CHARACTERISATION AND TRANSFORMATION OF PHOSPHORUS IN FOREST SOILS

by

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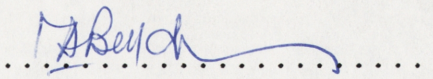
STATEMENT OF ORIGINALITY

I certify that, except for the assistance and sources acknowledged, the experimentation and analysis, interpretation and presentation of the results in this thesis are my own original work.



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ABSTRACT

Studies of phosphorus (P) in forest soils have been based on a procedure permitting fractions of soil P to be apportioned quantitatively along a gradient of lability.

The movement and transformation of P in a low sorbing soil suggested a long term residual effectiveness of fertiliser P. Where applied to soils of differing sorption capacity, P moved rapidly through a high P sorbing soil during initial leaching, but thereafter, the decline in the recovery of most-labile P was much faster than in a low sorbing soil.

The mineralisation of organic P in a sandy soil of a harvested pine forest was increased by cultivation. Removal of litter and slash resulted in a decline of labile organic P, and therefore less accumulation of mineralised inorganic P. Hence P nutrition problems might be expected when the next crop is established. Comparing different soils demonstrated that mineralisation was highest in soils with high organic P, but that the amount of P accumulated in labile forms depended on the P sorption capacity of the soil. There were no clear stoichiometric relationships between C, N and P during mineralisation. Plants in a bioassy showed that the estimated amount of mineralised P may not all be "available" for plant uptake. N uptake by plants was similar to mineral-N initially present in the soil and that resulting from mineralisation.

SUMMARY

Forest management practices are usually aimed at obtaining yield in perpetuity. It is desirable that the practices have minimal adverse effects on the biological, chemical and physical properties of forest soils. Selection of the suitable practices may be effected through quantification of nutrient and biological changes in forest soils associated with the various management practices. This study aimed to provide more definitive data regarding transformation of fertiliser and native P in contrasting forest soils under a number of different management practices.

A phosphorus (P) analytical procedure that permitted fractions of soil P to be apportioned quantitatively along a gradient of lability, was adopted. The procedure was used to assess under field conditions the movement and chemical transformations of P applied as superphosphate and sewage sludge. Experiments were conducted on a yellow podzolic soil supporting eucalypt and radiata pine stands. P applied as superphosphate was redistributed to more than 20 cm deep in the profile mainly by mass flow following the first rain . Subsequently, P was transformed into different fractions, most of the transformation occurring within one month of fertiliser application. After one month, the trend of decline in the recovery of labile and moderately resistant inorganic P fractions slowed. Twenty six months after fertiliser application, more than 35% of the added P was still in the labile inorganic pool, suggesting a long-term residual

effectiveness of fertiliser P in this soil.

Most of the P applied in wet sewage sludge was retained in lumps of the sludge that remained on the soil surface, and was therefore less effective in short-term improvement of soil P compared with superphosphate P. Application of sewage sludge in air dry and ground form increased its effectiveness as a source of P. About 8% of P applied in wet sewage sludge was recovered from soil as labile inorganic P after 14 months, compared with 46% when sludge was applied in air dry form. Most of this recovered P was in the surface 5 cm, possibly as a result of increased soil P sorption in the sewage sludge treatments.

The movement and chemical transformation of fertiliser P were also studied in soils of different P sorption capacities (yellow podzolic, red podzolic and yellow earth soils). Intact cores of these soils were incubated at 20°C and near field capacity. This study showed that the greatest movement of fertiliser P through the soil occurred with the first leaching water, and declined in the subsequent leaching episodes. Most P collected in the leachate was from soil with the highest P sorbing capacity, and this was attributed to preferential flow through soil macropores. However, the amount of P retained as resin extractable P (most-labile P) from added fertiliser was much less in soil with high P sorption capacity (7% in the yellow earth soil) compared with that retained in soil with low P sorption capacity (22% in the yellow podzolic soil). Also, the trend of decline in the recovery of the most-labile P was much faster in the high P sorbing soil than in the low P sorbing soil throughout the

incubation period.

A study on the transformation of native P under different treatments of litter and slash management after harvesting a pine forest was conducted with siliceous sand soils of South Australia. Where a cultivation treatment mixed litter P with the soil, mineralisation of soil P was increased. This resulted in an initially high rate of labile inorganic P accumulation in the soil. In contrast, in treatments where litter and litter+slash were retained and not disturbed, the rate of labile inorganic P accumulation was initially low, but subsequently increased after 8 months. When litter was removed by raking, labile organic P declined and this was accompanied by lower rates of inorganic P accumulation. The rates of accumulation of labile inorganic P were different in soils incubated under controlled conditions compared with soils sampled *in situ* in the field. However, excluding the control treatment, the lowest rate of labile inorganic P accumulation was 0.3 kg P ha^{-1} $month^{-1}$ in incubated soil from the raked treatment, while the highest rate was 1.2 kg P ha^{-1} $month^{-1}$ in soil from the slash treatment in the field. Both P values are quantitatively adequate for radiata pine seedling requirements during the first 3 years of growth.

The mineralisation of soil organic P was also studied in incubated yellow podzolic, red podzolic and krasnozem soils collected from radiata pine and eucalypt forests. The accumulation of labile P in these soils after 157 days of incubation was 2.7, 2.1 and 1.7 mg P kg^{-1} respectively. Corrected for sorption, these values give estimates of P mineralised of 4.3, 4.5 and 10.0 mg P kg^{-1} respectively,

which is consistent with the trend in soil organic P levels. Therefore, measurement of soil P sorption would be important in estimating levels of P cycling in the soil-plant systems.

Soil management practices also affected net mineralisation of soil P. Liming of the krasnozem soil increased accumulation of labile inorganic P by more than 3 times that in non-limed soil, while addition of a readily available C source to a fertilised yellow podzolic soil resulted in net immobilisation of P. Mineralisation could not be detected in soils with initially high inorganic P (fertiliser treatments) because continued chemisorption masked mineralisation effects.

Measurements of microbial biomass C, N, P, biomass respiration, and soil mineral N were also done in order to determine linkages between them and mineralisation of P. There was more biomass C in soils with high organic C, but the ratio of biomass C to soil organic C was not constant. This was also reflected in the soil respiration measurements. The ratio of C:N:P in the microbial biomass differed between soils. There was no clear relationship between mineralisation of C, N and P; and changes in the pools of C, N and P in the microbial biomass were also not related to the mineralisation of C, N, and P. These findings are consistent with the proposal by McGill and Cole (1981) that the relationship between C, N and P during cycling through organic matter is not stoichiometric.

The relationship between mineralisation of N and P was further tested using plants in a bioassy. The presence of growing plants resulted in the uptake of more P from the soil

than could be accounted for by changes in the labile fraction (P initially present and P resulting from mineralisation), whereas N taken up by plants was similar to the changes in soil mineral-N (N initially present plus N mineralised). The plants may have taken up some mineralised P prior to P fixation, and P from the non-labile P pool.

The results in the study are discussed in terms of optimal utilisation of forest floor and soil organic P, and efficient fertiliser programmes. P sorption partly determines the amount of P that can be retained in labile form after mineralisation. Management practices that can match mineralisation of P to its demand by trees are desirable to minimise losses through fixation and leaching. This, together with knowledge of soil P sorption could be useful in the selection of the timing, methods and rates of fertiliser P application to optimise returns from forest production.

CHAPTER 1 INTRODUCTION

CHAPTER ONE

INTRODUCTION

Phosphorus (P) is present in almost all components of the biosphere. According to Hutchinson (1970), P is an ecologically indispensable element because it functions as an agent of energy transfer, and a deficiency of available P is more likely to limit reproduction and hence productivity than any other material, except water.

In the soil-plant-animal system, commonly more than 90% of the P is in the soil (Ozanne, 1980), much of it in forms not readily available for plant use. This unavailabilty is because it is (a) physically buried within a solid matrix, (b) in organic forms which cannot be assimilated by the plant , or (c) is transferred very slowly from the soil solid phase to the plant root (Oglesby and Bouldin, 1983). The inability of plants to use most of the total P in soils has led to the widespread use of fertiliser P in agriculture and forestry.

Fertiliser P is derived mainly from mineral deposits which are a non renewable resource (Cathcart, 1980), and although world reserves of these are large (Cook, 1983), the cost of producing fertiliser P will continue to rise. Costs will increase because of the need to process P from lower grade phosphorus rock, and an increasing dependence on underground mines. Present rock phosphate that is high in phosphorus, low in undesirable elements, and from low cost open mines is being depleted rapidly.

Australian soils are relatively infertile and have particularly low P levels (Lamb, 1986), as a result of strong weathering and leaching over geologic time. (Williams and Raupach, 1983). The amount of available soil P, more than the amount of any other nutrient, governs the fertility status of Australian soils (Norrish and Rosser, 1983). The main artificial fertiliser used in Australia is superphosphate, and most of the rock phosphate used for its manufacture is imported. The cost of importing rock phosphate in 1981-1982 was A$ 109.5 million, and has been increasing (Australia Year Book, 1984; Cook, 1983). Utilisation of waste materials, such as sewage sludge, can reduce these costs.

Traditionally, it has been considered that trees grew without a need for fertilisation, with continued production largely dependent upon the recycling of nutrients. This partly explains why only about 0.5% of the fertiliser P used in Australia in 1980 was in forestry (Pulsford, 1981; Attiwill, 1983). Forests exhibit a number of nutrient recycling and conserving mechanisms (Attiwill, 1980; Florence, 1981; Miller,1981). To minimise investment in terms of nutrient inputs, foresters require much better understanding of these mechanisms. The effectiveness of native P and of P from fertiliser and other sources in the soil-tree P cycle may be enhanced by a better understanding of the transformations of P in forest soils.

Plantation forestry in Australia has been based on the replacement of the native vegetation with *P. radiata* and subsequent rotations of the same species. Experiments (Hamilton, 1965) have shown that the replacement of the native

eucalypt forest by exotic pines may only slightly affect fertility of the better forest soils during the first rotation. On poorer soils, organic matter in the upper soil horizons decreased, and changes in cation exchange capacity, nitrogen and total P occured, but these differences can mostly be accounted for by nutrient uptake (Feller, 1983; Hopmans *et al.*, 1979; Turner and Kelly, 1985). Since a high proportion of annual nutrient uptake in forests is returned to the soil as litter (e.g. Pritchett, 1979), significant nutrient losses at harvest resulting from certain management practices, such as hot slash burns or windrowing, could reduce nutrient supply to subsequent tree crops. Practices that induce changes in soil microbial populations may also change the cycle of nutrients. Recent advances in methods to measure P held in the microbial biomass (Brookes *et al.*, 1982; McLaughlin *et al.*, 1986) have created new opportunities to study the transformation of P in forest soils, where decomposition processes are critical to nutrient cycling during long (40-100 yr) forest rotations.

P deficiency problems are usually most severe in 6-15 year old trees (Gentle and Humphreys, 1967) which is the time of greatest nutrient demand (Turner and Lambert, 1986). Deficiency problems have been reported on soils derived from deposited sands (Boardman, 1972), sandstones (Appelton and Snow, 1966; Ballard, 1970) and volcanics (Turner and Lambert, 1985). Significant growth responses of trees to P application have been noted on many Australian soils (e.g. Gentle *et al.*, 1965, 1986; Turner, 1982; Woods, 1982; Cromer *et al.*, 1985) with potential for substantial residual effects into

subsequent rotations (Turner and Lambert, 1986).

In view of the importance of P as a growth limiting element in Australian forests, and of the soil or fertiliser as the main source of P for the trees, a more basic understanding of P in the soil is needed. There has been a large amount of empirical fertiliser research in Australian forestry, and fertiliser regimes determined for different soil types. However, there has been only a small amount of research into understanding the nature and distribution of P in forest soils, the way added fertiliser reacts with these soils, and the time period during which a response may be obtained. A more basic approach to the study of P in soils and reactions of fertilisers may lead to fertiliser regimes which are effective and efficient in the use of a finite resource.

It has been the objective of this project to develop a better understanding of the distribution, forms, movement and transformation of P in Australian forest soils, and the availabilty of this element to the plant.

A review of P in forest soils is given in Chapter 2. It is concerned largely with the forms of P in soils, its reactions with, and transformation in soil, and the responses of soil P to forestry management practices. Experimental materials and methods are described in Chapter 3. Five soils have been used, four from the Australian Capital Territory (yellow podzolic, red podzolic, yellow earth and krasnozem) and one from South Australia (a siliceous sand). Methods of extracting and determining different forms of P are described, together with some experiments to validate the methods.

Chapter 4 examines the distribution and forms of native P in a yellow podzolic soil supporting a pine stand and a native sclerophyll forest. Inorganic fertiliser and sewage sludge were applied to the soils *in situ*, and the movement of P, and the forms in which it occurs, monitored for 26 months. The same movement and transformation of applied P is further described in Chapter 5 in soils (yellow podzolic, red podzolic and yellow earth) known to differ in inherent P fertility and adsorption characteristics. This study was done under controlled conditions using largely undisturbed soil columns.

The effect of some forestry practices is examined in Chapter 6 in a study comparing the effect of soil P of different forest floor (slash and litter) management systems after clearfelling pine forest on dune sands in South Australia. This study aimed to improve our understanding of P dynamics during the marked site disturbance associated with clearfelling and re-establishment.

The investigations in Chapters 4 to 6 present only static pictures of P in soil at particular times, usually several months apart. The very small amounts of 'available'-P usually present in soil focussed attention on the need to explore the process by which P becomes 'available' through decomposition of soil organic matter. Thus the study in Chapter 7 sought to determine temporal linkages between soil microbial activity and the mineralisation of N and P derived from soil organic matter, and that in Chapter 8 to assess plant uptake of N and P derived from mineralisation during the early phase of plant growth.

The final Chapter (Chapter 9) draws together the work

undertaken in this project and presents a perspective of P in forest soils and its management.

CHAPTER 2 PHOSPHORUS IN SOIL: A LITERATURE REVIEW

CHAPTER TWO

PHOSPHORUS IN SOILS: A LITERATURE REVIEW

2.1 Introduction

The estimated reservoir of phosphorus (P) in the earth is of the order of $8.4*10^{17}$ kg (Richey, 1983) and most of this is in marine sediments. Erosion of terrestial P continues to enrich the marine P sediments. In living organisms, P is important because of its genetic role in ribonucleic acid and its function in energy transfer via adenosine triphosphate.

On the earth's surface, P principally occurs as orthophosphate (Stewart and McKercher 1982). Native P in soils is derived mainly from apatite in parent materials during weathering. The liberated P continually undergoes transformations between organic and inorganic forms as well as between plant available and unavailable forms. Differences between soils depend on the rocks from which the soils were derived, the climatic regime and the vegetation present. Soils from acid igneous rocks are generally low in total P while those derived from basic rocks contain moderate to high amounts (Stevenson, 1986).

On the global scale, variation in total soil P is a result of losses during soil formation by leaching, erosion and redistribution by wind, water, animals and plants (Larsen, 1967). Plant cycling, manuring and fertilisation have concentrated total P in the A horizon of soil profiles.

2.2 Forest soils

In their description of the distribution, morphology and genesis of Australian forest soils, Thompson and Beckman (1981) observed that podzolic soils were the most extensive with leached acid soils and shallow stony soils also occupying sizeable areas. They also noted that exotic pine plantations have been established on a wide range of soils. Generally these soils are old and highly leached so that pine plantations are grown on soils that are usually deficient in N and P (Waring, 1981).

A forest soil is a soil that has developed under the influence of a forest cover. According to Pritchett (1979) this definition recognises the unique effects of deep rooting by trees, specific organisms associated with the forest vegetation and the litter layer, and leaching promoted by the products of its decomposition. In natural forest ecosystems, the soils are normally stable and support some degree of forest productivity without artificial nutrient inputs. Man's disturbance of forest ecosystems by intensive silviculture can affect soil properties and long term productivity. These changes need to be well researched (Ballard, 1980; Pritchett, 1979).

Most forest soils are moderately to extremely acid as a result of the release of organic acids during decomposition of the litter layer and the subsequent leaching of bases from the surface soil (Pritchett, 1979). Soils supporting conifers tend to be more acid than those supporting hardwood

species because conifer leaves have a lower base content. The effects of soil acidity on the growth of trees result from indirect effects such as soil microbial activity and nutrient availability. Fungi are more tolerant of acid soils than bacteria (Alexander,1961; Haynes, 1984) and are therefore predominant in forest soils (Pritchett, 1979). Acid forest soils have pH-dependent charges and low base saturation (Pritchett, 1979). Soluble iron, aluminium and manganese are usually found in acid mineral soils and react with P rendering it insoluble (Talibudeen, 1981) and unavailable to most plants.

2.3 Coping with low P in soils

Plant species have evolved on nutrient poor soils to cope with low levels of soil fertility. The variety of ways in which they have done so have been discussed by Bowen (1981) and include:

1. improved methods of nutrient absorption, such as through mycorrhizal symbioses and formation of proteoid roots,
2. formation of fine, much-branched root systems,
3. efficient nutrient assimilation, assisted by low growth rates and evergreen perennial habits,
4. increased efficiency of nutrient use, i.e. increased organic matter production per unit of nutrient absorbed from the soil,
5. nutrient re-translocation prior to leaf senesence, and
6. storage of appreciable quantities of nutrients when not required for growth.

Eucalypts are the dominant tree species of Australia. They can form forests of commercial sawlog quality on soils in which most of the P is in Fe and Al complexes or resistant organic matter (Gentle and Humphreys, 1968; Kessel and Stoate, 1938). There is evidence of high efficiency of P uptake, circulation and conservation by eucalypts growing on such soils. The lignotuber of the eucalypt seedling is regarded as one adaptive mechanism enabling the seedling survive harsh environmental conditions (Florence, 1981). The ectomycorrhizae associated with the eucalypt roots probably play an important role in P uptake. Efficient withdrawal of P from leaves before abscision and efficient recycling within the tree has been used to partly explain high productive growth of *E. obliqua* forest with only 0.09 kg P per tonne of dry matter compared to 0.29 kg P per tonne for forests in North America, Africa, and New Zealand (Attiwill, 1972; 1975).

Conifers also possess special characteristics for efficient nutrient use. All conifers are ectomycorrhizal (Ballard, 1981) with the host plant benefitting from improved P uptake. Mycorrhiza inoculation of nursery soils for exotic pines has generally been practiced when introducing pines to the tropics and to areas with an agricultural land use history (Pritchett, 1979). The translocation of nutrients from older to younger tissues for re-use in the biosynthetic process is also important. Switzer and Nelson (1972) and Turner and Singer (1976) estimated that internal nutrient cycling in different pine forests accounted for about 60% of the P requirements for new tissue. This contrasts with 90-95% of P withdrawn from eucalypt during heartwood formation (Beadle and

White, 1968; Hingston *et al.*, 1979; Attiwill, 1980). Eucalypt and pine thus possess similar mechanisms of coping with low P but differ in degree of efficiency. Despite these mechanisms, experimental work reviewed by Boardman (1974) and Waring (1981) indicate a tree response to P fertiliser on soils that do not contain adequate soil nutrient reserves. In order to devise a nutritional strategy for Australian forest soils (Waring, 1981), it is necessary to understand the nature and distribution of native and fertiliser P in soils.

2.4 Consequences for forest P fertilisation

2.4.1 Forest fertilisation practice

Fertilisers are used in forestry either at the time of planting because of limited availability of soil nutrients, or at a later age, to overcome a more basic nutrient deficiency and increase production (Neilsen, 1982). Tree responses to fertiliser can be expected, and have been reported (Waring, 1969, 1972; Simpson and Lewty, 1982; Woods, 1982; Cromer *et al.*, 1985), during the years prior to canopy closure when growth is totaly dependent on soil nutrient reserves (Miller, 1981). Optimum response to fertilisers at this stage often requires weed control as well (Waring, 1972). Once the canopy has closed, the demands made on the soil nutrient capital will normally be reduced, and nutrient response may only occur after thinning, when the crown biomass is being re-established (Crane, 1981). Studies of nutrient cycling in *Eucalyptus obliqua* L'Herit (Attiwill, 1979) suggest stages of growth similar to those of pine.

Water soluble superphosphate and nitrogen-phosphorus mixtures are the most commonly used fertilisers in forestry (Pulsford, 1981). Slow release forms have limited demand because they are bulky (Rajan, 1980) and difficult to apply evenly (Mannion, 1977). They are less effective on forest soils of moderate acidity and P fixation capacity (Bengston, 1976; Mead, 1974). However, rock phosphate has been observed to be valuable on podzolised sands in which P can be leached quickly, and on soils of high fixation capacity (Bengston, 1976; Humphreys and Pritchett, 1971).

Experiments with waste materials overseas, including sludges, effluents, garbage and ash (Lutrick *et al.*, 1982; Brown and Fiskell, 1983; Dunavin and Lutrick, 1983; Neary and Comerford, 1983) on forested lands have produced positive results with respect to tree growth. Sewage sludges contain organic matter, substantial amounts of N and P and a number of other elements which at low concentrations are essential for plant growth (Coker, 1983). When compared to monocalcium phosphate in two different soils, McLaughlin and Champion (1987) observed that the anaerobically digested sludge exhibited characteristics of a slow release P fertiliser. It was 90-100% as effective as monocalcium phosphate in supplying P for plant growth. Thus the application of sewage sludge to forest soils may be beneficial to forest P nutrition, and an economical disposal method (Soon and Bates, 1982).

Experiments have shown that only 5-10% of applied P fertiliser is taken up by an annual crop (Cooke, 1981). Forest utilisation values of 1-4% (Ballard, 1978a), 15% (Will,

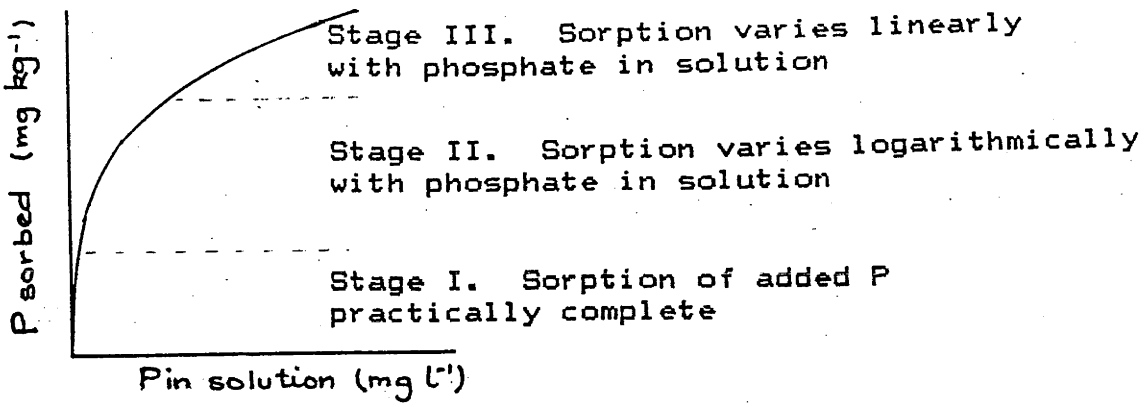
1965) and 30% (Pritchett and Smith, 1974) have been recorded after different periods of application and for different stages of growth. Because of low recovery values, residual fertiliser effects have been found to influence growth over an entire rotation (Pritchett, 1979) and have even carried over to the second rotation (Ballard, 1978). On some soils, however, residual effects are almost absent or levels of available P have declined significantly (Waring, 1981). Several factors are involved in the decrease in effectiveness of P fertiliser.

2.4.2 Reactions of P with the soil

When P is added to soil, its concentration decreases rapidly initially, followed by a slow decline (Munns and Fox, 1976). The amount of P retained by the soil therefore increases with time. With an increase in solution P concentration, adsorption increases following a curve that approximates the Langmuir plot at equilibrium (Mengel, 1985). The other commonly used plot to describe soil P sorption is the Freundlich isotherm (Larsen, 1967). Probert (1983) and Ratkowsky (1986) have examined and compared the above plots and their modifications, as well as other forms of equations used for describing P sorption by soils.

Barrow (1973,1974) found that during long term reaction, the retained P becomes less readily replaceable by other specifically adsorbing ions like arsenate, molybdate, and hydroxide. This is associated with a decrease in P availability (Larsen, 1967, Larsen and Widdowson, 1971)

* A plot of P sorbed against equilibrium solution P shows the three stages in the reaction.



Stage I involves high energy chemisorption of small amounts of P. Stage II involves the precipitation of a separate phosphate phase. Both the results for regions I and II are due to the exchange of the phosphate ion for the OH^- counter-ion of a positively charged edge Al atom formed by the hydrolysis of an OH group. The reaction consists of two steps:

$$\text{(step 1)}\quad \rangle AlOH + H^+ + OH^- \overset{K_1^I}{\rightleftharpoons} \rangle Al^+ \begin{matrix} \swarrow (H_2O) \\ \cdots OH^- \end{matrix}$$

$$\text{(step 2)}\quad \rangle Al^+ \begin{matrix} \swarrow H_2O \\ \cdots OH^- \end{matrix} + H_2PO_4^- \overset{K_2^I}{\rightleftharpoons} \rangle Al^+ \begin{matrix} \swarrow (H_2O) \\ \cdots (H_2PO_4^-) \end{matrix} + OH^-$$

where K_1^I is the equilibrium constant for the hydrolysis of the OH, K_2^I is the equilibrium for the exchange reaction, the symbol ↙ denotes a coordinate link and ⋰ an electrovalent link.

(from Muljadi *et al.*, 1966a).

Although the reactions are similar, the equilibrium constants for Stages I and II are different.

Stage III represents low energy sorption of phosphate, where sorption arises through penetration into some less crystalline region of the clay surface.

through precipitation or adsorption reactions (Sample *et al.*, 1980).

Two main methods have been used to identify the soil components that retain P (Norrish and Rosser, 1983). The first method involves measuring the capacity of the soil to retain P and correlating this with other soil properties. The second method involves a direct reaction of P solutions with pure minerals. Very few studies are made at normal soil solution concentrations (<0.001 ppm) or at high solution concentrations as would result from fertiliser application. Nevertheless, the studies have indicated that the following soil constituents are important in rendering soil phosphorus unavailable to plants.

(a) Iron and aluminium hydrous oxides

In their study of P adsorption by amorphous aluminium hydroxide, Hsu and Rennie (1962) found an initial rapid surface adsorption phase, followed by precipitation. Bache (1964), working with gibbsite, observed that surface adsorption occurred at low solution concentrations of phosphorus and that precipitation commenced at higher solution concentrations. Muljadi *et al.* (1966a,b,c,) also studied adsorption of P by kaolinite, gibbsite, and pseudoboehemite, and found three stage isotherms similar to those of Bache (1964)*. They suggested that the exchange of phosphate for surface OH groups was associated with a positively charged Al atom (region one) for the first OH and in "region two" for the second OH. Adsorption in "region three" was thought to be the result of P penetrating into some less crystalline region of

the clay surface, rather than being a result of precipitation. Kafikafi *et al.* (1967), Kyle *et al.* (1975) and Parfitt *et al.* (1977) suggested that the P in high energy adsorption was held as a bridging complex attached to two Al atoms to form a stable six membered ring.

Evidence also shows that iron oxides contribute to the P sorption capacity of soil. Parfitt *et al.* (1975) used infra red techniques to show that P was specifically adsorbed on Fe oxide surfaces by replacing two adjacent hydroxyl ions. Mattingly (1975) showed that the high energy adsorption capacity was correlated with dithionite extractable iron, while Bigham *et al.* (1978) observed that goethite adsorbed more P than hematite, but only because the goethite in their soils was very fine grained. Iron oxides have been observed to be important for P sorption in only some soils (Lopez-Hernandes and Burnham, 1974) and that it played second place in controlling sorption if exchangeable Al was present (Coleman *et al.*, 1969).

(b) Clay minerals

In his review of investigations into the reactions of phosphate with alumino-silicate clay minerals, Wild (1950) suggested that the mechanism for sorption was the same for the various clays- a ligand exchange reaction with the OH(H) groups coordinated to the exposed Al atoms on the edge face of the clay crystal (Beek and van Riemsdijk, 1978). This is analogous to the adsorption mechanism with metal oxides (Hingston *et al.*, 1972). Other possible mechanisms by which P reacts with clays are the replacement of silica from the

silicate framework (Rajan, 1975; Rajan and Perrott, 1975), and the formation of an Al-phosphate phase (Chen *et al.*, 1973). Both mechanisms occur in high solution phosphate concetrations.

(c) Soil carbonates

Phosphate adsorption on calcium carbonate or limestone has been measured by Cole *et al.* (1953), Amer and Ramsey (1971), Kuo and Lotse (1972), Griffin and Jurinak (1973) and Holford and Mattingly (1975). The reactive sites for adsorption of P by calcite are assumed to be the exposed surface Ca^{2+} ions in vacant coordinate positions which may be occupied by water molecules, bicarbonate ions or hydroxyl ions (Beek and van Riemsdijk, 1978). Phosphate ions may replace these molecules or ions. Holford and Mattingly (1975) concluded that below 0.5 mg P l^{-1}, the initial reaction between P and limestone is chemisorption. At higher concentrations, this is followed by physical adsorption of very low bonding energy, and by precipitation of octa-calcium phosphate in supersaturated solutions.

Field experiments have shown that liming increases P adsorption (Friesen *et al.*, 1980; Haynes, 1982) in moist soils. When limed soil was air dried before reaction with P, liming decreased adsorption. Haynes (1982) speculated that drying causes crystalisation of hydroxy-Al polycations as gibbsite, which would result in the charge characteristics of the soil being restored to a state similar to those of the unlimed soil. Sims and Ellis (1983) found that P availability of an acid Al-rich ultisol was best increased with low rates

of lime that increased soil pH slightly, but reduced exchangeable Al considerably.

(c) Soil organic matter

Soil organic matter has net negative charge and is therefore not thought to retain much P by itself, except when in association with cations like Fe^{3+}, Al^{3+} and Ca^{2+} (Wild, 1950). These ions are capable of adsorbing P when associated with organic matter. This association may explain the reported positive relationships between organic matter content of soils and P adsorption (Rennie and McKercher, 1959; Harter, 1969; Lopez-Hernandez and Burnham, 1974; Holford and Mattingly, 1975).

Appelt *et al.* (1975) prepared a hydroxy-Al-humic complex that was capable of adsorbing P. They reasoned that humic acid could react with Al from soil minerals to form complexes that would give rise to new surfaces for P adsorption. P was adsorbed by ligand exchange of phosphate for hydroxyl groups.

Weir and Soper (1963) also demonstrated the formation of a complex of Fe, P and humic acid in a neutral soil. They found that the addition of manure reduced the bonding energy of the soil, but increased the adsorption maximum. The above evidence suggests that the effect of an increase in organic matter content of the soil, such as through litterfall, would be to increase P adsorption rather than to decrease it by competing with P for adsorption sites.

In a study of 42 forest soils, Ballard and Fiskell (1974) and Ballard and Pritchett (1974) found that Al and Fe provided the best indices of P retention, consistent with P adsorption

in acid soils (Smith, 1965). For maximum utilisation of applied P, it would be convenient to characterise short term chemical forms into which it is converted, in order to be able to predict its effectiveness and residual effects for the entire rotation.

2.4.3 Immobilisation of P by the microbial biomass

Soil microorganisms affect available P by releasing available inorganic P by decomposition of organic P compounds, by promoting dissolution of insoluble inorganic forms and by immobilising available P into cellular material (Stevenson, 1986). During growth, microorganisms obtain inorganic nutrients from the soil in forms available to plants and from the decomposition of organic material. Net immobilisation of P occurs when the organic C:P ratio is 300 or more (Dalal, 1977; Stevenson, 1986). This is usually in the early stages of the decay process upon the addition of plant residues to the soil. This stimulates growth of microorganisms with the incorporation of P in microbial cell tissue. The P so immobilised would eventually be released upon death of the microbial cells, releasing the P in both inorganic and organic forms (Chauhan *et al.*, 1981). The life span of the microorganisms is relatively short, so most of the cellular P is only temporarily unavailable to plants.

There is evidence suggesting that soil organic P is predominantly microbial in origin (Cosgrove, 1967, 1977; Dalal, 1977; Anderson, 1980). On microbial death, a portion of the organic P is mineralised to inorganic P, but some of

the organic materials are stabilised with metal salts or organic polymers (Martin, 1964) or are resistant to decomposition (Anderson, 1980; Kapoor and Haider, 1982; Martin, 1964). This suggests that plant materials provide a source of inorganic P (and energy and other nutrients) for microbial synthesis of organic P compounds. Therefore conditions affecting the composition and activity of microorganisms would dictate the nature and amount of organic P compounds in soil.

2.4.4 Losses through leaching and erosion

It is generally agreed that in most soils P moves very little from the point of its application (Forsee and Neller, 1945; Russell, 1973; Schuman *et al.*, 1973; Baker *et al.*, 1975). The magnitude of movement depends on the fraction of soil P that is involved and the rate of movement of that fraction.

Appreciable movement of P from the top-soil down the profile has been reported in coarse-textured soils which have little capacity to retain water and have low buffering capacity for P (Neller *et al.*, 1951; Ozanne *et al.*, 1961; Mattingly, 1970; Humphreys and Pritchett, 1971; Paverill *et al.*, 1971), in organic soils (Larsen *et al.*, 1958; Miller, 1979) and in seasonally water logged soils (McGregor, 1953; Broomfield, 1955).

Other factors connected with P movement in soil include the rate of P addition (Logan and McLean, 1973; Miller, 1979) and water quantity and flux (Logan and McLean,

1973; Spencer and Wlasow, 1960; Bar-Yosef and Shekhosalmi, 1976) which encourage P movement at high rates. Initial soil wetness has been observed to affect P displacement by affecting the time needed to leach the soil (Sharma *et al.*, 1985). Preferential flow of water through biogenic channels (Scotter and Kanchansut, 1981) and cracks (Omoti and Wild, 1979) have been used to explain deeper penetration of P down the soil profile.

Losses of P through leaching, physical erosion and in surface runoff are low in forested areas (Megahan, 1972; Singer and Rust, 1975) although they can be increased during forest operations that expose surface soils.

2.4.5 Removal of P in forest products

The amount of P removed from a site during harvest is dependent on (i) the choice of tree species, since some species accumulate more nutrient than others (Attiwill, 1980; Crane and Raison, 1980); (ii) the length of crop rotation, as shortening of rotations has been shown to increase the quantity of nutrients removed (Kimmins, 1977; Crane and Raison, 1980; Crane *et al.*, 1981); and (iii) the completeness of harvest, as harvesting of bark, branches or leaves in addition to wood increases the quantity of nutrients removed (Kimmins, 1977; Cromer *et al.*, 1980).

The most common harvesting technique involves extraction of the merchantable bole and removes less than 1 kg P ha^{-1} yr^{-1} from coniferous forest sites (Ballard, 1980). Depending on the age of the trees, this may represent less than 20% of

the above-ground biomass (Ballard, 1980; Stewart *et al.*, 1981), because the greatest proportion of P is in the foliage. This emphasises the need for modifications to site management practices aimed at minimising losses and detrimental redistributions of slash and litter, especially after harvest.

2.5 Site disturbance and P economy

As discussed in section 2.4.3, harvesting results in nutrient losses arising from biomass removal. It also increases mineralisation through increases in solar radiation and moisture previously intercepted by the canopy. There may also be greater leaching losses as active roots are decreased (Ballard, 1980; Squire and Flinn, 1981).

Residue management during site preparation often includes cultivation, burning and windrowing. Cultivation increases the mineralisation of organic P, and leaching on sandy soils could occur. P losses during burning may occur in the form of particulates in convection columns (Harwood and Jackson,1975) and through movement of ash and surface soil by wind and as ash leacheate (Squire and Flinn, 1981). Increased availabilty of P after burning (Boyle, 1973) is a result of increased mineralisation of P from the organic layers.

Windrowing is used to remove debris from the site which interferes with cultivation or planting. This also removes top soil in addition to organic debris (Ballard, 1978a). Maceration of residues to reduce high volumes of logging residue to surface mulch has been used in South Australia (Squire and Flinn, 1981), but this is a costly practice.

In summary, site disturbance may result in losses of P reserves. However, some of the practices may at the same time improve P status of young planted stock (Haines, *et al.*, 1975). Such practices rely on P mineralising at the time of high plant P uptake, and in this way losses through leaching and sorption are minimised.

2.6 Mineralisation of organic P

Before becoming available for tree uptake, P in litterfall, other plant debris and in soil organic matter must first be mineralised, largely biologically. Will (1967) and Wells and Jorgensen (1975) have reported the release of 60-80% of the P from litter during the first year after deposition, after which the litter becomes part of the soil organic matter. Mineralisation of P is thought to be a result of the action of extracellular phosphatases. These are adaptive enzymes produced in response to the need for P by soil microorganisms and plant roots. Thus their activity is usually highest when the soil solution P concentration is low (Speir and Ross, 1978). Mineralisation may also be due to the process of utilising P containing organic substances by soil organisms for energy, analogous to N mineralisation (Thompson *et al.*, 1954; Cole and Heil, 1981). The rate of mineralisation rather than the amount of organic P in the soil is the main factor that determines its availability to plants (Tate, 1985).

Using a ^{32}P method to compare rates of labile organic P mineralisation in woodland soils, Harrison (1982a,b) could

account for over 90% of the variation in mineralisation rates by environmental and physical properties. These properties include temperature, moisture, aeration and soil reaction. These factors are interactive between themselves and with many other factors that control organic matter levels in soils.

a. Temperature

Temperature has a marked effect on microbial activity and thus mineralisation. Seasonal fluctuations in labile P levels in temperate grassland soils (Saunders and Metson, 1971; Helm *et al.*, 1972) and in temperate cultivated soils (Dormar, 1972) were attributed to the influence of temperature. In woodland soils, Harrison (1982) attributed higher rates of mineralisation in spring compared with autumn to higher phosphatase activities resulting from increased microorganism activity, or to increasing dominance of microorganisms actively secreting phosphatases. Temperature differences also account for the lower organic P in tropical soils than in temperate soils. Thus the mineralisation of organic P may contribute significantly to P nutrition of plants in the tropics, whereas in the temperate soils the contribution may be much smaller (Williams, 1967).

b. Moisture

Adequate moisture is necessary for mineralisation of organic matter and decomposition of plant residues, though these processes can occur at a range of moisture contents. Floate (1970) showed similar rates of mineralisation of sheep faeces and plant residues over a wide range of moisture

contents. Similarly, Jenkinson and Powlson (1981) found decomposition of organic residues at near maximum rates over a wide range of water contents. Dessication of soil causes the release of inorganic and organic P from dead microbial cells (Brookes *et al.*, 1982) and plant tissues (Newberg, 1979), as well as leakage of organic P from still viable microbial cells (Burns and Beever, 1978) and autolysed plant material (Jones and Bromfield, 1982). Flooding has been observed to cause more organic P mineralisation than holding soil at field capacity (Campbell and Racz, 1975). Wetting and drying cycles enhance organic P mineralisation (Birch and Friend, 1961).

c. Aeration

The effects of aeration on the transformation of organic P in soil are very complex (Tate, 1985) and difficult to separate from the effects of water (Dalal, 1977). Anaerobic conditions have been observed to encourage organic P mineralisation (Campbell and Racz, 1975; Ahmed, 1976; Racz, 1979) but usually it is slowed under such conditions (Dick and Tabatabai, 1978). It is possible that the increased solubility of Fe-P in anaerobic conditions inhibits phosphatase production and activity (Tate, 1985). More research to fully understand the effects of aeration on reactions of P in soil is essential.

d. Soil reaction

Generally, soil organic P is less stable in neutral or alkaline soils than in acid soils (Enwezor, 1967) because the diversity and activities of soil organisms decline with

increasing soil acidity. Van Diest and Black (1959) observed that variations in the levels of soil organic P in calcareous soils were correlated with the P contents of plants growing in them, whereas no such correlation was observed for acid soils. Increasing pH by liming usually, but not invariably, increases organic P mineralisation by encouraging microbial activity (Dalal, 1977; Tate, 1985) and solubility of the native organic P (Racz, 1979). Harrison (1982b) demonstrated a positive correlation between rates of mineralisation of labile organic P over the pH 3.1-7.5 range. Soil reaction, however, also affects reactions controlling P availability in soils (Sanchez and Uehara, 1980; Haynes, 1982).

e. Nature of organic substrates

C:P ratios in forest litters and organic matter affect nutrient levels and microbial activity in soil. When fresh organic matter is added to soil, the growth of saprophytic bacteria, fungi and actinomycetes is stimulated, and they attack the organic constituents of the added material (Stevenson, 1974), incorporating part of the C into the biomass and releasing a part as CO_2. The system develops toward a C:N:P:S ratio roughly equivalent to that in microbial cell tissue (100:10:1:1)(Alexander, 1961; de Haan and Zwerman 1978). The implication is that materials with C:N:P ratios of less than 100:10:1 will lead to mineralisation of organic N and P, whereas at higher values soil N and P will become immobilised within microbial cell tissue.

It is clear that microorganisms play an important role in the release of nutrients from organic matter. The amount and

stage of decomposition of organic residues, the solubilities of Al-, Fe- and Ca-phosphates and phosphate complexes with hydrous oxides and clay minerals play an important part in the ability of soil to provide P to plants. Knowledge of these forms of P in soils has come mainly from chemical extractions.

2.7 Characterisation of soil phosphorus

The characterisation of soil P has been done in many ways and for many reasons. Khanna (1981) has listed the aims of analysis of forest soils as (a) understanding soil processes in relation to vegetation, (b) prediction of growth responses, and (c) preparation of soil and land use maps. Olsen and Khasawneh (1980) classified P tests in two ways: those aimed at fertiliser recommendations, and which utilise correlations between soil test values and crop responses to fertilisers in field experiments; and those aimed at understanding the principles that govern the soil-phosphate-plant interactions. The latter require understanding of the chemical nature of soil P compounds, their physical and chemical interactions and transformations, and factors that affect the rate and direction of such transformations.

Kamprath and Watson (1980) have given the basic reactions by which P is removed from the soil solid phase as being anion replacement, and hydrolysis of cations binding P. In the anion replacement reactions, phosphate adsorbed on the surfaces of $CaCO_3$ and hydrated oxides of Al and Fe is replaced by other anions such as lactate, acetate, sulphate and bicarbonate. For example, the Chang and Jackson (1957)

fractionation procedure uses fluoride ions to complex Al ions, thus releasing P from Al-P. Citrate and lactate also complex Al ions, while extracting solutions containing the OH ion extract P from Al-P and Fe-P due to hydrolysis. Acids, generally in the pH range of 2 to 3, may also be used in the extraction of soil by dissolving Ca phosphates and some of the Al and Fe phosphates.

The extractants also remove organic P compounds of varying susceptibility to mineralisation. The sequential extraction procedure of organic P of Bowman and Cole (1978) uses 0.5M $NaHCO_3$, 1.0M H_2SO_4 and 0.5M NaOH to extract organic P pools ranging from labile to resistant forms.

Soil microbial biomass P used to be estimated from microbial biomass measurements (Van Veen and Paul, 1979). However, new methods have been developed based on chloroform fumigation (Brookes *et al.*, 1982; Hedley and Stewart, 1982) or hexanol fumigation (McLaughlin *et al.*, 1986). In these methods the P released from microbial cells following lysis with the fumigant is immediately extracted from the soil with $NaHCO_3$.

The ability of selective chemical reagents to dissolve discrete types of inorganic and organic P compounds has led to the development of various fractionation procedures to study the forms of P in soils. Since its inception, the Chang and Jackson (1957) P fractionation procedure has been modified (for example by Petersen and Corey, 1966; Williams *et al.*, 1971; Syers *et al.*, 1972) and widely used (Sharpley and Smith, 1985). A more elaborate fractionation scheme (Chauhan *et al.*, 1979, 1981; Hedley *et al.*, 1982) has recently been used to

separate microbial, inorganic and organic P into fractions that vary in their availability to plants. The scheme is summarised in Table 2.1.

Table 2.1 The sequential fractionation procedure for apportioning P into fractions that vary in availability to plants.
(From Hedley *et al.*, 1982a,b)

Extraction	Fraction and description of P measured
1. With anion (HCO_3) exchange resin	Resin-P. Most biologically available inorganic P
2. With 0.5M $NaHCO_3$, pH 8.5	$NaHCO_3$-P. Labile inorganic and organic P sorbed on soil minerals
3. Treatment with chloroform (24 h) followed by extraction with $NaHCO_3$, pH 8.5	Microbial biomass P. This is the difference between 2 and 3 resulting from lysis of microbial cells
4. With 0.1M NaOH	NaOH-P. Organic and inorganic P held more strongly by chemisorption to Fe and Al surfaces.
5. Sonification followed by extraction with 0.1M NaOH	Sonicate/NaOH-P. Inorganic and organic P of the internal surfaces of soil aggregates
6. With 1.0M HCl	HCl-P. Apatite-type minerals and occluded P.
7. Oxidation and acid digestion (H_2O_2, H_2SO_4)	Residual-P. Insoluble inorganic and stable organic P compounds

16 h extracts were used. Later methods (Section 3.3.4) prefer 30 min extraction particularly in biomass P measurements.

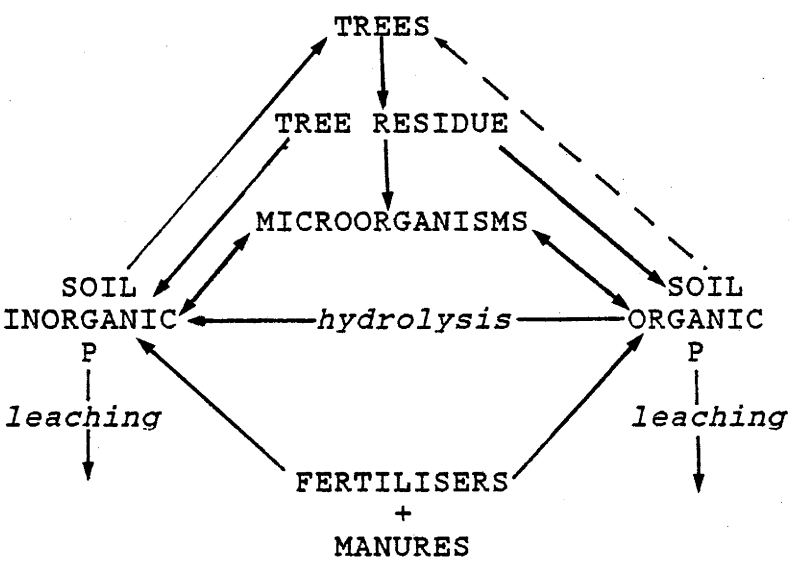
The method was applied to cultivated soils (Hedley *et al.*, 1982) and showed that cultivated soils were poorer in plant available P than pasture soils. The scheme appears to be also suitable for measuring short term changes in soil P

(Chauhan *et al.*, 1979), and to offer the advantage of apportioning P into forms of relative importance with respect to rates of mobilisation. Mobilisable forms of P that can be related to other forest soil parameters, such as mineral nitrogen, would reduce the extensive field and laboratory work commonly encountered in forest nutrition studies (Ulrich and Khanna, 1968; Khanna, 1981a).

2.8 General conclusions

P in the forest ecosystem involves its storage in living organisms, dead organic matter and inorganic forms. Trees take up P largely from the soil solution, which in turn is replenished by dissolution of inorganic forms, mineralisation of organic forms or by fertilising and manuring. The complex processes are summarised in Figure 2 in a simplified way.

Figure 2 P cycle in a forest soil.
Adapted from Anderson (1980).



Chemical extractants have been used to partition P in soil into pools based on their availability to plants. The importance of relationships between these pools and other soil parameters was emphasised in section 2.7. An appreciation of these relationships would be particularly useful in studying the mineralisation of P where mineralisation rates are low (Harrison, 1987), and the released P is subject to fixation.

P fixation reactions are also important from the standpoint of the efficiency of fertiliser use by plants. In acidic forest soils, P is likely to be precipitated as highly insoluble Fe or Al phosphates. Knowledge of the amount of P fixed and the rate of fixation in various soils and under different forest environments is useful in predicting the residual effects of the fertiliser and consequent application practices. Liming has been used in an attempt to correct soil acidity and improve availability of P (Pritchett, 1979). But liming has also exhibited contrasting results on microbial activity in forest soils (Bath *et al*, 1980; Lohm *et al*, 1984; Lang and Beese, 1985), which is important for the mineralisation of organic P.

Forestry practices are aimed at harvesting the forest at such a rate that the yield can be maintained in perpetuity. Nevertheless, these practices inevitably disturb the ecological equilibrium, affecting the soil physical, chemical and biological properties (Lamb, 1986). The extent of change might be minimised through selection of management regimes which encourage effective nutrient cycles. Quantification of nutrient changes associated with various management practices, including post-harvest practices, is one way of ascertaining

the suitability of management practices.

In view of the importance of P cycling to Australian forestry, more definitive studies of the movement and transformation of fertilisers, and the mineralisation P in forest soils were considered necessary, and are described in the following chapters.

CHAPTER 3 MATERIALS AND METHODS

3.1 Introduction

3.2 Sites and soils

3.3 Extraction and Determination of Soil P

- 3.3.1 Introduction
- 3.3.2 Extraction of phosphorus with anion (HCO_3) exchange resin
  - 3.3.2.1 Effects of time of extraction on resin P
  - 3.3.2.2 The effect of air drying on P extracted by the resin from soils
  - 3.3.2.3 Soil pH changes during P extraction with the resin
- 3.3.3 The extraction of P with 0.5*M* $NaHCO_3$
- 3.3.4 Extraction of microbial biomass P
  - 3.3.4.1 The effect of extraction time on the release of soil and microbial biomass P
  - 3.3.4.2 Correction for P sorbed during extraction for microbial biomass P
- 3.3.5 Extraction of P with NaOH
- 3.3.6 Kjeldahl digestion of phosphorus
- 3.3.7 Determination of soil P sorption capacity

3.4 Other Soil Chemical Analyses

- 3.4.1 Determination of soil pH
- 3.4.2 Extraction and determination of soil N
- 3.4.3 Determination of soil biomass carbon
- 3.4.4 Determination of O_2 uptake and CO_2 evolution by soils during incubation

3.5 Statistical Analysis and Presentation of Data

* The inclusion in this Chhapter of some tests to validate some of the experimental methods was criticised by one Examiner who believed they should have been the subject of a separate Chapter or a separate Part of the Chapter. This relates to Sections 3.3.1, 3.3.2, 3.3.3 and 3.3.4.

CHAPTER THREE

MATERIALS AND METHODS

3.1 Introduction

Studies of P in forest soils have been carried out in the field (movement and transformation of added P and under different forest floor management regimes); and under controlled environmental conditions (movement of P in soils and the mineralisation of organic P). This Chapter introduces the soils which have been used in these studies, and the analytical procedures. It also describes a number of tests carried out to validate some of these procedures.* More specific descriptions of soils, treatments and methods of sampling, preparing and storing soils are described in the individual chapters.

3.2 Sites and Soils

The experiments have been done with soils from the Australian Capital Territory and from South Australia. All experiments but one were conducted on soils from the Cotter Catchment Area of the Australian Capital Territory (ACT). The status of soil P under a number of forest floor (slash & litter) management regimes was examined on a site located within Softwood Holdings Ltd Caroline Pine Plantations (Block T101), Mt Gambier, South Australia. This site will be described at the beginning of Chapter 6.

The geology, geochemistry and soils of the ACT have been described in detail by Owen and Wyborn (1979) and Talsma (1983). The study area lies between 35°15' - 35°30'S and 148°40' - 149°E (Figure 3.1). The land surface is mainly over 500 m above MSL, with the uplands well over 1000 m. The present relief is primarily a result of post-Cretaceous erosion on fault blocks that have moved differentially in the Tertiary and Quaternary. The area encompasses a large range of altitude which influences both climate and native vegetation. Annual precipitation ranges from 1100 mm at Bulls Head to 820 mm at Uriarra (Talsma, 1983), and winter frosts are more common in high elevation areas. Highland areas are vegetated mainly by woodland *Eucalyptus pauciflora* (snow gum), while extensive lowland areas are covered by dry sclerophyll eucalypt forests. About 30% of the study area has been converted to pine plantations. There are 4 main soils in the area, with significant differences in profile morphology, chemical and physical properties (Talsma, 1983). The soils are the yellow podzolics, yellow earths, red earths and the krasnozems.

The Cotter Area experimental sites and soil sampling areas are shown in Figure 3.1, numbered 1 through 5, and briefly described below. Soil names are according to classifications by Stace *et al.*, (1968) and Northcote, (1979).

Sites 1 and 2

Site 1 is located within the Commonwealth Scientific and Industrial Research Organisation's (Division of Forest Research) Biology of Forest Growth study area (BFG) at Pierces

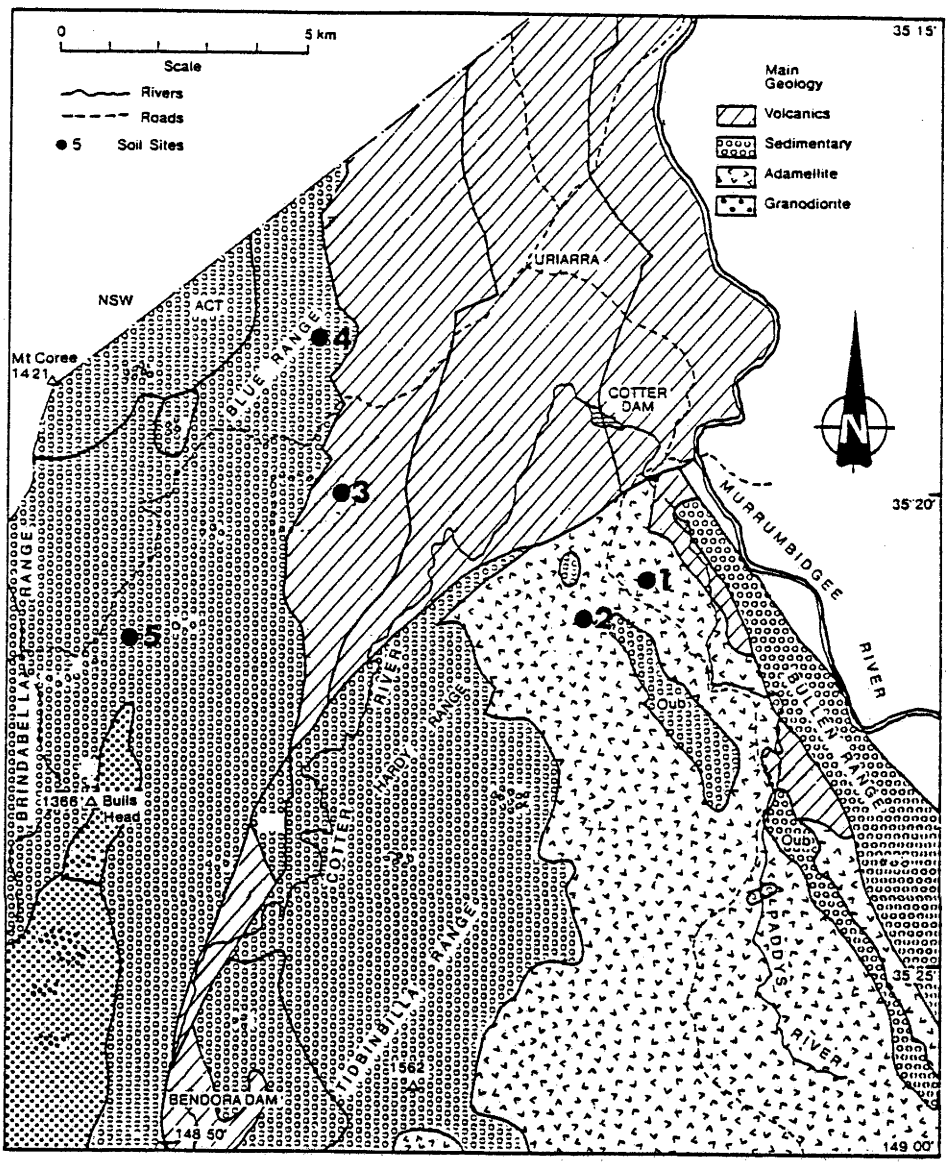


Figure 3.1 Location map of the Cotter Area, A.C.T. (Adapted from Talsma, 1983.)

Experimental and soil sampling sites:

1. Biology of Forest Growth (Pierces Creek)
2. Laurel Camp (Pierces Creek)
3. Greens Catchment
4. Blue Range
5. Picadilly Catchment

Creek. Site 2 is located on a similar yellow podzolic soil about 2 km away.

Soil: Yellow Podzolic (YP), Dy 3.61 (FAO, Orthic Luvisol).

Location: 31° 21'S; 148° 55'E. Alt., 594 m

Parent material: Adamallite (Coarse grained calcium rich granitic rock).

Vegetation: Site 1 supported a second rotation pine, planted in 1973. Site 2 contained natural dry sclerophyll eucalypt forest, first rotation pine (planted 1936) and second rotation pine (planted 1979) growing adjacent to one another.

Experiments: Field and soil column characterisation of P applied as fertiliser and sewage sludge (Ch. 4 & 5), and P transformation (Ch. 7 & 8).

Site 3. (Greens Catchment).

Soil: Red Podzolic (RP) soil, Dr 2.21 (FAO, Chromic Luvisol).

Location: 35° 20'S; 148° 52'E. Alt., 794 m

Parent material: Volcanic with metasediment overlay

Vegetation: First rotation pine.

Experiments: Movement of P in soil columns (Ch. 5), and P transformation (Ch. 7 & 8).

Site 4. (Blue Range)

Soil: Yellow Earth (YE), Gn 2.21 (FAO, Xanthic Ferralsol).

Location: 35° 18'S; 148° 52'E. Alt., 950 m

Parent material: Ordivician sediments from middle silurian volcanic rocks

* Where reference is made to formal soil types in this Thesis (eg Orthic Luvisol, Yellow Podzolic), upper case initials are used. Where referred to in a general descriptive sence, lower case initials are used (eg red or yellow podzolics).

** Additional climatic, soil profile and plant species information are given for the Cotter Catchment Area and the Caroline Forest Site in Appendix 4. However, some physical properties, including soil porosity, field permeability, hydraulic conductivity and water holding capacity were not analysed - although these could have contributed to the interpretation of the research data.

Vegetation: Second rotation radiata pine, planted 1959

Experiments: Movement of P in soil columns (Ch. 5).

Site 5 (Picadilly Catchment)

Soil: Krasnozem (K), Gn 4.11 (FAO, Humic Ferralsol).

Location: 35° 21'S; 148° 50'E. Alt., 1020 m

Vegetation: Wet sclerophyll (*E. delegatensis*) forest

Experiments: P transformation (Ch. 7 & 8).

Caroline Forest Site (Mt Gambier, South Australia)

Soil: Siliceous sand, Uc 4.21 (FAO, Cambic Arenosol).

Location: 35° 50'S; 140° 45'E

Parent material: Aeolian

Vegetation: Second rotation radiata pine, planted 1947

Experiment: Litter/slash management on
P transformation (Ch. 6)

*

A summary of the physical and chemical characteristics of these soils is given in Table 3.1.** There is a considerable range in both physical and chemical properties. For physical properties, the siliceous sand is composed mainly of coarse sand (>90%) and therefore has high bulk density. In the remainder of the soils, the yellow podzolic has the highest coarse sand fraction (54.2%) and the lowest clay content (8.9%) in the surface 0-10 cm. The krasnozem is the reverse, with 17% coarse sand and 28% clay. The krasnozem's low bulk density (0.8 - 1.1 g cm^{-3}) compared with that of other soils (e.g. 1.4 - 1.8 g cm^{-3} for the yellow podzolic soil) is associated with its high organic C content ranging from 10.1

Table 3.1 Some physical and chemical characteristics of the experimental soils.

Compiled from data of author, Talsma (1981, 1983), and CSIRO Plantation Forest Research Centre, Mt Gambier (Unpublished data).

Soil Type	Site	Depth (cm)	Stone > 2mm %	Physical Properties Fraction < 2mm					Chemical Properties									
									Exchangeable Cations me/100g									
				BD g cm^{-3}	CS %	FS %	SI %	Clay %	K	Ca	Mg	Mn	Al	CEC	%C	%N	pH Water	P mg/kg
Yellow Podzolic (YP) Dy 3.61	1 and 2	0-10	19	1.4	54.2	22.9	14.0	8.9	0.26	3.89	0.54	0.09	0	4.81	1.74	0.09	6.2	143
		10-30	26	1.5	52.2	23.1	15.8	8.9	0.13	0.83	0.31	0.01	0.19	1.54	0.32	ND	5.5	73
		30-50	31	1.8	50.5	13.5	20.8	15.2	0.21	1.50	0.54	0	0.91	3.31	0.14	ND	5.3	73
Red Podzolic (RP) Dr 2.21	3	0-10	17	1.1	20.3	23.1	35.5	21.1	0.48	2.33	0.94	0.23	1.88	6.03	1.98	0.11	5.2	232
		10-20	7	1.4	19.1	19.3	36.8	24.8	0.33	0.51	0.70	0.07	2.69	4.49	0.78	ND	5.2	ND
		20-35	7	1.5	16.8	14.6	35.3	33.3	0.65	0.48	1.15	0.02	2.87	5.35	0.58	ND	5.2	ND
Yellow Earth (YE) Gn 2.21	4	0-8	17	1.2	19.0	38.5	24.3	18.2	0.24	0.19	0.19	ND	7.98	6.40	2.69	0.2	4.7	164
		8-20	9	1.3	15.5	33.9	25.3	25.3	0.13	0.03	0.29	ND	4.56	5.06	1.34	0.1	5.3	99
		20-38	35	1.4	15.2	31.3	23.8	29.7	0.10	0.04	0.26	ND	3.84	4.31	1.02	0.1	5.4	108
Krasnozem (K) Gn 4.11	5	0-5	34	0.8	17.0	30.0	25.0	28.0	0.83	1.74	0.90	0.13	7.62	12.16	10.10	0.3	4.8	368
		5-15	14	0.9	13.0	24.0	28.0	34.0	0.56	0.52	0.41	0.04	6.48	8.18	5.80	0.9	ND	ND
		15-25	10	1.1	12.0	25.0	27.0	36.0	0.42	0.18	0.24	0.02	5.35	6.34	4.40	0.1	ND	ND
Siliceous Sand (Mt Gambier) Uc 4.21		0-15		1.3	> 90				0.27	1.13	0.86	0.07	0.21	2.82	0.80	0.04	6.1	60
		15-30		1.4					0.21	0.29	0.52	0.01	0.42	1.52	0.50	0.02	5.5	46

to 4.4% compared with 1.74 to 0.14% for the yellow podzolic soil. The red podzolic and yellow earth soils are intermediate in most physical properties.

On chemical properties, the siliceous sand has the lowest CEC, carbon, nitrogen and phosphorus levels, of the order of 50% or less when compared with other soils. The krasnozem has the highest level of exchangeable K, and the yellow and red podzolic soils have the highest levels of exchangeable Ca. There are considerable differences between exchangeable Al levels, ranging from 7.62 and 7.98 me 100 g^{-1} in the upper (0-5 cm) horizon of the krasnozem and yellow earth to, 0 and 0.21 me 100 g^{-1} in the yellow podzolic soil and siliceous sand respectively. The exchangeable Al levels reflect the differences in soil pH, being lowest (4.8) in the krasnozem and highest (6.2) in the yellow podzolic soil. Total P ranges from 60 mg kg^{-1} in the siliceous sand (0-15 cm) to 368 mg kg^{-1} in the krasnozem soil

3.3 Extraction and Determination of Soil P

3.3.1 Introduction

The range of methods which have been used in the extraction of soil P (Section 2.7; Sibbesen, 1983) give some indication of the difficulties in using a single extraction method for the universal study of soil phosphorus. Despite this, Donnelly (1970) was able to correlate various P tests by regression, and concluded that most P tests would be improved in their use for estimating "plant available P" if allowance

is made for soil constituents (e.g. exchangeable Mg, Ca and oxalate extractable Fe and Al) that affect the relationships of the different soil P tests.

