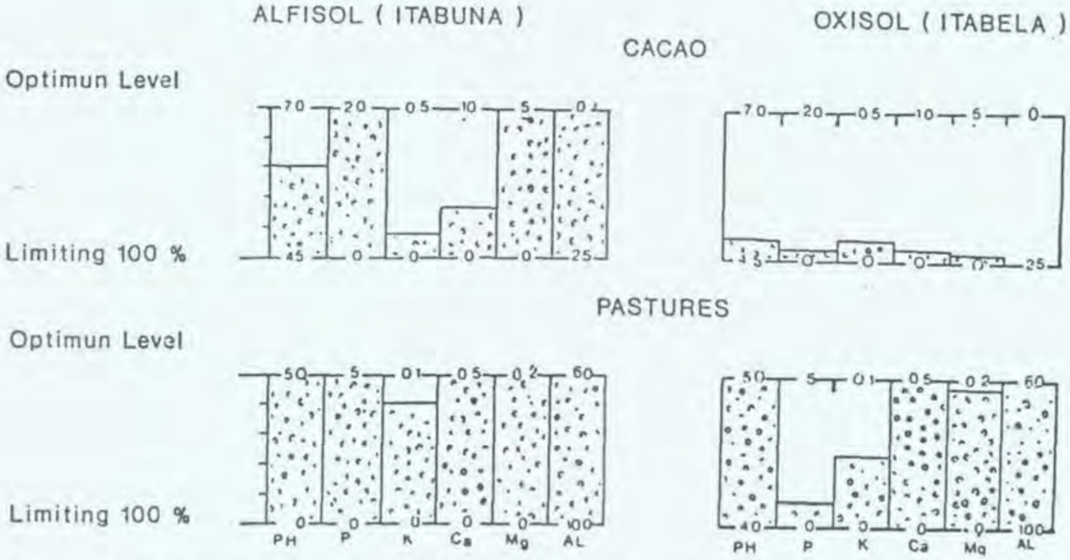

Figure 7. Relative soil fertility considering two contrasting crops (Cacao and pastures). (Spain and Salinas, 1985).

Pasture management. Management is the most important factor determining whether or not there will be adequate P recycling efficiency. Unfortunately, it is perhaps the weakest area in knowledge up to date, and consequently the one where mistakes are frequently made. Studies have begun on the interactions between stocking rates and grazing systems including different pasture species (Figure 8). Preliminary observations indicate that the conservation of balance between grass and legume and the persistence of the pasture for a long time can be managed. Pasture management systems and stocking rates seem to have an effect on the balance between species, but this interaction has not been well confirmed, and various field studies are in progress to come to a better understanding of this subject. Maintaining species balance is of primary importance, for productivity, stability and efficiency of pastures.

Residue management. During the rainy season, recycling of nutrients in a well kept pasture depends greatly on the litter at the soil surface, where humidity, temperature, aeration and nutrient concentration conditions are most favorable. To maintain this environment favorable, there has to be adequate plant cover and a constant return of both animal and plant residues. These undergo quick decomposition with an intense activity of micro flora and fauna. Earthworms, beetles, termites, ants and other meso fauna mix detritus and mineral soil. During the process they perform a very effective "tillage" of the first 5 to 10 cm of the soil profile. In this way, the soil is maintained porous and receptive to water infiltration, even under grazing with high production levels.

It is sometimes thought that animal trampling constitutes a waste of forage which should be reduced to a minimum. The hypothesis presented here says that one of the most important ways of nutrient recycling is the fall of green leaves and stems, as well as the plant detritus. The "waste" of trampelled plants and the animal residues are vital for the proper functioning of the system since they constitute the food for the micro and meso fauna encountered at the soil surface (Spain and Salinas, 1985).

Table 3. Annual production of plant residues and nutrient contributions in four tropical pastures.

