

Developing an index for phosphorus loss from sugarcane soils in Mauritius

**by
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DECLARATION

I declare that this thesis submitted by me for the Philosophiae Doctor degree at the University of the Free State is my own independent work and has not previously been submitted by me at another University. I furthermore cede copyright of the dissertation in favour of the University of the Free State.

A handwritten signature in black ink, consisting of a stylized 'S' followed by a dot and a horizontal line.

Signature

January 2015

PUBLICATIONS

The following publications have originated from research findings of this doctoral study and are as follows:

➤ Conference contributions:

1. **Mardamootoo, T.** and Du Preez, C.C. (2015). The need for a management tool to assess risks of phosphorus transport from sugarcane fields of Mauritius. Paper to be presented during the Agricultural Engineering, Agronomy and Extension workshop of the International Society of Sugarcane Technologists to be held in Durban, South Africa, August 24th-28th, 2015. (*Abstract submitted*)
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5. **Mardamootoo, T.**, Ng Kee Kwong, K.F. and Du Preez, C. C. (2011). Evolution of the agronomic and environmental phosphorus status of soils in Mauritius after a seven year sugarcane crop cycle. Sugar Tech 14(3): 266-274.
6. **Mardamootoo, T.**, Ng Kee Kwong, K.F. and Du Preez, C. C. (2010). History of phosphorus fertiliser usage and its impact on the agronomic phosphorus status of sugarcane soils in Mauritius. Sugar Tech 12(2). 91-97.

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ABSTRACT

Developing an index for phosphorus loss from sugarcane soils in Mauritius

Sugarcane is the major crop cultivated in Mauritius and currently occupies some 53,500 hectares of land with an average cane productivity of 74 t ha⁻¹. Phosphorus (P) fertilization on deficient soils can increase annual cane production up to 24 t ha⁻¹. Currently the sugarcane industry consumes some 3,350 tonnes of P₂O₅ in the form of soluble mineral fertilisers with an additional 1,110 tonnes P₂O₅ coming from filter muds. While the application of P to agricultural land is essential in maintaining crop productivity, non-point sources of P leaving the agricultural landscape can cause accelerated eutrophication of surface waters, thereby impairing water quality. Although these losses may not always be of economic importance to farmers, only small amounts of P can trigger eutrophic conditions in freshwaters. Past studies in Mauritius have shown that runoff P concentrations of limnological significance (> 0.1 mg P L⁻¹) can occur, particularly during high flow events. Hence, this study was initiated to provide a better understanding of P mobilization from cane fields and to integrate factors influencing P movement into a P risk assessment tool. To achieve these objectives, rainfall surface runoff simulations were conducted on 20 sites representing the main soils under which sugarcane is cultivated. The tests were done on runoff plots (2.1m by 0.75m in duplicate) at three slope categories (0 to 8%, 8 to 13%, 13 to 20%) and under three different rainfall intensities (50, 100, 150 mm hr⁻¹) for a duration of 30 minutes each.

In an attempt to evaluate the impact of continuous fertilizer application on the environmental soil P status of sugarcane soils in Mauritius, it was found that 0.01M CaCl₂-P of surface soils was linearly correlated to dissolved runoff P losses ($r^2 = 0.92$). Thus simple laboratory soil extractions with 0.01M CaCl₂ is a suitable estimate for dissolved P losses when field experimentation is not possible. Since the use of the routine agronomic soil test (0.1M H₂SO₄ soil extraction) for sugarcane in Mauritius provided a more accessible analytical tool for P management, the relationship between the 0.01M CaCl₂-P and 0.1M H₂SO₄-P was established and it was found that soils with 0.1M H₂SO₄-P above 160 mg kg⁻¹ can potentially impair runoff water quality. While soil testing provides a reliable pointer of desorbable P in surface soils, it gives no indication of its potential for transport during runoff and erosion.

Results from the simulation studies showed that runoff and erosion potential varied across the different soil types. It was also observed that with increasing rainfall intensities and field slopes, P mobilisation was enhanced due to increases in runoff and erosion rates. The results further showed that total runoff P was more strongly correlated with suspended sediments ($r^2 = 0.92$) present in runoff waters than with runoff volume ($r^2 = 0.49$) indicating that a greater proportion of the P transported in

runoff occurred mostly as particulate P rather than dissolved P. Actually, about 90% of total P loss in runoff waters was mobilised in particulate forms regardless of field soil type, rainfall intensity and field slope.

Using these research findings and historical data, the P index was developed to rank site vulnerability to P loss by accounting for source (dissolved P, particulate P, P application rate, method of application and application timing) and transport factors (soil erosion, surface runoff potential and precipitation) such that site-specific management practices can be implemented to critical source areas to minimize offsite P export. The proposed improved management practices to reduce P loss from fields in the P index include terracing, construction of diversions, field borders, field strips, grassed waterways, forest buffers and herbaceous cover.

Sensitivity analysis and edge-of-plot field testing were used to assess the behavior and performance of the P index. The results indicated that further evaluations at a watershed scale would be more insightful about the strengths and weaknesses of the P index as a risk assessment tool. Besides, further evaluations of this tool will eventually lead to improvements in estimating the impacts of agricultural P management on downstream water quality.

Keywords *critical source areas, diffuse pollution, eutrophication, phosphorus losses, sensitivity analysis, simulated rainfall, soil erosion, soil phosphorus testing, surface runoff, topography*

1 Introduction

1.1 Background

Phosphorus (P) is an essential nutrient for both plants and animals having, as reviewed by Higgs *et al.* (2000) an irreplaceable role in many physiological and biochemical processes. In most soils there is only a meagre supply of plant available P and hence the use of P fertiliser to remove P supply as a limitation to growth has long been practiced to achieve profitable crop production. Sugarcane production in Mauritius has been no exception. Indeed, five yearly averages of fertiliser P usage by the Mauritian sugarcane industry showed that from the 790 tonnes of P_2O_5 used at the beginning of the 20th century, fertiliser P usage peaked at of 5,675 tonnes in the 1970s (Figure 1.1).

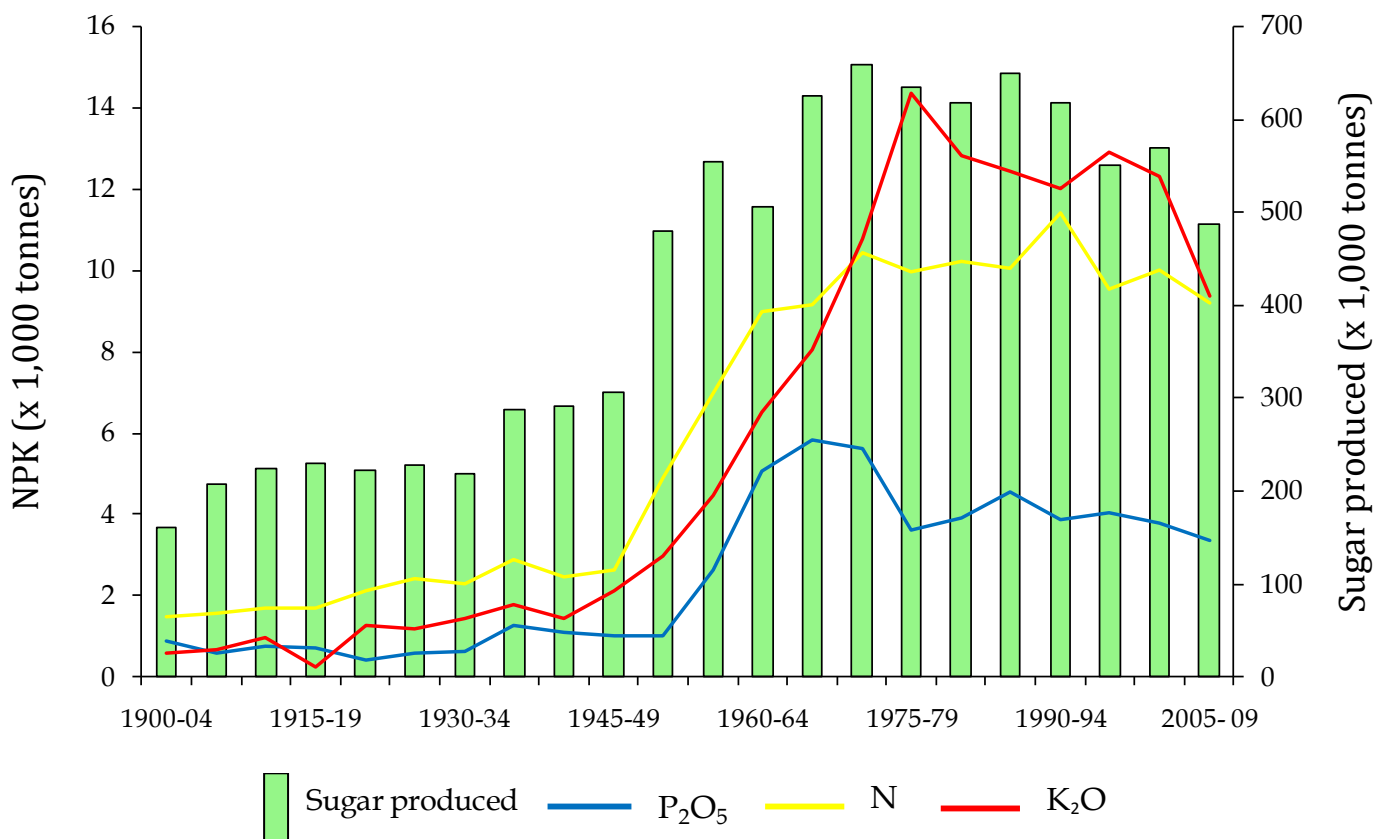


Figure 1.1: Five yearly averages of sugar production and NPK fertiliser usage in sugarcane soils of Mauritius from 1900 to 2009 (Mardamootoo, 2009).

It declined thereafter, not because it is less intensively used but due to a decreasing land area under sugarcane as indicated in Figure 1.2 (Mardamootoo, 2009). In fact, average P fertiliser rates have

risen from 13.6 kg ha⁻¹ P₂O₅ in the 1950s to 49.4 6 kg ha⁻¹ P₂O₅ in 2009. During the period 2005 to 2008, an average of 3,350 tonnes of P₂O₅ was applied annually to sugarcane fields mostly in the form of ammonium phosphates.

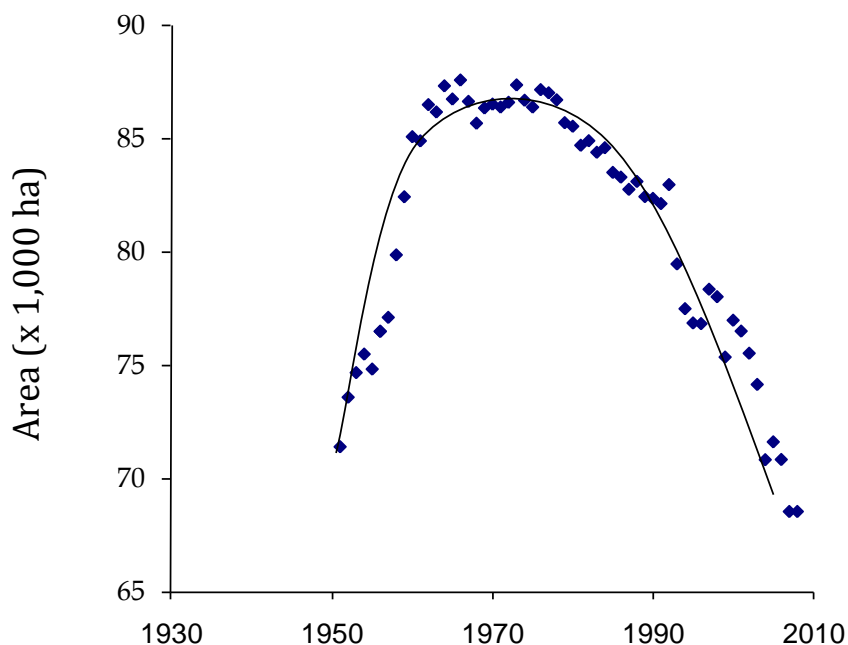


Figure 1.2: Evolution in area under sugarcane in Mauritius since 1951 to 2009 (Mardamootoo, 2009).

As reviewed by Chen *et al.* (2008), an intensive use of P fertilisers invariably results in an accumulation of P in the soil. Mardamootoo (2009) showed that in Mauritius approximately 52% (i.e. 32,000 hectares) of sugarcane lands replanted in 2005/2006 contained more P (i.e. $P \geq 100\text{ppm}$) than what was actually needed by the sugarcane crop (Figure 1.3). With the foreseeable continuing intensive use of P fertilisers, the area of land with levels of P in excess of those required for sugarcane growth will go on rising in the future if no remedial measures are taken. While this accumulation of P can be a long-term residual pool of P for crops, there is growing concern in Mauritius about its possible effect on surface water quality. Hence, laboratory experiments were initiated by Mardamootoo (2009) with the aim of extending the scope of the 0.1M H₂SO₄ extraction to also assess the resulting impact of the intensive use of fertilisers on the environmental P status of the main soils under sugarcane.

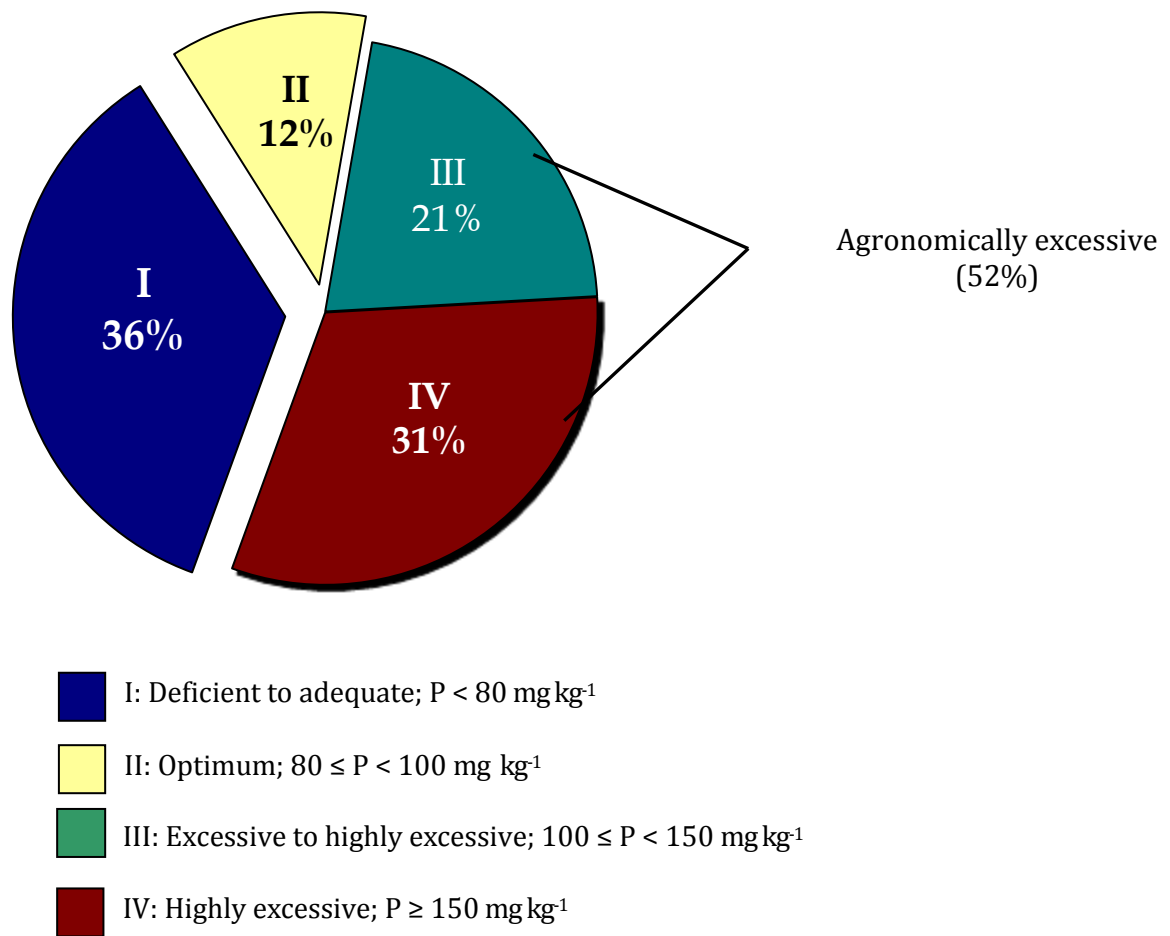


Figure 1.3: Phosphorus status of Mauritian sugarcane soils in 2005/2006 after categorisation into four arbitrary classes; no P fertiliser is recommended if these soils contain more than 80 mg P kg⁻¹ using the 0.1M H₂SO₄ extraction (Mardamootoo, 2009).

The laboratory data obtained showed that the agronomic threshold range of 80 to 100 mg P kg⁻¹ overlaps the environmental critical range of 85 to 95 mg P kg⁻¹ (Figure 1.4). This implies that soils in Mauritius that are agronomically suitable for sugarcane production are unsafe from the freshwater protection viewpoint. More specifically, on the basis of soil testing the excess P present in the 52% (32,000ha) of sugarcane lands of the island may impair the quality of the existing freshwaters and eventually the biodiversity of those resources if no remedial measures are taken.

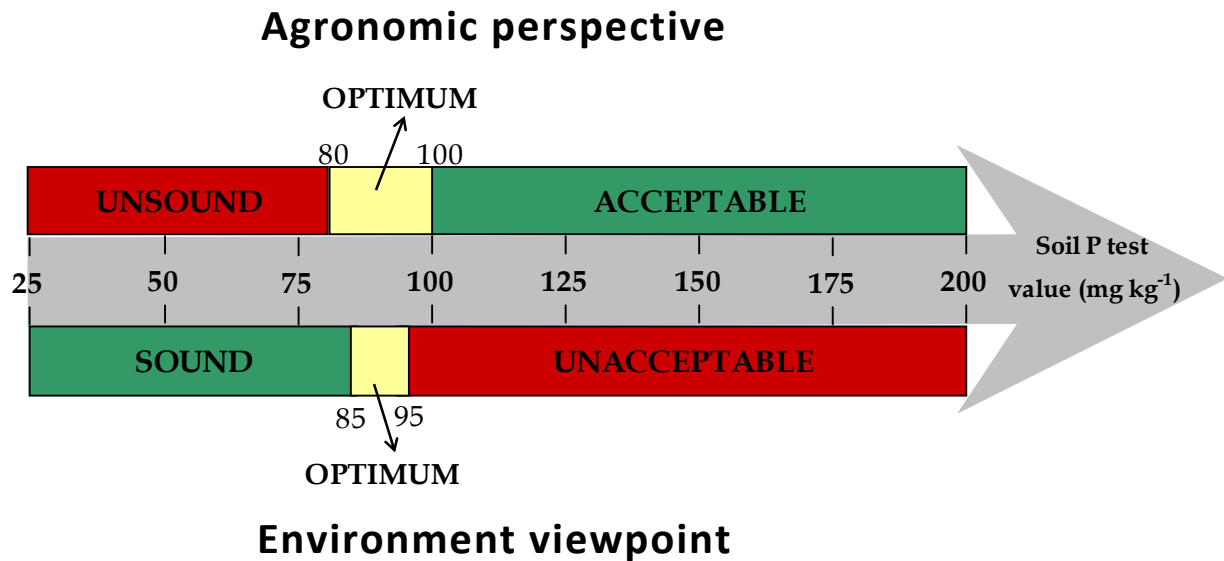


Figure 1.4: Interpretation of the 0.1M H₂SO₄ extractable P levels from the agronomic and environmental perspectives in Mauritius (Mardamootoo, 2009).

1.2 Problem statement

Soil test P on its own is insufficient to provide a complete assessment of the risk which soil P represents to freshwater resources because variables other than soil P levels contribute to P movement from fields and landscapes (Sharpley *et al.*, 1996). For P to pose an environmental problem, apart from the P source (e.g. soil P, fertilisers) the P must be transported (either by leaching, runoff or soil erosion) to water resources. Problems can only arise when the source and transport factors come together (Gburek *et al.*, 2000). Thus to determine if there is any need for new and improved management practices, not only source areas of P in the fields must be identified, but information on how current practices are inducing agricultural P transport to freshwater resources must be gathered.

Indeed advanced or accelerated eutrophication of surface water leads to problems with its use for fisheries, recreation, industry and drinking due to the increased growth of undesirable algae and aquatic weeds and oxygen shortages caused by their senescence and decomposition (Sharpley and Withers, 1994; Carpenter *et al.*, 1998). In addition, plant and animal communities may be directly affected by changes in water quality. Such changes may affect the biosphere by altering habitat, food, nutrient supplies and breeding areas. A common long-term effect will therefore be loss of both faunal and floral communities.

Usually to prevent eutrophication, total P should not exceed 0.05 mg L⁻¹ in streams entering lakes/reservoirs, or 0.025 mg L⁻¹ within the lakes/reservoirs as per directives of the United States

Environmental Protection Agency (Daniel *et al.*, 1998: U.S. Environmental Protection Agency, 2001). A four year study on agrochemical movement in sugarcane soils of Mauritius, which was undertaken jointly by the Mauritius Sugar Industry Research Institute and the Queensland Department of Natural Resources and Mines (Australia) with support from the Australian Centre for International Agricultural Research has shown that values higher than 0.05 mg P L⁻¹ in streams flowing past sugarcane fields particularly after high rainfall events were encountered (Ng Kee Kwong *et al.*, 2002).

From the above it is evident that while the productive agronomic potential of the soils under sugarcane in Mauritius must be safeguarded, the need also exists to protect the freshwater resources in the island from eutrophication. For an accurate knowledge of the possible risk of contamination of surface waters by P from sugarcane fields, susceptibility of the soil P to transport must therefore be assessed. Moreover inherent to risk assessment is the need to develop a simple decision support tool that can be adopted by farmers, extension officers, or agronomists to reliably predict the potential contribution of their farms and of their management practices to water eutrophication in their locale.

1.3 Hypotheses

- i. Eutrophication of waterbodies in sugarcane growing areas is associated with the transport of P from soils under that crop.
- ii. The risk of P movement in soils under sugarcane is highest when the soil water content is at field capacity and when the field has just been cultivated.
- iii. Most P transported from agricultural catchments to water bodies does not occur by leaching or by surface runoff but attached to soil particles coming during erosion from small but well defined areas of the landscape.
- iv. Knowledge of the risk of soil movement in Mauritius will provide an accurate indication of susceptibility of P to movement.

1.4 Objectives

- i. To evaluate factors influencing the vulnerability to movement of P from the main soils of Mauritius when they are at field capacity just after planting sugarcane.
- ii. To develop and validate a simple tool, the P index, which will combine the factors of source and vulnerability to transport, in order to identify areas within a watershed which are most prone to P mobilisation.

1.5 Expected outcome

The overall goal of the project is to maintain both a sustainable sugarcane industry and a sustainable clean environment. If sugarcane production is to stay profitable, it is evident that the sugarcane crop should receive all the nutrients that it requires, including P. This study is expected to provide a tool (i.e. the P index) to indicate to farmers where improved management practices must be targeted to reduce the incidence of accelerated eutrophication of freshwater resources. By and large the outcome of the project will be better management practices that are both agronomically, economically, and environmentally sound for producing sugarcane in Mauritius.

2 Literature review

2.1 Introduction

Phosphorus is an essential element for crop production, but as it is also one of the most difficult nutrients for plants to obtain from the soil, it is therefore often the limiting factor to optimum crop growth. This has led to the long-term application of P (either in the forms of chemical fertilisers or manure) to agricultural systems. Over the years on account of the very low mobility of the P applied, coupled with the fact that P is not subject to volatisation losses, an accumulation of P occurred in soils. From an agronomic point of view the accumulation of P in soils is desirable but from an environmental perspective, this build-up of P has been shown to represent a threat to the quality of fresh waters. The environmental significance of P lies in its dominant role in accelerating the eutrophication of aquatic ecosystems, particularly lakes (Foy, 2005). Phosphorus is in fact very often the most limiting nutrient influencing eutrophication of surface waters, generally at P concentration which is tenfold lower than that required for plant growth (Guidry *et al.*, 2006). In general, to maintain the quality of waters, total P should not exceed 50 $\mu\text{g L}^{-1}$ in streams entering lakes/reservoirs, or 25 $\mu\text{g L}^{-1}$ within lakes/reservoirs (U.S. Environmental Protection Agency, USEPA, 1996; 2001; 2002; Gibson *et al.*, 2000).

In view of the key role which P plays in determining the quality of freshwater resources, the significance of P in the environment has been extensively studied. Consequently, apart from the extensive literature which has always existed on the functions and needs of P in agricultural production, a vast literature is now being created in parallel on every aspect of P in the environment. Thus, it is now known that the transport of P from agricultural fields occurs primarily via surface flow when the water flowing across the soil surface either dissolves and transports soluble P or erodes and transports particulate P. Also, all forms of P whether they are soluble, adsorbed, precipitated or organic are susceptible to transport from soils to water bodies.

This chapter is an attempt to summarize the knowledge that has accumulated on P in the environment. Firstly, the dynamics of P in the soil, with an emphasis on the different factors and reactions affecting the availability and mobility of P in the soil is revised. A section then follows to highlight how agricultural soil P is transported to waterbodies and the related consequences, notably eutrophication. Phosphorus management strategies aimed at limiting surface water eutrophication caused by agricultural non-point P sources and at reducing the transport of P from agricultural land in runoff and erosion are afterwards summarised. For the review to be complete, in view of the importance of P in agriculture, it has included, right at the beginning, a section on the role and management of P in sustaining crop production.

2.2 Phosphorus in agricultural production

2.2.1 Importance of P in crop growth

Phosphorus is essential for plant growth by being involved in energy metabolisms, in cellular transfer mechanisms, in respiration, and in the photosynthesis of the crop. It is taken up by the plant as the orthophosphate ions (H_2PO_4^- or HPO_4^{2-}) and is incorporated into adenosine di- and tri-phosphate (ADP, ATP) required for the energy metabolism in the plant. As described by Ozanne (1980), through the combination of two photoreactions, light energy absorbed by the chlorophyll is used to reduce nicotinamide adenosine dinucleotide phosphate (NADP) to ATP. Indeed when the terminal phosphate from either ADP or ATP is split off from the molecules, a large amount of chemical energy is liberated for use in growth and reproductive systems. The high-energy phosphate compounds (ATP) in fact act as chemical intermediates which transfer energy rich H_2PO_4^- molecules from ATP to energy requiring substances (ADP) in the plant (Havlin *et al.*, 2005a). This energy transfer process is known as phosphorylation.