Soil phosphorus terminology. Because of the many extractants and extraction techniques used, and the different combinations and concentrations of extractants, fractions of soil P have tended to be identified by the chemical extractant used, or the author of the method. The different extractants have also been used to selectively extract P in a way which reflects, and may be identified as, particular chemical forms in soil. Generally, alkaline and ammonium based solutions preferentially extract Al- and Fe-P, and acid solutions preferentially extract Ca-P (Kamprath and Watson, 1980).

As the extraction methods are empirical, there has been a need to define soil P in terms of the ability of a soil to release P to a crop. In this context, the concept of labile P was defined operationally by relating the results from chemical extraction to those obtained by isotopic exchange (McAuliffe *et al.*, 1947; Olsen *et al.*, 1954). Association of labile P with plant P availability has led to identification of labile P with the amount of P removed from soil by a variety of extractants. More recently, resin and bicarbonate extractable P have been defined as labile forms of P (Hedley and Stewart, 1982; Hedley *et al.*, 1984), and hydroxide and acid extractable P as having lower plant availability.

In this study, it has been elected to define P in terms implying its relative availability to plants (most-labile, less-labile, moderately resistant, and residual P) to enable

appreciation of P transformation in relation to plant use. This in itself may be a subject of debate, since there appears to be a diversity of opinions in defining soil P in terms of plant P requirements.

Extraction and definition of P fractions. A procedure has been adopted for the study which permits levels of soil P to be apportioned quantitatively along a gradient of availability from labile at one end to resistant at the other. This enables a description to be made of the nature of P in unfertilised soil and of the movement and transformation of P in a fertilised soil. The analytical methods used have been adapted from those of Hedley *et al.* (1982a,b) and are summarised in Table 3.2.

The resin extraction method has been used to estimate P in its most biologically available form (most-labile P). The next step involved extracting a separate soil sample with 0.5*M* $NaHCO_3$ (pH 8.5) for 16 h. P measured in this extract represents the labile P pool. The 'less-labile inorganic P' was then determined by subtracting most-labile P from the amount of inorganic P in the $NaHCO_3$ extract. A succession of extraction steps in this way is shown in Table 3.2, and has permitted determination of labile organic P, microbial biomass P, moderately resistant inorganic and organic P and residual P. The use of separate soil samples for each extraction was considered to be more convenient than the successive extraction of one sample because of the large number of soil samples involved in the study, and eliminated losses of sample between extracts. Phosphorus in the extracts was determined

by automated colorimetric methods (Appendix 2). Procedures used for each extraction are given in the following sections.

Table 3.2 Extraction procedure to apportion soil P into fractions of varying lability

Form of extraction	Fraction Calculation	Descriptive term
Anion (HCO_3) exchange resin.....A		Most-labile inorganic P
0.5*M* $NaHCO_3$, pH 8.5 16 h extraction.....B	(B-A)	Labile inorganic P Less-labile inorganic P
0.5*M* $NaHCO_3$, pH 8.5 16 h extract, digested...C	(C-B)	Labile organic P
0.5*M* $NaHCO_3$, pH 8.5 fumigated and unfumigated soils, 30 min extraction	Difference	Microbial biomass P
0.5*M* NaOH.....D	(D-B)	Moderately resistant inorganic P
0.5*M* NaOH, digested extract.....E	(E-D-C+B)	Moderately resistant organic P
Kjeldahl digestion.....F	(F-E)	Residual P

More specific descriptions of soil sampling and preparations are given in individual studies

3.3.2 Extraction of phosphorus with anion (HCO_3) exchange resin

The resin bag procedure (Sibbesen, 1977, 1978) for estimating the most-labile form of inorganic P was used throughout this work. Details of the resin used, its conversion to the bicarbonate form and the extraction procedure are given in Appendix 1a. Briefly, 4 g of resin was used to extract P from 4 g of soil in 100 ml of water. The extraction period was 17 h (Sibbesen, 1978), and 80 ml of 0.5*M*

HCl was used to elute P from the resin.

Since its introduction by Amer *et al.* (1955), the resin method has been used widely to assess the P status of the soil (Cooke and Hislop, 1963; Vaidyanathan and Talibudeen, 1970; Tiessen *et al.*, 1983, 1984; Wolf *et al.*, 1986). It is considered to extract the most labile P pool (Amer *et al.*, 1955; Bowman *et al.*, 1978) in both high-P- and low-P-fixing soils (Wolf *et al.*, 1986). Sibbesen (1978) suggests reasons why the conversion to the bicarbonate anion is the most appropriate method for estimating the most labile P fraction. Enclosing the resin in the bag enables its easy separation from the soil (Zunnino *et al.*, 1972, 1973; Vaidyanathan and Talibudeen, 1970; Sibbesen, 1977, 1978). This was appropriate for use in these studies which involved analysis of fresh soils with different characteristics (sandy to organic).

Barrow and Shaw (1977) suggested that the rate of sorption of P by the resin was restricted by the rate of transport of ions between the bulk solution and the surface of the resin. Therefore, tests were done on some of the soils used in the study to determine the extraction period, and to examine the effect of soil drying and soil pH on the amount of P extracted.

3.3.2.1 Effects of time of extraction on resin P

Experimental procedure. The yellow podzolic, red podzolic and yellow earth soils$_{A}^{(0-5\,cm)}$ were used in this experiment. Samples from the fertilised yellow podzolic field treatment were also tested because they had the highest P content. Resin

extraction was done on 2 mm sieved soils for 0.5, 1, 2, 4, 8, 16, 24, and 40 hours; there were four replications for each extraction period.

Results and discussion. Figure 3.2 shows that resin P extracted after 16 h was less than 10% of that extracted before 16 h, and would not increase significantly by increasing the extraction period for the soils used in this study. The 17 h extraction period (Sibbesen, 1978) was therefore used throughout the study. The results also suggest that 2.0 mm sieved soils can be used reproducibly, so that the method can be used for extracting P from fresh soil samples without grinding to pass 0.3 mm, as recommended by Sibbesen (1978). The fresh soils used could not be ground to pass the 0.3 mm sieve.

3.3.2.2 The effects of air drying on P extracted by resin from soils

The resin method for P extraction has normally been done on ground, air dried soils, which aids passage of soil through the mesh of the resin bags (Sibbesen, 1978). Both the drying and the fine grinding of soils may cause qualitative and quantitative chemical changes in the soil, and hence affect the P values obtained. The influence of air drying has been observed to affect the $NaHCO_3$ extractable P pool due to the release of P from the killing of the soil biomass (Sparling, 1985; Sparling *et al.*, 1985). Sparling also suggested that this effect will vary between soils, because organisms of drier soils are more resistant to air drying, thus releasing less P. An experiment was done to determine if a relationship

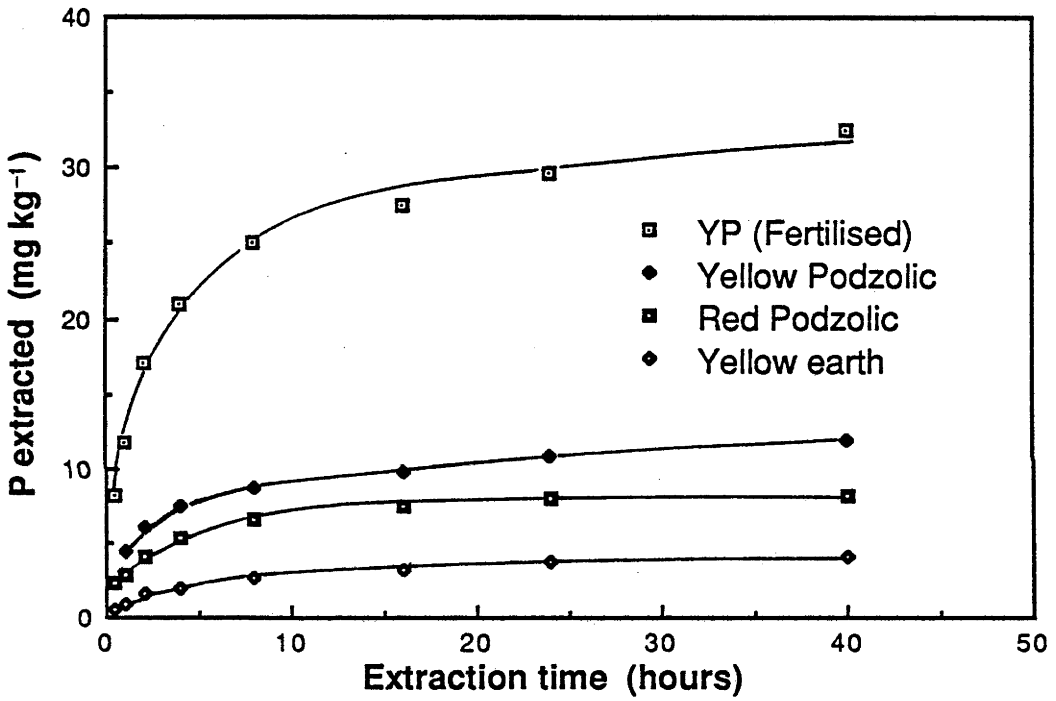


Figure 3.2 The effect of the period of shaking on P sorbed by the resin in mesh bags from 2 mm sieved soils.

existed between values of resin extractable P before and after air drying.

Experimental procedure. Eleven surface soil samples (0-5 cm) and four subsurface soil samples (5-20 cm) were collected from sites with different soils and land uses. Of the eleven soils, five* were used in this study. The field fresh soils were sieved (2 mm) and resin extractable P was determined on the soils before and after air drying. The values obtained were all expressed on an air dry basis.

Results and discussion. Figure 3.3a represents the regression analysis of the mean values of six replicate determinations on each soil. On average, air drying increased the amount of resin extractable P. Such increases have been attributed to P derived from soil microorganisms killed during air drying (Birch, 1958; Bowman and Cole, 1978; Marumoto *et al.*, 1977, 1978; Sparling *et al.*, 1985). There was a high correlation between the quantities of P determined before and after air drying the soils. However, Figure 3.3b shows that the differences were comparatively large for soils with low most-labile P (resin extractable P) and insignificant for soils with high most-labile P. The untreated soils used in this study had low most-labile P (< 10 mg kg^{-1}). The air dried samples had lower variability ($Se = 0.24$) than the fresh soils ($Se = 0.65$), consistent with Anderson's and Bervely's (1985) suggestion that air drying of soils brings about greater control of sample variability.

Preparation of field fresh soil samples for analysis is

* yellow and red podzolics, krasnozem, yellow earth and siliceous sand

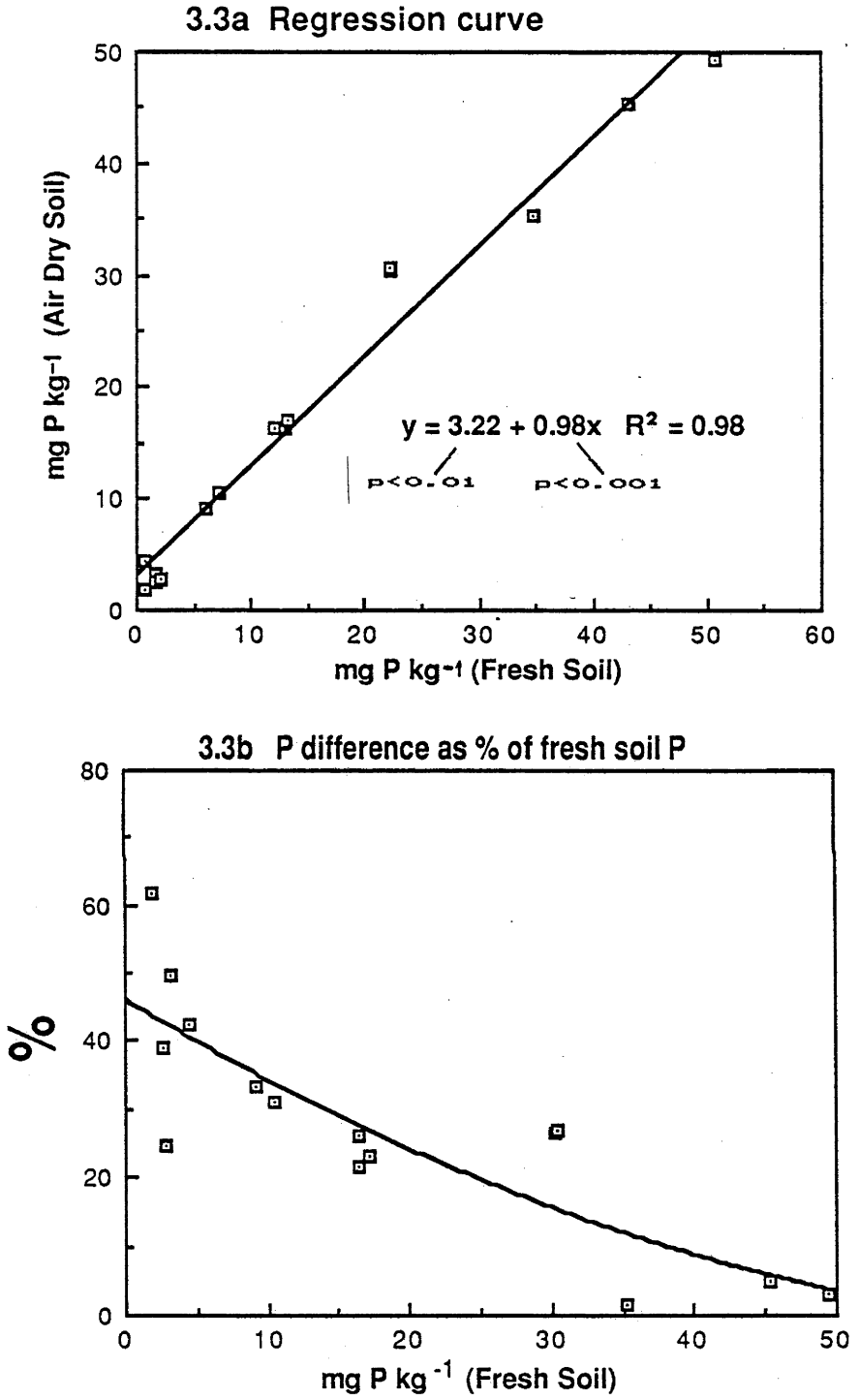


Figure 3.3 The relationship between P determined from fresh and air dried soil samples (3.3a), and the difference as % of P determined from fresh soil (3.3b).

labour intensive if sampling errors are to be minimised. The ease of sieving fresh soils differs with soil type and the moisture content. It was therefore concluded that for routine P analysis, air drying prior to sample preparation for analysis is preferable. However, where mineralisation studies are involved, careful handling and use of field fresh soil should give a better representation of the processes occurring at the time of sampling. The latter approach was used in the P transformation studies (Ch. 5, 6, 7 & 8).

3.3.2.3 Soil pH changes during P extraction with the resin

When the anion exchange resin used in the extraction of P is in the bicarbonate form, some of the bicarbonate ions are adsorbed onto the soil particle surfaces and some are released into the solution. This will increase or decrease the pH depending on the soil (Sibbesen, 1978). Tests were conducted to see the extent of pH change on some of the soils used in the study.

Experimental procedure. Soil pH changes during the extraction of resin P were examined in triplicate with five surface soils (0-3 cm). Samples were shaken in water for 17 hours in the presence and absence of the bicarbonate resin. After the shaking, the resin bags were removed and the pH determined in the supernatant solution (1:20) after settling for 30 minutes. The amount of P extracted was also determined.

Results and discussion. The results are shown in Table 3.3 and they confirm Sibbesen's (1978) observation. As the pH of the soil increased towards neutral, the effect of the resin on pH decreased. For the soil with the pH in the alkaline range (sewage treated yellow podzolic, YPS), extraction of P with the resin decreased its pH. Sibbesen (1978) suggested that these changes resemble the activity of plant roots which accumulate bicarbonate in the rhizosphere resulting in an increase in rhizosphere pH in acid to neutral soils and a decrease in rhizosphere pH in calcareous soils. As this bicarbonate is likely to have an effect on the P uptake by plants, the resin (HCO_3 form) is likely to be better correlated with plant P uptake.

Table 3.3 pH changes in the soil extract after P extraction with the resin. Means of triplicate analyses.

Soil	pH without resin bag	pH with resin bag	Change in pH	P extracted mg kg^{-1}
YE	5.20	5.81	0.61	4.45
K	5.23	5.64	0.41	0.96
YP	5.74	6.06	0.32	6.52
RP	5.95	6.09	0.14	7.73
YPS	7.78	6.53	-1.25	19.35

See Section 3.2 for abbreviations used for the soils

3.3.3 The extraction of P with 0.5M $NaHCO_3$

In experiments involving soil P characterisation after fertiliser application and forest harvest (Ch. 4 & 6), its extraction with sodium bicarbonate and subsequent colorimetric analysis were done according to Colwell (1965) (Appendices 1b

and 2b). 1.0 g of air dry soil in 100 ml extractant was used. Where the soil P content was low, the amount of soil used was increased and is reported in the text.

To speed up the phosphomolybdate complex development for colorimetric analysis, the mixed solutions are sometimes passed through an oil bath at about 95°C. This also improves the sensitivity of the measurements. Experiments (Salt, 1968) have shown that when the phosphate content of extracts containing organic material is measured using heat for colour development, inorganic P together with an amount of heat hydrolysable organic P is measured as well. Tests were done on some soils to determine the extent to which hydrolysable organic P affected measurement of inorganic P from the $NaHCO_3$ and NaOH extracts.

Experimental procedure. Phosphorus in 0.5*M* $NaHCO_3$ (pH 8.5) and 0.5*M* NaOH extracts was determined in triplicate for each of eight yellow podzolic soils taken from different treatments (control, fertiliser and sludge) and at different depths. The automated methods of Fogg and Wilkinson (1958) and of Murphy and Riley (1962) were used to find out how much P was released by the hydrolysis of organic matter in the extracts during analysis. The method of Fogg and Wilkinson uses ascorbic acid and ammonium molybdate for phosphomolybdate colour development at 95°C, whereas that of Murphy and Riley uses ascorbic acid, ammonium molybdate and potassium antimony tartrate at 37°C.

Results and discussion. In the analysis of $NaHCO_3$ extracts, the organic material precipitated by neutralisation

was removed by a 3.0 um membrane filter before adding the reagents. Table 3.4 shows that after the removal of the precipitated organic matter, inorganic P determined at 95°C was slightly higher than that determined at 37°C. This could be partly explained by the improved sensitivity in the measurement done at 95°C. In this thesis, measurement of inorganic P in the $NaHCO_3$ was with the automated method of Fogg and Wilkinson (1958) following filtration of the neutralisation precipitate through a 3.0 um membrane.

Table 3.4 The influence of heat on organic P hydrolysis during the phosphomolybdate colour development. Units, mg P kg^{-1} air dry soil, means of 3 analyses.

Soil (depth, cm)	$NaHCO_3$ Extracts			NaOH Extracts		
	37°C	95°C	% increase	37°C	95°C	% increase
YP (5-10)	3.2	3.9	21.9	11.6	16.0	36.8
YPS(2.5-5)	3.6	4.3	19.4	12.6	17.9	42.1
YP (2.5-5)	9.4	10.3	9.5	16.8	22.5	33.9
YPF(10-15)	15.6	16.1	3.2	26.7	31.9	19.5
YPS(0-2.5)	21.2	22.1	4.2	35.6	39.1	9.8
YP (0-2.5)	19.6	21.4	9.2	40.7	49.7	22.1
YPF (5-10)	30.6	30.9	1.0	52.3	57.0	9.0
YPF (0-5)	46.0	48.8	6.1	64.8	70.3	8.5

YPF: Yellow podzolic, fertiliser treatment
YPS: Yellow podzolic, sewage sludge treatment

NaOH extracted greater amounts of organic matter than $NaHCO_3$, and filtration with a 3.0 um membrane filter was inpracticable. Therefore, removal of the precipitated organic material was done by centrifuging at 4000 rpm. Table 3.4 shows large differences between the amounts of P determined in

the centrifuged extracts at the different temperatures. This suggests that centrifuging did not remove all the organic matter, and the hydrolysis of the remaining (probably soluble) fraction caused the increase in P values measured at 95°C. The automated method of Murphy and Riley (1962) was considered a better estimate of inorganic P from NaOH extracts and was used in these studies.

3.3.4 Extraction of microbial biomass P

Measurement of the P content of the soil microbial biomass was made to assess its importance in P cycling and replenishment of available P in forest soils. P contained in microorganisms can represent a large soil pool (Anderson and Domsch, 1980), and is a relatively labile fraction of soil organic P. Techniques for estimating biomass P have been developed (Brookes *et al.*, 1982; Hedley and Stewart, 1982) and utilise chloroform as a biocidal agent. Microbial biomass P is the difference between P extracted by $NaHCO_3$ from soil with and without fumigation. McLaughlin *et al.* (1986) used hexanol as a biocide and found it to be as effective as chloroform which is more carcinogenic. Therefore hexanol was the biocide used in microbial biomass P studies.

Hedley and Stewart (1982) recommended 16 h $NaHCO_3$ extraction after fumigation of the soil with the biocide, while Brookes *et al.* (1982) found no increase in P by prolonging the extraction beyond 30 minutes. Tests were conducted on the effect of the two extraction times on the release of soil and microbial P in some of the forest soils

used in the study.

3.3.4.1 The effect of extraction time on the release of soil and microbial biomass P

Experimental procedure. 2 ml hexanol were added to 5.0 g fresh soil, mixed and incubated for 24 h at 25°C. The soils were then extracted with 0.5*M* $NaHCO_3$ (pH8.5) for 30 minutes (Brookes *et al.*, 1982; McLaughlin *et al.*, 1986) and 16 h (Hedley and Stewart, 1982).

Results and discussion. Plate 3.1 shows that the amount of organic matter extracted during a 16 hour period was much more (darker colour) than that extracted during a 30 minute period. The effect was the same for the extracts from fumigated and unfumigated soils. The increased amount of extracted organic material necessitated frequent changes of the autoanalyser microfilter which removes the precipitated organic matter following neutralisation. It was for this reason that Colwell (1965) reduced the amount of soil sample from 5.0 g to 1.0 g. However, while the 16 hour extraction period minimises errors resulting from non uniform periods of exposure of the soil to the extracting solution, the 5.0 g in the 30 minute procedure give a more representative sample, especially where fresh soil is to be extracted.

Table 3.5 shows that the 16 h extraction released more inorganic P (IP) and organic P (OP) from the soil than the 30 minute extraction. There was no general trend of change in the total microbial biomass P (MPt). The changes were slight, suggesting that the microbial P was independent of the period

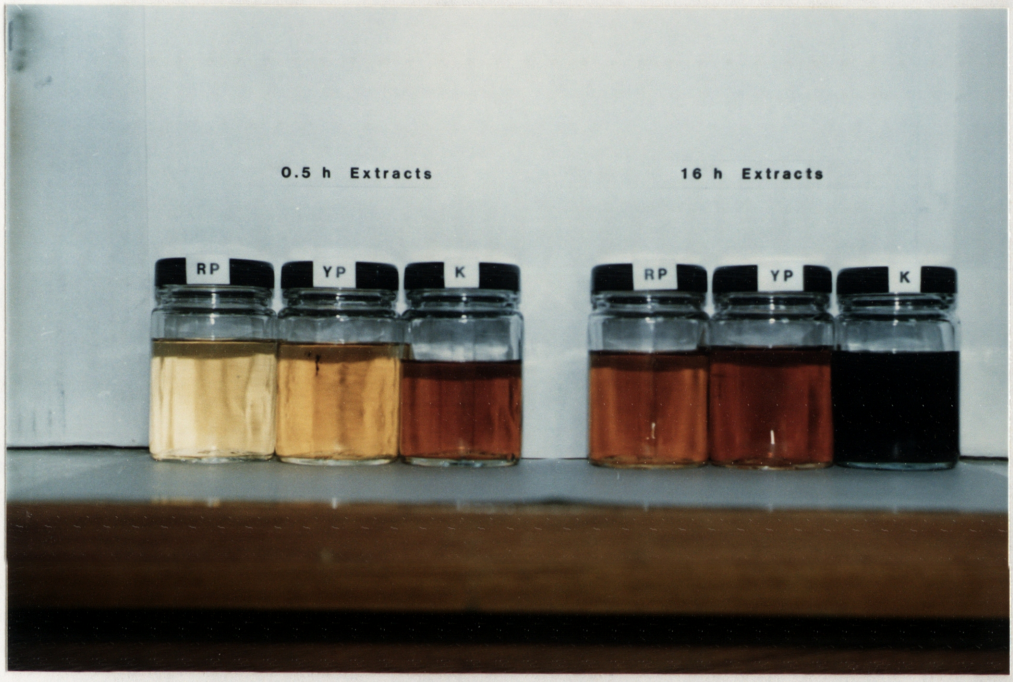


Plate 3.1 Visual differences in the amount of organic matter extracted in different soils during two periods of extraction with 0.5M $NaHCO_3$.

RP = red podzolic soil
YP = yellow podzolic soil
K = krasnozem soil

of extraction with sodium bicarbonate. It is also possible that exposure of the released microbial P to the soil during the different periods of extraction could have resulted in its differential sorption. Bowman *et al.* (1978) also noted that longer periods increased the background against which biomass P is measured as a result of large releases of P from live bacterial and fungal cells. The observed increase in organic matter extracted would result in large errors in soils with low biomass P. Because of these reasons, the 30 minute extraction was used in the biomass P studies.

Table 3.5 A comparison of the amount of P extracted during the 30 minute and 16 hour extraction periods. Means of 3. Data not corrected for sorption

	Extracted P ($mg\,kg^{-1}$)					
Soil	IP	OP	MPi	MPo	MPt	MPi/MPt
			30 minute extraction			
K	0.7	19.1	12.7	5.5	18.1	0.70
RP	3.9	7.0	6.1	2.3	8.4	0.73
YP	5.2	7.1	5.1	1.5	6.6	0.77
YPS	24.7	9.9	1.9	1.2	3.3	0.57
YPF	27.5	9.0	2.8	0.8	3.6	0.78
			16 hour extraction			
K	4.4	42.2	11.5	3.6	15.2	0.76
RP	11.8	17.5	7.4	1.9	9.3	0.80
YP	12.2	19.2	3.9	1.0	4.9	0.80
YPS	62.7	26.4	3.7	1.7	5.4	0.69
YPF	38.4	17.4	2.1	1.6	3.7	0.57

YPF = Yellow podzolic, fertiliser amended;
YPS = Yellow podzolic, sewage sludge amended.

The results also show that most of the released microbial biomass P after fumigation was in the inorganic form (MPi)

rather than in organic forms (MPo) as had been observed earlier by Brookes *et al.* (1982). MPi fraction was on average 70% of the MPt as compared to 91% observed by Brookes *et al.* (1982, 1984). These values, however, are not corrected for sorption.

3.3.4.2 Correction for P sorbed during extraction for microbial biomass P

Traditionally, biomass P has been estimated by assuming that 40% of the P in the biomass is rendered extractable from soil by 0.5*M* $NaHCO_3$ after cell lysis (Anderson and Domsch, 1980; Brookes *et al.* 1982, 1984; Hedley and Stewart, 1982). This value (termed K_P) is an average estimate as the composition of the microbial biomass and the soil sorption capacities are not the same between different soils. The different K_P values for three soils in the experiments of McLaughlin (1986) confirm this observation. However, the nature of microbial biomass in a given soil is not at a steady state, and will change with time, disturbance or cultural treatment. This suggests that for very accurate measurements of the biomass P, recovery factors should be determined regularly. This would be very laborious, and for practical reasons the experiments involving biomass calculations reported in this thesis (which covered many treatments and times) are based on the K_P factor of 0.4, corrected for sorption according to Brookes *et al.* (1982).

Brookes *et al.* (1982) suggested that a standard "spike" of 25 mg P kg^{-1} soil be added to soil during extraction and the percentage recovery of added P determined. Sorption

isotherms were constructed for P sorbed against extraction by 0.5*M* $NaHCO_3$ (30 min) for soils used in the mineralisation experiments (Ch. 7 & 8), so that, if necessary, improvements could be made by matching the spike to the amount of P released by hexanol fumigation. The different ranges of P additions were added in the extracting solution at a soil solution ratio of 1:20.

The sorption curves are shown in Figure 3.4. Different soils showed different abilities to sorb P. However, the percent recovery of the added P for each soil did not vary over the range used in this experiment. The average percent recoveries for the soils are shown in Figure 3.4 and were used to correct for microbial P sorbed during the extraction of the fumigated soil. The calculation of Brookes *et al.* (1982) was on the basis that most of the biomass P (91%) was recovered in the inorganic form. Results in Table 3.4 show substantial amounts of microbial P extracted as organic P, which is consistent with data obtained by Hedley and Stewart (1982) and McLaughlin *et al.* (1986). The calculation was therefore modified on the assuption that the microbial organic P released during fumigation was not sorbed by the soil. The following equation was used to calculate the microbial biomass P (MP) content of the soil:

$$MP = ((MPi/s)+MPo)/0.4$$

where: MPi is the microbial inorganic P

s is the soil P sorption recovery factor (Figure 3.4)

MPo is the microbial organic P

0.4 is a constant representing P in the biomass rendered extractable by hexanol fumigation.

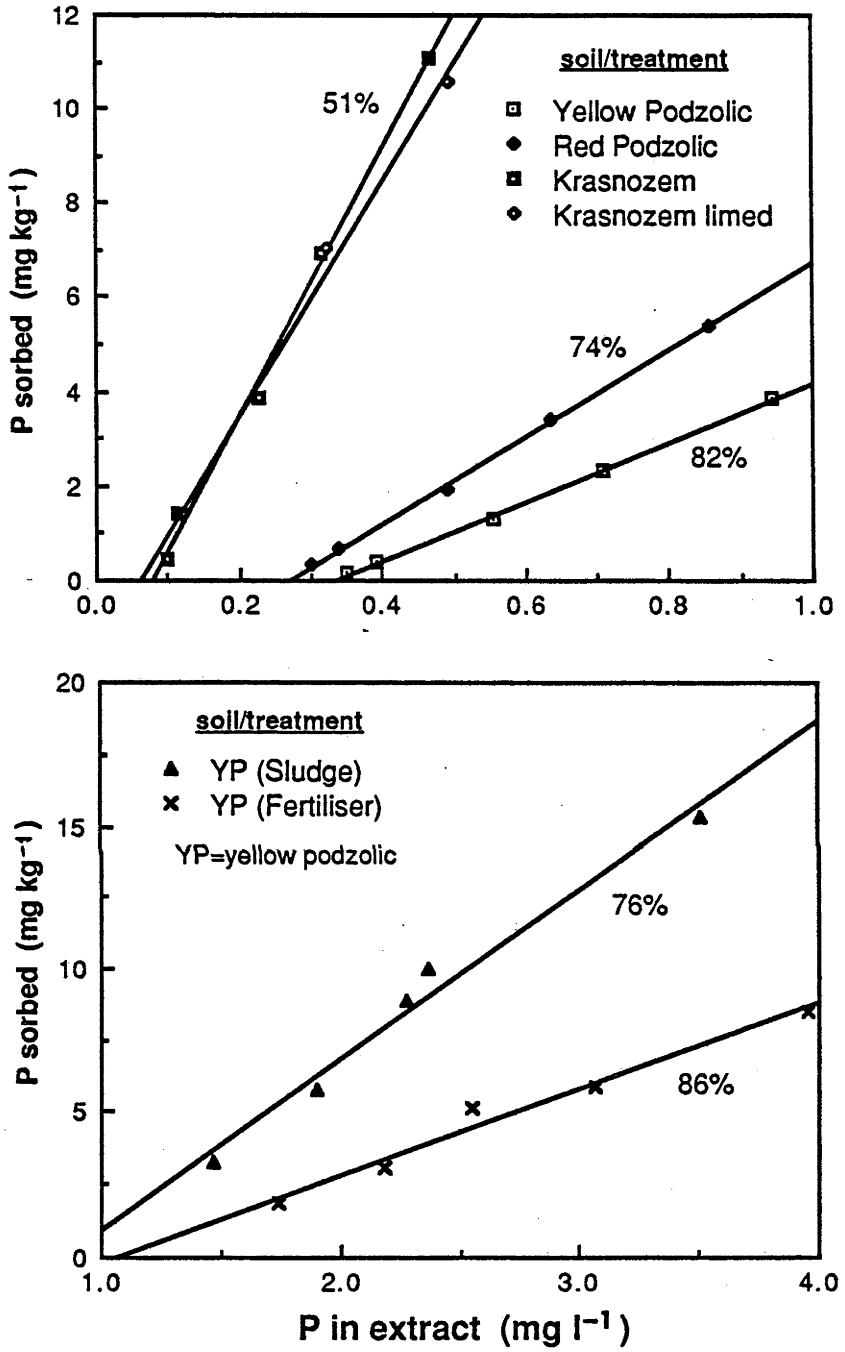


Figure 3.4 Sorption of added P by different soils during $NaHCO_3$ 30 minute extraction.

(Means of triplicate analyses)

When this equation was applied to the 30 minute extraction biomass P in Table 3.4, the ratios MPi/(MPi+MPo) increased to 0.82, 0.78, 0.80, 0.66, and 0.80 for the K, RP, YP, YPS and YPF respectively, but were still below the 0.91 value of Brookes *et al.* (1982, 1984). As the K_P value used was a mean value, the corrected values can only be regarded as approximate values for the different soils.

3.3.5 The extraction of P with NaOH

The procedure of Lambert (1978) was used and aimed at maximising extraction of organic phosphorus. 1.0 g of soil was extracted with 50 ml of 0.5*M* NaOH for 16 hours and filtered through two No. 42 filter papers. When the filtrate was neutralised, the organic precipitate from the NaOH extract was removed by centrifuging at 4000 rpm, and the inorganic P was determined by the automated method of Murphy and Riley (1962), at 37°C. The NaOH extracted organic P was obtained as the difference in P between digested (Thomas *et al.*, 1967) and non digested filtered extract.

3.3.6 Kjeldahl digestion of phosphorus

Kjeldahl digestion of soil and plant P was performed according to Vogel (1958). 0.5 g of oven dried plant material or 1.0 g of oven dried soil were digested at 370°C in 5.0 ml of the 'digest acid' comprising of an initially heated mixture of 375 ml sulphuric acid, 0.375 g selenium powder and 37.5 g

potassium sulphate. The phosphorus in the digest was determined colorimetrically.

3.3.7 Determination of soil P sorption capacity

Soil P sorption was determined according to the method of Ozanne and Shaw, (1967) in which 100 ml of 0.01*M* $CaCl_2$ solution containing 100 ppm Hg and graded amounts of P were added to 5.0g of 2.0 mm sieved and air dried soil and shaken for 24 hours at 25°C. The extract was filtered and the concentration of P in the filtrate determined. This was used to calculate the amount of P retained by the soil at the solution concentration of P in the extract. The relationship X+Q vs C was used to establish the sorption curves. Where,

X = amount of P sorbed per unit of soil

Q = P initially present in the soil before it is brought into contact with a phosphate solution (Probert, 1983). Resin extractable P was used (Fitter and Sutton, 1975) as the initial value.

C = concentration of P in soil solution

3.4 Other Soil Chemical Analyses

3.4.1 Determination of soil pH

The soil pH was determined on air dry soils sieved through 2.0 mm, according to Tucker and Beatty, (1974). 10 g of air dry soil were shaken in 50 ml of distilled water for 1 hour and allowed to settle for 30 minutes. The pH was read

using a combined glass calomel electrode.

3.4.2 Extraction and Determination of Soil N

The mineral fraction of soil N was determined using the method of Bremner, (1965). 10.0 g of fresh soil were shaken in 50 ml of 2*M* KCl solution for 1 h. For hexanol fumigated soils 0.5*M* K_2SO_4 was used as the extractant to determine microbial biomass N as digestion was easier in this extractant. An aliquot of the filtrate from this extraction procedure was digested according to Thomas *et al.* (1967) to determine total microbial biomass N. N in the filtrate and digested filtrate was determined colorimetrically. A K_N value of 0.57 proposed by Jenkinson (1987) was used to estimate soil biomass N. The K_N factor represents the fraction of biomass N extractable by 0.5*M* K_2SO_4 from soil after fumigation with hexanol. As for microbial P, this is an average factor and represents approximate biomass N for individual soils.

3.4.3 Determination of soil biomass carbon

Changes in the quantity of microbial biomass C in soils was estimated by the chloroform fumigation-incubation method (CFIM) proposed by Jenkinson and Powlson (1975). Microorganisms in the soil were killed by fumigation with chloroform vapour. Subsequent mineralisation of the lysed cells after re-inoculation during a 10 day incubation period was measured from the evolved CO_2.

Field fresh soils were fumigated with purified chloroform vapour (Stotsky, 1965) for 24 hours. After fumigation, the chloroform vapour was removed by repeated evacuation under suction, and the soil was inoculated with one gramme of the original soil. The flush of CO_2-C from the decomposition of the killed microbial cells was calculated as the difference between the amount of CO_2-C evolved from fumigated and unfumigated soil after 10 days of incubation at 25°C. This method assumes that the basal respiration is not altered in the fumigated sample (Anderson and Domsch, 1978). The incubation after fumigation was in a 2 l wide neck jar with an air tight rubber closure. 20 ml of 10% KOH (w/v) was used to trap the evolved CO_2, which was in turn determined by the titration of the remaining KOH. The unneutralised alkali was titrated with 1*M* HCl after adding 20 ml of saturated barium chloride solution (Stotsky, 1965) to a pH end-point of 7.2.

A weighted K_c factor of 0.41 (Domsch, 1978; Anderson and Domsh, 1978; Voroney and Paul, 1984) was used to estimate total biomass C in the soils.

3.4.4 Determination of O_2 uptake and CO_2 evolution by soils during incubation

Soil microbial respiration was determined using manometric respirometers. The respiration apparatus used are illustrated in Plate 3.2 and were assembled as described by Klein *et al.* (1972).

Fresh soil (200 g) was weighed into plastic bags and sealed. Three holes, one cm in diameter, were made in the bags to allow for gas exchange while minimising evaporation.

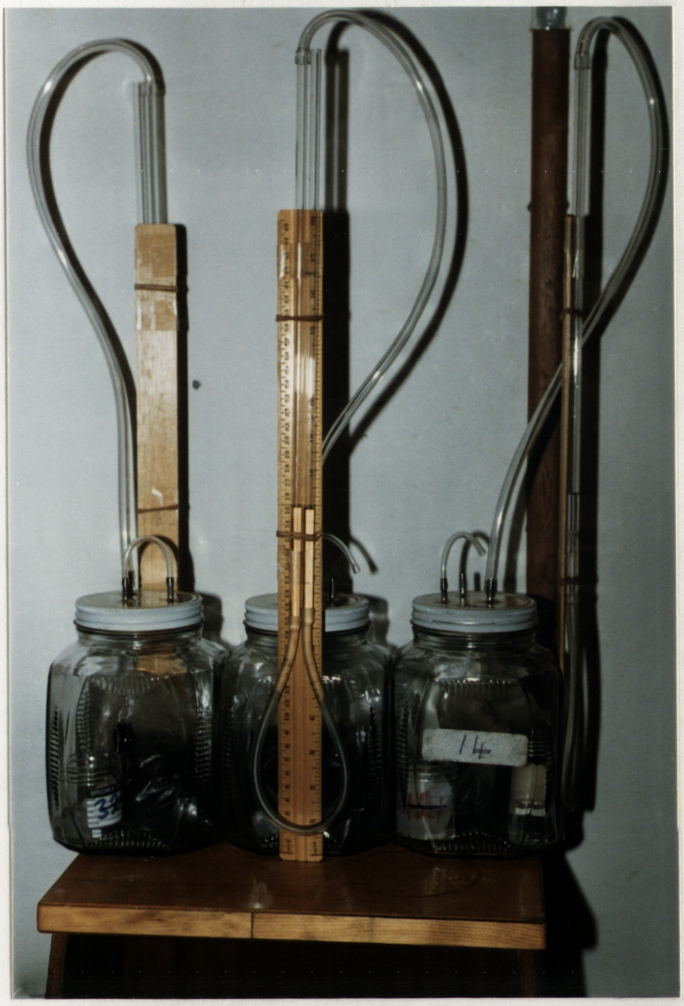


Plate 3.2 Apparatus used in soil respiration studies.

The soil was then put into the respirometers together with a vial containing 10 ml of distilled water, and another vial containing 20 ml of 20% (w/v) KOH. The water was used to maintain a constant humidity and to minimise evaporation from the soil, and the drying effect of KOH. Account was taken of fluctuations in ambient atmospheric pressure and temperature by using control respirometers (thermobarometers) in which soil was replaced by water of equivalent volume.

The respirometers were maintained in the dark at 25°C and measurements were started after a 6 h equilibration period. Pressure equilibration and aeration were done as required by opening one of the inlets to the bottle. A four day periodic flushing was done with a vaccuum pump for about one minute to replenish O_2. The KOH traps were replaced about every 10 days.

O_2 uptake was calculated by converting the manometric readings using standard manometric constants, according to the following equation:

$$k = \frac{Vg(ml)273/T + Vf(ml)a}{Po}$$

where k is the conversion factor for volume change to ml O_2
Vg is gas volume in the respirometer apparatus
Vf includes soil moisture and dry soil volume and any volumes of the liquids and solids used
T is temperature (°K)
a is the oxygen absorption coefficient at T
Po is the standard pressure

(From Stotsky, 1965; Klein *et al.*, 1972)

The CO_2 absorbed in the KOH traps was determined by titration (as in section 3.4.3). Respiratory quotient (RQ) was calculated from:

RQ = ml CO_2 evolved / ml O_2 consumed (Stotsky, 1965).

3.5 Statistical analysis and presentation of data

Generally, mean values of treatment or analytical replicates are presented in Figures and Tables together with a measure of variability. Most of the data were analysed by analysis of variance (Anova) using the GENSTAT Computer Program (GENSTAT Manual, 1977). Significant difference as used in the text refers to the range $p<0.05$ to $p<0.001$ and is specified in the text. Standard errors of difference (sed) of means, multiplied by 2, were used in the graphs. The GENSTAT Program was also used for regression analyses between measured parameters.

Where data were not analysed by Anova, the standard error (se) was used as the measure of variability. It represents the standard error of the treatment mean and is calculated from the square root of s^2/n, where s is the sample standard deviation and n is the number of observations from which the mean was calculated.

CHAPTER 4. PHOSPHORUS MOVEMENT AND TRANSFORMATION IN A YELLOW PODZOLIC SOIL

CHAPTER FOUR

PHOSPHORUS MOVEMENT AND TRANSFORMATION IN A YELLOW PODZOLIC SOIL

4.1 Introduction

Crop utilisation of inorganic P fertilisers in the year of application is generally less than 25% (Wagar *et al.*, 1986). The effectiveness of the remainder as a source of P for plants depends on the amounts reacting with the soil components, immobilised into microbial material, and leached from the root zone.

The availability of native and residual fertiliser P in forest soils has been assessed by a variety of methods (e.g. Kadeba and Boyle, 1978; Hopmans *et al.*, 1978), including the Chang and Jackson (1957) P fractionation procedure (Gentle *et al.*, 1965; Kelly *et al.*, 1983; Turner and Lambert, 1985). Because of the long-term nature of the forest crop, a dynamic approach to the analysis of forest soils is needed for successful predictions of the uptake of P, and other nutrients, by trees (Khanna, 1981a). Qualitative laboratory data have shown that water soluble phosphate fertilisers react with soils to produce metastable P minerals which may be readily available sources of P for plants, and may persist in soil for months or years (Huffman, 1962; Olsen and Flowerday, 1972). Comparable quantitative information, particularly regarding the rate and extent of transformation of these reaction products to increasingly less soluble forms in forest soils, would help predict fertiliser P effects during

rotations. The information would also indicate if established plantations will respond to fertiliser application according to species, establishment methods and stage of growth.

There has been a general trend for P levels in sewage sludges to increase, in some cases to levels comparable to inorganic fertilisers (Kirkham, 1982). This makes sewage sludge a potential source of P fertiliser for forest soils, and research is needed to test its effectiveness as a source of plant available P in forest soils.

The objectives of the studies in this Chapter were therefore to determine the distribution of different pools of native P, and of P from superphosphate fertiliser and sewage sludge over different periods after application to a yellow podzolic soil. Short term transformations among different forms of P may help in the understanding of long term availability of residual fertiliser P. The first experiment of this Chapter examines the movement and transformation of the labile (apportioned into most-labile and less-labile fractions) and moderately resistant inorganic P fractions over a period of 26 months in soils supporting a second rotation pine forest. The same theme is further developed in the second experiment, this time including organic and residual P fractions, and three different soil-litter and vegetation situations.

4.2 Experiment 1 Evaluation of surface applied fertiliser and sewage sludge P as sources of labile and moderately resistant inorganic P in soil

4.2.1 Soil used in Experiment 1

A detailed description of the yellow podzolic soils, and their distribution in Australia, is given by Stace *et al.* (1968) and Northcote (1979). The yellow podzolic soil used in the study is derived from adamallite (coarse grained, calcium rich, granitic rock). The A horizon is down to 40 cm deep, but bulk density increases quickly with depth. This horizon has modest permeability, with water storage of around 50%. The B horizon has very poor permeability and a bulk density between 1.7 and 1.8 g cm^{-3}.

The soil has low organic matter, and hence low total nutrient reserves. Fertility is concentrated in the surface 10 cm of the soil profile - organic carbon content decreases from 2.4% in the 0-2.5 cm region to 0.5% in the 10-15 cm region. Soil reaction is mildly acid with a pH (1:5 water) of 6.0 in the 0-10 cm and decreasing to 5.3 at 40 cm. Ca is the dominant exchangeable cation in the surface soil. The associated natural vegetation is mainly the dry sclerophyll eucalypt forest, some of which has been cleared and replaced with *P. radiata*.

4.2.2 Experimental procedure

The study has been part of the Biology of Forest Growth Research Project (BFG), being conducted by the CSIRO Division of Forest Research in collaboration with the Division of

* Holes left after soil sampling were filled with soil from elsewhere to prevent movement of water into them. The filled holes were marked for identification during subsequent sampling.

Soils, and the Department of Forestry of the Australian National University (Figure 3.1, Site 1). The project is concerned essentially with an understanding of growth and growth processes (e.g. nutrient cycling, photosynthesis, water relations) in a *Pinus radiata* stand.

The soil phosphorus experiment consisted of three treatments; control, solid fertiliser and sewage sludge treatments. The layout of the treatments is shown in Figure 4.1. Each treatment was applied to one 0.25 ha plot in a 10 year old second rotation radiata pine plantation. The fertiliser treatment consisted of a split application on 28 September and on 18 November, 1983. Each dose contained 200 kg N ha^{-1} as ammonium sulphate, 100 kg P ha^{-1} as single superphosphate, 50 kg K ha^{-1} as potassium sulphate and 5 kg B ha^{-1} as borax. The sewage sludge treatment consisted of a single application on 5 October, 1983, of 13.3 tons ha^{-1} (dry weight) of undigested lime-treated wet sewage sludge, with a chemical composition of 23.5% organic carbon and 43.8% calcium carbonate. The sludge added the equivalent of 176 kg N and 237 kg P ha^{-1}.

Both materials were evenly applied by hand to the soil surface, with special attention to four internal 2m * 2m subplots in each treatment. From each of these subplots, six soil cores were taken at the following sampling times: before treatment and 1, 4, 14, and 26 months after treatment*. The cores were subdivided into a number of depth sections and then bulked. Cores were 7.6 cm in diameter and 20 cm (for post-treatment control and sludge samples) or 30 cm deep. The soil samples were air dried, ground and sieved through 2.0 mm and stored in

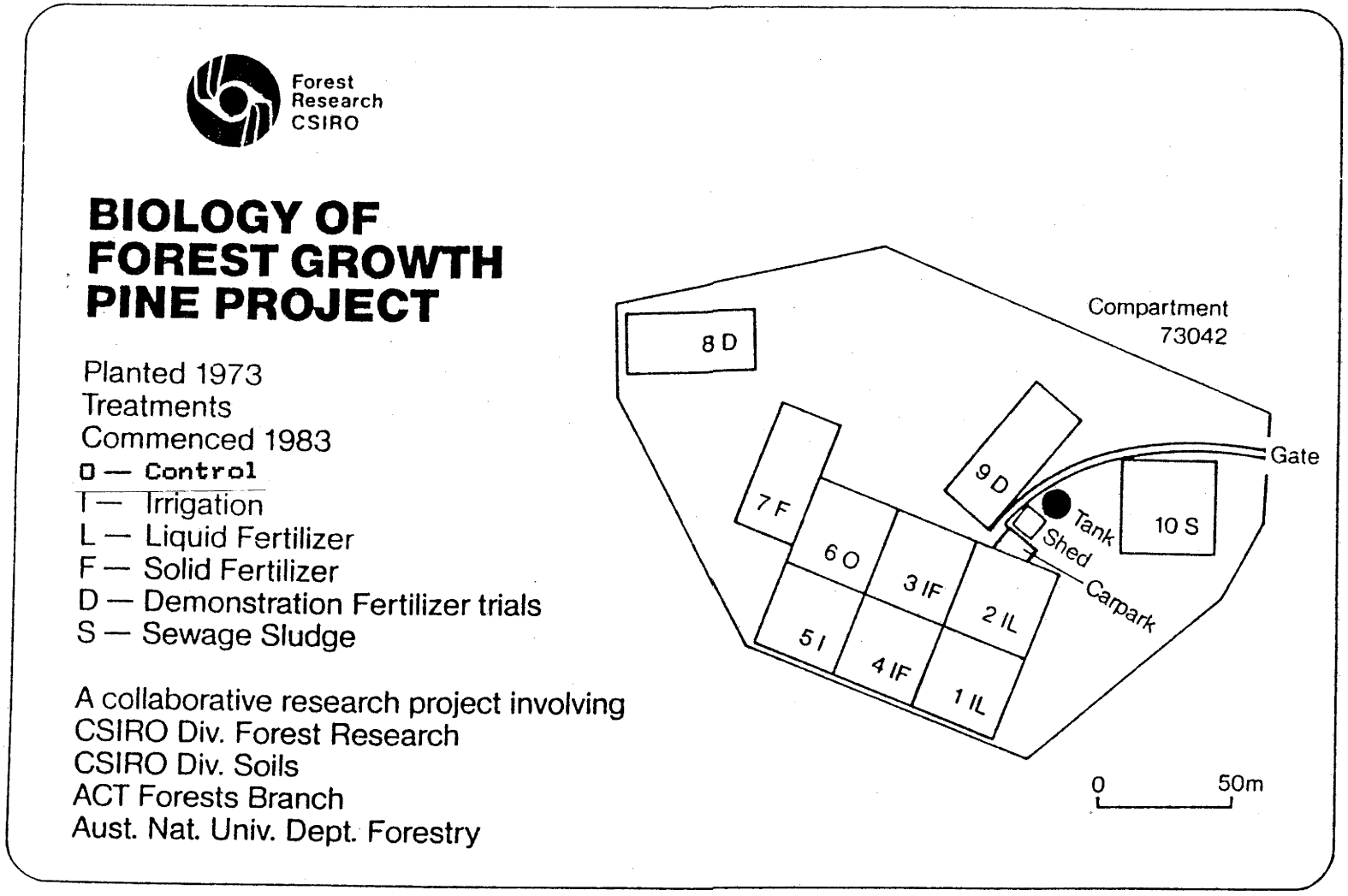


Figure 4.1 Layout of the Biology of Forest Growth Research Project. Plots 6 O (Control), 7 F (Fertiliser), and 10 S (Sewage sludge) were used in this study.

closed plastic bags until analysis.

The soils were analysed for resin extractable P (most-labile P), $NaHCO_3$ extractable inorganic P fraction (less-labile inorganic P), and NaOH extractable inorganic P fraction (moderately resistant inorganic P). The methods of analysis are described in Chapter 3. Soil P sorption was measured on soil samples from the three plots before, and 26 months after the treatments were applied. This was done for samples taken from 0-2.5, 2.5-5 and 20-30 cm depths. Sorption capacity was determined using the method by Ozanne and Shaw (1967) and described in Section 3.3.7.

As there were no external replications, treatments were analysed individually (ANOVA) for the effects of time on changes of different P fractions and their distribution in the soil profile. The use of pre-treatment data as covariate to adjust for starting point effects did not change the interpretation. Depth values in figures are mid-points of the soil profile sections sampled, and the P values are expressed on a unit depth of one cm.

4.2.3 Results and discussion

4.2.3.1 Comparison of the plots before treatment application

Generally the amount of all P fractions decreased with depth (Table 4.1). P content (per unit of depth) in the surface 0-2.5 cm was more than 4 times that between 20-30 cm. This reflects the effect of plant cycling on concentrating P in the surface of the soil profile, and the slow movement of P down the profile.

Table 4.1 Profile distribution before treatment of three P fractions in the top 30 cm of the yellow podzolic soil (means of all treatment plots)

P fraction (mg kg^{-1})	Depth (cm)					p<	sed
	0-2.5	2.5-5	5-10	10-20	20-30		
Most-labile IP	1.08	0.83	0.58	0.35	0.18	0.001	0.09
Less-labile IP	0.93	0.87	0.42	0.28	0.20	0.001	0.14
Mod. res. IP	3.04	2.00	1.18	0.88	0.61	0.001	0.42

IP = inorganic P; Mod. res. = moderately resistant

Pre-treatment comparison of the three plots is given in Table 4.2. There was significant between-plot variation for the most labile P ($p<0.001$) and labile P (0.01) fractions (Table 4.2). There were 26.1 kg P ha^{-1} of most-labile P in soil from the control plot to 30 cm - more than double that in soil from the sludge plot. Most-labile P in soil from the fertiliser plot was intermediate. Less-labile inorganic P was similar in soils from the fertiliser and sludge treatment plots and significantly less than in the control, while differences in the moderately resistant P were not significant (Table 4.2). The total of the three fractions to a depth of 30 cm shows initial plot differences. Soil from the sludge treatment plot had the smallest quantity of P in the three fractions (66 kg P ha^{-1}) while soil from the control plot had the highest (92.1 kg P ha^{-1}). There was also within-plot variation (especially in the sludge plot) as shown by the large standard errors of means of the fraction totals (Table 4.2).

Table 4.2 Quantities of P fractions before treatment application (totals of 0-30 cm depth).

P fraction (kg ha^{-1})	Treatment plot			p<	sed
	Control	Fertiliser	sludge		
Most-labile IP	26.1	16.8	11.4	0.001	2.1
Less-labile IP	23.1	13.2	12.3	0.01	3.3
Mod. res. IP	42.9	53.4	42.3	ns	
Fraction total	92.1(6.3)	83.4(5.3)	66.0(12.0)		

se of means in brackets

Most-labile P was the only fraction that showed significant plot-depth interaction, and Figure 4.2 shows that there was more resin extractable P in soil from the control than that in soil from other plots down to about 10 cm depth. The results show that the main treatment plots were different, and that this would have to be taken into account when comparing them.

4.2.3.2 Temporal variations of soil P in the Control Plot

Figure 4.3 shows the levels of the three P fractions at different sampling periods on the control plot. The most-labile P fraction declined by about 5 kg ha^{-1} after 14 months from the start of the experiment. Both less-labile and moderately resistant inorganic P fractions increased slightly over the 26 month period. Temporal variability in soil chemical composition has been observed by other researchers. Kemp *et al.* (1985) obtained significant temporal variability on 49 pasture sites over a 3 year period, which was correlated with a general estimate of soil moisture and thermal index for

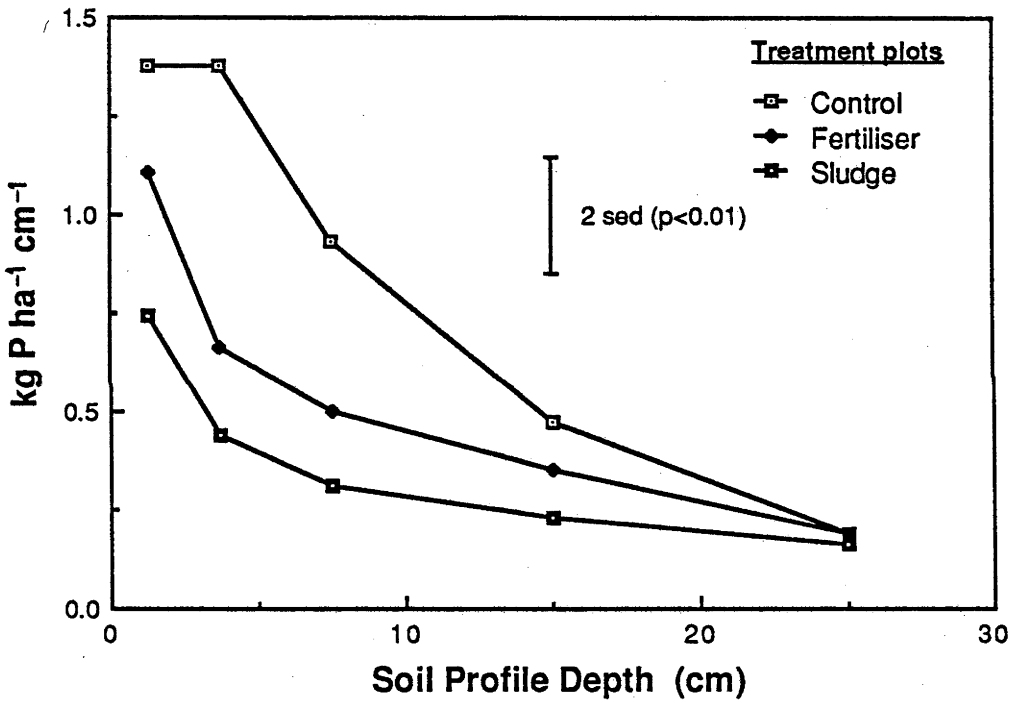


Figure 4.2 Resin extractable P at various depths in plots before treatments were applied.

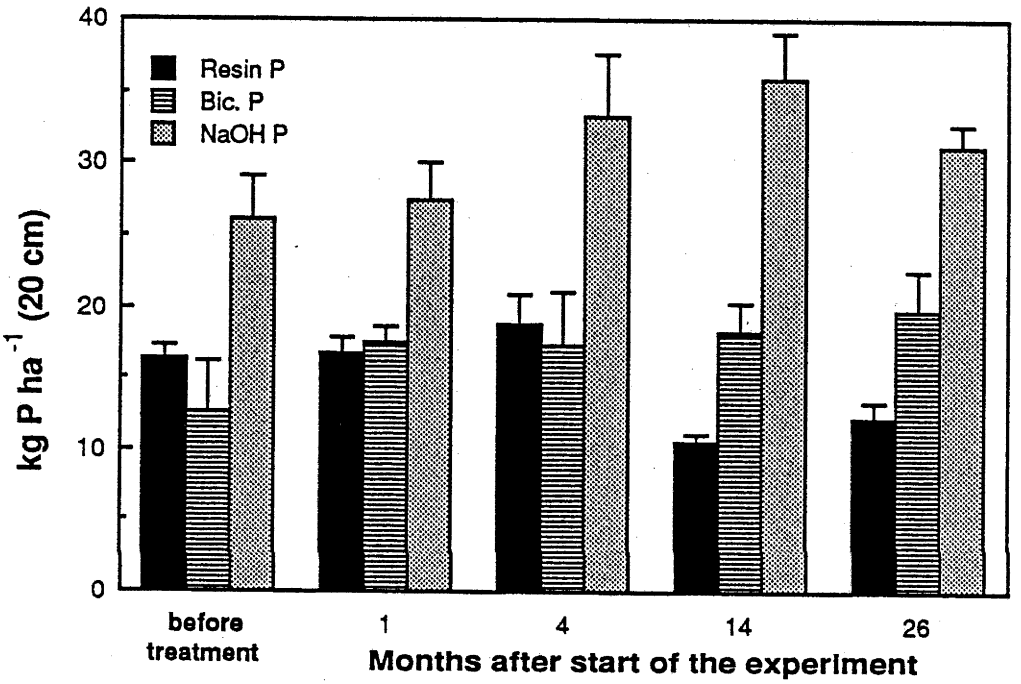


Figure 4.3 Temporal changes in the three P fractions in soils from the Control plot.

Resin P = most-labile P
Bic. P = less-labile P
NaOH P = moderately resistant P
(all three are inorganic fractions)

se of means presented as vertical T bars.

the sampling month. Based on detailed studies on the spatial pattern and seasonal variation in the amount of nutrients in a Scots pine forest soil (humus iron podzol), Usher (1970) recommended that soil samples for survey work should be taken in spring or early autumn at a standard distance from trees. This was in recognition of the effect of seasonal changes in the nutrient cycles.

Haines and Cleveland (1981) obtained significant monthly variations in Bray-P (and other nutrients) of 5 forested sites, and suggested that lower nutrient values were a result of both nutrient uptake during the growing season and leaching. Leaching of P will be tested in the fertiliser and sewage sludge treatments. It would be difficult to ascribe the change in the most-labile P to a contribution to plant P uptake, since plant depletion of the labile pool invariably results in mobilisation of less mobilisable P. Research needs to be done on the rate of recovery of labile P after depletion, for its importance lies in the fact that "available" P in the absence of applied P does not drop as fast as would be expected from crop removal (Larsen, 1967). Therefore, in addition to plant P uptake, changes in soil P of this magnitude could also result from soil variation (Ballard, 1980) or climatic influence (Usher, 1970; Harrison, 1979; Kemp *et al.*, 1985).