Pasture association	Litter Production	Nutrients					
		N	P	K	Ca	Mg	S
		Kg ha^{-1} y^{-1}					
P. phaseoloides	3562	77.5	3.3	12.1	59.5	9.9	6.8
+ *A. gayanus*							
+ *B. decumbens*	7085	86.6	6.9	26.5	46.2	16.6	12.3
D. ovalifolium	7537	60.3	4.9	11.8	59.8	13.5	8.3
+ *A. gayanus*							
+ *B. humidicola*	7014	78.3	6.1	17.8	32.9	15.3	10.9

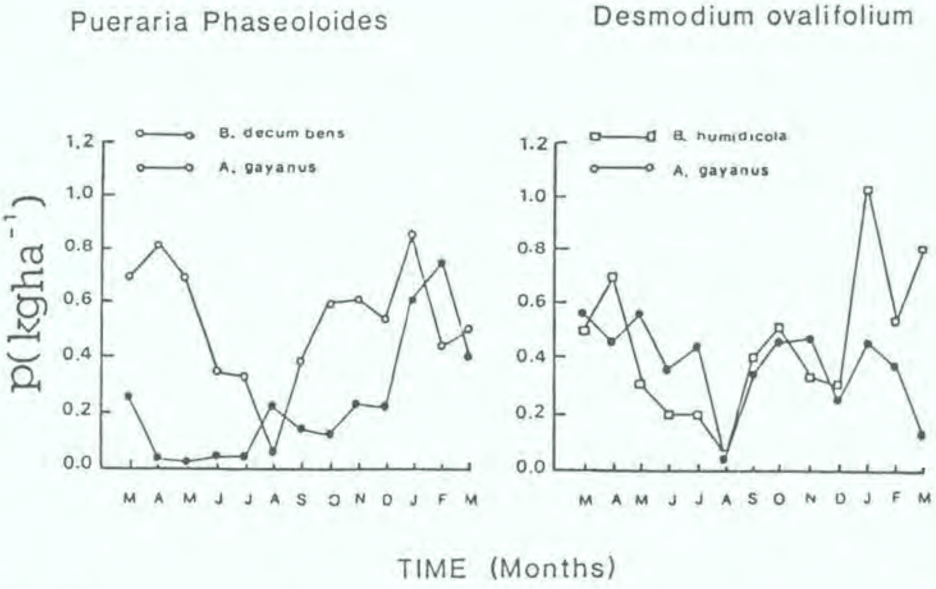

Figure 8. Differential contribution of P from the litter as a function of time in four tropical pastures under grazing (CIAT, 1985).

On soils of low fertility, with low cation exchange capacity and high P fixation capacity, the concentration of this recycling process at the surface could be essential for the preservation of nutrient availability. The process functions almost independently of mineral soil, with important advantages. P fixation by colloids would be minimal. Losses by leaching are small, even at high plant production levels, since nutrient concentrations in the soil solution continue to be low without damaging the plant which is taking most of the nutrients from the detritus in an almost completely closed recycling. In a rainy environment with highly porous soils, the potential for leaching is high, and the only way to counteract it, is by using production systems not requiring high nutrient concentrations in soil the solution. This situation is not much different from the one encountered at a humid forest floor where recycling is also concentrated on litter and on the first centimeters of soil.

In summary, the objectives of pasture management are to assure a constant covering of soil, to keep the balance between grass and legume and, to provide to the soil a constant return of residues (Table 3). This creates an optimal environment and nutritional condition for decomposers and soil fauna, which is essential for an adequate functioning of the system.

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PHOSPHORUS DYNAMICS IN A SUGAR CANE FIELD CROP

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Introduction

During several decades the Río Yaracuy Valley has been traditionally cultivated with sugarcane. Phosphorus appears not to be a limiting factor for sugarcane production in the area, and therefore fertilization with P is rather uncommon (Sequera et al. 1984). However, annual stem cropping constitutes a significant net P loss from the system. Persistence of adequate supply of P to the crop, in spite of large yearly P outputs in the crop suggests that internal processes such as decomposition-mineralization, play an important role in maintaining the soil P-status.

This paper is an attempt to quantify the distribution of P during the growing season of a sugar cane field plot, focusing on cycling processes in the soil. As a result, a balance model of net in-and outputs was developed to predict fertilization requirements in the study area.

Study area and procedures

The study area is located in the Rio Yaracuy Valley (Yaracuy State, in mid western Venezuela), at 70 m above sea level, with a mean annual rainfall of 1600 mm and a mean annual temperature of 26.5° C (Sequera, 1988). The soil under study is a Mollisol with a pH around 7, and with substantial amounts of calcium carbonate.