Phosphorus is also an essential element in deoxyribonucleic acid (DNA) and ribonucleic acid (RNA) that contain the genetic code of the plant and which play a role in producing proteins, other compounds essential for plant structure, seed yield, and in genetic transfer (Havlin *et al.*, 2005a). Phosphate also occurs in phospholipids including those of membranes, in sugar phosphates, and in various nucleotides and co-enzymes. Phytic acid, the hexaphosphate ester of myo-inositol, or its calcium or magnesium salts (phytin), serves as a storage form of phosphate in seeds (Sanchez, 2007).

One of the first symptoms of P deficiency of many plant species includes darkening of the leaves resulting in blue-green foliage. As described by Epstein (1972), often red, purple, or brown pigments develop in the leaves, especially along the veins. With increasing P deficiency, the dark green colour changes to a grayish-green to bluish-green metallic lustre. The visual P deficiency symptoms usually appear on lower leaf tips and progress along leaf margins until the entire leaf turns purple. The purple colour is due to accumulation of sugars that enhances synthesis of anthocyanin (a purple pigment) in the leaf (Ozanne, 1980).

In the absence of adequate amounts of P, plants fail to get off to a quick start, their root systems do not develop satisfactorily, and the plants become dwarfed or become stunted as illustrated in Figure 2.1 showing narrower and shorter leaves in the sugarcane plant (Korndörfer, 2005). Phosphorus deficiency may also reduce seed numbers, their viability and size (Ozanne, 1980). Other symptoms of P deficiency in small grain crops such as wheat include poor tillering, and delayed maturity (Prasad and Power, 1997).

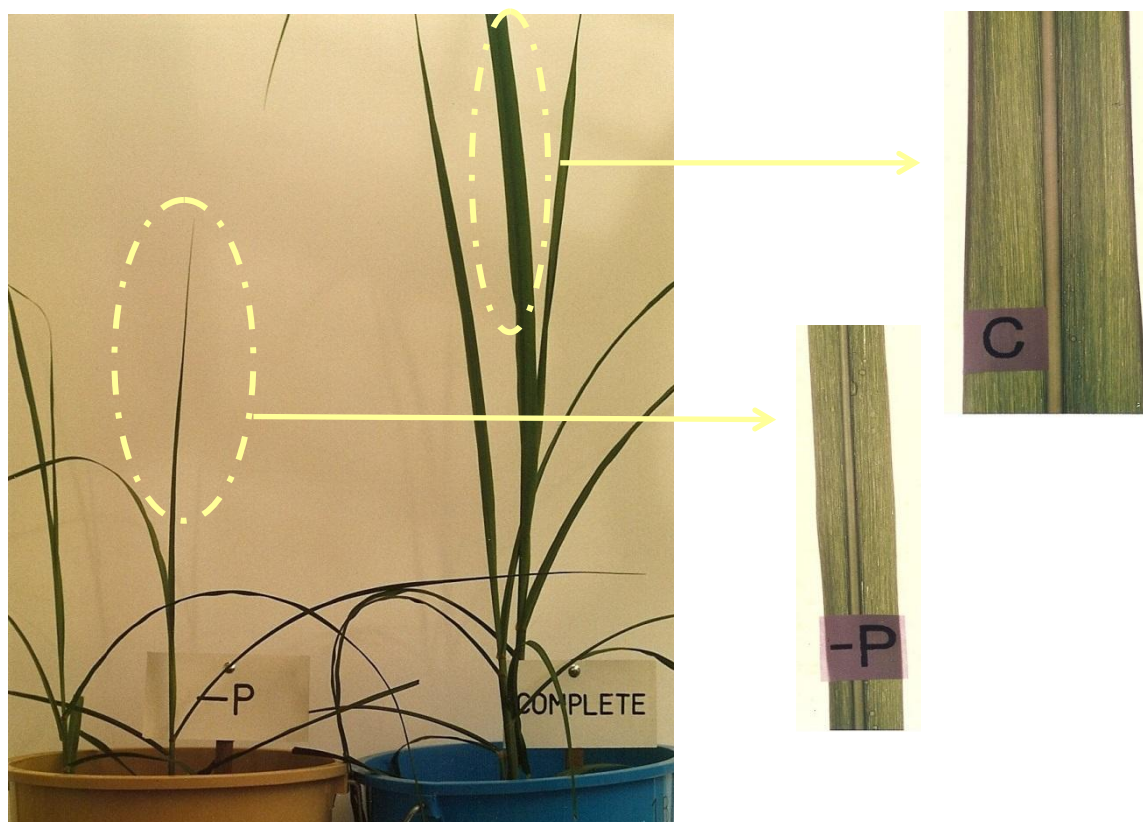


Figure 2.1: Stunted growth observed in potted sugarcane crops as a result of P deficiency.

Studies carried out by Hanway and Olson (1980) on the phosphate nutrition of maize, sorghum and soybeans showed that the total amount of P taken up was from 7 to 15 kg P ha⁻¹ with 2 to 8 kg P ha⁻¹ being returned to the soil in the crop residues left in the field. Research on P nutrition of cereals (e.g. maize, rice and wheat) showed that for every tonne of grain produced, the total crop contained about 4.2 kg P, the range of P in the grain is given as 2.7 to 3.3 kg grain and 0.83 to 1.6 kg P in the stover (Johnston, 2005). In addition, research done in Mauritius on the uptake of nutrients by sugarcane has shown that for each tonne of millable cane, the crop needed on average 0.29 kg P (Anon, 1994). Concerning vegetable crops such as celery, garlic, asparagus, cucumbers, and watermelons, the amount of P removed by the harvested portion of the plant is usually less than 10 kg P ha⁻¹ (Lorenz and Vittum, 1980).

2.2.2 Phosphorus dynamics in the soil-plant continuum

The dynamics of P in soils is best described by showing the soil P cycle such as the one proposed by Pierzynski *et al.* (2005) and reproduced in Figure 2.2.

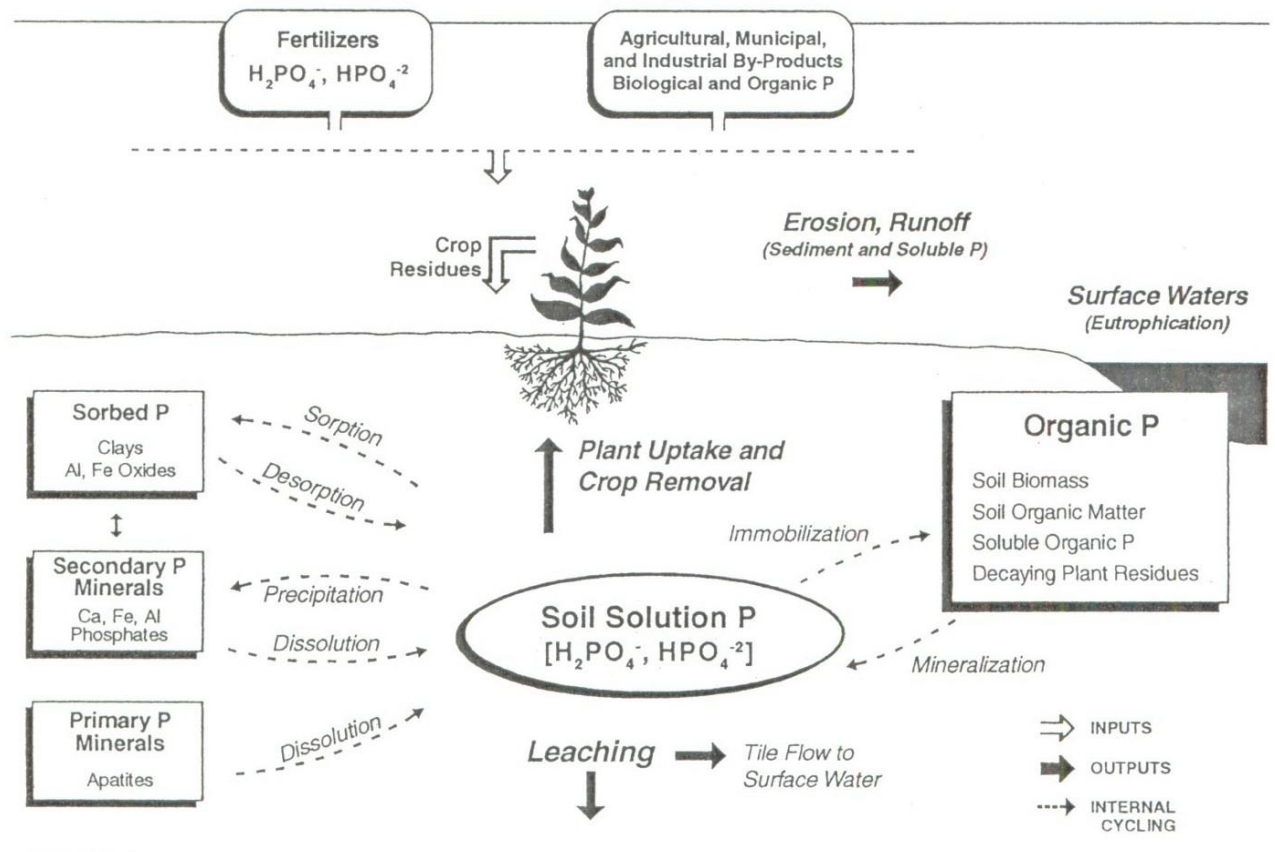


Figure 2.2: The soil P cycle as described by Pierzynski *et al.* (2005).

From the viewpoint of plant nutrition and availability to crops, the P in soil has most conveniently been categorised into three forms, namely *solution P*, *labile P* and *non-labile P* (Pierzynski *et al.*, 2005). The relationship among these three forms of P is often simplified to the following equilibrium equation (Havlin *et al.*, 2005a).



Phosphorus occurs in soil solution as orthophosphate ions, H_2PO_4^- and HPO_4^{2-} , which are in fact the only forms of P that can be taken up by crops. To maintain the concentration of P in soil solutions at an optimum value for plant growth (about 0.2 mg L^{-1}), all the chemical and biochemical processes of the soil P cycle must come into play to release P rapidly enough for crop uptake through dissolution-precipitation, sorption-desorption, mineralisation-immobilisation, and oxidation-reduction reactions (Pierzynski *et al.*, 2005).

While the inorganic P in soils equilibrates with the soil solution P through adsorption-desorption reactions and through dissolution-precipitation, the organic P component influences the P

concentration in the soil solution through mineralisation and immobilisation (Pierzynski *et al.*, 2000). Both P mineralisation and immobilisation rates are affected by factors such as temperature, moisture, aeration, pH, cultivation intensity and P fertilisation (Havlin *et al.*, 2005a). The extent of P mineralisation over immobilisation depends on the C:P ratio of the residues deposited in the soil (Stevenson, 1964). Mineralisation occurs rapidly if the C:P ratio of the organic matter is less than 200:1, while immobilisation will be predominant if the C:P ratio exceeds 300:1 (Pierzynski *et al.*, 2000).

2.2.2.1 Transformations of P fertiliser in soil

When soluble phosphatic fertilisers are applied to soils, they initially dissolve causing an immediate rise in the concentration of soil solution P, which then participates primarily in adsorption and precipitation processes (Prasad and Power, 1997). The reactions that occur among the phosphate ions present in the soil solution, the soil constituents, and the non-phosphatic components in the fertilisers, primarily remove the P from the solution phase and render the phosphate less soluble over time (Sample *et al.*, 1980). This phenomenon is commonly referred to as *P fixation*, *adsorption* or *retention*. As a consequence of fixation, P becomes highly immobile in soils and generally stays near the point of application (Prasad and Power, 1997). In fact, at the beginning the sorption processes are easily reversible and the added P remains readily available for plant uptake, thereby imparting a high residual value to the phosphate fertilisers (Havlin *et al.*, 2005a).

The solid labile phases which are formed initially however gradually revert to less soluble P forms (non-labile) and adsorption continues to decrease soil solution P concentration with time and to cause a reduction in plant available P (Pierzynski *et al.*, 2005). Fixation of P by soils thus plays an important role in determining the ultimate availability of fertiliser P to crops and its mobility in soils. On account of its significant role in affecting the availability and mobility of P, an understanding of the different reactions underlying P fixation in soils is a first step towards obtaining optimum P nutrition and towards achieving efficient management of the fertiliser P to protect freshwater sources.

2.2.2.2 Fixation of P by hydrous oxides of Fe and Al

The most active soil constituents involved in the retention of P in the soils are the hydrous oxides of iron and aluminium. These oxides occur either as discrete compounds in soils or as coatings on soil particles or as amorphous Al hydroxyl-compounds between the layers of expanding Al silicates. Studies carried out (e.g. Sample *et al.*, 1980) have shown that these hydrous oxides of Fe and Al retained large amounts of P from soil solution, the amount of P sorbed by hydrous oxides of iron and aluminium being dependent upon the time of reaction, the temperature, pH and the P concentration in the soil solution. Bache (1964) studied P sorption by gibbsite and hydrous ferric oxide and showed

that the mechanism of P retention in soils by the Al and Fe oxides followed three distinct stages which occur at different P concentrations in the solution: (i) a high energy chemisorption, (ii) precipitation of a separate phosphate phase, and (iii) a low energy sorption of P onto the precipitate.

In acid soils, the predominance of positive charges on Al and Fe oxides/ hydroxides facilitates the attraction of negatively charged orthophosphate H_2PO_4^- and HPO_4^{2-} ions (Havlin *et al.*, 2005a). The mechanism of P adsorption on Al/Fe oxide surface involves the exchange of phosphate for OH groups as shown in Figure 2.3.

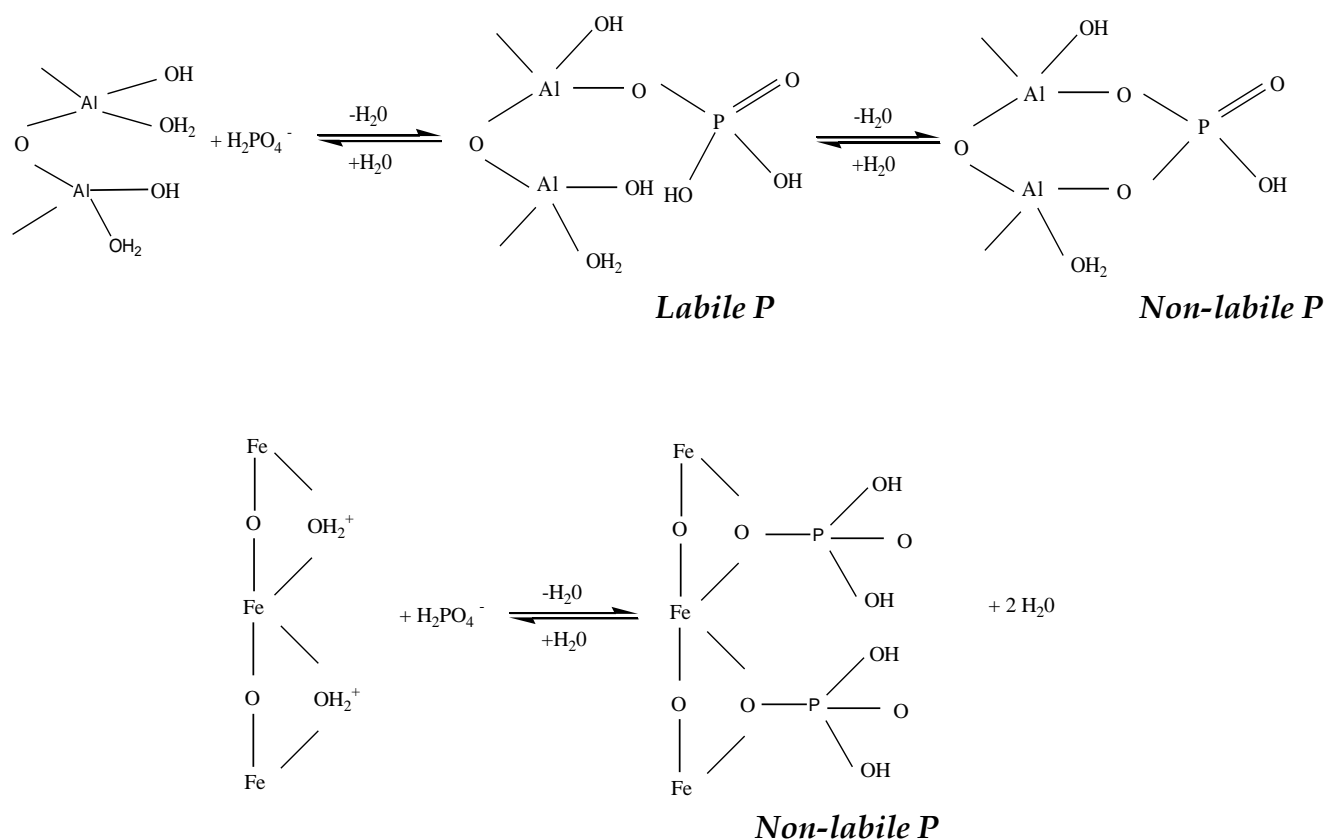


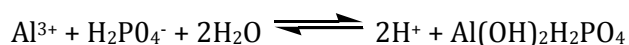
Figure 2.3: Mechanism of P adsorption to Fe/Al oxide surface (Havlin *et al.*, 2005a).

When the orthophosphate ion is bonded through one Al-O-P bond, the H_2PO_4^- is considered as labile as it can readily be desorbed from the mineral surface to soil solution. But when the H_2PO_4^- is bonded to the Fe/Al hydroxides through two Al-O bonds, a stable six-membered ring is formed and the H_2PO_4^- is regarded as non-labile and unavailable for plant uptake.

As reviewed by Sample *et al.* (1980), at low solution P concentrations hydrous oxides retain P through sorption-type reactions but at higher P concentrations, that is when the concentration of P and

associated cations in the soil solution exceeds that of the solubility product (K_{sp}) of the mineral, precipitation reactions are favoured. In neutral and calcareous soils, Ca being the dominant cation, the addition of soluble P initially results in the precipitation of di-calcium phosphate dihydrate $\{CaHPO_4 \cdot 2H_2O\}$ which, with time slowly reverts to other more stable but less soluble Ca phosphates (Pierzynski *et al.*, 2000). The precipitates in Ca systems as described by Sharpley (2000a), usually occur in the following sequence: mono-calcium phosphate $\{Ca(H_2PO_4)_2\}$, di-calcium phosphate dihydrate $\{CaHPO_4 \cdot 2H_2O\}$, octa-calcium phosphate $\{Ca_8H_2(PO_4)_6 \cdot 5H_2O\}$ and finally hydroxy-apatite $\{Ca_{10}(PO_4)_6(OH)_2\}$ or fluoro-apatite $\{Ca_{10}(PO_4)_6F_2\}$.

In acid soils, the chemical equation given below summarises the precipitation reactions involving soluble Fe or Al with $H_2PO_4^-$ to form Al or Fe hydroxyl-phosphates (Brady, 1974).



As reviewed by Sharpley (2000a), P in the soil solution generally reacts with Al oxides to form amorphous Al-P organized phases such as sterretite $\{Al(OH)_3 \cdot HPO_4 \cdot H_2PO_4\}$; and with Fe oxides to precipitates such as tenticite $\{Fe_6(PO_4)_4(OH)_6 \cdot 7H_2O\}$ or griphite $\{Fe_3Mn_2(PO_4)_2 \cdot 5H_2O\}$.

2.2.2.3 Fixation of P by soil alumino-silicate minerals and carbonates

Alumino-silicate minerals, such as kaolinite, montmorillonite and illite also play a significant role in P fixation (Brady and Weil, 1996). Phosphorus is adsorbed to a larger extent by 1:1 clays (e.g. kaolinite) than by 2:1 clays (e.g. montmorillonite). This can be explained by the presence of higher amounts of Fe/Al oxides associated with kaolinitic clays. Moreover in the kaolinitic clays, a larger number of OH groups are exposed in the Al layer to exchange with P (Havlin *et al.*, 2005a). In addition, the presence of pH-dependent charges on kaolinitic clays also contributes to P adsorption. The mechanisms of P adsorption by alumino-silicate minerals are in fact the same as described above for the oxides of Al and Fe. Thus at low P concentrations, the P is adsorbed onto the silicate clays with the replacement of surface hydroxyl groups as illustrated in Figure 2.3. High P concentrations, such as soon after application of soluble mineral P fertilisers, favour the release of Si and Al with the subsequent precipitation of Al-P compounds (Sample *et al.*, 1980).

In calcareous soils, P adsorption may also occur on the carbonates ($CaCO_3$) present. As reviewed by Prasad and Power (1997), the interaction of P with the $CaCO_3$ involves two reactions: the first reaction occurs at low P concentration and consists of adsorption of P on $CaCO_3$ surfaces, while the second reaction is a nucleation process to form phosphate crystals.

The different above-mentioned reactions of added P in soils, described in this section and in the preceding one, explain the high residual values of P fertilisers that are often reported in the literature

(Havlin *et al.*, 2005a). As reviewed by Morel and Fardeau (1989), 80-99% of P applied as fertilisers remains in the soil. In fact as summarised by Barrow (1980), the literature available on the residual value of P fertilisers has two contrasting strands. First, a reported decline in effectiveness of the P fertilisers during the first few months (or years) after their application implying that repeated applications of P is required and second, mention is frequently made about the continuing uptake of P by the crops over several years after its application. Thus as observed by Cavalot *et al.* (1988), a single application of P fertilisers at planting was sufficient to meet the needs of a sugarcane crop cycle for six to seven years. The data in the literature in effect support the conclusion of Havlin *et al.* (2005b) that the residual availability potential for immobile nutrients such as P could only be assessed through soil testing.

2.2.3 Factors and reactions affecting P availability and mobility

It follows from the preceding sections that in general, P retention or fixation in soils is a continuous process involving precipitation, chemisorption and adsorption (Prasad and Power, 1997). At low solution P concentrations, P retention follows mainly an adsorption mechanism while at high P concentrations in solution, precipitation predominantly occurs following solubility product principles. As the availability and mobility of P in soils are highly influenced by P retention, the soil properties influencing P retention and solubility need to be known and are discussed in this section.

2.2.3.1 Soil P buffering capacity

The soil P buffering capacity is an important soil property providing a suitable indication of available P in the soil (Holford, 1997). McDowell *et al.* (2001) added that since soil P buffering capacity is a function of sorption capacity and sorption strength, it controls the rate of desorption and diffusion of P from soil to solution. The higher the soil P buffering capacity, the slower but the longer P will be replenished in the soil solution following its absorption by plant roots. As explained by Holford (1997), this replenishment capacity depends on the quantity of P in the labile pool and the ease with which this P is released into solution.

2.2.3.2 Soil mineralogy and clay content

Adsorption and desorption reactions are affected by the type of mineral surfaces in contact with P in the soil solution (Havlin *et al.*, 2005a). As explained in section 2.2.2.1 and 2.2.2.2, P is adsorbed most extensively by Al and Fe oxides and to a greater extent by 1:1 clays (such as kaolinite) as compared to 2:1 clays (e.g. montmorillonite) due to the presence of higher Fe/Al oxides content in the 1:1 clay minerals (Havlin *et al.*, 2005a). Apart from the nature of the minerals, the clay content of soils also affects the degree of P fixation. Among soils of similar clay mineralogy, P fixation will obviously increase with rising clay content (Kamprath and Watson, 1980). Thus soils with a sandy texture have

low P adsorption capacities with the P more susceptible to leaching than soils of a clayey texture (Pierzynski *et al.*, 2000).

In calcareous soils, the presence of CaCO_3 with large surface area also shows a high adsorption and a rapid precipitation of Ca-P minerals (Havlin *et al.*, 2005a). Calcareous soils with highly reactive CaCO_3 and a high Ca-saturated clay content have in this context been shown to exhibit low solution P levels, since the P in the soil solution is instantaneously precipitated or adsorbed (Havlin *et al.*, 2005a).

In addition, the type of cations on the cation exchange sites of the clays has an influence on P adsorption (Havlin *et al.*, 2005a). Ca-saturated clays have been shown in this context to exhibit greater P adsorption than their Na-saturated counterparts. As reviewed by Kurtz (1953), even at pH levels below neutrality, where calcium precipitation would not be expected, calcium clays retain more phosphate than sodium, ammonium or potassium clays. This observation was explained by a possible precipitation of calcium phosphate at the colloid surface or a binding of phosphate to the soil colloid through Ca^{2+} on the exchange complex (Kurtz, 1953).

2.2.3.3 Soil pH

Phosphorus fixation in acidic soils is more pronounced than in calcareous/alkaline soils. The P adsorbed is moreover held more strongly. In fact, in most soils, maximum P retention occurs at the low pH values of 3.0 to 4.0 because of adsorption by Fe/Al oxides. As the pH increases, P adsorption decreases resulting in a higher concentration of P in soil solution (Havlin *et al.*, 2005a). In general, P availability to plants in most soils will be at its maximum when the soil pH is maintained in the range of 6.0 to 7.0 (Brady and Weil, 1996). Above pH values of 7, the presence of CaCO_3 accounts for P fixation, resulting in a decline of soil solution P.

2.2.3.4 Soil organic matter

Soil organic matter in association with cations such as Fe, Al and Ca is capable of retaining significant amounts of P (Prasad and Power, 1997). Humic acid dissolves Al from soil minerals to form complexes which give rise to new surfaces for P adsorption by ligand exchange of the phosphate ions for the hydroxyl groups (Sample *et al.*, 1980). Hence the overall effect of an increase in organic matter content of the soil would be an increase in P adsorption. On the other hand, as also described by Sample *et al.* (1980) in calcareous soils, organic matter and P compete for the same adsorption sites on CaCO_3 , thereby decreasing the ability of the calcareous soils to adsorb P.

The presence of organic compounds in soils has also been reported to increase P availability by maintaining the P in solution through the formation of stable complexes with Fe and Al (Prasad and

Power, 1997). The organic anions known to be most effective in competing and replacing H_2PO_4^- are citrate, oxalate, tartrate and malate (Havlin *et al.*, 2005a). In soils with very high organic matter, P mobility is further enhanced by the organic matter forming a coating on the colloidal surfaces responsible for P adsorption (Pierzynski *et al.*, 2000). This explains why organic compounds tend to move P to a greater depth than would inorganic P alone in soil solution. In this context, the continuous application of manure has been found to result in elevated P levels at 0.6 to 1.2 m soil depths while the application of the same amount of P as inorganic fertilisers resulted in much less downward movement of P (Havlin *et al.*, 2005a).

2.3 Phosphorus management for crop production

2.3.1 Management of P fertilisers

The efficiency with which P fertilisers are used by crops depends not only on the extent of P deficiency in soils and on crop P requirements but also on factors such as the time of application, placement, rate and frequency of the fertiliser P applications (Havlin *et al.*, 2005c). All of these factors, by influencing P fixation reactions in the soil, eventually determine P availability and uptake by crops.