4.2.3.3 The effect of time on the distribution of fertiliser P in a yellow podzolic soil profile

Figure 4.4 presents changes in the three P fractions with time in soil from the fertiliser treatment. Figure 4.4a shows

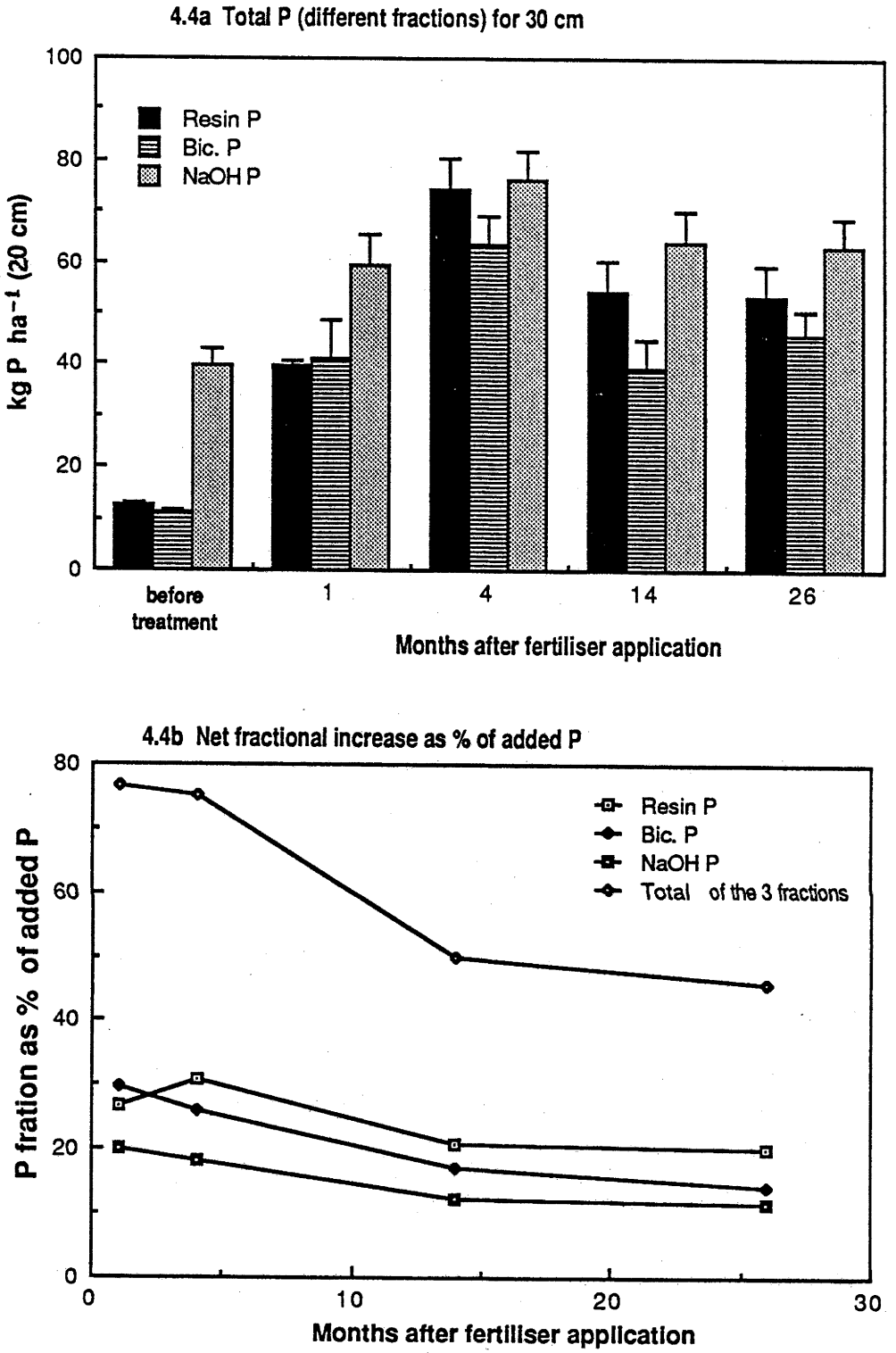


Figure 4.4 Temporal changes in the three P fractions in soils from the Fertiliser treatment plot.

the total P for each fraction to 30 cm, while Figure 4.4b presents the different P fractions as percentages of P added in fertiliser. The data for Figure 4.4b were obtained as the difference between P after treatment and P before treatment, expressed as a percent of the added P in fertiliser (this was 100 kg P at 1 month after treatment, and 200 kg P after 4 months). This procedure was used for the three P fractions.

Figure 4.4a shows that all three fractions increased after each fertiliser application (1 and 4 month analyses). P fractions in soils sampled at 14 and 26 months after treatment show that P in different fractions was more or less steady but had declined from the level at 4 months.

Figure 4.4b shows that fertiliser P was recovered in all three soil fractions soon after application (1 and 4 months) and accounted for about 75% of the added P. Holford (1977) suggested that initial P reactions with the soil solid phase probably reach a stage of metastable equilibrium in most soils within a week or two. More than 20% of the added P was not accounted for in the fractions, 1 month after the first fertiliser application, indicating that reactions to more resistant inorganic fractions or conversion to organic forms may equally be fast. However, some of this P may have moved deeper than 30 cm. Profile movement of P is discussed later in this section. There was a gradual decline in the amount of fertiliser P recovered in the three fractions. This decline may be a result of transformation of metastable reaction products to less soluble P forms (Sample *et al.*, 1980), and retention or immobilisation of P by biological pathways (Anderson, 1980).

Figure 4.4b also shows that the trend of decline in the recovery of the P fractions slowed with time, i.e. between 4 and 14 months, most-labile P dropped by 10% while between 14 and 26 months, it dropped by 0.5%. Similar data, showing that the decline in fertiliser P effectiveness becomes slower with time, was obtained by Arndt and McIntyre (1963) and Barrow (1974b).

P uptake by plants also contributes to the decline in fertiliser P. Because fertiliser P is initially readily available, it encourages vigorous growth, and therefore increases P uptake. P uptake by trees of the order of 1.7 to 13.8 kg ha^{-1} yr^{-1} (Ballard, 1980) is low, and therefore its contribution to the observed decline in the recoverable P fractions would be small.

Microbial growth in soils involving uptake of inorganic P (in addition to other elements) may also contribute to the decline in recoverable P. Blair *et al.* (1976) found that 28% of labelled fertiliser applied to soil was in organic forms after 7 days, possibly microbial. McLaughlin (1986) demonstrated that about 65% of residue P under controlled microbial growth conditions, and 25% residue P in the field situation, had entered and remained in the microbial P pool 50 days after addition of plant residue. In a forest situation, litter fall may encourage microbial immobilisation of inorganic fertiliser P. Changes in soil organic and residual P were studied in experiment 2 of this Chapter.

Figure 4.4b shows that during the chemical transformation of (fertiliser) inorganic P, it was not quantitatively transferred from one fraction to another. Thus the decline in

most-labile P did not result in an increase in less-labile P. This indicates that there are either different pathways involved, or the rates of conversion from one fraction to another are the same.

Figure 4.5 shows the profile distribution of P after fertilisation. At one month after the first addition of 100 kg P ha^{-1} all the three fractions were significantly increased down to about 7.5 cm of the soil profile. Following the second application of 100 kg P ha^{-1} there was a further increase in all fractions. At 4 months, the effect was significant at the depths of 3.75 to 17.5 cm for the most-labile P fraction (Figure 4.5a), and at the depths of 3.75 to 7.5 cm for the less-labile P fraction (Figure 4.5b). It should be noted that the soil was not sampled until 3 months after the second application of 100 kg P ha^{-1}, so that sorption reactions may have reduced the amount of these fractions more than that recorded after the first fertiliser application. This may partly explain the small increase of P in the surface 0-2.5 cm. Alternatively, sorption sites may have been saturated, allowing excess P to move down the profile. This is supported by data from the sorption studies (Section 4.2.2.5) which show maximum sorption at less than 100 mg kg^{-1}. This is about 200 kg P ha^{-1} for the 2.5 cm depth - the amount supplied in the fertiliser and equivalent to that required to 'saturate' soil sorption sites to a depth of 0-25 cm.

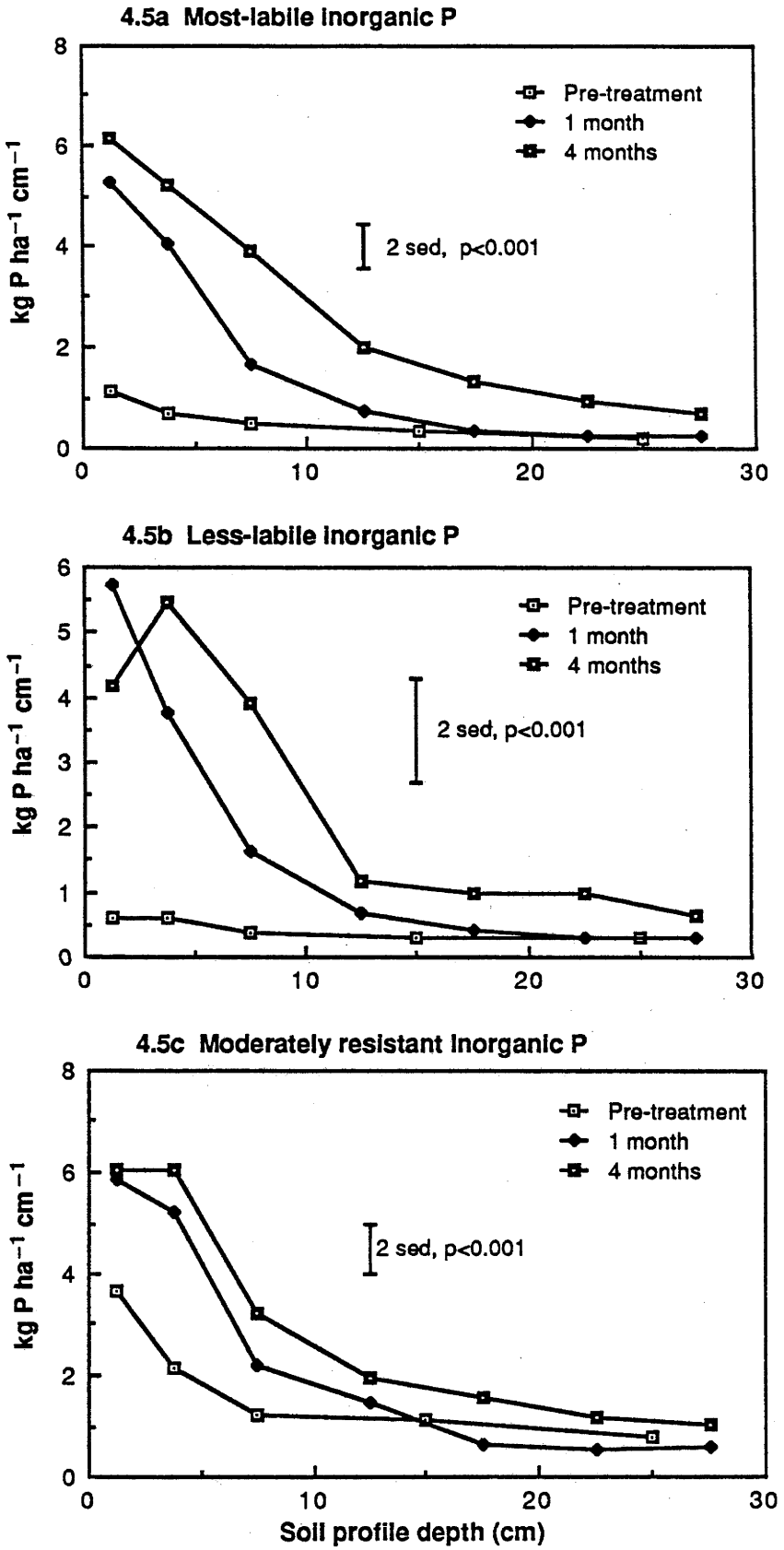


Figure 4.5 Profile distribution of P in soils soon after fertiliser application

Figure 4.6 shows the profile distribution of the P fractions in soil from the 4th to the 26th month. There was less P in all three fractions, in the samples taken after the 4th month, than in soils at the 4th month, particularly in the top 10 cm of the soil profile. This is the region where the increases in P were greatest and therefore the region where the most changes are likely to occur. It is the region with highest root and microbial activity, and soil constituents (e.g. Ca, CEC, silt, clay and organic matter) which contribute to the reduction of inorganic fertiliser P.

There was no increase with depth in P fractions after the 4th month. This suggests that profile movement of P was immediately after fertiliser addition, probably by mass flow of the dissolved fertiliser. Experiments by Kanchanasut *et al.* (1978) and Scotter and Kanchanasut (1981) showed rapid fast profile P movement could be due to preferential flow through soil macropores. Other researchers (Bar-Yosef and Shekhosalmi, 1976; Logan and McLean, 1973a,b; Spencer and Wlasow, 1960) suggested that in the presence of moisture, there will be movement of applied P by mass flow until its concentration in soil solution falls considerably due to rapid sorption reactions. The yellow podzolic soil has relatively low P sorption capacity (Section 4.2.2.5), and both the above processes could have operated.

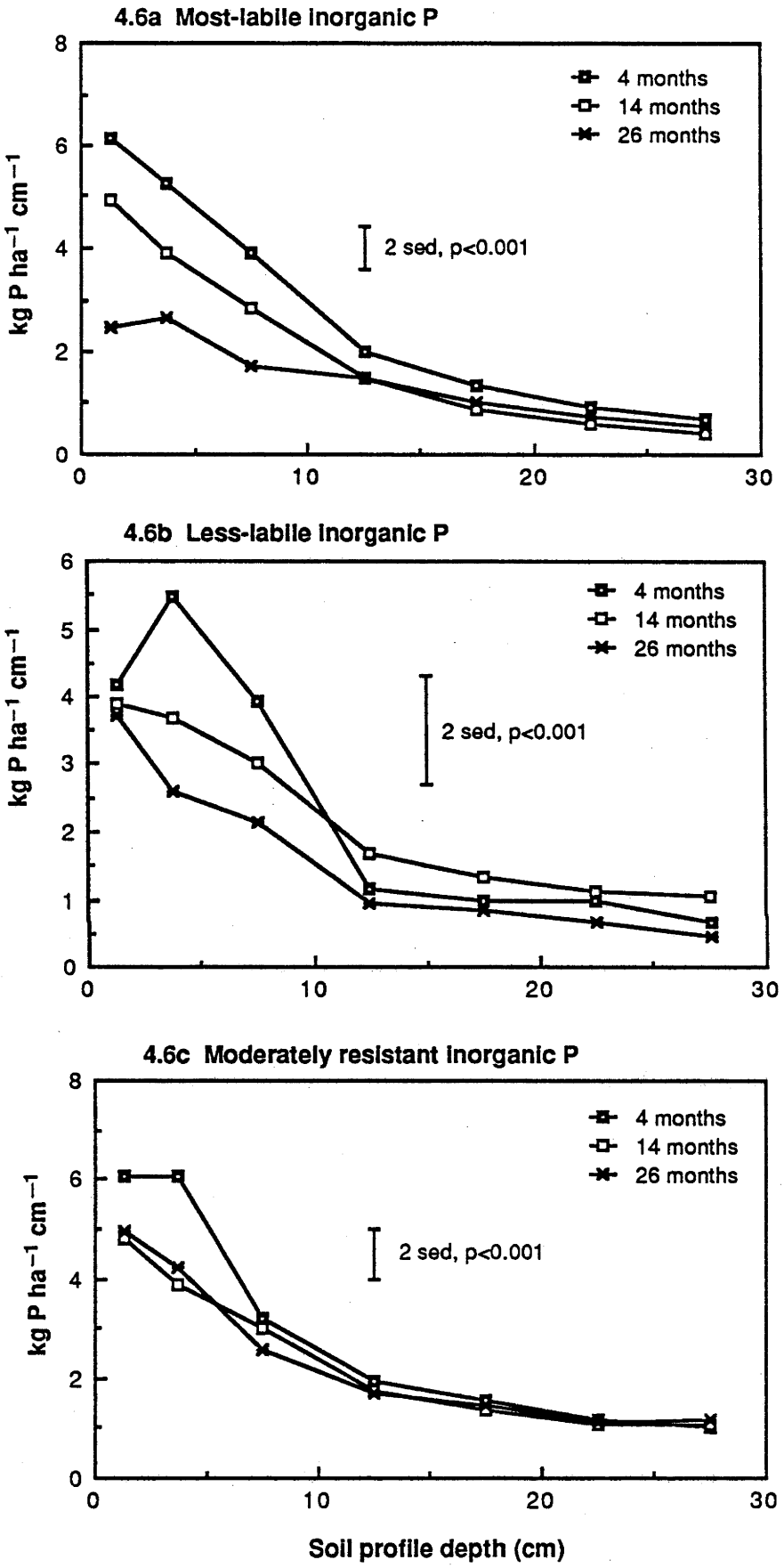


Figure 4.6 Profile distribution of P in soils after the second and final fertiliser application.

4.2.3.4 Effects of addition of sewage sludge on soil P

Figure 4.7 presents changes of P in soil from the sewage sludge treated plot. Figure 4.7a shows total P for each fraction to 20 cm, while Figure 4.7b presents the different fractions as percentages of P added in sewage sludge. Data for Figure 4.7b were obtained as the difference between P after treatment and P before treatment, expressed as a percent of the 237 kg P added in sewage sludge. The responses of the soil P fractions to the sewage sludge treatment were small, and mainly in the labile fraction (Figure 4.7). After 26 months, only 13.6% of P added as sewage sludge was recovered in the three P fractions, 11.5% of which was in the labile fractions.

Figure 4.8 shows the profile distribution of P in the sewage sludge treatment. There was a slight increase in the most-labile P fraction in the surface 0-2.5 cm (Figure 4.8a). The less-labile P fraction increased significantly in the surface 0-2.5 cm before 14 months, and to about 10 cm at the 26th month (Figure 4.8b). Sludge had no effect on the moderately resistant P fraction. It was observed that some of the sludge was still on the soil surface in lumps which were removed, together with the litter, before soil sampling. It is therefore possible that a large part of the added P was still being held in the sludge on the soil surface. King and Morris (1973) and Touchton *et al.* (1976) reported that large amounts of P could be retained in sludge crusts without increasing extractable P levels in soil.

It appears that plant availability of sludge P is

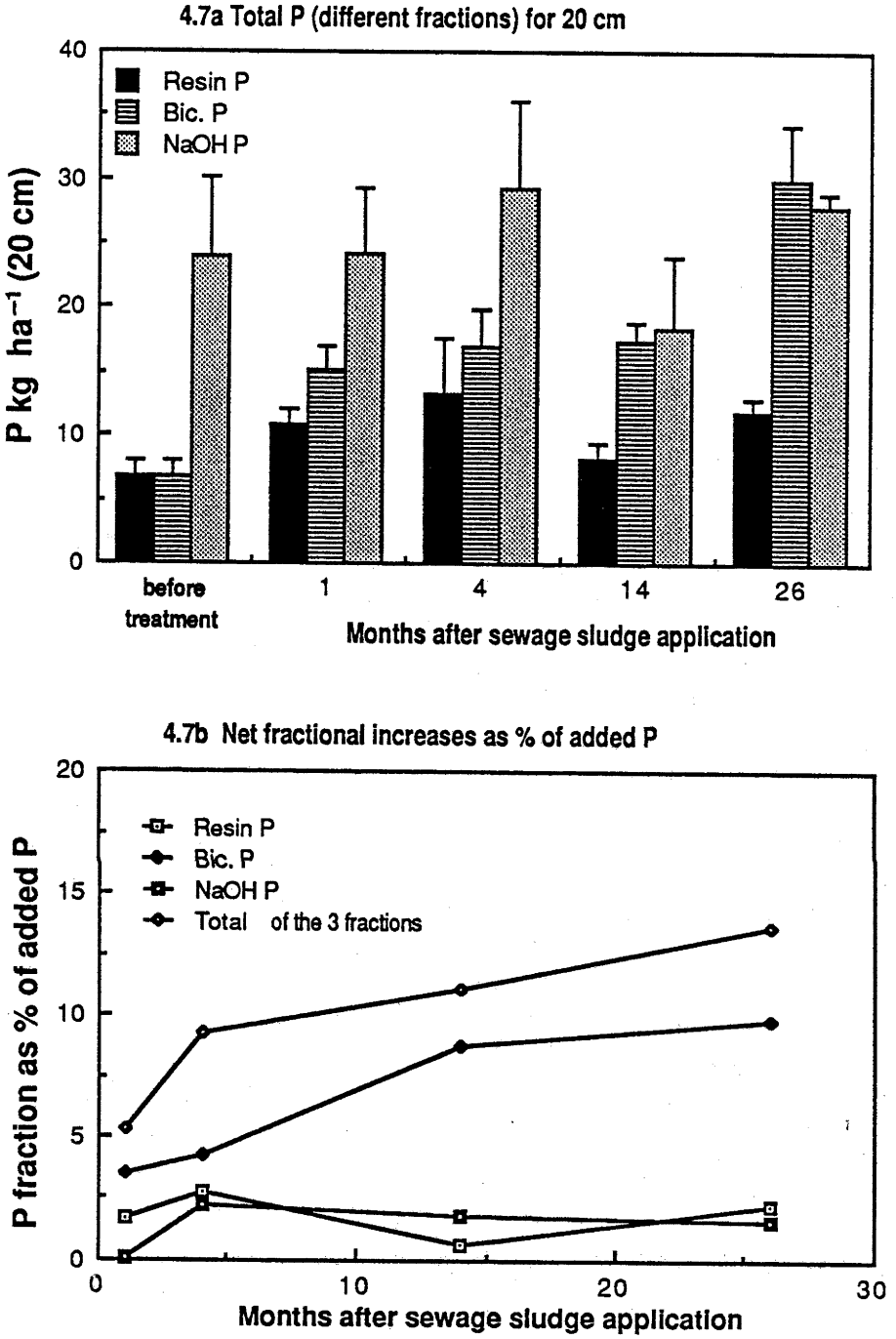


Figure 4.7 Temporal changes in the three P fractions in soils from the Sewage sludge treatment plot.

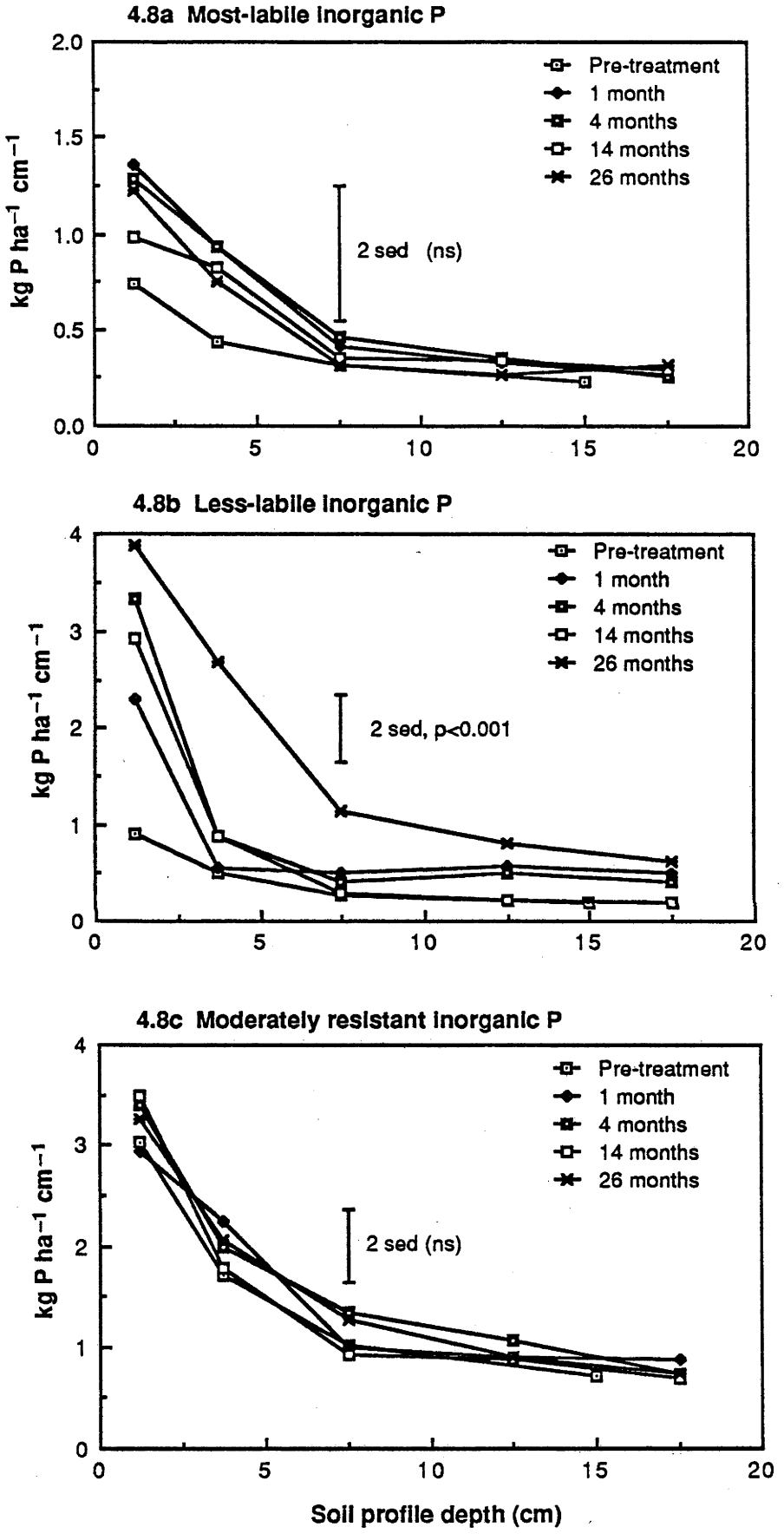


Figure 4.8 Profile distribution of P in soil after sewage sludge application

(months = months after treatment)

dependent on the form and the method of application of the sludge to soil. Willett *et al.* (1986) obtained response to P by lucerne, and an improved P status in the surface soil, using a similar sewage sludge on a P deficient soil. The sludge had been air dried, crushed to pass a 20 mm screen, and incorporated by hand cultivation into the surface 10 cm of the soil. This may have aided microbial breakdown of the sludge. Air dried and crushed sludge was used in second experiment to be reported in this Chapter. Taylor *et al.* (1978) observed that where most of the P in the sludge is in organic form, e.g. in biological sludges, slow release of P to inorganic (plant available) forms may occur. Thus the method of sewage application in this experiment may provide a slow but sustained supply of P over a long period of forest growth as it becomes progressively incorporated in the soil.*

The amount of $CaCO_3$ added in sewage sludge to soil was high (5.8 tons ha^{-1}) and was expected to change the chemical characteristics of the soil. Ca is an element which contributes to P sorption in soil, and could have led to the low amounts of labile P. Fertiliser P also changes the soil chemical characteristics, possibly affecting P sorption. It was decided to determine the extent to which the two treatments affected sorption of P by the yellow podzolic soil.

4.2.3.5 Soil P sorption as affected by fertiliser and sludge treatments

Soil P sorption was determined in soil samples at 3 depths (0-2.5, 2.5-5 and 20-30 cm) before and 26 months after treatment. The three depths were chosen to test treatment

*See note p. 80

effects down the soil profile.

Figure 4.9a shows no change in the soil's sorption capacity with time in the surface 5 cm of the control plot. The reduction in sorption at 20-30 cm was probably due to soil sample variation. Sorption was higher in surface (0-5 cm) than subsurface soils (20-30 cm) because of the higher content of P retaining soil constituents, especially Ca and organic matter (Table 3.1). This also partly explains why more P was retained in the soil surface in the field treatments.

Fertiliser application had no effect on P sorption (Figure 4.9b). The amount of fertiliser P was more than the amount of P required to bring the soil solution to 0.2 mg l^{-1}. At this concentration the growth of plants should not be limited by the supply of P (Beckwith, 1964). Barrow (1973) found reduced sorption capacity in soils with previous additions of P because the measured amount of sorption did not include the native sorbed P (Fitter and Sutton, 1975; Holford, 1977), which in this method was represented by resin extractable P.

Figure 4.9c shows that application of sludge markedly increased the sorption capacity of the surface soil (0-2.5 and 2.5-5 cm) to more than double that of soil before treatment. P sorption by the subsurface soil (20-30 cm) was not affected. Previous studies (Lee *et al.*, 1981; Soon and Bates, 1982) have shown increased P sorption on a loam soil amended with chemically treated sludges. They associated the increase in P adsorption to the presence of $CaCO_3$ and hydrous Fe and Al oxides resulting from sludge application. The sludge used in this experiment contained 43.8% $CaCO_3$ and 51.8 mg Al kg^{-1}.

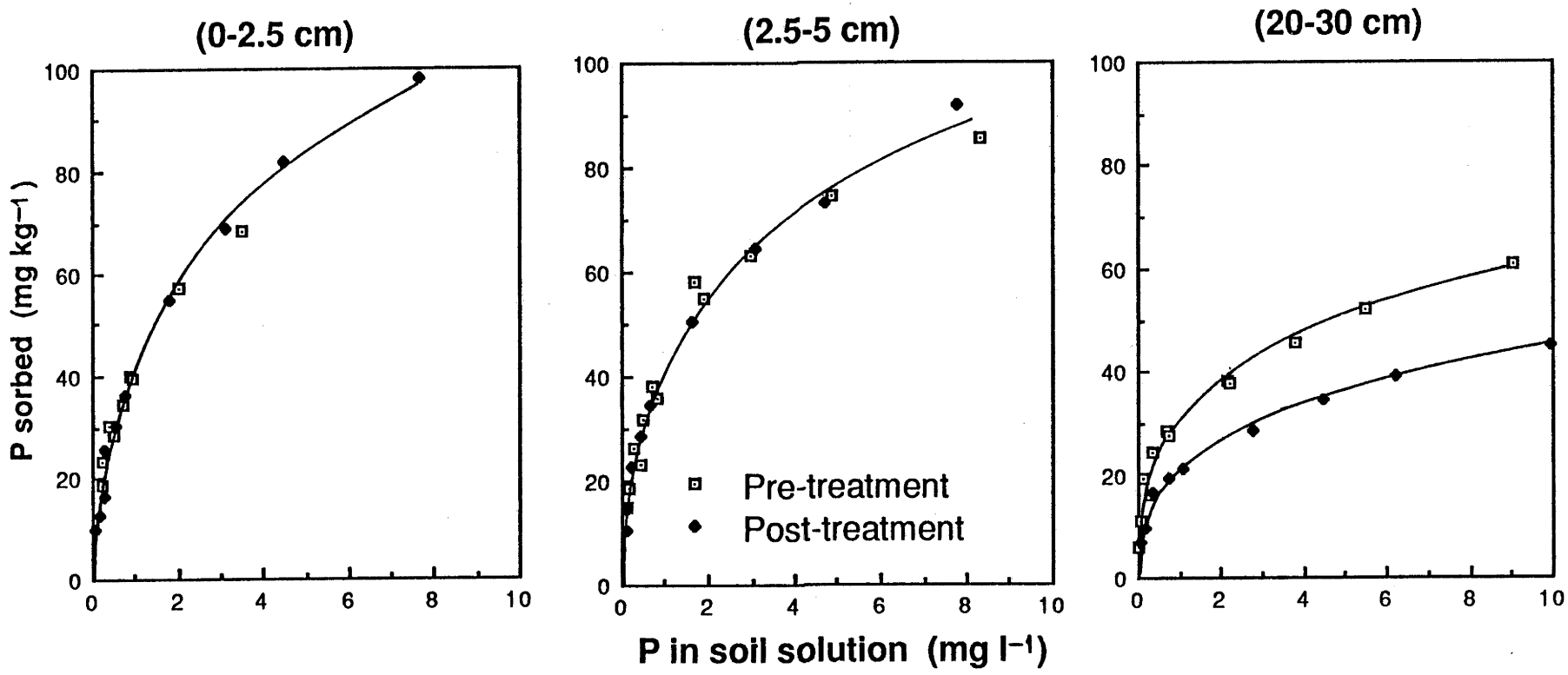


Figure 4.9a P sorption curves for soils from the Control plot before and 26 months after treatments commenced.

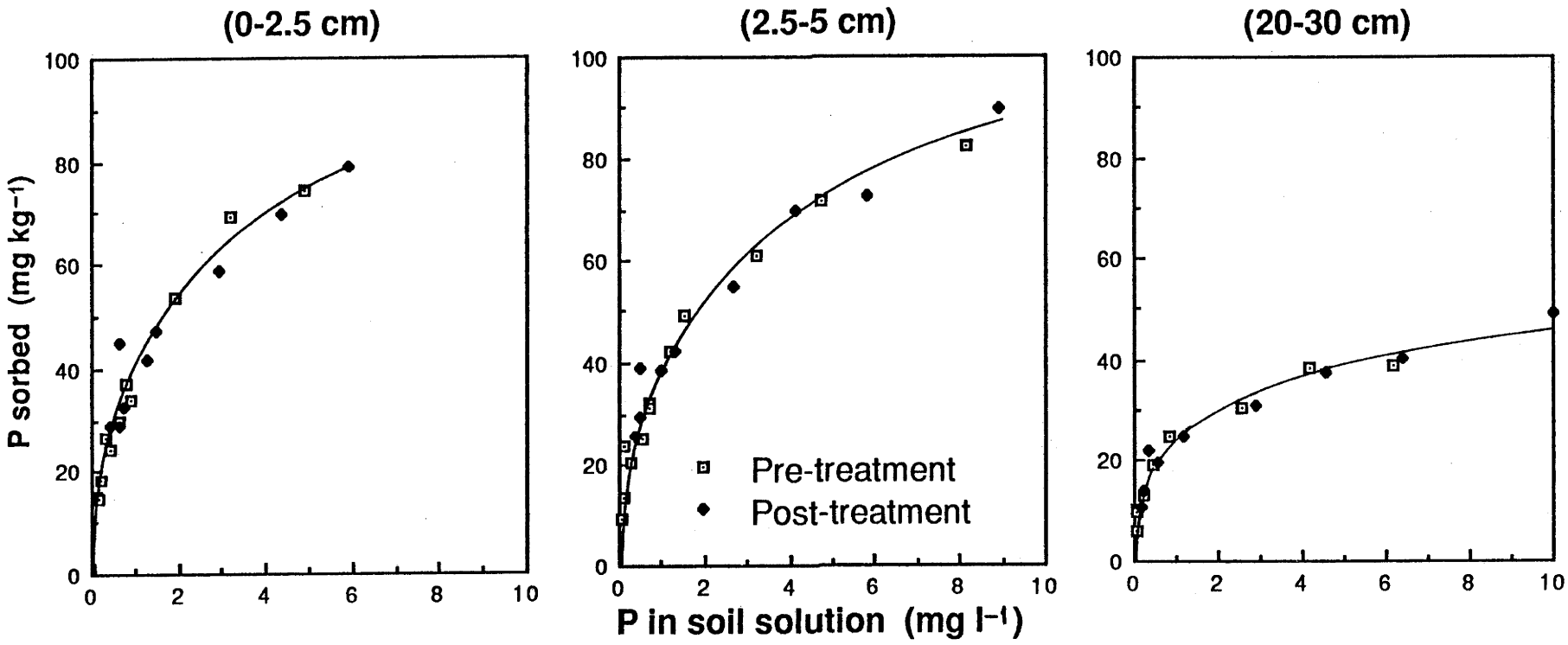


Figure 4.9b P sorption curves for soils from the Fertiliser treatment plot before and 26 months after treatment.

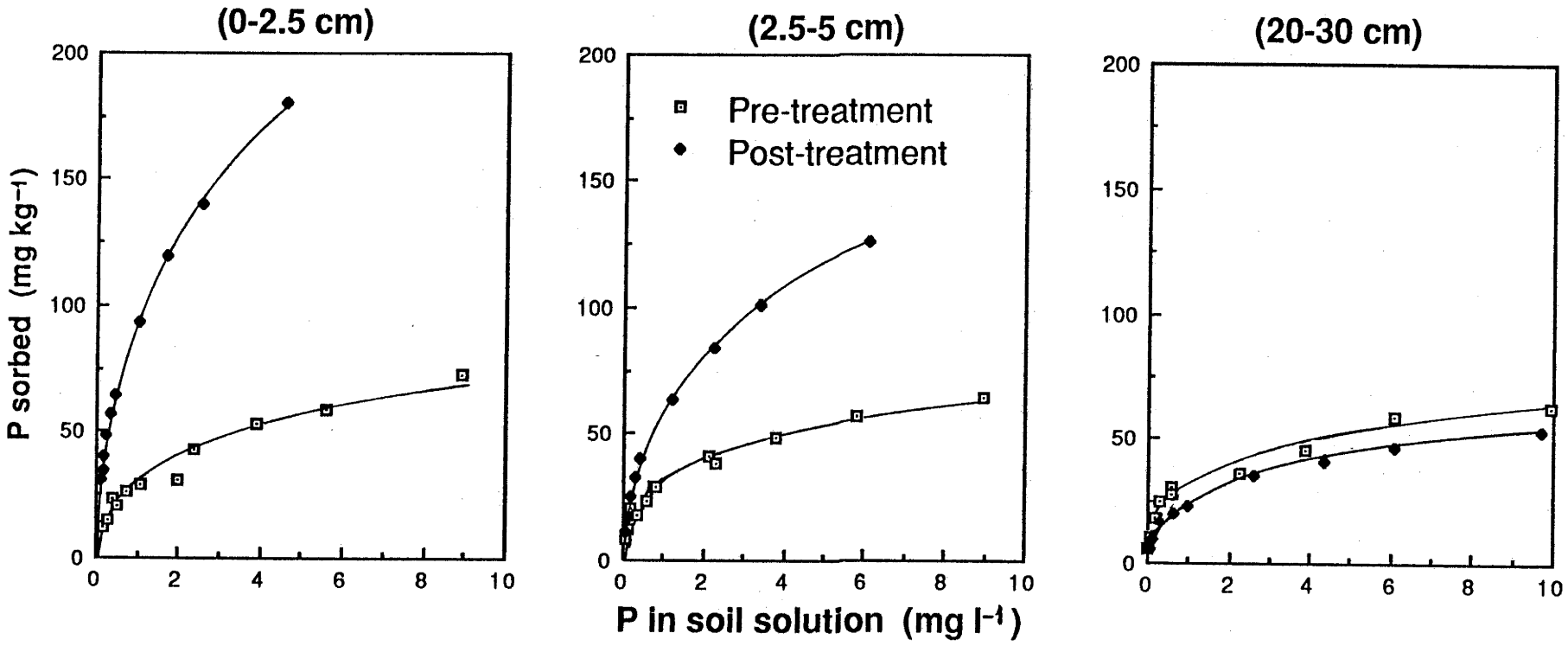


Figure 4.9c P sorption curves for soils from the Sewage sludge treatment plot before and 26 months after treatment.

Fe was not determined. The increase in P sorption may be responsible for the accumulation of sludge derived P only in the surface soil and, therefore, useful against leaching losses of P in soils with low sorbing ability.

4.2.4 Conclusions on Experiment 1

The rate of P used in this experiment was high compared to the 50-100 kg P ha^{-1} commonly used in broadcast fertilisation at establishment and to established stands (Ballard, 1980). However, the yellow podzolic soil showed the ability to conserve a large fraction of the applied P (about 35% over 2 years) in the most-labile and less-labile P pools. These fractions could remain for long periods as the rate of decline of the labile fraction was low. Thus the large labile P in soil from the fertiliser treatment, together with the efficient use of applied P through recycling, give a basis for suggesting that a single P broadcast application of fertiliser P (200 kg P ha^{-1}) in the early growth phase is suitable for pine growth on the yellow podzolic soil experimented with.

There was some movement of P in the soil profile, but this seemed only enough to cause profile distribution of P within the root zone. Only a very small fraction, if any, could have been lost from the root zone.

Sewage sludge applied in wet form was not an immediate source of labile P in soils, but showed trends of a long term effect. Its effect on sorption was evident in the 2.5-5 cm zone of the soil profile, implying that the chemical

components involved in sorption had been washed out of the sludge lumps into the soil, while the amount of P released was small. It is possible that the mechanisms of releasing nutrient ions from sludge are different, and need to be examined.

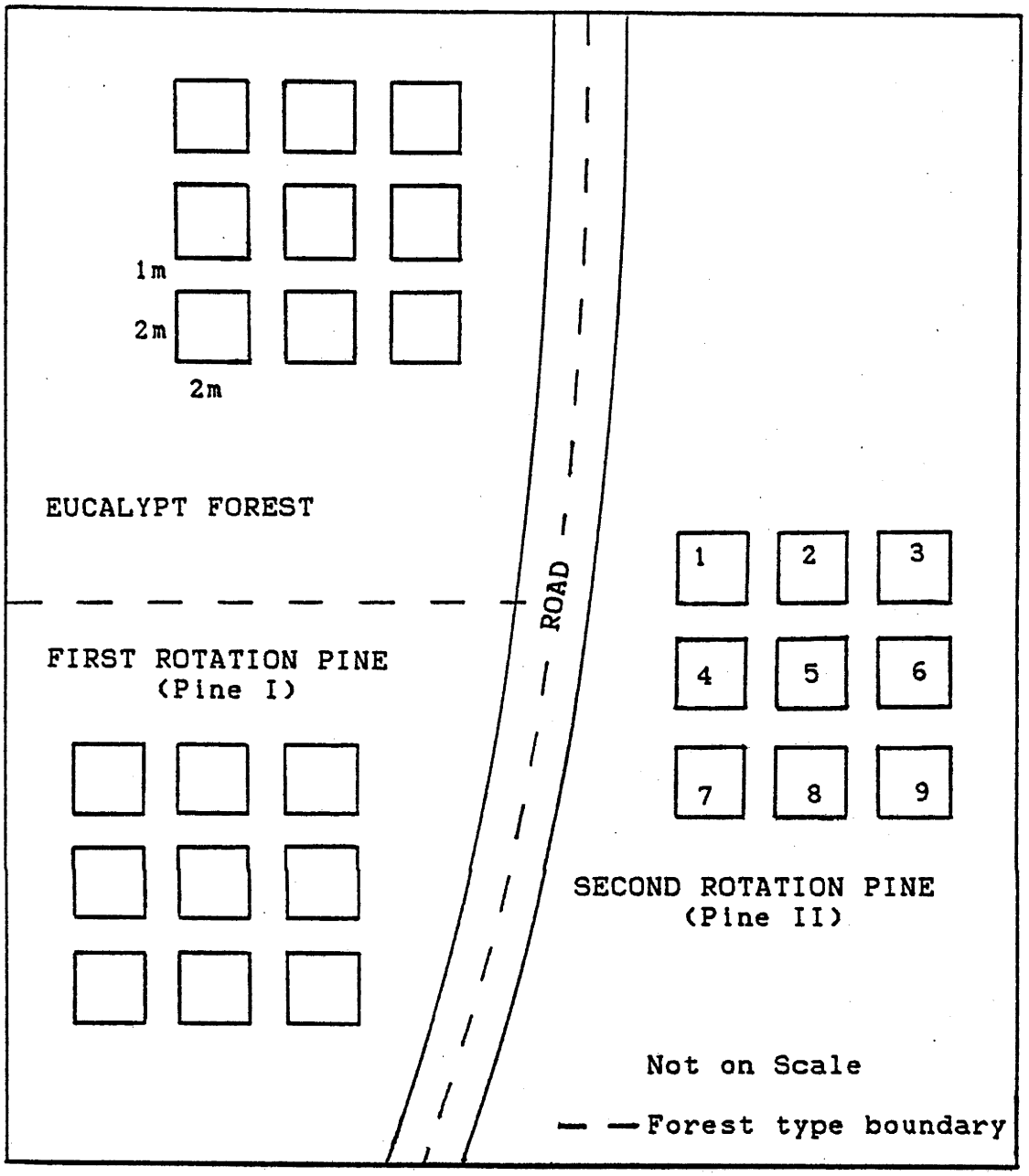
Fertilising pine plantations at planting, rather than later age fertilisation, is the widespread practice in Australian forestry (Crane, 1981; Nambiar and Cellier, 1985). This apparently starts the plantation off at an optimum growth rate and free of obvious deficiencies (Woods, 1976). Late age fertilisation experiments have provided evidence of lowered P requirement with age (Brockwell and Ludbrook, 1962; Waring, 1980) and delayed, low and variable response (Flinn *et al.*, 1979). The fate of fertiliser P in soil was not studied in the late age experiments, and could partly explain some of the differences observed. This was tested in the following experiment by monitoring changes in soil P after surface application of fertiliser P to forest floors with forest vegetations of different ages. In addition, a sewage sludge treatment was included in which air dried and ground sewage sludge was used, so as to test its effectiveness as a source of soil P compared with wet sludge used in experiment one.

4.3 Experiment 2. Effect of different soil-litter and vegetation situations on surface applied fertiliser and sewage sludge phosphorus

4.3.1 Experimental procedures

This experiment was based on a small area in which natural dry sclerophyll eucalypt forest, 49 yr old first rotation radiata pine stand and 5 yr old second rotation pine forest were growing in close proximity on a yellow podzolic soil (Figure 3.1, Site 2). These presented different litter and vegetation situations in which the movement and chemical transformation of P could be further tested. The eucalypt and first rotation pine stands represented 'later age' situations, and the second rotation pine stand represented an 'early age' fertilisation situation. The eucalypt floor consisted of scanty material composed mainly of bark, twigs, branches and a few leaves and capsules. The floor of the first rotation pine stand supported mainly a uniform litter layer with litter at all stages of decomposition, and an average thickness of about 8 cm. The floor of the second rotation pine was still free of litter.

Under each vegetation type (site), an orthogonal latin square experiment was established, consisting of nine 2m * 2m plots, with three replicates of control, phosphate fertiliser and sewage sludge treatments. The layout of the experimental plots is shown in Figure 4.10. The experimental sites were sprayed with a herbicide (Roundup) to stop P uptake by the undergrowth. Single superphosphate and air dried and ground lime-treated sewage sludge were carefully spread by hand to the undisturbed forest floors at a rate of 160 kg P ha^{-1} on



Control treatment in replicates 1, 5 and 9
Fertiliser treatment in replicates 2, 6 and 7
Sewage sludge treatment in replicates 3, 4 and 8

Plot set up was the same for the three forest sites.

Figure 4.10 Layout of the experimental plots for the study of soil-litter and vegetation situations on soil P.

21 September, 1984. The chemical composition of the sewage sludge was similar to that used in Experiment one (Section 4.2.1).

Soil samples were taken before and 1, 4 and 10 months after application of the treatments. A 2.8 cm diameter corer was used to take a soil sample down to 50 cm depth, and this was subdivided into 0-2.5, 2.5-5, 5-10, 10-20, 20-30, 30-40 and 40-50 cm depth segments. Three cores were taken from each plot and the depth segments from each were bulked. Soils were air dried, ground, sieved through 2.0 mm and stored in closed plastic bags until analysis. The soils were analysed for:

1. pH (soil:water ratio of 1:5)
2. Resin extractable P (most-labile inorganic P)
3. $NaHCO_3$ extractable inorganic and organic P
   (less-labile inorganic and labile organic P fractions)
4. NaOH extractable inorganic and organic P
   (Moderately resistant inorganic and organic P fractions)
5. Kjeldahl digested P (residual fraction)

All the methods of analysis have been described in Chapter 3. The individual fractions were determined by difference as described in Section 3.3.1. *

4.3.2 Results and discussion

4.3.2.1 Examination of P in soils under the three sites

The three sites of the experiment are denoted as follows: eucalypt; Pine I (first rotation radiata pine); and Pine II (second rotation radiata pine).

* Data were analysed by Anova to test for treatment, time and depth interactions

The initial site differences in profile distribution of the different P fractions are shown in Figure 4.11. Apart from the residual P fraction, all other P fractions showed depth effects, that is there was more P in the surface 15 cm, and the amount declined with depth (Figures 4.11a to 4.11e). Alternatively, residual P declined from the surface to 15 cm, and then increased with depth (Figure 4.11f).

Significant effects ($p<0.001$), of depth-site interaction were mainly in the surface of the soil profiles (0-5 cm), and are summarised as follows:

Most-labile P	Pine I > Eucalypt > Pine II
Less-labile inorganic P	Pine I > Eucalypt > Pine II
Labile organic P	Eucalypt > Pine I = Pine II
Moderately resistant inorganic P	Pine I > Eucalypt > Pine II
Moderately resistant organic P	Pine II > Pine I = Eucalypt
Residual P (0-5 cm)	Eucalypt = Pine I > Pine II
Residual P (40-50 cm)	Pine I = Pine II > Eucalypt

For the most-labile P and moderately resistant organic P fractions, significant differences also occurred at depths of 5-15 cm in the same order as for the surface 0-5 cm. The three sites were therefore not comparable and the differences between the three sites were not consistent for the different P fractions. Pine II site tended to have the least P in the surface soil, except for moderately resistant organic P where it was highest.

Differences were also observed in the soil profile pH (Figure 4.12a) and bulk density. pH in soils from the Pine II

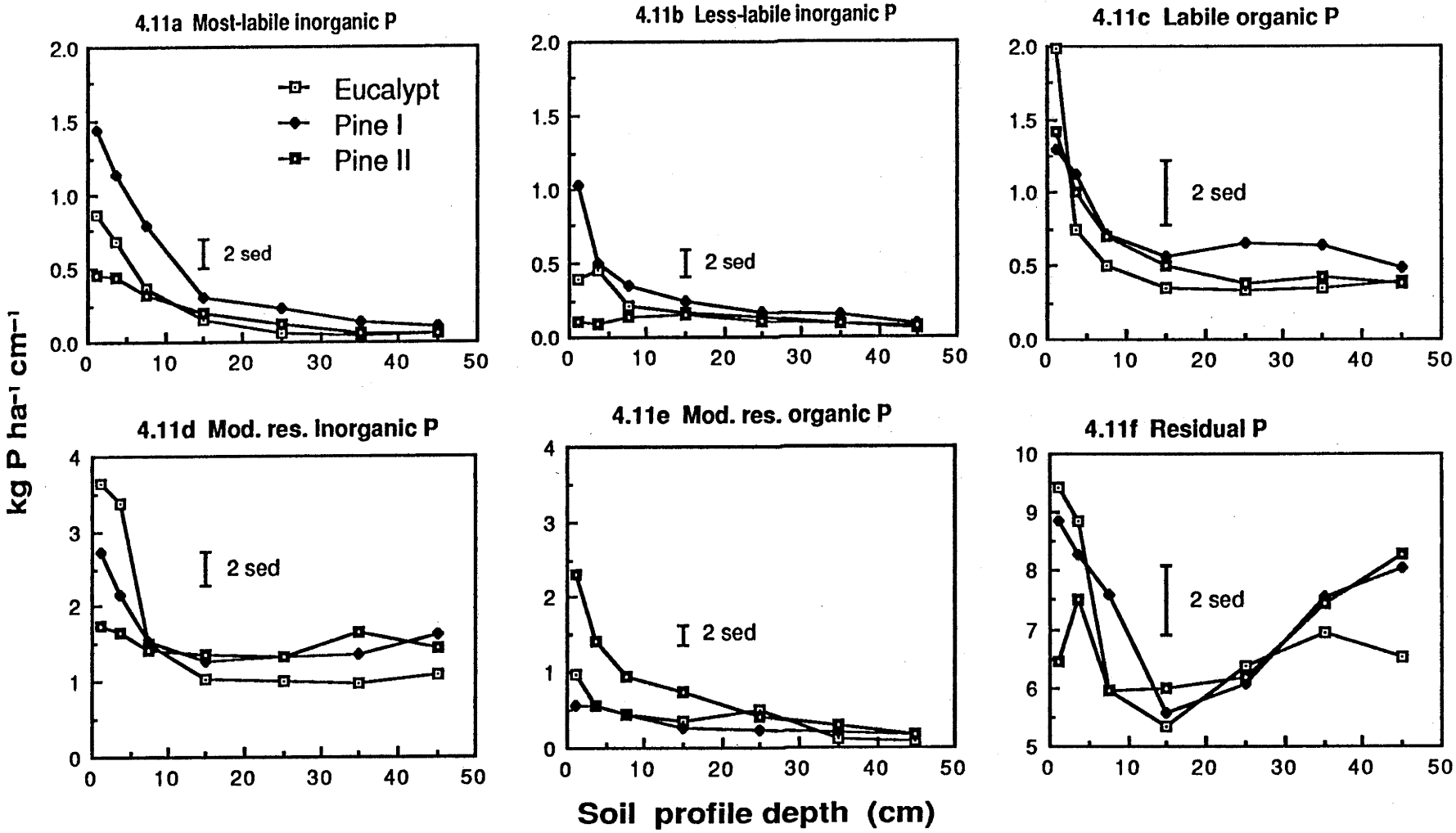


Figure 4.11 Site differences in soil P before treatments were applied.

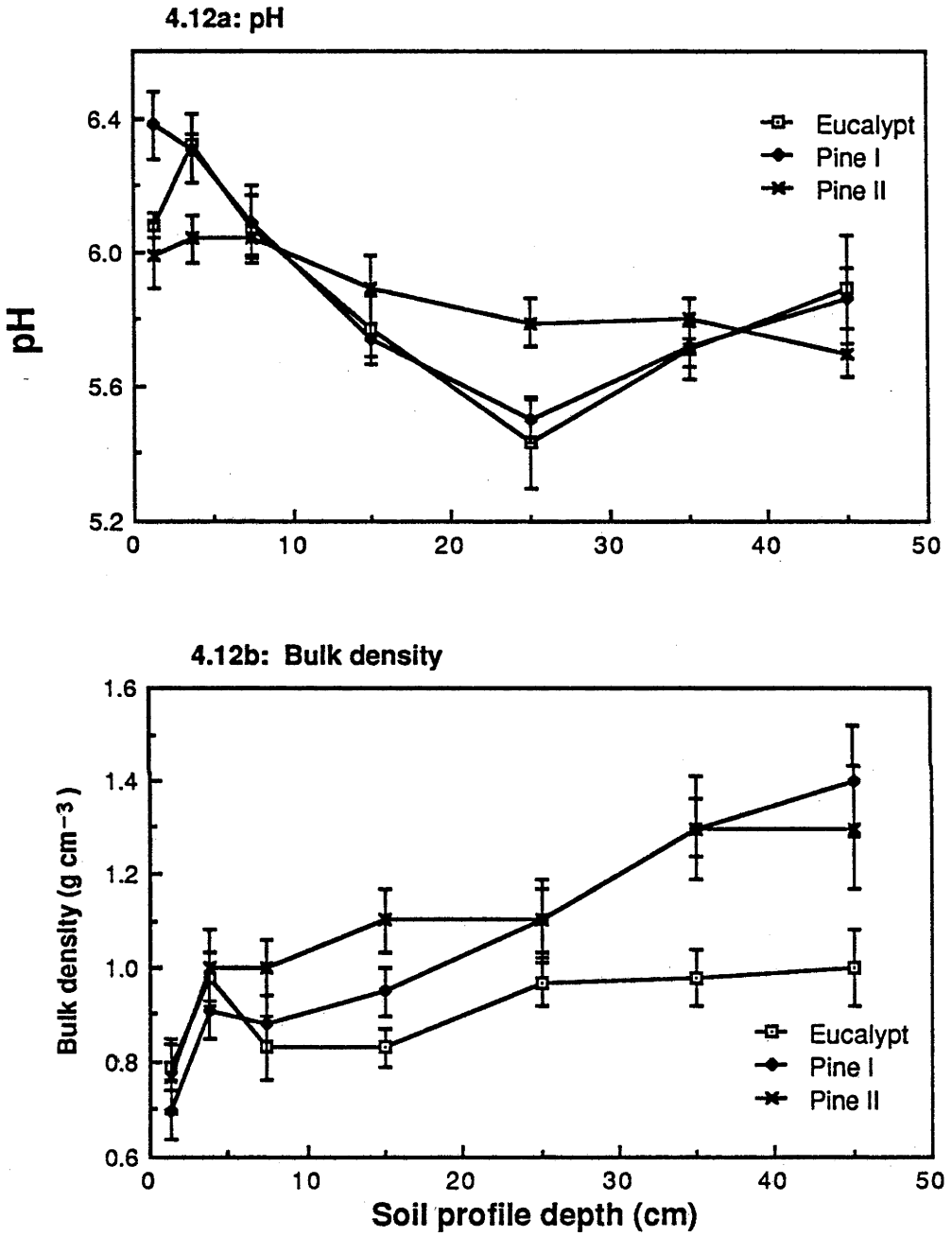


Figure 4.12 Site differences in soil pH (4.12a) and bulk density (4.12b) before treatments were applied.

site was slightly lower in the surface soil (0-5 cm) compared with that in soil from the eucalypt and Pine I sites. At 15-35 cm, pH of soil from Pine II was markedly higher (by about 0.5 pH units at 25 cm) than that of soils from the other sites. Figure 4.12b shows that soils from the eucalypt site had lower bulk density values below 10 cm than soils from the other sites. Bulk density was less in soils from Pine I than that in soils from Pine II only at 5 to 25 cm. Because of these differences, the site effects of adding fertiliser and sewage sludge P have been analysed individually.

For a given site, illustrations of the different P fractions are expressed on the same scale for both fertiliser and sewage sludge treatments for ease of comparison.

4.3.2.2 P distribution in soil on the eucalypt site

Table 4.3 shows temporal changes in different soil P fractions (total, to a depth of 50 cm) on the eucalypt site. The different P fractions in the control treatment were relatively stable with time, and the treatment-time interactions were mainly the result of fertiliser and sewage sludge additions. Total P in the control treatment soil was about 550 kg P ha^{-1}, most of it being in the residual fraction (about 65%). The most-labile, less-labile inorganic and labile organic P fractions formed about 12 % and the rest was in the moderately resistant fractions.

In the fertiliser treatment, most of the added P was in the most-labile and less-labile inorganic P fractions. Most-labile P increased from 15 (pre-treatment) to 99 kg P

ha^{-1} after 10 months, while less-labile inorganic P increased from 10 to 79 kg P ha^{-1}. Both of the labile fractions therefore contained more than 90% of the added P at the 10th month. The amount of P recovered in these fractions at 1 month after treatment was higher than the added P, possibly because of errors resulting from incomplete mixing of the fertiliser with soil. The moderately resistant inorganic P increased slowly with time. The organic fractions and the residual P were less affected by the fertiliser treatment.

Table 4.3 Comparison of changes in the different P fractions on the eucalypt site

Treatment	P fraction	Time after treatment (months)				treatment-time interaction	
		0	1	4	10	sed	p<
		kg P ha^{-1} (50 cm)					
Control	M. labile IP	18	20	17	18	13	0.001
	L. labile IP	15	19	16	19	15	0.01
	Labile OP	33	30	33	32	5	ns
	Mod. res. IP	103	119	104	87	13	ns
	Mod. res. OP	19	17	20	22	7	ns
	Residual P	355	354	376	351	32	ns
	Total	541	558	556	528		
Fertiliser	M. labile IP	15	141	107	99	As above	
	L. labile IP	10	61	65	79		
	Labile OP	36	36	34	32		
	Mod. res. IP	81	90	96	105		
	Mod. res. OP	12	14	14	16		
	Residual P	359	364	339	354		
	Total	512	706	655	685		
Sewage sludge	M. labile IP	16	65	63	56	As above	
	L. labile IP	9	42	75	89		
	Labile OP	33	35	33	30		
	Mod. res. IP	88	84	85	88		
	Mod. res. OP	35	44	44	47		
	Residual P	347	389	394	405		
	Total	526	658	693	718		

M. labile = most labile; L. labile = less labile
IP = inorganic P; OP = organic P
Mod. res. = moderately resistant

* Sludge was not analysed but was presumed from reactions in soil (Table 4.3) to have a large inorganic component. There may have been a much larger component of organic P in the wet sludge (see also addition p. 71 & p. 105). The process of air drying and grinding may have killed microorganisms in the sludge releasing P in inorganic form.

Figure 4.13 shows the profile distribution of the different P fractions with time in the fertiliser treatment on the eucalypt site. The significant increase in the most-labile and less-labile inorganic P fractions (Figures 4.13a and b) was in the surface 5 cm, but the graphs also show that P moved deeper (> 20cm) in the profile. This is consistent with the results of Experiment one which also showed profile movement of P. The increase in moderately resistant P (Figure 4.13d) was slight and only in the surface 0-2.5 cm. None of the other fractions show significant fertiliser effects on the profile.

In the sewage sludge treatment, most of the increases were also in the most-labile and less-labile inorganic P fractions (Table 4.3), and increased with time for the less-labile fraction. Both of the labile fractions formed about 75% of the P added in sewage sludge, and the other 25% of the added P appears to be in the residual fraction. There was little effect of sewage sludge treatment on labile organic P and moderately resistant inorganic and organic P. These results suggest that most of the P in air dried and ground sewage sludge is in inorganic form and only a small fraction is in the resistant organic fraction. According to Sommers *et al.* (1976) most of the P in sewage sludges is inorganic in form, and this may range from 70-90% of the total sewage sludge P.*

The profile distribution of the various P fractions in the sewage sludge treatment is presented in Figure 4.14, and

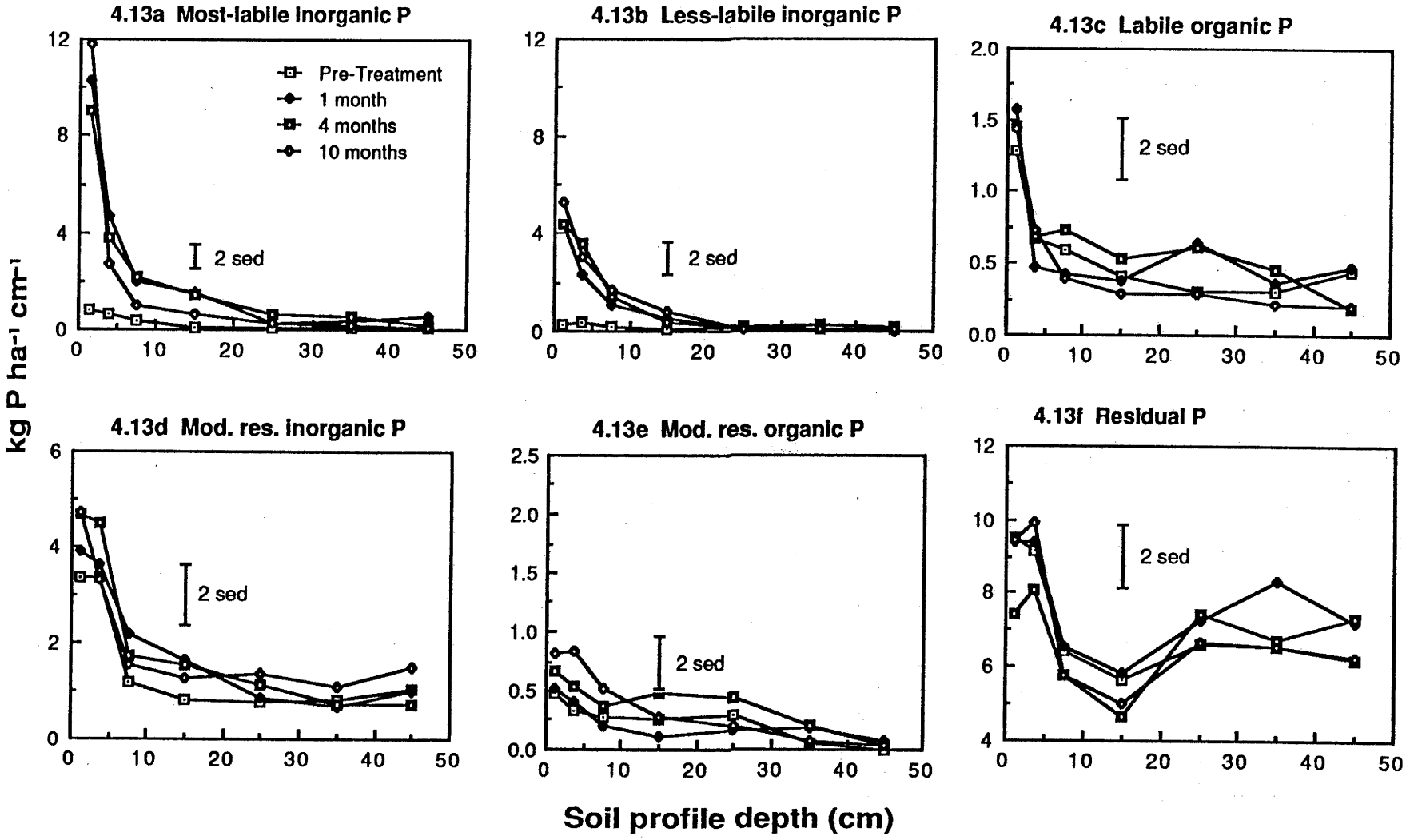


Figure 4.13 Profile P distribution in soil of the Fertiliser treatment on the Eucalypt site. (sed for treatment*time*depth interaction).