The area under cultivation is affected by acid rains. The pH of the rain waters varied between 3.5 and 4.5. Acid rains in alkaline or neutral soils can induce a significant solubilization of calcium phosphate (Cole and Stewart 1983; Cook 1983). Acid rains may increase the amount of P extracted by the crop and the leaching losses after heavy rains.

Dynamics of P was measured through two growing seasons in a 0.12 ha plot planted with sugar cane. Results reported here are for the variety Venezuela 58-4. Methodology of biomass collection, and for P measurement in plant, soil, and water samples has been described elsewhere (Sequera et al. 1984, Sequera 1988).

The approach consisted in measuring: a) input of P in rainfall (wet and dry deposition) and fertilization; b) output through leaching, harvesting and pre-harvest burning; c) estimation of P content in dead and live biomass, and in the soil at different times during a growing season.

Phosphorus accumulation and distribution in biomass during a growing season

The amount of P accumulated in biomass increased rapidly and almost linearly during the first 160 days (Figure 1). Afterwards it stabilized reaching a total accumulation of 48.8 kg ha^{-1} at harvest. From this amount 72.2% were found in stems, 10.5% in living leaves, 9.2% in dead aboveground material and 3% in roots. Almost all the P contained in the stems was harvested and removed from the system (38 kg ha^{-1} yr^{-1}).

Phosphorus accumulation and distribution in the soil

Total P in the soil (down to 30 cm depth) amounted to 2498 kg ha^{-1}, of which 76% were inorganic (Pi) and 24% organic (Po) fractions. Within the Pi fraction, P-Ca is the main reserve (López-Hernández et al. 1989b), whereas in the Po the moderately labile forms dominate, as measured with the Bowman and Cole (1978) procedure.

Phosphorus inputs and losses

Previous work in the area showed that the sugarcane crop did not respond to applications of P fertilizer (Sequera et al. 1984). It appears then that supply of P to the crop is covered by the available and potentially mineralizable pools in the soil. However, maintenance doses of P of 50 kg ha^{-1} of triple superphosphate (11 kg P ha^{-1}) were applied in the present experiment.

Table 1. P concentration (mg l^{-1}) and P inputs (kg ha^{-1}) in rainfall during the sugar cane growing cycle

Days of growth	Month	P conc. (± s.d.)		P inputs (range)	
		mg l^{-1}		Kg ha^{-1}	
30	April[†]	1.40	(±1.17)	2.41	(0.02 - 5.05)
62	May	0.69	(±1.36)	1.06	(0.00 - 4.19)
96	June	0.41	(±0.69)	0.14	(0.01 - 0.45)
127	July	0.10	(±0.07)	0.11	(0.10 - 0.20)
162	August	0.41	(±0.33)	0.94	(0.16 - 2.80)
180	September	0.82	(±0.83)	1.20	(0.00 - 2.62)
218	October	0.11	(±0.41)	0.25	(0.00 - 0.65)
248	November[††]	1.28	(±0.93)	2.76	(0.15 - 5.04)
270	December[††]	1.64	(±0.99)	5.84	(2.81 - 8.83)
308	January[††]	1.22[†††]		0.33	-
	Total			15.04	

[†] Experimental plot just burned.

[††] Burning operations in surrounding areas.

[†††] Single data.

An average input of 15 kg ha^{-1} yr^{-1} was estimated from monthly measurements of P concentrations and volumes of rainfall (Table 1). However, this figure is unreliable. Variability of P concentrations within collectors (n=5) and dates was high, indicating the heterogeneous influence of dry deposition. There is evidence that the higher concentrations of P in rainfall correspond to those months when sugarcane is burned in areas in the neighborhood of the experimental plots. Ash deposition could account for a large fraction of the P input measured in those months. Pulido (1987) for the same growing season showed that 42% of the ashes in a pre-harvest burn were lost from the site of burning.

This result shows that estimations of nutrient input in rainfall in agricultural areas submitted to seasonal burning are liable to large errors (Table 1). Perhaps the only alternative to obtain reasonably accurate estimates might be to considerably increase the number of rainfall collectors within the study area.