The *timing of P fertilisation* from an agronomic perspective is optimised if adequate amounts of P are available at all times to meet plant requirements (Bundy *et al.*, 2005). Phosphorus is needed as from the earliest stages of crop growth since it is important in nearly all energy-requiring processes in the plant. As indicated by Bundy *et al.* (2005) the use of starter P fertilisers is known to promote early plant growth and development. As P stress early in the growing season reduces crop productivity more than P restrictions later during the crop season, P fertilisation is usually best carried out just before or at planting.

The *placement of starter P fertiliser* also plays an important role in its effectiveness to crops (Bundy *et al.*, 2005). Phosphorus is relatively immobile in the soil and so remains near the site of fertiliser placement (Grant *et al.*, 2001). Surface application after the crop has been planted will not place the P near the root zone and will thus be of little value to annual crops in the year of application (Havlin *et al.*, 2005c). For optimum P management, the question of band placement over broadcast application is an important consideration. As pointed out by Havlin *et al.* (2005c), band placement of P reduces fertiliser-soil contact, resulting in less fixation than broadcast P. This implies that P is maintained in a plant-available form for a longer period of time.

In fact to ensure maximum P efficiency, a compromise has to be found between reducing the volume of soil fertilised so as to minimise fixation and providing a large enough fertilised soil volume to encourage root-fertiliser (Grant *et al.*, 2001). Being immobile and moving mostly by diffusion which occurs over short distances in the soil, the P must be placed in a position where the plant roots can take it up early in the season. Therefore, soluble fertiliser P is most efficient when seed-placed or placed in a band close to the seed, especially for crops having poorly developed root systems (Grant *et al.*, 2001). However, seedling damage is often observed when P is placed in the seed-row. Such damages are more prominent when mono-ammonium or di-ammonium phosphates are used and to avoid seed damage, banding the fertiliser below the seed-row is the best practice (Grant *et al.*, 2001).

The question of *frequency of fertiliser P application* in crop rotations or in permanent pastures is pertinent throughout the range of P-deficiency levels encountered in soils. Frequent P applications are likely to be more important in soils of high P fixation capacities than in soils with low P fixing properties (Stanford and Pierre, 1953). In general the repeated application of large amounts of P fertiliser to crops eventually result in a point where a single application at the time of planting will suffice to give optimum crop yields. Recent studies on the response to P fertilisation by ratoon cane have even shown that on some sugarcane farms the application of P to ratoon cane can be reduced during several years without a decline in productivity (Korndörfer, 2005).

The utilisation of rock phosphate to supply plant-available P in strongly weathered and P deficient acidic soils has been found to be agronomically responsive (Bolland *et al.*, 2001). The efficacy of the rock phosphate depends upon its dissolution which is influenced by many factors, including soil pH (Yusdar *et al.*, 2007) and fineness of grinding of the phosphate rock. From an agronomic point of view, while water-soluble fertilisers should be applied in bands as explained above, rock phosphates on the contrary will be most effective when broadcast throughout the field (Gilkes and Bolland, 1990). This is so because the dissolution of the rock phosphate will be enhanced when its degree of contact with the soil H⁺ ions is increased (Gilkes and Bolland, 1990).

2.3.2 Assessment of P needs of crops

The quantity of soil P available to crops depends on many interacting factors such as the environment (water, temperature) and the soil management practices (Havlin *et al.*, 2005b). The quantity of P necessary to optimise crop yield further depends on both the plant P requirements and on the P-supplying capacity of the soil. Diagnostic techniques commonly employed to assess the P status of a soil include the identification of plant P-deficiency symptoms, plant testing and soil analysis (Sumner, 2006). The following sections outline how plant testing and soil analysis techniques are utilized to evaluate the nutrient status of the soils. Though visual deficiency symptoms provide an indication of P

deficiency, they are seldom conclusive. Consequently accurate diagnosis typically requires a tissue test (Sanchez, 2007). In conjunction with soil tests, plant testing may also aid in determining the P-supplying capacity of the soil.

2.3.2.1 Plant testing

Plant testing in a narrow sense can be described as the determination of the concentration of an element such as P or of an extractable fraction of the element in a particular part of a crop when the latter is sampled at a certain time or stage of morphological development (Walsh and Beaton, 1973). Plant testing involves either field tests which are performed on fresh tissue or laboratory-based analysis of the plant tissue. It is based on the premise that the amount of a given nutrient in a plant is related to the availability of that nutrient in the soil (Havlin *et al.*, 2005b). In essence plant testing helps to identify deficiency symptoms or to determine nutrient shortages before they appear.

Diagnostic standards with respect to plant testing involve the utilisation of critical levels or sufficiency ranges (Sanchez, 2007). These levels or ranges are usually determined by developing a response curve relationship as shown in Figure 2.4. As reviewed by Westermann (2005), the critical nutrient range is the range of concentrations above which the crop is amply supplied and below which the crop is deficient in the nutrient investigated. Usually the diagnosis of a nutrient deficiency, including P deficiency, by tissue analysis is a post-mortem of the current fertilisation practices (Sanchez, 2007) and is used to correct impending shortages of the nutrient in future crops.

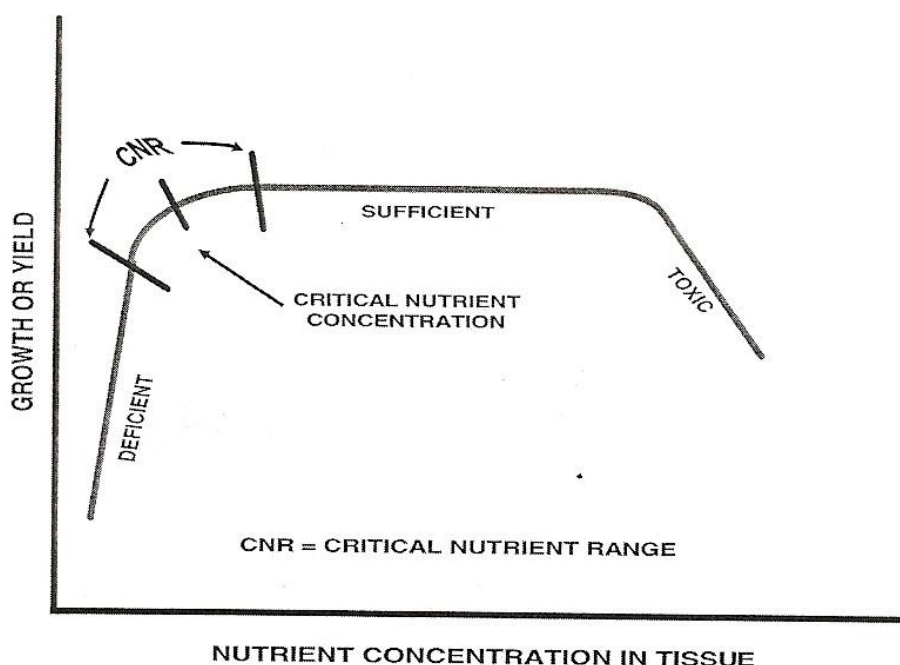


Figure 2.4: Growth or yield of plants in relation to nutrient concentration in plant tissue (Westermann, 2005).

2.3.2.2 Soil testing

Soil testing is an essential and integrated part of crop management in present-day agricultural systems (Fageria *et al.*, 1997). Crop response is poorly related to the total amount of P in a soil and therefore a successful soil test should represent some index of P availability (Sanchez, 2007). Agronomic soil tests to indicate available P have been designed such that (i) they are simple for routine application, (ii) they extract sufficient P to be easily measurable, (iii) they extract sufficient P to represent a significant portion of the soil P potentially available for plant uptake, and (iv) they do not extract significant amounts of P that are not available to plants (Tiessen and Moir, 1993).

To assess the soil P available to crops, several extracting agents are being used, with the most commonly utilised ones listed in Table 2.1. These extractants cover a broad range of soil conditions, from acid to alkaline, from low to high cation exchange capacity (CEC), and from arid to humid soil conditions (Fageria *et al.*, 1997). Many of the soil test extractants employ acids to dissolve the Ca, Al, and Fe phosphates, which have been shown to be the main inorganic sources of labile P (Beegle, 2005). Extractants such as the Bray-1 and -2, and Mehlich-1 and -3, are dilute solutions of the strong acids, namely HCl, HNO₃, and H₂SO₄. As discussed by Holford (1997), a soil test should extract a quantity of P that is positively related to extractable P and negatively related to the P buffering capacity of the soil but in practice the selection of the extractant is based on the degree of correlation between the soil P extracted and a measure of crop growth (Fageria *et al.*, 1997). The preferred extractant is normally the one giving the best correlation between the soil P extracted and the measure of crop growth used.

The test based on Bray-1 extractant, which is a mild-acid solution, has been found to be reliable for predicting crop response to P fertilisation on neutral to acidic soils but it has been less effective on alkaline soils, where the acid from the extractant is neutralised quickly by the bases present while the fluoride ions are precipitated by Ca (Sanchez, 2007). The Bray-2 extractant has the same concentration of NH₄F (0.03M) as Bray-1, but the HCl concentration has been raised to 0.1M to give it an increased capacity to extract the less soluble Ca-P (Fageria *et al.*, 1997).

The Mehlich-1 extractant has the advantage of simultaneously extracting P, K, Ca, Mg, Cu, Mn, Fe, and Zn (Sanchez, 2007). The Mehlich-2 extractant was developed to allow simultaneous determination of the same nutrients over a still wider range of soil properties (Sanchez, 2007). However, the corrosive nature of the Mehlich-2 extractant discouraged its use and its composition was ultimately slightly modified to become Mehlich-3 which has been found to be reliable across a wide range of soil and crop production circumstances (Sanchez, 2007).

Table 2.1: Common soil tests for estimating available soil phosphorus
(adapted from Sanchez, 2007).

Extractant	Extracting reagents
AB-DTPA	1M NH_4HCO_3 + 0.005M DTPA, pH 5
Bray-1	0.025N HCl + 0.03N NH_4F
Bray-2	0.1N HCl + 0.03N NH_4F
Citric acid	1% citric acid
Colwell	0.5M NaHCO_3 , pH 8.5
Egner	0.02N Ca-lactate + 0.02N HCl
Mehlich-1	0.05N HCl + 0.025N H_2SO_4
Mehlich-2	0.2N CH_3COOH + 0.25N NH_4Cl + 0.015N NH_4F + 0.012N HCl
Mehlich -3	0.2N CH_3COOH + 0.25N NH_4Cl + 0.015N NH_4F + 0.013N HNO_3 + 0.001M EDTA
Modified Morgan	0.62N NH_4OH + 1.25 N CH_3COOH , pH 4.8
Morgan	0.54N HOAc + 0.7 NaOAc, pH 4.8
Olsen	0.5M NaHCO_3 , pH 8.5
Truog	0.002N H_2SO_4 buffered at pH 3 with $(\text{NH}_4)_2\text{SO}_4$

Whichever extractant or method is selected for the available soil P analyses, the P soil-test levels need to be converted into P fertiliser recommendations. A useful starting point for that conversion is the determination of critical soil test P levels, which refer to the soil P value above which there will be no response to P fertiliser (Sanchez, 2007). In fact, to convert soil test P values into fertiliser P recommendations, two sets of calibration information for each combination of crop-soil type-climate are required, namely: (i) the soil P test level that produces the maximum yield, and (ii) the quantity of fertiliser P that is required to reach that test level (Thomas and Peaslee, 1973). Thus, in situations where the soil test P levels are below the critical P value, fertilisation is required and the rate of P fertiliser will depend on the soil test P level.

2.4 Phosphorus in the environment

2.4.1 Eutrophication and related problems

The environmental significance of P lies in the dominant role it plays in accelerating eutrophication of aquatic ecosystems (particularly lakes), where P is commonly regarded as the limiting nutrient governing primary production (Foy, 2005). *Eutrophication* is the natural aging of lakes or streams brought on by a nutrient enrichment that increases the biological productivity of the water body. As nutrient inputs to surface waters gradually increase (Table 2.2), the trophic state of the water evolves

through four stages of eutrophication, namely *oligotrophic*, *mesotrophic*, *eutrophic* and *hypereutrophic*; with changes occurring at each stage in the ecology of the water (Pierzynski *et al.*, 2000).

Table 2.2: Average nitrogen (N) and phosphorus (P) characteristics of lakes, streams and coastal marine waters at different trophic states (adapted from Smith *et al.*, 1999).

	Trophic state	Total P (mg m⁻³)	Total N (mg m⁻³)
Lakes	Oligotrophic	<10	< 350
	Mesotrophic	10-30	350 - 650
	Eutrophic	30-100	650-1200
	Hypereutrophic	>100	>1200
Streams	Oligotrophic	<25	<700
	Mesotrophic	25-75	700-1500
	Eutrophic	>75	>1500
Marine	Oligotrophic	<10	<260
	Mesotrophic	10-30	260-350
	Eutrophic	30-40	350-400
	Hypereutrophic	>40	>400

Over time (thousands of years), an initially clear (oligotrophic) lake gradually eutrophies, with filling of sediment to eventually becoming a marsh, and then a dry land (Spiro and Stigliani, 2002). Low biological productivity is associated with low nutrient concentration in the oligotrophic state and once eutrophic conditions are attained (high nutrient concentration), changes (most often undesirable) occur in the aquatic ecosystem.

Although eutrophication is a natural process, it can be accelerated by changes occurring in the use of land in a watershed, particularly those changes which cause the amount of nutrients added to an aquatic system to be increased abruptly (Sharpley and Beegle, 2001). Anthropogenic inputs of P may occur from sewage, agricultural runoff (especially where fertilisers and manures are applied) and from wastewaters (presence of polyphosphates in detergents). The major nutrients required for algal growth are carbon (C), nitrogen (N) and P in the atomic ratios of 106:16:1 to reflect the average composition of the molecules in biological tissues (Spiro and Stigliani, 2002). Atmospheric inputs generally satisfy the demands of algal growth for C and N, the latter through the capacity of certain cyanobacteria to fix nitrogen (N₂), thus bypassing the limiting concentrations of dissolved N in the waters (Foy, 2005).

For P, on the other hand, there is no atmospheric source since there is no naturally occurring gaseous P compound. Consequently, unlike P, N is not usually limiting to eutrophication although it may be in some regions. However in moving from freshwaters to saline oceans, through transition zones of brakish waters and estuaries, there is a shift from P to N limitation, whereby N becomes the nutrient controlling aquatic productivity (Correll, 1998). Moreover, the relative importance of N and P can be assessed by comparing the ratio of the two elements in surface water by using as benchmark an atomic N to P ratio of 16:1 (termed the Redfield ratio). Deviation of the ratio above 16 will indicate P to be in short supply for growth while lower ratios will suggest that N is potentially limiting (Foy, 2005).

On specific occasions, in some ecosystems moderate nutrient enrichment can be beneficial because increased primary production can lead to increased fish populations and harvest (Committee on the Causes and Management of Eutrophication, Ocean Studies Board, Water Science and Technology Board, National Research Council, 2000). However when nutrient enrichment is sufficiently pronounced, the effects are invariably detrimental from an ecological, economic and animal/human health perspectives. As highlighted by Sharpley *et al.* (1994), the principal impacts of eutrophication relate to increased aquatic plant growth, oxygen depletion, pH variability, and plant species quality and food chain effects. The ecological effects of accelerated eutrophication can have societal impacts as well, and these include loss in aesthetic values of the beaches, seagrass beds and coral reefs causing diminished recreational value (Committee on the Causes and Management of Eutrophication, Ocean Studies Board, Water Science and Technology Board, National Research Council, 2000). The cyanobacteria or blue-green algae commonly associated with eutrophic waters may produce toxins with potential adverse impacts on human (as well as on animals) health resulting in livestock deaths and human sickness.

Once P enters a waterbody and combines with the nutrients already in the aquatic ecosystem, its removal is tedious and expensive (USEPA, 1986). Complicating the problem further is the fact that eutrophication sometimes occurs a long distance away from where the P enters the waters and by the time the water quality effects become noticeable, remedial strategies are difficult to implement (Sharpley and Beegle, 2001). It is generally less expensive to cure the cause of eutrophication than to treat its effects. The practices proposed for correcting problems associated with excess P in water include removal of sediment from water bodies, stimulation of aerobic conditions, enhanced vegetative growth in littoral zones to decrease water-column mixing, introducing vegetative mining of sedimentary P and harvest of aquatic vegetation (Sharpley *et al.*, 2006). However, the economics of remediation remain a key limitation to substantial changes in agricultural management for environmental protection (Sharpley and Wang, 2014).

Best management practices are encouraged to limit the generation or delivery of pollutants from agricultural activities to water resources and to prevent impacts to the physical and biological integrity of surface and ground waters. Moreover, to reduce the risks and impacts of eutrophication, a range of international and national strategies and legal measures has been introduced to control point and diffuse nutrient inputs to the water environment.

2.4.2 Acceptable P concentration in waters

Several conditions need to be taken into consideration when setting water quality criteria. The designated use of the water, for instance, will dictate the desired or tolerable nutrient loadings, e.g. lakes used primarily for water consumption, swimming and multipurpose recreation should benefit from low P loadings while lakes mainly used for fish production would tolerate higher P inputs (Daniel *et al.*, 1998). Moreover as other factors such as flow rate, light, turbidity, temperature, and N levels also limit plant and algae growth, it is difficult to recommend absolute total P concentrations to prevent plant and algal blooms in aquatic environments (Australian Department of Sustainability, Environment, Water, Population and Communities, 2012).

Total P concentrations in non-polluted natural waters cover a very wide range, from less than 1 $\mu\text{g L}^{-1}$ to more than 200 mg L^{-1} in some closed saline lakes but in most uncontaminated surface waters they vary from 10 to 50 $\mu\text{g L}^{-1}$ (Wetzel, 2001). As highlighted by the European Environment Agency (EEA, 2004), natural concentrations of orthophosphates vary from one catchment to another, depending upon such factors as geology and soil but they usually vary between 0 to 10 $\mu\text{g P L}^{-1}$ only. At times however, background concentrations of P in runoff from undisturbed areas may already exceed the quality thresholds and in yet other instances accelerated eutrophication has been observed to occur in waters where the P concentration is less than 100 $\mu\text{g P L}^{-1}$ (USEPA, 1986). Most relatively pristine lakes in the USA are known to have surface waters that contain between 20 to 30 $\mu\text{g L}^{-1}$ total P (USEPA, 1986). In general, surface waters that are maintained at 10 to 30 $\mu\text{g L}^{-1}$ of total P tend to have low algal productivity and are termed *oligotrophic* (USEPA, 1986).

Although for a designated use, the USEPA usually proposes only one critical limit for a specific pollutant, it has set no such national criterion for P in waters to control eutrophication. Instead the USEPA has divided the USA into ecoregions and has established criteria for lakes/reservoirs (varying between 10 to 130 $\mu\text{g L}^{-1}$ total P) and rivers/streams (varying between 8 to 40 $\mu\text{g L}^{-1}$ total P) in each ecoregion (USEPA, 2000a and b). In general, however the USEPA (2000a and 2000b) states that total P should not exceed 50 $\mu\text{g L}^{-1}$ in streams entering lakes/reservoirs, and 25 $\mu\text{g L}^{-1}$ within lakes/reservoirs. Moreover for the prevention of plant nuisances in streams or other flowing waters

not discharging directly to lakes/impoundments the concentration of total P should not exceed 10 $\mu\text{g L}^{-1}$ (Daniel *et al.*, 1998).

The Australian Water Quality guidelines (Australian and New Zealand Environment and Conservation Council, ANZECC, 1992), which are comparable to the standards set by the USEPA (1988) recommend total P concentrations of 10 to 100 $\mu\text{g L}^{-1}$ for rivers and streams, 5 to 50 $\mu\text{g P L}^{-1}$ for lakes and reservoirs and 1 to 10 $\mu\text{g P L}^{-1}$ (as orthophosphate P) for coastal waters. In Mauritius, the guidelines for inland surface water quality are similar to those decided by the USEPA (1988). Even more specific guidelines based on their utilisation purposes have been proposed for coastal water quality, for example for conservation of the coral community in Mauritius, the total P should not exceed 40 $\mu\text{g L}^{-1}$ in coastal waters (Republic of Mauritius, 1999).

2.4.3 Phosphorus movement in the landscape

All forms of P in the soil whether they are soluble, adsorbed, precipitated, or organic are susceptible to transport to water bodies (Pierzynski *et al.*, 2000). Transport of soil P occurs primarily via surface flow when the water flowing across the soil surface either dissolves and transports soluble P or erodes and transports particulate P (Sharpley and Withers, 1994). The soluble P can be in both inorganic and organic forms but its transport in most soils is usually low due to the low solubility of soil P, the high adsorption capacities of the clays for P, and the general strong bonding of the P with the soil organic matter (Hansen *et al.*, 2002). In consequence, the majority of total P transport occurs as particulate P. However, in soils with excessive P, for example in soils where organic wastes are frequently applied, the amount of soluble P may be high (Pierzynski *et al.*, 2000). Moreover when conservation practices (e.g. grassed waterways, crop residues in reduced tillage systems) are adopted to reduce the risk of erosion, the transport of soluble P may become more pronounced (Pierzynski *et al.*, 2000).

Of particular importance for P movement from soils is the fact that dissolved (soluble) P in runoff originates from the release of P from a thin zone of surface soil (1 to 2.5 cm) and/or from vegetative material that interacts with rainfall (Sharpley *et al.*, 1996). The interaction of surface runoff with soil is greatest at the surface and declines exponentially with depth (Dougherty, 2006). Sharpley (1985a) showed that the effective depth of interaction between surface soil and runoff increased with rainfall intensity and soil slope. For soluble P transport by surface flow an initial desorption or dissolution of P bound to soil particles first occurs, followed by movement of the P in the water. Soluble organic P that is not adsorbed by soil particles may also be carried by surface or subsurface runoff.

Leaching and subsurface lateral flow of P are a concern only in some specific circumstances, for example in soils with a high degree of P saturation or in fields where the artificial drainage systems provides a pathway for the water and dissolved solutes to move through the soil (Sharpley and Withers, 1994). Under such circumstances the water and the dissolved solutes would percolate slowly through the soil profile. Generally, any form of water flow that bypasses the soil matrix is of environmental concern because it decreases the likelihood that P will be retained by the soil and, as a corollary, it increases the potential for soluble P to enter ground and surface waters (Pierzynski *et al.*, 2000).

The concentration of P in subsurface flow is, in general, noted to be quite low, well below the eutrophication threshold, thus reflecting the considerable sorption capacity of soils for P, particularly in P-deficient subsoil horizons (Pierzynski *et al.*, 2000). Consequently, P leaching is seldom viewed as an important environmental issue except in heavily manured fields, in sandy soils with shallow water tables and in fields with artificial drainage systems as mentioned above. It is to be further noted that while artificial drainage in organic and poorly drained soils increases infiltration and percolation of water, thereby increasing the likelihood of P leaching, it nevertheless decreases P movement in runoff.

As stated above, the transport of particulate P from fields results from soil erosion, that is from the detachment and movement of soil particles in response to the intensity of either the falling rainfall or to the energy of flowing water resulting from the rainfall or snowmelt (runoff). The amount of particulate-P loss will therefore depend on the rainfall characteristics as well as upon the topography of field (greater slope more energy) and soil texture (Pierzynski *et al.*, 2000). Finer and lighter soil constituents such as clays and organic matter will be preferentially transported in runoff (Dougherty, 2006). One important aspect observed is the fact that particles transported in runoff are normally higher in P (as well as in other nutrients) and organic matter than in the soil from which they originated (Jin *et al.*, 2009).

Once particulate P enters a water body, many processes of the P cycle as later discussed in Section 2.4.5 will continue to operate, although at different rates. Particulate (suspended or sediment-bound) P exists not only in association with soil but also with the vegetative material eroded during runoff (Zaimes and Schultz, 2002). In agricultural systems prone to erosion, P in runoff will predominantly be in particulate form, unless, as mentioned above conservation practices (e.g. grassed waterways, crop residues in reduced tillage systems) have been implemented in the fields to reduce erosion in which cases the transport of soluble P may then become more significant.

In fact, the magnitude and forms of P transferred to waterbodies will vary widely depending on the land from which it originated, on the transport route and on management practices (Haygarth *et al.*, 1998). During transport, soluble and particulate P interact with soils bordering the fields with flowing waters and other water bodies. In so doing, the potential for movement of the P originally in runoff is altered, making it difficult to determine accurately the pathways involved in the transport of the P. Furthermore, in-stream processes (e.g., uptake of dissolved P by aquatic biota, changes in equilibrium between particulate and dissolved P forms, deposition of suspended particulate P, and re-suspension of streambed particulate P) are continuously happening from the time the P (as both dissolved and particulate forms) is transported in runoff from the edge of the field to the receiving water bodies, thereby blurring even more the mechanisms and pathways by which the P is moved from the individual fields to the freshwater systems (Sharpley *et al.*, 2000).

2.4.4 Conceptual model of P transfer

The development of a conceptual model of P transfer is an important step for understanding the differences in P movement occurring at different scales (Dougherty, 2006), for providing the basis in modeling P movement to surface waters and for developing risk assessment tools (such as the P index which will be discussed in Section 2.5.4). A conceptual model of P transfer as illustrated in Figure 2.5 should incorporate the hydrological pathways, the P sources, the forms of P, the mobilisation processes and their inter-relationships in P transport.

Haygarth and Jarvis (1999) defined the *potentially mobile forms of P* as being a non-quantitative concept that describes the P in terms of its chemical, biological or physical forms and in terms of its potential for transport to water bodies. For example fields with either a high P status or a high erodibility index are in both instances considered to have a high potentially mobile P status but no P transport would occur if there is little or no rainfall. In short, Haygarth and Jarvis (1999) defined the potentially mobile P as a conceptual reservoir of soil P which may be vulnerable to transport or loss to waters. They thus differentiated the potential P for transfer from the P that would actually be transported. The potentially mobile form of P that is actually transported to surface waters by the various hydrologic pathways is referred to as the *total P transported* (i.e. the problem in Figure 2.5). The conceptual model in Figure 2.5 is therefore intended to help in the designing of strategies for mitigation of diffuse (non-point) P pollution.

In the conceptual model, the sources of P from agricultural lands include soil P, P fertilisers, soil amendments and crop residues left in the fields after harvest. The contribution of each of these P sources is governed by their availability and the rate at which the P in each of them can be mobilised. The processes of mobilisation in the conceptual model, for the past, has been categorised according to

three distinct mechanisms, notably *chemical* (e.g., leaching), *incidental* and *physical* (e.g., soil erosion) transfers. Incidental modes of transport refer to the short-term P losses, for example when rainfall occurs shortly after application of P fertilisers and/or soil amendments resulting in high concentrations of P in runoff.