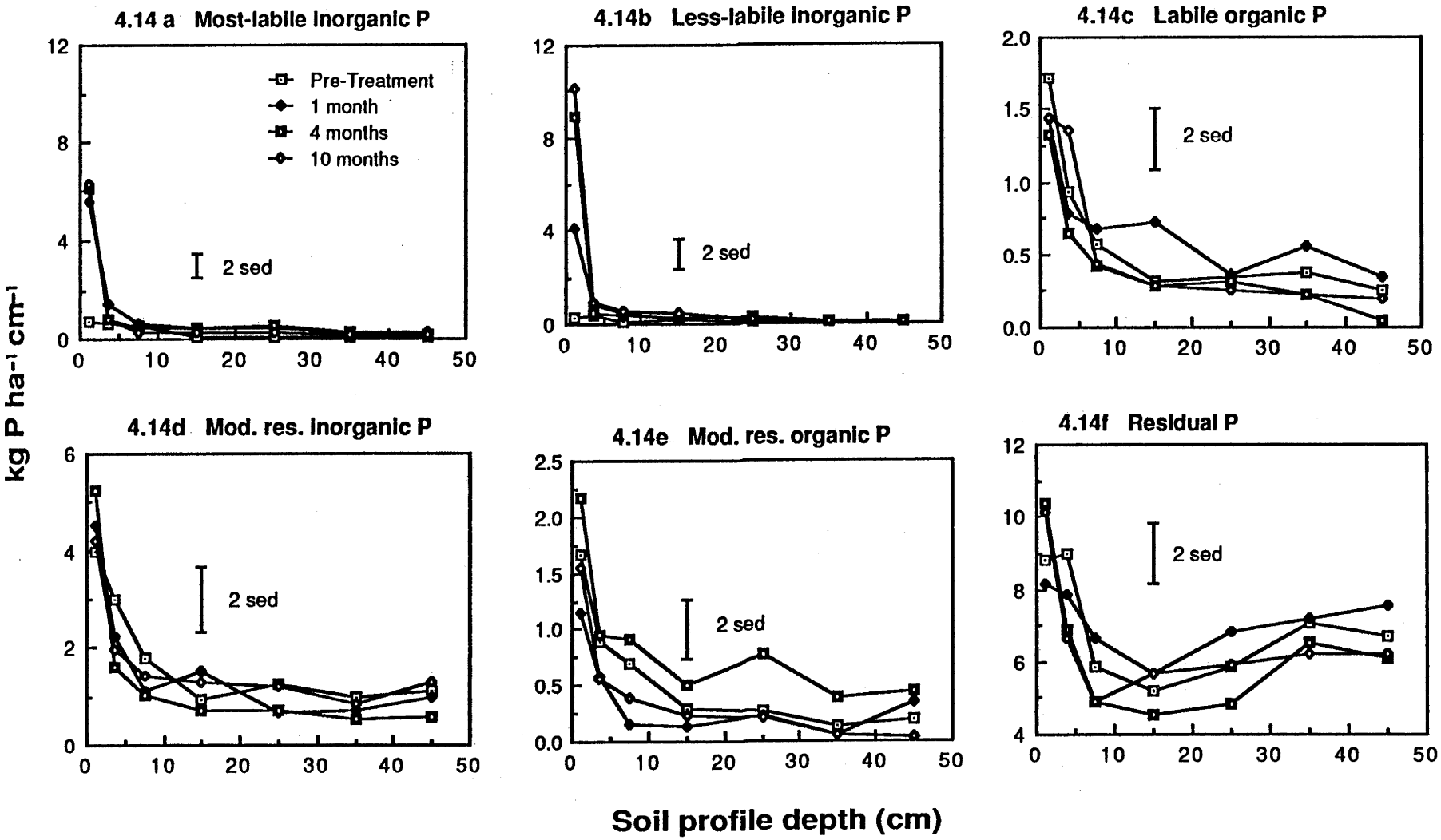


Figure 4.14 Profile P distribution in soil of the Sewage sludge treatment on the Eucalypt site. (sed for treatment*time*depth interaction).

shows that the increase in the most-labile and less-labile inorganic P fractions was only in the soil surface (0-2.5 cm). This leads to two suggestions. Firstly, the results may reflect direct measurements of the sludge still lying on the soil surface; the sludge was in powder form and could not be removed before soil sampling, and therefore formed part of the 0-2.5 cm soil samples. Secondly, P may have been retained in the surface due to increased sorption. Results in Section 4.2.2.5 showed that sewage sludge application increased the soils's P sorption capacity. With the increases in the most-labile and less-labile inorganic P fractions being only in the soil surface (0-2.5 cm), the tree must be capable of having roots close to the soil surface so as to utilise P from sewage sludge.

4.3.2.3 P distribution in soil on the first rotation pine site (Pine I)

Table 4.4 shows changes in the total amount of the different P fractions to a depth of 50 cm with time. There were large sampling differences in the control treatment, and therefore any significant effects observed may be due to both sampling and treatment effects. In the control treatment, there were about 520 kg P ha^{-1} (total), 70% of which was in the residual fraction. About 14% was contained in the labile fractions, and the remaining 16% was in the moderately resistant fractions.

In the fertiliser treatment, most-labile P increased from 35 kg P ha^{-1} before treatment, stabilising at about double this level. Less-labile inorganic P also increased from about

22 kg P ha^{-1} to about double this quantity. Together, the labile fractions represented about 50% of the initial added P at 10 months. The other P fractions were less affected by fertiliser addition. Significant differences in the moderately resistant inorganic and organic P fractions appear to be resulting from temporal and spatial variations.

Table 4.4 Comparison of changes in the different P fractions on the first rotation pine site

Treatment	P fraction	Time after treatment (months)				treatmet-time interaction	
		0	1	4	10	sed	p<
		kg P ha^{-1} (50 cm)					
Control	M. labile IP	30	25	16	21	15	0.05
	L. labile IP	16	26	10	23	11	ns
	Labile OP	40	35	25	26	5	ns
	Mod. res. IP	76	53	55	68	10	0.05
	Mod. res. OP	17	16	41	17	4	0.01
	Residual P	374	368	371	354	35	ns
	Total	553	523	518	509		
Fertiliser	M. labile IP	35	85	70	90	As above	
	L. labile IP	22	36	57	45		
	Labile OP	39	43	28	26		
	Mod. res. IP	94	103	102	103		
	Mod. res. OP	18	20	24	18		
	Residual P	391	364	363	357		
	Total	598	651	644	639		
Sewage sludge	M. labile IP	25	27	25	20	As above	
	L. labile IP	18	19	17	22		
	Labile OP	39	39	26	24		
	Mod. res. IP	92	69	66	82		
	Mod. res. OP	18	25	30	15		
	Residual P	342	363	377	365		
	Total	534	541	541	528		

Figure 4.15 shows the profile distribution of P in the fertiliser treatmentat the Pine I site. The increases were only in the most-labile and less-labile inorganic P fractions, and were confined to the surface 10 cm. The other P fractions

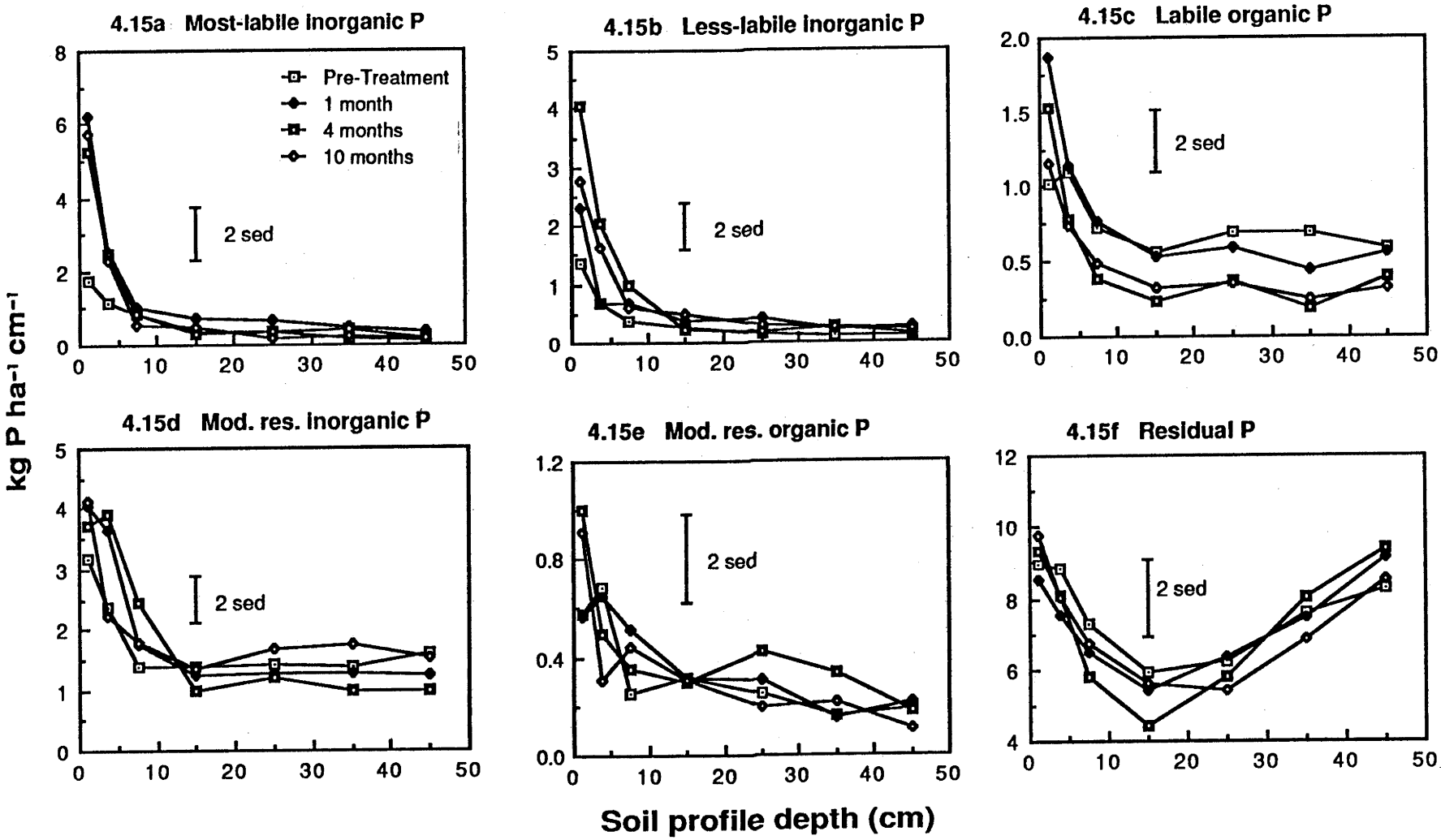


Figure 4.15 Profile P distribution in soil of the Fertiliser treatment on the Pine I site. (sed for treatment*time*depth interaction).

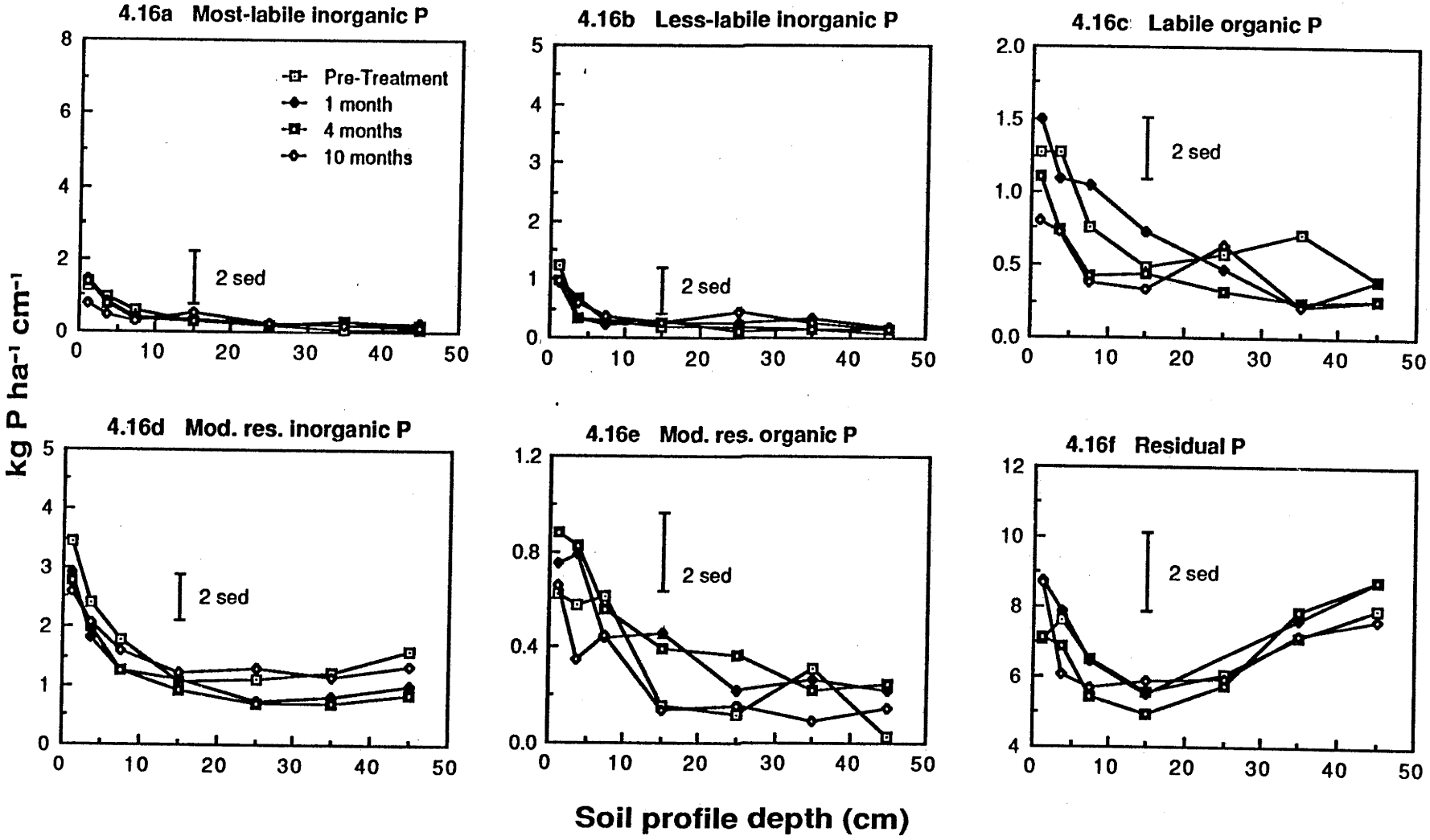


Figure 4.16 Profile P distribution in soils from the Sewage sludge treatment on the Pine I site. (sed for treatment*time*depth interaction)

were not influenced by fertiliser addition. Some degree of variation is evident throughout the profile distribution of various P fractions.

Sewage sludge application to soil on the Pine II site had no effect on soil P (Table 4.4). Accordingly, Figure 4.16 does not show any variation with depth in different P fractions resulting from this treatment.

The lack of influence of sewage sludge on soil P, together with the low amount of fertiliser P recovered in the fertiliser treatment (compared with the eucalypt site), suggests that part of the fertiliser and all the sewage sludge applied to the first rotation pine site may have been physically retained by the litter on the forest floor. Alternatively, the P released may have been chemically held by the litter. In view of this, the ability of litter to sorb P is examined in the following section.

4.3.2.4 Phosphorus sorption by radiata pine litter

Measurement of P sorption by litter was based on the method used for soils, described in Section 3.3.7, using litter from the first rotation pine site. Attempts were made to use homogeneous samples of all litter fractions without any form of grinding so as to minimise the effects of physical and chemical disturbance. Each analysis was replicated 4 times. The sorption curve was constructed as for soils and is shown in Figure 4.17. There were large variations, as shown by the standard errors of mean values, due to the difficulty in obtaining uniform representative samples from all grades of

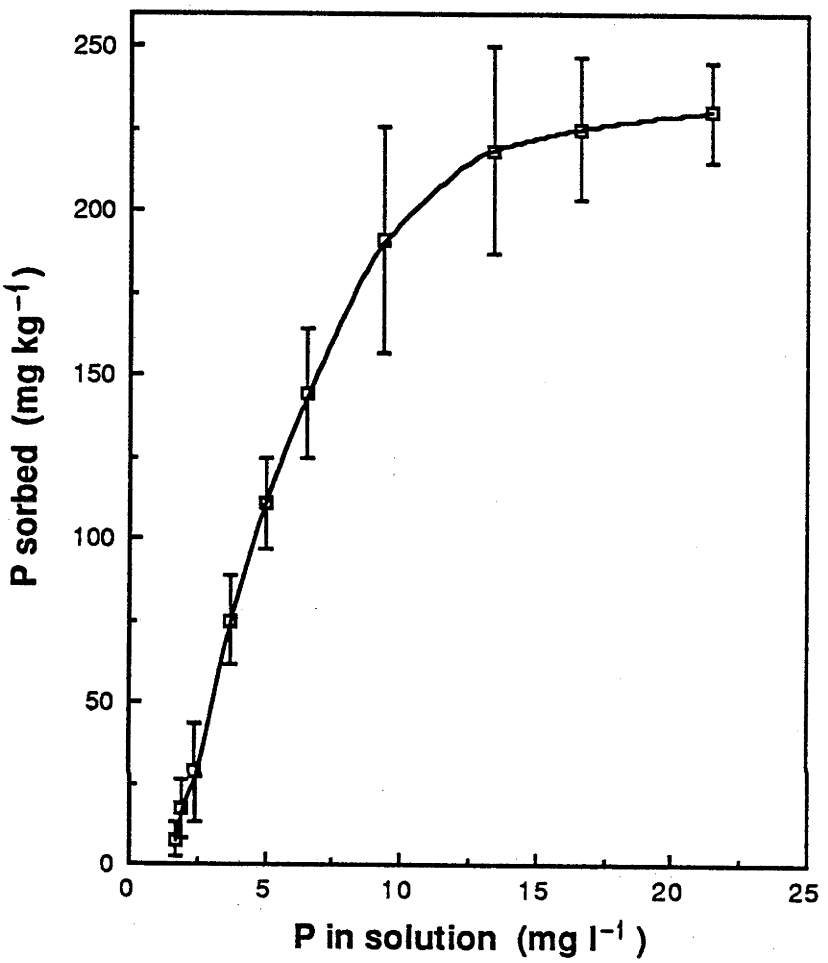


Figure 4.17 P sorption curve for radiata pine litter.

(se of means shown as vertical bars; n=4)

litter.

Figure 4.17 shows that the litter was able to chemically retain some P. However, the maximum P sorbed was only about 250 mg kg^{-1} (0.25 kg P ton^{-1}), and would represent only a small fraction of the fertiliser P retained in the field experiment. It is therefore concluded that most of the fertiliser and sewage sludge was physically retained by the litter. In late forest fertilisation practices, it may be necessary to leach the fertiliser through the litter (e.g. with irrigation) or to apply fertiliser in liquid form. It is likely that fertiliser P added to the eucalypt soil would have moved down the profile in rain water, as sparce cover of litter would have allowed fertiliser to be in immediate contact with the soil. Litter has been observed to intercept 5-9% of incident precipitation (Feller, 1981). This would reduce the amount of rain water available to leach fertiliser P to the soil. Again, canopy interception by radiata pine can be as much as 10-11% greater than in the eucalypt forest (Feller, 1978, 1980), further reducing the amount of rain water available to leach fertiliser from litter. Thirdly, the canopy also affects distribution of the throughfall and stemflow, which would cause uneven leaching of the fertiliser through the litter. These factors could lead to less P reaching the soil, and consequently to less response by trees to fertiliser P under similar conditions.

4.3.2.5 P distribution in soil on the second rotation pine site (Pine II)

Table 4.5 shows changes with time in the total of

different P fractions on the Pine II site to a depth of 50 cm. The treatment-time effects were similar to those on the eucalypt site, reflecting largely the influence of fertiliser and sewage sludge treatments. The most-labile, less-labile and moderately resistant inorganic P were the fractions affected most. In the control treatment, total soil P was about 520 kg P ha^{-1}, with about 68% in the residual fraction; 10% was contained in the labile P fractions and the rest was in the moderately resistant fractions.

Table 4.5 Comparison of changes in different P fractions on the second rotation pine site

Treatment	P fraction	Time after treatment (months)				treatment-time interaction	
		0	1	4	10	sed	p<
		kg P ha^{-1} (50 cm)					
Control	M. labile IP	12	13	15	18	9	0.001
	L. labile IP	6	5	5	8	12	0.001
	Labile OP	32	27	30	32	5	ns
	Mod. res. IP	74	73	72	67	14	0.05
	Mod. res. OP	46	38	40	45	10	ns
	Residual P	357	361	365	346	37	ns
	Total	527	517	527	516		
Fertiliser	M. labile IP	11	90	62	63	As above	
	L. labile IP	6	54	82	92		
	Labile OP	38	33	32	35		
	Mod. res. IP	76	107	119	129		
	Mod. res. OP	39	35	39	30		
	Residual P	341	345	358	354		
	Total	511	664	692	703		
Sewage sludge	M. labile IP	14	56	48	43	As above	
	L. labile IP	5	29	37	49		
	Labile OP	34	37	38	34		
	Mod. res. IP	80	76	89	96		
	Mod. res. OP	39	35	33	33		
	Residual P	327	390	373	399		
	Total	499	623	618	653		

In the fertiliser treatment, about 86% of the P added as fertiliser to the soil on the Pine II site was retained in the most-labile and less-labile inorganic P fractions at the 10th month, most of it (54%) being in the less-labile inorganic P fraction. The most-labile fraction peaked and then declined after one month, while the less-labile inorganic P increased with time from 6 kg P ha^{-1} (pre-treatment) to 92 kg P ha^{-1}. The moderately resistant inorganic P also increased with time, while the organic and residual P fractions were not affected by fertiliser addition.

Figure 4.18 shows the profile distribution of the P fractions in soil of the fertiliser treatment on the Pine II site. There were significant profile increases for the most-labile, less-labile inorganic and moderately resistant inorganic P fractions to about 10 cm by the 10th month. There were only slight increases to about 25 cm. Residual P also increased in the surface 5 cm whereas the organic fractions were less affected by fertilisation.

Sewage sludge treatment increased the most-labile, less-labile inorganic and residual P fractions (Table 4.5). After 10 months, about 46% of the added P in sewage sludge was in the most-labile and less-labile inorganic P fractions, while the rest appears to have been in the residual fraction. The labile organic and the moderately resistant inorganic and organic P fractions were less affected by the sewage sludge treatment.

The profile distribution of the different P fractions in the sewage sludge treatment are shown in Figure 4.19. The increase in P in the most-labile, less-labile inorganic and

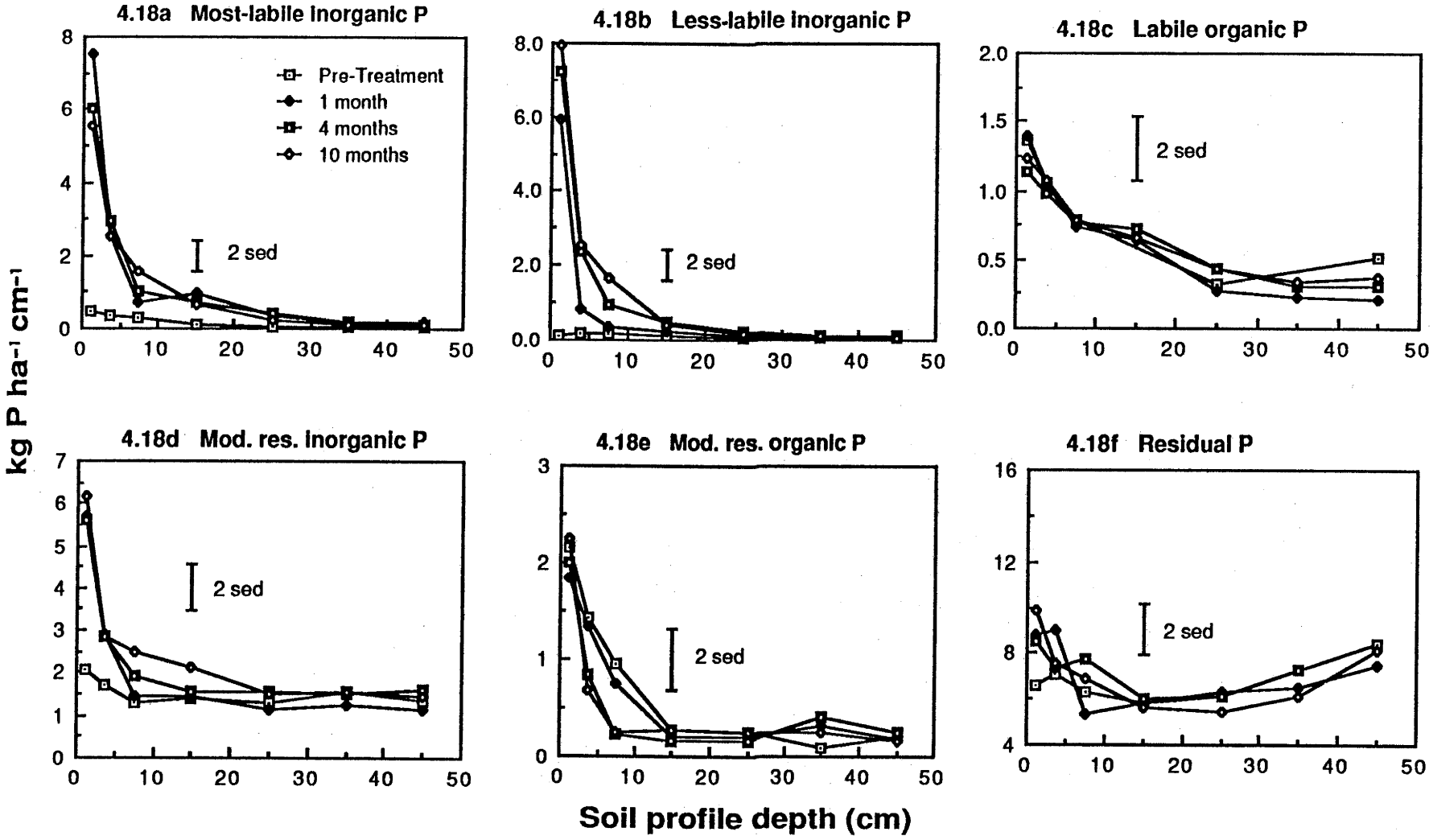


Figure 4.18 Profile P distribution in soils of the Fertiliser treatment on the Pine II site. (sed for treatment*time*depth interaction).

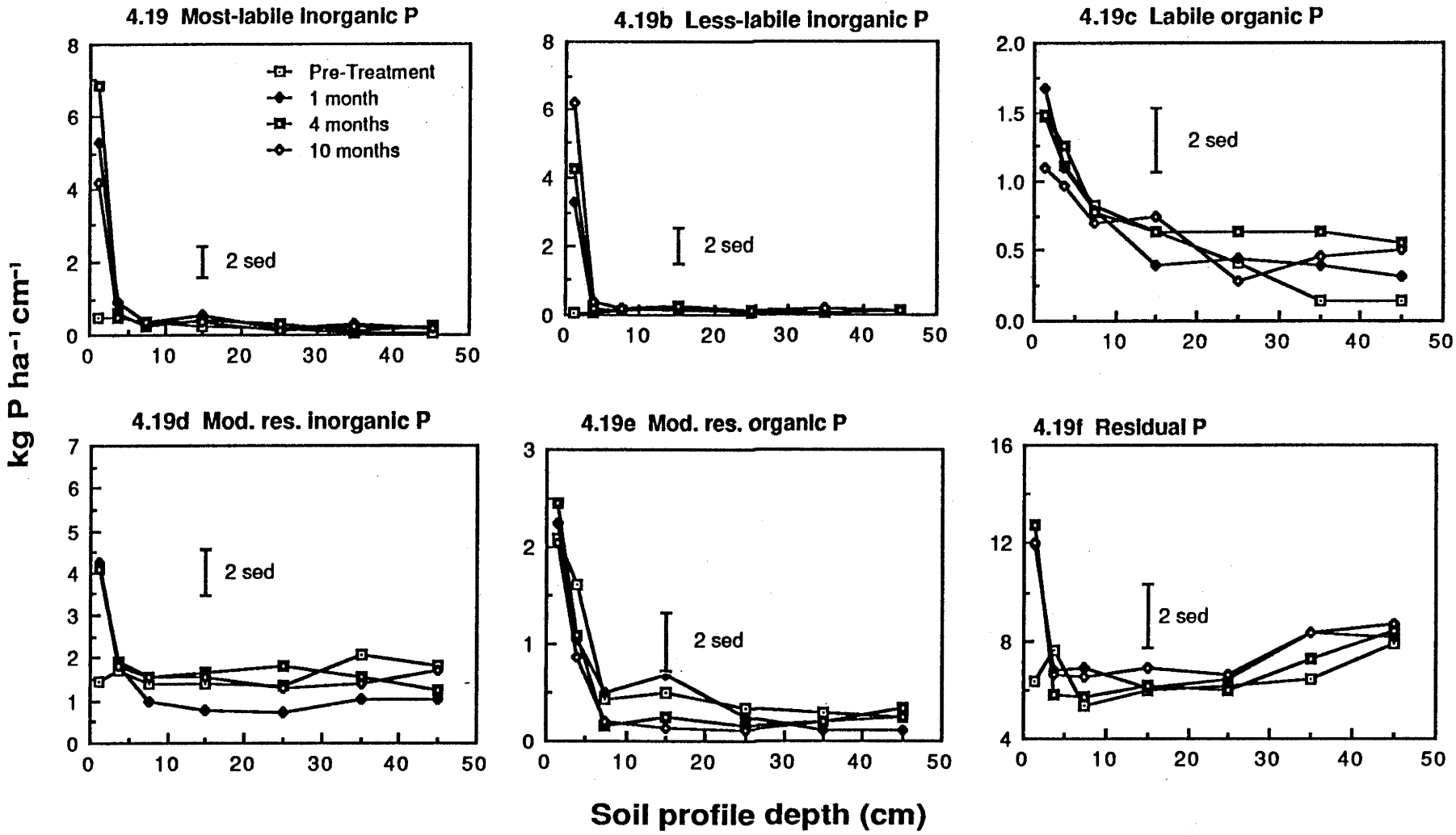


Figure 4.19 Profile P distribution in soils of the Sewage sludge treatment on the Pine II site. (sed for treatment*time*depth interaction).

residual P fractions was limited to the surface 2.5 cm of the soil profile. The profile distribution of the other fractions were not affected by sewage sludge addition.

The responses of added fertiliser and sludge on the Pine II site are comparable with those obtained on the eucalypt site, and therefore interpretations given for the eucalypt site also apply for the Pine II site. Again, these results are generally comparable to those of Experiment 1. It is presumed that the responses to fertiliser and sludge on the Pine I site soil differed from that of the Pine II and eucalypt site because of the litter cover rather than any inherent different soil properties affecting the movement and transformation of P.

It was observed in Experiment 1 that although most P was not released from the sewage sludge lumps, constituents that affected P sorption were released to soil, such that P sorption increased in the 2.5-5 cm region. The Ca ion was suggested as contributing to the increased sorption. $CaCO_3$ from this kind of sewage sludge also increases soil pH (Willett *et al.*, 1986). Therefore, changes in soil pH were measured to determine the depth to which sewage sludge could affect soil chemical properties other than increasing soil P. Soil pH measurements on the Pine I site would also show if other chemical constituents from sewage sludge were withheld from the soil.

4.3.2.6 Changes in soil pH after fertiliser and sewage sludge treatment

pH was measured in the soil-water suspension (1:5) as

described in Section 3.4.1, on soils from all three sites. Figure 4.20 shows the profile changes in pH with time in the fertiliser and sewage sludge treatments. For ease of comparison between different sites, the same scale has been used for drawing the pH curves.

Figure 4.20 shows that when compared with that of sludge, the effect of superphosphate fertiliser on soil pH was not significant. However, pH in the soils of the fertiliser treatments declined slightly in the surface 0-2.5 cm between 1 and 4 months after fertiliser application, before returning to the original level by 10 months. This happened in soil from all three sites, but was less pronounced on the Pine I site. This is probably because there was less fertiliser in contact with soil on the Pine I site. Most phosphate fertilisers, including superphosphate, are acidic from the acidulation of phosphate rock during manufacture (Engelstad and Terman, 1980), and would therefore decrease soil pH. In the vicinity of the fertiliser granule, the saturated solution may have a pH of as low as 1 to 1.5 (Rajan, 1977).

The sewage sludge significantly increased pH in soils from the eucalypt and Pine II sites only in the surface 2.5 cm. There were non significant increases down to about 5 cm. pH increased with time, and by about 2 units at the 10th month. This shows that there were effects of sewage sludge only in the soil surface (0-5 cm) during the 10 month after application. Similar trends were observed for the wet sewage sludge treatment in Experiment one (Figure 4.21), where the soil surface (0-2.5 cm) pH increased from the pre-treatment level of 6.1 to 7.4 in 26 months. This suggests that the

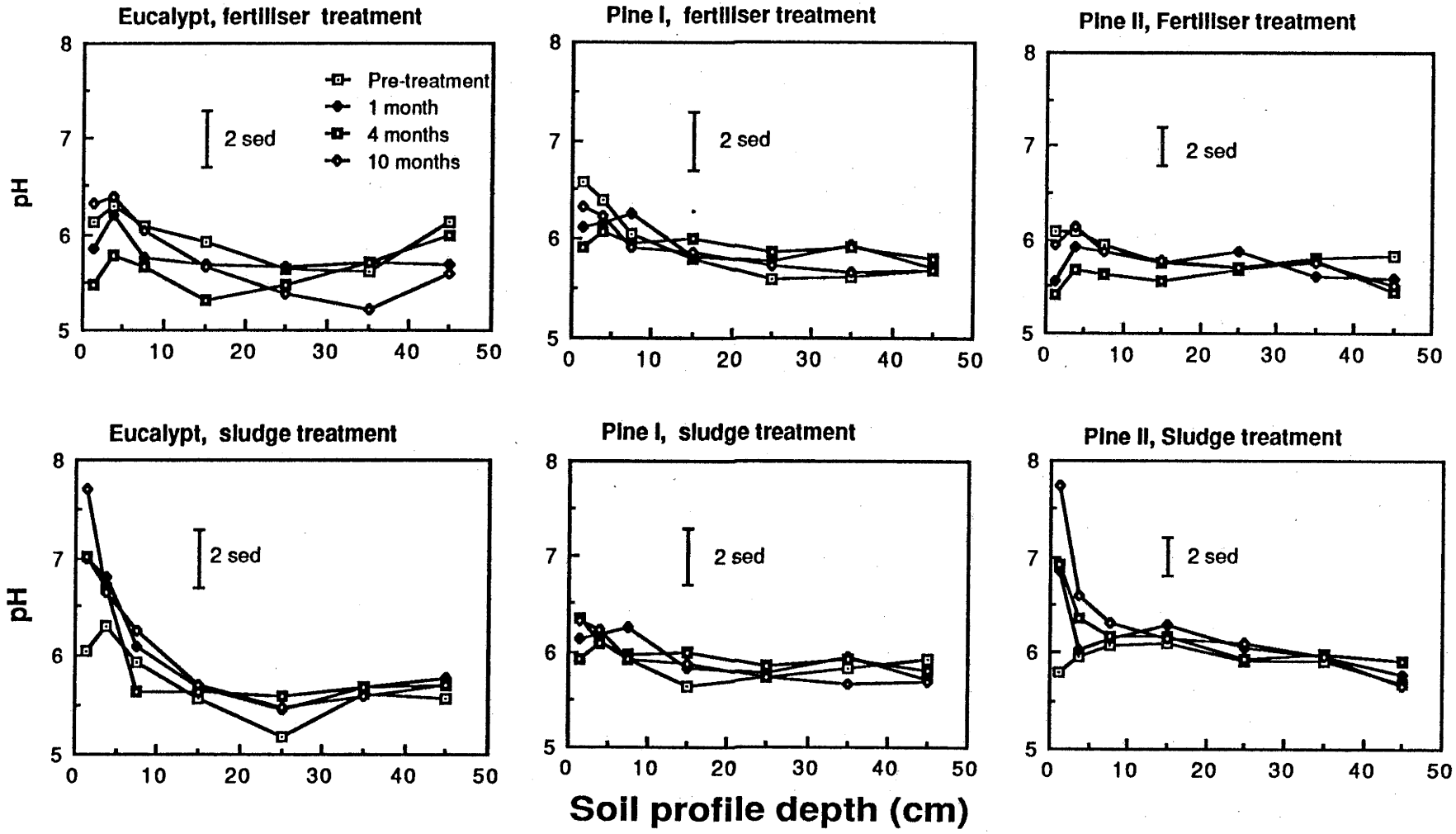


Figure 4.20 Effects of fertiliser and sewage sludge treatments on pH of soils from the three experiment sites.

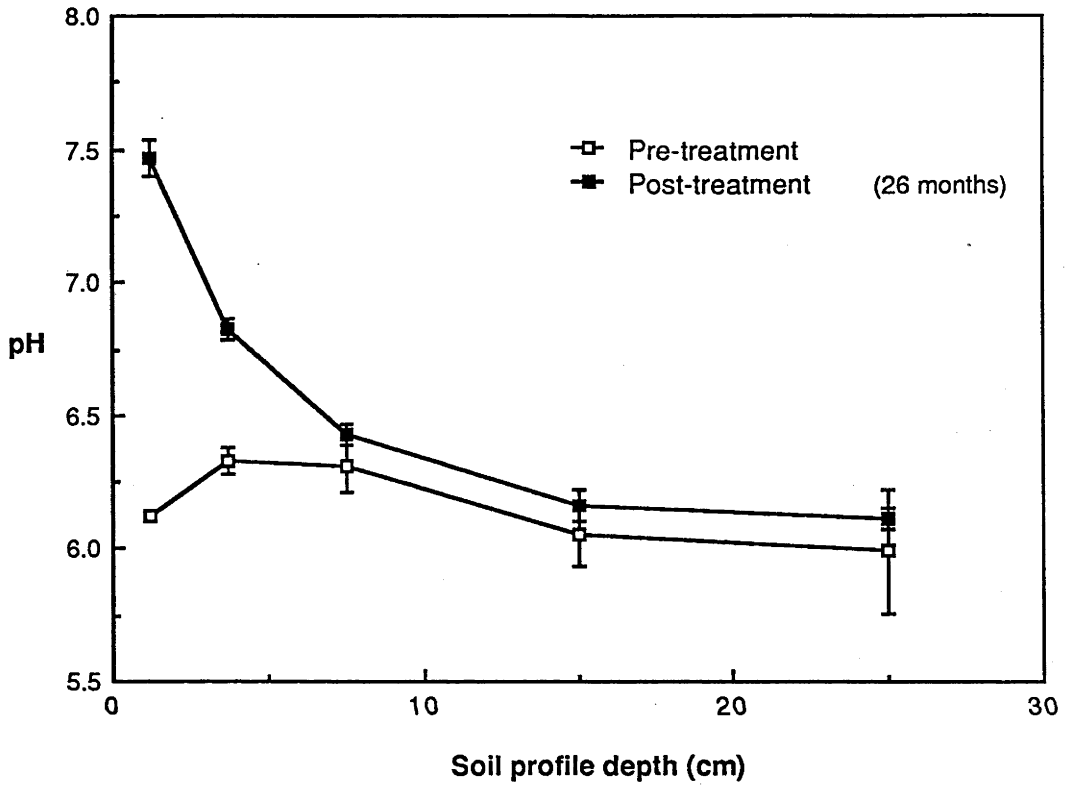


Figure 4.21 The effect of wet sewage sludge on soil pH 26 months after application

chemical effects of both wet and air dry sewage sludge on soils may be similar.

There was no change in pH of soil from the sewage sludge treatment on the Pine I site. This again may reflect the physical capacity of litter to hold sewage sludge from reaching the soil.

4.4 Conclusions on experiments on the movement and transformation of P in a yellow podzolic soil

Results of experiment two are consistent with those of Experiment 1, confirming that adsorption processes are dominant in short-term immobilisation of fertiliser P. Tree uptake of P is known to be small in relation to that applied (e.g. Ballard, 1980). Soil microbial immobilisation of fertiliser P is a feature of the mineralisation-immobilisation turnover (Jansson and Persson, 1982), but the net immobilisation in live microorganisms is small. Thus there was no significant increase in soil organic P. Transformation of microbially immobilised P to resistant forms is slow and could not be detected in the residual P fraction. The build up of organic P (plant and microbial) to significant levels would require longer periods of monitoring than the period of this study.

Both Experiments 1 and 2 have shown evidence of P movement in the soil profile. In this soil (yellow podzolic), movement was limited to the root zone. It is possible that under heavy rainfall after fertiliser application, there could be some considerable loss of P to the subsoil in mass flow through soil channels. It is also possible that movement in

the profile was partly due to the decrease in the surface soil's buffering capacity, such that excess P moved to subsurface depths.

The movement of fertiliser and sewage sludge P on the first rotation pine site indicates the significance of the presence of litter on the capacity to respond to added P. For the added fertiliser to be effective, there must be means of leaching the fertiliser through the litter to the soil. Rainfall may not be enough. In the next Chapter, the effect of litter on P added in solution will be studied.

Sludges have been reported to produce positive results with respect to tree growth (Luttrick *et al.*, 1986). However, results of the experiments in this Chapter suggest that the form of application is important. Air dried and ground sludge had greater effects than wet sludge in raising the level of surface soil inorganic P.

Perhaps the most important observation in these experiments is that a large proportion of P added in fertiliser and air dry sewage sludge stayed in the yellow podzolic soil in labile forms considered available to plants. Labile inorganic P in soil from the wet sewage sludge treatment increased slowly with time. Since soils differ in their sorption characteristics, it is conceivable that the amount of fertiliser P retained in labile forms in soils would be different. However, Barrow (1980) warned of the uncertainty about the importance of differences between soils in their sorption characteristics. For example, while testing the decline in effectiveness of previously applied P to 22 soils with different sorption capacities, Barrow (1973)

observed a slight and negative trend between residual effectiveness and the ability to adsorb phosphate. Thus the following Chapter examines the amount of fertiliser P retained in the most-labile form by forest soils with different sorption capacities. The soils are also compared for their effects on movement of fertiliser P in the soil, and the rate of decline of the most-labile fraction over a period of 12 months.

CHAPTER 5 CHANGES IN THE MOST-LABILE P POOL IN SOILS WITH DIFFERENT P SORPTION CAPACITIES

CHAPTER FIVE

CHANGES IN THE MOST-LABILE P POOL IN SOILS WITH DIFFERENT P SORPTION CAPACITIES

5.1 Introduction

In Chapter 4, it was shown that substantial amounts of fertiliser P (> 20%) were still present in soil in the most labile P fraction after two years, but that it was slowly declining. Larsen *et al.* (1965) measured the rate of P immobilisation in terms of the time in years required for half the applied P to become non-labile. There was no obvious relationship between the half-life values and soil properties normally associated with the adsorption complex (e.g. clay and organic matter content, extractable Al and Fe). This would appear to conflict with some reports that the residual effectiveness of P is least on very highly buffered soils and greatest on weakly buffered soils (Anderson and McLachlan, 1951; Hughes and Searle, 1964). Movement of P has also been attributed to the nature of soil as it relates to the chemical reactions that occur in the soil (Lindsay and Moreno, 1960; Logan and McLean, 1973). In order to relate these findings to Australian forest soils, three soils with different sorption capacities have been tested for their ability to retain P in the most labile form. These are the yellow podzolic soil used in Chapter 4, a red podzolic soil and a yellow earth soil. The study has been based on the addition of fertiliser P to soil columns under controlled environmental conditions, and an examination of the extent to which the P has moved in these cores. The P has been applied

in solution form, followed by controlled leaching of the soil columns with distilled water.

In the experiment described in Chapter 4 where fertiliser was added to different soil-litter situations (Section 4.3), litter was observed to be able to physically hold fertiliser P against contact with the soil. Because of this, the movement and chemical transformation of P in soils under litter was not established. In order to examine this question further, different levels of litter have been applied to the soil columns containing the yellow podzolic soil.

5.2 Experimental procedures

The soils used in this study are the yellow podzolic, the red podzolic and the yellow earth, from sites 1, 3 and 4 respectively (Figure 3.1). These soils were chosen on the basis of their different P sorption capacities. The yellow earth soil has the highest sorption capacity, and the yellow podzolic soil the lowest (Figure 5.1). Radiata pine was growing on all three sites from where the soils were collected (Section 3.2).

Plastic tubes (7.6 cm diameter) were gently driven into the soil (using a drop hammer), 20 cm deep for the yellow podzolic soil, and 10 cm deep for the red podzolic and yellow earth soils. Different core lengths were used on the assumption that P would not move deep in columns of soils with high P sorbing capacities. The soil cores were taken close to one another to minimise lateral soil differences (Plate 5.1). The

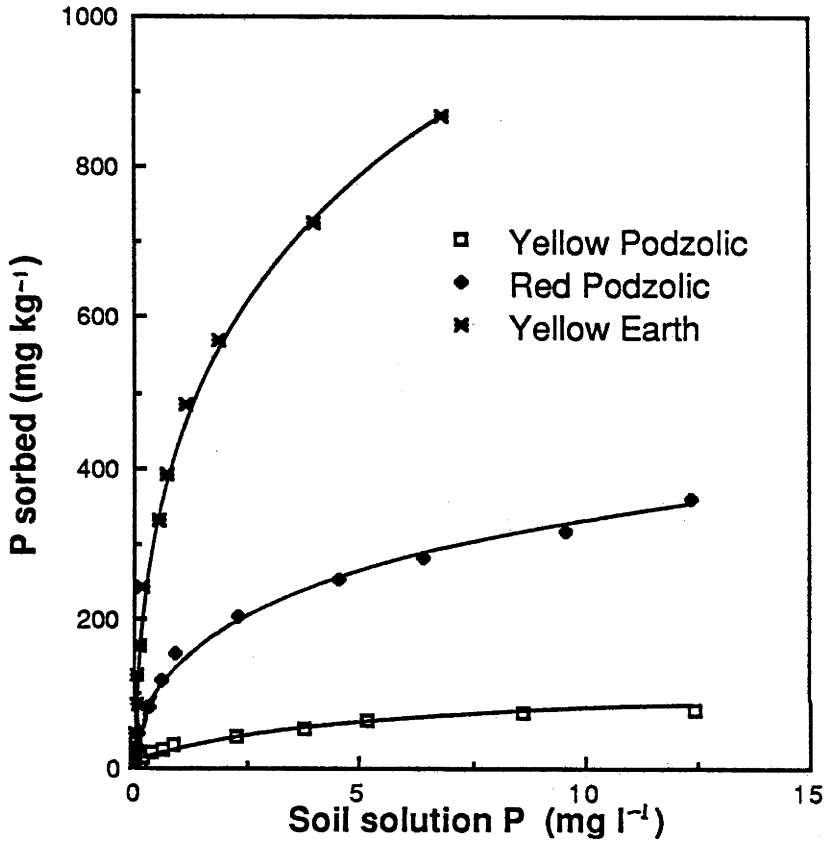


Figure 5.1 A comparison of P sorption curves for the three soils used in the experiment.

P sorbed at the soil solution concentration of 0.2 mg l^{-1}

Yellow podzolic soil = 14.0 mg kg^{-1} (0-5 cm)
Red podzolic soil = 67.0 " (0-3 cm)
Yellow earth soil =230.0 " (0-3 cm)

Plate 5.1 Soil core sampling

tubes containing intact, relatively undisturbed soil columns were carefully dug out and transported to the laboratory. The lower end of the tubes was covered with a nylon cloth to stop loss of soil during subsequent leaching. When not being leached, the soil columns were placed on close fitting plastic plates to minimise evaporation. The top end was covered with plastic film in which a 2.0 mm hole was made for gaseous exchange.

There were four treatments for the yellow podzolic (YP) soil:-

1. Control
2. + 100 kg P ha^{-1} as fertiliser treatment (YPF)
3. As in 2, but with surface litter removed (YPF-L)
4. As in 2, with 4 g extra litter (L layer) added on top of the original litter layer (YPF+L).

The red podzolic and yellow earth soils were either untreated or fertilised with P, without change to the natural litter layer.

The fertiliser was added as a solution of monocalcium phosphate ($Ca(H_2PO_4)_2.H_2O$) over a 4 day period. 5 ml portions were added each day, using a pipette and spreading it evenly over the surface. Litter was initially wetted by spraying with distilled water (hand spray pump) and incubating the soil columns for a few days to overcome the non wetting characteristics of litter. This was to ensure that there was no preferential flow of the added fertiliser through dry litter.

Each treatment consisted of 12 replicates, six of which were sampled after six months of incubation, and the remaining

six after 12 months. The replicates were arranged in random blocks in a dark room maintained at 20°C. Soil moisture was maintained at field capacity and checked by weighing and using a hand spray pump to replace the deficit.

During the first six months of incubation, the soil columns were leached once every two months with one pore volume equivalent of distilled water. This was 450 ml for the yellow podzolic soil and 270 ml for the other two soils. The water was applied evenly over the surface of the soil columns core at a rate of 100 ml per hour. The leachate was collected, its volume measured, and analysed for inorganic and organic P. After six months, the soils were incubated without further leaching.

At each sampling time (6 and 12 months), the litter was carefully removed and analysed for resin extractable P.* The soils were subdivided into 0-2, 2-4, 4-6, 6-8, 8-10 cm depth sections (and 10-13, 13-16, and 16-20 cm for the yellow podzolic soils). Two cores from each treatment were bulked to give three replicates per treatment at each sampling time. The soils were sieved (2 mm) and analysed for resin extractable P (most-labile inorganic P) immediately after sampling, without drying the soils and litter.

Resin extractable P was chosen for use in the study on the assumption that it preceeds formation of any other soil P fraction when fertiliser P is added to soil. It is the fraction most likely to determine the P concentration in soil solution and the rate at which the concentration is replenished upon leaching or removal of P by plant roots (Mattingly, 1974). The previous experiments (Chapter 4) have

* 2.0 g of litter were used without drying or grinding. Data were expressed on oven dry (70°C) basis.

shown that resin extractable P moved deeper or as deep as any other P fraction, and would therefore be a good indicator of P movement in a profile. Freshly collected soils were used so as to minimise the contribution from the microbial biomass P killed during air drying (Sparling, 1985; Section 3.3.2.2).

Statistical analysis was done on individual soil types because different soil core lengths were used, and the amounts of water used for leaching were different.

5.3 Results

5.3.1 The effect of different litter levels on fertiliser P movement in the yellow podzolic soil.

Table 5.1 summarises the quantities of most-labile P in the yellow podzolic soil associated with the different litter treatments after two periods of incubation. There were no large differences of the most-labile P in soils under different litter treatments after addition of fertiliser P. Of the 100 kg P ha^{-1} added, about 20% was in the most-labile pool after 6 months of incubation, and decreased to 17% after 12 months. The results confirm the field observations reported in Chapter 4 of an initial high reaction rate and a subsequent slowing down of that rate.

Table 5.1 Resin extractable P from the yellow podzolic soil under different litter treatments (se in brackets)

Months of incubation	Treatment			
	Control	YPF	YPF-L	YPF+L
		kg P ha^{-1}	(20 cm)	
6	5.1(0.7)	24.8(3.9)	27.9(2.2)	27.4(0.7)
12	3.0(0.5)	21.8(3.5)	19.8(2.6)	21.9(0.9)

Figure 5.2 shows the distribution of most-labile P in the profiles of different litter treatments. The significant depth-treatment interaction was a result of higher levels of P resulting from fertiliser P addition. The graphs show that the most-labile P in the control treatment was small (less than 0.3 kg P ha^{-1} cm^{-1}). The addition of fertiliser P increased most-labile P down the whole length of the soil columns. The increase in the surface 2 cm was about 10 times the control, while at 2-4 cm the increase was only about three times. There were no significant effects of the different litter treatments on profile distribution of most-labile P. This implies that that any restricted movement of P in the soil profile of the field treatment in the second rotation pine (Section 4.3.2.3) was limited by litter acting as a physical barrier to both fertiliser and water. As for the previous experiments, most sorption reactions were in the surface soil, and Figure 5.2 shows that between 6 and 12 months, P had declined by 1 to 2 kg P ha^{-1} cm^{-1} in the top 2 cm of the soil.

The results obtained from the soil column experiment were comparable to those of the field experiment in the amount of P retained in the most-labile form (Chapter 4). After 12 months incubation, 18.8% of the added P remained in the soil as most-labile P compared with just over 20% in the field experiment. Both experiments showed movement of P with depth, but the soil column experiment suggests that losses of P below the root zone through leaching would be small. Because

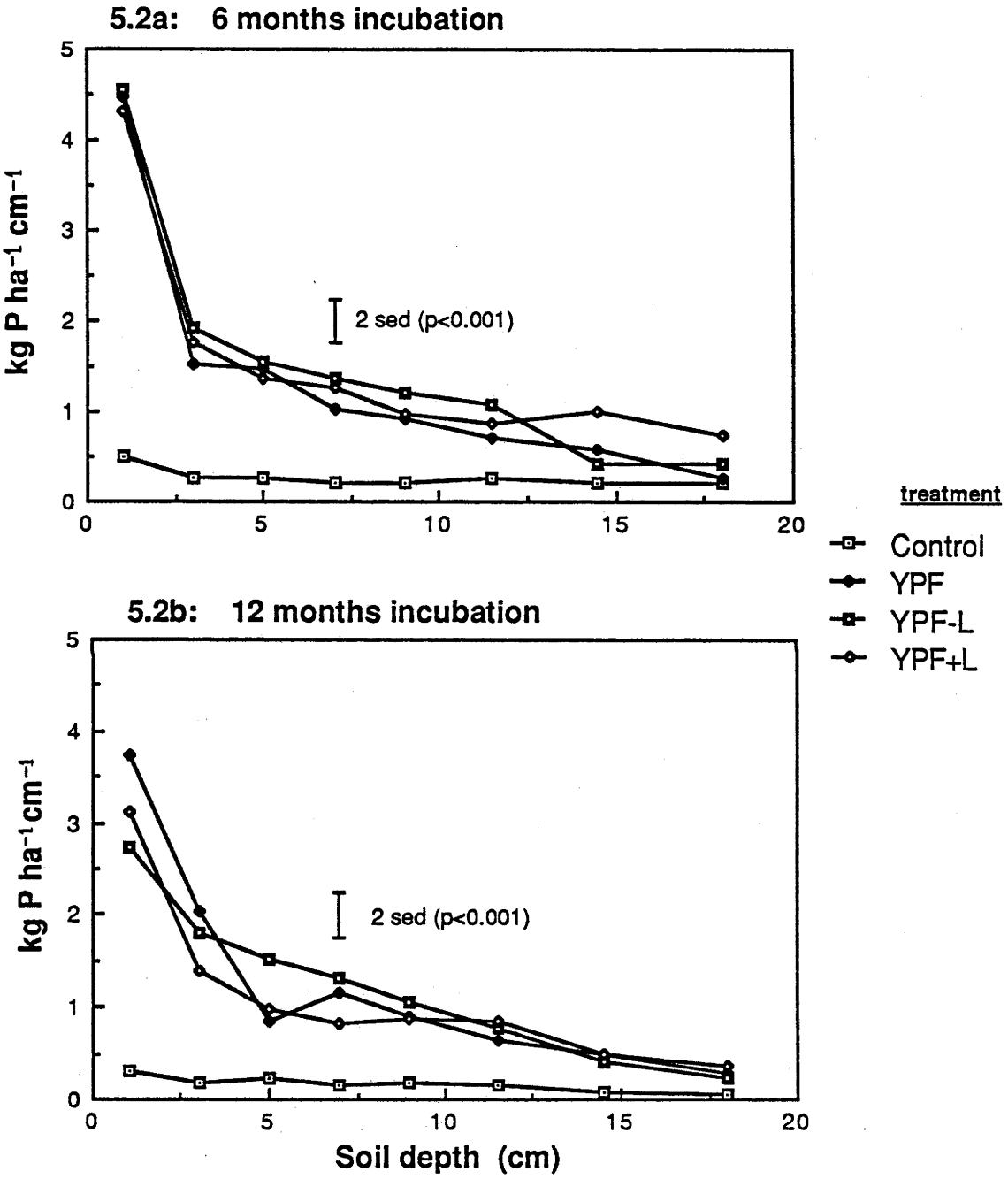


Figure 5.2 The effect of different levels of litter on fertiliser P in a yellow podzolic soil after two periods of incubation.

(YPF=fertilised; YPF-L=fertilised, litter removed; YPF+L=fertilised, extra litter added)

transformation of P to non-labile forms is not a geometric progression process (Barrow, 1980), fertiliser P in the yellow podzolic soil is most likely to be available for tree uptake for long periods.

5.3.2 The behaviour of fertiliser P in the red podzolic and yellow earth soils

The red podzolic and yellow earth soils had higher sorption capacities compared with the yellow podzolic soil (Figure 5.1). The amounts of most-labile P in the two soils after 2 periods of incubation are given in Table 5.2. The amount of P retained in resin extractable form in the yellow earth soil was slightly less than that retained in the red podzolic soil after 6 months (12.1% compared to 13.5%). The difference widened by the 12th month (6.6% compared to 12.6%).

Table 5.2 Resin extractable P in the red podzolic and yellow earth soils after adding fertiliser

Months of incubation	Soil/treatment			
	Red podzolic		Yellow earth	
	Control	+Fertiliser	Control	+Fertiliser
		kg P ha^{-1}	(10cm)	
6	2.6(0.2)	16.1(1.3)	0.5(0.1)	12.7(1.3)
12	2.3(00)	14.9(1.3)	0.4(00)	7.0(1.1)

se of means in brackets.

The profile distribution of most-labile P is given in Figure 5.3 and shows that the addition of fertiliser P increased the most-labile P fraction throughout the entire column lengths of both soils (significant to about 5 cm). Between the 6 and 12 month incubation period, there was no

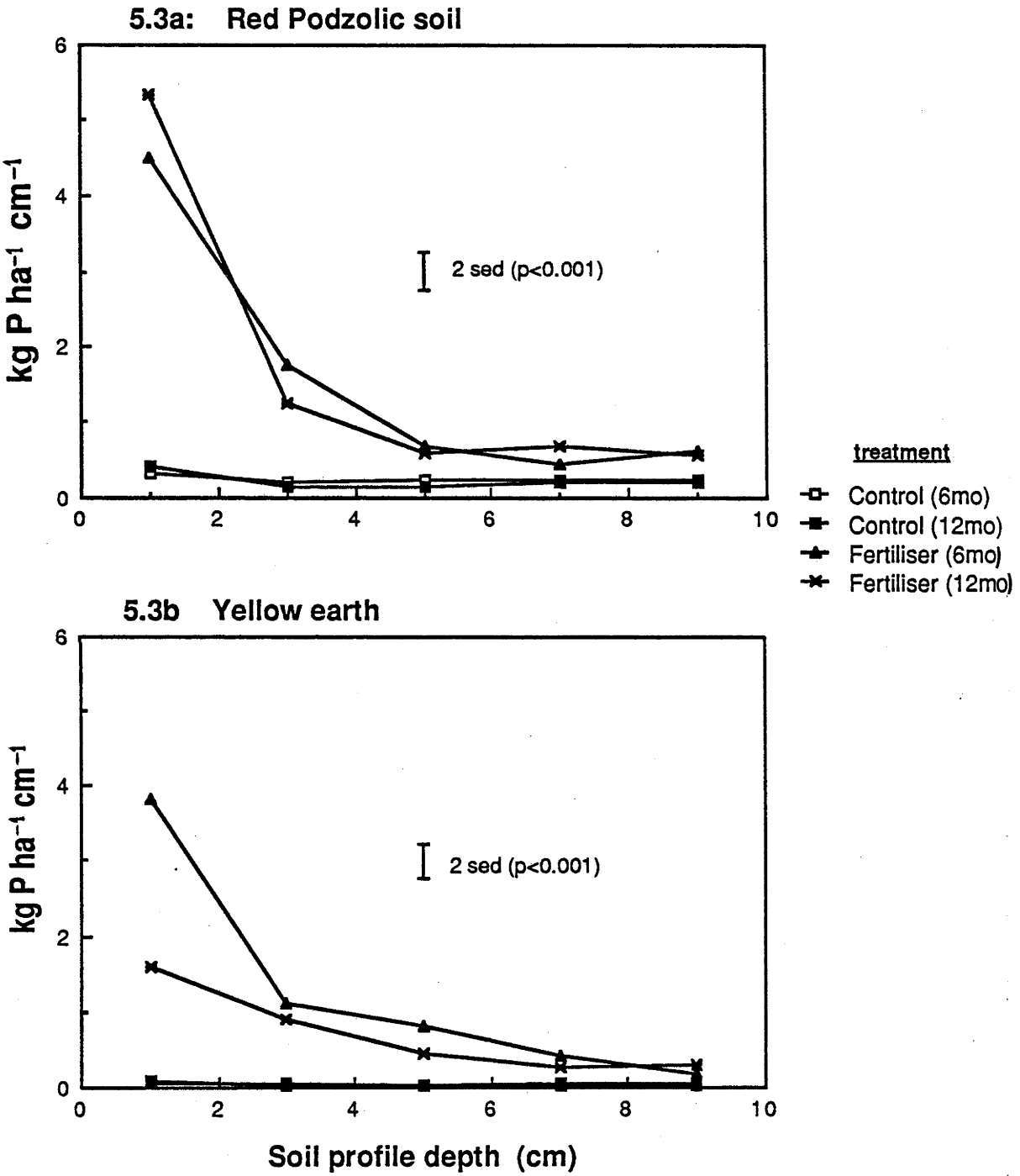


Figure 5.3 Fertiliser P distribution in soil cores after two periods of incubation.
(mo=months of incubation)

significant change in most-labile P in the red podzolic soil, while it declined by more than 50% in the surface 0-2 cm of the yellow earth soil.

It appears that changes in most-labile P with time are a result of differences between soils in their ability to reduce most-labile P by the slow reactions that follow sorption. For the same reason, the amount of P recovered in the yellow podzolic soil (18.8%; Table 5.1) was greater than that in the red podzolic and yellow earth soils even though there was more soil in the yellow podzolic cores (i.e. double core depth). The results therefore are consistent with the observations that the residual effectiveness of P is least on very highly buffered soils and greatest on weakly buffered soils (Anderson and McLachlan, 1951; Hughes and Searle, 1964). However, Holford and Mattingly (1976) studied soils in which they observed that the largest long term increase in labile residual P, following fertilisation, was in the soil with the largest adsorption capacity, whereas the smallest increase was in soil with the greatest adsorption energy (described by the slope of the sorption isotherm). Holford (1977) therefore suggested that loss of exchangeability of adsorbed P may be a function of the adsorption energy rather than the adsorption capacity. This would explain the results of Larsen *et al.* (1965) where chemical immobilisation of P was not related to soil adsorption properties. Effectiveness of fertiliser P on weakly buffered soils may also be reduced by losses through leaching. Some of the fertiliser P moved through the soil columns and was collected in the leachates.

5.3.3 P in water leachates from different soils

Table 5.3 is a summary of the amounts of P leached out during the first six months of incubation. Fertiliser application increased the amount of inorganic P in the leachates. In the yellow podzolic soil, fertilisation increased inorganic P from 0.06 (control) to more than 2 kg P ha^{-1} in the first leachate (2 months). The amount of leached inorganic P decreased in the successive leachates. There was a large variation in the amount of inorganic P leached at any one time, but there was a tendency for the soil without litter to leach less P than the treatments with litter.

The three soils showed different losses of P in the leachates (Table 5.3). Inorganic P present in the leachate from the high sorbing soil (yellow earth) was about double that from the low sorbing soil (yellow podzolic). This was about 9% of the added fertiliser P collected in 3 pore volumes of leaching water from the yellow earth. This relatively large amount of leachate P in a high sorbing soil, together with movement in the soil columns can only be explained by preferential flow of P (Scotter and Kanchanasut (1981), where solution P moved along channels in the soil without substantial contact and reaction with soil surfaces. As there was more litter on the yellow earth soil columns, it is possible that the litter provided a steady source of P (retained from the fertiliser) which was washed down during the leaching process. More than 10% of the initial added P was still found in the litter as resin extractable P after the 6 month leaching period (Section 5.3.4).