In any case, estimated P input in rainfall in the study area is unusually high, and much larger than figures reported from other lowland and montane areas in Venezuela (López-Hernández et al. 1986; Steinhardt and Fassbender 1979).

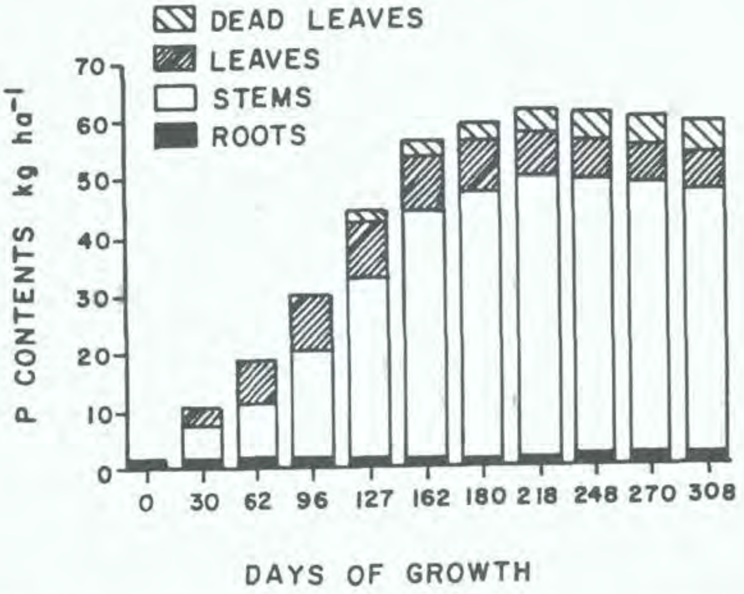

Figure 1. Total P contents and distribution in sugarcane during the cropping season.

Leaching losses are a function of the P concentration in the soil solution and the volume of water percolating beyond the root level. Tension free lysimeters were located at 30 cm depth, thus including the zone where 80% of root biomass of sugar cane occurs (Avilán et al. 1984), and at 60 cm depth. Water collected in these lysimeters was used to measure the P concentration of the soil solution. Concentration in the latter lysimeters was used to estimate the net P losses through leaching. The volume of percolated water was estimated through daily water balances (Sequera 1988). Leaching losses of soluble P reached 3 kg ha^{-1} yr^{-1} (Table 2), an amount considerably larger than what was found in natural ecosystems but not unusual in agroecosystems (Ryszkowski et al. 1989). Monthly leaching losses ranged from 0.01 to 0.23 kg ha^{-1} throughout the growing season, however, most of the P losses from the ecosystem occurred in an unusual wet month at 270 days of growth (Table 2).

Harvesting of cane stems removed a total 38 kg P ha^{-1}, more than 3.5 times the amount of P supplied as maintenance fertilization.

Another cause of P loss from the system is the pre-harvest burning of the dead aboveground plant material (Pulido 1987). Fire produces large convective air currents which transport ashes and burning materials away from the plantation site. This material could be redeposited in other cane fields or blown away to other systems. The pre-harvest burning eliminates all the dead plant material accumulated during the growing season. About 42% of the ashes produced are blown away, while the rest is incorporated into the soil at the same site. Burning results in a net loss of 2.3 kg ha^{-1} yr^{-1}, which eventually may be recovered through the transport of ashes originated in burning of neighboring cane fields (Pulido 1978). The ashes incorporated into the soil amount to 17.1 kg P ha^{-1}. This deposition could be detected through the measurement of available P (Olsen et al. 1954) in the soil after burning (Table 3).

Table 2. Soluble P concentration in lysimeter waters and soluble P losses by leaching during the sugar cane growing cycle

Day of growth	P concentration (30 cm depth)	P concentration (60 cm depth)	P losses (60 cm depth)
	mg l^{-1}	mg l^{-1}	kg ha^{-1}
30	0.24	1.36	0.23
62	0.13	0.16	0.11
96	0.21	0.09	0.01
162	0.43	0.31	0.33
180	0.05	0.16	0.01
218	0.07	0.16	0.13
248	1.06	0.28	0.20
270	0.53	0.76	2.07

Table 3. Available P in soil before and after burning (mean of 5 plots)