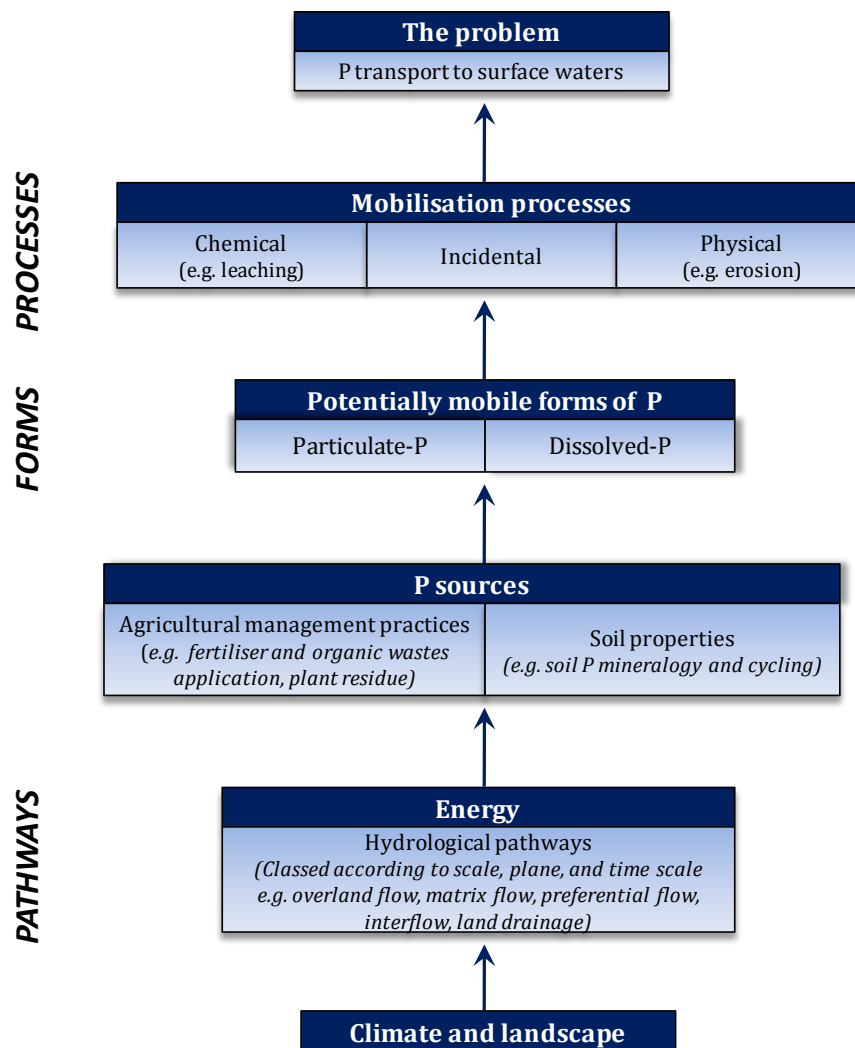


Figure 2.5: A conceptual model describing how non-point source P from agricultural land reaches surface waters (modified from Haygarth and Sharpley, 2000).

The hydrological transfer pathways is summarised in Table 2.3 and are illustrated in Figure 2.6. They are intended to cover a wide range of spatial scales in water flow, as well as the variations in plane and time scale of the flows (Haygarth and Sharpley, 2000). In their conceptual model, Haygarth and Sharpley (2000) categorised leaching as a process rather than a pathway and they defined runoff as the lateral movement of water from the land, at and below the soil surface, that cause a short-term increase in water flow at the catchment outlet (Table 2.3 and Figure 2.6). Surface runoff, as highlighted

by Kleinman *et al.* (2006), may be generated by two non-exclusive mechanisms notably by *infiltration excess surface runoff* which occurs when rainfall intensity exceeds soil infiltration rate and by *saturation excess surface runoff* takes place when the water table rises to the soil surface to exceed the soil's water storage capacity. The occurrence of infiltration excess surface runoff is limited to during rainfall events whereas saturation excess surface runoff may occur during and immediately after rainfall events (Srinivasan *et al.*, 2002).

Table 2.3: Hydrological processes, their approximate timeframes of occurrence and their variations in the plane of water movement (adapted from Haygarth and Sharpley, 2000).

Term	Scale	Timescale	Definition
Interflow	Slope, field	Min, h ^a	<i>Lateral</i> flows below the soil surface.
Matrix flow (also by-pass flow)	Soil		<i>Vertical</i> movement of water downward along subsoil pathways such as wormholes and fissures, often occurring under unsaturated conditions. Also common in very porous media (e.g. sandy texture soils).
Runoff	Slope, field	Min, h ^a	General hydrological term describing the <i>lateral</i> movement of water off the land above and below ground, causing a short term increase in flow at the watershed outlet.
Overland flow (also referred to as surface runoff)	Slope, field	Min, h ^a	<i>Lateral</i> movement of water exclusively over the soil surface, down slope, during heavy rain.
Return flow	Slope, field	Min, h ^a	Where a sub-surface flow pathway emerges at the soil surface.

^a Min: Minutes; h: hours

Moreover, as reviewed by Kleinman *et al.* (2006), saturation excess runoff includes both rain and soil water, while infiltration excess runoff is comprised predominantly of rainwater. Furthermore with infiltration excess surface runoff, rainfall first infiltrates and accumulates at various positions in the landscape. Any subsequent rain received on these saturated areas will no longer be absorbed and will lead to runoff. As reviewed by Dougherty (2006), the saturated zones expand and contract according to changes in the watershed water balance and they are referred to as the variable source areas in the watershed (Figure 2.6). The variable source area concept when extended gives rise to the critical source area concept discussed in Section 2.5.4.

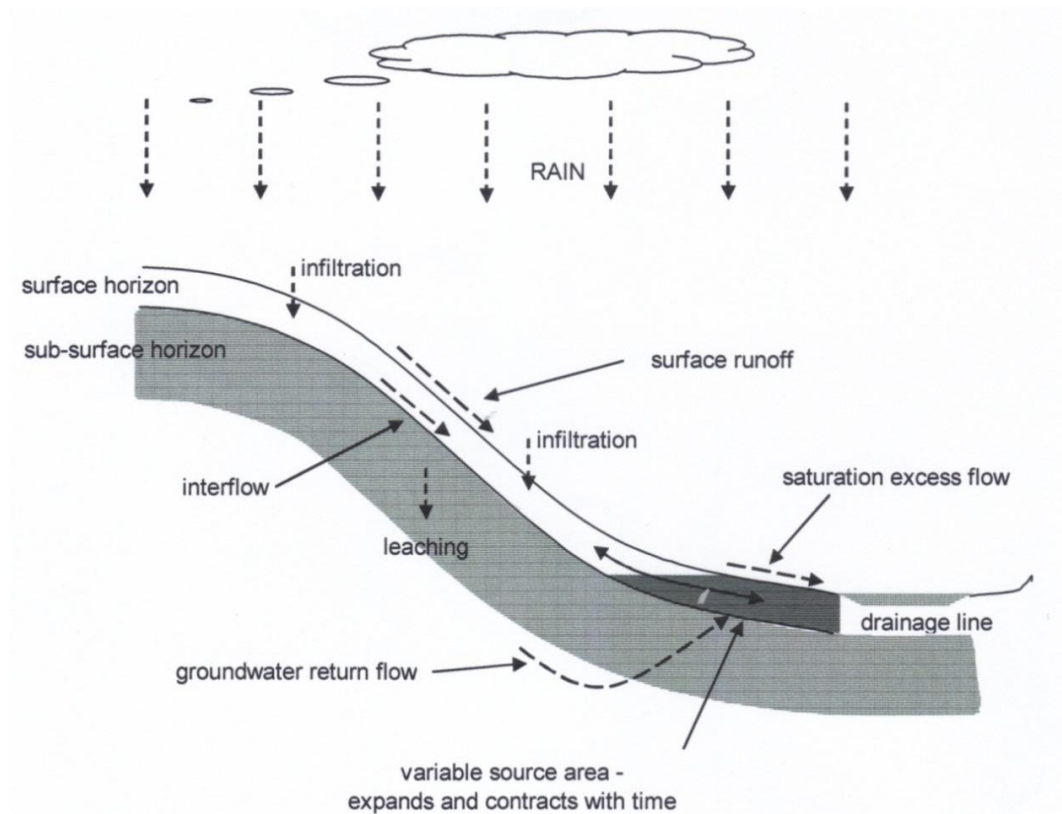


Figure 2.6: Basic components of hillslope hydrology (Dougherty, 2006).

2.4.5 Terminology for the mobile forms of P

Research on P transfer from soils to waters, as can be seen from the conceptual P transfer model described in Section 2.4.4, is multidisciplinary in nature. One consequence is that terminologies used for the different forms of P concerned with transfer to waterbodies have varied according to research disciplines and have led to misinterpretations and confusions. As a remedy, Haygarth and Jarvis (1999) have provided a simple classification of terms in P transfer, based on *processes*, *pathways* and *forms of P*.

Any form of P measured in soil is in fact just a snapshot of its potential for transfer to water. A distinction among the different P forms is necessary to highlight their differences in adsorption and desorption properties because these differences in properties determine the vulnerability of the P forms to transport and their eventual bioavailability to aquatic organisms. In this context, the total P content of waters (be it soil solution, runoff, leachate, streams or lakes) can be separated into fractions that are best defined in terms of analytically determined methodologies. This fractionation is particularly important in the understanding of the fate and transport of P (Haygarth and Jarvis, 1999).

Although in Section 2.4.3, physical and chemical terms have been used to describe the different P forms in soil and water matrices, operational definitions need to be adopted in order to avoid confusion and ambiguity. The operational definitions of the different forms of P transported in water have been based on filtration and chemical methods of analysis. Typically, a 0.45µm membrane filtration is adopted to separate the '*dissolved*' from the '*suspended*' forms of P. However as reviewed by Haygarth and Jarvis (1999), there exists undisputed evidence that P can be associated with colloids less than 0.45 µm in size (such as oxides, clay minerals, organic matter) thereby casting further doubt over the term 'dissolved' P. Nevertheless, filtration through 0.45µm filters does provide a convenient and replicable analytical technique (American Public Health Association, 1992).

The most common method for determining P is the molybdenum-blue method of Murphy and Riley (1962). Confusion however often arises when the P measured in unfiltered and undigested samples by the latter method is considered as free 'orthophosphate' or 'inorganic' P (McDowell *et al.*, 2001). The Murphy and Riley procedure has been shown to also determine loosely bound inorganic and organic forms of P, as a result of either a molybdenum-enhanced hydrolysis or a hydrous ferric oxide-orthophosphate reactions, or an interference with silica (Tarapchak, 1983; Ciavatta *et al.*, 1990). Hence taking the Murphy-Riley determined (i.e., Mo-reactive) fractions to be orthophosphate is technically incorrect (Haygarth and Sharpley, 2000) and the term '*reactive*' P (i.e. reactive to Mo-blue reaction) should be more appropriate. An accurate estimate of the orthophosphate can be obtained by chromatographic separation but this technique is expensive and time-consuming (McDowell *et al.*, 2001).

Total P in waters can either be determined directly by using inductively coupled plasma (ICP) analysis or by the Murphy and Riley method (1962) following an appropriate method of digestion. A suitable digestion method to determine total P in water matrices must be able to oxidize organic matter effectively so as to release the P as orthophosphate which can then be determined by the Murphy and Riley method (1962). Three digestion methods have been proposed by the American Public Health Association (1992), namely the *perchloric acid method*, a most drastic and time-consuming method, the *nitric acid-sulphuric acid* method which is usually recommended for most samples and the *persulfate oxidation technique*, the simplest digest method among the three mentioned. The most important factors to be considered when choosing the digestion method for determining total P accurately and precisely in soil solution and other water matrices are the sensitivity of the procedure, the simplicity of the oxidation conditions and the ability to control the reaction to avoid contamination (Rowland and Haygarth, 1997). The difference between the total P and reactive P provides an estimate of the '*unreactive*' P, which is generally considered to represent the organic P forms, but this is subject

to some qualification because some condensed forms of P, such as the polyphosphates will also be included within the unreactive fraction.

In line with the discussion above, a systematic nomenclature, which is shown in Figure 2.7, has been proposed by Haygarth and Sharpley (2000) in the endeavour to remove the ambiguity and inaccuracies associated with such terms as '*soluble*', '*dissolved*', '*particulate*', '*inorganic*' and '*organic*'. The equivalence of the terms proposed in the systematic nomenclature of Haygarth and Sharpley (2000) in relation to those commonly encountered in the literature is given in Table 2.4.

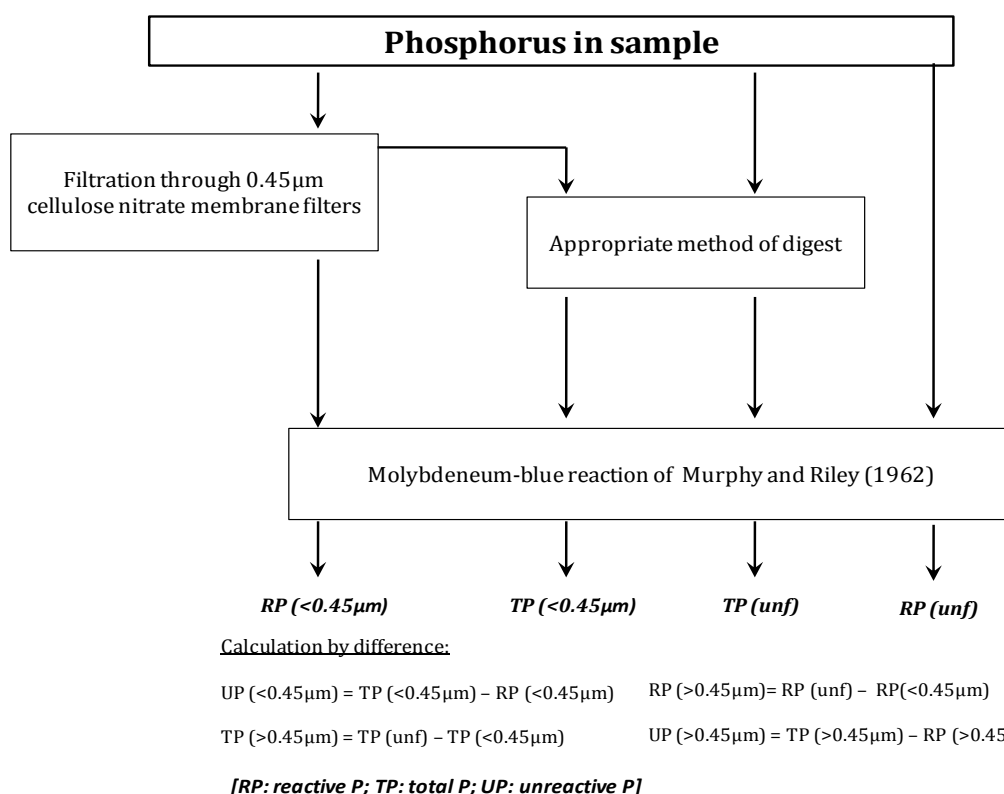


Figure 2.7: Nomenclature for the operationally defined forms of mobile P (adapted from Haygarth and Sharpley, 2000). Samples are defined specifically according to filter size. The suffix in parentheses relates to the micron size of the filter used.

Table 2.4: Suggested methodologically defined terms of P forms in waters with their equivalence commonly found in the literature (adapted from Haygarth and Sharpley, 2000).

Classification ^a (proposed by Haygarth and Sharpley, 2000)	Equivalent terms employed in literature ^b
TP (unf)	Total P from an unfiltered sample
TP (>0.45µm)	Particulate P, sediment-bound P, suspended P
TP (<0.45µm)	Total dissolved P
RP (unf)	Total reactive P
RP (<0.45µm)	Molybdate-reactive P, dissolved reactive P, soluble reactive P, dissolved molybdate reactive P, orthophosphate, inorganic P, phosphate
RP (>0.45µm)	Molybdate-reactive particulate P, particulate reactive P
UP (unf)	Total organic P
UP (<0.45µm)	Dissolved organic P, soluble organic P, dissolved nonreactive P
UP (>0.45µm)	Particulate organic P

^a [RP: reactive P; TP: total P; UP: unreactive P]

^b May not necessarily be correct

Another term which is frequently encountered when discussing water quality issues related to P is *bioavailable P* (also algal-available P). The bioavailable P represents the P fraction which is potentially available for algal uptake. It consists of soluble P and a variable portion of particulate P (Sharpley *et al.*, 1992). Almost all soluble P transported via runoff is in effect biologically available while particulate P that enters streams and other surface waters must first undergo some type of solubilisation reaction before becoming available to the aquatic biota (Figure 2.8).

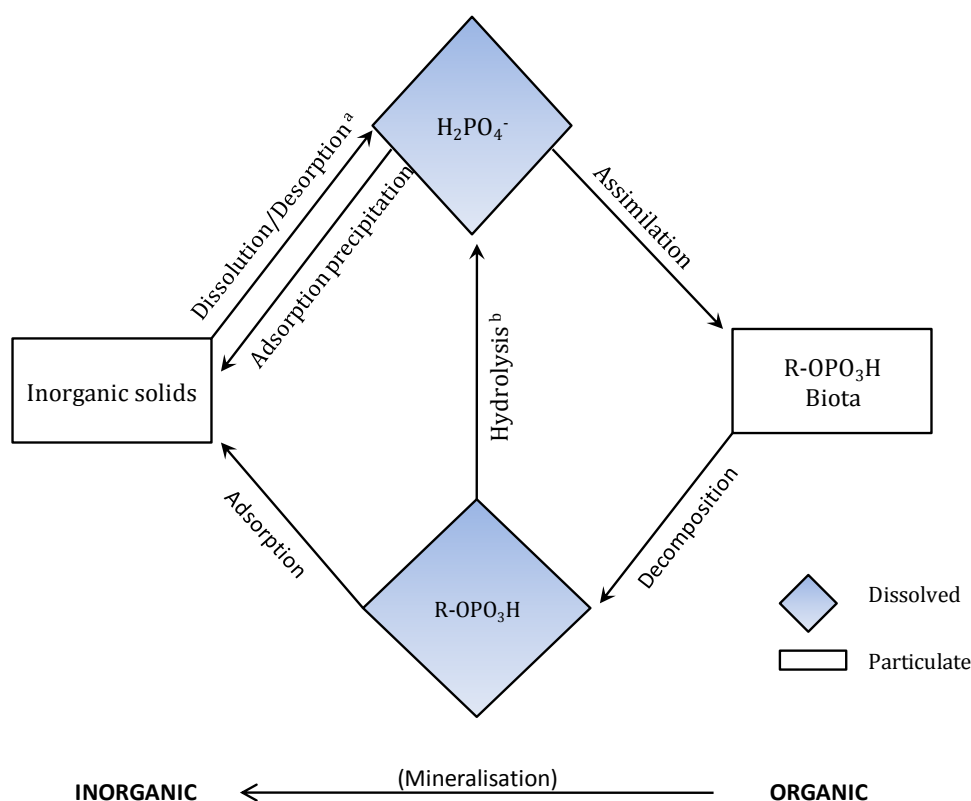


Figure 2.8: The biochemical P cycle in aquatic systems (Glennie *et al.*, 2002). [^a reductive, photochemical, pH variability; ^b enzymatic, photochemical, pH variability].

Algae can instantaneously consume dissolved reactive P while particulate P (in organic or inorganic forms) represents long-term P sources (McDowell *et al.*, 2001). The bioavailability of particulate P varies between 10 to 90% depending on the nature of the eroding soil and of the receiving lake (Daniel *et al.*, 1998). Monitoring the amount of bioavailable P (both soluble and particulate) transported in agricultural runoff is important since bioavailable P plays an important role in the eutrophication of receiving water bodies. As iron oxide strips can be used to quantify bioavailable P with precision in soil and agricultural runoffs, the impact of agricultural runoff on biological productivity of receiving waters can be accurately evaluated (Sharpley, 1993a).

2.5 Management of agricultural P for environmental protection

The continued input over the years of fertiliser and manure, very often in excess of crop requirements, have led to a build-up of soil P levels to a point where the P has become an environmental concern, particularly in areas of intensive crop and livestock production (Sharpley *et al.* 1994). Though the loss of agricultural P is not of economic importance to a farmer, it has often lead to the deterioration of water quality due to accelerated eutrophication which has significant off-site economic impacts

(Sharpley *et al.*, 1994). As a result, in agricultural P management, there is an increasing shift towards strategies that protect the environment, in particular strategies that recognize the fact that it is less expensive to treat the cause of eutrophication rather than to remedy its effect that acknowledge the findings that remedial measures in the watershed takes several years to become effective (Sharpley *et al.*, 2006). Haygarth *et al.* (2014) associated this delay in the intended reduction in catchment P fluxes to result from the legacy effect associated with the buildup of P in the topsoil and the complex release patterns in catchments and rivers.

In search of strategies that will minimize the environmental impact of P, several important factors are taken into account. For P to cause an environmental problem, there must be a source of P (e.g. soil P, manure or fertiliser applications) and the P must be transported to a sensitive water body (Sharpley and Tunney, 2000). As summarised by Higgs *et al.* (2000), the three main routes by which P can be lost from land to water are in eroded soil, by surface runoff and in leachate. The different strategies and management techniques elaborated to prevent agricultural P loss revolve around those source and transport factors. In principle, by carefully matching fertiliser P applications with crop needs and yield goal potentials, the accumulation of P in soil will be minimised, with most of the added P being removed in crop produce as grain or forage (Sharpley *et al.*, 2006). While management of the source factors seeks to prevent the buildup of P in the soil to a level above that sufficient for optimum crop growth, management of the transport parameters aims at controlling the movement of P from agricultural lands to sensitive locations such as freshwater bodies (Sharpley and Beegle, 2001).

In practice, remedial P management strategies seek to identify critical areas of P export from the agricultural landscape that is those areas in the landscape where the source and transport factors converge to favor P loss. The critical areas are reported to change rapidly with time, expanding and contracting quickly during a storm as a function of rainfall intensity and duration, antecedent moisture conditions, temperature, soils, topography, ground water, and moisture status over a watershed (Sharpley *et al.*, 2001a). This section describes the effective P management approaches that have been targeted towards those critical source areas to minimize agricultural P losses to water bodies.

2.5.1 Soil P testing

As stated above in Section 2.5, one strategy to protect freshwater sources from accelerated eutrophication is to identify critical source areas of P in the agricultural landscape. In this context, as we move from agronomic to environmental concerns, it is reasoned that soil P in excess of crop requirements is vulnerable to removal by surface runoff or leaching. Accordingly, soil P tests can be used to indicate when P enrichment of runoff may become unacceptable (Sharpley *et al.*, 2001a). The

common approach to-date has been to extend the interpretation of agronomic soil P tests. Indeed a soil test P level above which a crop response is no longer expected must be in excess of crop needs and can therefore be considered polluting (Figure 2.9). It could however not be assumed that the calibration of the soil test for crop response is equally valid for indicating runoff P enrichment potential.

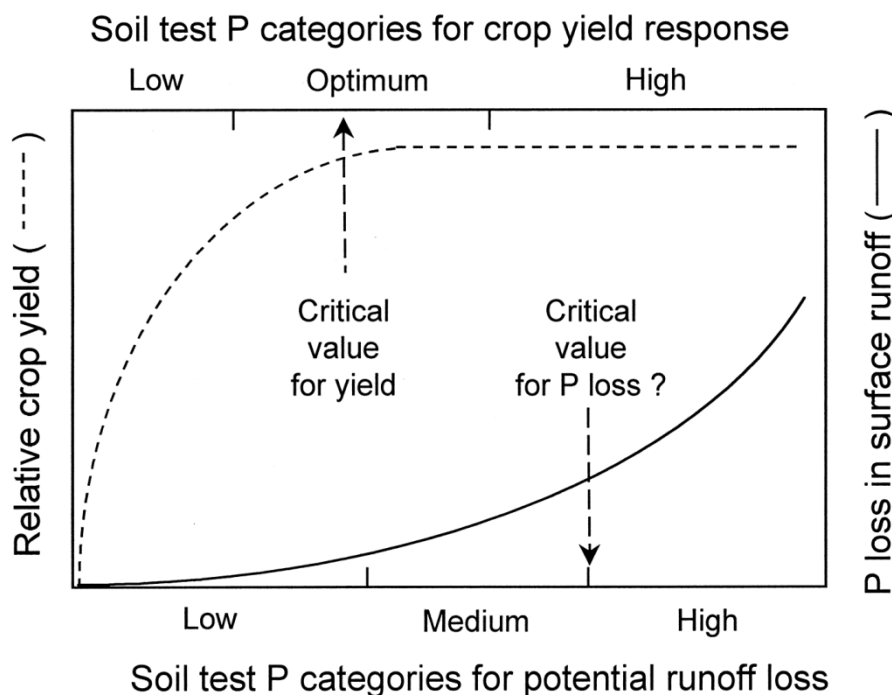


Figure 2.9: Interpretation of soil test P levels for agronomic and environmental purposes (Sharpley *et al.*, 2001a).

A variety of soil extractants (e.g., Mehlich-3, Olsen) have been evaluated as indicators of P loss potential, by relating the soil P extract to P in surface runoff or subsurface leachate (Sims *et al.*, 2002; Carmo Horta and Torrent, 2007). Numerous studies (e.g. Sharpley *et al.*, 2001a; Weld *et al.*, 2001; Schindler *et al.*, 2009) have in fact shown that the dissolved P concentrations in actual runoffs are closely correlated with the P content of surface soils except when the latter have recently been fertilized. This finding has led to extractions of surface soils with water or 0.01M calcium chloride to be frequently used for estimation of P concentrations in either runoff or leaching waters (Casson *et al.*, 2006). Apart from water and calcium chloride extractions, the concept of degree of soil P saturation has also been used as an indicator of P loss potential since it integrates the dominant properties controlling the P sorption-desorption status of soils (Beck *et al.*, 2004). The concept of soil P saturation is based on the premise that more P will be released from soil to surface runoff or leaching water as P saturation or amount of P fixed increases (Sharpley, 2000b).

The split line model is one innovative approach proposed by McDowell and Sharpley (2001) for the determination of a soil environmental P threshold. In the model the relationship between soil P and dissolved P in runoffs is split into two sections, one with greater P loss per unit increase in soil P saturation than the other. The split line model has been applied in many instances to identify soils whose P status is considered to be excessively high and likely to result in significant environmental risk (Moody, 2011). In addition it has permitted the scope of agronomic soil P tests, such as Mehlich-3 and Olsen, to be enlarged for environmental interpretations (Sims *et al.*, 2002; Carmo Horta and Torrent, 2007). However, in other instances as reported by Moody (2011), there is only a single linear relationship between the degree soil P saturation and the P extracted in 0.01M calcium chloride solution with no indication of a change point.