Table 5.3 P removed from "intact" soil cores during leaching with distilled water.

(Units: kg P ha^{-1} ; se of means in brackets; n = 12).

P fraction	Leaching time (months)	Yellow Podzolic Soil				Red Podzolic Soil		Yellow Earth Soil	
		Control	+Fertiliser	+Fertiliser +Litter	+Fertiliser -Litter	Control	+Fertiliser	Control	+Fertiliser
Inorganic	2	0.06(0.03)	2.50(0.61)	2.59(0.62)	2.06(0.34)	0.01(00)	1.81(0.20)	0.02(00)	4.13(0.85)
	4	0.04(0.01)	1.85(0.26)	1.81(0.58)	1.51(0.43)	0.02(00)	1.24(0.14)	00	3.22(0.18)
	6	0.05(0.01)	1.45(0.20)	1.56(0.28)	0.98(0.21)	0.02(00)	1.04(0.11)	0.01(00)	1.98(0.44)
	Total	0.15	5.80	5.96	4.55	0.05	4.09	0.03	9.43
Organic	2	0.04(0.01)	0.21(0.04)	0.14(0.03)	0.21(0.03)	0.03(00)	0.13(0.03)	0.03(00)	0.15(0.02)
	4	0.05(0.01)	0.17(0.05)	0.18(0.04)	0.19(0.04)	0.03(00)	0.08(0.03)	0.01(00)	0.05(0.01)
	6	0.06(0.01)	0.08(0.02)	0.11(0.04)	0.07(0.03)	0.04(00)	0.12(0,03)	0.03(00)	0.12(0.03)
	Total	0.15	0.46	0.43	0.47	0.10	0.33	0.07	0.33

Fertiliser P also increased organic P in the leachates by about three times (Table 5.3). This was still only about one-tenth of the increase in leachate inorganic P. There were no litter treatment effects on leachate organic P in the yellow podzolic soil, and this soil, in turn, produced more leachate organic P than the other two soils. Hannapel *et al.* (1964) suggested that the movement of organic P in soils is associated with microbial cells and cell debris serving as vectors of organic P. It is possible that fertiliser treatment coupled with incubation enabled microbial growth and therefore increased microbial immobilisation of P. This would lead to increased microbial organic P displaced in the leachates.

5.3.4 Retention of P by litter

The amount of resin extractable P in the litter from different treatments is given in Table 5.4. The values of litter are based only on litter retained on the soil columns during coring. There were increased levels of resin extractable P from the litter after fertilisation with 100 kg P ha^{-1}. The total resin extractable P retained increased with the quantity of litter; thus litter on the yellow earth soil columns (43.4 tons ha^{-1}) retained 10.7 kg P from the added fertiliser at 6months, compared with 0.6 kg P retained by 5.9 tons ha^{-1} of litter on the yellow podzolic soil columns. However, the amount of P retained per unit of litter weight was different because the litter differed in composition. Thus the quantity of litter on the YPF+L

treatment was the same as that on the red podzolic soil (about 14 tonnes ha^{-1}) but with about 2 kg P ha^{-1} less. This was because the litter added in the YPF+L treatment was mainly the L fraction, and probably had less P holding capacity. The amount of fertiliser P retained by litter (e.g. 0.21 kg P $tonne^{-1}$ for the litter on the yellow earth soil columns at 12 months) is consistent with the P sorption curve established in Chapter 4 (Section 4.3.2.4), where maximum sorption was about 250 mg kg^{-1} (0.25 kg $tonne^{-1}$). This shows that a thick litter layer may also chemically retain substantial amounts of fertiliser P and should be taken into account in late fertilisation practices. In this experiment, litter on the yellow earth soil retained about 10% of the added fertiliser.

Table 5.4 Retention of fertiliser P by litter from different treatments (se of means in brackets)

Soil	Treatment	Total litter tonnes ha^{-1}	Litter resin P kg P ha^{-1}	
			6 months	12 months
Yellow Podzolic	YPC	5.9(0.3)	0.1(00)	0.1(00)
	YPF	5.9(0.3)	0.7(0.1)	0.4(0.1)
	YPF+L	14.1(0.6)	2.9(0.3)	2.6(0.1)
Red Podzolic	Control	14.4(3.7)	0.4(0.1)	0.5(0.1)
	+ Fertiliser	14.4(3.7)	4.9(0.4)	4.5(0.3)
Yellow Earth	Control	43.4(9.0)	0.4(0.1)	0.3(0.1)
	+ Fertiliser	43.4(9.0)	11.1(0.3)	9.0(0.4)

YPC = yellow podzolic, control
YPF = yellow podzolic, fertilised
YPF+L = yellow podzolic fertilised after adding extra litter

During the 6 to 12 months incubation period, the amount of resin extractable P from litter on the fertiliser treatment columns showed a tendency to decline, albeit slightly (Table

5.3). It is possible that the decline was partly due to release of P to the soil or being microbially immobilised. Bromfield and Jones (1972) observed that wetting plant material (hayed-off pasture) resulted in the formation of water insoluble P by microorganisms. Birch (1961) also demonstrated that microbial growth on dead plant material was rapid, but was unable to detect any release of P. Birch presumed this to be due to microbial immobilisation.

5.4 Discussion

It was noted in Chapter 4 that short-term accumulation of residual fertiliser P was by physico-chemical reactions (adsorption) in the soil. Soils vary in their capacity to immobilise P, and usually immobilise more P than is normally applied in fertiliser (Holford, 1977), except for coarse textured soils. Results of experiments in this Chapter agree with this view in that the reduction of most-labile P during the second 6 months of incubation was faster in the high sorbing yellow earth soil than in the other soils.

During and after immobilisation, a proportion of P remains in the soil solution because of the equilibria which are established between the solid and solution phases. This solution P may be lost by leaching to greater depths in the soil profile and beyond the reach of plant roots (Holford, 1977). Leaching was observed in both field and soil column experiments. Profile movement of P as a function of the soil sorption appears to be important where P is leaching from the equilibrium situation. Where P is a result of fertiliser

dissolution, preferential flow of the dissolved fertiliser through biogenic macropores and along aggregate surfaces may dominate (Scotter and Kanchanasut, 1981). This explains the high mobility of P in the yellow earth soil. Since the porosity and structure of field soils appear to influence the amount of P sorbed, it might be more appropriate to express P sorption on a unit volume of soil for field situations, rather than on a unit weight used in laboratory practices.

While using soils with different sorption capacities packed in columns, Logan and McLean (1973a) observed that with rapid leaching, P dissolved from the surface leached through the soil faster than the soil sorption processes could operate. This emphasises the function of water moving through the profile as a factor determining movement of P in the soil.

CHAPTER 6 EFFECTS OF POST-HARVEST LITTER AND SLASH MANAGEMENT ON SOIL P

CHAPTER SIX

EFFECTS OF POST-HARVEST LITTER AND SLASH MANAGEMENT ON SOIL P

6.1 Introduction

The study has been concerned thus far with forms of P occurring in some Australian forest soils and the fate (movement and transformation) of P added to soils either as inorganic salt (superphosphate fertiliser) or organic material (sewage sludge)*. It is important to understand such processes in predicting the response of forests to P fertilisers, and in determining optimum fertiliser regimes for soils with different physical and chemical characteristics.

The study now focuses on the nature and transformation of native soil P, that is the processes by which P becomes available from organic matter, and may be subject, in turn, to further immobilisation through either microbial metabolism or sorption.

Many forest operations in wood production do not use any fertiliser (Ballard, 1981) to compensate for site nutrient drain associated with the harvest of forest products (Crane and Raison, 1980; Raison *et al.*, 1982). Rather, the regenerating stand will draw in part on the pool of available P mineralised from the organic matter maintained or accumulated within the soil. Studies by Nambiar and Cellier (1985) indicate that with appropriate silvicultural practices, organic matter could supply sufficient N to young trees. This may also apply to P. Harrison (1987) has pointed out the low

*See note p. 80

annual P uptake by trees and ground flora (deciduous woodland) of about 11 kg P ha^{-1} yr^{-1} and the low rates of mineralisation of organic matter (4 mg kg^{-1} soil $month^{-1}$) that would be required to meet this uptake. Ballard (1980) has compiled from various authors gross annual P uptake values of coniferous species, ranging from 1.7 kg P ha^{-1} yr^{-1} (29 - 30 year old *Pinus banksiana*) to 13.8 kg P ha^{-1} yr^{-1} (4 - 8 year old *Pinus radiata*). Therefore, variations in soil conditions or site management factors that affect mineralisation could be important in influencing P supply from the organic pool to young trees.

Little is known of the processes by which the organic P pool in Australian forest soils becomes available to trees. However, it is known that Australian forest ecosystems may be able to maintain themselves with low levels of soil P supply by making efficient use of relatively small quantities of P within the tree (Attiwill, 1972; Florence, 1981) and due to efficient recycling within the soil-tree system.

As a forest stand develops, there is an accumulation of nutrients and dry matter within different components of the ecosystem. For example, in a pine forest canopy, closure shades out understorey plants, and pine litter becomes the major source of organic matter on the forest floor. About 30% of the above ground P is in the litter layer (Switzer and Nelson, 1972; Lamb and Florence, 1975) which is therefore a major path in the biogeochemical cycle of P in the forest ecosystems (Ballard, 1980).

Where tree crowns are left on site after harvesting, further litter input to the forest floor can be expected. In

such cases, P losses from the site in harvested wood could be less than 1 kg ha^{-1} yr^{-1} (Ballard, 1980) since only the more nutrient poor components of the forest stand are removed. Following the harvest there may be higher soil temperature and moisture leading to increased mineralisation of the organic layers (Pritchett, 1979). The harvest also effectively eliminates uptake of P, and any increased input of P from forest floor to the soil due to increased mineralisation, may react with soil or be leached.

The relative increase in mineralisation, P sorption or profile redistribution will depend on the method of site preparation after harvest. Common methods have included broadcast burning or windrowing and burning of logging slash. In this case there could be appreciable losses of P and other nutrients to the site. The retention and management of litter and logging residues as mulch and as a source of nutrients could reduce this loss. Such practices have been proposed as an alternative to residue disposal and a means of increasing site productivity (Farrell *et al.*, 1981; Squire *et al.*, 1979, 1985).

Optimal management of the forest floor between harvest and re-establishment should lead to the retention of most of the site P in that part of the profile from which it can be released at the rate suitable for the next crop. Tests need to be done on various management systems to establish their effects on soil P. This Chapter is concerned with the way labile P was influenced by different litter and slash management treatments after the harvest of a radiata pine forest. The study was done at Caroline Forest Site, Mt

Gambier, where Dr. S. Nambiar and co-workers are studying the dynamics of organic matter and nitrogen between crop cycles of radiata pine. Both N and P studies consisted of field and laboratory experiments.

6.2 Field experiment

6.2.1 The study area

There is perhaps no more pertinent site on which to study the soil organic P contribution to forest nutrition than the dune sands in the south east of South Australia. The soils are mainly infertile podzolised sands of aeolian origin, low in organic matter and fertility (Nambiar and Cellier, 1985). The soils have high permeability and low water holding capacity so that leaching is an ongoing active process. Consequently, the leaching of salts and bases has left the soils generally acidic. Total soil P is usually less than 0.01% (Stace *et al.*, 1968). On the study site, total P to a depth of 30 cm was 164 kg ha^{-1}, with more than 35% being held in the above ground slash and litter (Table 6.1). $NaHCO_3$ extractable (16 h) inorganic and organic P were 6 and 17 kg P ha^{-1} respectively, in the control treatment, 8 months after start of the experiment (Figure 6.2).

Despite the low fertility levels, high levels of wood production have been obtained in first rotation stands of *P. radiata*, but this is usually followed by decline in site productivity in succeeding rotations (Keeves, 1966; Boardman, 1978; Woods, 1980). The greater part of site production loss

occurs early in the second rotation (Florence and Lamb, 1975) when apparently *P. radiata* is unable to fully express its outstanding capacity for vigorous growth. This is the period of high nutrient uptake.

While it has been possible to reverse production decline through sophisticated site preparation regimes and use of fertiliser (eg Woods, 1976), it is claimed that production might also be maintained through appropriate litter and slash management regimes after the felling of the stand (Squire *et al.*, 1979; Farrell *et al.*, 1981; Nambiar and Cellier, 1985). Supporting evidence for this suggestion arises from the fact that some of the reported decline in productivity was after burning of the slash (Keeves, 1966; Woods, 1980), which results in nutrient loss. Thus we are concerned in this Chapter with the nature of the soil labile P pool under a number of slash and litter management regimes.

6.2.2 Experimental procedures

The experiment, established by Dr. S.E.K. Nambiar and co-workers, consisted of four post-harvest radiata pine litter and slash (litter/slash) management treatments applied in quadruplicate plots of 42.5m * 22m each. The forest had been planted in 1947 and had a productivity level of SQ IV (Classification as described by Lewis, 1954). The forest was thinned three times before harvest in August, 1984, when litter and slash treatments commenced. The treatments comprised:

1. surface retention of slash and litter (slash) where clearfelling slash was chopped by a high speed rotary slasher (Hydro-axe) and redistributed over litter by hand.
2. surface retention of litter only (litter), that is slash was removed but litter retained. Trees and crowns were removed from the plots by a forwarder with a hydraulic arm operated from outside the plot. Woody material (> 10 cm) was removed by hand.
3. retention of litter with site cultivation (cultivated). Trees, crowns and slash were removed as in 2, and the site was disc ploughed and harrowed, causing mixing of litter with soil to a depth of about 25 cm.
4. removal of both slash and litter by raking (raked); slash was removed as in 2, followed by litter removal with a rake mounted on a bulldozer.
5. Control (control) where plots were set in the unharvested part of the plantation, about 100 m from the litter and slash treatments.

The layout of the treatments is shown in Figure 6.1. The phosphorus composition of the accumulated above ground surface material, and total P in soil down to 30 cm, at the beginning of the experiment are given in Table 6.1. Of the total 164 kg P ha^{-1}, 8% was contained in the wood, bark and cones, another 8% was in the needles while the original litter layer contained 20%. Total P in the forest floor debris was about 36% of the total P to 30 cm of the soil profile, and therefore an important P pool.

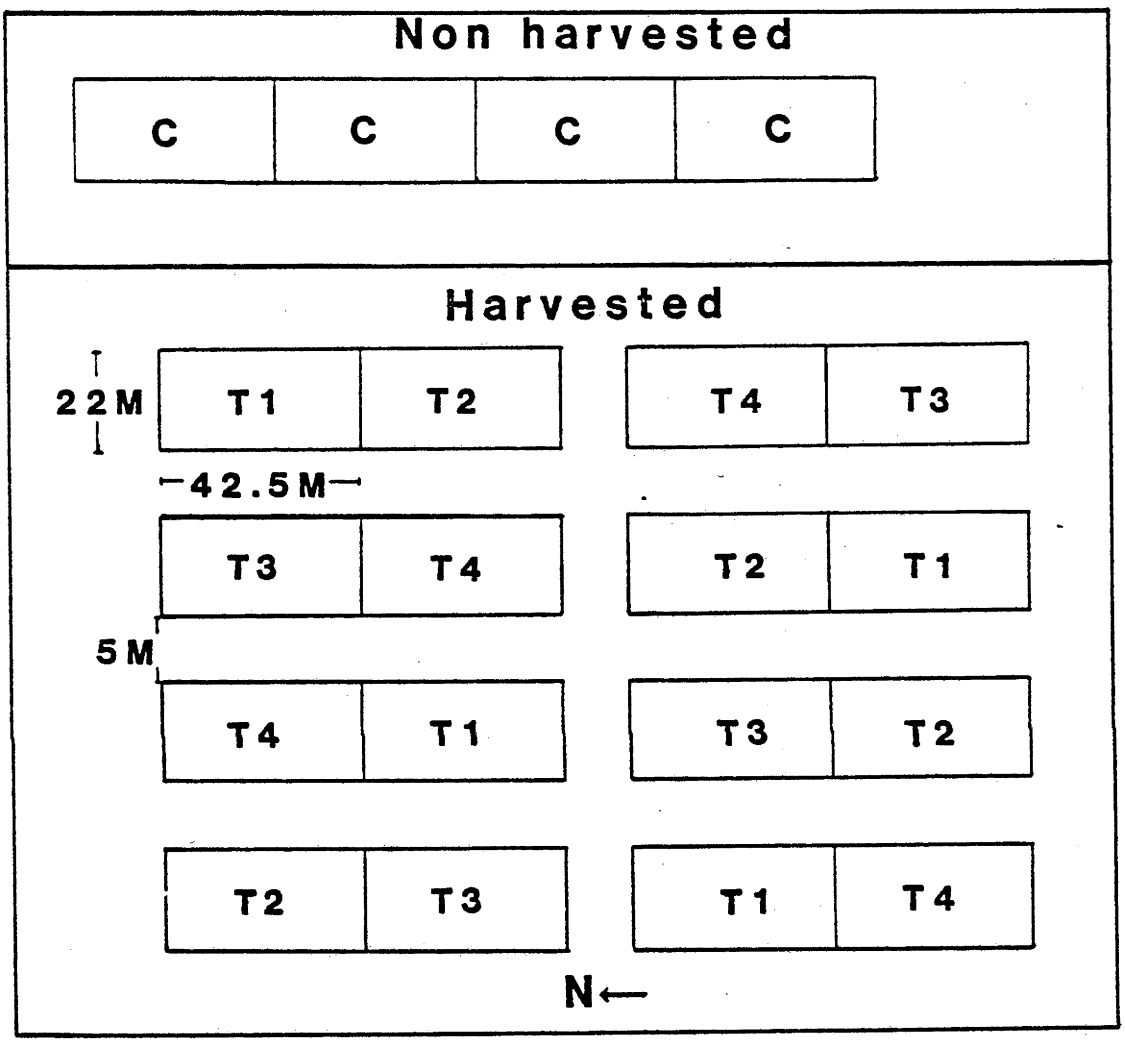


Figure 6.1 Layout of the experiment used to evaluate the effects of clearfelling and slash and litter management on soil P.

Treatments: T1= slash and litter retained
T2= litter only retained
T3= cultivated with litter retained
T4= raked to remove slash and litter
C = control (unharvested forest)

Table 6.1 Phosphorus composition of the forest floor debris and soil at the Mt Gambier Experimental Site. Source: S.E.K. Nambiar and Co-workers, Plantation Forest Research Centre, CSIRO, Mt Gambier (Pers. Commun., 1985).

Material	Weight kg ha^{-1}	Total P kg ha^{-1}
Cones	1860	0.6
Wood + bark >10 cm	740	0.1
Wood + bark 1-10 cm	23860	4.9
Wood + bark <1 cm	14020	7.0
Needles	11760	12.9
Litter, L	3660	2.3
Litter, F+H >1 mm	26320	27.5
Litter, F+H <1 mm	2070	2.9
Total	84290	58.2
Soil (0-15 cm)		60.0
(15-30 cm)		46.0

Soil samples were taken in the fallow period, 8 and 14 months after treatment application. The samples were taken using 5.04 and 2.21 cm diameter corers for 0-5 and 5-30 cm depth samples respectively. Six soil cores were taken per plot and bulked at the depth intervals of 0-2.5, 2.5-5, 5-10, 10-15 and 15-30 cm. All samples were kept field moist at 4°C until analysis. The soils were analysed without sieving, and without drying due to their uniform sandy nature and because drying increases inorganic P (Sparling, 1985; Section 3.3.2.2) and could mask mineralisation effects. Samples of 5 g of soil were used in the $NaHCO_3$ 16 hour extraction (labile inorganic and organic P) because of the low P content of the soil. P fractionation was not carried out in experiments in this Chapter, and labile inorganic P includes both the most-labile and less-labile P fractions.

6.2.3 Results

Although data for the control treatment are included in the statistical analysis, interpretation is treated cautiously as trees growing on this plot are expected to take up mineralised P from the soil. However, removal of the control plot data from the experimental analysis did not affect the statistical trend observed in the other treatments.

Figure 6.2 represents the total P of the labile inorganic and organic fractions contained in the soil 8 and 14 months after treatments commenced. The level of labile organic P is higher than that of labile inorganic P indicating that organic P could be important in P nutrition of forests on these soils. Both fractions comprise about 25% of the total soil P (30 cm depth).

The level of labile inorganic P was somewhat higher in the cultivated treatment compared with that of other treatments at the first sampling (8th month). This may have been due to increased mineralisation, as cultivation stimulates mineralisation (Squire and Flinn, 1981) through exposure of soil organic matter to more vigorous microbial attack (Anderson, 1980). It has also been noted that the release of inorganic P from plant material is partly an autolysis/release process (Martin and Cunningham, 1972). It is therefore possible that the increase of P in the cultivated plot soil was partly inorganic P released from litter by autolysis and mixed with the soil. Resin extractable P was determined in litter (needles, L and F+H layers) only at the beginning of

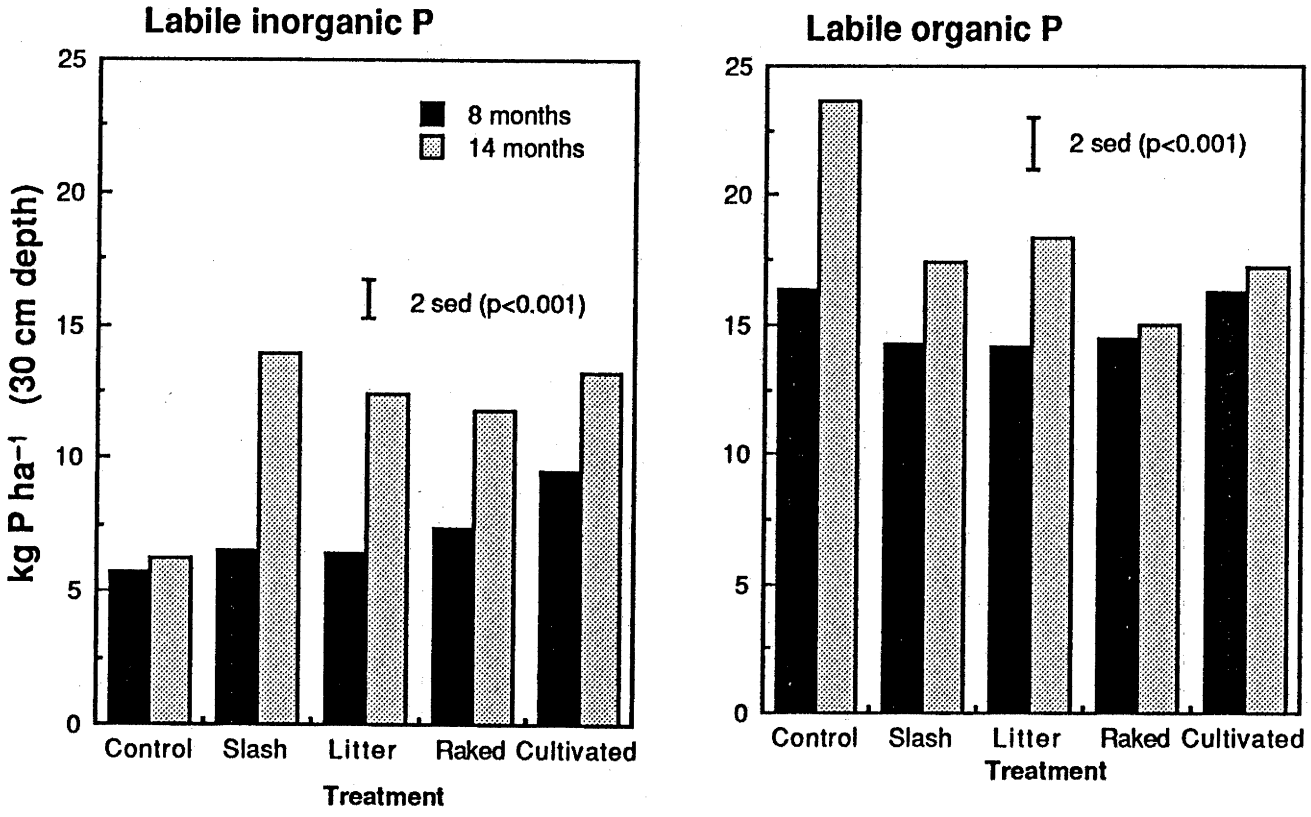


Figure 6.2 Litter/slash treatment and time effects on labile P in soil of a harvested radiata pine forest.

months= months at which soils were sampled after harvest

the incubation experiment (Section 6.3) and was found to be 40.5 mg kg^{-1}. This is equivalent to 1.8 kg P ha^{-1}, and could be significant in contributing to the differences observed between litter and slash treatments.

At the second sampling (14 months), there was no increase of labile inorganic P in the control plot soil, while it increased in soils from all the other treatments (Figure 6.2). Table 6.2 shows that the rates of change for labile inorganic P in the 8 to 14 month period were higher in the slash and litter treatments and lower in the raked and cultivated treatments. For example, there was very little change of labile inorganic P in the control (0.1 kg P ha^{-1} $month^{-1}$), and the rate of change in the cultivated plot (0.6 kg P ha^{-1} $month^{-1}$) was only half of that in the slash plot (1.2 kg P ha^{-1} $month^{-1}$). In contrast with inorganic P, the rate of change of organic P was greatest in the control treatment (1.2 kg P ha^{-1} $month^{-1}$) and least in the cultivated plot (0.1 kg P ha^{-1} $month^{-1}$). Rates of change in the other treatments were in the range 0.5 to 0.7 kg P ha^{-1} $month^{-1}$.

Table 6.2 Rates of change of labile inorganic and organic P in different post harvest litter treatments
Units: kg P ha^{-1} $month^{-1}$; Depth 30 cm.

Treatment	Control	Slash	Litter	Raked	Cult.
Inorganic P	0.1	1.2	1.0	0.7	0.6
Organic P	1.2	0.5	0.7	0.6	0.1

A factor common to the cultivated and raked treatments was the exposure of the soil to the atmosphere. This could mean that as the second sampling was done after the winter

period, cold temperatures could have reduced mineralisation rates particularly in the surface soil. It is also possible the temperature effect may have been less in the mulched plots (litter and slash treatments). Again, on this type of sandy soil, surface exposure in the raked and cultivated treatments might also encourage leaching compared with treatments where litter is retained (slash and litter treatments), affecting the amount of P retained in the mineralisation zone.

Labile organic P increased in all treatments and was highest in the control treatment (Figure 6.2, Table 6.2) suggesting that its conversion from moderately labile and resistant organic P was faster than its mineralisation.

Figure 6.3 shows the distribution of labile inorganic P in the soil profiles of different treatements at two periods of sampling. Compared with the control, there was a significant increase in inorganic P in all treatmments in the surface 2.5 cm at the 8th month. By the 14th month, there was a significant increase down the profile to 10 cm. Compared to the other treatments, the cultivated soil had higher labile inorganic P deeper in the profile at both 8 and 14 months after treatment. This could have been due to increased mineralisation, the contribution from the litter incorporated into the lower part of the soil profile during cultivation, or to leaching from surface horizons during winter.

Figure 6.4 shows the distribution of labile organic P in the soil profiles of different treatments. There were no significant differences between treatments at the eighth month. At the 14th month, the level of labile organic P from the control treatment was higher than that from the other

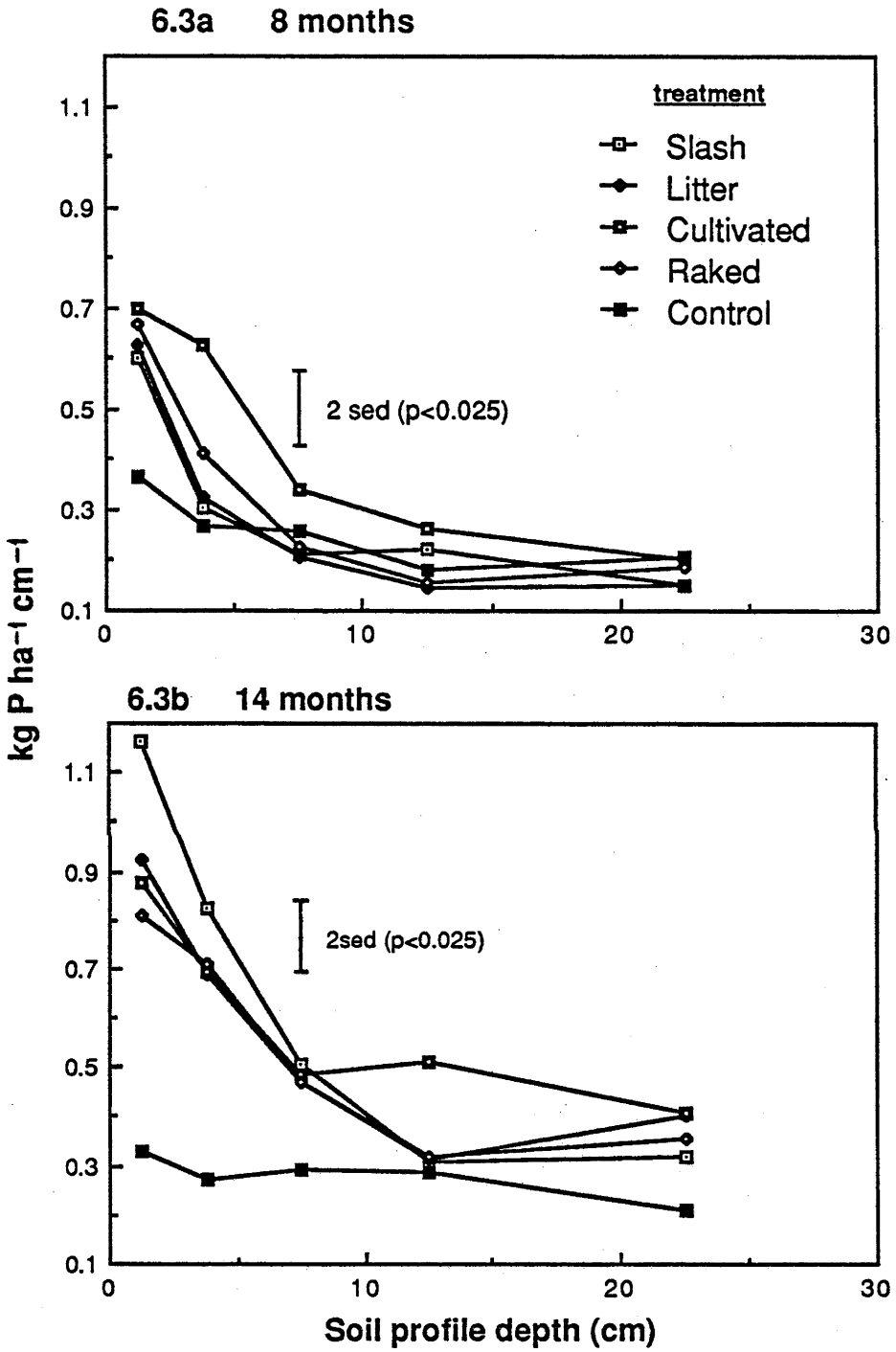


Figure 6.3 Labile inorganic P in soil profiles of different litter/slash treatments at 8 and 14 months after harvest.

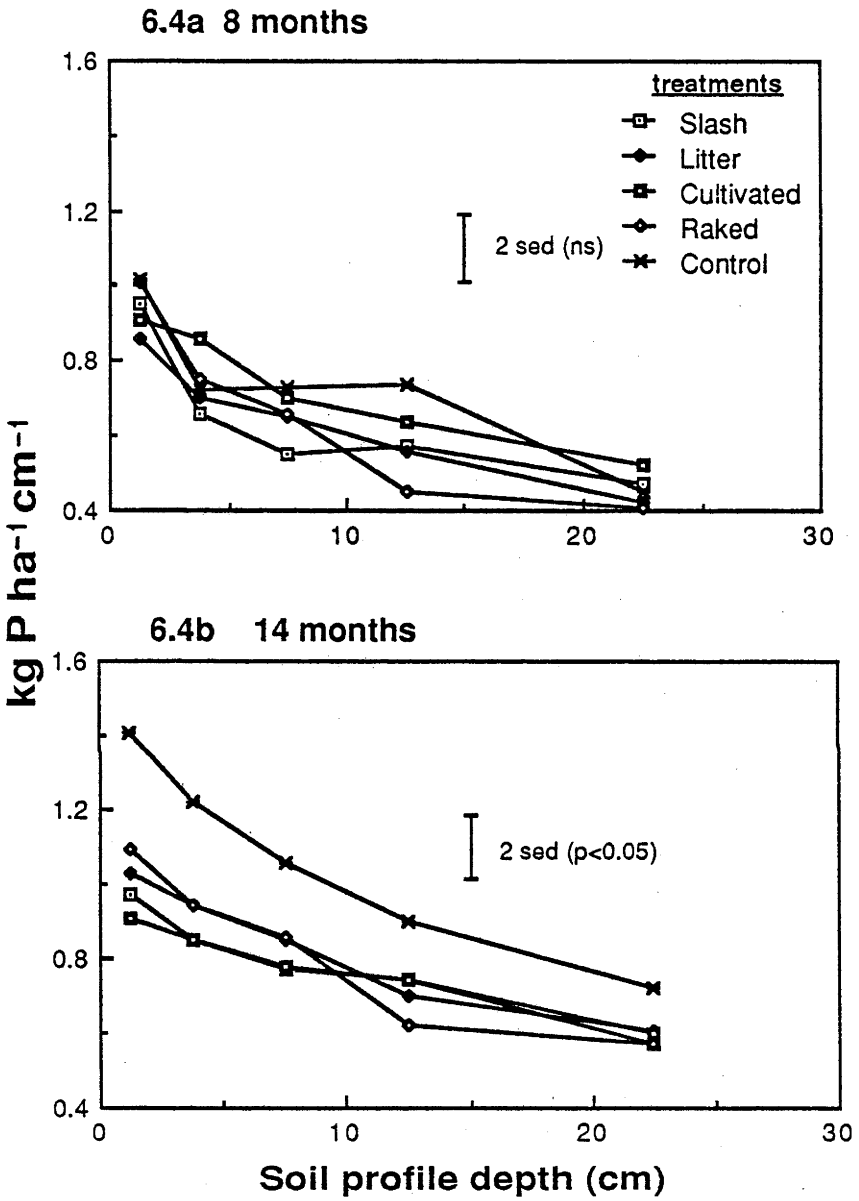


Figure 6.4 Labile organic P in soils of different litter/slash treatments at 8 and 14 months after harvest.

treatments over the entire 30 cm of the profile. It is difficult to attribute this to the treatments. The increase could be a soil variation error.

In order to examine whether there was any leaching of P in the cultivated treatment, a test was done based on the premise that there is a reduction in biological activity in winter, and thus any accumulation of P in the lower part of the profile during winter would mainly be due to leaching by winter rains (compared with autumn which is dry). Soils were sampled from the cultivated treatment during winter, 1985 (12 months after treatment), and labile inorganic and organic P distribution in the profile compared with that in the previous autumn and in the following spring.

Figure 6.5a shows that in winter, labile inorganic P increased in the soil below 5 cm relative to the P values of the previous autumn, while there were no significant differences between seasonal values above 5 cm. In spring, inorganic P levels increased slightly in the surface 0-2.5 cm. This suggests that the conditions in the soil below 5 cm remained favourable for mineralisation during winter. Parker (1962) and Shields and Paul (1972) observed that the rate of decay of plant material on the soil surface was more subject to environmental factors than when incorporated in the soil. Since the amount of P that had been accumulated prior to the onset of winter is not known, leaching may also have contributed to P increase below 5 cm. Labile organic P was not affected by seasonal changes (Figure 6.5b).

The field experiment suggests that there will be some increase in soil inorganic P with time after forest harvesting

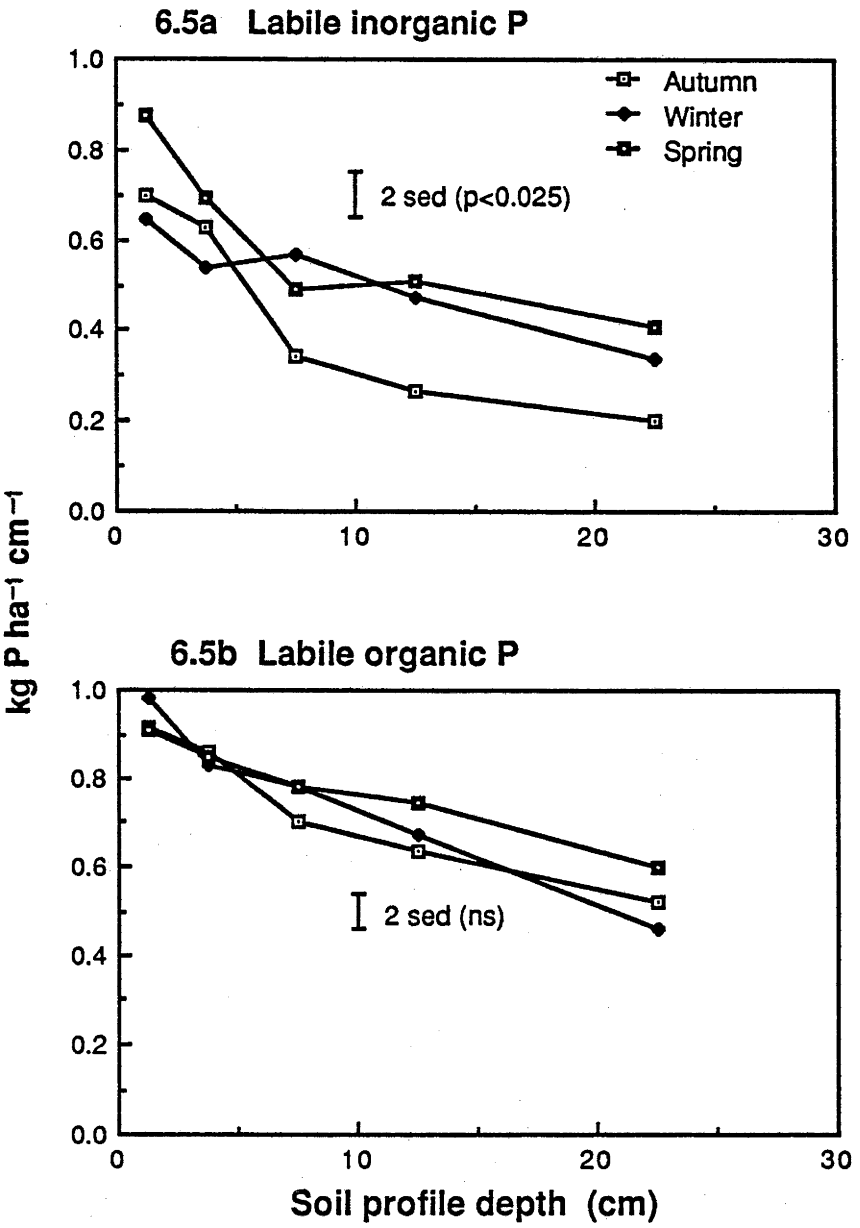


Figure 6.5 Seasonal effects on the profile P content in the cultivated treatment.

irrespective of the management applied to the forest floor. However, the P measured represented the concentration at different sampling points in time, and differences between the treatments may have resulted partly from uptake (in case of control), leaching or microbial immobilisation. As the soil was sandy, only a small fraction of P could have been sorbed. Because of the low rates of P mineralisation observed in any treatment, an experiment was also done using soil cores from the litter and slash treatment plots and incubated under defined environmental conditions for comparison with the field results.

6.3 Laboratory incubation study

The field experiment showed net increase in labile inorganic and organic P after harvest of a radiata pine forest. However, it has often been noted that soil P determined using standard tests can vary appreciably within and between seasons (Harrison, 1979; Kemp *et al.*, 1985). This temporal variation arises due to changes in moisture and temperature, and should be taken into account when interpreting soil test results in field experiments. In the above field experiment, it was suggested that leaching, variable surface exposure and different soil temperatures (surface compared with subsurface) could have influenced soil P content. It was therefore decided to examine inherent differences between treatments under defined environmental conditions and to compare them with results obtained under field conditions.

6.3.1 Experimental procedures

The experiment was based on intact soil cores taken from the slash, litter, raked and cultivated treatments described in Section 6.2.2.

Four months after the start of the field treatments, intact cores (PVC tubes, 5 cm in diameter and 40 cm deep) were brought to field capacity in the field. This was achieved by adding water slightly in excess of that required to bring the soil to field capacity (to minimise leaching) and allowing free drainage. 12 cores per treatment were collected, sealed and incubated at 20°C. At 0, 5 and 7 months incubation, four cores per treatment were divided into 2 depths (0-15 and 15-30 cm) after removing slash and litter from the relevant treatments. The soils were analysed without drying for $NaHCO_3$ extractable (labile) inorganic and organic P.

6.3.2 Results

The results of this experiment are given in Figure 6.6. In all the treatments, there was more P (about twice) in samples from 0-15 cm than in the 15-30 cm, and the difference in labile inorganic P between the two depths increased significantly ($p<0.001$) with time. The increase of inorganic P in the 0-15 cm samples demonstrates higher mineralisation activity in the top 0-15 cm than in the 15-30 cm part of the soil cores.

Figure 6.6 also shows that of the incubated samples,

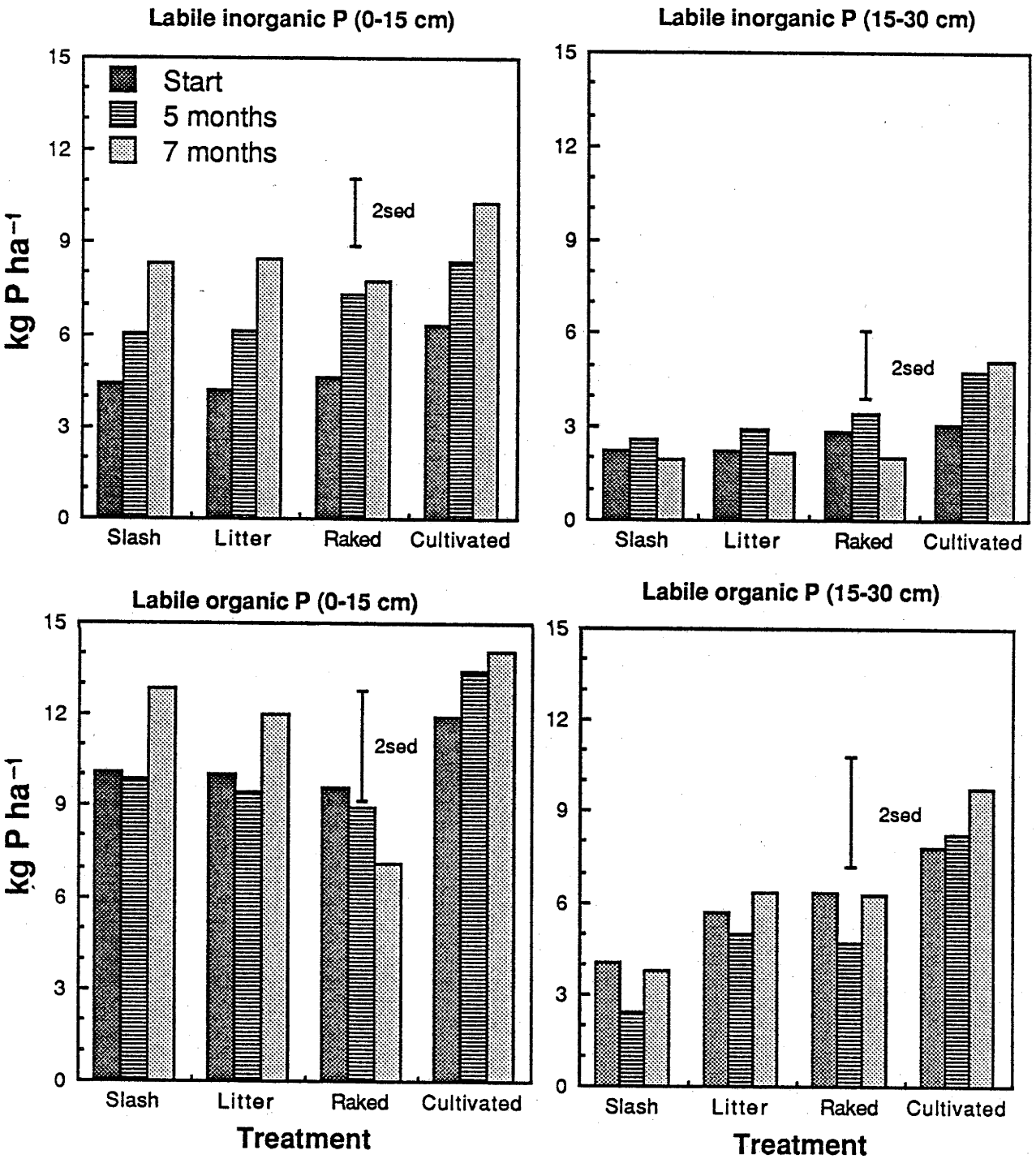


Figure 6.6 Litter/slash treatment effects on labile soil P in laboratory incubated soils.

(treatment*time*depth interaction significant at p<0.05; months=months of incubation)

those taken from the cultivated treatment at both 0-15 and 15-30 cm depths had more P (inorganic and organic) than those from the other treatments at the start of the experiment. There were no significant differences between the other treatments (slash, litter raked) at the start of the incubation. The accumulation of labile inorganic and organic P was significantly different ($p<0.001$) between the litter treatments. At 0-15 cm, inorganic P in the slash, litter and cultivated treatments increased at a faster rate between 5 and 7 months than between 0 and 5 months of incubation. Soil from the raked treatment showed a large increase between 0 and 5 months, and thereafter only a small increase. At 15-30 cm, only the cultivation treatment showed a slight increase in inorganic P with time.

Labile organic P was also higher in soil from the cultivated treatment at the start of the experiment, and it continued to accumulate with time (Figure 6.6). Labile organic P at 0-15 cm from the raked treatment was lower at the start of incubation and continued to decline with time. The slash and litter treatments showed an increase in labile organic P only at the final sampling. At 15-30 cm, labile organic P was again higher in the cultivated treatment and increased slightly with time. Labile organic P in the other treatments declined slightly at 5 months but increased to the original level at 7 months. The soil from the slash treatment had the lowest organic P at this depth.

Table 6.3 shows the rate of change in labile P during 7 months of incubation of soils. Labile inorganic P accumulated in soil from the cultivated treatment at a rate of 0.9 kg P

ha^{-1} $month^{-1}$, which was 3 times faster than that for soil from the raked treatment. Labile inorganic P in soils from the slash and litter treatment soils accumulated at intermediate rates of 0.5 and 0.6 kg P ha^{-1} $month^{-1}$ respectively. The slash and litter treatment soils had the same rate of labile organic P accumulation (0.4 kg P ha^{-1} $month^{-1}$), and this was slightly less than that of the cultivated treatment (0.6 kg P ha^{-1} $month^{-1}$). There was a decline of labile organic P in soil from the raked treatment at a rate of 0.4 kg P ha^{-1} $month^{-1}$.

Table 6.3 Rates of change of labile inorganic and organic P in incubated soils from different litter/slash treatments. Units: kg P ha^{-1} $month^{-1}$ (30 cm depth)

Treatment	Slash	Litter	Raked	Cultivated
Inorganic P	0.5	0.6	0.3	0.9
Organic P	0.4	0.4	-0.4	0.6

6.4 Discussion

Both laboratory and field experiments show an increase in labile inorganic P in all treatments after harvesting a pine forest. As noted earlier, the trees in the control treatment could have utilised mineralised P. Accordingly, the noted increase in inorganic P compared with the 'control' treatment could have resulted from either increased mineralisation or from similar mineralisation as the control but with accumulation in the absence of plant uptake. Nevertheless, it has been suggested that an increase in the temperature and moisture content of the litter and soil surface (Squire and

Flinn, 1981) and the enhanced development of saprophytic microflora following clearfelling (Gadgil and Gadgil, 1978) could accelerate the decomposition of soil organic matter in the field and hence release of P. The accumulation rates of labile inorganic P ranged from 0.3 to 1.2 kg P ha^{-1} $month^{-1}$ and fell within the annual tree P uptake range of 1.7 - 13.8 kg P ha^{-1} yr^{-1} cited by Ballard (1980)

Cultivation appears to have accelerated mineralisation of organic P as inorganic P was consistently higher than in the other treatments. This was supported by a higher accumulation rate of labile inorganic P observed under controlled conditions. Cultivation stimulates mineralisation (Squire and Flinn, 1981) by mixing and therefore exposing previously inaccessible organic matter to microorganisms for decomposition (Dalal, 1977; Chauhan *et al.*, 1981). In the field, the rate of mineralisation could be affected by environmental conditions. Thus the accumulation rate of inorganic P in the field was lowest in the cultivated treatment possibly due to a temperature effect, since the amount accumulated before the winter period was higher than in the other treatments. The inorganic P of the litter released by autolysis could also have contributed to the initial increase in soil inorganic P in the cultivation treatment.

The initial high rate of inorganic P accumulation in the 0-15 cm soil from the raked treatment was not maintained after the 5th month (Figure 6.5), possibly due to the absence of litter and slash as sources of organic P for mineralisation. This was supported by the continuing decline in labile organic P. Losses of up to 21% of organic C (Allegre and Cassel,

1986) and 20% of N (Burger and Pritchett, 1984) from forest site reserves have been observed to result from residue removal. Removal of litter and slash from the Caroline study site would result in about 35% removal of P from the site (30 cm profile).

Except for the incubated soil from the raked treatment, labile organic P slightly increased with time. Sharpley (1985) suggested that during mineralisation a shift occurs from moderately labile and resistant organic P in response to the labile organic P mineralised. Since labile organic P declined (incubated soil) and did not increase (field soil) in the raked treatment, the main source of labile organic P in the other treatments must have been litter and slash retained on site, emphasising the advantage of maintaining slash and litter on site after harvest.

Data from Neilsen *et al.* (1984) on the growth of superphosphate fertilised (and non fertilised) *P. radiata* seedlings showed P uptake of 526.3 mg P per plant (1 m spacing) at the end of 3 years, giving a total removal of about 5 kg ha^{-1}. Data from the same experiment gives about 0.2 kg P ha^{-1} removal in the first 8 months of seedling growth. This suggests that the amount of P accumulated in soil of all four incubation and field treatments, ranging from 2.4 (incubated raked treatment soil) to 9.6 kg P ha^{-1} (field slash treatment soil) calculated for 8 months from Tables 6.3 and 6.4, was enough to meet the requirements of pine seedlings in the initial stages of growth. The field experiment showed that most of the increase in P was in the surface 10 cm of the soil profile where seedling root activity

would be most active. The cultivated treatment would probably encourage deeper root penetration as there was more P down to about 20 cm.

In conclusion, the experiments showed increased labile P in the soil after harvesting a forest. Given that the amount released was in excess of the uptake expected by *P. radiata* seedlings, losses to the plant may be occurring concomitantly through fixation and leaching. This may explain the observation that in many instances the productivity increases attributable to cultivation are short lived (Post, 1974). In view of the excess labile P accumulated in the soil, revision of the strategies of fertiliser P application on the sandy soils of South Australia could be necessary. For example, P fertilisation could be applied at later stages of seedling growth when P requirements are high, instead of pre-planting fertilisation when plant uptake is low and leaching losses may be high.

Management practices that could allow slow P release in the early stages of seedling growth, and higher release at peak nutrient requirements, are therefore appropriate for tree nutrition, and need to be further researched. Slash and litter treatments appear to be favourable over the cultivated and raked treatments in the slow release of P. Where the litter and slash are left undisturbed, less P (and other nutrients) is released early when plant requirements are low, and the rest is held against leaching or sorption. Other beneficial factors associated with retention of litter on site are conservation of soil moisture (Squire *et al.*, 1979) and the protection of soil structure and provision of a more

favourable environment for root development in the surface soil (Farrell, 1982). In this study, there were no differences observed between the slash and litter treatments possibly because the slash was deposited on to the litter where it was physically retained. Cultivation would supply P early, and encourage root development deeper in the profile, but this effect may be short lived. Raking would remove an important source of P from which inorganic P could be continually generated by mineralisation. Turvey and Cameron (1982) have identified this operation (windrowing) as not only contributing to the decline in productivity of subsequent pine rotations, but also increasing the growing cost, as it is an expensive operation. Long term studies are necessary to establish if the short term differences would be maintained, especially between the slash, litter and cultivation treatments.

The experiments in this Chapter and data of Neilsen *et al*, (1984) suggest that the best method for managing the forest floor between crop cycles of radiata pine would be that which aims at retaining the litter and harvest residue on site, and allowing minimal mineralisation until the crop is established. The quantities of P released were small and this raises questions about the importance of mineralisation of P in soils with high sorbing capacity. The following chapter is addressed to this question and to the role of microorganisms in the mineralisation-immobilisation process.

CHAPTER 7 MICROBIAL TRANSFORMATION OF N AND P IN SOIL DURING INCUBATION

7.1 Introduction

7.2 Experimental procedures

7.2.1 Preliminary study of the relationship between N and P in soil after incubation

7.2.2 Microbial transformation of N and P in soil during incubation - the main study

7.3 Results

7.3.1 Changes in soil microbial biomass carbon during incubation of soil

7.3.2 Soil biological activity during incubation

7.3.2.1 Comparison of respiration rates in the untreated soils

7.3.2.2 The influence of different tretments on respiration in the yellow podzolic soil

7.3.2.3 The influence of lime and lime+fertiliser on respiration in the krasnozem soil

7.3.2.4 Conclusions on soil respiration

7.3.3 Changes in soil and microbial biomass N during incubation

7.3.3.1 Comparison of changes in the untreated soils

7.3.3.2 The effect of different treatments on N mineralisation and microbial biomass N in the yellow podzolic soil

7.3.3.3 The effect of lime and lime+fertiliser on N mineralisation and microbial biomass N in the krasnozem soil N

7.3.3.4 Conclusions on soil and microbial biomass N

7.3.4 Changes in soil and microbial biomass P during incubation

7.3.4.1 Comparison of changes in the untreated soils

7.3.4.2 The effect of different treatments on P in the yellow podzolic soil P

7.3.4.3 P transformations in the limed and limed+fertilised krasnozem soil

7.3.4.4 Correction for sorption of inorganic P produced during mineralisation

7.4 Discussion:
Relationships between soil properties, microbial properties and N and P fluxes during incubation

CHAPTER SEVEN

MICROBIAL TRANSFORMATION OF N AND P IN SOIL DURING INCUBATION

7.1 Introduction

The field and laboratory incubation studies on a dune sand described in Chapter 6 showed accumulation of very small pools of labile P during the re-establishment phase, and highlighted the advantages of retaining plant residues and hence improving the labile P pool between crop cycles of radiata pine. Because of the low uptake of P by plants at establishment (Neilsen *et al.* 1984), only a small proportion of the soil organic P may need to be mineralised to provide sufficient inorganic P for growth. However, some of the inorganic P released by mineralisation may become immobilised in microbial cells or chemisorbed or adsorbed by soil. It is therefore important to be able to determine the rate of P mineralisation and to explore the fate of the mineralised P in different soils.

The microbial biomass is the agent of breakdown of organic materials in soils (Vance and Brookes, 1987). The biomass is also a highly labile nutrient reservoir compared with other soil components (Tate, 1984) and thus plays a key role in the cycling of plant nutrients. For example, Brookes *et al.* (1985) calculated that the annual flux of P through the biomass in grassland soils averages 23 kg P ha^{-1}. Therefore factors that control the population structure and activities of microorganisms also control the balance between

mineralisation and immobilisation of P in soil.

The work in this Chapter seeks to determine linkages between mineralisation of P, and microbial respiration, microbial biomass C, N and P content, and N mineralisation. It was envisaged that if mineralisation of P could be predicted from biomass population changes, microbial activity or N mineralisation for which measurement techniques are better developed, this would be useful in analysing P dynamics in soils where changes in P fluxes are usually small and difficult to determine. In addition, identification of the dynamics of the short-term processes controlling P mineralisation are essential for improving our understanding of long-term soil nutrient relationships.

7.2 Experimental procedures

An examination of the relationship between N and P mineralised in 5 major soils was examined first in order to select soils with a range of conditions appropriate for the main study.

7.2.1 Preliminary study of the relationship between soil N and P after incubation

In a preliminary experiment, the mineralisation of N and P was examined in 12 soils representing five major soil types (Table 7.1). This was done to help select an appropriate range of soils for the more detailed investigation described in this Chapter. The soils were incubated for 68 days, at 20°C and at near field capacity moisture content. The soils

were analysed for inorganic N extractable in 2*N* KCl, and for resin extractable P (most-labile), and $NaHCO_3$ (labile) extractable inorganic and organic P before and after incubation. The $NaHCO_3$ extractable inorganic P also included resin extractable P.

Table 7.1 Characteristics of the soils used in the preliminary study

Soil	Depth (cm)	pH (KCl)	%C	%N	%P
Siliceous sand	0-5	4.51	1.40	0.05	0.005
	5-10	4.19	0.87	0.03	0.003
Yellow podzolic	0-2.5	4.67	2.39	0.08	0.013
	2.5-5	4.64	1.30	0.04	0.010
Siliceous sand	0-10	4.39	2.16	0.06	0.006
(ploughed)	10-20	4.22	1.01	0.03	0.004
Terra rossa	0-5	4.58	4.82	0.21	0.032
	5-10	4.53	2.59	0.16	0.030
Red podzolic	0-5	4.29	5.68	0.13	0.020
	5-10	4.19	2.49	0.08	0.017
Krasnozem	0-2.5	3.65	16.27	0.39	0.037
	2.5-5	3.73	10.28	0.27	0.034

The relationship between changes in inorganic N and resin extractable and $NaHCO_3$ extractable P during incubation is shown in Figure 7.1. Both inorganic N and $NaHCO_3$ extractable inorganic P increased during incubation. Resin extractable P was relatively steady and labile organic P decreased. The changes in inorganic P were very small (less than 10% of the N increase) and the only significant relationship ($p<0.01$) was a negative one between N and labile organic P.

While mineralisation of both N and P occurred during incubation, the absence of a clear relationship between them may be due to the fact that, unlike N, the end product of P

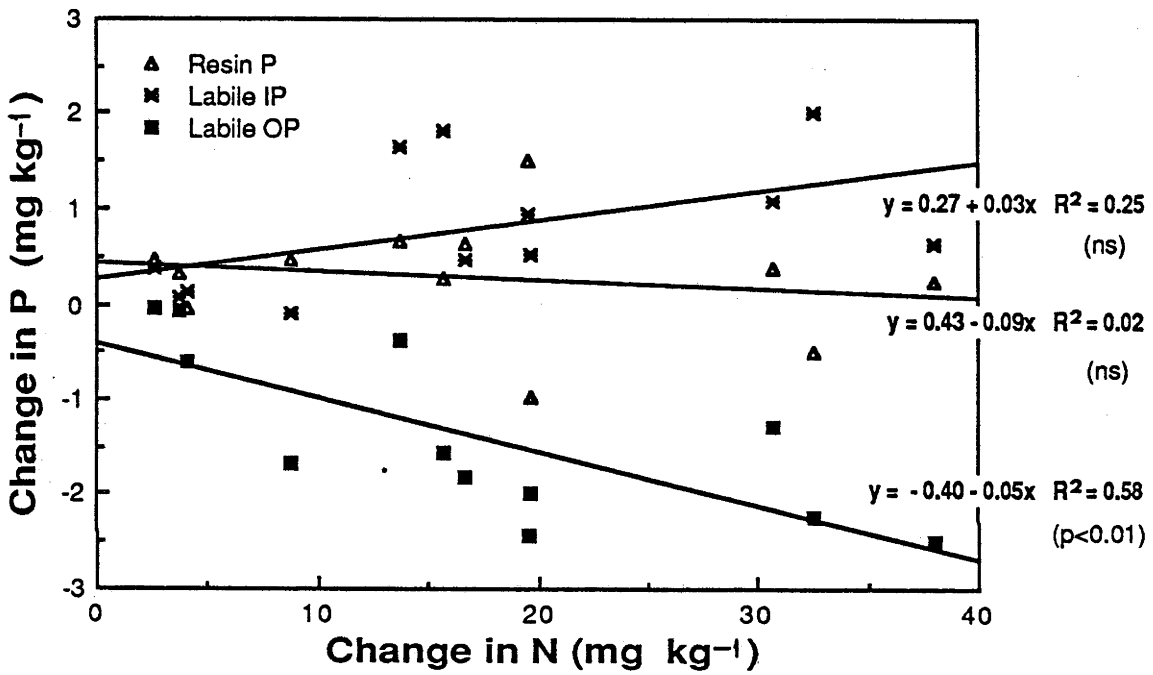


Figure 7.1 The relationship between changes in soil inorganic N and labile inorganic and organic P in different soils after 68 days of incubation. (Means of 4 replicates). Both resin P and $NaHCO_3$ (labile) inorganic P are not corrected for sorption.

IP = inorganic P; OP = organic P

mineralisation can be adsorbed or can precipitate as insoluble phosphates. As the soils were different, the sorption and precipitation reactions would occur at different rates and to different degrees, thus confounding the relationship. The changes of P were not corrected for sorption.

The major experiment reported here used a longer period of incubation (with several samplings), measurements of microbial biomass nutrient changes, and correction for P fixation to elucidate further the relationship between N and P mineralisation. Resin extractable P was not measured separately because it showed least response to incubation in the preliminary study. The forms of P described are the 30 minute $NaHCO_3$ extractable inorganic (labile inorganic), organic (labile organic) and biomass P. This extraction procedure has been used widely in the study of microbial biomass P (e.g. Brookes *et al.*, 1982; McLaughlin 1986; McLaughlin and Alston, 1986), and is described in Section 3.3.4. Although P in the 30 minute $NaHCO_3$ extracts is also referred to as labile inorganic and organic P in Chapters 7 and 8, it is appreciably less than that in the 16 hour $NaHCO_3$ extracts (Section 3.3.4.1). The major soils selected were the yellow podzolic, the red podzolic and the krasnozem.

7.2.2 Microbial transformation of N and P in soil during incubation - the main study

The experiments in the main study entailed temporal measurements of microbial activity (O_2 uptake and CO_2 production), microbial biomass C, N and P, and changes in soil mineral N and labile P fractions.

Soils and soil treatments were chosen so as to provide a wide range of conditions under which temporal changes in N, P and the microbial biomass could effectively be compared. The yellow podzolic (YP), red podzolic (RP) and krasnozem (K) soils were used in the study. In addition, soils were collected from the yellow podzolic-sewage treated plot (Figure 4.1, Plot 10 S) and the yellow podzolic-irrigated + liquid fertilised plot (Figure 4.1, Plot 2 IL) from the Biology of Forest Growth Experiment. The fertilised plot had received weekly inputs of a balanced nutrient solution (equivalent to 330 kg N and 41 kg P ha^{-1} yr^{-1}) for two years prior to sampling. The fertiliser (YPF) and sewage sludge (YPS) treatment soils were included in this study to examine the long term effects of these treatments on the soil biomass, N and P.

Some of the properties of the soils used are shown in Table 7.2. The krasnozem had the highest soil C, N and P and lowest pH, and the yellow podzolic soil had the lowest C, N and P and highest pH. The red podzolic soil was intermediate in these properties. The C:N ratio was lower in the yellow podzolic soil than in the red podzolic and krasnozem soils, in the same way as the C:organic P ratio. The soils were also known to have different P sorption capacities (Section 3.3.3.2), with the krasnozem soil having the highest sorption capacity and the yellow podzolic having the lowest. Compared with the untreated yellow podzolic soil, the sewage sludge treatment increased soil pH, C, N and P, whereas except for P the fertiliser treatment decreased them.