Time of sampling	(mean)	(sd)	(mean)
	mg kg^{-1}		Kg ha^{-1}
Before burning	10.17	3.34	33.56
After burning	17.16	3.69	56.22

Changes in the soil organic P fractions

Total Po measured by the Bowman and Cole (1978) method decreased until the pre-harvest burning (Table 4). Maximum Po values occur during the first 30 days of growth which is the period when the soil receives an important Pi input from fertilizer and ashes. Po increases after fertilization have been reported by López-Hernández et al. (1989b) for the same soil and by Doormar (1972) and Sharpley and Smith (1985) in other natural and cultivated ecosystems although at larger time scales. Sharpley and Smith (1985) attributed the Po decrease along the growing season to an active uptake after mineralization.

In the Mollisol studied neither the labile Po fraction nor the highly resistant Po forms could be detected (Table 4). The former is mainly associated with phospholipids and nucleic acids (Cole et al. 1977, Bowman and Cole 1978, Borie and Barea 1981) and is supposed to be readily mineralized (Harrison 1982). Lack of this fraction might be a consequence of active mineralization, although more information should be obtained. The dominant fraction is the moderately labile Po (82%-92% of total Po) (Table 4).

McGill and Cole (1981) have postulated a biochemical mineralization for Po which involve endogene phosphohydrolase. The enzyme is a "repressible enzyme" affected by Pi levels. In natural ecosystems Po mineralization is the main source of Pi for primary producers (López-Hernández et al. 1989a). Harrison (1982) indicates that a rate of Po mineralization of 4 mg P kg^{-1} $month^{-1}$ can supply 11 kg P ha^{-1} yr^{-1} which is enough to support a British Northern hardwood forest. Any Pi supply by mineralization in a sugarcane plantation under a low P input should be above the value presented by Harrison (1982). In fact, a preliminary result obtained with the isotopic dilution technique of López-Hernández et al. (1989a) gives a high P mineralization rate for the Mollisol studied (5 mg P ha^{-1} d^{-1}) which, in turn, is associated with low C/Po ratios at the different stages of plant growth (Table 5). The role of microbial biomass and microbial P in this agroecosystem might be essential. This is currently being studied.

Table 4. Organic P fractions (mg kg^{-1}) in soil during diverse stages of the sugarcane growing cycle.

Day of growth	Total Po	Labile Po	Moderately labile Po	Moderately resistant Po	Highly resistant Po
30	277	0	250	27	0
150	260	0	233	27	0
330bb†	163	0	136	27	0
332ab††	179	0	147	32	0