While the studies to develop an environmental soil P test in general have shown promise in describing the relationship between the level of P in soil and surface runoff P concentration, it must be recognised that they have limited application in environmental P risk assessment (Sharpley, 2000b). This is because soil testing does not consider transport processes that connect a site with surface waters, nor the management factors and the proximity of the source area to a waterbody sensitive to P inputs (Maguire *et al.*, 2005). Consequently, critical soil P levels alone have little meaning with regard to P loss potential unless they can be used in conjunction with an estimate of potential surface runoff, erosion and leaching (Sharpley *et al.*, 2012).

Even though soil P testing on its own has limited usefulness, there is nevertheless scope for using it in a comprehensive assessment of P loss risk, for example for screening soils with potential P movement where a complete site evaluation is warranted (Maguire *et al.*, 2005). Best management practices to control P movement can only be most effective if they are targeted to the hydrologically active source areas of the watershed during storm events (Gburek and Sharpley 1998; Gburek *et al.*, 2007). A more holistic approach which integrates the source of P, as indicated by soil test with site characteristics affecting P transport and management factors should therefore be elaborated to predict the risk of P loss from fields in a farm or in a watershed. Such holistic approaches would include modeling and the P indexing system which are discussed in Sections 2.5.3 and 2.5.4 respectively.

2.5.2 Best management practices

2.5.2.1 Management of fertiliser P sources

The most common sources of P in agricultural runoff are mineral fertilisers and organic materials such as manures and composts (Sharpley *et al.*, 1994). Doses, timing and method of application of these P nutrient sources represent the most obvious practices that can be managed to minimize the potential

risks of P movement from agricultural lands. As reviewed by Mullins *et al.* (2005) and already stated in Section 2.5.1 dissolved P in runoff from simulated rainfall is highly correlated with soil test P levels (except in recently fertilized plots where the dissolved P in surface runoff is primarily related to the amount of soluble P in the fertiliser source), with P rate and with timing of application. Sharpley *et al.* (1994) have previously reported that the dissolved P concentration in runoff waters from fields receiving broadcast fertiliser P was much higher than from fields where comparable rates were incorporated 5cm below the soil surface. Fertilisers that are applied on the surface of the soil will be most prone to loss especially when they are applied at high rates or in high rainfall areas or just before a storm or irrigation event (Withers *et al.*, 2003). Furthermore, fertilisers that are less water-soluble should be preferred in some regions (e.g. high rainfall regions) to minimize transport of P in runoff waters (Zaimes and Schultz, 2002).

As mentioned, apart from the dose, form of P used and method of placement, timing of the fertiliser P applications to coincide with dry weather had been observed to be of fundamental importance in reducing incidental P loss from the fertilisers, especially in high rainfall areas or where frequent irrigation is practiced (Withers *et al.*, 2003). As the major portion of annual P loss in runoff generally occurs during one or two intense storms, avoiding P applications during periods of the year when intense rain storms are expected would reduce the potential for P loss (Sharpley *et al.*, 2001a). In this context, an increase in the time interval between applying P nutrient sources and a rainfall/runoff event has been found to reduce P transport in runoff (Sharpley *et al.*, 2001a). Zaimes and Schultz (2002) have on the other hand, reported that nutrient concentration in runoff can be more dependent on the number of rainfall events after the time of application than on the annual quantity of runoff or rainfall.

2.5.2.2 Cultural practices

The ultimate goal when managing P for environmental protection is to prevent a connection from being formed between the sources of P and the transport pathways. From this perspective, the strategies that have been developed to reduce P transport can be categorised into *preventive* and *interception* measures. While the preventive remedies such as cover crops are aimed towards reducing surface runoff and erosion at its source and at minimizing the exposure of applied P, interception technologies such as the use of buffer strips are designed to remove P originating elsewhere in the landscape in the throughflow (Sharpley, 2000b). Conservation tillage and crop residue management, buffer strips, terracing, contour tillage, cover crops, grassed waterways, and creation of riparian zones are the other practices that have been proposed for reducing P movement by erosion and runoff (Sharpley and Withers, 1994; Sims and Kleinman, 2005). Basically, these

practices would reduce the impact of the rain on reaching the soil surface or would decrease runoff volume and velocity, or would increase soil resistance to erosion (Sims and Kleinman, 2005).

Conservation tillage practices, especially when most of the crop residue is left on the soil surface, have been shown to reduce soil and P losses considerably (Jin *et al.*, 2009). However, conservation tillage would also increase the P status of surface soil arising from the stratification of the P in the soil profile, with the highest concentrations of soluble P in the upper few centimetres due to the repeated surface applications of fertiliser or manure P without the mixing associated with plowing (Sims and Kleinman, 2005). To avoid an accumulation of P in the upper few centimetres of soil, the intermittent use of conventional tillage to mix the topsoil with the subsoil has been recommended (Sims and Kleinman, 2005). Although conservation tillage practices would reduce runoff and erosion, they may show differential effects on dissolved and particulate P losses (Sharpley *et al.*, 2006). Indeed though in conservation tillage, the residue left on the soil surface would decrease surface runoff and soil erosion, infiltration of water through the soil profile may be increased during rainfall events, thereby enhancing P losses by leaching (Sims and Kleinman, 2005).

Furthermore, mulching with crop residues would minimize the risks of runoff and soil erosion not only by enhancing the water infiltration through the soil profile and decreasing runoff volume but would reduce soil detachment and particle transport by dissipating the velocity of impact of the rain at the soil surface (Singh and Lal, 2005). A new farming approach, stubble mulch tillage has been developed specifically to keep the soil surface protected at all times, by either leaving crop residues on the surface during fallow periods to protect the soil surface from raindrops or by growing cover crop (e.g. grasses, legumes or other herbaceous plants) to scavenge excess nutrients remaining in the soil profile after harvest of the previous crop, to break up root restricting layers, to improve soil structure and to increase soil organic matter during periods when the primary row crops or cash crops are not being grown (Sims and Kleinman, 2005).

Grassed waterways have in addition been recommended to trap sediment and to reduce water flow to a non-erosive velocity from the landscape without causing flooding (Sims and Kleinman, 2005). In some cases, grassed waterways are established as cross slope diversions to intercept runoff and break up effective slope length (Sharpley *et al.*, 2006). In addition different riparian buffer models in the form of filter strips, contour grass strips, field borders, alley cropping and vegetative barriers have been developed to fit various landscape and land management scenarios (Zaimes and Schultz, 2002). The buffers implemented are primarily viewed as means to trap sediment leaving agricultural lands, to promote infiltration of runoff through the soil and thus as ways of limiting particulate P transport. Dissolved forms of the P in the runoff may be sequestered through assimilation and immobilisation of

the P by the same vegetation composing the buffers (Sharpley *et al.*, 1994, Sims and Kleiman, 2005). The effectiveness of vegetative buffers is however reported to decrease with time as sediment accumulates in the buffer so that eventually instead of acting as sinks for P, they become sources of P (Sharpley, 2000b; Sheppard *et al.*, 2006; Zaimes and Schultz, 2002).

2.5.3 Modeling P transport

The identification of critical source areas i.e. areas with disproportionately high potential P losses, is essential in order to be able to efficiently manage on a routine basis P movement at the field and also at the watershed scale. Tools which have been proposed for the identification of the critical source areas of P movement include as reviewed by White *et al.* (2009), comprehensive hydrological and water quality models such as the Soil and Water Assessment Tool (SWAT).

In fact, many of the basic approaches of modeling P transport were developed in the 1980s and despite other advancements in modeling, such as graphic user interfaces and the use of geographic information system (GIS) layers, the basic approaches to modeling P movement have changed very little (Radcliffe and Cabrera, 2007). Modeling of P transport generally implies the simulation of soil erosion, runoff, leaching and in-stream processes and the necessary input data for the simulation most commonly include land use, soil texture, topography and management practices. Sharpley (2007) divided the different modeling approaches into three broad categories, namely *processed based*, *export co-efficient* and *statistical* models.

Processed based models simulate watershed processes through mathematical representation of parameters such as rainfall, infiltration, P application method, rate and timing and land management. The Soil and Water Assessment Tool (SWAT) is an example of a catchment scale processed based model and it was developed for the United States Department of Agriculture's Agricultural Research Service to predict the impact of land management practices on water, sediment and agricultural chemical yields in complex watersheds (Ekstrand *et al.*, 2010). The SWAT model reflects the soil P, water and plant interactions (e.g. mineralization, decomposition and immobilization), P sorption, leaching, organic and inorganic fertiliser P application (Chaubey *et al.*, 2007). In addition to those processes, movement of P as both dissolved and sediment bound P in surface runoff and the in-stream P cycles are also simulated in the SWAT model (Chaubey *et al.*, 2007). Other examples of processed based models include the Hydrologic Simulation Program-Fortran (HSPF), Areal Nonpoint Source Watershed Environmental Response Simulation (ANSWERS) and the Watershed Ecosystem Nutrient Dynamics-Phosphorus (WEND-P). While most processed based models primarily focus on hydrologic and soil chemical processes to predict P movement from agricultural systems, WEND-P is designed to examine the long-term storage, process and cycling mechanisms by which the mass balance of P can

change in various land uses and under various management scenarios throughout a specified watershed (Kort *et al.*, 2007).

Export co-efficient models, for their part, rely on simple and empirical formulations to represent transport processes, for example the use of a runoff co-efficient for generation of runoff from rainfall. One example of the export coefficient models is the Generalised Watershed Loading Functions (GWLF) model which conceptualizes the watershed as a number of small land units (often referred to as hydrologic response units) that produce surface runoff and erosion. The dissolved and suspended nutrients in stream flow are then estimated at the watershed outlet by loading functions that empirically relate nutrient concentrations in runoff and sediment to the characteristics of the watershed and of the hydrologic response units (Schneiderman, 2007). Such models are commonly utilized to predict how stream flow and nutrient loads from a watershed are affected by land use, watershed management and climate conditions.

The statistical models range from simple regression equations to relatively sophisticated derived-distribution approaches for prediction of the frequency distribution of nutrient loadings and concentrations. The Annual P Loss Estimator (APLE) is an example of a statistical model and has been proposed for prediction of field-scale P loss in runoff, erosion and under different management conditions (Vadas *et al.*, 2012). This model has a user-friendly structure and requires as inputs the annual precipitation, the annual runoff and erosion, the soil test P level, clay content and organic matter. The APLE model simulates sediment bound and dissolved P loss in surface runoff and is intended to simulate edge-of-field P loss from uniform fields (Vadas, 2011). In the APLE model, the dissolved P concentration in runoff from surface-applied fertilisers as explained by Vadas *et al.* (2008) is calculated as follows:

$$\begin{aligned} \text{Runoff dissolved P in mg L}^{-1} = & (\text{available fertiliser P in mg}) \times (\% \text{fertiliser P released}) \\ & \times [(\text{P distribution factor})/(\text{total rain during the event in L})] \end{aligned}$$

The P distribution factor is a unitless factor which is obtained in the following way:

$$\text{P distribution factor} = 0.034 \exp [(3.4) (\text{runoff to rain ratio})]$$

As reviewed by Sharpley (2007) statistical relationships are commonly employed in the P index concept (discussed in Section 2.5.4) which may itself be considered as a statistical model.

The use of models such as SWAT to identify critical source areas have been shown to be powerful tools for the management of P and sediment but as indicated by White *et al.* (2009), they have several possible limitations. In general, the major drawback in modeling resides in the uncertainties in model

computations which occur due to an imperfect or incomplete understanding of the physics, chemistry and biology of the real world leading to numerical approximations and to inaccurate parameter estimates (Sharpley, 2007). Furthermore, the quality of the output from the models depends to a great extent on the accuracy of the input data required by the models and then on the considerable expertise, talent and proficiency of the modeler in understanding what the problems are while designing the model. It is thus imperative that the modeler defines clearly what the model can be useful for and what it is not designed to do (Sharpley, 2007). The SWAT model for instance considers infiltration excess runoff mechanisms (White *et al.*, 2009). In the generation of runoff other mechanisms such as saturation excess may also be significant. It is moreover worth emphasizing as indicated by Gburek *et al.* (2005), that models for nutrient transport are often developed for a specific region or country alone and therefore their wider applicability on account of their deterministic nature and exact forms of the output is to a large extent not certain. Indeed the process of calibrating a model at discrete locations as found by White *et al.* (2009) for the SWAT model, does not necessarily improve the spatial accuracy of the model. In view of the shortcomings of modeling in the studies of P transport from agricultural fields, the researchers are seeking more flexible, reliable and meaningful techniques of predicting P movement from soils, one of this is the P index approach which is reviewed in Section 2.5.4 below.

2.5.4 Phosphorus index

2.5.4.1 Origin of the P index

A P index is a field assessment tool specifically designed to enable farmers, extension officers and field managers to identify critical source areas that is those agricultural fields or part of them which are most vulnerable to P loss in the watershed. It integrates the major source and transport factors controlling P movement. The P index is now widely adopted in the USA, as well as in several countries in Europe, to estimate the risk of P loss from agricultural areas to surface waters (Berzina and Sudars, 2010). Currently in the USA, 47 states have implemented the P index as a site assessment tool to identify critical source areas where remedial practices are to be targeted (Sharpley *et al.*, 2011). In Europe, countries such as Denmark, Norway, Sweden (especially those surrounding the Baltic Sea) have integrated the P index to improve the management of agricultural P.

The original P index was developed by Lemunyon and Gilbert (1993) with the initial goals of (i) assessing the risk of P transport from a field to a waterbody, (ii) identifying the critical factors that influence P loss and (iii) helping to select management practices that would decrease P loss from a site. In the following two decades since it was introduced by Lemunyon and Gilbert (1993), the P index has evolved considerably from being a critical source area identifier to serving now as best

management practice selectors in manure application scheduling tools, in manure application rate calculators, and regulatory tools of some states in the USA (Sharpley *et al.*, 2012). As many as 34 site variables have to-date been included in the different P indices developed across USA and Europe (Nelson and Shober, 2012). The original P index comprised only eight characteristics namely soil erosion, irrigation erosion, runoff class, soil P test, P fertiliser application rate, P fertiliser application method, organic P source application rate and organic P source application method and the five value categories of none, low, medium, high and very high (Table 2.5).

Table 2.5: Framework of the original P index as proposed by Lemunyon and Gilbert (1993).

Site characteristic (<i>Weighting factor</i>)	P loss rating (value)				
	Negligible (0)	Low (1)	Medium (2)	High (4)	Very high (8)
Soil erosion, tons acre ⁻¹ (1.5)	None	< 5	5 to 10	10 to 15	> 15
Irrigation erosion (1.5)	None	Infrequent irrigation on well-drained soils	Moderate irrigation on soils with slopes of 2 to 5%	Frequent irrigation on soils with slopes of 2 to 5%	Frequent irrigation on soil with slopes > 5%
Runoff class (0.5)	None	Very low or Low	Medium	Optimum	Excessive
Soil P test (1.0)	None	Low	Medium	Optimum	Excessive
P fertiliser application rate, lb P acre ⁻¹ (0.75)	None	< 15	16-40	41 to 65	> 65
P fertiliser application method (0.5)	None	Placed with planter deeper than 2 inches	Incorporated immediately before crop	Incorporated > 3 months before crop or surface applied < 3 months before crop	Surface applied to pasture or applied > 3 months before crop
Organic P source application rate, lb P acre ⁻¹ (1.0)	None	< 15	16 to 40	41 to 65	> 65
Organic P source application method (0.5)	None	Injected deeper than 2 inches	Incorporated immediately < 3 before planting	Incorporated > 3 months before crop or surface applied months before crop	Surface applied to pasture or applied > 3 months before crop

The weighing factor assigned to each site characteristic was based on professional judgment and/or past experience and assumes that some site characteristics may be more prominent than others in influencing potential P movement from the site. For the derivation of the P index, the site characteristic weighing factor is first multiplied by the P loss rating value to obtain a weighted score for that site characteristic. The weighted scores for the eight site characteristics were then summed to obtain a P index which is then interpreted against an arbitrarily established site vulnerability chart (Table 2.6). A low P index for a field would imply a low risk of P loss in that field while a high P index would indicate a high risk of P loss.

Table 2.6: Site vulnerability chart for interpretation of the original P index
(Lemunyon and Gilbert, 1993).

P index*	Site P loss vulnerability
< 8	Low
8-14	Medium
15-32	High
>32	Very high

*P index = \sum (Site characteristic P loss rating value x weight)

2.5.4.2 Evolution of the P index over the past two decades

The initial P index as proposed by Lemunyon and Gilbert (1993) was essentially an edge-of-field screening tool. It has since been considered that to be more realistic, edge-of-field P losses must be evaluated with respect to their proximity, or connectivity to a stream or receiving water body (Gburek *et al.*, 2000). Accordingly, the P index of Lemunyon and Gilbert (1993) has been modified by Gburek *et al.* (2000) to include hydrologic return period that quantifies the probability that runoffs from a field would impact on the stream. The hydrologic return period is incorporated in the modified P index in as return period/distance among the site transport characteristics. A return period describes the recurrence interval of floods/storms, hence highlighting the discharge (amount of water reaching channel via surface runoff, streamflow and base flow) that originated from rainfall, while the distance caters for the hydrological connectivity to drainage networks or surrounding water bodies. For example, in regions where subsurface flow pathways are dominant, the areas contributing P to drainage waters would be localized in high P soils hydrologically connected to drainage networks.

In addition, in the original P index of Lemunyon and Gilbert (1993), a site could be incorrectly ranked as highly vulnerable to P loss on the basis of the site characteristics alone, though no surface runoff or erosion would occur (Sharpley *et al.*, 2003). This shortcoming showed that the additive nature in the P

index calculation as proposed by Lemunyon and Gilbert (1993), poorly reflected P transport from the dynamic and variable source areas on a watershed scale. To correct that shortcoming, Gburek *et al.* (2000) further reformulated the original P index of Lemunyon and Gilbert (1993) by evaluating separately the P source and transport characteristics of a field before combining the two in a multiplicative manner.

Apart from the additive and multiplicative P indices, a third type of P index, namely the component P index, has been proposed by Reid *et al.* (2012) and Osmond *et al.* (2012). The component P index sums the P losses from each of the individual components contributing to the P load, with each P loss component taken as the product of both the transport and source factors (Osmond *et al.*, 2012). The possible sources of P and transport factors considered are listed in Table 2.7. In the component P index, if every possible combination of source and transport is taken into account for each of the two forms of P (particulate and dissolved P), the resulting P index will comprise 48 separate components. The challenge then for the P index developers is to determine, for the soil, landscape, and climatic conditions within their jurisdictions which of the 48 components need to be taken into account and the relative weighting of each of them to reflect the actual risk of P movement into surface water (Reid *et al.*, 2012).

Table 2.7: Phosphorus source and transport factors (Reid *et al.*, 2012).

P form	P source	P transport
Particulate P	Soil P	Surface runoff
Dissolved P	Fertiliser P	Preferential flow to tiles
	Manure P	Matrix flow to tiles
	Other organic P	Matrix flow to surface drains
		Deep percolation to groundwater
		Wind erosion

The Minnesota P index in the USA is one example of the component P index. As described by Lewandowski *et al.* (2006) it assesses P loss risk after modeling the three major pathways of P movement from fields to water, namely erosion, rainfall runoff and snowmelt (Figure 2.10).

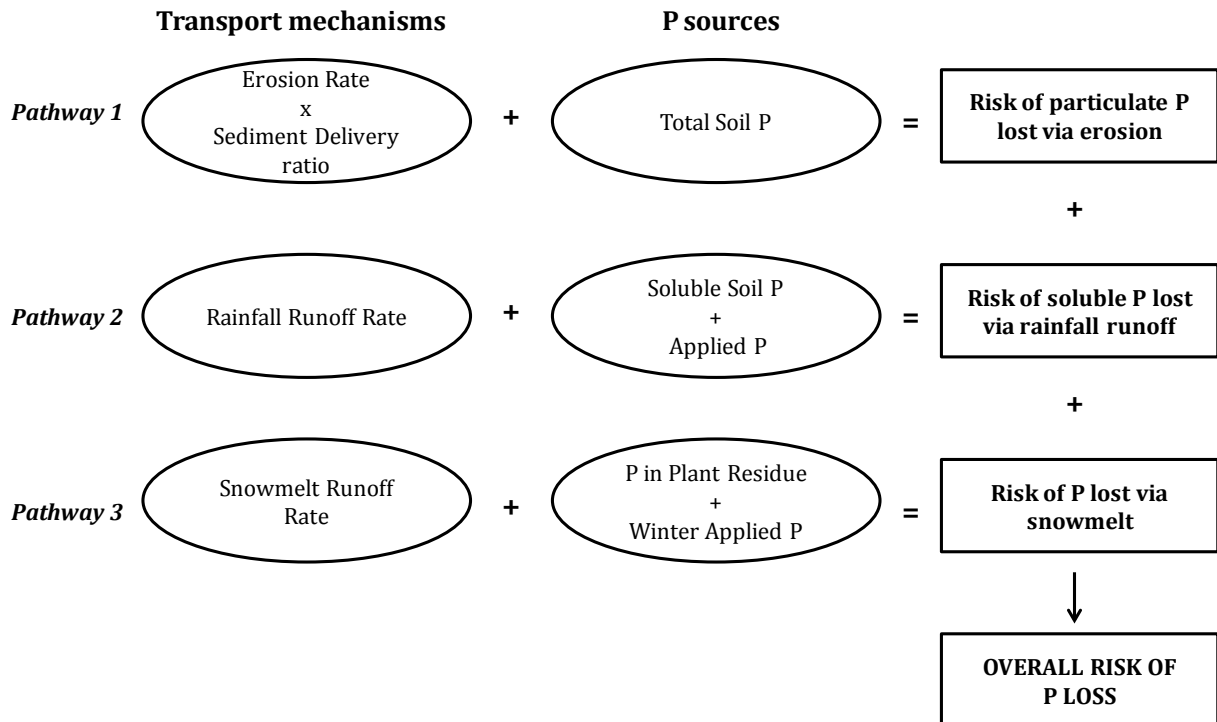


Figure 2.10: The Minnesota P index model (Lewandowski *et al.*, 2006).

The Wisconsin P index adopts an approach similar to that in the Minnesota P index model. It uses a set of equations (e.g. soluble P component = soil runoff dissolved P + direct dissolved P losses from manure/fertiliser applied to the surface) to quantify P loss processes from an individual field to nearby surface water (Good *et al.*, 2012). The P index obtained includes two primary components, namely a particulate P index to estimate delivery of sediment-bound P and a soluble P index that estimates annual dissolved P loads in runoffs.

Different P indices have in fact been developed in different countries to reflect regional variations in soil types, land management, climate, physiographic and hydrologic controls, manure management strategies, and policy conditions (Sharpley *et al.*, 2012). For example, the P index in Norway, though based on the Pennsylvania P index in the USA, includes a ‘plant residue’ factor to account for the P released upon repeated freezing and thawing of plant residues occurring frequently in the cold region of Norway (Bechmann *et al.*, 2005). Other modifications brought to the P index include the addition of factors to account for previously omitted ones, e.g. P loss processes (such as subsurface losses), the conversion of categorical variables to continuous variables, and the revision of P loss weighting factors to reflect more accurately P losses (Nelson and Shober, 2012). The fact that many different versions of P indices exist today highlights not only the flexibility but also the robustness of the P index framework.

Even though a great deal of research has to-date been carried out to justify the various source and transport factors in the P index, little site evaluation of the index ratings has been carried out (Sharpley *et al.*, 2003). Yet as stated by Sharpley *et al.* (2012), placing more emphasis on science-based P index weights will likely lead to meaningful improvements in the index performance as a P loss estimator. Evaluation of the accuracy of a P index in identifying fields that are at risk to P losses will require comparison of the P index ratings with a separate independent and valid assessment of P loss such as the data collected from runoff simulation studies or measured data from field-scale P loss investigations (Nelson and Shober, 2012). In one of the few studies done, Sharpley *et al.* (2003) reported that in a watershed assessment of the P index in Texas and Arkansas in the USA, the P indices adopted in those regions accurately estimated P export at a farm or watershed scale. The Norwegian P index tested at the field and sub-catchment scale has also clearly revealed the potential of the P index to detect areas with the highest risk of P loss (Bechmann *et al.*, 2007).

2.6 Conclusion

Even though the loss of agricultural P may not necessarily be of economic importance to the farming community, the deterioration of water quality from the accelerated eutrophication caused by the P transported from agricultural fields has caused extensive studies to be undertaken on P movement from fields and watersheds during the past two to three decades. Those studies have shown that once P enters a waterbody its removal becomes tedious and very expensive (USEPA, 1986). Accordingly, much of the research done has focussed on a better understanding of P movement from the agricultural fields to waterbodies to prevent accelerated eutrophication by the P loading. The review of literature has shown that though regular soil testing, selection of appropriate P application rates to meet reasonable crop yield expectations, and prescriptive application of mineral fertiliser using methods that maximise the availability of applied nutrients to growing crops are all useful for minimizing losses by erosion, runoff, and leaching; effective management measures to reduce the impacts of agricultural P on water quality require holistic approaches, that integrate source and transport factors in the prediction of P movement from a specific landscape. Indeed, although soil testing and establishment of environmental thresholds are suitable to identify fields with undesirably high soil P levels, it is not suitable on its own to indicate where in the field the P is most prone to be lost because site hydrology, as highlighted by Sharpley *et al.* (2012), may overwhelm source factors (e.g. high P soil) in determining P losses.

Two holistic approaches have been proposed and adopted for elaboration of improved management practices in agricultural fields, namely P modeling and the P index concept. In view of the shortcomings of P modeling as highlighted by the uncertainties in the accuracy of the input data in

model computation, the P index concept on account of its flexibility has often been preferred and has now been integrated in the P management system in many countries e.g. the USA and Norway. In the two decades since its introduction, the P indexing concept has evidently evolved and has been expanded to reflect regional differences, namely in soil type, climate, land management and so on. The success of the P index approach in USA as well as in Europe, where validation research has confirmed its accuracy in identifying field at risk to P transport, indicates that there is plenty of scope for the P index to be used for assessing P losses from sugarcane soils in Mauritius. The study proposed will take advantage of the scope offered by the P index and will in addition provide a simple soil test to indicate the areas where P fertilization can be withheld on account of an excess P already present. The proposed study will moreover indicate to the farmers where improved practices must be targeted to minimize the incidence of accelerated eutrophication of lakes and rivers.