The soils were collected from an average depth of 0-5 cm,

after removal of the litter layer, using a spade. Microbial activity was expected to be highest in this part of the soil profile. For each soil, four random sites were sampled and the soil mixed, sieved through a 4.75 mm sieve and stored at 4°C until used, usually within 7 days.

Table 7.2 Soils and treatments for the C, N and P mineralisation study

Soil	MC as % of FC	pH (water)	%C	%N	C:N	OP mg/kg	C:OP	P_t mg/kg
YP	70.4	5.91	2.1	0.09	22	85.5	246	143
YPS	76.1	6.94	2.4	0.12	20	87.5	274	292
YPF	67.4	5.19	1.4	0.07	22	36.3	386	161
RP	84.8	5.20	3.2	0.11	28	98.6	324	232
K	71.0	4.80	9.3	0.28	33	256.2	363	368

MC = moisture content; FC = field capacity
OP = organic P; P_t = total (Kjeldahl) soil P

A number of treatments were imposed on these soils. Glucose was added to soil from the fertilized plot (YPF+G), and lime and lime+fertiliser (N and P) were added to the krasnozem soil (KL and KLF respectively). The three treatments were designed to stimulate microbial activity. Glucose, N and P were added in solution forms (4 ml 100 g^{-1}) as glucose, $(NH_4)_2SO_4$ and KH_2PO_4 respectively at the rates of 3 g glucose kg^{-1}, 200 mg N kg^{-1} and 50 mg P kg^{-1}. Liming was with 20 g $CaCO_3$ kg^{-1} in powder form. The chemicals were added and the materials mixed thoroughly, just before incubation.

Samples equivalent to 100 g oven dry soil were incubated at 25°C (±2) and at the moisture contents at which

they were collected from the field, as they were about 70-80% of field capacity moisture content (Table 7.2). Soils were incubated in PVC tubes, 13 cm long and 5 cm in diameter, which were covered at both ends with plastic film to minimise water loss. A 2 mm hole in the film at the top end allowed gaseous exchange. Four replicates of each treatment were incubated for 0, 8, 20, 40, 60 and 90 days. Four replicates of 200 g of soil from each treatment were used for measurement of soil respiration studies (O_2 uptake, CO_2 evolution) during 157 days.

After each incubation period, the soils were analysed for:

1. Cumulative soil respiration (Sections 3.4.4)
2. Microbial biomass C, N and P (Sections 3.4.3, 3.4.2, and 3.3.4 respectively)
3. Inorganic N (Section 3.4.2)
4. $NaHCO_3$ (30 minute) extractable P (Section 3.3.4.1).

All analytical methods have been described in Chapter 3. In order to retain clarity of figures, the highest standard error alone is shown for each curve at the point for which it was calculated.

7.3 Results

The effects of different soils and of soil treatments are discussed in the order: Soil biomass C, Soil biological activity, Soil and microbial N, and Soil and microbial P. The final section draws together, and discusses, work undertaken

in the different sections.

7.3.1 Changes in microbial biomass carbon during incubation of soil

The chloroform fumigation-incubation method (CFIM), described in section 3.4.3, was used for the quantification of the microbial biomass C. This was to enable determination of the relationship between the biomass as a transforming agent and as a source and sink for P and N during soil incubation. Figure 7.2a shows that biomass C was highest in the krasnozem (about 175 mg C 100 g^{-1} soil), less in the red podzolic soil and lowest (about 25mg C 100 g^{-1} soil) in the yellow podzolic soil. This is in the same order as soil organic C (Table 7.2). Biomass C in the yellow podzolic soil increased slightly at the 20th day, but was essentially steady throughout the incubation period. In the red podzolic and krasnozem soils, it increased slightly during incubation. The stability of biomass C in the untreated soils suggests presence of a high proportion of resting forms of organisms with a slow turnover. Investigations to explain such slow turnover have pointed to the presence of toxins, such as ammonia in alkaline soils (Ko *et al.*, 1974), ethylene in anaerobic soils (Smith and Cook, 1975), and Al ion (Ko and Hora, 1972), as possible limits to microbial growth in soils. However, Lockwood and Filonow (1981) suggested that substrate deprivation, particularly readily utilisable C (Barber and Lynch, 1977; Lockwood, 1977; Paul and Voroney, 1980) is the main restriction to microbial growth in soils. This

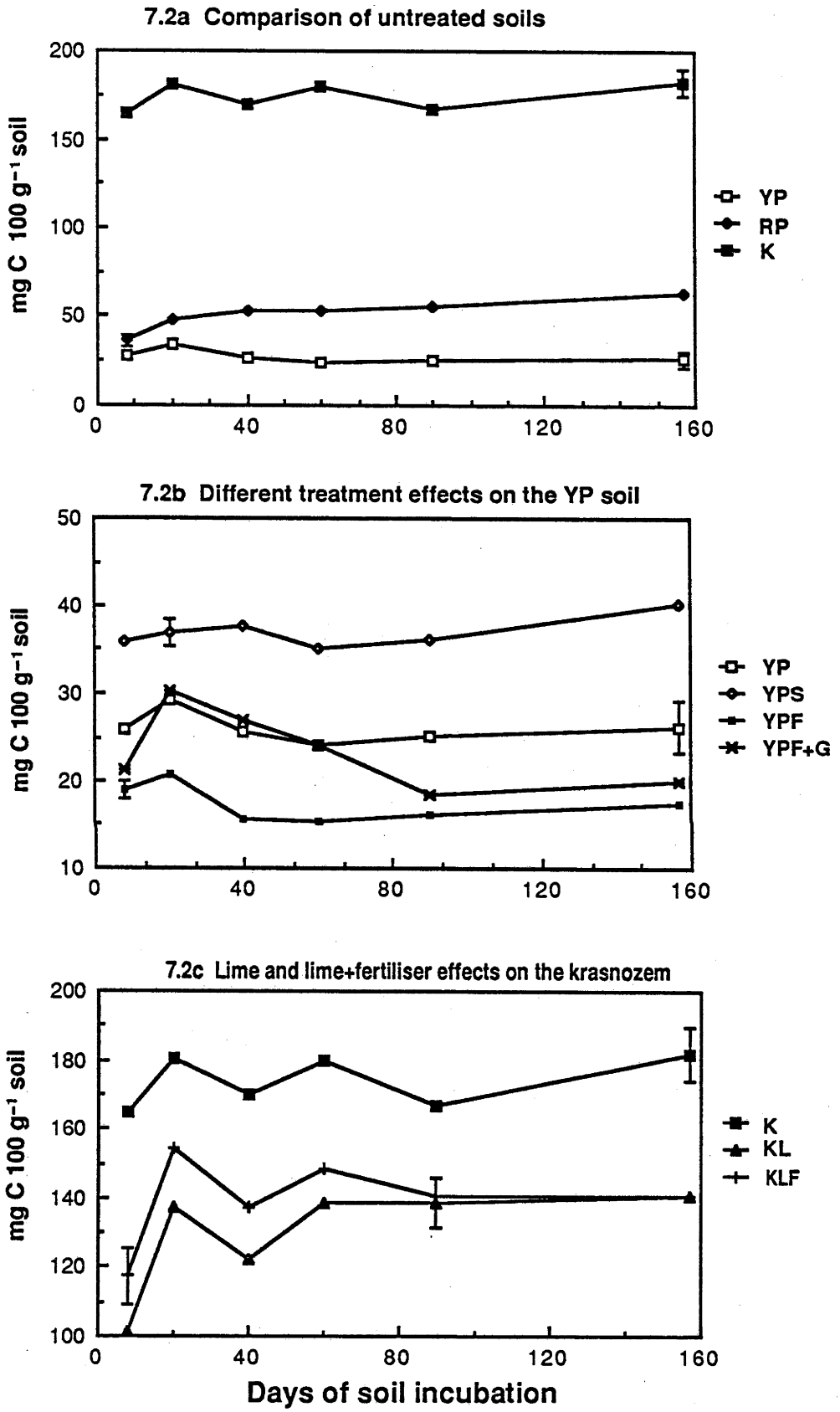


Figure 7.2 Changes in soil microbial biomass C during soil incubation (as measured by CFIM).

(Explanation of legend is given on the following page).

Legend to Figure 7.2 in full

YP = Yellow podzolic soil, untreated
RP = Red podzolic soil, untreated
K = Krasnozem soil, untreated
YPS = Yellow podzolic soil, sewage sludge (field) treatment
YPF = Yellow podzolic soil, fertiliser (field) treatment
YPF+G = Yellow podzolic soil, fertiliser (field) treatment and glucose (laboratory) amended
KL = Krasnozem soil, limed (laboratory)
KLF = Krasnozem soil, limed+fertilised (laboratory)

These abbreviations have been used on other illustrations in Chapters 7 and 8.

suggestion is consistent with the observed increase in microbial biomass C and respiration resulting from the glucose treatment and described later. Humic materials present in soil do not serve as a major source of energy, since their turnover rate is very low (Barber and Lynch, 1977).

The effects of different treatments on biomass C in the yellow podzolic soil are shown in Figure 7.2b. The glucose treatment appears to support the substrate deprivation hypothesis since it increased biomass C above the level of its control (YPF) treatment for at least 90 days. In more intensive measurements on microbial growth rate in glucose-amended soil, Behera and Wagner (1978) observed that rapid microbial multiplication occurred within the first 24 hours after glucose addition, followed by a stationary phase and finally by about 52 hours, a period of slow decline. The early peak in biomass C was therefore attributed to the fast decomposition of the readily available glucose C and its partial incorporation into microbial tissue. After utilisation of the glucose C substrate, utilisation of non renewable microbial products resulted in a slow decline in biomass C to the level of the unammended soil between 90 and 157 days (Figure 7.2b). Because fertiliser and sludge application had been made to the soil three years before sampling (Biology of Forest Growth Experiment), their effects on microbial biomass C had stabilised, and as with the untreated soils there was little change during incubation (Figure 7.2b).

Figure 7.2c shows that liming and liming+fertilisation decreased biomass C overall by more than 20% of that of the

control. After 8 days the reductions were much greater (30-40%). Liming and liming+fertilisation could have influenced microbial composition in the krasnozem soil. Microbial biomass of acid forest soils is predominantly fungal (Pritchett, 1979), and therefore, the very low biomass C at the 8th day may have resulted from the shift of microbial populations from fungi through actinomycetes to bacteria as soil pH increased (Haynes, 1984). By the 20th day, biomass C had increased to a higher level (Figure 7.2c) as the new microbial populations established themselves. Liming alone initially reduced biomass C more markedly than liming+fertilisation possibly because the pH rise with fertiliser addition was less than for lime alone.

In a separate experiment, pH was measured in both the limed and limed+fertilised krasnozem soil at different periods of incubation, ranging from 6 h to 32 days after treatment. The results are given in Figure 7.3. Liming increased soil pH from the pre-liming value of 4.8 to 7.1 within 6 h, and the maximum pH of 7.4 was reached at the 8th day. After 8 days, pH declined slowly. Liming and fertilisation also increased soil pH substantially, but by about 0.5 of a unit less than liming alone, possibly as a result of the acidifying effects of the ammonium salt (Thompson *et al.*, 1983).

In conclusion, the quantity of soil microbial biomass as measured by biomass C is partly dependent upon the amount of soil organic C; there was more biomass C in soils with high organic matter. However, biomass C as a percent of total soil C varied little (1.3, 1.5, 1.5, 1.1, and 1.8 for YP, YPS, YPF, RP and K respectively). The marked increased growth of the

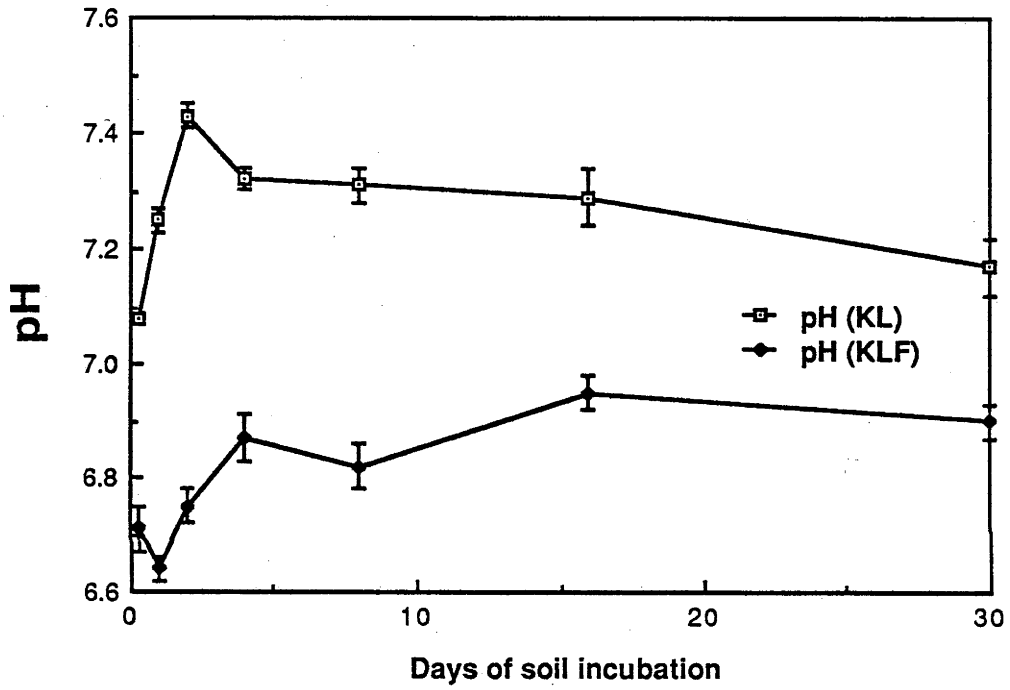


Figure 7.3 Changes in soil pH (1:5 water) after liming and liming+fertilising a krasnozem soil. pH before liming was 4.80.

organisms brought about by addition of substrate C indicated that available C in soils limits microbial growth and activity (see Section 7.3.3 on respiration). Liming the krasnozem reduced soil biomass C probably by killing organisms that could not withstand the increased pH.

7.3.2 Soil biological activity during incubation

The majority of microorganisms obtain their energy through oxidation-reduction reactions in which the oxidation is accomplished through dehydrogenation with a subsequent transfer of hydrogen (or electrons) through a series of mediators to its final acceptor (Stevenson, 1964). In biological oxidation (termed respiration) molecular oxygen acts as the ultimate hydrogen acceptor, and the net result in the oxidation of organic substances is the release of energy, CO_2 and water. Therefore the respiration of microorganisms has been used as an index of soil microbial activity (Neller, 1922; Stotsky and Norman, 1961; Raison, 1976; Raison and McGarity, 1980b) usually by measurement of O_2 consumed by, or CO_2 evolved from, known quantities of soil (Stotsky, 1965). In the following experiments, respiration was measured by both O_2 uptake and CO_2 production (Section 3.4.4).

CO_2 production by soil in respirometers was determined after 34, 45, 60, 89, 128 and 157 days, and respiratory quotients (RQ) were also calculated at these times (Section 3.4.4). CO_2 was also measured over 10 day periods in the control (unfumigated) soils during biomass C determinations (Section 3.4.3). O_2 uptake by soil in respirometers was

measured daily (Section 3.4.4). Together these data provided the general trend for soil biological activity for the whole incubation period.

7.3.2.1 Comparison of respiration rates in the untreated soils

Cumulative O_2 uptake and CO_2 production rates are shown in Figures 7.4a and 7.4b. The three soils had significantly different respiration rates, with the krasnozem having the highest rate followed by the red podzolic and yellow podzolic soils, corresponding to the order for biomass C content (Figure 7.2a). For all three soils, both O_2 consumption rate (Table 7.3) and CO_2 production rates (Figure 7.4b) were highest at the beginning of the experiment and decreased with time of incubation. Because there was no appreciable increase in biomass C with incubation (Figure 7.2a), it is likely the initial high rates of respiration were due to higher metabolic activity following an increase in substrate availability as a result of soil mixing during field sampling and preparation. Chauhan *et al.* (1981) observed stimulated microbial activity with monthly mixing of soil. The gradual decline in respiration was due to depletion of the substrate that had been brought into microbial contact through disturbance.

The respiratory qoutients (RQ) (Figure 7.4c) show a three stage substrate depletion trend. The RQ was about 0.7 - 0.8 during the initial 40 days of incubation, after which it steadied at 0.8 - 0.9, and then increased to about 1.0 in the later stages. This suggests changes in the nature of organic compounds metabolised during incubation. The similar RQ

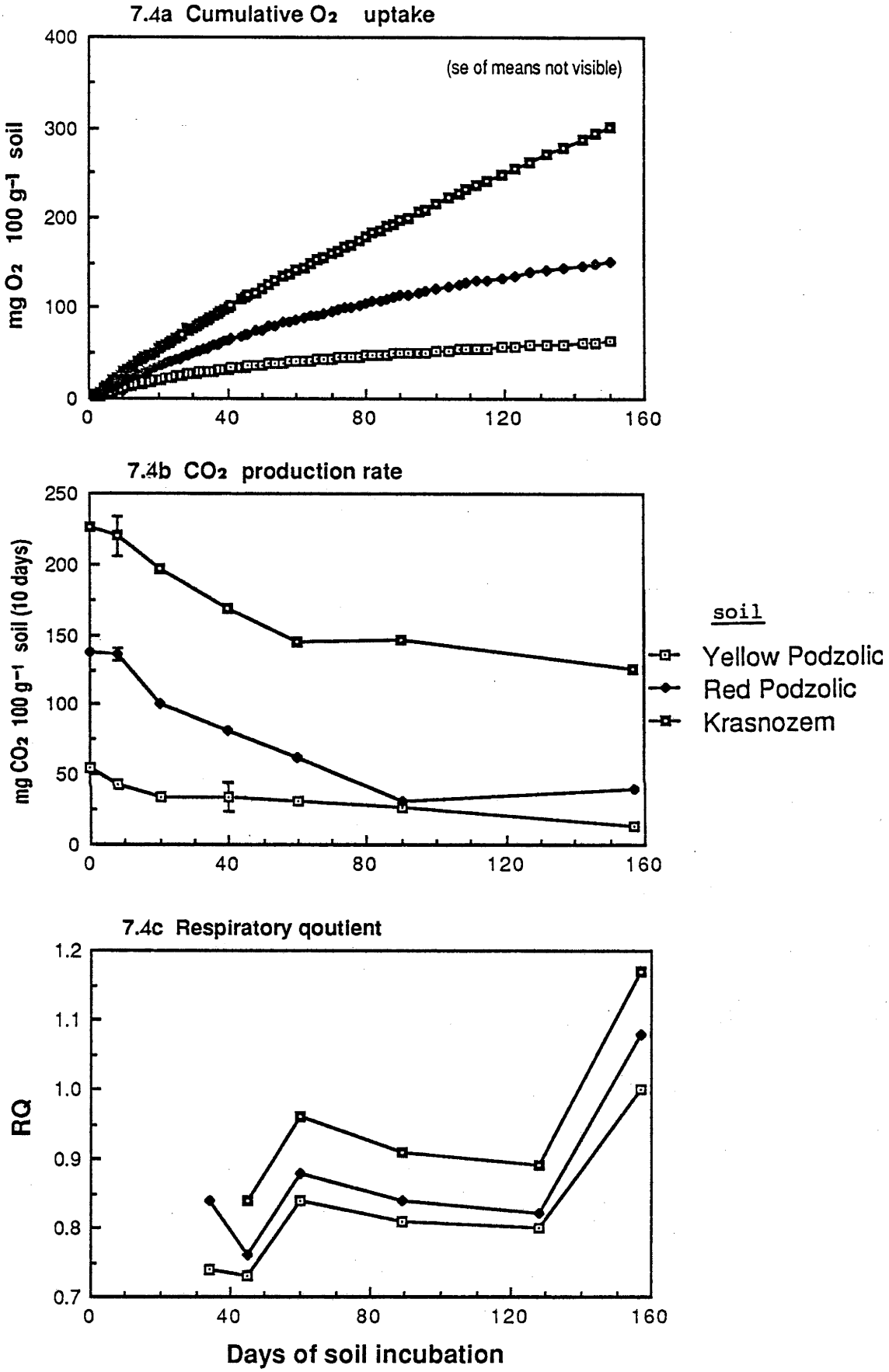


Figure 7.4 A comparison of respiration activity in the untreated soils during incubation

trends suggest that organic compounds degraded in the three soils were of similar nature, although the RQ values were always slightly higher in the krasnozem soil.

7.3.2.2 The influence of different treatments on respiration in the yellow podzolic soil

There were clear differences in respiration between the variously treated soils (Figure 7.5). In the glucose amended (fertilized) soil, O_2 uptake (Figure 7.5a, Table 7.3) and the CO_2 production rates (Figure 7.5b) increased by 4 - 5 fold in the initial days of incubation. The very rapid respiration lasted only for about 3 days (Table 7.3) after which it slowed down. Peak respiration rates have been observed within the first two days after glucose addition (Behera and Wagner, 1974; Wilson and Griffin, 1975). Since biomass C peaked after 10 days (Figure 7.2b), initial vigorous respiration was partly due to greater activity of the existing microbial populations. Microorganisms in the early phase of growth are known to respire about 2.5 times more CO_2 per unit of biomass C than organisms in late or stationary phases of growth (Anderson and Domsch, 1973). Behera and Wagner (1974) observed that glucose was utilised primarily by bacteria, and that fungal growth increased after bacterial counts began to decline. The breakdown of microbial products and cells that had been synthesised with the addition of glucose C acted as a steady source of C so that respiration was always more than double that in the control (fertilised) soil (Figure 7.5b and Table 7.3).

Comparing other treatments, respiration rates were

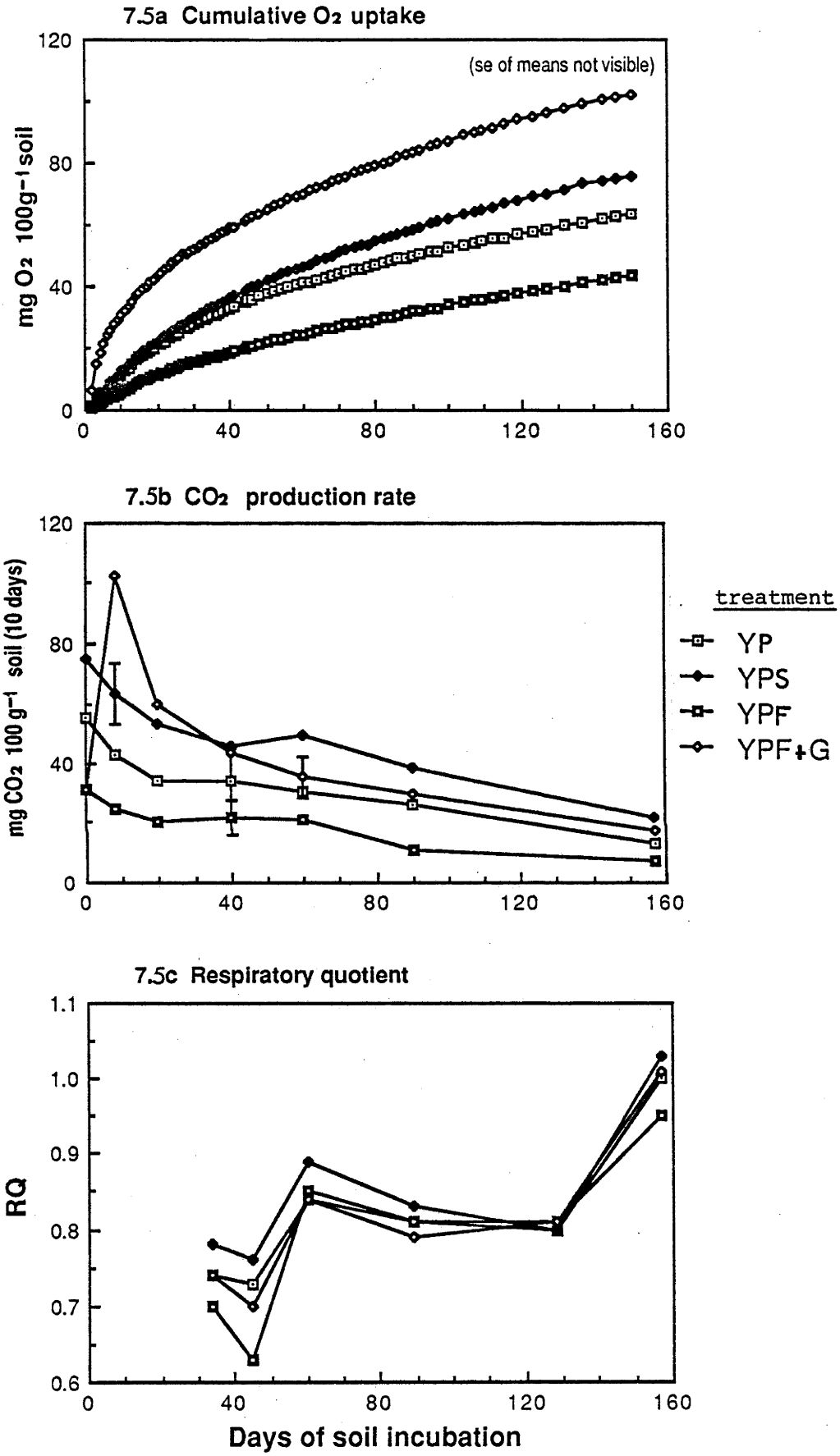


Figure 7.5 The effect of different treatments on respiration in a yellow podzolic soil

Table 7.3 O_2 uptake rates for selected days. Units: mg O_2 h^{-1}

Treatment	Days of incubation									
	2	3	4	5	6	8	20	40	60	90
YP	0.07	0.08	0.06	0.06	0.05	0.05	0.03	0.02	0.02	0.01
YPS	0.07	0.08	0.08	0.07	0.06	0.06	0.04	0.03	0.03	0.02
YPF	0.03	0.04	0.04	0.03	0.03	0.02	0.02	0.02	0.02	0.01
YPF+G	0.34	0.37	0.22	0.15	0.10	0.07	0.05	0.06	0.04	0.02
RP	0.12	0.11	0.10	0.09	0.08	0.08	0.07	0.06	0.04	0.03
K	0.20	0.18	0.16	0.14	0.14	0.12	0.12	0.13	0.10	0.07
KL	0.36	0.32	0.24	0.24	0.20	0.20	0.18	0.14	0.08	0.04
KLF	0.32	0.29	0.28	0.20	0.19	0.18	0.15	0.12	0.07	0.04

highest in the sludge treated soil, followed by the control and and the fertilized soils. This was the same order as for soil organic C and biomass C.

The RQ trend for the glucose amended soil (Figure 7.5c) was similar to that for the other treatments (Figure 7.4c), suggesting a shift from utilisation of easily decomposable substrate (comparable to glucose) in the early stages of soil incubation, to more reduced compounds during later stages.

7.3.2.3 The influence of lime and lime+fertiliser on respiration in the krasnozem soil

The effects of lime and lime+fertiliser on soil respiration is shown in Figure 7.6. There was an immediate increase in O_2 uptake (Figure 7.6a; Table 7.3) and CO_2 production (Figure 7.6b) after liming. This was in contrast to the reduction in biomass C after liming (Section 7.3.2). Lime+fertiliser had slightly less effect compared to lime alone, and this may have been due to the lower pH increase in the latter treatment (Figure 7.3). Table 7.3 shows that peak O_2 uptake occurred within the first 2 days of incubation and remained above the control for about 40 days. After about 60 days both O_2 uptake and CO_2 evolution declined below that of the untreated soil. CO_2 production may have been exaggerated, particularly in the first half of the experiment by the liberation of CO_2 from the $CaCO_3$ used for liming, according to the equation:

$$CaCO_3 + H^+ \rightarrow Ca(OH)^- + CO_2$$

This extra CO_2 would have been absorbed by the KOH traps.

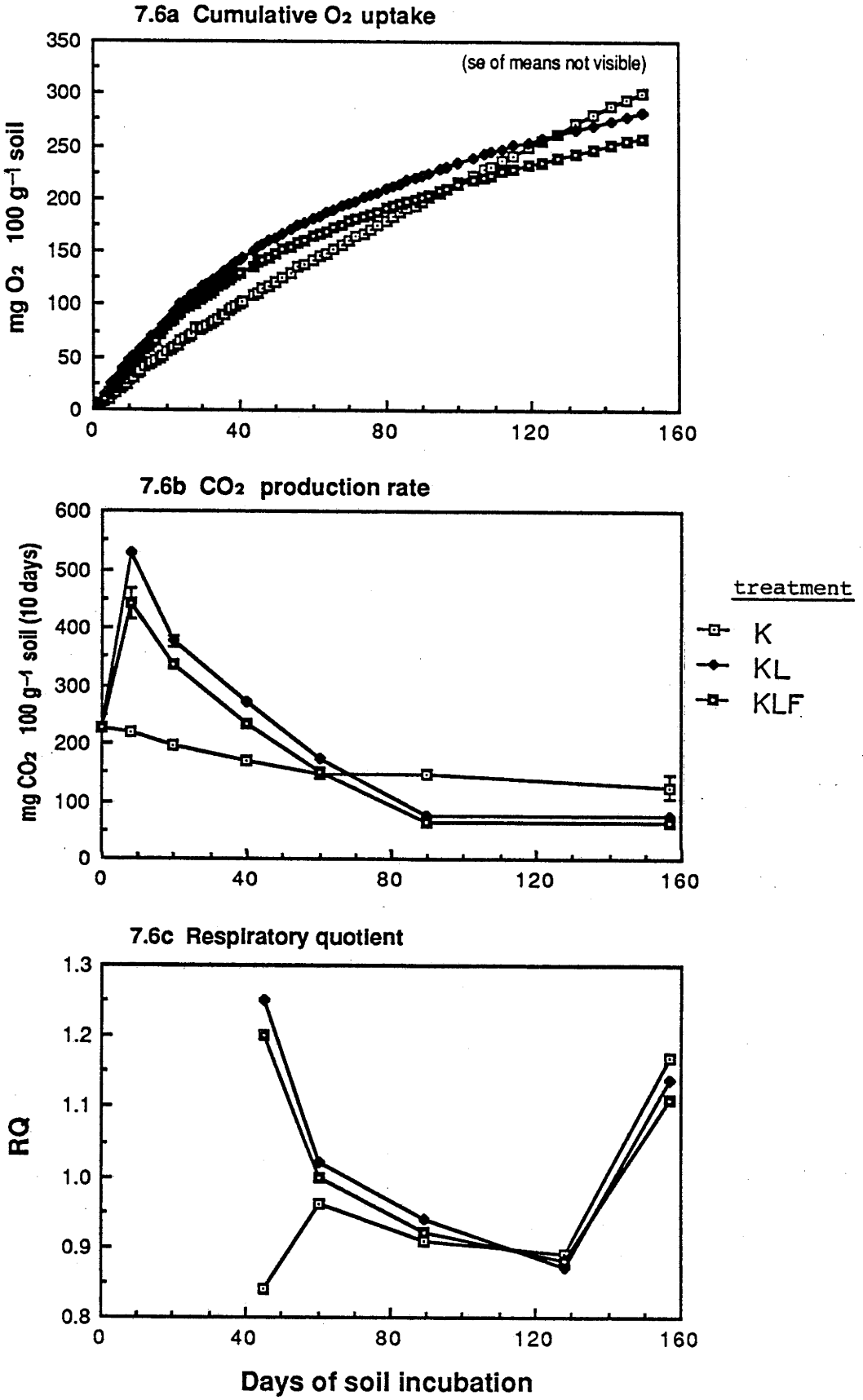


Figure 7.6 The effect of lime and lime+fertiliser on microbial respiration in a krasnozem soil.

Abiotic CO_2 interference has been observed in calcareous soils during the chloroform fumigation method for measuring soil biomass (Jenkinson and Powlson, 1976b). O_2 uptake is therefore the most reliable short term indicator of respiration in limed soils.

Eventually, substrate depletion resulted in respiration below that of the control. Substrate depletion could occur in two ways. Firstly, it is possible that the new microbial population established after liming was capable of decomposing the biomass killed by liming but was incapable of much mineralisation of non-biomass soil organic matter after the flush. Secondly, liming has been observed to reduce solubility of soil organic C (Raison and McGarity, 1980a). Plate 7.1 shows the amount of organic matter (dark colouring) extractable by $NaHCO_3$ during P extraction of the limed and non limed krasnozem. The colour was less intense in the extracts from the limed krasnozem than in the extracts from the non limed krasnozem, which is in support of the observation by Raison and McGarity (1980a). The mechanism that reduces solubility of the organic C may lead to its reduced availabilty for microbial respiration.

The effect of $CaCO_3$-derived CO_2 also appeared to affect the calculated respiratory quotients. Because of high rates of CO_2 production, more than $^2/_3$ of the KOH (CO_2 trap) was neutralised in the first two measurements and RQ could not be calculated for these periods. For alkali solutions to be efficient absorbers, no more than $^2/_3$ of the alkali should be neutralised (Stotsky, 1965). Figure 7.6c shows that the RQ for the two missing values would probably have been greater

Plate 7.1 The effect of liming on the amount of $NaHCO_3$ extractable organic matter from the krasnozem soil. The darker the extract, the higher the organic matter content.

than unity as the high values on the 45th day for the lime and lime+fertiliser treatments was attributed to CO_2 from $CaCO_3$. The RQ values of the incubation periods after the 89th day were the same as for the control implying that the stepwise substrate use in the limed soil was the same as in other soils and treatments.

7.3.2.4 Conclusions on soil respiration

Like biomass C, respiration rate was in part dependent on the initial total soil C content; there was higher respiration in soils with high organic matter. Respiration rates following disturbance, glucose addition and liming, did not relate to biomass C. Sparling *et al.* (1981) observed that biomass C calculated by fumigation-incubation methods was generally less than that estimated by substrate-induced respiration in glucose ammended soils, and lack of comparability between the two methods has also been reported by Ross *et al.* (1984). Behera and Wagner (1974) suggested that during cell growth, substrate utilisation rate is not proportional to microbial numbers (mass). Therefore, respiration rate cannot be used to estimate biomass C in such situations.

The effect of lime on the krasnozem soil and the observed changes in RQ values during incubation suggest that measurement of respiration by O_2 uptake is more reliable than measurement of CO_2 production.

7.3.3 Changes in soil and microbial biomass N during incubation

7.3.3.1 Comparison of changes in the untreated soils

Figure 7.7a shows that inorganic N increased in all three soils during incubation. Whereas the soils had about the same inorganic N at the beginning of the experiment, it accumulated at a faster rate in the yellow podzolic soil during the first half of the experiment. The reverse was true after 90 days. This implies that the yellow podzolic soil contained more readily mineralisable organic N substrates (soil had a lower C:N ratio, Table 7.2), and exhibited rapid mineralisation during the initial 80 days, after which substrate depletion occured. The amounts of inorganic N mineralised were neither related to the initial quantity of soil N (Table 7.4), nor to the cumulative O_2 uptake (Figure 7.4a).

Table 7.4 Inorganic N accumulated during 157 days of incubation as % of initial total N

	Treatment							
	YP	YPS	YPF	YPF+G	RP	K	KL	KLF
Total N (mg kg^{-1})	0.9	1.2	0.7	0.7	1.1	2.8	2.8	2.8
Mineralised N (%)	8	7	10	9	6	3	10	8

Table 7.5 shows that most of the inorganic N in the yellow podzolic soil was converted into the NO_3^- form while all the inorganic N in the krasnozem was in the NH_4^+ form. For the red podzolic soil, most of the inorganic N was in the NH_4 form during the initial 40 days after which nitrification

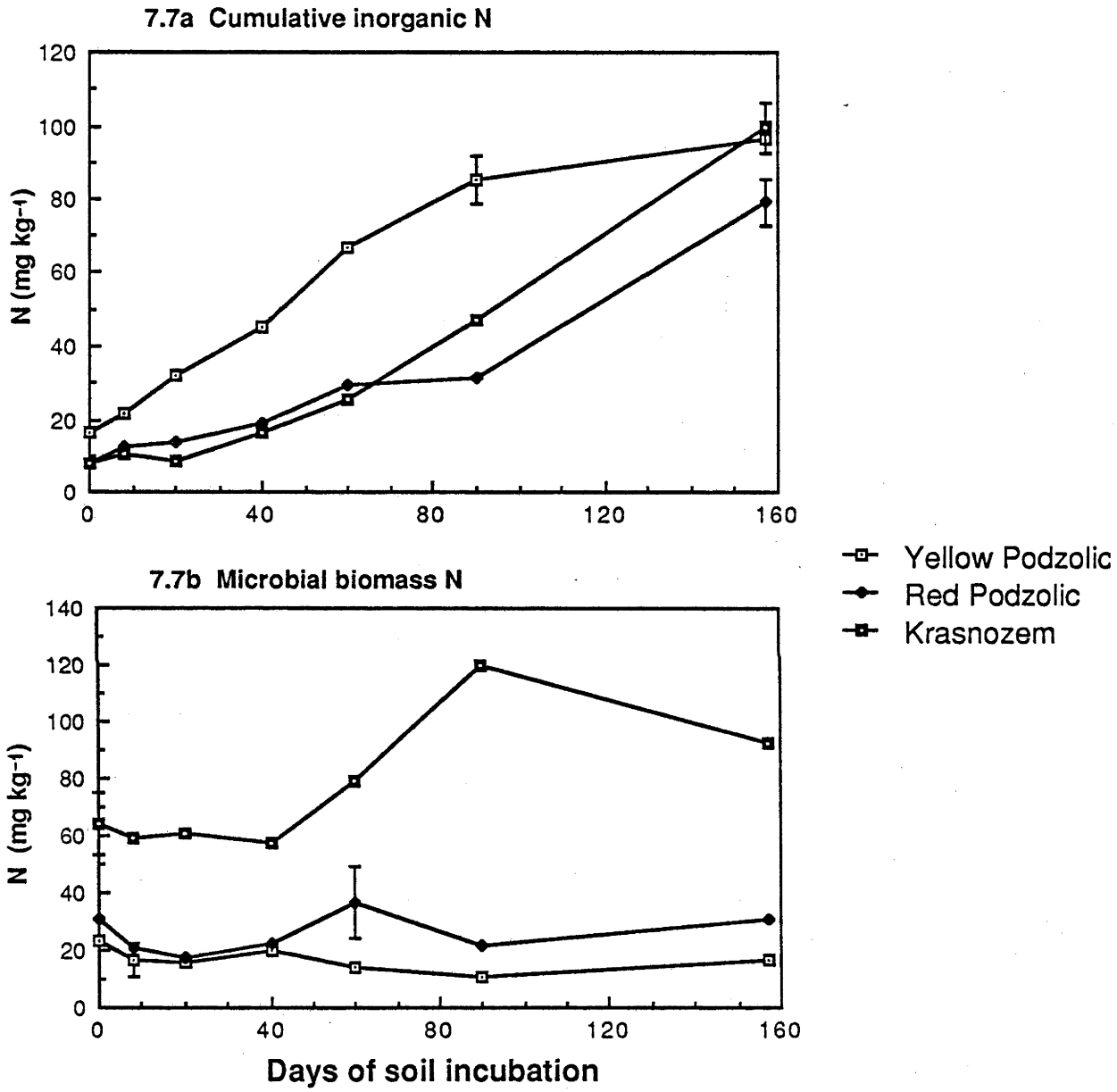


Figure 7.7 Changes in soil inorganic N and microbial biomass N during incubation of the untreated soils.

became dominant.

Table 7.5 NO_3-N as % of N accumulated in the untreated soils during incubation. Means of 4 replicates.

Soil	Days of soil incubation						
	0	8	20	40	60	90	157
Yellow Podzolic	31.5	82.0	91.9	94.9	96.0	98.5	95.9
Red Podzolic	00	00	00	0.1	65.1	90.0	93.4
Krasnozem	00	00	00	00	00	0.1	00

Ammonification is carried out by a broad spectrum of heterotrophic organisms, and nitrification mostly only by a few autotrophic bacteria (Mengel, 1985). Acid conditions tend to inhibit nitrifying bacteria (Stevenson, 1964), so that the low pH of the krasnozem soil may have been the reason for the lack of nitrification. Schmidt (1982) has summarised observations that indicate an arbitrary lower limit for nitrification of pH (water) 4.0, obvious nitrification in the pH range of 4 to 6, and a pH independent nitrification range of 6 to 8. The pH of the krasnozem was 4.81 (Table 7.2) but showed no nitrification. The effect of acidity may be an expression of Al toxicity (Brar and Giddens, 1968) and the high Al in the krasnozem (Table 3.1) may have been the factor limiting nitrification, but this has not been studied here.

Microbial biomass N in the yellow and red podzolic soils was steady during incubation (Figure 7.7b) conforming to biomass C. The krasnozem had the highest microbial N and this remained steady for the first 40 days and then increased for

the remainder of the incubation. No explanation can be given for this increase in biomass N in the untreated krasnozem, except that perhaps organisms of a lower C:N ratio grew as mineral-N became available during incubation as shown in Table 7.6.

Table 7.6 Biomass C:N ratio calculated from biomass C and N determined during incubation of the untreated soils

Soil	Days of soil incubation					
	8	20	40	60	90	159
Yellow podzolic	9.1	10.5	7.6	9.9	12.9	9.1
Red podzolic	10.0	11.3	12.8	9.5	12.9	12.0
Krasnozem	16.2	17.3	17.2	13.2	8.1	11.5

Jenkinson (1976) obtained for a range of organisms C:N ratios of 3.5:1 - 4.2:1 for bacteria, 11.4:1 and 12.8:1 for fungi and 4.4:1 for actinomycetes, while Ross and Tate (1984) obtained a mean value of 9:1 for the ratio of biomass C to mineral N flush after chloroform fumigation of fresh samples. The values in Table 7.6 were estimated using average K_C and K_N values of 0.41 and 0.57 respectively (Section 3.4). They however, suggest that compared with other soils, the dominant microbes in the krasnozem soil during the first half of the incubation period were fungi.

7.3.3.2 The effect of different treatments on N mineralisation and microbial biomass N in the Yellow Podzolic soil

The initial mineral N content of the yellow podzolic soil samples was similar for all treatments (Figure 7.8a). Subsequently, the control and sewage sludge treated soils mineralised in a similar way. Mineralisation was slower in the soils collected from the fertilised field plot, and addition of glucose to this soil immobilised most inorganic N for about 20 days. Immobilisation of N was a result of increased microbial metabolism and growth resulting from the addition of readily assimilable C. Figure 7.8b shows that microbial biomass N in the glucose ammended soil was slightly more than in its control (YPF) during the initial 50 days by about 5 - 10 mg kg^{-1}. This is about half the amount of N immobilised as measured by the difference in soil inorganic N between the control (YPF) and the glucose treatment. It is possible that some of the inorganic N immobilised eventually was converted to more humified organic material during mineralisation-immobilisation turnover. Haider and Farook-e-Azam (1983) also observed rapid immobilisation of N when glucose was added to the soil, and that only about 60% had remineralised in 24 weeks.

The sewage sludge treatment did not increase N mineralisation beyond that of the control (Figure 7.8a), possibly because the most labile components in the sludge had been mineralised in the 3 years since it was applied. The liming effect of sludge also had no effect on N mineralisation, and this is consistent with the results of

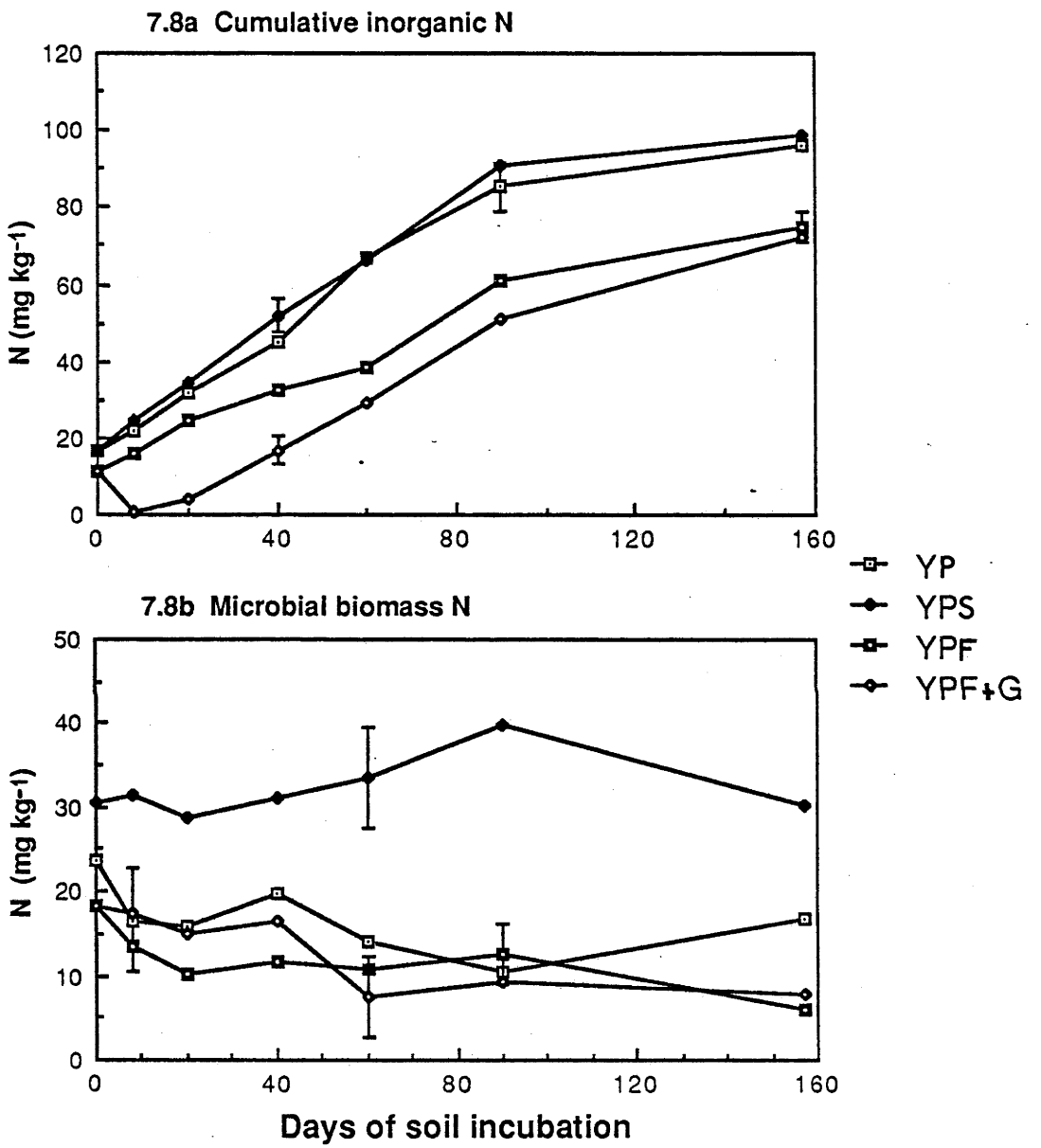


Figure 7.8 Field and laboratory treatment effects on inorganic N and microbial biomass N of a yellow podzolic soil.

Nyborg and Hoyt (1978) who found that lime added in the field 1 to 2 years previous to sampling had little or no effect on the release of mineral N during incubation of two Gray Luvisolic soils. Lower mineralisation in the field fertilised soil (Figure 7.8a) was probably due to its initially lower total N and C contents and the lower pH due to rapid nitrification and leaching over an extended period in the field (R.J. Raison, 1987, Pers. Commun.).

Figure 7.8b shows that the sludge-treated soil had the most biomass N and that this increased slightly during incubation. The fertilised soil had slightly less biomass N than the control, and this decreased in both soils during incubation. Given that biomass N was higher in the sludge treatment than in the control and that both treatments mineralised N at similar rates, it was concluded that mineralisation depended on the amount of labile substrate present rather than on microbial biomass. This conclusion is based on the assumption that the K_N values were the same for both treatments.

Table 7.7 shows that nitrification was dominant in the yellow podzolic soil, irrespective of the treatment.

Table 7.7 NO_3-N as % of N accumulated in different treatments of a yellow podzolic soil.

Treatment	Days of soil incubation						
	0	8	20	40	60	90	157
YP	81.5	82.0	91.9	94.9	96.0	98.5	95.9
YPS	62.1	87.4	94.4	92.9	94.0	96.8	99.1
YPF	85.5	83.4	94.2	93.1	94.9	95.8	98.7
YPF+G	85.5	-	58.5	85.2	92.3	94.1	96.3

7.3.3.3 The effect of lime and lime+fertiliser on mineralisation and microbial biomass N in the krasnozem soil

Figure 7.9a depicts the accumulation of inorganic N in the variously treated krasnozem soils. Except for the period between 8 and 20 days when inorganic N accumulation slowed, the rate of inorganic N accumulation in the lime and lime+fertiliser treatments was higher during the first part of the incubation period, and gradually decreased in the second part. At the end of the incubation period, there was about 3 times more inorganic N in the limed treatment than in the control. Liming modifies soil pH, and consequently affects the composition of the microflora as previously discussed (Section 7.3.2) as well as microbial activities. Mineralisation of N is often increased (Alexander, 1961; Awad and Edwards, 1977; Cullen and Grigg, 1971; Harmsen and van Schreven, 1955; Nyborg and Hoyt, 1978). Mulder (1950) and Nyborg and Hoyt (1978) have observed that increased mineralisation in field treatments may last for 1 to 5 years.

Subtraction of the amount of added fertiliser N (200 mg kg^{-1}) from the inorganic N accumulated in the lime+fertiliser treatment shows less net N mineralisation compared to liming alone (Figure 7.9a). This may have been the result of lesser pH rise in the lime+fertiliser treatment (Figure 7.3), or possibly because more N was converted into humic compounds during mineralisation-immobilisation turnover in the high N environment. Addition of N has often been found to stimulate net mineralisation of soil N (Filimonov and Rudelev, 1977) due

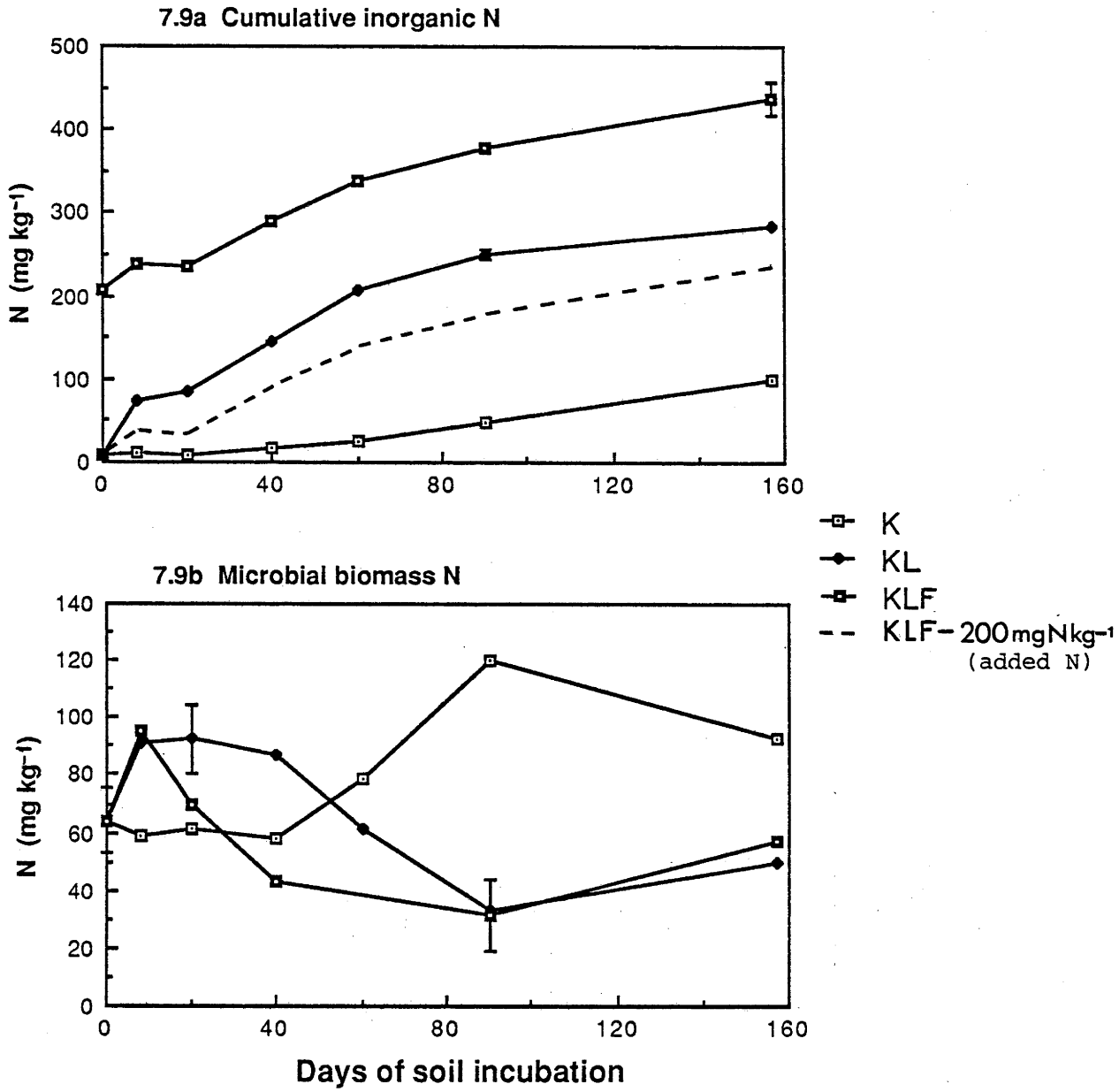


Figure 7.9 Lime and lime+fertiliser effects on inorganic and microbial biomass N of a krasnozem soil.

to physical and chemical causes (Broadbent and Nakashima, 1971; Laura, 1974; Westerman and Tucker, 1974). These causes include salt effects, pH changes, protolytic action on NH_4^+ formation and of NH_4^+ on nitrogenous bases in soil organic matter, and other side effects produced by fertilisers (Jansson and Persson, 1982). Liming may have obscured the fertiliser effects.

Figure 7.9b shows that within 8 days of treatment there was an increase in biomass N resulting from both liming and liming+fertilisation. Peaking of biomass N in the lime+fertiliser treatment was sharp whereas with liming alone, the maximum was broader, covering the period from 8 to 40 days. After the peak, biomass N in both treatments declined to the same level at 90 days, and this level was lower than initial biomass N. At the end of the experiment, biomaass N had again increased slightly. These changes parallel the initial high rates of mineral N accumulation followed by low rates (Figure 7.9a) and the respiration rates (Section 7.3.3.3) but not the changes in biomass C (Figure 7.2c). The difference between biomass N content of unlimed and limed soils was 40 - 90 mg kg^{-1} in the latter phases of the experiment, but this was much less than the differences (about 200 mg kg^{-1}) in accumulated inorganic N between soils. This suggests that the additional N mineralised was not derived solely from killed biomass N.

All the inorganic N extracted from the krasnozem and the limed and limed+fertilised krasnozem during the incubation period was in the NH_4^+ form. This implies that the change in microbial population, as shown by the change in microbial C

(Figure 7.2c), was only among the ammonifiers. Chase *et al.* (1968) reported that acid forest soils in Ontario did contain nitrifying bacteria, but their numbers and activity were extremely low until the soils were limed. Although liming increased soil pH to more than 7 (independent nitrification pH range = 6 - 8; Schmidt, 1982) there was no nitrification, implying that colonisation of the limed soil by nitrifiers did not occur within the experimental period. A lag period of nitrification was observed in the red podzolic soil, and possibly the limed krasnozem needed a longer period for significant colonisation and proliferation of nitrifiers.

7.3.3.4 Conclusions on soil and microbial biomass N

The amounts of organic N mineralised during soil incubation were an appreciable fraction of, but not proportional to, the total soil N (Table 7.4). Liming improved N mineralisation in the acid krasnozem soil, but it also decreased microbial biomass N. Addition of glucose confirmed the effect of readily available substrate C on the immobilisation of N.

7.3.4 Changes in soil and microbial biomass P during incubation

Results of P mineralised are not corrected for sorption in the initial discussion, but this is subsequently addressed in Section 7.3.5.4.

7.3.4.1 Comparison of changes in the untreated soils

Changes with time in soil P and soil microbial biomass P in the three soils are shown in Figure 7.10. The yellow podzolic soil had the most $NaHCO_3$ extractable inorganic P (labile inorganic P) while the krasnozem soil had the least (Figure 7.10a). Generally, labile inorganic P increased slowly with incubation time in all three soils, with some periods of net immobilisation.

$NaHCO_3$ extractable organic P (labile organic P) from the yellow podzolic and red podzolic soils were similar and much less than that from the krasnozem soil (Figure 7.10b). In all three soils, however, labile organic P was reasonably constant throughout the incubation period; there was a slight decline on the 8th day followed by an increase to the original level by the 40th day, and then another slight decline. Sharpley (1985) and Sharpley and Smith (1985) have observed that labile organic P in soil is maintained at a fairly constant level as a result of a dynamic equilibrium, such that a shift occurs from moderately labile and resistant organic P during mineralisation (Sharpley, 1985).

The behaviour of biomass P (Figure 7.10c) appears to be related to respiration by the soil biomass (Figure 7.4b), with high P at the start of the experiment and a decline during the incubation. The initial high biomass P could have resulted from microbial uptake of P from the soil inorganic P pool during increased activity after disturbance. Subsequent release of this P may have contributed to the increase in labile inorganic P.

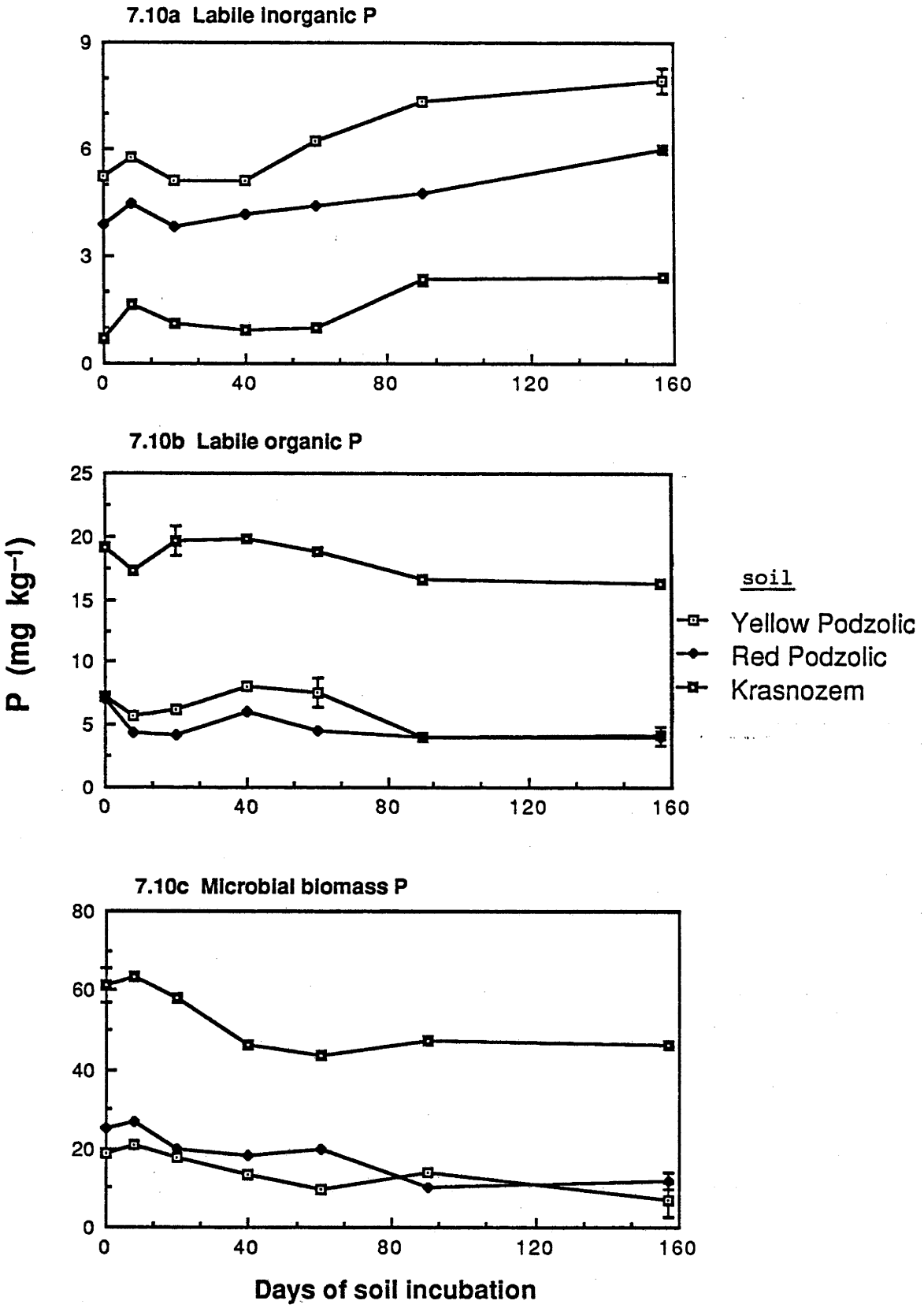


Figure 7.10 Changes in labile and microbial biomass P during incubation of the untreated soils.

The krasnozem soil had higher microbial biomass P than the other two soils - a pattern similar for biomass C and N. Biomass C:P ratios are shown in Table 7.8, and vary markedly between soils. Chauhan *et al.* (1981) found C:P ratios ranging from 12:1, under conditions of high P availability, to 45:1, where P was in low supply. Contrary to the findings of Chauhan *et al.* (1981), the sludge ammended soil (Table 7.8) had the highest C:P ratio although the labile inorganic P was high. This indicates that there are other factors, in addition to soil inorganic P, that determine biomass P.