†bb before burning
††ab after burning

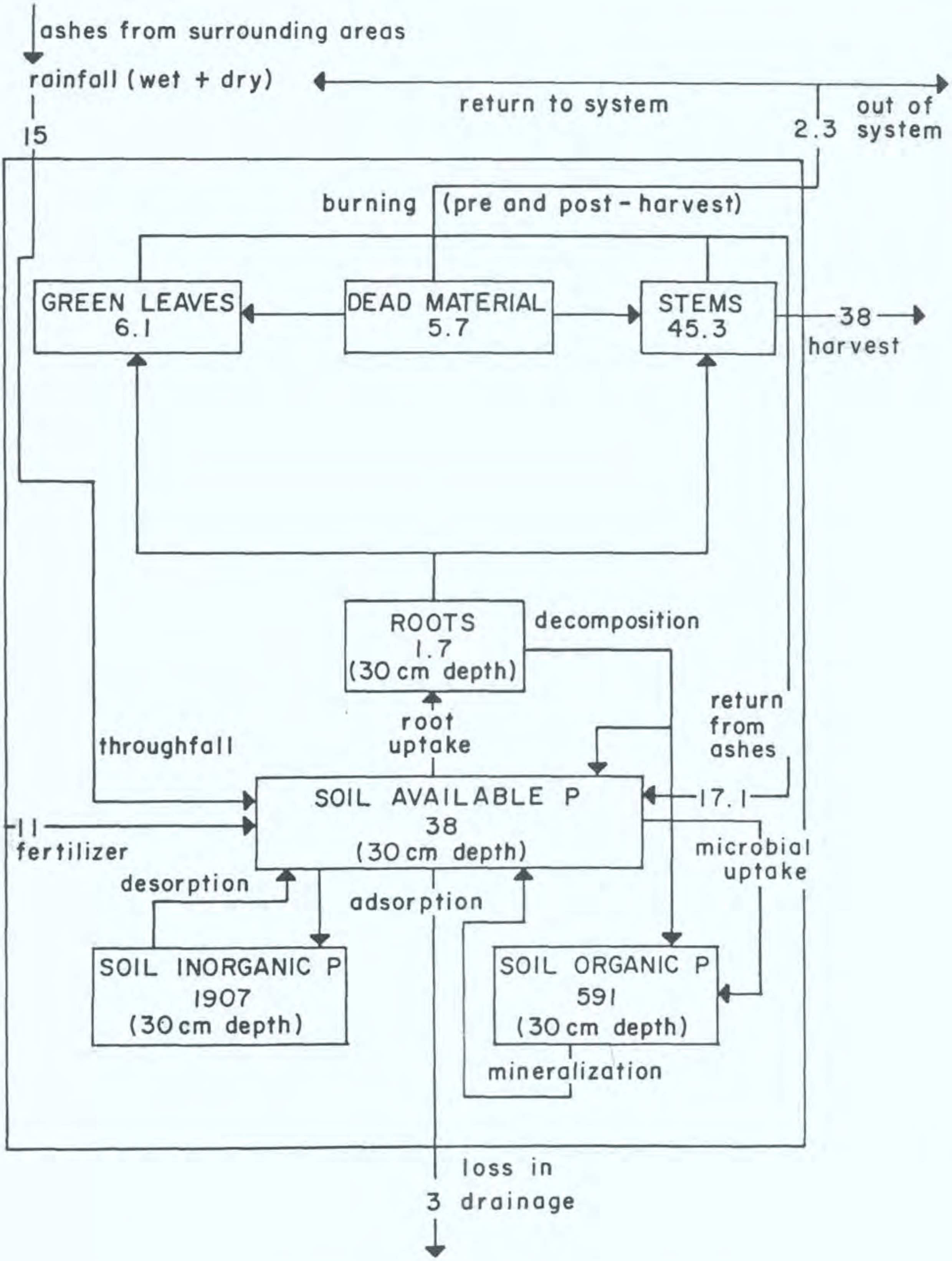

Figure 2. Phosphorus cycle in a sugarcane field. Contents in kg ha^{-1}, transfers in kg ha^{-1} yr^{-1}.

Table 5. C:Po ratio in soil during diverse stages of the sugar cane growing cycle

Day of growth	Po (mg kg^{-1})	C (mg g^{-1})	C:Po
30	277	17	39.35
150	260	11	59.23
330bb	163	15	82.21
332ab	179	20	112.29

bb before burning
ab after burning

The P cycle in the sugarcane agroecosystem

Figure 2 summarizes the information presented above as a general diagram for the phosphorus cycle in a sugar cane field. The general budget (P inputs - P outputs) gives a negative value of 17 kg ha^{-1} yr^{-1}, which suggest that fertilizer maintenance doses must be increased in order to avoid P depletion in the system. However, this net annual loss, is less than 3% of the mineralizable pool, which reflects the high content of P in the soil and may explain the lack of response of sugarcane to P fertilization in the area.

Conclusions

The P cycling in the sugarcane agroecosystem studied, is characterized by an active role of internal mechanisms which maintain the P supply demanded by the crop. Of notorious importance is the recirculation of ashes by increasing P availability in soil and P levels in precipitation. Changes in organic P fractions along the crop growing cycle and low C/Po ratios are indicators of a good potential for Po mineralization, although more research is needed to determine Po mineralization rates.

Acknowledgements

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PHOSPHORUS RESOURCES AND REQUIREMENTS IN LATIN AMERICA

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Present fertilizer situation

Latin America consumes less than half the N-P-K* fertilizer materials of North America. In 1987/88 consumption totalled 7,223,000 versus 16,635,000 metric tons (Mg), respectively. Of the three plant essential major nutrients (N, P and K), P consumption was the least (Bumb 1989).