3 Materials and methods

3.1 Introduction

Mardamootoo (2009) extended the scope of the soil P test (0.1M H₂SO₄ as extractant) currently used in Mauritius for predicting how much fertiliser P must be applied to the soil to meet sugarcane P crop requirements such that it can also predict the potential risk of the movement of soil P. However, this agro-environmental soil P test by itself provides an incomplete assessment of the risk which the agricultural P represents to freshwaters. Indeed as indicated by Sharpley and Tunney (2000), adjacent fields having similar soil P levels but different susceptibilities to surface runoff and erosion, due to contrasting topography and management, may have substantially different P loss potential. For example, a high P source with little opportunity for movement will not constitute an environmental threat to freshwater ecosystems. On the other hand, a situation where there is a high vulnerability for transport, but no source of P to move, is also not a threat to the freshwaters. Hence to know the potential risk of soil P to freshwaters, information on the transport processes that will markedly influence the movement of P from soil to water must be acquired. Since surface runoff and erosion are the main processes by which P is exported from agricultural lands in Mauritius (Ng Kee Kwong *et al.*, 2002), simulated rainfall studies were carried out to establish the conditions under which surface runoff and erosion are most prone to occur.

As reviewed by Dougherty *et al.* (2008), rainfall simulation runoff studies on small plots not only provide rapid results but are also an efficient way to collect data and to test hypotheses. In addition they allow greater control of the experimental conditions (both treatments and rainfall) and are adaptable to a range of research approaches. In Mauritius, rainfall simulation runoff studies were carried out as described in the National P Research Project (2001) on 30 sites representing the five main soil groups and the three slope categories under which sugarcane is cultivated. The methods for collecting the runoffs and for determining the different P fractions in the runoff samples are described in this chapter. The data gathered from the simulated rainfall runoff studies were interpreted and combined as elaborated in Chapter 9 to establish a P index risk assessment tool to assess the likelihood of P losses from sugarcane fields in Mauritius. Moreover, to complement the information gained from the rainfall simulation studies, soil at each study site was sampled to assess its P status and to characterize for basic soil properties, notably particle size distribution, pH, organic matter and exchangeable bases.

3.2 Description of study area and sites

3.2.1 Soils

Mauritius, an island of volcanic origin situated in the South-West Indian Ocean, covers a total land area of about 1,860 km². On account of its relief, the island is divided into three agro-climatic rainfall zones, namely the sub-humid (< 1,600 mm annual rainfall), humid (1,600 to 2,600 mm annual rainfall) and super-humid (> 2,600 mm annual rainfall). On the basis of the differences in annual rainfall and age of parent materials, Parish and Feillafé (1965) using a genetic classification system previously adopted in Hawaii, mapped 13 soil groups and the prominent ones being shown in Figure 3.1.

However, 90% of the agricultural land under sugarcane cultivation is located on five soil groups, namely the Low Humic Latosol, Humic Latosol, Humic Ferruginous Latosol, Latosolic Reddish Prairie or Latosolic Brown Forest soil groups (Table 3.1). A detailed description of the different soil groups is presented in Section 4.2.

Table 3.1: Sugarcane land area in Mauritius as per the existing soil group (Mardamootoo, 2009).

Soil group	Sugarcane land area (hectares)
Low Humic Latosol (<i>Humic Nitosol</i>)*	16,289
Humic Latosol (<i>Humic Nitosol</i>)*	6,032
Humic Ferruginous Latosol (<i>Humic Acrisol</i>)*	7,668
Latosolic Reddish Prairie (<i>Eutric Cambisol</i>)*	18,174
Latosolic Brown Forest (<i>Dystric Cambisol</i>)*	12,694
Grey Hydromorphic (<i>Gleyic Cambisol</i>)*	833
Dark Magnesium Clays (<i>Pellic Vertisol</i>)*	549
Mountain Slope Complexes	3,487
Lithosols	3,499
Coral Soil	102
Total land area (hectares)	69,327

* Soil group as per FAO classification (Arlidge and Wong You Cheong, 1975).

3.2.2 Topography

As stated by Arlidge and Wong You Cheong (1975), land slope is a factor of major importance in agricultural land use in Mauritius, not only on account of its strong influence on soil development, but also because of its bearing on the rate and amount of runoff and on the erosion susceptibility of the soil. Arlidge and Wong You Cheong (1975) have accordingly categorized the land slopes in Mauritius

into six classes on the basis of degree of erosion susceptibility and of the measures of control required to minimize erosion (Table 3.2). For practical reasons, the simulation runoff studies as described next in Section 3.2.3 were confined to sites with slope categories ranging from flat to gently sloping (0-8%), sloping (8-13%) and moderately steep (13-20%).

Table 3.2: Classification of land slopes in Mauritius in relation to their degree of erosion susceptibility and measure of control required (Arlidge and Wong You Cheong, 1975).

Slope class	Range of slopes (%)	Description	Degree of erosion susceptibility	Measure of control
1	0 to 2	Flat or almost flat	Nil to very slight	Easily controlled
2	2 to 8	Gently sloping	Slight	Control easily feasible by proper use of selected crops
3	8 to 13	Sloping	Fairly moderate	Control similar to that of class 2 but more extensive and at greater cost
4	13 to 20	Moderately steep	Moderate	Control mostly feasible but more extensive and involving more investment, technical knowledge and maintenance
5	20 to 30	Moderately steep to steep	Strong	Control will be mostly difficult and expensive
6	> 30	Steep and very steep	Strong to very strong	Control will be very difficult and expensive, or neither technically nor economically feasible.

3.2.3 Site selection

Selection of the sites was restricted as mentioned in Section 3.2.1 to the five main soil groups under which sugarcane is cultivated on the island. Six sites in each soil group were selected making a total of 30 study sites. The field measurements generated at 20 sites were utilised to study the factors influencing P mobilisation and thereby develop the P index framework, while the remaining 10 sites were utilised to validate the P index as a risk assessment tool. The study sites were chosen such that they were distributed throughout the island and were located as illustrated in Figure 3.1.

Legend

- Isohyets (mm/yr)
- Lakes/Reservoirs
- Other soils
- Latosolic Brown Forest (B)
- Grey Hydromorphic Soils (D)
- Humic ferruginous Latosols (F)
- Low Humic Gleys (G)
- Humic Latosols (H)
- Low Humic Latosols (L1)
- Low Humic Latosols (L2/L3)
- Low Humic Latosols (L4)
- Dark Magnesium Clays (M)
- Latosolic Reddish Prairie (P1/P2)
- Latosolic Reddish Prairie (P3)
- Mountain Slope Complexes (S)
- Lithosols (P1/P2)
- Location of study site

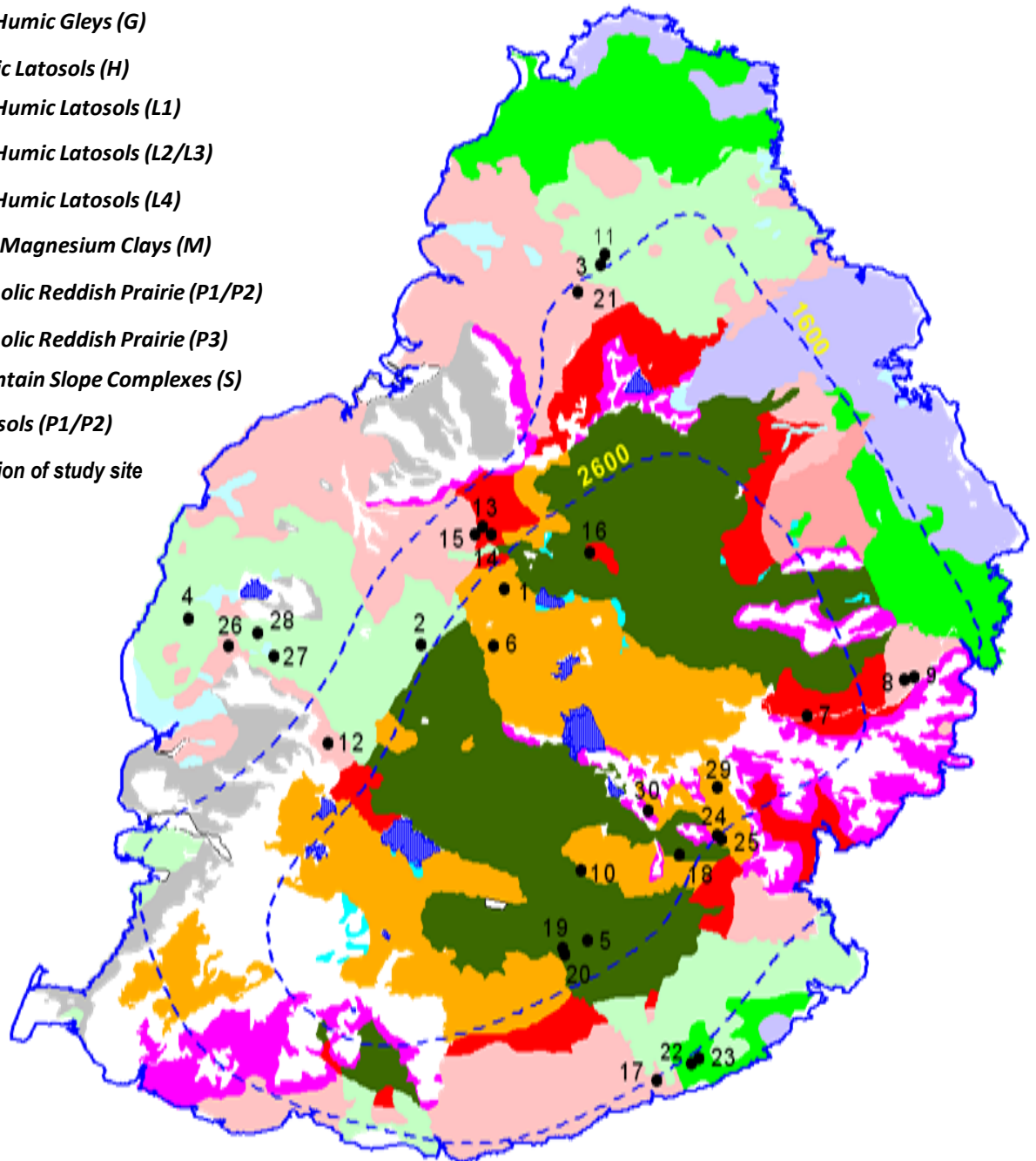


Figure 3.1: The different soil groups in Mauritius according to Parish and Feillafé (1965) and the location of study sites.

In Mauritius, the sugarcane crop (*Saccharum hybrid sp.*) is grown by planting cut pieces of cane (cane setts) and this provides the plant cane 15 to 18 months later. After harvesting the plant cane by cutting at ground level during the period June to December, the regrowth gives rise to the first ratoon crop which is in turn similarly harvested 12 months later. The crop is ratooned repeatedly thereafter until the yield declines to such an extent that replanting is worthwhile. In general, a field is only replanted every seven to eight years, that is, after a plant crop and six or seven ratoons. Phosphorus fertilization is practiced at planting (application in furrows) for the whole crop cycle and fertiliser application rates are based on soil P levels (soil extraction using 0.1M H₂SO₄ as discussed in Section 3.5.6). In the present study, the rainfall simulation studies were carried out just after planting the cane setts that is at a stage in the crop cycle when the soil is still bare (i.e. no crop cover) and is most susceptible to erosion and is richest in plant available P due to recent fertiliser additions.

3.3 Field experimentation to generate surface runoff and bedload

3.3.1 Rainfall simulation

As stated in Section 3.2.2, the simulated rainfall studies focussed on fields having an undulating topography with slopes ranging from 0 to 8%, 8 to 13% and 13 to 20%. These studies as mentioned in Section 3.2.3 were carried out just after planting since it is at this time of the sugarcane crop cycle of seven to eight years that plant cover is absent and so the risk of soil detachment or movement as a result of intense rainfall is most probable. The studies at the 30 sites followed the protocol described in the National P Research Project (2001) and were conducted with soils at field capacity to eliminate antecedent soil moisture as a variable. Moreover, it is at field capacity that the soil would be most prone to movement by erosion.

Three different rainfall intensities (50, 100, 150 mm hr⁻¹) were chosen for simulation at each site, with each simulated rainfall event lasting 30 minutes. Although Mauritius occupies an area of 1,860 km², there is owing to its topography a high variation in rainfall over very short distances, hence the necessity of studying the impact of different rainfall intensities on runoff events. Rainfall intensities of 50 and 100 mm hr⁻¹ were chosen since they are representative of rainfall intensities generally encountered over the island (Seeruttun, 2006). Rainfall intensity of the order of 150 mm hr⁻¹ was also studied as it occurred periodically, particularly during cyclones. It is recognized that this intense rainfall is most destructive to soils, causing soil particles to break up, forming rills and rivulets in the fields which thereafter carry the particles as surface runoff (Padya, 1989). The three rainfall intensities were simulated on separate runoff plots on each slope at each site such that the initial field conditions were those of undisturbed ones prior to the simulated rainfall.

3.3.2 Rainfall simulator used

A rainfall simulator similar to that described by Loch *et al.* (2001) which was primarily designed for field infiltration and erosion studies was utilized in the present study in Mauritius (Figure 3.2). The rainfall simulator consisted of three Veejet 80100 nozzles mounted 1m apart on a manifold, with the nozzles controlled to sweep across a plot width of 1.5m (Loch *et al.*, 2001). The rainfall intensity was varied by adjusting the frequency of nozzle oscillation, the faster the oscillation of the manifold, the higher the rainfall intensity. The rainfall simulator was controlled by an electronic module (Figure 3.2a) which consisted of a circuit board, a stepper motor power supply and driver module and a manifold position feedback module (Loch *et al.*, 2001).



Figure 3.2: Rainfall simulator viewed from (a) the front; (b) sideways and (c) the top.

The frame of the rainfall simulator was made of aluminum tubes having a diameter of 38mm and a wall thickness of 3mm. The operational sequence of the rainfall simulator relied on a continuous water flow through the nozzles. Water was supplied to the rainfall simulator from an electric water pump (0.55 kW) operating at a pressure of 70kPa. Catch trays were positioned on either side of the oscillating nozzles to collect and recycle water from the nozzles that is not sprayed onto the runoff plot (Figure 3.2c). Provision was made for slight up and down adjustments of the plots while at the same time avoiding wasteful and excessive water application over the study area (Loch *et al.*, 2001).

The rainfall simulator was calibrated by placing rain gauges at regular intervals (25cm) in a square grid at ground level as shown in Figure 3.2a. The calibration tests were performed at the recommended operating pressure of 70kPa with the timer setting of the rainfall simulator adjusted so as to correspond with the desired rainfall intensity. The volume of the water collected in the catch cans was measured after 30 minutes and the rainfall intensity in mm hr⁻¹ was calculated. The uniformity of water application by the rainfall simulation was assessed by the Christiansen's coefficient of uniformity (Christiansen, 1942) which is expressed as follows:

$$CUC = 100 (1-S)/nm \quad \text{[Equation 3.1]}$$

Where CUC is the Christiansen's coefficient of uniformity, S is the sum of the absolute difference between the individual catch-can readings and the mean, n is the number of readings measured and m is the mean value of the measured readings.

The calibration data showed that the rainfall distribution within the one meter square plot had a CUC of about 90% (Table 3.3).

Table 3.3: Calibration data of rainfall simulator.

Expected rainfall intensity (mm hr⁻¹)	Observed rainfall intensity (mm hr⁻¹)	Christiansen Coefficient of Uniformity (%)
150	153.5	88.98
150	148.4	89.07
100	91.8	92.45
100	105.9	90.38
50	48.2	92.09

A complete analysis of the source water feeding the rainfall simulator was performed as recommended in the National P Research Project (2001) and as further described in Section 3.7. The electrical conductivity, pH, concentration of soluble calcium (Ca), iron (Fe) and silica (Si) were

determined in the source water. On average the source water had an electrical conductivity of $200\ \mu\text{S cm}^{-1}$, a pH of 6.9, soluble cations less than $1\ \text{mg L}^{-1}$ and an average soluble Si concentration of $9\ \text{mg L}^{-1}$. Also the source water was found to have a total P concentration of $0.05\ \text{mg L}^{-1}$ and an orthophosphate-P concentration of $0.04\ \text{mg L}^{-1}$. Based on the above analytical results, the water was deemed suitable for performing the rainfall simulation on the runoff plots.

3.3.3 Runoff plots and sample collection

Runoff plots (2.1m by 0.75m) were established in duplicate with their longer side in the direction of the slope and along the cane rows (Figure 3.3). Metal frames were installed with their edges 5cm above the ground to isolate runoff plots from the rest of the field. The surface runoff originating from the plots was diverted towards a down slope runoff gutter into a collection vessel. Also a protective cover (not shown in Figure 3.3) was laid on the runoff gutters to prevent source water from entering the collection vessel during the simulated rainfall event. Whenever necessary, tarps (or tarpaulin) were hooked to the sides of the rainfall simulator to avoid wind from disturbing the simulated rain distribution and uniformity over the plot area.

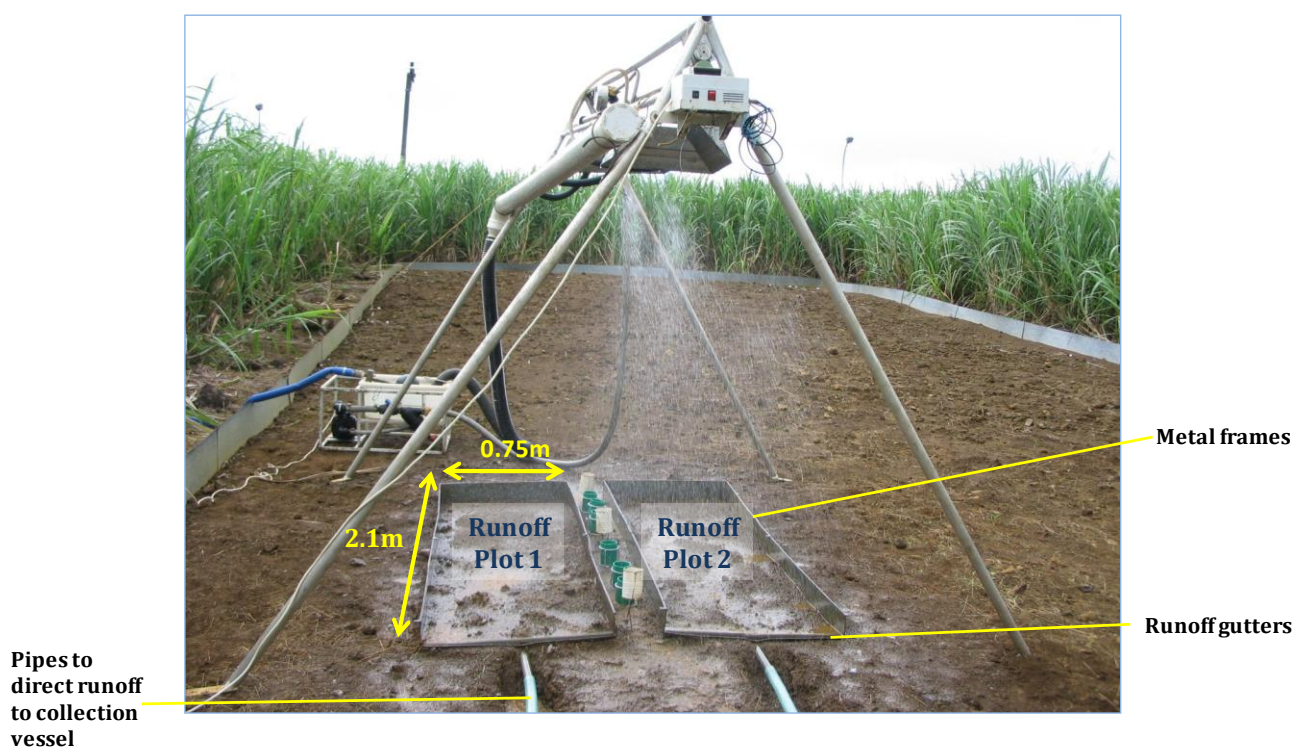


Figure 3.3: Runoff plot establishment with rainfall simulator.

Whenever it occurred, the runoff was collected and its volume measured at five minute intervals during the 30 minutes simulated rainfall event. The individual runoff samples collected were then

thoroughly mixed to ensure a uniform sediment suspension before a subsample (1L) was collected either in a glass amber or a Nalgene bottle and transported to the laboratory for analysis.

The soil lost during the 30 minutes simulated rainfall event (henceforth referred to as bedload) was estimated as outlined in Figure 3.4 by measuring the amount of soil carried during runoffs and that deposited in the runoff gutters. A sample of bedload collected was dried to constant weight in crucible placed in a forced draught oven at 105°C. The resulting dry weight was used to estimate the soil loss per unit area.

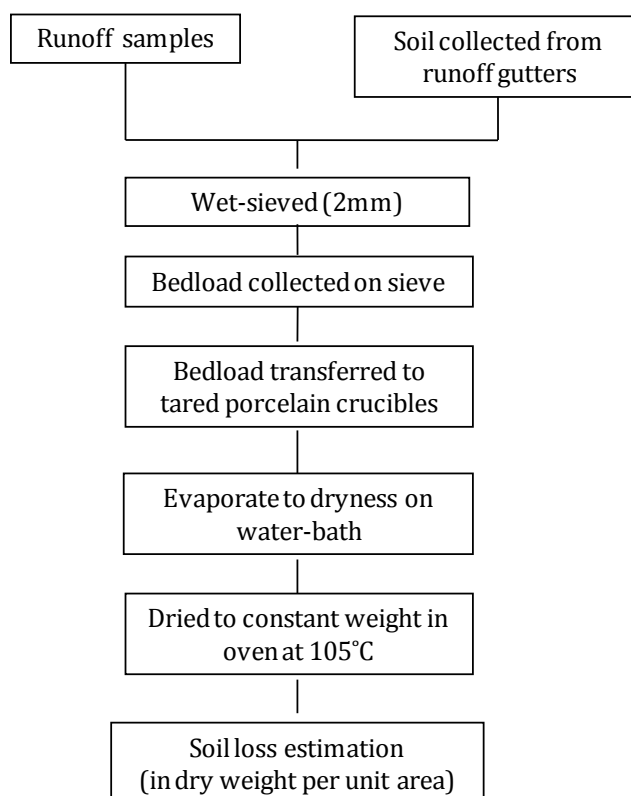


Figure 3.4: Bedload collection and soil loss estimation.

3.4 Analysis of runoff and bedload samples

A flowchart summarizing all the analyses is given in Figure 3.5. The bedload and runoff samples were analyzed for different P fractions. The methodology used for each determination is outlined in this section. The suspended load in the runoff samples was also determined as discussed in Section 3.4.4. For micro scale analysis (such as determination of total P in bedload and organic carbon in soil as described in Sections 3.4.5 and 3.5.4 respectively), the 2mm sieved samples were further ground in a mortar with a pestle to pass through a 60 mesh sieve.

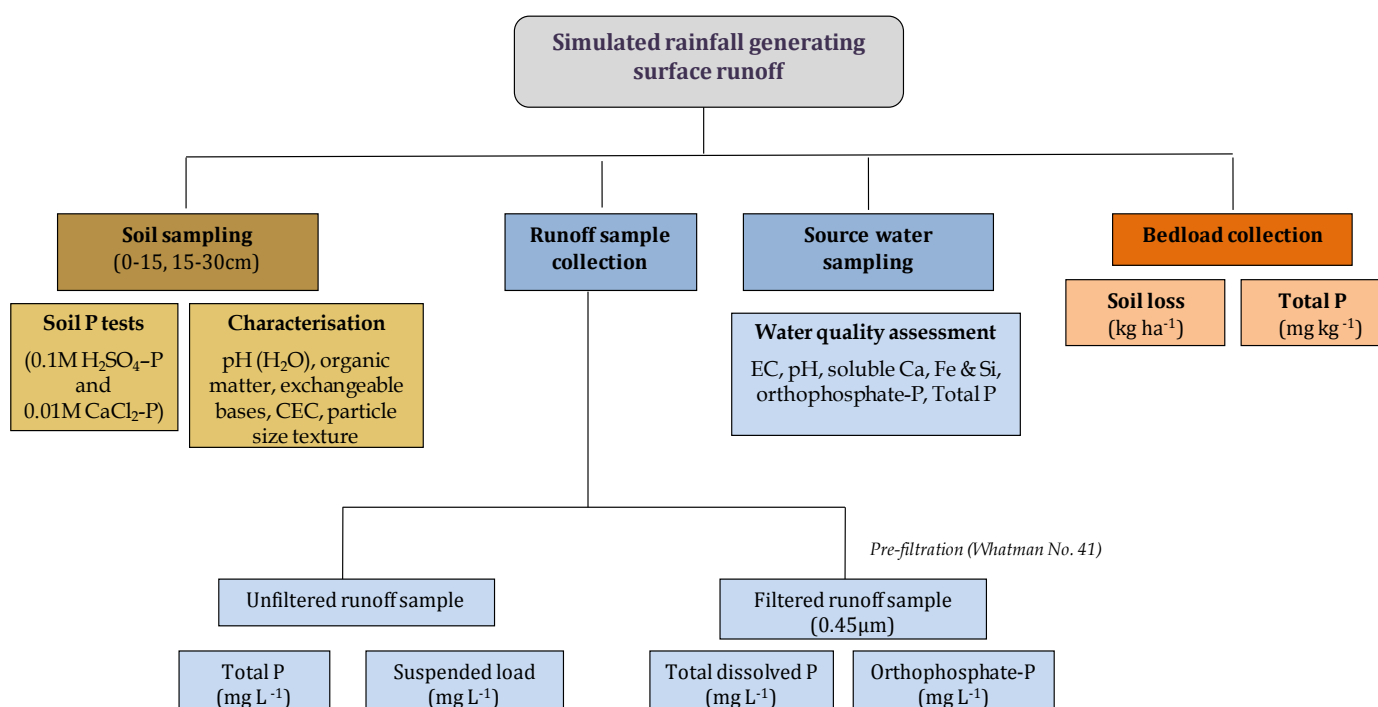


Figure 3.5: Scheme summarising the analyses of soil, runoff and bedload samples.

3.4.1 Runoff sample preparation

Once in the laboratory, half the volume of the runoff samples was vacuum-filtered through a 0.45µm cellulose nitrate membrane filter and kept at 4°C and if analysis was not possible within 48hrs after sample collection then the samples were kept frozen until analysis.

3.4.2 Total P in runoff

Total P in the runoff samples were determined as per the standard methods adopted for the examination of water and waste water (American Public Health Association, 1992). In the present study, 50ml of the filtered samples (10ml for unfiltered samples) were first digested with concentrated nitric acid (1ml) and then with perchloric acid (70%, 5ml) to ensure complete conversion of the total P into soluble forms. After cooling to room temperature, the digest was diluted to about 50ml and neutralized with 6N sodium hydroxide using phenolphthalein as indicator. The neutralized digest was then filtered through Whatman No. 50 filter paper and the P concentration in the solution determined by the method of Murphy and Riley (1962).

3.4.3 Orthophosphate-P in runoff

The orthophosphate P concentration was determined in the 0.45µm filtered runoff samples by the method of Murphy and Riley (1962). The determination first required dilution of a 50ml aliquot of the filtered sample to about 80ml and adjustment of its pH to 5 using *p*-nitrophenol as indicator.