Table 7.8 Biomass C:P ratios at the beginning of soil incubation

Soil	YP	YPS	YPF	RP	K
C/P	11.3	48.6	24.3	14.5	26.9

7.3.4.2 The effect of different soil treatments on P in the yellow podzolic soil P

The effects of soil treatments on labile inorganic, labile organic and microbial biomass P are shown in Figure 7.11. Figure 7.11a shows that there was a very slight increase in labile inorganic P in the untreated yellow podzolic soil during incubation. Labile inorganic P in the fertilized soils declined from 27.5 to 24.3 mg kg^{-1} over the 157 day incubation period, and the addition of glucose to this soil further reduced labile inorganic P by about 2 mg kg^{-1}. The general decline in labile inorganic P may be attributale to the slow transformation of the added fertiliser P from labile into non labile forms (Holford, 1977). The extra 2 mg

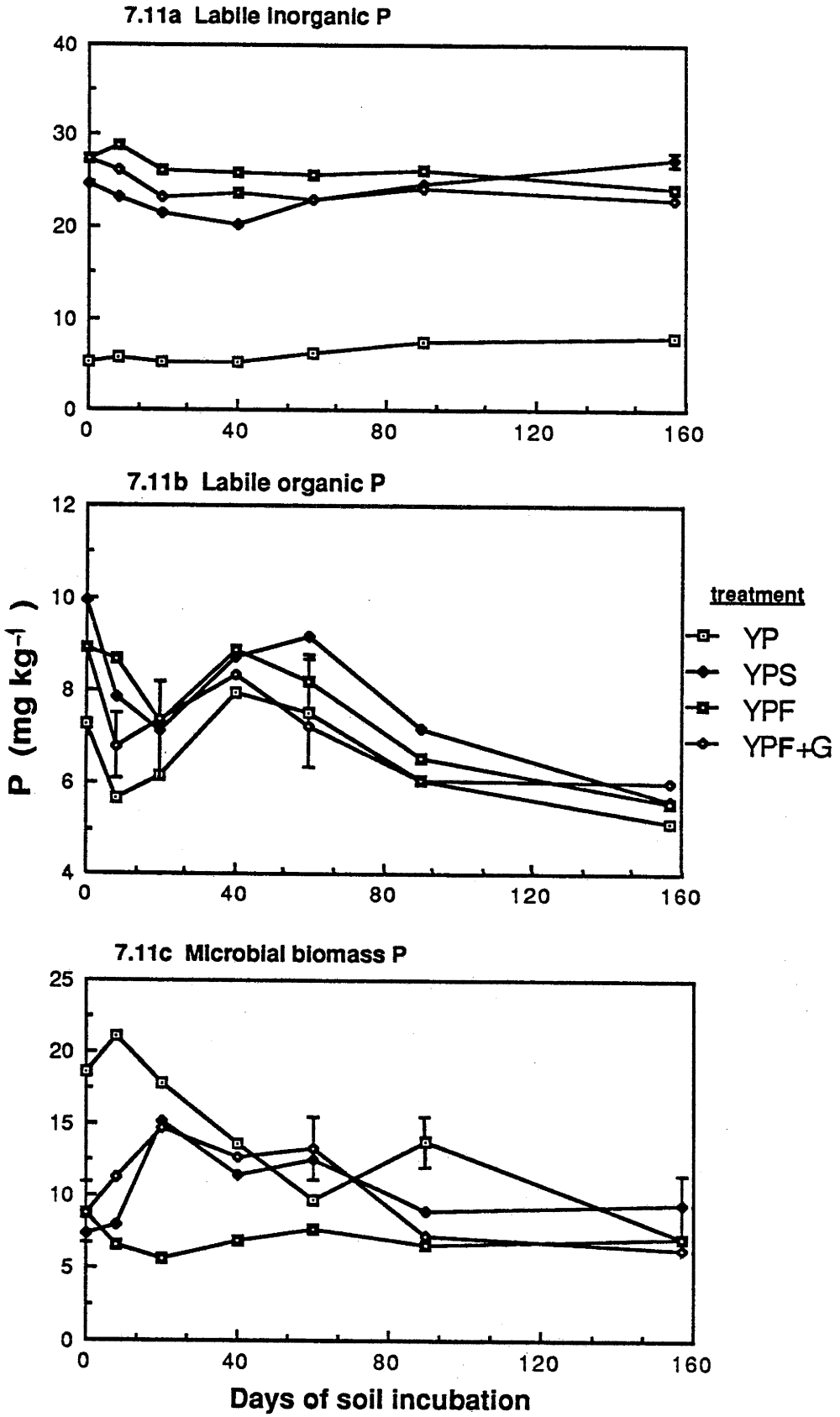


Figure 7.11 Field and laboratory treatment effects on labile and microbial biomass P during incubation of a yellow podsolic soil.

kg^{-1} reduction with glucose amendment is assumed to have been immobilised as a result of microbial growth. Labile inorganic P in the sludge treated soil declined during the first 40 days of incubation and then increased.

Labile organic P did not differ between treatments (Figure 7.11b). There was an initial decline followed by a recovery during the first 40 days of incubation, and this was followed by a slight decline.

Microbial biomass P declined in the untreated soil from about 20 to 10 mg kg^{-1}, while that from the fertilised soil was steady at about 7 mg kg^{-1} (Figure 7.11c). Biomass P in the sewage treated soil and glucose amended soil was similar - it initially increased by about 7 mg kg^{-1} at the 20th day and then gradually declined. The increase in biomass P in the glucose amended soil was (at the highest value) about 3 times greater than the deficit in labile inorganic P, suggesting either utilisation of other P sources by the biomass, or rapid replenishment of the labile P pool from the non-labile sources. Biomass P declined to its original level by the 90th day, corresponding to the decline in biomass C, and about 50 days later than N .

There appears to have been microbial immobilisation of P in the sludge treated soil. Labile inorganic P gradually declined by about 5 mg kg^{-1} on the 40th day and then increased (Figure 7.11a). About the same amount of P was recoverable as biomass P (Figure 7.11c). The mixing in of some sludge C could have given this effect as respiration was higher in the sludge treated soils. This effect was not reflected in biomass C, which increased slowly, nor in

inorganic or biomass N.

7.3.4.3 P transformations in the limed and limed+fertilised krasnozem soil

Liming caused a marked increase in inorganic P in the first 8 days followed by a more gradual increase (Figure 7.12a). The initial increase is consistent with the observation by Russell (1973) that liming increases solubility of Al phosphates. However, the stimulation of microbial activity following liming (Anderson, 1980; Haynes, 1984) is also likely to affect the mineralisation-immobilisation turnover in favour of mineralisation (Thompson *et al.*, 1954; Halstead *et al.*, 1963; Awan, 1964; Islam and Ahmed, 1973). According to Anderson (1980), liming increases solubility, and therefore microbial accessibility, of some phosphate esters. Data reported in section 7.3.2 suggests that the stimulated microorganisms are only those favoured by the new pH conditions, and that the increased biological activity lasts only as long as the substrate (probably including the solubilised phosphate esters) is available.

The steady decline in labile organic P after liming (Figure 7.12b) is suggestive of increased organic P mineralisation. Decline in organic P extracted from soils after liming has also been observed (Ghani and Aleem, 1942; Halstead *et al*, 1963) but has been partly attributed to the repressive effect of lime on the solubility of organic compounds (Dyer and Wrenshall, 1941; Halstead *et al*, 1963; Raison and McGarity, 1980a). Visual evidence (Plate 7.1) shows that less organic matter was extracted by $NaHCO_3$ in the

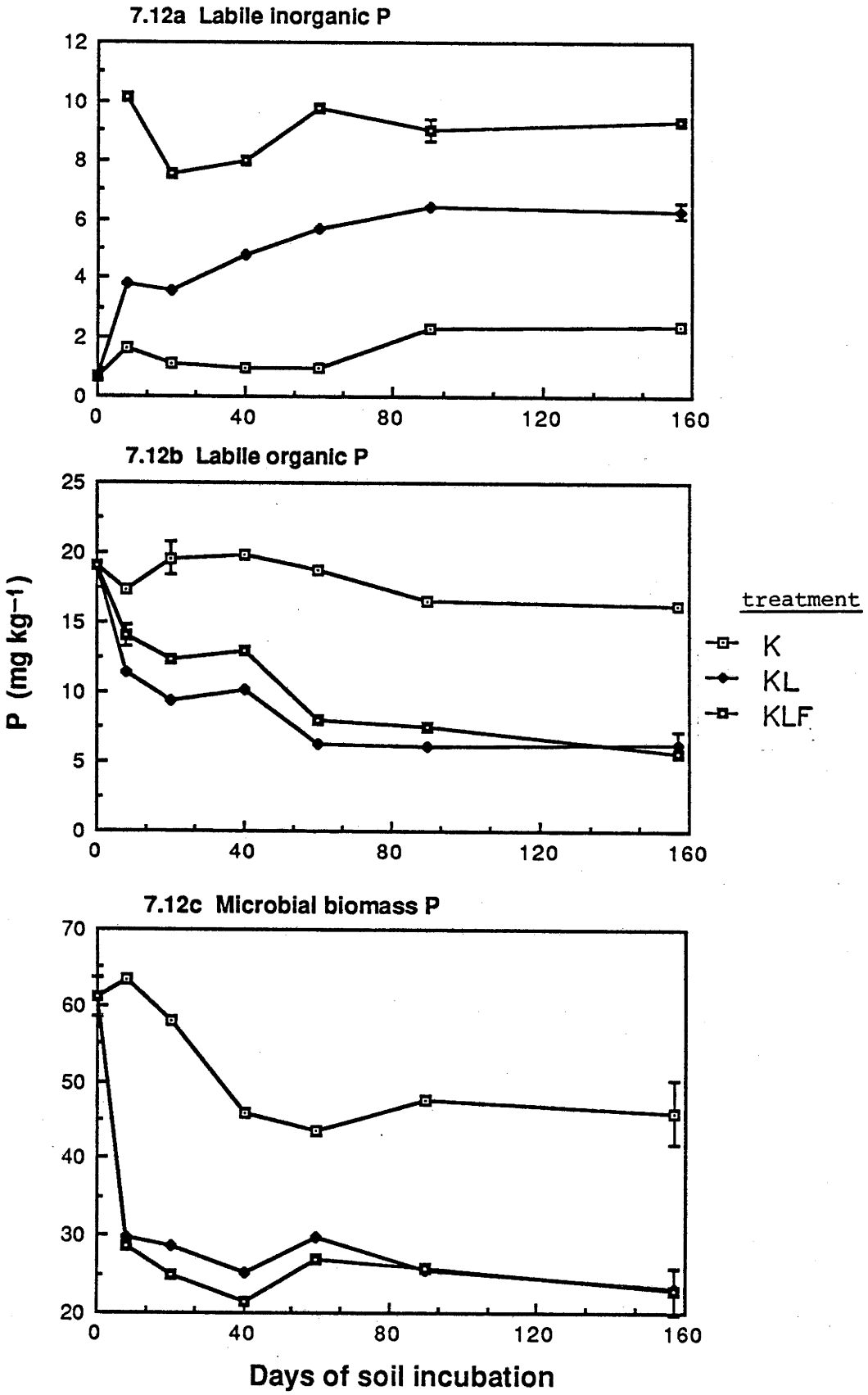


Figure 7.12 Effects of lime and lime+fertiliser on labile and microbial biomass P during incubation of a krasnozem soil.

limed krasnozem when compared to the control, and this may have led to low organic P being extracted.

The decreased biomass P due to liming (Figure 7.12c) is a reflection of the death of microorganisms (mainly fungi) in the changed soil pH environment, and this may also have contributed to the observed increase in labile inorganic P. Biomass P was halved within 8 days of liming, after which it remained constant. The biomass P content was not affected by greater quantities of labile P present in the soil (i.e. after liming and liming+fertilising). Biomass P decreased by an average of 24 mg kg^{-1} while inorganic P increased by an average of only 3.4 mg kg^{-1}. The wide difference is attributed to the high sorption capacity of this soil, which is also shown by the reduction of added fertiliser P (50 mg kg^{-1}) to about 10 mg kg^{-1} within 8 days (Figure 7.12a). Therefore the resulting increase in inorganic P in the limed krasnozem may be a result of: (i) solubilisation of mineral P compounds such as Al phosphate, (ii) increased mineralisation as a result of increased solubilisation of P esters, and (iii) mineralisation of lysed microbial cells resulting from the marked pH change.

The lime+fertiliser treatment did not appear to promote mineralisation more than lime added alone. Changes in biomass P and labile organic P were similar, although the decline in labile organic P was slightly more in the lime only treatment. The difference between inorganic P from the lime and lime+fertiliser treatments was fairly steady after the 8th day of incubation, indicating that very rapid sorption of added fertiliser P occurred within 8 days of its addition.

7.3.4.4 Correction for sorption of inorganic P produced during mineralisation

When P is added to soil it undergoes precipitation and sorption reactions (Section 2.4.1). This too is expected of P released from mineralisation (Halstead *et al.*, 1963). Thus accounting for all the mineralised P would necessitate determination of changes in the different soil P fractions, but this is impossible to perform with sufficient precision to be able to detect the small amounts of P mineralised in most soils. A better approach is to determine sorption correction factors for individual soils. Construction of the sorption isotherm, such as that constructed for P released from microbial biomass (Section 3.3.3.2) was deemed unsuitable because it is done in a solution medium and lacks the rate of reaction component. It was decided that incubation of soil, to which P was added, and repeated sampling over time was a better indicator of the reactions of P released during mineralisation.

Dilute solutions of P (KH_2PO_4) were mixed with samples of soil (4 ml/100g). The yellow podzolic, red podzolic and the limed krasnozem soils were used. The levels of added P were based on the soil organic N:P ratio, which is about 10 (Russell, 1973). Similar rates of mineralisation for the two nutrients were assumed in order to obtain working levels of P in the different soils. Therefore P added to soil was 2.5, 5 and 10 mg kg^{-1} for the yellow and red podzolic soils, and 15, 30 and 60 mg kg^{-1} for the limed krasnozem soil, being about 5%, 10% and 20% respectively of total N mineralised in soils

after 157 days (Section 7.3.3). The soils were incubated at 25°C in plastic bags with small holes for aeration, and were sampled 6 h, and 1, 2, 4, 8, 16, and 32 days after P addition. Duplicate analysis was done for each soil for $NaHCO_3$ (30 min) extractable inorganic P. The amount of P recovered was obtained as the difference between soils incubated with and without P added.

The results of this experiment are shown in Figure 7.13. For all the three soils, there was an immediate conversion of the added P into forms not extractable by $NaHCO_3$ (30 min extraction). After 6 h, the recovery of the added P was about 70% for the yellow podzolic soil, 65% for the red podzolic soil and 30% for the krasnozem soil. This was followed by a rate of decline during the next 8 days to recovery values of 65%, 55% and 20% respectively. After 8 days, the decline was slow. In the red podzolic and limed krasnozem soils, the recovery values of the added P was similar for the different amounts of P added. In the yellow podzolic soil, recovery of the added P was much lower for the low level of P added, but was the same for the higher levels.

A number of observations can be made from the results of this experiment. Firstly, the rate of conversion of added P into non-labile forms was consistent with field observations (Chapters 4 and 5), and with the limed+fertilised krasnozem soil (Section 7.3.5.3), in which the initial rate was high followed by a gradual decline. Secondly, the rates of decline and the amounts of P recovered were soil dependent. Less P was recovered from the limed krasnozem and most P was recovered in the yellow podsolic soil. This is consistent

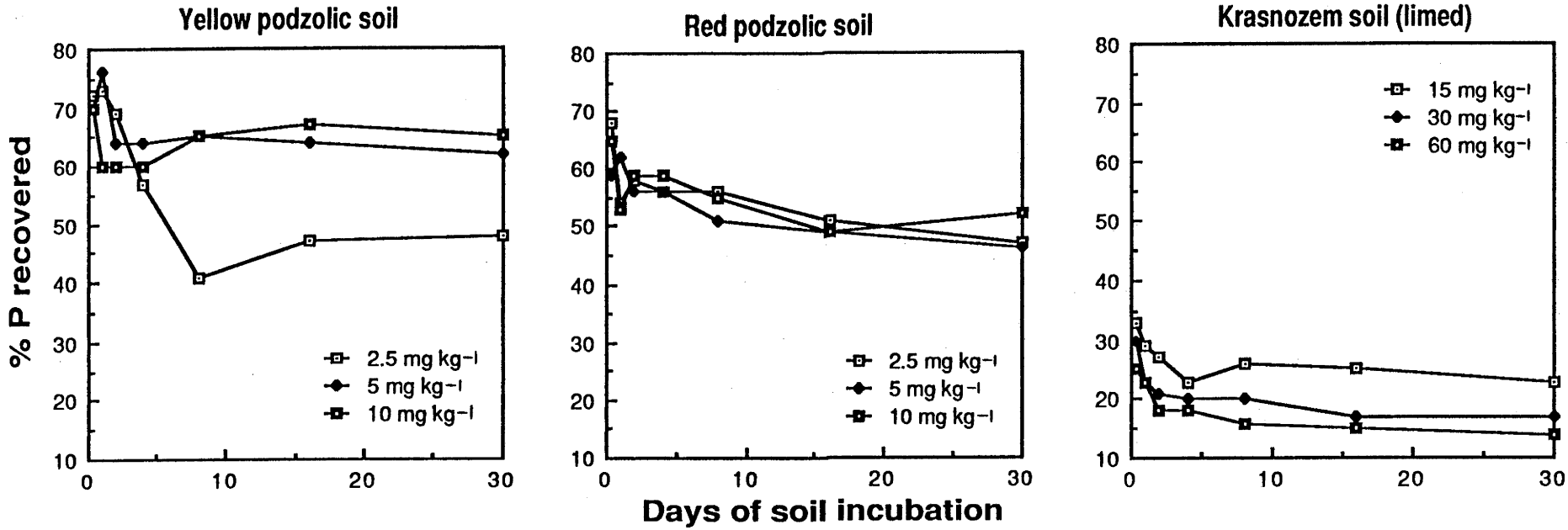


Figure 7.13 Recovery (%) of P added to soil and incubated for different periods.

with differences in the soils' sorption capacities, and implies that different amounts of P between soils may be mineralised than can be shown by increases in labile inorganic P. This will be discussed in Section 7.4. Thirdly, the recovery values were less than those obtained when P was added in the extracting solution - method used for P sorbed after microbial lysis (Section 3.3.3.2). For example, 51% P was recovered when P was added in the extracting solution to a limed krasnozem soil, compared with 30% (after 6 h) in the incubation method. The difference could have been due to the time of contact being only 30 minutes in the former method. However, results of the incubation method for sorption highlights the potential variation in the K_P factor (the fraction of biomass P extracted after fumigation). Microbial biomass P sorbed during the 24 hours of fumigation will depend on the soil sorption characteristics. P sorption during fumigation has been largely ignored in method of biomass P determination (Section 3.3.3.2) where a 'spike' of P is added to soil during extraction (Brookes *et al.*, 1982). For example, the recovery values after 24 h of incubation of soil with added P (equivalent to 24 h of fumigation) were 67% for the yellow podzolic soil, 56% for the red podzolic soil and 25% for the limed krasnozem soil, compared with 82%, 74% and 51% respectively when 'spiked' during extraction (Section 3.3.4.2). This suggests that 'spiking' during extraction might underestimate soil microbial biomass P. Further work needs to be done on the incubation of soil with added P as an alternative method of correcting for microbial biomass P sorbed during fumigation and extraction.

The recovery values of added P appear to be similar for the different levels of added P (except for the low level in the yellow podzolic soil). This needs to be further tested as it would be advantageous to be able to use one level of P in the determination of the amount of mineralised P and microbial biomass P sorbed. Recovery values for the mineralised P may be higher than those obtained in the incubation method because of the initial mixing of the soil after P addition. Mineralisation is also a dynamic process whereby P is not released to soil at one time. However, the method enables appreciation of the amount of P that may be recovered in labile forms after different soil incubation periods.

7.4 Discussion - Relationships between soil properties, microbial properties and N and P fluxes during incubation

Microbial activity and transformation of N and P and the way they respond to various soil treatments and amendments have been examined in a number of soils. This section examines the possible relationships between these processes.

Table 7.9 represents the cumulative O_2 uptake and N and P mineralised during 157 days of incubation. In estimating the P sorbed, the recovery values of P added as 10% of the mineralised N at the end of 32 days (Section 7.3.5.4) were used. The recoverly values were 62% for the yellow podzolic soil, 46% for the red podzolic soil and 17% for the krasnozem soil. The recovery value for the yellow podzolic soil was also used to estimate sorption for the sewage sludge treated yellow podzolic soil, while that of the limed krasnosem was

used for the non treated krasnozem. Labile inorganic P in the fertilised yellow podzolic soil (and its glucose treatment), and in soil from the limed+fertilised krasnozem soil showed a negative trend during incubation. The continuing conversion of added P from labile to non-labile forms in these soils may have masked any effects of mineralisation of P.

Table 7.9 A comparison of inorganic N and P accumulated and O_2 taken up after 157 days of soil incubation

Treatment	%C	N	P	P*	O_2	O_2/N	O_2/P	N/P
		mg kg^{-1}						
YP	1.4	79.9	2.7	4.0	634	7.9	159	20
YPS	2.1	82.1	2.7	4.0	760	9.3	190	21
YPF	2.4	63.8	-3.2	**	434	6.8	**	**
YPF+G		60.6	-4.5	**	1023	16.9	**	**
RP	3.2	71.5	2.1	4.6	1525	21.3	332	16
K	9.3	91.6	1.7	10.0	3001	32.8	300	9
KL		276.0	5.6	32.9	2820	10.2	86	8
KLF		230.6	**	**	2574	11.2	**	**

* P corrected for sorption; ** Periods during which there was net P immobilisation.

Before correcting for sorption, both yellow podzolic and red podzolic soils showed higher levels of labile inorganic P from mineralisation (2.7 and 2.1 mg kg^{-1} respectively) than the krasnozem soil (1.7 mg kg^{-1}). However, when these values are corrected for sorption, it is shown that P mineralised in the krasnozem (10 mg kg^{-1}) is more than double that mineralised in the other two soils. Correcting for P sorbed during mineralisation gives an appreciation of the degree of its mineralisation in different soils. Chapter 8 will examine if plants can make use of the this P before it is sorbed.

Table 7.9 also shows that inorganic N and P accumulated, and O_2 taken up during 157 days of soil incubation varied markedly. The highest amount of N accumulated was in the limed krasnozem (276 mg N kg^{-1}) while the lowest was in the glucose treated soil (60.0 mg N kg^{-1}). The highest amount of P accumulated was also in the limed krasnozem (32.9 mg P kg^{-1}) while the lowest was in the yellow podzolic and in the sewage sludge amended soil (4.0 mg P kg^{-1}). O_2 uptake ranged from 434 mg kg^{-1} in the fertiliser ammended yellow podzolic soil to 3001 mg kg^{-1} in the untreated krasnozem.

The differences in N and P accumulated and O_2 uptake may be a result of both the variability in the soil substrates and the soil microorganisms. Generally, the krasnozem and red podzolic soils with high ratios of organic C:N and C:P (Table 7.1) required more O_2 uptake per unit of N accumulated. This can be explained by the fact that nutrient mineralisation usually represents the excess over microbial demands. Thus during decomposition of organic matter with high C:N and C:P ratios, the decomposers will respire more C until the C:nutrient ratios are lower than that required for microbial growth (Jenkinson, 1981). This is supported by the effect of glucose which caused N and P immobilisation as a result of widening C to nutrient ratios. Liming reduced the requirement for O_2 per unit of N and P probably because the organic matter initially mineralised was mainly low C:N biomass killed by liming. Microorganisms (e.g. bacteria and fungi) differ in the forms of nutrients immobilised (Hunt *et al.*, 1983), and consequently may bring about variation in nutrient mineralisation between soils.

Over the incubation period, O_2 was used at different rates compared with amounts of N and P mineralised, and this varied with treatment. For example, in treatments for which P sorption was corrected (Table 7.10), the ratio of cumulative O_2 uptake to inorganic N accumulated in soil declined in the yellow podzolic soil, varied in the red podzolic and was steady in the limed krasnozem. With respect to labile inorganic P, the incubation exhibited net mineralisation at some periods, and net immobilisation at some others. Thus the differences in the ratio of cumulative O_2 uptake to labile inorganic P accumulated in soil, and in the inorganic N:P ratio (cumulative) were large and variable.

Table 7.10 Variation in the cumulative O_2, N and P ratios during soil incubation.

Soil	Ratio	Days of soil incubation					
		8	20	40	60	90	157
YP	O_2 :N	19.2	13.5	11.5	8.1	7.3	7.9
	O_2 :P	114	**	**	257	148	159
	N:P	6.7	**	**	32.6	20.3	18.6
RP	O_2 :N	38.6	59.8	58.4	39.9	47.6	21.3
	O_2 :P	145	**	1081	794	601	332
	N:P	3.7	**	18.5	19.9	12.6	15.5
KL	O_2 :N	6.6	11.1	10.4	9.1	9.2	10.2
	O_2 :p	21	50	59	62	58	87
	N:P	3.5	4.5	5.6	6.8	7.1	8.4

** Periods during which there was net P immobilisation

The differences observed may be explained by the differences and dynamic nature of the biomass that carry out

the mineralisation-immobilisation processes. The biomass continually transfers inorganic nutrients into organic products of synthesis, and immobilised nutrients back into inorganic decay products (building up and dying away of the biomass; Knowels and Chu, 1969; Campbell, 1978; Shields *et al.*, 1973). The inorganic N and P measured were net effects of the mineralisation-immobilisation turnover which do not reflect gross flows in the soil (Hiltbold *et al.*, 1951). A small net effect may be the result of low overall biological activity in the ecosystem, or it may be the result of a high activity in which the processes work in opposite directions (Jansson and Persson, 1982). This is consistent with the proposal against any stoichiometric relationships between C, N and P cycling through organic matter (McGill, 1979; McGill and Cole, 1981).

To conclude, the basic activity of heterotrophic organisms in soil is the dissipation of organically bound energy, thus enabling net mineralisation to occur (Jasson and Persson, 1982). Measurement of net N and P mineralised is of practical significance from the standpoint of forestry production since mineralisation-immobilisation turnover is an important source of these nutrients. In this experiment, mineralised N over 157 days ranged from about 37 kg N ha^{-1} in the untreated krasnozem to about 110 kg N ha^{1} in the limed krasnozem, to a depth of 5 cm. Net P mineralised in 157 days ranged from about 3 kg P ha^{-1} in the yellow and red podzolic soils to 13kg P ha^{-1} in the limed krasnozem (corrected for sorption). These amounts are well above the pine seedling requirements at establishment as determined in the experiments by Nielsen *et al.* (1984). However, relationships between C, N

and P are not stoichiometric during cycling and they appear to need to be measured independently. Correction for P sorption is necessary in order to have a better estimate of net P mineralised, and of microbial biomass P in different soils. The method reported in Section 7.3.4.4 appears to offer a more realistic estimate of P sorbed and needs to be further tested.

One of the factors that influence the nature and rate of P transformation processes is the presence of plants. Plants have a major impact on the functional behaviour of decomposers (i.e. rhizosphere effects). They are sources of decomposable substrates, and act as competitors with microbes for soil nutrients. For example mean microbial population density around root surfaces rises to 10 or more times than that in the bulk of the soil (Tinker, 1980). This may be counteractive to plant uptake if the nutrients are immobilised when the plant demand is high, but may be an asset in retarding leaching and sorption of nutrients. There is general agreement, however, that an appropriate population of the rhizosphere can increase the growth of plants in some circumstances (Tinker, 1980), and that laboratory incubation procedures used to measure organic P mineralisation usually underestimate the potential utilisation of this P by plants (Dalal, 1979). The following Chapter assesses the effects of the presence of plants on the mineralisation, and subsequent uptake of N and P.

CHAPTER 8 PLANT UTILISATION OF PHOSPHORUS AND NITROGEN RELEASED BY MINERALISATION

CHAPTER EIGHT

PLANT UTILISATION OF PHOSPHORUS AND NITROGEN RELEASED BY MINERALISATION

8.1 Introduction

The incubation experiments (Chapter 7) showed that N mineralisation can be assessed directly by chemical extraction. This was less so with P, largely because methods for *in situ* correction for sorption of part of the mineralised P require more development. The quantities of mineralised N and P estimated were greater than those required by pine seedlings during the establishment phase. It seemed that a more appropriate estimate of the amount of P released in the soil may be obtained by determining the amount of P taken up by plants, and the amount of residual labile inorganic P remaining following plant harvest. Thus an experiment was performed in which, plants were used as a bioassy of chemical indices of N and P availability. The amounts of P which might be adsorbed after mineralisation were estimated using P recovery factors obtained in Section 7.3.5.4 - that is net P mineralised was divided by 0.62, 0.46 and 0.17 for the yellow podzolic, red podzolic and krasnozem soils respectively.

Growing plants may influence the net effect of the mineralisation-immobilisation turnover and thus the amount of P sorbed by the soil. According to Barber (1980), plant roots change concentrations of ions in soil adjacent to the root because of (i) mass flow and diffusion of ions resulting from water and ion adsorption; (ii) exudation of H^+ or OH^- (or

HCO_3^-) as a result of imbalance between cation and anion uptake; (iii) exudation of organic substances from root into the soil; and (iv) differential microbiological activity due to the presence of the root and its effects under (i), (ii), and (iii). A decrease in rhizosphere pH increases P solubility, and usually occurs when cation uptake dominates anion uptake (Riley and Barber, 1971). The reverse occurs when pH increases due to anion uptake becoming dominant. pH changes also influence microbial population composition (Haynes, 1984) and hence mineralisation rates.

Microorganisms utilise root exudations in addition to other substrates present in the soil (Tinker, 1980), and this may increase microbial growth and consequently nutrient mineralisation-immobilisation reactions. Microbial nutrient immobilisation is advantageous where inorganic nutrients are in excess of plant requirements because they are temporarily held against leaching or sorption. It is disadvantageous where available plant nutrients are in short supply. Because most of the biomass is close to the root surface (Newman and Watson, 1977) any products of its turnover may be in easy reach of the root.

As Blair *et al.* (1977) noted, mineralisation can vary markedly depending on whether plants are present or not, indicating that results from laboratory incubation experiments may not be safely extrapolated to the field. The following experiment studied mineralisation of soil organic N and P in the presence and absence of plants, and the subsequent uptake of these elements by plants.

8.2 Experimental procedures

8.2.1 Preparation of soils

The yellow podzolic, the red podzolic and the krasnozem soils were used in this study. The soils were collected from the top 5 cm of the soil profiles (at different times to the samples used in Chapter 7) and sieved without drying through a 4.75 cm sieve. There were two treatments applied to the krasnozem soil: (a) control, and (b) the addition of lime to the fresh soil (20 g $CaCO_3$ per 1 kg equivalent of oven dry soil). Bioassy (measurement of plant growth and nutrient uptake) and mineralisation studies were then conducted on the freshly treated krasnozem soil. The yellow podzolic and red podzolic soils were subjected to different pre-planting incubation periods with the aim of generating different quantities of mineral N and P in the soil at planting time. Portions of each of the two soils were incubated in bulk (5 kg) for periods of 0, 15, 30, 60 and 90 days at 25°C. The incubation was in plastic bags with a few holes made in them to allow for gaseous exchange while minimising moisture loss. After the incubation periods, eight replicates of 500 g of soil (on oven dry basis) were made for each period of incubation and for the limed and non-limed krasnozem treatments. The soils were put in 10.5 cm diameter pots with sealed bottoms to prevent leaching. Four of the replicates were used to assess changes in soil N and P where the soil was further incubated in the glass house without plants. The other four replicates were planted with rye grass (*Lolium*

perenne L.) to estimate the effects of plants on the mineralisation of N and P and uptake of these elements by plants.

8.2.2 Sowing and growth conditions

Four replicates of all treatments were sown with rye grass (*Lolium perenne* L.) at a rate of 0.5 g of seed per pot. Rye grass was chosen as a test crop because of its fast establishment compared to tree seedlings. The seeds were sown by removing about 1 cm of the soil, evenly spreading the seeds and sprinkling back the soil. The soil was kept moist and the pots were covered with a plastic sheet until germination (6 days). The pots were then arranged in a randomised block design.

The plants were allowed to grow for 40 days in a glass house. The soils were kept at field capacity by weighing and replacing the deficit with distilled water. The growth period ran through spring , and both temperature and light varied with external weather changes. The average maximum day temperature was 21°C (minimum 12°C; maximum 31°C) with daylight being more than 10 h. Temperatures inside the glasshouse were not measured.

8.2.3 Harvesting and plant sample preparation

The plant shoots were harvested by cutting them 1 cm above the soil surface to avoid soil contamination. The roots were carefully separated from the soil, and as much of the adhering soil as possible was washed off with distilled water.

Both shoots and roots were oven dried at 70°C, and ground using a stainless steel mill to pass through a 1 mm sieve, prior to analysis for total P and N after Kjeldahl digestion.

8.2.4 Analysis of soils

The soils were analysed without drying for the 30 minute sodium bicarbonate extractable inorganic, organic and biomass P, and K_2SO_4 extractable inorganic N. Soils were analysed at the following times:

1. After they had been incubated for various periods under controlled conditions (pre-glass house*(pre-plant) N and P)
2. After continued incubation in the glass house. This involved soils without plants (post-glass house N and P) and soils with plants (post-plant N and P)

The terms pre-glass house and post-glass house are used when describing soils incubated in the glass house without plants. Pre-plant and post-plant terms are used when describing soils in which plants were grown. As the soils were incubated in bulk under the controlled conditions, data for pre-glass house analysis is the same as for pre-plant analysis.

8.3 Results

8.3.1 Changes of N and P in soils incubated without plants

The yellow and red podzolic soils. Figures 8.1a and 8.1b show that changes of labile inorganic P following different

* "Glasshouse" is one word. In the text, it has been wrongly written as two words (glass house).

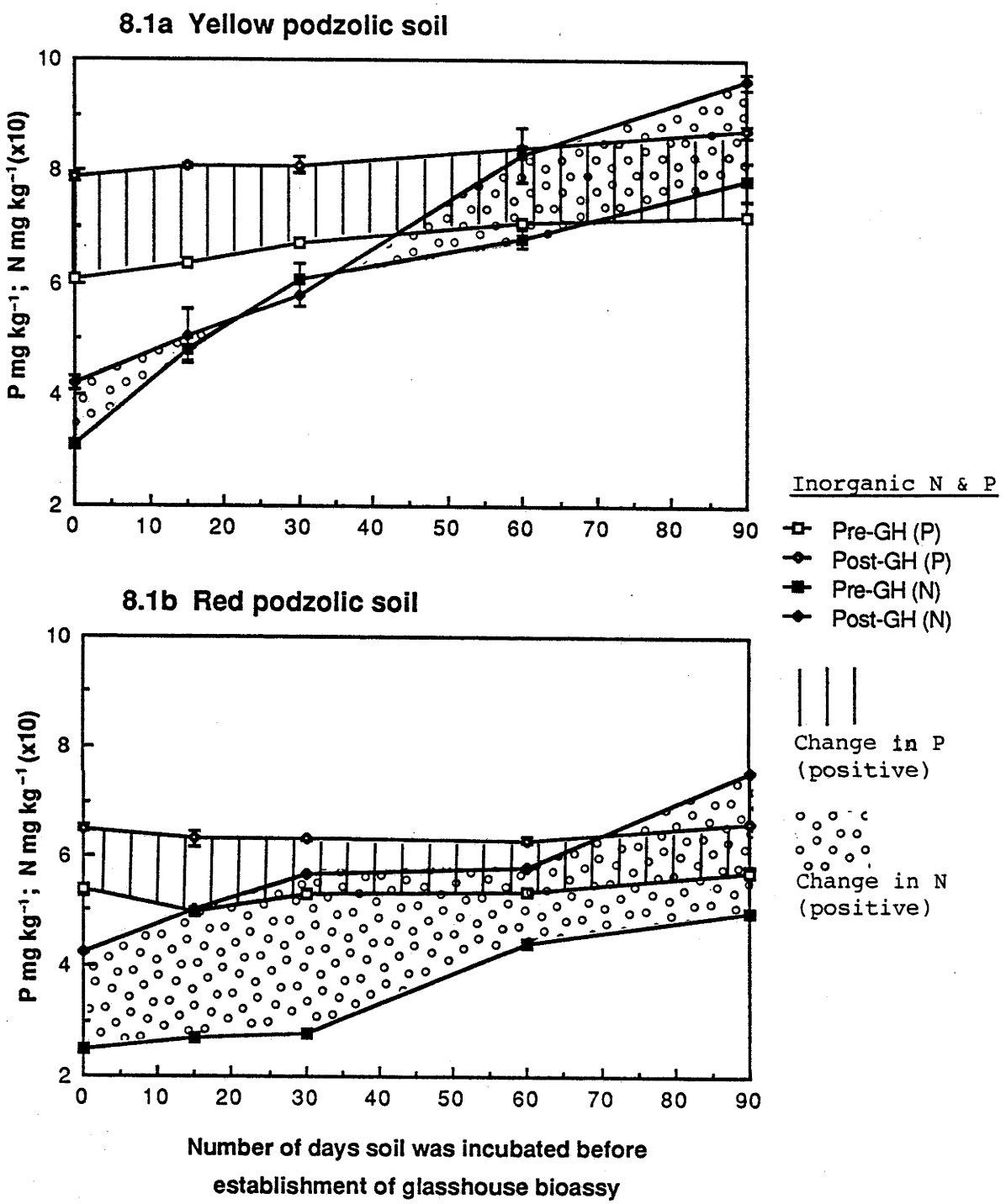


Figure 8.1 Changes in N and P during laboratory and glasshouse incubation of soils without plants. Shaded areas represent the effect of glasshouse incubation (40 days) over pre-glasshouse (different days) incubation levels.

FIGURE 8.1 CONTINUED ON NEXT PAGE

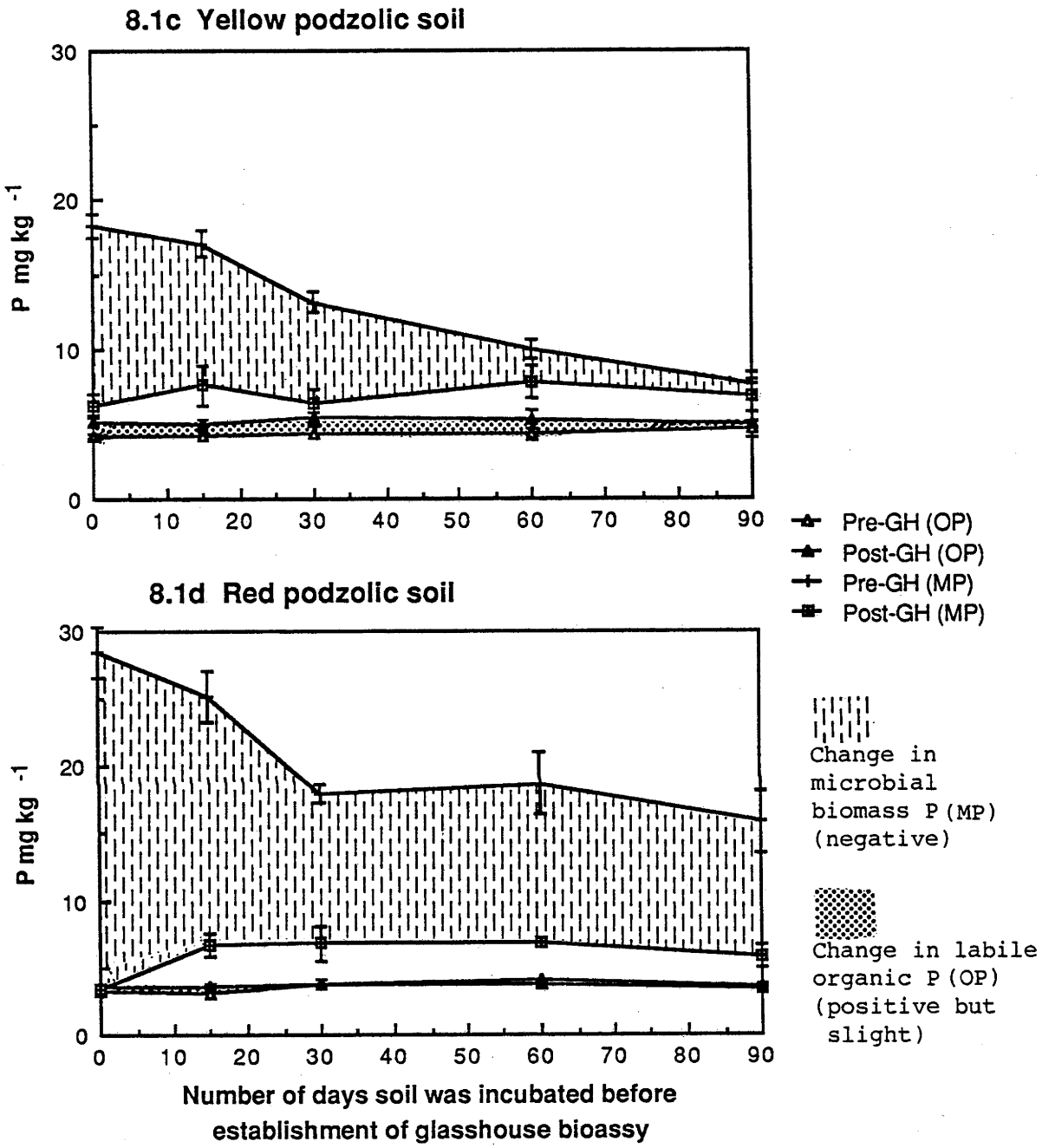


FIGURE 8.1 CONTINUED FROM PREVIOUS PAGE

periods of pre-glass house (Pre-GH) incubation were small in both the yellow podzolic and red podzolic soils. The period of Pre-glass house incubation also had little effect on the amount of labile inorganic P in soils following subsequent incubation in the glass house. Instead, glass house incubation resulted in a uniform increase in labile inorganic P in each of the two soils irrespective of the different periods of prior incubation.

In contrast to labile inorganic P, inorganic N accumulated in both soils with increased periods of pre-glass-house incubation (Figures 8.1a and 8.1b) but at a faster rate in the yellow podzolic soil than in the red podzolic soil. Similar soil effects were observed in Chapter 7. Inorganic N also increased with glass house incubation, although not uniformly for all the pre-glass house incubation levels.

Changes in labile organic P with various pre-glass house incubation periods were not significant (Figures 8.1c and 8.1d), and glass house incubation increased it only slightly in the yellow podzolic soil. Microbial biomass P declined markedly with pre-glass house incubation (Figures 8.1c and 8.1d), and during glass house incubation microbial biomass P was decreased significantly also to similar levels irrespective of the pre-glass house value.

The rate of change in inorganic and biomass P fractions under glass house incubation was greater than when the soils were incubated under controlled temperature and moisture (pre-glass house incubation). For example, the amounts of inorganic P (corrected for sorption) accumulated in the yellow podzolic and red podzolic soils for 90 days under controlled

conditions (pre-glass house) were 1.1 and 0.3 mg kg^{-1} respectively, compared with 1.56 and 2.3 mg kg^{-1} after 40 days of glass house incubation. The rate of change was also greater for N accumulation in the red podzolic soil but not in yellow podzolic soil. The increased inorganic P accumulation was attributed to the uncontrolled fluctuations in soil temperature and moisture. Day temperatures outside the glass house varied from 12° to 31°C during the growth period. Temperatures inside the glass house (not recorded) were expected to have been higher at peak day temperatures than the external temperatures, and may have contributed to higher mineralisation than under the controlled conditions. Bunt and Rovira (1955) showed that the rate of CO_2 production in a subtropical soil increased to a maximum at 37°C and then declined. According to Dalal (1977), high organic P mineralisation rates are observed above 30°C, at which temperature the growth of bacteria are optimal.

Changes in temperature also affected the rate of drying and thus the need to rewet soils. Wetting and drying cycles enhance P mineralisation possibly by exposing inaccessible humic matter to microorganisms for decomposition (Dalal, 1977). The mineralisation flush depends upon the intensity and period of drying. In this experiment, the drying conditions were not as severe as those in the experiments of Birch (1958, 1959, 1960) and of Birch and Friend (1961). Nevertheless, the dry-wet cycles may still have enhanced mineralisation.

The reduction in microbial biomass P during soil incubation was large, and up to 5 times the increase in

inorganic P during glass house incubation. Thus only part of the biomass contributed to the increase in labile inorganic P after the glass house incubation period. The remaining P appears to have been transformed into less mobilisable organic P. The relatively constant level to which biomass P was reduced after glass house incubation (Figures 8.1c and 8.1d) suggests a minimum level to which soil microbial biomass P can be reduced.

In conclusion, there was increased mineralisation of soil organic matter in the glass house incubated soils compared with the controlled conditions, and this was attributed to the drying-weting cycles and to temperature changes. These results show the need to take into account climatic effects in assessing nutrient mineralisation. A technique developed for *in situ* measurement of field N mineralisation (Raison *et al*, 1987) appears to be suitable for this purpose.

The krasnozem soil. There was no pre-glass house incubation treatments with the krasnozem soil. Table 8.1 shows that both labile organic P and microbial biomass P decreased after liming (1 day) and declined further during glass house incubation. The effects of lime on inorganic N and P were not immediate, but occured during glass house incubation. Thus inorganic N increased by 43.6 mg kg^{-1} in the non-limed krasnozem compared with 138.6 mg N kg^{-1} in the limed krasnozem, while P increased by 7.8 and 15.2 mg kg^{-1} (corrected for sorption) in the non-limed and limed soils respectively. As in the yellow and red podzolic soils, biomass P declined with incubation.

Table 8.1 The effects of liming on inorganic N and $NaHCO_3$ extractable P in a krasnozem soil during glass house incubation (40 days). Means of 4 replicates.

Treatment	Pre-incubation				Post-incubation			
	N	IP	OP	MP	N	IP	OP	MP
	mg kg^{-1}							
non limed	4.0	1.8	17.8	79.4	47.6	3.9	19.7	36.3
limed	6.6	1.7	14.9	31.4	145.2	5.8	7.9	29.9

IP = inorganic P; OP = organicP; MP = microbial biomass P

8.3.2 Effects of plants on N and P in soils that had been incubated for different periods

In this section, nutrient contents in soils before (pre-plant) and after (post-plant) plant growth are compared.

Figures 8.2a and 8.2b show that for both soils, the presence of plants markedly reduced the amount of labile inorganic P. This effect was more pronounced in the yellow podzolic soil than in the red podzolic soil. This may be due to differences in the soils' sorption capacity for P, because the more strongly the P ion is adsorbed by the soil, the smaller is its effective diffusion to the plant (Wild, 1981). The red podzolic soil had higher P sorption capacity than the yellow podzolic soil (Section 5.2).

Plants utilised nearly all the inorganic N initially present at planting (Figures 8.2a, 8.2b). This implies that the plants growing in soils that had been incubating for longer periods had a better supply of N, and that inorganic N

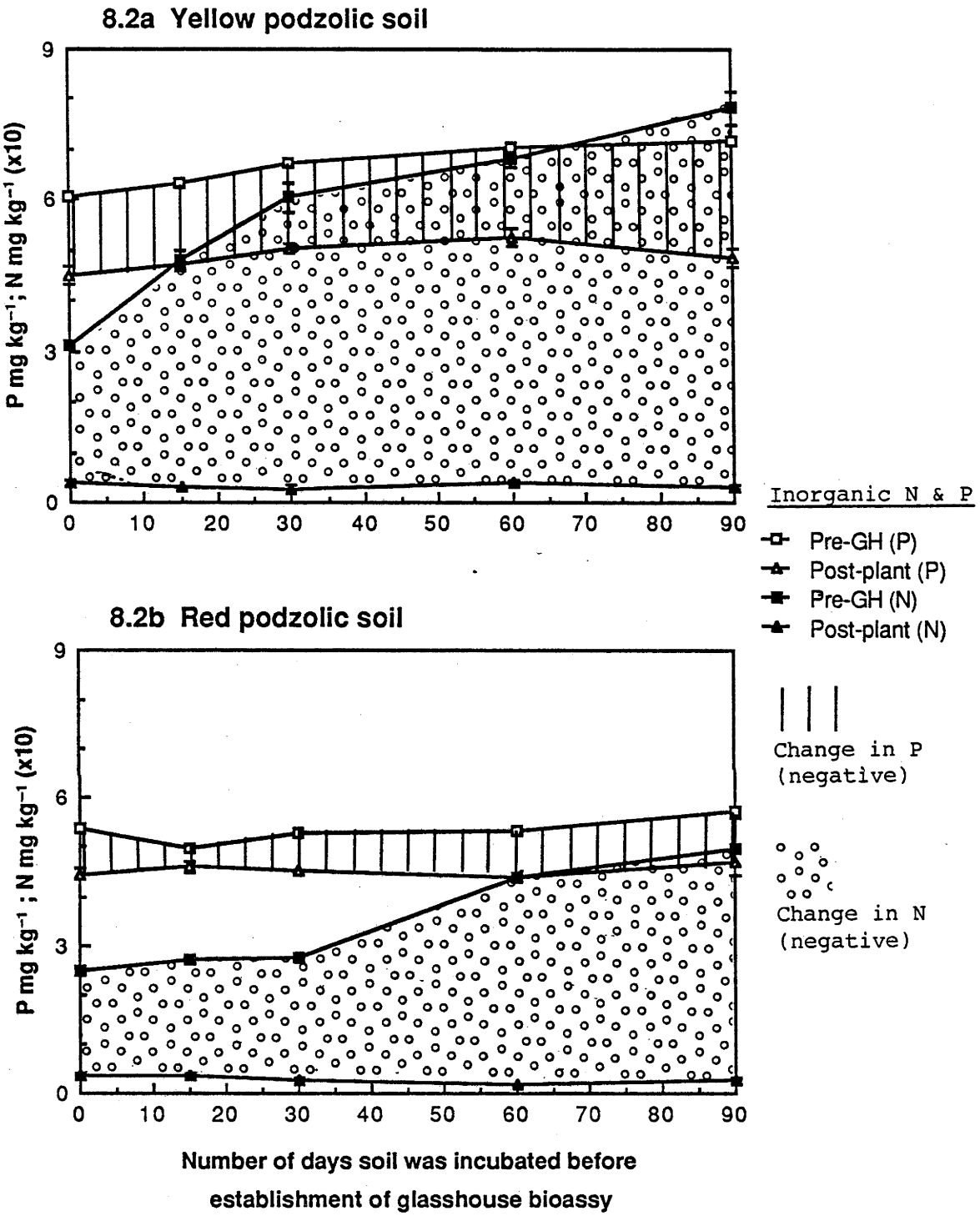
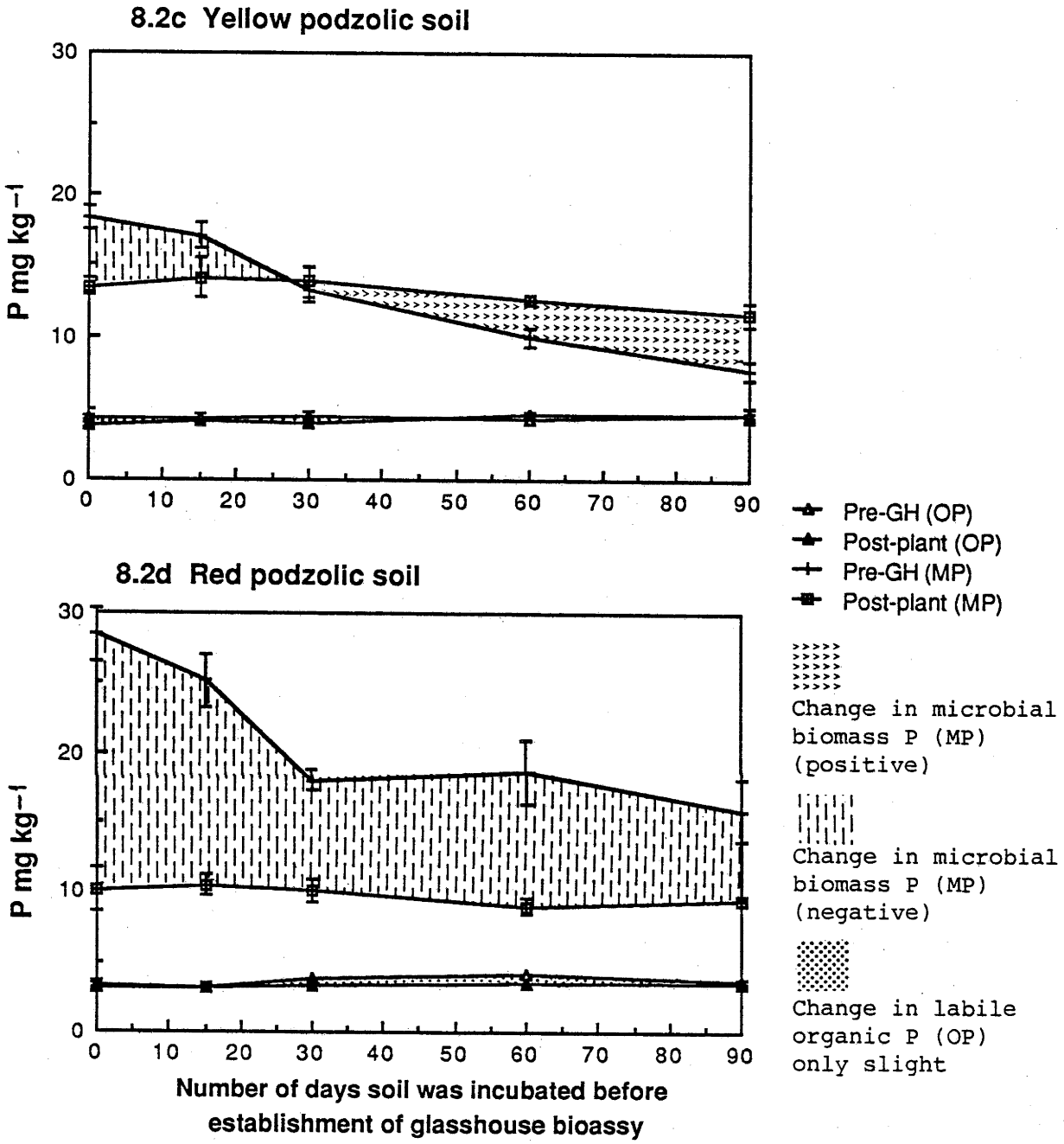


Figure 8.2 Changes in soil N and P during growth of potted plants in the glass house. Shaded areas represent the effect of plants on pre-planting soil nutrient levels.

FIGURE 8.2 CONTINUED ON NEXT PAGE



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is a good index of N availability.

Soils in which plants were grown showed slight changes in organic P (Figures 8.2c, 8.2d). Sharpley (1985) hypothesised that labile organic P may remain fairly constant because mineralisation of this pool could be replenished at similar rates from the moderately labile organic P pool. Figure 8.2c shows that post-plant microbial biomass P in the yellow podzolic soil was lower than that in the pre-plant soils in samples with short pre-plant incubation periods (0 and 15 days) and higher in samples incubated for longer periods. Biomass P in the red podzolic soil declined to levels below those of pre-plant incubation samples, but with the largest decline ocurring in the short incubation periods. In both soils, however, the decline in microbial biomass P in the planted soils was less than that in soils incubated in the glass house without plants (Table 8.2) by 5.6 - 7.8 mg kg^{-1} in the yellow podzolic soil, and 2.9 - 16.7 mg kg^{-1} in the red podzolic soil. It is therefore clear that the glass house

Table 8.2 Changes in microbial biomass P during 40 days of glass house incubation of soils with and without plants. Means of 4 replicates.

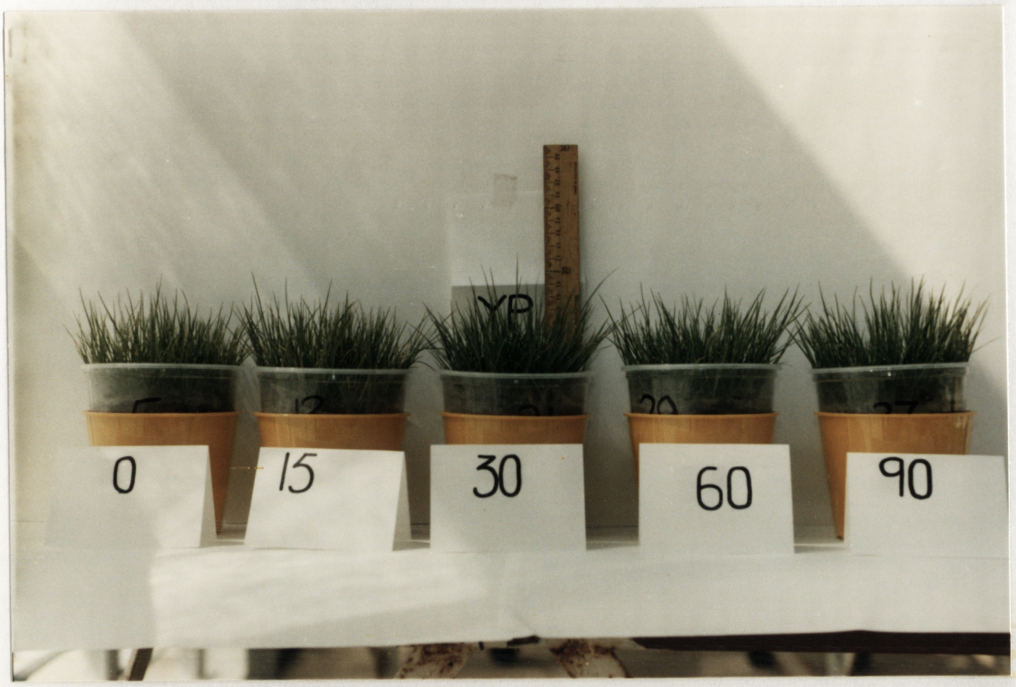
Treatment	Pre-glass house incubation days				
	0	15	30	60	90
	Units: mg kg^{-1}				
YP (not planted)	-12.0	-9.5	-6.9	-2.2	-0.8
YP (planted)	-4.7	-3.0	+0.9	+3.4	+5.2
RP (not planted)	-25.1	-18.4	-11.1	-11.8	-10.1
RP (planted)	-8.4	-14.7	-7.9	-8.9	-6.6

YP = yellow podzolic soil; RP = red podzolic soil

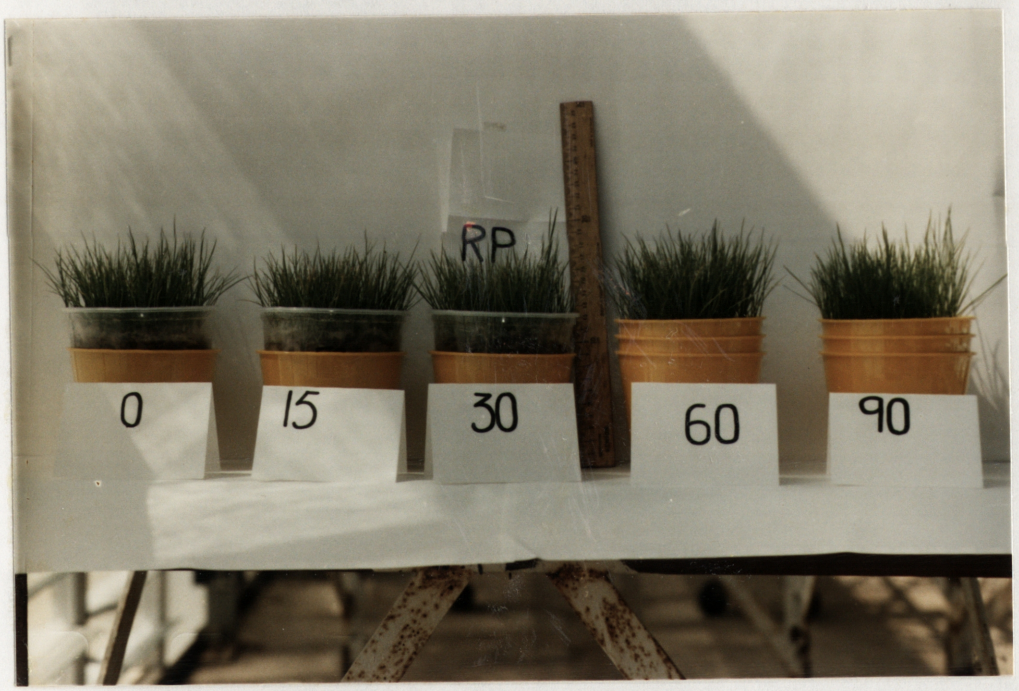
incubation reduced microbial biomass P, but that the reduction was minimised by the presence of plants. Two reasons that may explain the minimisation are the contribution of P from plant roots during fumigation of soil, (i.e. an artifact of the methodology) or increased subtrates for microbial growth and maintainance by plant roots (i.e rhizosphere effect). McLaughlin and Alston (1985) found that plant root and shoot materials influenced microbial P determined using the chloroform fumigation technique. Attempts were made to remove all the roots from soil, but it is possible that some root material may have remained and contributed to the higher microbial biomass P determined in soils in which plants were grown. Materials supplied by the plant root (including soluble exudates, sloughed-off parts of the root cap, mucigel, root hair residues and abraded epidermal cells) are utilised by microorganisms, increasing their activity (Tinker, 1980) and hence P immobilisation (Barber and Laughman 1967). This increases extractable microbial biomass P.

8.3.3 Plant dry matter yield

An effect of period of soil incubation before planting on plant size was not easily distinguishable in either the yellow or red podzolic soils (Plate 8.1). The shade in the photographs exaggerates the plant colour in some of the treatments. Figure 8.3 shows that root dry matter yield was more than shoot dry matter yield, and slightly declined with increasing length of pre-planting incubation. Plants grown in



Y P



R P

Plate 8.1 Effects of pre-plant incubation time (days) of soil on plant growth.
YP = yellow podzolic soil
RP = red podzolic soil

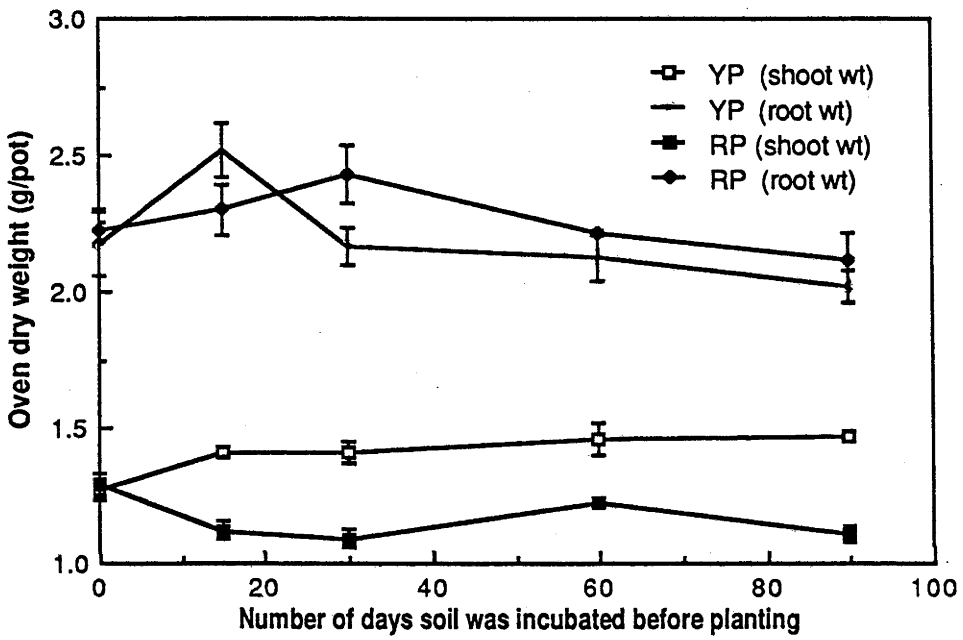


Figure 8.3 The effect of pre-plant soil incubation on plant dry matter yield

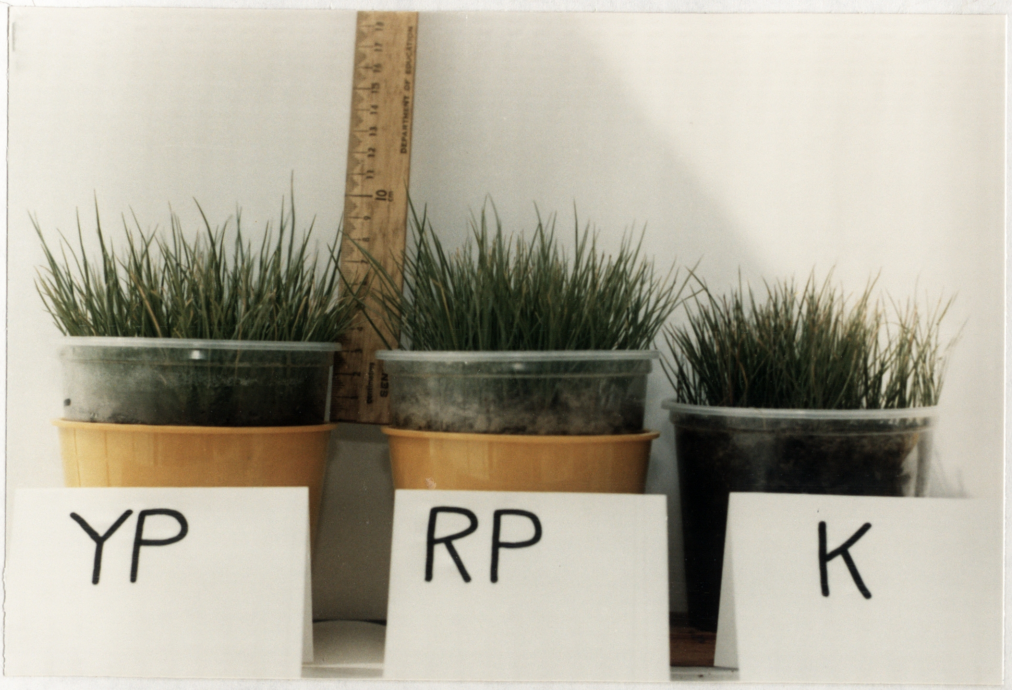
(YP=yellow podzolic soil; RP=red podzolic soil; wt=weight)

the yellow podzolic soil had more shoot dry matter than those grown in the red podzolic soil while root dry matter was similar. Generally, the effect of pre-planting incubation was minimal on dry matter yield. This is contrary to data in Figures 8.2a and 8.2b which indicate higher N removal from soils that had been incubated for long periods. Plant nutrient composition is discussed Section 8.3.4.