Overall the balance of the three major nutrients consumed in Latin versus North America is very similar (Table 1). The N:P:K ratio for Latin America is 2.0:0.6:1.0. This compares to 2.7:0.5:1.0 for North America. The largest difference is with N, where more is used in North America relative to P and K. Four Latin countries deviate significantly from the average. These are Argentina, Chile, and Mexico which use considerably more N compared to P or K, and Brazil which uses less N relative to P or K. At least part of the explanation for these differences lies in the inherent soil fertility of the individual countries.

Table 1. Fertilizer consumption in Latin America and North America during 1987-88 (Bumb 1989)

	N	P	K	N : P : K
	Consumption, 1000 Mg			Ratio[1]
Latin America	3,997	1,252	1,974	2.0 : 0.6 : 1.0
North America	10,689	1,906	4,040	2.7 : 0.5 : 1.0

[1]Countries considerably different from average (1985-86):

Argentina,	9.5 : 2.2 : 1.0
Chile,	8.7 : 3.5 : 1.0
Mexico,	22.5 : 3.0 : 1.0
Brazil,	9.0 : 0.6 : 1.0

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* Data dealing with production and consumption of phosphate and potash fertilizers at the commercial level are normally reported on the oxide basis. However, the organizing committee of this conference requested all fertilizer data be printed as elemental N, P, and K. Conversion factors were: P205 x 0.437 = P and K20 x 0.830 = K.

Latin America consumes substantially less fertilizer per hectare of arable land and per capita than North America (Table 2). The discrepancy in fertilizer use between North and Latin America is even greater than these values suggest, since there are substantial arid and semi-arid land areas in North America with low annual production capacity, including some that can be cropped only on alternate years due to moisture limitations.

Table 2. Fertilizer consumption per hectare of arable land and per capita, 1985 (FAO, 1987)

	N	P	K	N + P + K
	kg ha^{-1}			kg/capita
Mexico	12.7	1.7	0.6	20.9
Central America				
Costa Rica	17.0	1.8	4.9	27.2
Guatemala	18.9	3.9	3.4	10.0
South America				
Venezuela	8.7	2.6	4.0	18.5
Colombia	5.2	1.1	2.2	10.5
Ecuador	5.9	1.3	0.9	6.4
Peru	1.6	0.2	0.3	3.3
Argentina	0.5	0.1	0.1	4.1
Chile	6.1	2.4	0.7	13.3
Brazil	3.5	2.4	3.7	19.1
North America				
Canada	15.8	4.0	4.2	73.5
U.S.	21.0	3.8	8.8	62.5

Canada has the highest P use per hectare across all the Americas, followed by Guatemala, USA and Venezuela. Highest per hectare consumption of fertilizer in Latin America is in Costa Rica, Guatemala and Venezuela, where crops with high nutrient requirements plus high cash value on the export market support greater fertilizer use. Also, in the case of Venezuela, fertilizer has received heavy government subsidies during most of the 1980's.

The lowest P use is in Argentina. Nitrogen and K consumption also rank lowest for Argentina. Although the soils of Argentinian Pampas, on which substantial areas of wheat, corn and soybeans are grown, are among the most fertile soils of Latin America, reports in recent years indicate that both N and P are, or are becoming more frequently deficient. During the 1990's, fertilizer use is expected to increase sharply.

The data per capita emphasize the dramatic differences between Latin America and North America. Consumption ranges from 3.3 to 27.2 kg/capita in Latin America compared to 62.5 and 73.5 for the U.S. and Canada, respectively. Assuming that the North American pattern is closer to ideal (production efficiency is among the best in the world and food costs among the cheapest), then Latin America would be expected to substantially increase fertilizer consumption as an essential component to increased productivity. Phosphorus consumption would increase at least as fast as N and K since tropical and subtropical soils frequently contain little plant available P.

History of P production and consumption

Both production and consumption of P fertilizers have grown dramatically over the past four decades in Latin America (Table 3). Production in 1960 was only 40,000 Mg of P, and increased to 845,000 Mg by 1987, a 21-fold increase. Consumption increased 9.3-fold from 132,000 Mg to 1,232,000 Mg,.

Table 3. Latin American production and consumption of P, 1960-87 (Bumb 1989).