3.4.4 Suspended load in runoff

The suspended load in the unfiltered runoff sample was determined by transferring a known volume (usually between 50 to 100ml) of the sample into a tared glass crucible and then heating the samples to dryness over a water-bath. The glass crucible was then oven-dried at 105°C to constant weight and the difference in weights was calculated and expressed as weight of suspended load per unit volume of runoff, resulting in the suspended load of the runoff.

3.4.5 Total P in bedload

Total P in the bedload samples was determined using the method described by Tandon *et al.* (1968). It involved digesting 0.5g of finely ground (< 0.25mm) oven-dried bedload with 5ml of perchloric acid (70%). Once the digest was complete, it was brought to a volume of 50ml with distilled water and then filtered through a Whatman No. 50 filter paper. The diluted digest (5ml) was then reacted with 5ml of vanadate-molybdate reagent and the intensity of the resulting yellow coloured complex was read at a wavelength of 440nm.

3.5 Soil sampling and analysis

Soils were sampled at each of the 30 study sites to characterize their properties in terms of particle size distribution, pH, organic matter, exchangeable bases and P status. The methods used for determination of those properties are outlined in the subsections below.

3.5.1 Soil sampling and preparation

Soils were sampled at depths of 0 to 15cm and 15 to 30cm not within but just outside and adjacent to the runoff plots so as to avoid disrupting the hydrologic conditions of the soil inside the plots (National P Research Project, 2001). Six individual samples were taken at each depth in a zigzag pattern adjacent to the actual runoff plots as shown in Figure 3.6. The individual samples were spread on a plastic sheet, mixed and composited into one sample. The composite sample was brought to the laboratory where it was air-dried, passed through a 2mm sieve and stored for analysis.

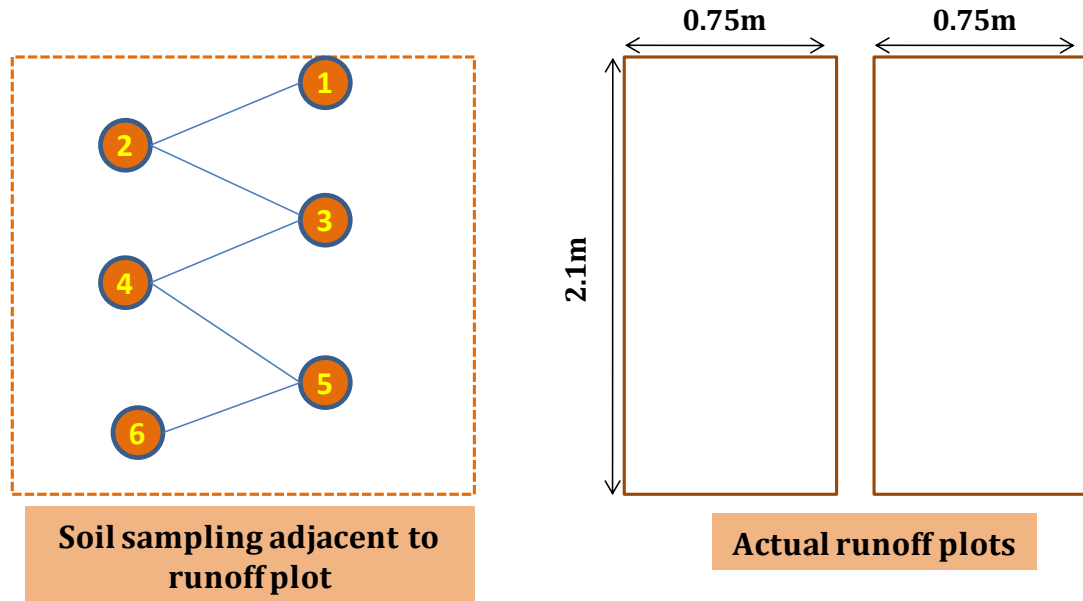


Figure 3.6: Soil sampling pattern adjacent to actual runoff plots.

3.5.2 Particle size distribution

The soil aggregates, for particle size analysis, were first dispersed by chemical and mechanical means as indicated by Gee and Bauder (1986) before the individual particles were separated according to size limits by sieving (sand fraction) and sedimentation (silt and clay fractions). Sedimentation relies on the relationship between settling velocity and particle diameter (Stoke's Law), the smaller particles taking longer to fall to the bottom of a suspension.

As organic matter is a powerful aggregating agent in the soil, the first step was to chemically destroy it by heating 10g of air-dried soil (< 2mm) with 2g sodium metabisulphite and 50ml 10% H_2O_2 until the contents of the beaker was reduced to half its original volume. The oxidation process using H_2O_2 was repeated twice to ensure that all organic matter had been destroyed. Once the oxidation process was complete, the soil was dispersed by shaking with sodium hexametaphosphate for 16 hours and maintained in the dispersed state until sedimentation was completed.

The silt content was determined in a constant temperature room at 20°C by pipetting a 20ml sample at a depth of 10cm from the surface of the suspension after the critical time required (approximately 4.5 minutes after shaking by repeated inversions for 30 seconds) for the silt fraction to settle. The clay content was similarly determined through pipetting after the critical time of 7 hours. The remaining silt and clay were thereafter siphoned off leaving the sand particles which were collected, dried and weighed.

3.5.3 pH

The pH was determined electrochemically in a suspension of 20g soil in 50ml distilled water on a pH meter. In effect the potential of the hydrogen ion electrode in the suspension was read against that of a calomel reference electrode as described by STASM (2003). The pH meter was first calibrated using buffer solutions of pH 4 and 7.

3.5.4 Organic matter

The organic carbon content of the soils was determined in the laboratory by the modified Walkley-Black procedure as described by Anderson and Ingram (1989). In this procedure, soil organic carbon was partially oxidized by treating 0.5g of finely ground (< 0.25mm) air-dried soil with 10ml of 5% potassium dichromate solution acidified with 20ml concentrated sulphuric acid and was allowed to stand for at least two hours to ensure complete oxidation. Barium chloride solution (50ml, 0.4%) was added and the mixture was centrifuged at 2500rpm for 10 minutes. The concentration of chromium ions in the reduced state in the supernatant obtained was measured colorimetrically at 600nm. Since this concentration is directly proportional to the amount of organic carbon in the sample, the latter was deduced from chromium ion concentration and converted to organic matter by multiplying by a factor of 1.724.

3.5.5 Exchangeable bases

The amount of exchangeable bases was determined in the laboratory by leaching the soils with unbuffered ammonium acetate for the determination of cation exchange capacity as described by Peech (1945). Initially, 5g of air-dried soil (< 2mm) was leached with 1M unbuffered ammonium acetate solution to displace the exchangeable Ca, Mg, K, and Na. The concentrations of the Ca and Mg in the leachate were determined by atomic absorption spectrophotometry while the concentration of K and Na was read on a flame photometer. The CEC of the soil was estimated by the summation of the exchangeable cations Ca, Mg, K and Na obtained.

3.5.6 Agronomic soil P status

As stated in Section 3.2.3, the P extractable in 0.1M H_2SO_4 (0.1M H_2SO_4 -P) is adopted in Mauritius to assess the availability of the soil P to the sugarcane crop. Its determination, as described by Cavalot *et al.* (1988) involved shaking 1g of the air-dried soil (< 2mm) with 50ml of 0.1M H_2SO_4 for one hour. The suspension was centrifuged at 2000rpm for 10 minutes before filtering through a Whatman No. 41 filter paper. The P concentration in the extract was determined by the method of Murphy and Riley (1962).

3.5.7 Environmental soil P status

McDowell and Sharpley (2001) showed that a laboratory based extraction of soil with 0.01M calcium chloride provides a reliable representation of the P that will be observed in runoff following rainfall events. The calcium chloride extractable-P ($\text{CaCl}_2\text{-P}$) in the soils was accordingly determined by shaking 10g of the air-dried soil (< 2mm) in 50ml of 0.01M CaCl_2 solution for 30 minutes as outlined by McDowell and Sharpley (2001). The suspension was then centrifuged for 10 minutes at 2000rpm and filtered through a Whatman No. 42 filter paper. The P concentration in the extract was determined by the method of Murphy and Riley (1962).

3.6 Source water analysis

As mentioned in Section 3.3.2, the water utilized to feed the rainfall simulator was analysed to determine the quality of the water used. Specifically the pH, electrical conductivity (EC), concentrations of soluble calcium (Ca), iron (Fe) and soluble silica (Si) were determined in samples of the source water. Total P as well as orthophosphate P were determined in the source water as per analytical procedures previously described in Section 3.4.2 and 3.4.3 respectively.

3.6.1 pH

The pH of the source water was determined electrochemically by measuring the potential of a hydrogen ion electrode in the source water sample against that of a calomel reference electrode (Rayment and Higginson, 1992). The pH of the source water was measured by immersing the pH electrode which was previously calibrated against buffer solutions of 4, 7 and 10 into the test sample. The pH values were recorded when equilibrium was reached and readings stabilised.

3.6.2 Electrical conductivity

As EC is a measure of the extent to which water conducts an electrical current, it depends upon the total concentration and relative proportions of the dissolved ionized substances present as well as upon the temperature at which the measurement is made (Rayment and Higginson, 1992). The electrical conductivity of the source waters were read on a calibrated conductivity meter using KCl reference solutions. The electrode was dipped into the water samples ensuring that the electrode was well covered and the EC values noted only when the readings had stabilized.

3.6.3 Soluble calcium and iron

The soluble Ca and Fe in the source water samples were determined directly on an atomic absorption spectrophotometer. However for the determination of Ca, lanthanum chloride (10%) was used to

reduce chemical interference caused by chemical components such as phosphate ions that could decrease the extent of atomization of the Ca (Harris, 1998).

3.6.4 Soluble silica

The soluble Si was determined as described by STASM (1990). Ammonium molybdate-sulphuric acid reagent (1ml) was added to a 5ml aliquot of the source water sample. After 10 minutes, citric acid (3ml, 10%) and ascorbic acid (3ml, 10%) were added and the volume brought to 50ml. The Si in the solution first formed with the acidified ammonium molybdate a yellow complex which was then reduced with ascorbic acid to form a blue complex. The intensity of this blue complex was read on a UV-Vis spectrophotometer at a wavelength of 650nm.

3.7 Data analysis and interpretation

The experimental data were processed using Microsoft Office Excel 2007 program in Windows 7. This program was used for calculation and graphical presentation of the data such as regression models and charts. The flow weighted concentration of P in the runoff waters during the 30 minutes simulated rainfall event was calculated as follows:

$$\text{Flow weighted concentration of P in runoff waters} = \frac{\sum_{i=1}^6 (\text{runoff } P_i \times V_i)}{\sum_{i=1}^6 (V_i)} \quad [\text{Equation 3.2}]$$

Where runoff P_i ($i=1, 2, 3, 4, 5, 6$) and V_i ($i=1, 2, 3, 4, 5, 6$) represent respectively the concentration of P and the volume of runoff water collected in the 0 to 5, 5 to 10, 10 to 15, 15 to 20, 20 to 25 and 25 to 30 minute intervals.

4 Soils in Mauritius and as a source of phosphorus transfer

4.1 Introduction

On account of its resilience to drought and to cyclone, sugarcane production is considered to be the only sustainable agricultural system in Mauritius, so much so that Mauritian agriculture has become synonymous with sugarcane production. Sugarcane cultivation began in 1639 and by the 1970s, out of the total land area of 186,500 hectares, sugarcane was cultivated on some 90,000 hectares. As a result of competition from other land uses such as housing, tourism and infrastructure construction coupled with land abandonment by an aging farming population, the area under sugarcane has gradually receded to about 53,500 hectares in 2013. Sugar production which varies from year to year depending on the vagaries of climate, declined from 700,000 tonnes in 1970s to only 404,700 tonnes in 2013.

Though climate and edaphic factors are conducive to good sugarcane growth in Mauritius, the success of sugarcane production on the island can to a large extent be attributed to the adoption of good crop management practices (Ng Kee Kwong and Deville, 1992) such as an optimum fertilization to meet sugarcane crop requirements. As stated by Parish and Feillafé (1958), out of the three major elements (i.e. N, P and K), P may have the least dramatic effect on the appearance of the sugarcane crop, but in terms of yield, P is just as vital as the other two major nutrients. When lacking, P can impose a limit to sugarcane yields which no amount of N or K can overcome. This being the case, it goes without saying that P fertilisation has been and will remain a regular feature in the management of the sugarcane crop in Mauritius.

This chapter attempts to provide an overview of the impact which the regular or repeated applications of the P to sugarcane have had on the P status of the soils in Mauritius. Information of this nature is essential for obtaining a better perspective on the soils under sugarcane acting as potential sources of P to freshwater pollution in Mauritius. As a pre-requisite to that attempt, a detailed description of the climate, geology and the nature of the soils in Mauritius is first presented to help understand the varying capacity of the soils to act as sources of P.

4.2 Geology, climate and soils

Except for the coral reefs, beaches or dune sands fringing the greater part of the coastline, Mauritius is entirely volcanic in origin and was created by two main phases of volcanic activity, namely an *Older Volcanic Series* dating back from 5 to 8 million years ago and a *Younger Volcanic Series* starting 3.5 million years ago (Arlidge and Wong You Cheong, 1975). The remnants of the basaltic lavas of the Older Volcanic Series now stand as a discontinuous ring of mountain ranges and as isolated peaks (Figure 4.1) rising from 600 m to 900 m above sea level. They enclose a *Central Tableland* created by

volcanic eruptions during the period of the Younger Volcanic Series, particularly from the lava flows dating from 0.7 to 0.2 million years ago (*Intermediate and Late Lavas*). Except in the southwest part of the island, the mountain ranges of the Older Volcanic Series are now surrounded by the flat to gently undulating *Coastal Plains* which resulted from outpourings of the *Intermediate and Late Lavas* with the latest lava flows occurring very recently in geological time (< 25,000 years) in the East at Plaine des Roches.

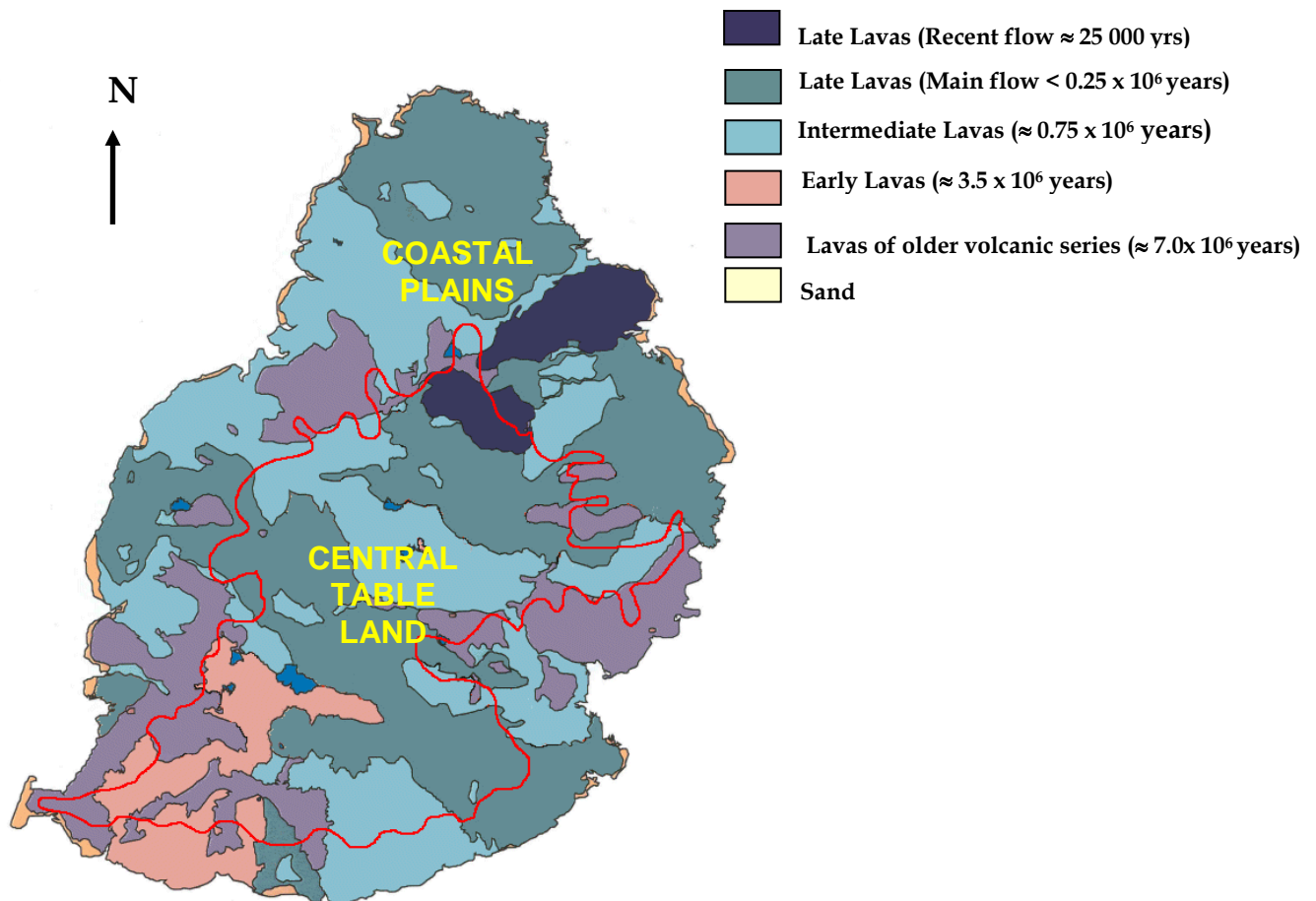


Figure 4.1: Geology of Mauritius (adopted from Arlidge and Wong You Cheong, 1975).

Variations in wind exposure, in altitude and in distance from the sea have created a succession of climates in Mauritius. On the basis of Thornthwaite classification, where emphasis is placed on the factor of evapotranspiration, a total of 24 different micro-climates have been differentiated in Mauritius (Halais and Davy, 1969). In general, however, the climate may be considered to comprise of two seasons, namely a subtropical winter from May to October during which the island experiences a cool and comparatively dry season, and a tropical summer from November to April which is warm and wet and often affected by tropical cyclones (Padya, 1989). July is the coolest month with mean temperatures of 21°C in the coastal regions and 16.5°C in the Central Tableland at an elevation of

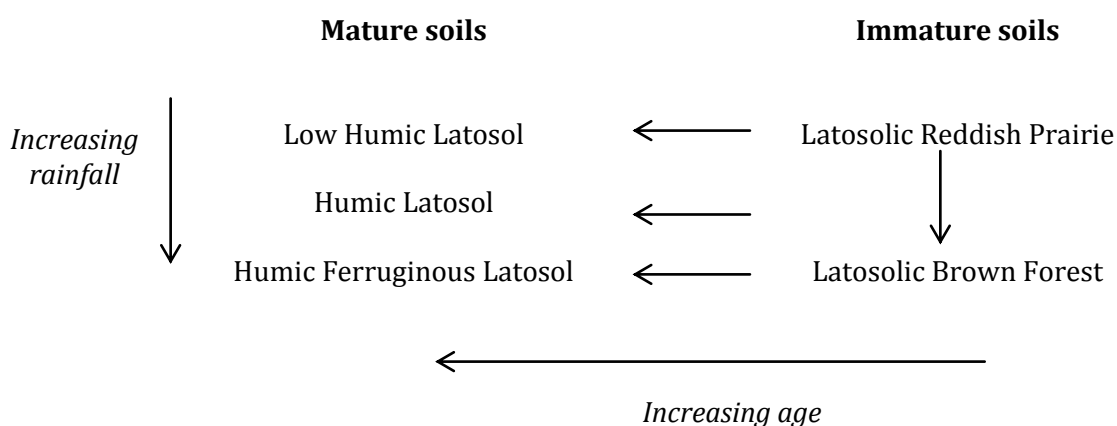
about 600 m. February is generally the warmest month with corresponding mean temperatures of 27.8°C and 22.3°C in the coastal region and central tableland respectively, the difference in the mean temperatures between the coolest month of July and the warmest month of February being only 5.5°C.

Mean annual rainfall changes abruptly from 800 mm on the west coast to over 4,000 mm in the Central Tableland, over a distance of only 20 km. From May to October rainfall expressed as a percentage of annual rainfall varies from 30 to 35% in locations exposed to the southeast trade winds and between 10 and 30% in the western leeward areas. The uneven distribution and frequency of rainfall coupled with high rates of evaporation (1,870 mm annually in the northern and western coastal areas and about 1,379 mm annually in the highest parts of the island) give rise to moisture deficits of moderate to severe degree and sustained sugarcane production is not possible without irrigation on the Coastal Plains.

It is against the climatic background described above that the soils of Mauritius have developed from basic volcanic rocks (mostly olivine basaltic lavas) of very different ages. Agriculturally important soils of Mauritius have been classified by Parish and Feillafé (1965) into two main groups, namely (a) the typical *mature ferratillitic soils* or *latosols* in which the decomposition of the parent basaltic lava rock has proceeded to such an extent that, except for large rounded boulders and stones, there are now no undecomposed minerals in the soil complex, and (b) the typical *immature latosolic soils*, the properties of which are affected by the presence of minerals still in the process of weathering and which, in the field, are characterized by the presence of more or less high proportions of angular stones and gravels of vesicular lava.

According to Parish and Feillafé (1965), Mauritius in fact provides a very fine example of the *zonality of soils*, that is, the progressive intensity of weathering and soil development with an increase in the intensity of the soil forming factors, notably rainfall. Using a genetic classification adopted for the soil survey of Hawaii and on the basis of differences in rainfall and age of parent material, Parish and Feillafé (1965) recognized 13 soil types which are shown in Figure 3.1, with subdivision at a lower category into families.

Parish and Feillafé (1965) moreover described the interrelationships among the five main soil types as follows:



In essence in Mauritius, sugarcane is cultivated mainly on the above five main soil types, which in brief may be described as follows:

The *Low Humic Latosols* have developed on the Intermediate Lavas (Figure 3.5) and they occur in zones receiving 800 mm to 2,750 mm rainfall per year. Because the clay fraction in the soil is composed of kaolinite cemented with oxides, they have, the texture of silty clays or silty clay loams in the field (Parish and Feillafé, 1965). A characteristic feature of this soil type, which covers 16.4% of the whole island, is the presence of manganese dioxide in its profile.

The *Humic Latosols* occur in the humid and superhumid zones with a mean annual rainfall ranging from 1,500 mm to 3,750 mm. This soil group, which covers 5.2% of the island, is in fact, a transitional group between the *Low Humic Latosol* and the *Humic Ferruginous Latosol* (Parish and Feillafé, 1965). The clay fraction of the Humic Latosols consists mainly of more or less equal proportions of kaolinite, goethite and gibbsite.

The *Humic Ferruginous Latosols* are the strongly weathered soils occurring in regions which receive between 2,500 mm to over 5,000 mm rainfall annually (Parish and Feillafé, 1965). This soil group makes up 11.4% of the whole island and is highly leached to the extent that its mineralogy is dominated by goethite and gibbsite.

The *Latosolic Reddish Prairie* and the *Latosolic Brown Forest* soils are the *immature intrazonal* soils developed from the late lavas under conditions where the effects of climate and vegetation had been masked by local factors of relief, drainage and age of the parent material (Parish and Feillafé, 1965).

The *Latosolic Reddish Prairie* soils, which cover 19.9% of the island, occur in the same rainfall zone as the Low Humic Latosol but can also be slightly acid to neutral in reaction. The *Latosolic Brown Forest*

soils cover 16.5% of the island and have on the other hand, been formed in the super-humid area where Humic Ferruginous Latosols are also encountered.

To provide a picture of the properties of the above five main soil types under which sugarcane is grown in Mauritius, 40 representative soil samples were chosen and were analysed for pH, organic matter, particle size texture and cation exchange capacity.

4.3 Research procedure

Soils were sampled at depths of 0 to 15 and 15 to 30 cm at each of the 30 study sites which cover the five main soil groups under which sugarcane is cultivated in Mauritius. The five major soil groups make up to 88% of cane lands in Mauritius and include the Low Humic Latosol, Humic Latosol, Humic Ferruginous Latosol, Latosolic Reddish Prairie and Latosolic Brown Forest.

The pH of the soils sampled was determined electrochemically using a pH meter (calibrated using buffer solutions of pH 4 and 7) by measuring the potential of the hydrogen electrode in a suspension of 20g soil in 50ml distilled water against that of a calomel reference electrode (STASM, 2003). The organic carbon content of the samples was determined in the laboratory through partial oxidation using the modified Walkley-Black procedure (Anderson and Ingra, 1989). The amount of exchangeable bases was determined in the laboratory by leaching the soils with 1M unbuffered ammonium acetate solution to displace the exchangeable Ca, Mg, Na and K (Peech, 1945). An atomic absorption spectrophotometer was then used to measure the concentration of Ca and Mg in the leachate, while the concentration of Na and K was determined by flame photometry. For particle size analysis, the soil aggregates were first dispersed by chemical and mechanical means as indicated by Gee and Bauder (1986) before the individual particles were separated according to size limits by sieving (sand fraction) and sedimentation (silt and clay fractions). Sedimentation relies on the relationship between settling velocity and particle diameter (Stoke's Law), the smaller particles taking longer to fall to the bottom of a suspension. A detailed description of the methodology used to measure each of the above-mentioned parameters as well as the soil sampling procedure is given in Chapter 3 (Section 3.5).

4.4 Properties of the main soils under sugarcane in Mauritius

4.4.1 Soil pH

The soils of Mauritius are generally acidic to near neutral with pH values ranging between 5.3 and 6.2 (Table 4.1 and Appendix 1). At these pH values, the P in the soils would exist mainly as iron (Fe) and

aluminium (Al) phosphates as explained in Section 2.2.3.3. As stated by Pierzynski *et al.* (2005), the speciation of P in soil solution is primarily a function of pH and in soils with pH values in the range of 4.1 to 7.3, P fixation would be dominated mostly by the presence of hydrous oxides of Fe and Al and to a lesser extent by calcium carbonates. An increase in soil pH through liming as usually practiced would decrease the activity of Fe and Al thereby resulting in a lower P adsorption/precipitation and a higher P concentration in solution (Havlin *et al.*, 2005a).

4.4.2 Soil texture

In general, the texture of the main soils in Mauritius ranges from clay loam to clayey with an average of not less than 32.6% clay (Table 4.1 and Appendix 1). The clay content in the Low Humic Latosols is on average as high as 54.4%. As expected the amount of clay decreased from the Low Humic Latosol to the Humic Latosol and to the Humic Ferruginous Latosol (38.6% clay) for the *Latosol group* and from Latosolic Reddish Prairie to the Latosolic Brown Forest soils for the *Latosolic group* indicating that as expected with leaching the clay content tends to diminish with increasing rainfall. The latosols generally have higher clay contents than the Latosolic soils.