Table 8.3 is a comparison of plant dry matter yield from soils that were not incubated before planting. There was low dry matter yield from the krasnozem when compared with the other two soils, but it improved after liming. Plates 8.2a and 8.2b show that plants grown in the non limed krasnozem soils were shorter than those grown in the yellow and red podzolic soils. Liming of the krasnozem markedly improved the mass of plants. Several workers have suggested the beneficial effect of liming on the mineralisation of soil organic matter as a reason for lime responses in the field (Harmsen and van Schreven, 1955; Awad and Edwards, 1977; Edmeades *et al.*, 1981, 1986). Liming also reduces P sorption and toxic levels of Al and Mn, which in turn affect soil charge characteristics, availability of exchangeable cations and micronutrients, and soil physical properties (Haynes, 1984). These factors can all affect plant growth.

Plants often adapt to a P deficient soil by depressing top growth more than root growth so that the root:shoot ratio increases (Loneragan and Asher, 1967) thereby reducing nutrient demand by the shoot per unit of root. The root:shoot ratios in Table 8.3 show that P was most limiting in the krasnozem and least in the yellow podzolic soil, and that

8.2a



8.2b

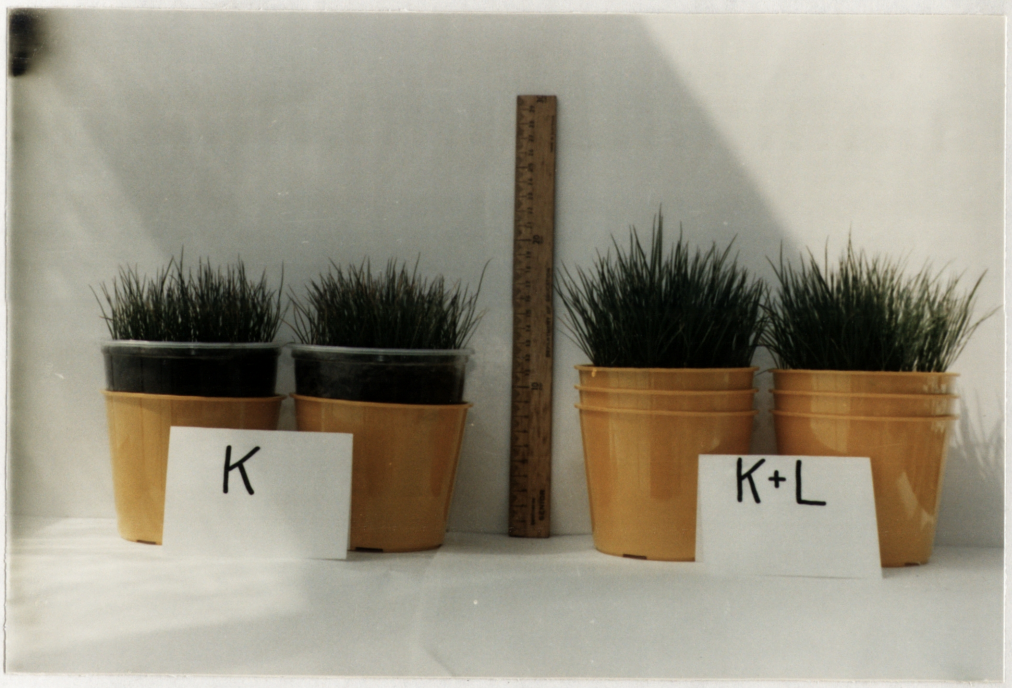


Plate 8.2 Size differences of plants grown in different soils (8.2a) and in the limed and non-limed krasnozem soil.

YP = yellow podzolic soil;

RP = red podzolic soil; K = krasnozem soil

K+L = limed krasnozem soil

liming reduced P limitation to plant growth.

Table 8.3 A comparison of dry matter (DM) yield from different soils. Means of 4 replicates.

Soil	Shoot DM (g)	Root DM (g)	Total (g)	Root:Shoot ratio
Yellow Podzolic	1.27	2.18	3.45	1.72
Red Podzolic	1.29	2.32	3.61	1.80
Krasnozem	0.63	1.27	1.90	2.02
Krasnozem (limed)	1.33	2.20	3.53	1.65

The soils were not incubated before planting

8.3.4 N and P content of the plants

Figure 8.4 compares the amounts of N and P in plants grown on soils incubated for various periods before planting. Nutrient units are based on nutrient uptake from 1 kg of soil for later comparison with mineralisation rates in soils. In both the yellow and red podzolic soils, extended periods of pre-plant incubation decreased P uptake but increased N uptake (Figure 8.4). A possible reason is that the P mineralised during long periods of incubation became fixed and unavailable for plant uptake, and the quality of mineralisable substrate was reduced so that mineralisation during the period of plant growth was reduced. Halm *et al.* (1971) found a decline in organic P in summer (due to increased mineralisation) which coincided with an increase in aluminium bound P. The more readily mineralisable organic matter present in soils incubated for shorter periods enabled P to be mineralised in the presence of plants and for it to be taken up before it was

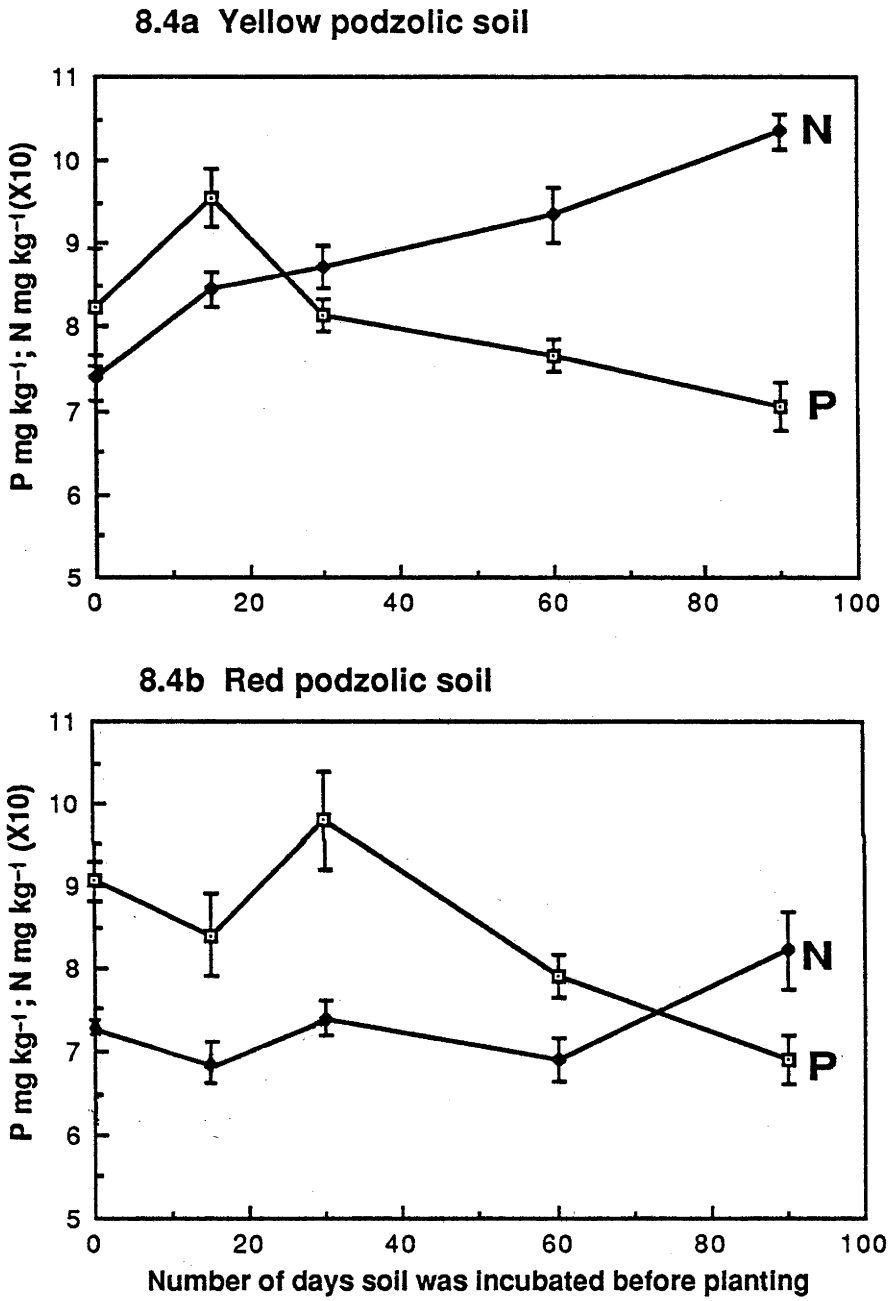


Figure 8.4 N and P taken up by plants from soil

fixed.

N uptake increased with longer periods of pre-plant incubation because it accumulated in the soils in plant available forms after mineralisation.

The amount of N and P taken up by plants grown in the three different soils and the limed krasnozem are shown in Table 8.4 (data on concentration basis are given in Appendix 3). Apart from the amount of N in the roots of plants grown in the limed treatment, there was more P or N harvested in the roots than in the shoots because there was more root material harvested.

Table 8.4 N and P uptake by plants grown in different soils
Means of 4 replicates.
KL not included in the statistics

Source of nutrient	Soil				p<	sed
	YP	RP	K	KL		
	Units: mg kg^{-1} soil					
Shoot N	28.4	31.9	32.6	64.0	ns	
Root N	44.9	42.2	50.0	57.4	0.01	1.4
Total	74.4	74.1	82.5	121.4	ns	
Shoot P	3.1	3.1	0.7	1.8	0.001	0.2
Root P	5.1	6.0	2.3	3.8	0.001	0.5
Total	8.2	9.1	3.0	5.5	0.001	0.6

YP = yellow podzolic; RP = red podzolic; K = krasnozem;
KL = limed krasnozem

P uptake was the same for the plants grown on the yellow podzolic and red podzolic soils and this was about three times that of plants grown on the krasnozem soil. Liming increased P uptake over the non limed krasnozem soil, but this was still

much less than for the yellow and red podzolic soils. The increase in P uptake due to liming may be due to various reasons. Liming encouraged root growth (Table 8.3), thus enabling a larger volume of soil to be exploited. The plants may also have benefited from increased mineralisation of organic P and from a decrease in Fe and Al ion concentrations as a result of liming. Grunnes (1959) observed that application of ammonium in association with phosphate seemed to encourage plant P uptake. Results in Chapter 7 showed that N mineralised in the krasnozem accumulated in the ammonium form even after liming. This ammonium may also have encouraged P uptake by plants.

N uptake was generally greater (significantly so in the root fraction;Table 8.4) in plants grown on the krasnozem soil than in those grown on the yellow or red podzolic soils confirming that P was the element most limiting growth on the krasnozem soil. Liming also increased N uptake as a result of greater N mineralisation and improved soil P availabilty for plant uptake.

8.4 Discussion

8.4.1 Relationship between plant uptake and the amounts of inorganic N and labile inorganic P in soils without plants after glasshouse incubation

This section compares the overall inorganic N and P (initial plus that liberated during incubation) in soils without plants, with N and P taken up by plants. This is to examine these values as a soil test parameter for plant N and P uptake. P values from soils samples that had not been

incubated at planting were used as a control to obtain net mineralised P for which sorption was corrected. These P values were 6.06 and 5.38 mg P kg^{-1} for the yellow podzolic soil and red podzolic soil respectively. The overall inorganic N and labile inorganic P in non-planted soils were compared with N and P taken up by plants and are given in Table 8.5. The relationships are shown in Figure 8.5.

Table 8.5 Comparison of inorganic N and P in soils without plants after glass house incubation with N and P harvested in plants. Means of 4 replicates.

Soil	Pre-glasshouse incubation period (days)	P in soil		Plant P	N in soil	Plant N
		P	P*			
		Units: mg kg^{-1}				
Yellow podzolic	0	7.19	7.88	8.2	42.2	74.4
	15	7.32	8.09	9.6	50.4	84.5
	30	7.33	8.11	8.1	57.9	87.1
	60	7.51	8.40	7.7	83.0	93.5
	90	7.72	8.73	7.1	96.2	103.5
Red podzolic	0	6.51	7.83	9.1	46.7	62.5
	15	6.24	7.24	8.4	48.4	59.5
	30	6.31	7.40	9.8	50.4	74.0
	60	6.31	7.40	7.9	54.1	69.1
	90	6.60	8.03	6.9	75.5	82.2

* Net P mineralised corrected for sorption using data from Section 7.3.5.4.

$NaHCO_3$ extractable P showed very little P mineralisation, and in some cases immobilisation with pre-glass house incubation of soils, and was therefore poorly related to plant P uptake (Figure 8.5a). There was a tendency for high plant P to be from soils that had been incubated for shorter periods, and as P accumulated was low in these soils,

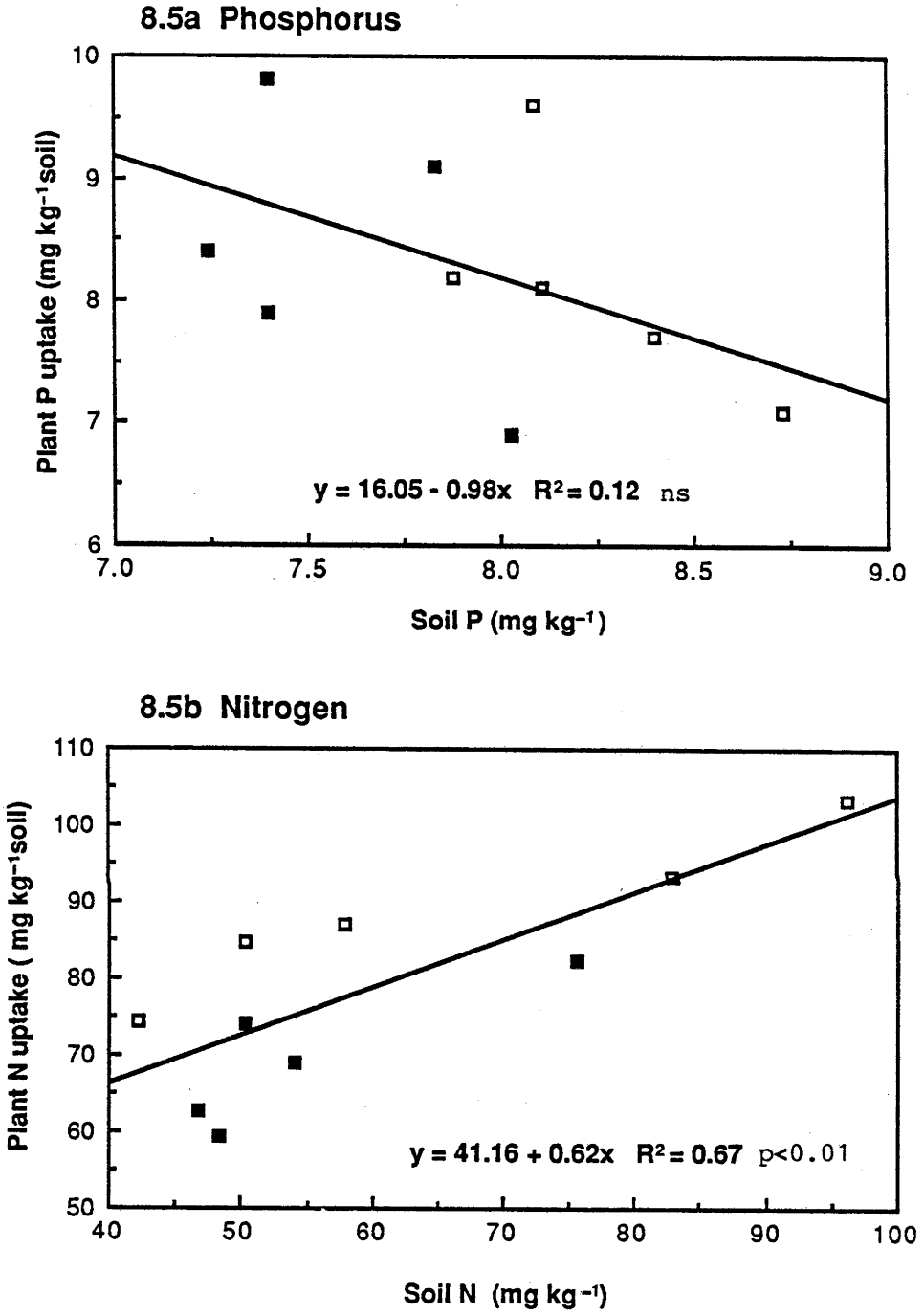


Figure 8.5 The relationships between nutrients (P and N) accumulated in soils that were incubated before planting and P and N taken up by plants after 40 days of growth. The open points represent data from the yellow podzolic soil, while the dark points are data from the red podzolic soil.

plant P uptake was negatively related to increased periods of incubation. The negative relationship between mineralised P and that taken up by plants is consistent with the observation in Section 8.3.4 that P was transformed into forms not readily available for plant uptake during long incubation periods without plants. This transformation occurred in both soils which had different P sorption characteristics. The effect seems to affect the root system as the the root mass declined with longer periods of incubation before planting. (Figure 8.3). Both N mineralised, and N taken up by plants, increased with pre-planting incubation periods and were significantly ($p<0.001$) related (Figure 8.5b).

8.4.2 The effect of plants on mineralisation of N and P

It has been suggested that the presence of plant roots may improve the availability of soil P (Barber, 1980). In this section, a comparison is made between the amounts of N and P in "plant available" forms present in soils in the presence and absence of plants. Table 8.6 shows the amounts of inorganic N and P "mineralised" in soils with and without plants during the plant growth period (40 days). Data was obtained as follows:

Net "mineralisation" = N or P uptake + final soil N or P
- initial soil N or P - seed N or P
(soils with plants)

= Final soil N or P - initial soil N or P
(soils without plants)

Table 8.6 Effect of plants on net "mineralisation" of soil N and P

Soil	Pre-plant incubation (days)	N (mg kg^{-1})		P (mg kg^{-1})		
		soil - plants	soil + plants	soil - plants		soil + plants
				A	B	
Yellow podzolic	0	10.9	24.3	1.1	1.8	3.2
	15	2.4	17.1	1.1	1.8	4.7
	30	-2.6	7.0	1.0	1.7	3.3
	60	14.9	7.0	1.1	1.8	4.5
	90	17.9	6.1	0.7	1.1	1.8
Red podzolic	0	17.3	19.0	1.1	2.5	4.7
	15	23.4	13.7	1.3	2.9	5.4
	30	28.9	26.6	1.0	2.2	5.6
	60	14.0	4.3	1.3	2.7	5.5
	90	25.9	13.0	0.8	1.7	2.7
Krasnozem	0	13.6	11.0	2.9	16.8	1.9
Krasnozem (limed)	0	108.6	91.1	4.1	24.1	4.0
Mean of all 12 cases		22.9	20.0	1.5	5.1	3.9
(se of means)		(8.2)	(6.8)	(0.3)	(2.1)	(0.3)

A = soil P not corrected for sorption; B = same soil corrected for sorption.

Table 8.6 shows that net N mineralisation in soils with plants was similar in magnitude to that in soils without plants. Sometimes more N was mineralised in the presence of plants but often the reverse was true. The overall means of "mineralised" N with and without plants were 20.0 and 22.9 mg kg^{-1}, and were not significantly different . As more N was taken up by plants grown in soils incubated for long periods before planting, the results in Table 8.6 suggest that fallow periods allowed better establishment of the rye grass.

However, tree seedlings have a lower establishment rate than rye grass, and the excess N could be lost through leaching. Fallow periods may therefore not be necessary in the establishment of a forest. This is also consistent with the loss of P to the plant through sorption associated with long periods of incubation (Section 8.4.1).

Except for the krasnozem (and its limed treatment), the amount of P mineralised in soil with plants was more than double that in soils without plants (Table 8.6). Correcting P for sorption on soils without plants increased P level slightly to about half that in soils with plants. It is possible that plant roots were able to take up the mineralised P before it could be fixed by the soil, or that there was increased P mineralisation due to the addition of readily assimilable substances excreted by the roots. Thompson and Black (1970) have shown that growing plants decrease organic P content of soils near the roots. In addition to releasing inorganic P from organic phosphates, microorganisms may also have the ability to dissolve insoluble inorganic P (Hayman, 1975) although the liberated soluble P may only contribute slightly to plant nutrition.

On the other hand, roots could have extracted mineral forms of P not measured in the bicarbonate extracts (labile P). Experiments have shown that plants may take up P from sources other than the labile pool ($NaHCO_3$ extractable P), such that the increase of P uptake in the presence of plants may not be wholly from mineralisation. The fall of rhizosphere pH causes dissolution of P compounds (Riley and Barber, 1971; Hedley *et al.*, 1982). Accordingly, plants have

been observed to take up P from the HCl extractable P pool (Hedley *et al.*, 1982; McLaughlin and Alston, 1986; Wagar *et al.*, 1986) suggesting that slightly soluble calcium and magnesium phosphates may have high potential availability to plants. It is also known that plant roots produce phosphatase enzymes which catalyse hydrolysis of some organic compounds (Bartlett and Lewis, 1973). These enzymes increase considerably under P deficiency (Bielski and Johnson, 1972).

In the krasnozem soil, root development was limited and less able to affect root functions that might increase P release in the soil. In addition, the soil's high sorption capacity would have limited P uptake. Liming is known to improve soil chemical and physical conditions for root development (Russell, 1973; Haynes, 1984) and consequently plant growth. However, even after liming P was strongly adsorbed before plant uptake as only 2.2 mg P kg^{-1} (Table 8.7) was taken up by plants out of the estimated 24.1 mg P kg^{-1} mineralised. This shows that plants may not use some of the amount of mineralised P estimated to be adsorbed. This is supported by the sorption studies (Section 7.3.5.4) which suggest that sorption of P may take place faster than the time needed for it to diffuse to the plant roots.

A comparison of the uptake of N and P by plants and the net change in soil inorganic N and P is shown in Table 8.7. Plant data were obtained as the difference between total plant and seed nutrients, while soil N and P were obtained as the difference between soils incubated with and without plants. The table shows that the change in soil N accounts for almost all N taken up by plants, while the change in P acccounts for

only about half of the plant uptake. In the unlimed krasnozem, plants only utilised seed P.

Given that a substantial amount of $NaHCO_3$ extractable inorganic P was still present in the soils after plant growth (Figures 8.2a and 8.2b), and that the amount of P taken up by the plants could not be accounted for by the difference in soil $NaHCO_3$ extractable P (Table 8.7), further work is needed on the compounds in the acid and other P fractions that may readily supply P to plants and on the redistribution of labile P to zones of depletion by roots.

Table 8.7 Contribution of soil inorganic P ($NaHCO_3$ extractable) and N to plant uptake

Soil	Pre-plant incubation (days)	Change in soil P	Plant P	Change in soil N	Plant N
			Units: mg kg^{-1}		
Yellow podzolic	0	2.7	4.9	38.2	45.9
	15	2.6	6.2	47.9	61.9
	30	2.5	4.8	55.1	64.7
	60	2.5	4.3	79.1	71.1
	90	2.5	3.7	92.9	81.1
Red podzolic	0	2.1	5.7	38.4	39.7
	15	1.7	5.0	46.9	37.0
	30	1.8	6.4	42.3	52.4
	60	2.2	4.6	56.4	46.6
	90	1.6	3.5	72.6	59.8
Krasnozem	0	2.9	00	43.9	60.0
Limed krasnozem	0	2.2	2.2	116.5	98.9

8.5 General conclusions on plant utilisation of N and P released by mineralisation

The experiments in this Chapter have shown that unlike N, P mineralisation under fluctuating temperature and soil moisture in the glass house was greater than that under undisturbed laboratory conditions. The differential effect of laboratory and glass house incubation on the rates of mineralisation of P and N is consistent with that shown by Birch and Friend (1961) where in a laboratory experiment, repetition of the wetting-incubation-drying cycle eventually led to complete mineralisation of organic P but only to partial (54%) mineralisation of organic N.

The difference between the amount of P uptake and the reduction in labile soil P during plant growth could be a result of a number of factors including increased mineralisation, reduced time of P contact with soil compounds between mineralisation and plant uptake, or the direct chemical and physical effects of roots and microorganisms. Unlike N, the diffusion of P to the root surface is a major limiting step to its plant uptake (Barley, 1970), and may also limit the use of the plant as a bioassay for P mineralisation. Lewis and Quirk (1967) calculated that after 4 days 95% of the P taken up by a root came from the soil solution within 0.1 mm of its surface. Therefore any P mineralised outside this zone would have little chance of being measured in the plant. In the short-term, however, root associations with microorganisms ensure that a significant amount of mineralised P is near the

root surface.

The implication of these experiments for forest management may be summed up by Khanna's (1981b) proposal that research is necessary to define phases of N (and P) demand by trees (especially seedlings), and to match the mineralisation of N and P with them. The experiments suggest that long periods of mineralisation in the absence of plants will lead to losses of P to the plant mainly through fixation while losses of N would be through leaching. As N and P uptake by pine seedlings is low at establishment (Neilsen *et al.*, 1984; Nambiar and Cellier, 1985), it is suggested that planting immediately after clearfelling would enable maximum utilisation of the mineralised nutrients.

CHAPTER 9 GENERAL DISCUSSION

CHAPTER NINE

GENERAL DISCUSSION

9.1 Introduction

Forestry practices have the potential to adversely affect the physical, chemical and biological characteristics of soils, including the processes of nutrient cycling. Thus it is important to minimise disturbance to forests through selection of management regimes based on the understanding of their long-term impacts. Quantification of changes in soil-nutrient supplying capacity associated with the various management practices would be one way of ascertaining their suitability.

In this thesis, field and laboratory studies have been used to examine the distribution and transformation of forms of P in contrasting forest soils under some different management regimes. The thesis was concerned with P at two broad levels; the fate of applied fertiliser P, and the nature of transformation of organic P. Both studies show that soil P sorption capacity is an important characteristic influencing the effectiveness of P as an applied fertiliser or released by mineralisation. The following discussion seeks to place the findings of this study within the broader framework of (i) soil analysis in forestry, (ii) the impact of fertilisation on forest soil, (iii) the forest floor as a source of P, and (iv) P turnover from organic matter in forest soils.

9.2 Soil analysis in forestry

Observations on some of the methods used in this study. The P analytical procedures used in the study were adapted from Hedley *et al.* (1982a,b), and a number of tests were carried out to validate some of the extraction and analytical methods of the procedures. Extraction of soil P with resin enclosed in a bag (Sibbesen, 1977) was adopted for use to minimise errors associated with preparation of moist soil samples for analysis. The resin method was used to test the effect of air drying forest soils on the amount of P extracted. Air drying of soils increased resin extractable P. The increases ranged from 1.5% to 61.8%, with high percentages being in soils with low resin extractabe P. These findingsare consistent with the conclusion that the use of air dry soil may give an over-estimation of plant available P levels in fresh soils (Sparling *et al.*, 1986). The source of this extra P may be derived from microorganisms killed by drying (Bowman and Cole, 1978a). Sparling *et al.* (1986) found the increase in $NaHCO_3$ extractable inorganic P of dry soil to be related to microbial biomass P of the moist soil. In this context, the assessment of soil microbial biomass P should be done immediately after soil sampling to minimise quantitative or qualitative changes in the microbial biomass during storage (McLaughlin and Alston, 1986).

Over-estimation of soil inorganic P could also occur when alkaline extracts, which contain high levels of organic matter, are analysed for P with certain methods. For example, analysis of P in NaOH extracts using the automated method of

Fogg and Wilkinson (1958) gave inorganic P values 8 to 42% higher than those obtained using the automated method of Murphy and Riley (1962). The higher P in the former method is due to the hydrolysis of organic P at the high temperature (95° C) used in the development of the phosphomolybdate colour complex. Most forest surface soils have high organic matter content, and care should be taken to minimise its contribution to the inorganic P fraction. This can be achieved by analysing the extracts at low temperatures, and by microfiltration or centrifugation of the neutralised extracts before measurement of P. The analysis of the organic P fraction is in itself important in forest nutrition studies as it is a source of P either through mineralisation or direct utilisation by mycorrhizae (Turner and Lambert, 1983).

The importance of soil analysis in forestry. Soil testing as a diagnostic method in forestry has been somewhat limited (Ballard, 1980) because of difficulties associated with obtaining representative samples from throughout the rooting zone and defining mobilisable forms of nutrients (Ballard, 1980; Khanna, 1981a). A wide range of tests for soil P status (e.g. water and resin extraction, Bray 1 and 2, Olsen, NH_4OAc, lactate, total P) have been found to be significantly correlated with tree growth or response under some conditions (Baur, 1959; Humphreys, 1964; Ballard, 1970; Ballard and Pritchett, 1975; Hopmans *et al.*, 1978; Kadeba and Boyle, 1978). Variation in soil properties and period of growth may influence the suitability of extractants used, and points to the need for a dynamic approach to soil analysis in forestry.

An example of how variation in soil P sorption

characteristics affect the rate of transformation of fertiliser P has been given in Chapter 5. The rate of loss of resin extractable P after fertiliser application was much faster in the high P sorbing yellow earth soil than in the lower P sorbing yellow podzolic soil. The importance of the different soil P fractions at different periods of growth is shown in the experiment by Ballard and Pritchett (1975) in which early response (1 yr) of planted or potted *P. elliotti* seedlings was best predicted by P in mild extractants (H_2O, NH_4OAc), while response over 3 to 5 years was best predicted by P in stronger extractants (0.5*M* $NaHCO_3$, 0.05*N* HCl + 0.025*N* H_2SO_4, 0.03*N* NH_4F + 0.025*N* HCl). In the first year of growth, only the current P uptake is involved in the response. In subsequent years, both current uptake and reserves accumulated in tissues from the previous years' uptake are involved, and include P mobilised from the P pool extractable by stronger extractants.

The results in Chapter 5 also emphasise the need for soil analysis as a diagnostic tool. For example, the most-labile P (resin extractable P) represented only 7% of the fertiliser P remaining in the yellow earth soil after 12 months of incubation compared with 20% in the yellow podzolic soil. An appreciation of such inherent differences in P retention could help in predicting the likely duration of response by plants to fertiliser P, and therefore in selecting P sources, methods and rates of application for efficient utilisation (Ballard, 1978d).

Again, soil analysis might be used to determine the effects of forest management practices on soil P. In this

study, the effects of cultivation, raking of litter, liming and fertiliser application have been quantified using soil analysis, and are discussed in the following Sections.

9.3 Impact of fertilisation on P in forest soils

When fertiliser P was added to the yellow podzolic soil (Chapter 4), some of it moved deeper than 20cm. It has also been reported in Chapter 5 that some of the fertiliser added to incubated soil cores was collected in water leaching through 10 and 20 cm deep cores. The highest concentration (9% of the added P) was in the leachate from the high P fixing yellow earth soil. P movement in the soil profiles was probably in solution (mass flow) percolating through the soil without allowing contact and reaction of P with soil surfaces. Logan and McLean (1973a,b) observed that with rapid leaching, soluble P dissolved from the soil surface layer could leach through the soil faster than the soil could adsorb it. Scotter and Kanchanasut (1981) also demonstrated that P can sometimes move deep in the soil profile, mainly through biogenic macropores such as root and worm channels, and incipient fracture planes. That the highest amount of P collected in the leachate came from the high P sorbing yellow earth soil can be attributed to preferential flow.

Rain and leaching water also caused redistribution, and therefore contact and reaction of the unleached P with soil compounds. This minimised any further movement with subsequent water flow in the soil profiles. According to Sample *et al.* (1980), a complex sequence of reactions follows

the addition of fertiliser P to soil, initially to metastable forms and eventually to stable reaction products. The field experiments in Chapter 4 showed that most of fertiliser P added to a yellow podzolic soil was initially in the labile inorganic P fractions which gradually declined with time. An exponential decline in P effectiveness for plant growth relative to freshly applied P was obtained in a study in which high temperatures were used to increase the rate of reaction between soil and P to give a range of effective periods (Barrow, 1974b).

Ballard (1978c) and Gentle *et al.* (1986) obtained large responses in a second rotation pine crop to a residual P applied to a first rotation crop 19 and 34 years previously. However, Ballard (1978d) warned against extrapolation of the results to other sites, as residual effectiveness of P fertilisers depends on a number of factors, including P retention capacity. The results in Chapter 5 in which soils with different sorption capacities were incubated, also support this. The rate of loss of the most-labile P in the yellow earth soil (high P sorbing) was more than 6 fold that in the yellow podzolic soil (lower P sorbing) during the second half of the incubation period, and therefore may not have a long term residual effect. This is reflected in a field study on a similar yellow earth soil. Snowdon and Waring (1985) obtained a response in foliar P concentration in the first 4 years of growth. However, after 4 years, foliar P concentration declined, suggesting that subsequent application of fertiliser might be necessary to maintain the high productivity of the treated plots. Mead and Gadgil (1978)

reported a similar example with a New Zealand soil of moderate P retention capacity. Foliar analysis indicated a repeat application was required only 3 years after an application of 120 kg P ha^{-1} as single superphosphate.

Ballard (1980) cited from various sources similar rates of broadcast fertilisation ranging from 50-100 kg P ha^{-1} at establishment and to established coniferous stands. Ballard proposed that the similarity in rates of fertiliser application was due to (i) the practice of adjusting the "frequency" of fertiliser applications rather than "rates" of individual applications to sustain response throughout a rotation, (ii) similar P requirement for most conifers, and (iii) the ability of established tree crops, irrespective of soil and climatic conditions, to conserve nutrients within their ecosystem by means of nutrient cycling. Sorption differences between soils (Chapter 5) and data by Snowdon and Waring (1985) suggest that the range 50-100 kg P ha^{-1} should not be taken literally. Rates of P application to sustain response very much depend on the soil's ability to retain P in plant available forms.

There have been reports of large growth responses to fertiliser P with different levels of fertiliser P recovered in trees. In the experiment by Gentle *et al.* (1965), only 2% of the 96 kg P ha^{-1} was estimated to be present in the biomass of the first rotation at harvest. Ballard (1978) recorded utilisation figures ranging from less than 1% to 4% of the P applied at 11 and 22 g P per seedling 3 years previously. Will (1965) estimated that 15% of the fertiliser P applied (224 kg P ha^{-1}) was in 25 year old radiata pine

trees and litter 8 years after application. Pritchett and Smith (1972) showed that 30% of 118 kg P ha^{-1} applied at establishment, could be accounted for in a 15 year old slash pine stand. It is clear that utilisation of fertiliser P by trees is different between soils. However, it is not clear from which soil fraction(s) P has been recovered, and more research needs to be done in this area. For example, in the report by Gentle *et al.* (1965), superphosphate transformed mainly into Al-phosphate while Ca-phosphate was maintained in the rock phosphate treatment. Both treatments continued to give similar yields after 34 years.

The low utilisation of fertiliser P by plants means that most P remains in the soil. Evoking Miller's (1981) proposal that fertilisers should be of benefit to trees, not the site, it is suggested that there is a need to search for means of improving fertiliser P utilisation, such as through manipulation of application methods, rates and frequency of application, and of using different sources of fertiliser. In soils where the initial P status is low, or the sorption capacity is high, the amount of P required to maintain a given level of plant available P may be prohibitive (Larsen, 1967). Under these circumstances, localised fertiliser placement has been used to restrict the amount of soil that the fertiliser contacts, in order for at least part of the growing medium to reach a satisfactory level. Alternatively, a given quantity of P may be spread in time to coincide with the different periods of nutrient demand by trees. This routine was advocated by Woods (1976) for the marginal *P. radiata* sites in the south-eastern region of South Australia.

A factor noted in this study which could have an important bearing on fertilisation of established stands is the chemical and physical ability of litter to retain fertiliser P, as demonstrated in Chapter 4. Maximum P sorption by litter from the first rotation pine (Chapter 4 was estimated to be 0.25 kg P $tonne^{-1}$. A large quantity of litter would be required to retain a significant amount of P by sorption. Therefore, the large amount of fertiliser P (about 50%) that was not recovered in soil on the first rotation pine site was mainly physically retained by litter. Fertiliser P applied to the litter surface could therefore result in its low utilisation, unless measures to wash the fertiliser through the litter are applied during fertilisation.

Recent studies (McLaughlin and Champion, 1987) have shown that sewage sludge exhibited characteristics of a slow release P fertiliser to ryegrass, possibly due to its slow release of P from organic to inorganic form. In the present study, a slow release of P to soil occurred under the wet sewage treatment on a yellow podzolic soil (Chapter 4). By comparison, air dried and ground sludge increased labile inorganic P immediately. The increase was more than 5 times that of wet sewage sludge. These responses suggest that sewage sludge can be an adequate source of soil P, and can be manipulated to vary the rate of supply of P to soil (and therefore to plants). When applied in fresh form, sewage sludge as a source of P may be beneficial to soils of both low and high P retention capacities, particularly in the duration of response. Willett *et al.* (1986) have demonstrated that the

efficiency of sewage sludge derived P can be increased by mixing sludge with soil.

It is worthwhile to note that interactions between response to P fertilisation and other soil properties and forest management practices may occur. Interactions attributable to moisture, rooting depth, soil bulk density, weed competition and other nutrients (especially nitrogen) have been reported (Ballard, 1978b; Pritchett and Smith, 1972; Waring, 1973; Woods, 1976).

9.4 The forest floor as a source of plant P

One strategy that has been proposed for avoiding a decline in the productivity of *Pinus* plantations on second rotation sites is the retention and management of logging residue and litter as a mulch and source of nutrients (Squire *et al.*, 1979; Woods, 1980). This is supported by the results of the slash/litter study described in Chapter 6. In the incubation study of soils from the different slash and litter treatments, the rate of increase in labile inorganic P was lowest, and labile organic P declined in soil from the raked treatment, compared with soil from treatments where litter was retained. Similar results have been reported from a field study on radiata pine in New Zealand where the soil had relatively high reserves of P and N, but the litter had been regularly raked from the site (Ballard and Will, 1981). Soil from the regularly raked treatment had lower concentrations of total N, total P, Bray 2 P and lower moisture, consistent with the removal of organic matter. Loss of productivity over 16

years as a result of raking was estimated at 12%. Crane *et al.*, 1981) also noted that in addition to reduced nutrient status, regular raking of a radiata pine plantation floor changes soil physical properties such as decreased time in which the soils set hard in dry periods, reduced infiltration of water, increased bulk density, and an increased range over which temperature fluctuates.

Growth of first rotation and second rotation stands of radiata pine were compared on sandy soils at Rennick in south-west Victoria (Squire *et al.*, 1979), where the second crop was established by hand planting into the residue of the first crop. The measurement of mean predominant (75 tallest trees ha^{-1}) height on both first and second rotation sites for 5 year old trees indicated that second rotation pine growth had not declined (Farrell *et al.*, 1981). This contrasts with earlier evidence of decline in yields of successive crops (e.g. Keeves, 1966; Bednall, 1968) when it was common practice to prepare sites by broadcast burning or by gathering and burning slash in heaps or rows. It is therefore desirable to retain litter and logging residue on site as a means of conserving organic matter, moisture and nutrients, irrespective of the soil's nutrient reserves. Ballard (1978b) has also suggested one way of ensuring continued response of P applied to the first rotation into the second rotation was through the return of P in slash and litter from the first fertilised crop. Fertilisation also results in higher nutrient concentrations in the litter (Crane *et al.*, 1981).

The accumulation rates of labile inorganic P (Chapter 6)

and mineral-N (Smethurst and Nambiar, 1986) in incubated soils from the three litter treatments are compared in Table 9.1. The labile inorganic P and mineral-N are for soils sampled at the one time. The highest accumulations of N and P were in soil from the cultivation treatment, and contrast with lower accumulations in soils from treatments where litter was retained without disturbance (litter) or removed (raked). This suggests that the mineralisation of N and P could be similar under certain management regimes and equally important in tree response. The importance of the mineralised P lies in the fact that the absolute requirements of P by trees is small and could possibly be met through mineralisation.

Table 9.1 Comparison of the mineralisation rates of N and P in siliceous sand soils under three litter treatments

Treatment	N	P	N:P
	kg ha^{-1} $month^{-1}$ (30 cm)		
Cultivated	30.3	0.9	33.7
Raked	12.7	0.3	42.4
Litter	7.9	0.6	13.1

A comparison of mineralisation rates of N and P in the different treatments (Table 9.1) shows that N was accumulating at a faster rate compared with P (high N:P ratio) in the raked treatment and at a slower rate in the 'litter' treatment. This lack of stoichiometric relationship was confirmed in the

experiments in Chapter 7 in which the ratio of N:P mineralised changed with time in individual treatments and was different between treatments. These findings support McGill and Cole's (1981) proposal that element cycling can best be interpreted within the framework of their stabilisation with carbon. In this system, N is stabilised as a result of covalent bonding with C (C-N) and is mineralised during C oxidation to provide energy, whereas P exists as an ester (C-O-P) and may be considered to be mobilised by the need for it. Therefore the mineralisation pattern of N in forest soils may not be linked to that of P.

The high rate of P and N accumulation in soil from the cultivation treatment reported in Chapter 6 was partly due to the maceration of litter and mixing of this and surface organic matter with the soil, which stimulates mineralisation (Johnston, 1982). The resulting increase in nutrient availability, and the reduction in weed competition and bulk density have been reported as being responsible for increased growth responses to cultivation (Waring, 1968; Squire, 1977; Lewty and Francis, 1982). However, cultivation may also cause rapid depletion of the mobile component of the organic matter, so that subsequent rates of P mineralisation may decline. This was demonstrated in Chapter 6 where lower rates of inorganic P accumulation occurred during the latter part of soil incubation compared with those in the early incubation period. Cassells (cited by Johnston, 1982) presented evidence supporting the hypothesis that the increased mineralisation following cultivation may result in the release of more N than can be utilised by the forest stand, with much of it being

lost through leaching. A corollary can be expected with P, where excess P may be mineralised but subsequently lost through leaching or fixation. This is consistent with Post's (1974) observation that in many instances the productivity increases attributable to cultivation can be short lived.

It has been observed that the release of inorganic P from plant material does not require that microbial decomposition take place (Martin and Cunningham, 1972)- so that conversion of plant P to inorganic P is partly an autolysis/release process. Pine litter on the site of the experiments reported in Chapter 6 consisted of 1.8 kg ha^{-1} of resin extractable P. Thus it was suggested in Chapter 6 that part of the initial P increase in the cultivation treatment was a direct result of extraction of P from litter mixed with the soil. This could cause a faster increase in soil inorganic P after cultivation than the net mineralisation which depends on the turnover of microbial P. The direct contribution of P to soil from organic residues that results from cultivation is a factor that has not been given much consideration in interpretation of growth responses.

9.5 Mineralisation of organic P in forest soils

Low values of labile P accumulating in soils (ranging from 1.7 to 2.7 mg P kg^{-1} during 157 days of incubation) as a result of mineralisation of soil organic P are reported in Chapter 7. Nevertheless, in studies on P forms in a range of northern New South Wales forest soils (Kelly *et al.*, 1983), and in a long term *P. radiata* phosphate fertiliser trial

(Turner and Lambert, 1985), it was acknowledged that organic P could be the most important form of this element in maintaining long term forest productivity. Only a small proportion of soil organic P needs to be mineralised to provide significant quantities of inorganic P for plants (Greb and Olsen, 1967). For example, a net mineralisation rate as low as 0.3 kg P ha^{-1} $month^{-1}$ obtained in the raked treatment (Chapter 6) was still quantitatively greater than the rate of P uptake by seedlings of *P. radiata* (0.2 kg P ha^{-1}, 8 months) recorded by Neilsen *et al.* (1984). However, microbes exist in relatively close association with the entire nutrient substrate in litter and soil, but plants can only exploit that part of the soil immediately sorrounding their roots. Otherwise plants must rely on diffusion and other environmental processes to transfer P (and other nutrients) outside the rhizosphere. It can therefore be expected that not all mineralised P in soil is available for tree uptake, and research is needed to establish its utilisation.

It has been shown in Chapter 7 that the krasnozem soil with low pH (4.8) mineralised less N and P (91.6 and 10.0 mg kg^{-1}, respectively) in proportion to the high soil organic C (9.3%) during 157 days of incubation, compared with the yellow podzolic soil (pH=5.9, N and P mineralised = 79.9 and 4.0 mg kg^{-1} respectively, organic C= 1.4%). Similarly, Harrison (1983) found a positive correlation between soil pH and the rate of organic P mineralisation for 50 woodland soils with pH ranging from 3.1 to 7.5. The converse is that decreasing pH appears to make organic P more stable (Enwezor, 1967) through formation of Al and organic matter complexes (Hargrove and

Thomas, 1981) and low microbial populations and activity (Alexander, 1980).

Forest soils tend to be acidic (Pritchett, 1979) and this may lead to slow rates of P cycling. This may be alleviated through liming. Liming the acid krasnozem soil (Chapter 7) increased both N and P accumulated in soil during incubation by a factor greater than 3. However, whether the effects of increasing pH with lime on P is comparable with pH differences in natural soils needs to be further researched. The increase in microbial activity after liming was short lived and declined to below that of the unlimed soil after about 50 days of soil incubation. The rate of inorganic N and P accumulation also declined with time. There is evidence that the increase in N mineralisation is mainly restricted to the first year after liming of acid soils from cultivated and wooded sites (Nyborg and Hoyt, 1978), and this may equally apply to P. Black (1968) attributed the short term effect on mineralisation to a portion of organic matter that becomes susceptible to mineralisation after liming. When this portion is used up, mineralisation returns to near original levels. However, liming may affect soil P in other ways, such as increased solubility of Al phosphates (Russell, 1973), which were not examined in this study.

One factor that may lead to immobilisation of large amounts of P in forest soils is the continuous addition of litter with wide C:P ratios (Pritchett, 1979). The effect of wide C:P ratios on P immobilisation was demonstrated in Chapter 7 where the addition of glucose led to an increase in the microbial biomass P and a decrease in soil labile

inorganic P. Part of the P so immobilised may be converted to organic compounds resistant to decomposition (Anderson, 1980; Martin, 1964).

In the glass house incubation of soils (Chapter 8), it was observed that biomass P declined in soils, and that the decline was greater in soils without plants than in soils with plants. It is possible that the excretion of readily assimilable substances by roots (Dalal, 1977) maintained a higher microbial population in soils with plants. The reaction of microorganisms to the addition of nutrient substrates, especially readily assimilable substrate C (Jansson, 1971), has been used to explain stimulation of microflora that mineralise organic P in soil near plant roots. This rhizosphere population could have increased mineralisation of organic P, thereby contributing to P uptake by plants that could not be accounted for by P mineralised in soil without plants. However, it has been shown that plants may take up P from 1.0*M* HCl extractable inorganic P (Hedley *et al.*, 1982; McLaughlin and Alston, 1986), which has been termed 'more stable inorganic phosphorus composed probably of apatites' (Stewart and McKercher, 1982). Indeed, Black (1968) suggested that all forms of P in soils are of some significance in supplying P for plants on a long-term basis. Thus P fractionation to show changes in different P pools during tree growth may provide a better appreciation of the relative value of different P fractions in the nutrition of trees.

9.6 Summary of general discussion

Studies of the P characteristics of Australian forest soils are few, and there is much scope for future research. On forest fertilisation, there is need for research towards increasing effectiveness of applied P through rationalising the method, rate and timing of application. Soil studies that identify the P fraction to which P is converted, and from which plants obtain some of their requirements, are of significant value in this area. This study has provided evidence suggesting that the amount of fertiliser P remaining in the labile P pool ($NaHCO_3$ extractable P) could be a good indicator of residual P effectiveness. The study has also provided some basic data on the effects of some silvicultural practices on the fluxes and pools of N and P in several forest soils. Soil measurements gave net P mineralisation values that were quantitatively adequate for plant nutrition during the early (seedling) growth phase. However, large fertiliser P responses reported in a number of experiments suggest that trees may be unable to obtain all their P requirements from the mineralised pool, either due to physical inability to reach all the P mineralised, or to most P being sorbed before uptake takes place. Future research should aim to provide information on the relationship between the rate of release of P (and other nutrients) and tree uptake especially during different stages of growth.

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APPENDICES

Appendix 1a: Soil phosphorus extraction using the anion exchange resin enclosed in bags (Sibbesen, 1977, 1978).

Conversion of the resin to bicarbonate form.

The resin used in the experiments was 'Dowex' 1-X8(CI), TAEC=2.7 me/g. The cloth used was Nytal Brand 45GG, with a mesh opening of 0.4mm. The amount of resin used was 4.0 g (60° C) per bag. It had been washed over the cloth material to get rid of the small sized beads.

1. Transfer bags containing resin (resin bags) into a vessel containing 0.5*M* $NaHCO_3$ solution, with an amount equal to 100 ml per bag
2. Stir from time to time for 30 minutes
3. Repeat the treatment in a fresh 0.5*M* $NaHCO_3$ solution
4. Wash twice with distilled water, and allow to drain for 10 minutes before use

Extraction procedure

1. Weigh 4 g air dry, 2mm sieved soil into extraction bottle
2. Add 100 ml distilled water
3. Add resin bag (containing the bicarbonate treated resin)
4. Stopper the bottle and shake end over end for 17 h at 25° C
5. Discard the soil suspension

6. Rinse the bag and bottle with distilled water to get rid of the remaining soil. Invert to drain for 10 minutes.

A few resin bags are weighed out, drained overnight at 60° C and weighed again to determine the amount of moisture remaining in the resin bag at step 6.

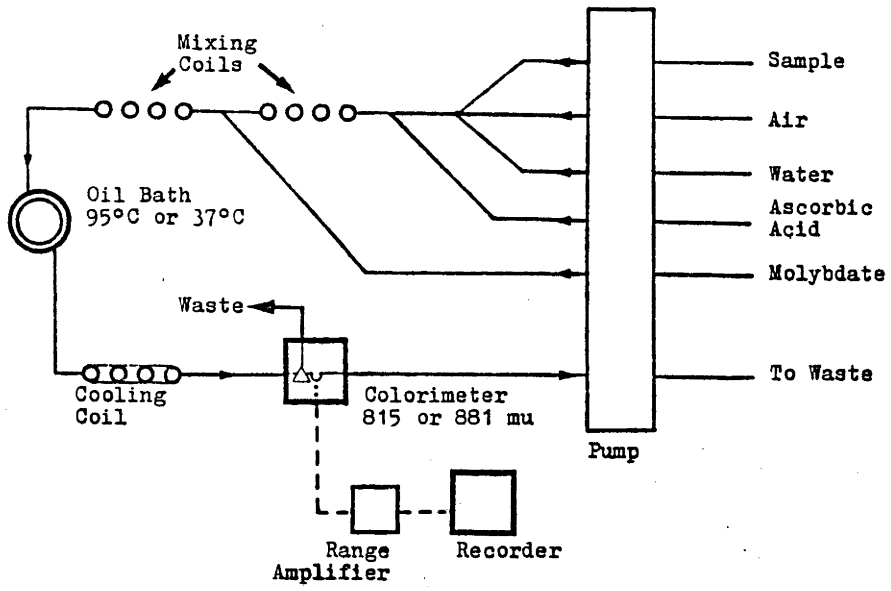
7. Add 80 ml of 0.5*M* HCl
8. Allow the release of CO_2 to subside, stopper and shake end over end for 1 h.
9. Determine P in the acid eluate (now about 0.4*M*).

Note. The volume of the eluate = 80ml + amount of water in resin bag after Step 6.

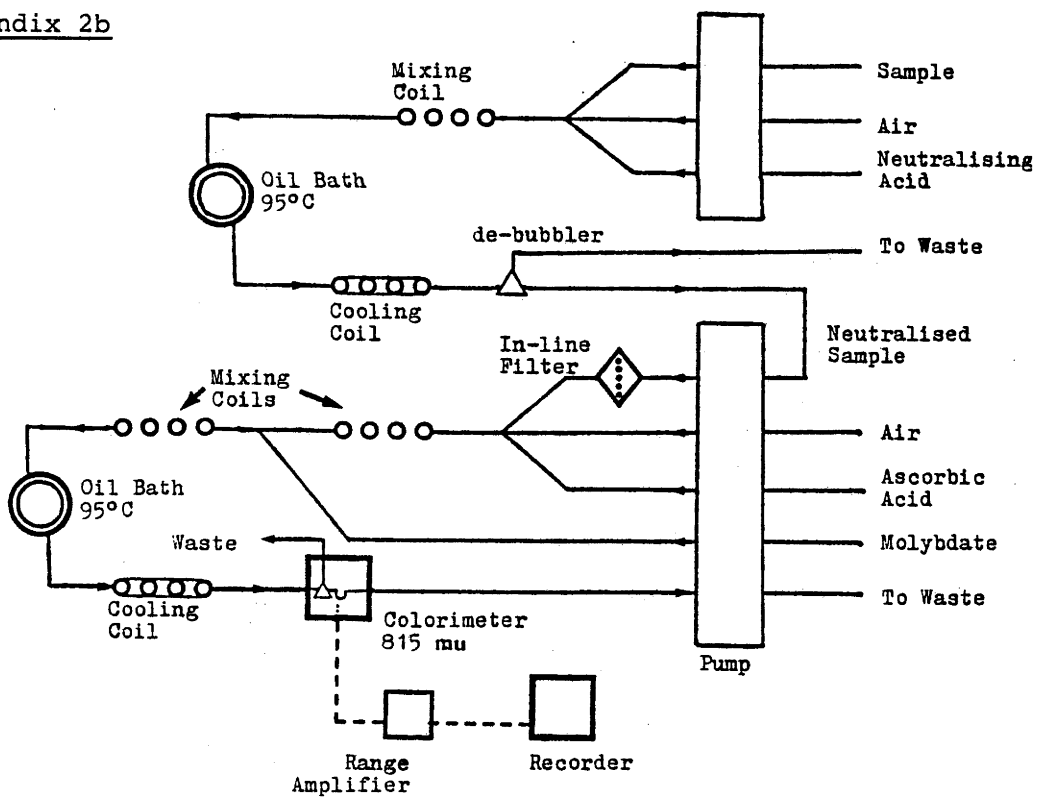
Appendix 1b: Sodium bicarbonate extraction of soil phosphorus (Colwell, 1965)

1. Weigh 1.0g air dry, 2.0 mm sieved soil into a 150 ml conical flask
2. Add 100ml of 0.5*M* $NaHCO_3$ solution, at pH of 8.5
3. Shake (wrist action) for 16 h at 25° C
3. Filter through two filter papers (Whatman No 42)
5. Determine P in the filtrate using autoanalyser (Appendix 2b)

Appendix 2a



Appendix 2b



Appendix 2 Schematic flow diagram for determining P in soil extracts (2a) and that adapted for $NaHCO_3$ extracts (2b).

Appendix 3

Nutrient ($mg\ g^{-1}$)	Number of days soil was incubated before establishment of glasshouse bioassy				
	0	15	30	60	90
YELLOW PODZOLIC SOIL					
Shoot P	1.23(0.06)	1.10(0.06)	1.01(0.01)	1.02(0.05)	0.91(0.00)
Shoot N	11.57(0.48)	12.88(0.49)	14.02(0.28)	14.63(0.66)	18.29(0.47)
Root P	1.17(0.01)	1.23(0.02)	1.22(0.03)	1.13(0.03)	1.08(0.02)
Root N	10.40(0.45)	9.58(0.40)	10.63(0.73)	11.91(0.32)	12.32(0.17)
RED PODZOLIC SOIL					
Shoot P	1.20(0.03)	1.16(0.06)	1.23(0.08)	1.11(0.07)	1.04(0.02)
Shoot N	12.50(1.09)	10.82(0.13)	10.39(0.23)	12.84(0.41)	16.47(0.30)
Root P	1.29(0.01)	1.35(0.06)	1.43(0.07)	1.19(0.03)	1.07(0.03)
Root N	9.12(0.20)	7.98(0.34)	8.69(0.56)	8.44(0.23)	10.72(0.63)
KRASNOZEM (NON-LIMED)					
Shoot P	0.52(0.02)				
Shoot N	23.58(0.21)				
Root P	0.83(0.04)				
Root N	18.65(0.41)				
KRASNOZEM (LIMED)					
Shoot P	0.66(0.01)				
Shoot N	24.03(0.79)				
Root P	0.85(0.05)				
Root N	13.00(0.72)				

Nutrient concentrations in plant shoots and roots after 40 days of growth in soils that had been incubated for different days before planting. se of means in brackets, n=4.

Appendix 4

Appendix 4a: CLIMATIC DATA FOR THE COTTER CATCHMENT AREA

Climatic data relevant to the Cotter Catchment Area are available from three stations - Pierce's Creek Forest Office (35° 20'S, 148° 55E), Yarralumla (36° 18'S, 149° 06'E and the BFG site (Site 1, Figure 3.1). A summary of the data is given in the following Tables.

1. Climatic data for Yarralumla and Pierce's Creek Meteorological Stations

Yarralumla Forestry (mean 1928-1979)

	J	F	M	A	M	J	J	A	S	O	N	D	Total
mean rainfall (mm)	61	61	60	54	49	43	43	52	52	75	57	53	659
mean max. temp. (°C)	28	27	24	20	15	12	11	13	16	19	23	26	
mean min. temp. (°C)	14	14	12	8	4	2	1	2	4	7	9	12	
mean sunshine (h)	8.9	8.3	7.4	6.9	5.6	4.8	5.2	6.2	7.3	7.9	8.8	9.1	
no. of frosts	0	0	0	3	12	17	20	17	11	4	1	0	

Pierce's Creek (mean 1929-1982)

	J	F	M	A	M	J	J	A	S	O	N	D	Total
mean rainfall (mm)	67	62	68	61	58	57	63	73	67	86	69	60	791

2. Total annual rainfall at Pierce's Creek Forest Office during the life of the BFG experimental stand

Year	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986
Rainfall	978	1239	967	736	700	957	474	636	883	359	1142	1072	797	690

3. Monthly rainfall for BFG site for 1982-83 to 1986-87 growing seasons (for Pierce's Creek prior to Oct. 1983)

Growing Season	J	A	S	O	N	D	J	F	M	A	M	J	Total
1982-83	8	13	89	14	1	19	25	47	107	66	198	50	637
1983-84	63	115	100	116	164	96	385	50	35	105	23	4	1256
1984-85	117	105	80	79	37	23	2	7	210	25	66	32	783
1985-86	48	152	80	70	79	36	52	3	1	70	78	11	680
1986-87	156	66	46	117	100	21	21	76	26	17	54	65	765
Mean*	63	73	67	86	69	60	67	62	68	61	58	57	791

* 30 year means for Pierce's Creek

Appendix 4b: SOIL PROFILE DESCRIPTIONS AT SITES 1, 3, 4 & 5 (FIGURE 3.1)

SITE 1: yellow podzolic soil (Dy 3.61)

Parent Material: adamellite (coarse grained granite) of Silurian age; part of Murrumbidgee batholith.

Vegetation: originally eucalypt woodland; second rotation of P. radiata planted in 1973.

Profile Drainage: poor in B horizon

Morphology:

Hor.	Depth (cm)	Description
A1	0-10	10 YR 6/4, organically enriched coarse loamy sand; merging into -
A2	10-45	very bleached at 15 cm, 10 YR 6/3 at 30 cm; coarse loamy sand. Waterlogging at bottom
B	45-80	mottled (7.5 YR 6/6, 10 YR 6/4); gritty light clay or coarse clay loam; apedal yellow massive unstructed coarse material.
BC	80-100	gritty loam, gravel increasing

SITE 3: red podzolic soil (Dr 2.21)

Parent Material: Silurian volcanics (main constituents dacite and tuff) - Paddys River member.

Vegetation: originally eucalypt forest; second rotation of P. radiata naturally regenerated about 1974.

Profile Drainage: reasonable, mid-slope location

Morphology:

Hor.	Depth (cm)	Description
A1	0-10	10 YR 5/2, loam, hardsetting
A1	10-20	10 YR 6/3, loam
A2	20-35	10 YR 6/6, loam, bleached
B	35-80	5 YR 4/6, light clay, pedal small blocky, stone mottling
C	80+	2.5 YR 4/6, medium clay, greyish yellow weathered stone

SITE 4: yellow earth soil (Gn 2.21)

Parent Material: metamorphosed (low Ca) sediments of Ordovician age, Nungar beds.

Vegetation: originally dry sclerophyll eucalypt forest. The failed first rotation pine plantation has been invaded by eucalypts.

Profile Drainage: adequate

Morphology:

Hor.	Depth (cm)	Description
A1	0-20	7.5 YR 5/6, silty loam
A2	20-30	7.5 YR 5/6, silty loam, unbleached
B	30-65	7.5 YR 5/8, silty clay loam, stoney
C	65+	7.5 YR 5/8, silty clay loam, very stoney

SITE 5: krasnozem soil (Gn 4.11)

Parent Material: metamorphosed sediments of Ordovician age, Nungar beds

Vegetation: subalpine eucalypt forest of Eucalyptus pauciflora and E. dalrympleana with an understorey dominated by Poa sp. (snowgrass) and Daviesia mimosoides.

Profile Drainage: good

Morphology:

Hor.	Depth (cm)	Description
A	0-10	10 YR 2/2, organic loam, worm casts
A	10-30	5 YR 3/3, loam, massive
A	30-40	5 YR 3/3, light clay loam
B	40-50	5 YR 4/7, light clay loam
B	50-70	5 YR 4/7, light clay, massive

Appendix 4c: CLIMATIC DATA AND SOIL PROFILE DESCRIPTION AT CAROLINE FOREST SITE, MT. GAMBIER

Long term averages of rainfall and temperature at Mt. Gambier

	J	F	M	A	M	J	J	A	S	O	N	D
Rainfall (mm)	34	29	36	63	86	99	106	100	78	64	46	41
Temperature (°C):												
Daily mean max.	26	25	23	20	16	14	13	14	16	18	20	23
Daily mean min.	11	11	10	9	7	5	5	5	6	7	8	10

Soil profile description at the study site

Soil: siliceous sand (Uc 4.21)

Parent material: aeolian

Vegetation: second rotation *P. radiata*, planted in 1947

Profile drainage: well drained

Morphology:

Hor.	Depth (cm)	Description
A11	0–15	Dark grey sand; 7.5 YR 4/0
A12	15–30	Grey sand; 7.5 YR 6/0
A13	30–40	Light grey sand; 7.5 YR 7/0
A2	40–100	Pale orange sand; 7.5 YR 8/6
B1	100–130	Dark orange sand; 7.5 YR 6/8
B2 hnx	130–170	Dark orange sand; 7.5 YR 6/8; with small amounts of loose ironstone gravel < 1 cm diam. and organic staining; weak to moderately cemented
	170–190	Bleached sand; 7.5 YR 8/0
	190–(200)..	Bleached sand with thin lenses of grey mottled pale orange structureless sandy clay and clayey sand