	1960	1970	1980	1987
	1000 Mg of P			
		-------------- PRODUCTION	--------------	
Latin America	40	167	702	845
Central America	--	74	105	116
South America	40	93	598	729
		-------------- CONSUMPTION	-------------	
Latin America	132	372	1,084	1,232
Central America	20	136	177	264
South America	112	236	907	968

Production has been increasing in Latin America at a faster pace than consumption. Presently Latin America is approximately 68% self-sufficient compared to only 30% in 1960 (Table 4). The percentage figure is misleading, however, since the gap in actual tonnage between production and consumption has been increasing. Imports needed to fill this gap were 92,000 Mg in 1960 and 387,000 Mg in 1987.

Table 4. Latin American P production/consumption ratios, 1960-87 (Bumb 1989)

	Production/Consumption	% Sufficiency
1960	40,000/132,000	30
1970	167,000/372,000	45
1980	702,000/1,084,000	65
1987	845,000/1,232,000	68

Fertilizer projections

Fertilizer forecasting is about as reliable as any other type of long-range projections based on world economic and political ebbs and flows (or the weather). None-the-less, based on present cropping patterns in Latin America, potential for expansion of cropped areas, population requirements (including growth projections), individual government's attitude toward domestic agricultural development versus food imports, fertilizer consumption during the past few decades, etc., the following projections are offered (Table 5).

Table 5. Fertilizer consumption in Latin America: Present and projected (Armelin 1988)

	N	P	K	N : P : K
	Consumption, million Mg			Ratio
1987-88	4.0	1.3	2.0	2.0 : 0.6 : 1.0
2000	5.6	2.0	3.2	1.8 : 0.6 : 1.0

In the case of P, the projection is for an increase of about 700,000 Mg by the year 2000. This is about 54% in a little over 10 years. The ratio of N:P:K is expected to stay reasonably close to that of today, with considerably less N relative P and K than in North America (Table 1). The production deficit for P in Latin America is expected to continue into the next century. In 1987 the deficit was 387,000 Mg (Table 4) and is expected to be about 880,000 Mg in 2000 (Table 6).

Table 6. Projected fertilizer production deficit in Latin America in the year 2000 (Armelin 1988)

	Tons	% of consumption
Nitrogen	3,000,000	54
Phosphorus (P)	880,000	44
Potassium (K)	2,500,000	79

Phosphorus resources

A number of countries in Latin America have commercial P reserves or at least the potential for developing commercial reserves. Many areas have not been evaluated in detail and, therefore, the tonnage reported in this section for reserves (deposits profitably mineable with todays economics and technology) and resources (potentially profitable with a positive shift in economics or technology) are likely much lower than actuality.

The Brazilian deposits are the most developed at present. In fact, Brazil is essentially self-sufficient in P fertilizer production. Known deposits are widely scattered around the country and total 167,000,000 Mg of P. Of this, 66,000,000 Mg are considered commercial reserves. It is projected that these reserves will last Brazil 25-30 years at an annual growth rate in consumption of 4%.

Other countries with P resources are given in Table 7. This is a partial list derived from several sources (Casanova and Valderrama 1986, Elizalde 1989, Meninato 1988, Grossi 1988, Ricaldi and McClellan 1987). The authors of this paper had difficulty separating reserves from resources, and obtaining consistent values among references. This is simply a reflection of the present dynamic situation in Latin America with frequent updated information, the problem of converting geologist information into meaningful agronomic information, and perhaps subtleties missed in translating the Spanish literature. There are deposits only recently discovered, such as some in Ecuador in 1984, and not yet fully evaluated. More intensive and extensive studies will certainly lead to additional discoveries in numerous locations in Latin America.

Table 7. Countries with P deposits of commercial or potential commercial value (references listed above)

	Tons of reserves/resources
Colombia	164,000,000
Ecuador	205,000,000
Peru	260,000,000
Venezuela	460,000,000
Mexico	5,000,000,000

Summary

Latin America contains significant deposits of P suitable for commercial production. In fact, substantial amounts of P are being mined and converted into phosphate fertilizer material. However, consumption continues to outpace domestic production so that, with the notable exception of Brazil, Latin countries must import P to meet internal demand. Consumption of P per hectare and per capita is low compared to north America and other developed countries, reinforcing the present trends for increasing demand for P as well as N and K into future decades.

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