As soils with higher clay contents are generally known to fix more P than soils with low quantities of clay (Havlin *et al.*, 2005a), it is expected that soils such as the Low Humic Latosol located in the low rainfall zone will be fixing more P than the soils in the wet regions of Mauritius such as the Humic Ferruginous Latosol. From an agronomic point of view, for such soils in the low rainfall zone, it might therefore be necessary to add larger quantities of P fertilisers in order to maintain an optimum level of soil solution P for sugarcane growth. From the environmental viewpoint however, the likelihood of P transfer from the Low Humic Latosols to freshwater courses would be minimal given their generally flat topography and the lower rainfall they receive.

4.4.3 Soil cation exchange capacity (CEC)

The mean CEC was lowest (4.8 cmol⁺ kg⁻¹) for the Humic Ferruginous Latosols and was highest (18.2 cmol⁺ kg⁻¹) in the Latosolic Reddish Prairie soils (Table 4.1 and Appendix 1). Differences in the CEC and concentration of the exchangeable bases among the five main soil groups in Mauritius are related to rainfall which affects the intensity of leaching in the soils (Cavalot *et al.*, 1988). Thus, it can be noted that the Low Humic Latosol and Latosolic Reddish Prairie which are located in the low rainfall regions (800 to 2,600 mm) have the highest CEC among the Latosols and between the two Latosolic soils, respectively. Correspondingly, the highest concentrations of exchangeable bases (K, Na, Ca, Mg) were encountered in those two soils. The data obtained further show that Ca²⁺ is the predominant exchangeable base present, indicating that in Low Humic Latosols and Latosolic Reddish Prairie soils there would be more P present as precipitated Ca-P than in the other three soil groups. Moreover as

the presence of divalent cations on the cation exchange sites enhances P adsorption (Havlin *et al.*, 2005a), the P adsorbed induced by the divalent cations would tend to be lowest in soils with the least amount of exchangeable Ca^{2+} , namely in the Humic Ferruginous Latosol and Latosolic Brown Forest soils of the high rainfall zones. However the latter two soils have a greater abundance of oxides to offset the lack of divalent cations in promoting P retention.

4.4.4 Soil organic matter

The organic matter content of the main soils varies between 3.7 and 6.5% with the Low Humic Latosols, as the name suggests having the lowest soil organic matter (Table 4.1 and Appendix 1). The Latosolic Reddish Prairie has the highest organic matter content. As described in Section 2.2.3.4, the presence of organic compounds increases not only P availability to plants but also its mobility. Moreover, organic compounds increase P availability and mobility by firstly anion replacement of H_2PO_4^- on adsorption sites, thus releasing the P into solution, and secondly by coating the Fe/Al oxides with a humus layer to form a protective cover that reduces P adsorption (Havlin *et al.*, 2005a). With the generally higher organic matter content in the latosolic soils, a greater mobility of P could be expected on those soils, especially in the Latosolic Reddish Prairie than in the latosols.

Characterisation of the chemical properties of the five main soils under sugarcane cultivation in Mauritius thus shows that the soils, being generally acidic in nature with a high clay content composed mainly of the oxides of Fe and Al, have a high capacity to retain P. This being the case, it was therefore not surprising that as inferred by Ng Kee Kwong *et al.* (2002), the movement of P from the soils in Mauritius, when it occurs would be mostly attached to the sediment carried by the runoff during erosion.

The presence of a generally high organic matter status would further inhibit the transfer of the applied P though it should also be acknowledged that the retentive action of the organic matter would be offset to some extent by the ability of the organic compounds to block the retention sites on the oxide surfaces and to keep the P in solution in the form of organo-P complexes. The extent to which the P is retained by the soils after being intensively and continually applied to sugarcane over more than one century is best reflected by the current soil P status. Accordingly the latter is described after first highlighting in the next section the different sources from which the sugarcane received the P in Mauritius.

Table 4.1: Pertinent characteristics (mean \pm SE) of the five main soil groups under sugarcane in Mauritius.

				Particle-size analysis ³			Exchangeable bases ⁴				Cation
Soil group	Soil depth (cm)	pH ¹ (H ₂ O)	Organic matter ² %	Clay	Silt	Sand	Ca	Mg	K	Na	exchange capacity ⁵
				%			cmol ⁺ kg ⁻¹				
Low Humic Latosol <i>(Humic Nitosol)</i> *	0-15	6.2 ±0.3	54.0 ±5.5	3.9 ±0.2	23.4 ±2.0	22.6 ±3.7	7.40 ±1.87	2.23 ±0.83	1.72 ±1.15	0.70 ±0.34	12.0 ±3.6
	15-30	6.0 ±0.3	58.1 ±5.0	3.8 ±0.2	21.8 ±1.6	20.1 ±3.4	7.38 ±1.71	2.24 ±0.94	1.30 ±0.77	0.48 ±0.15	11.4 ±3.4
Humic Latosol <i>(Humic Nitosol)</i> *	0-15	5.5 ±0.2	40.5 ±3.5	4.5 ±0.1	31.0 ±2.4	28.6 ±2.4	3.62 ±0.84	0.91 ±0.13	0.31 ±0.05	0.28 ±0.03	5.1 ±1.0
	15-30	5.5 ±0.2	42.4 ±4.3	4.3 ±0.2	29.0 ±4.3	28.6 ±1.6	3.38 ±0.86	0.78 ±0.06	0.23 ±0.05	0.23 ±0.03	4.6 ±0.9
Humic Ferruginous Latosol <i>(Humic Acrisol)</i> *	0-15	5.9 ±0.2	34.0 ±1.9	4.4 ±0.5	30.2 ±2.1	35.8 ±2.4	4.12 ±1.76	0.83 ±0.12	0.37 ±0.09	0.29 ±0.02	4.8 ±1.9
	15-30	5.6 ±0.2	31.4 ±3.4	3.7 ±0.4	32.5 ±2.0	36.1 ±2.3	3.10 ±1.73	0.59 ±0.13	0.18 ±0.07	0.20 ±0.03	4.1 ±1.9
Latosolic Reddish Prairie <i>(Eutric Cambisol)</i> *	0-15	6.3 ±0.2	47.0 ±3.4	4.0 ±0.3	22.5 ±1.1	30.6 ±2.9	10.68 ±0.97	4.92 ±0.32	1.95 ±0.46	0.67 ±0.04	18.2 ±1.4
	15-30	6.3 ±0.1	48.5 ±2.0	3.9 ±0.3	24.4 ±1.2	27.1 ±2.5	10.76 ±0.74	5.27 ±0.57	1.97 ±0.47	0.63 ±0.08	18.6 ±1.1
Latosolic Brown Forest <i>(Dystric Cambisol)</i> *	0-15	5.5 ±0.1	30.2 ±6.8	6.5 ±0.7	34.1 ±2.4	35.7 ±5.0	3.63 ±1.10	1.91 ±0.50	0.52 ±0.10	0.33 ±0.03	6.4 ±1.1
	15-30	5.3 ±0.1	32.7 ±1.2	6.3 ±0.1	30.5 ±0.8	36.8 ±2.0	2.51 ±0.50	1.13 ±0.08	1.80 ±1.56	0.25 ±0.04	5.7 ±1.2

* Soil group as per FAO classification (Arlidge and Wong You Cheong, 1975).

¹ 2:5 Soil water suspension using a calomel pH electrode (STASM, 2003).

² As outlined by Anderson and Ingram (1989).

³ By sieving and sedimentation following dispersion of aggregates according to Gee and Bauder (1986).

⁴ As outlined by Peech (1945).

4.5 Sources of P to sugarcane

The sugarcane crop is grown by planting cut pieces of cane (cane setts) and this produces the *plant cane* 15 to 18 months later. After harvesting the plant cane during the period June to November, the regrowth gives rise to the first *ratoon crop* which is in turn similarly harvested 12 months later. In Mauritius, generally a field is only replanted after six or seven ratoons, that is when the sugarcane yield has declined to such an extent that replanting the field is economically warranted. Concurrently, studies on P fertilization have shown that when sufficient amount of P had been applied to sugarcane at planting, supplementary P fertilization in the ratoons would be superfluous (Cavalot *et al.*, 1988). Accordingly, the current practice in Mauritius is to apply P to the sugarcane only at planting. This section provides an overview of the different sources of P used to meet the P requirements of sugarcane and by extension it indicates the potential sources of P which could be carried to waterbodies during runoff events.

4.5.1 Phosphorus mineral fertiliser usage

Phosphorus fertiliser usage in Mauritius, as can be anticipated, has been intimately linked to the intensive cultivation of sugarcane. This implies not only high inputs of P fertilisers, but inevitably of N and K fertilisers as well. In fact as the aim is to achieve and to sustain a high level of sugarcane production in Mauritius through elimination of all limiting factors to growth, including nutrients, P fertiliser usage in sugarcane cannot be dissociated from that of N and K. The different types of mineral P fertilisers which have been utilized by the Mauritian sugarcane industry are given in Table 4.2.

During the first half of the 20th century, the sugar planting community was generally not concerned about P fertilisation and its importance in enhancing crop yields. Among the possible causes for neglecting P fertilisation during that period, as mentioned by Parish and Feillafé (1959), was the fact that with the introduction of higher yielding cane varieties, sugar production had increased to mask the negative effect of P deficiency on yields. Accordingly the amount of P fertilisers consumed was then fairly low with an annual of 790 tonnes of P₂O₅.

Table 4.2: Phosphate fertilisers utilised by the sugarcane industry in Mauritius.

Type of fertiliser	% N	% P ₂ O ₅	%K ₂ O	Formula of main compound
<i>Straight fertilisers</i>				
Rock phosphate	0	30	0	Ca ₃ (PO ₄) ₂
Guano phosphate	0	23	0	Ca ₃ (PO ₄) ₂
Single superphosphate	0	19	0	Ca(H ₂ PO ₄) ₂
Triple superphosphate	0	46	0	Ca(H ₂ PO ₄) ₂
<i>Complex fertilisers</i>				
Mono-ammonium phosphate	12	52	0	NH ₄ H ₂ PO ₄
Di-ammonium phosphate	18	46	0	(NH ₄) ₂ H ₂ PO ₄
13-30-18	13	30	18	-
16-22-22	16	22	22	-
17-8-25	17	8	25	-
14-20-20	14	20	20	-
17-8-20	17	8	20	-

Usage of P fertilisers experienced a boom in the 1950s when large areas of ratoon and even virgin cane showed the typical discoloration of P deficiency and yearly foliar diagnosis data were showing a worsening P status of the Mauritian cane lands (Parish, 1964). It was established that about 30% of the sugarcane lands in Mauritius were deficient in P and P fertilisation then became recognised as being essential for maintaining a viable sugar industry. As a consequence, over the period 1950 to 1970, P fertiliser imports rose from 570 tonnes to 5,675 tonnes indicating clearly the big effort made to replenish P reserves of sugarcane soils. At present with land under sugarcane having dwindled from about 90,000 hectares in the 1970s to only 53,500 hectares in 2013, some 3,350 tonnes of P₂O₅ are still being used in sugarcane plantations.

During the first 50 years of the 20th century, insoluble mineral P in the form of guano phosphate was the predominant if not the only form of mineral P fertiliser (tri-calcium phosphate) used in Mauritius, mainly because it was a cheap source of P that was obtained from the coral islands and atolls in the South West Indian Ocean, several of which belong to Mauritius (Parish *et al.*, 1956). It was then a common practice to mix guano phosphates with farmyard manure before application at planting. Subsequent studies to test the efficacy of various forms of P fertilisers had showed that, the water-soluble forms of P were superior to such insoluble phosphate as guano or rock phosphate, especially in soils with pH above 6.0 (Anon, 1994), the utilisation of water soluble

forms of P fertilisers which at that time were available as superphosphates, become favoured. As a result, the amount of superphosphates used in sugarcane production rose in the 1960s and that rise continued until the late 1980s, as shown in Figure 4.2.

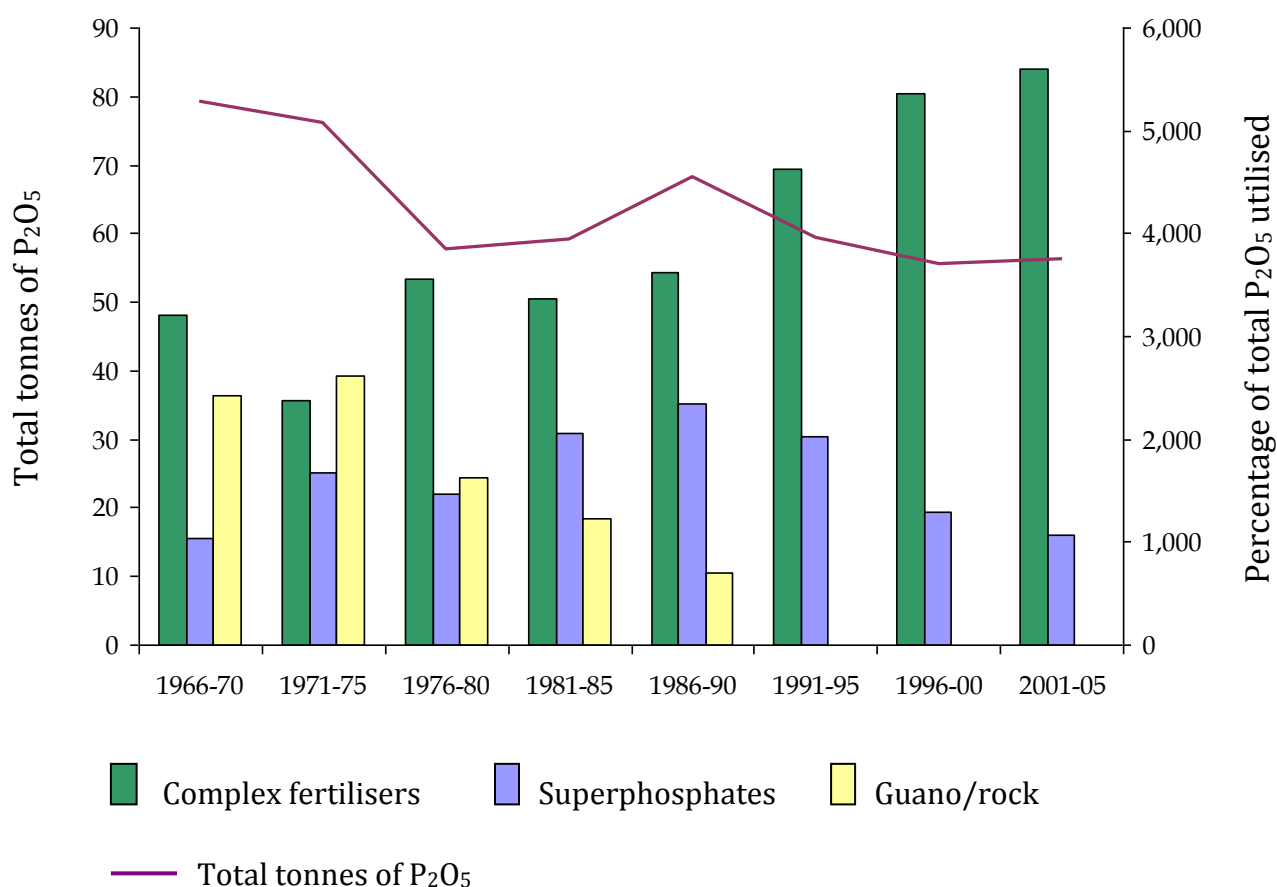


Figure 4.2: Types of mineral P fertilisers utilised from 1966 to 2005 (Mardamootoo *et al.*, 2010).

During that time span, usage of guano/rock phosphates evidently declined. Indeed while in 1966, guano/rock phosphate accounted for 40% of the P₂O₅ utilised by the sugar industry it had declined to only 9% in the late 1980s. Eventually in the early 1990s insoluble phosphates were no longer utilised, because the local market for the guano/rock phosphates had shrunk to the extent that it became too small to be of interest to the importers of that commodity.

The use of complex fertilisers with mono-ammonium phosphate or di-ammonium phosphate as the source of P in their turn, rose steeply as from 1986 before plateauing in the mid 1990s (Figure 4.2). Concomitantly, the amount of superphosphates, which in the mid 1980s supplied 35% of the P to sugarcane, gradually decreased as their utilization became confined to correcting P deficiencies when diagnosed by foliar diagnosis in ratoon canes.

4.5.2 Organic sources of P

While the previous section highlights the utilisation of mineral phosphate fertilisers, this section discusses the relevance of the different organic 'waste' materials as P sources to sugarcane in Mauritius. As reviewed by Deville (1999), studies carried out in Mauritius had shown that organic P sources gave yields of sugarcane similar to those obtained with inorganic fertilisers. The main organic sources of nutrients that have been used in sugarcane production in Mauritius were poultry litter as a source of N, scums (filter muds) as sources of P and molasses as sources of K (Deville, 1999).

Scums are the wastes resulting from the clarification of sugarcane juice with lime, while molasses are obtained during the evaporation and crystallisation processes to extract the sugar crystals. In fact, organic fertilisers in the form of farmyard manure and wastes from the sugar manufacturing processes such as scums and molasses had been disposed of in sugarcane fields since the early years of the 20th century. Though the agricultural re-use of these 'waste' materials produce economic benefits as they contain essential plant nutrients (Ng Kee Kwong *et al.*, 2003), their application required considerable transport to the fields because large amounts of the material had to be applied to meet crop requirements. Consequently, there was then a general tendency to apply these 'wastes' only to fields close to sugar factories whilst outlying fields did not receive those supplements (Parish, 1964).

In the 1960s, around 40% of the total yearly production of some 111,000 tonnes molasses was used as fertiliser at an average rate of 3.5 t ha⁻¹ applied in the furrows before planting. The quantity of P added to soils through the molasses was however insignificant in view of their inherently low P content (0.19% P₂O₅). With the diversification of the sugarcane industry, most of the 140,000 tonnes molasses now produced are fermented to ethanol. The resulting production of 30,000 tonnes ethanol generates as much as 400,000 tonnes of stillage waste (vinasse) which are disposed of in sugarcane fields. Recent studies have shown that application rates of 100 m³ ha⁻¹ of vinasse, on account of its low P content (0.10% P₂O₅) can be disposed of in fields without any negative impact on P content of water draining from the soil at one meter depth (Soobadar, 2009). To offset the high cost of transporting raw vinasse from the distillery to distant farms, the vinasse is evaporated to around 45 to 50% total solids to obtain a product commonly referred to as concentrated molasses stillage (CMS). As the CMS or the vinasse is not a significant source of P, fertiliser companies in Mauritius are now supplying vinasse and/or CMS based liquid fertilisers where the vinasse or CMS is enriched with straight fertilisers such as urea and mono-ammonium phosphate to ensure that the total nutrient composition of the liquid fertiliser meets the requirements of the sugarcane crop.

Bagasse ash is another waste product of the sugarcane industry which is disposed of in sugarcane fields and is thus a potential source of P for crop uptake or for mobilization into freshwater sources in Mauritius. The ash is obtained following the cogeneration of electricity from bagasse, which is the fibrous material remaining after sugarcane juice has been extracted from cane stalks. Currently about 18% of the national electricity supply in Mauritius is produced by cogeneration with bagasse. The 1.5 million tonnes of bagasse after cogeneration produced about 20,000 tonnes ash which may be disposed of in sugarcane fields. Indeed studies by Soobadar (2009) have shown that up to 50 t ha⁻¹ ash may be applied without any significant negative impact on sugarcane yield. Apart from an average 4.3% K₂O, bagasse ash also contains some 2.7% P₂O₅ (Soobadar, 2009).

Sources of P which have and may still be used for fertilizing sugarcane in Mauritius and thus constitute potential sources of P for freshwater pollution also include poultry litter and sewage sludge. In this context, studies initiated in the 1980s to determine the value of poultry litter as a nutrient source for sugarcane had showed that the application of poultry litter did not significantly give better yields or improve the NPK nutritional status of the sugarcane when compared with the corresponding mineral fertilisers (Anon, 2006). Nevertheless, as poultry litter is a waste product supplying a cheap source of nutrients, its use as a fertiliser to meet the nutrient requirements of the sugarcane, has been recommended to the planting community, generally at rates based on N needs of the crops.

The feasibility of using sewage sludge as a source of nutrients for sugarcane has been evaluated in Mauritius. In view of the expected production of 120,000 tonnes of sewage sludge by 2015 (Mohee and Mudhoo, 2010), studies initiated in the 1990s have shown that the disposal of sewage sludge (which in addition to its 4.8% N, also contains 1.9% P₂O₅) in sugarcane fields can only be a rational and cost-effective management option if the cost of transporting the sewage sludge to sugarcane fields are not prohibitive. The studies have shown that as much as 40 t ha⁻¹ of the sludge would be needed to substitute for mineral fertilisers.

The average nutrient composition in terms of N, P₂O₅ and K₂O of the different organic materials discussed above is given in Table 4.3. It is important to note that such organic 'waste' materials are quite heterogeneous in nature, and the nutrient composition may vary considerably. For example the P content of vinasse may vary from 0.02 to 0.45 g P₂O₅ L⁻¹ (Soobadar, 2009).

Table 4.3: Average nutrient composition (expressed in percent weight by weight on a dry weight basis) of organic materials available for application to sugarcane fields in Mauritius.

Organic material	% N	% P ₂ O ₅	%K ₂ O	Reference/source data
<i>'Wastes' from sugarcane industry</i>				
Bagasse ash	0.05	2.80	4.29	Soobadar (2009)
Concentrated molasses stillage	0.58	0.16	4.51	MSIRI ²
Molasses	0.92	0.19	5.29	MSIRI ²
Poultry manure	3.04	2.04	1.72	MSIRI ²
Scums (filter muds)	1.00	1.64	0.30	MSIRI ²
Vinasse ¹	0.30	0.10	1.60	Soobadar (2009)
<i>Other materials</i>				
Composts of poultry manure and cane trash	1.07	2.09	3.40	MSIRI ²
Compost of vinasse, scums, bagasse ash, poultry litter, green wastes	1.57	2.06	2.36	MSIRI ²
Composts of poultry manure and bagasse	0.96	1.03	2.37	MSIRI ²
Sewage sludge	4.82	1.91	0.10	Toory <i>et al.</i> (2003)

¹ Vinasse being a liquid, its nutrient composition is expressed in percent weight by volume.

² Data obtained from the analysis of samples at the Mauritius Sugar Industry Research Institute (MSIRI).

Of all the organic waste materials enumerated above, scums (filter muds) are the only one which has been consistently applied to sugarcane in Mauritius. They have a relatively high percentage of P₂O₅ (1.6%) and it is a common practice to return most of the scums produced in the sugar factories to sugarcane fields. The evolution of the annual production of scums in Mauritius is shown in Figure 4.3 to provide an illustration of how much P₂O₅ has been returned to sugarcane fields from this material.

From 2005 to 2011, the sugarcane fields received an average of 1,110 tonnes of P₂O₅ (assuming that the scums contain 0.55% P₂O₅ on a fresh weight basis) from some 200,000 tonnes scums generated by the sugarcane industry in Mauritius. It is a recommended practice to apply scums at a rate of about 20 t ha⁻¹ in the furrows at planting to substitute for mineral P fertilisers. A review by Cheesman (2004) showed that scums were in fact superior to other organic materials (farmyard and poultry manure) in sustaining crop yield and soil properties. In addition, they are a perfect illustration of the fact that the extensive utilisation of wastes from the processing of sugarcane crops is a notable, positive environmental aspect of the industry.

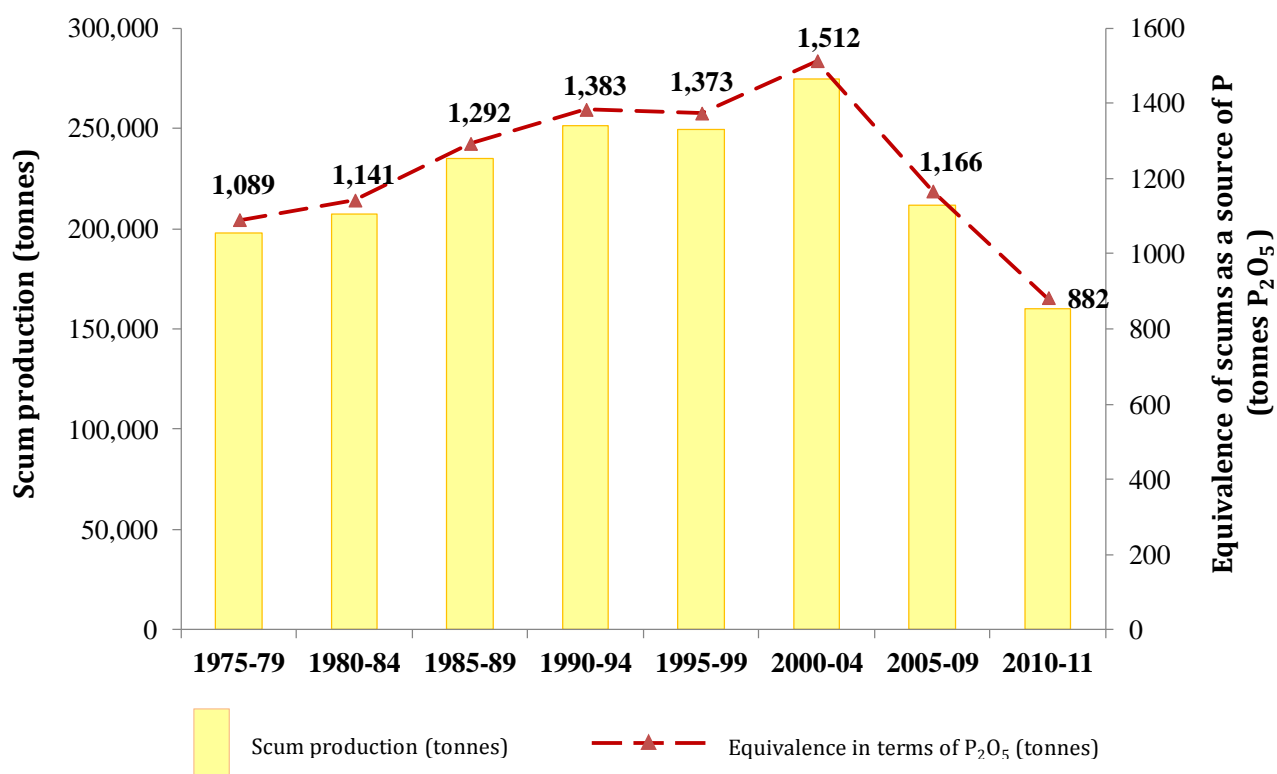


Figure 4.3: Five yearly averages of scums (filter muds) produced by sugar factories in Mauritius and their potential equivalence as a source of P to sugarcane. [Source data: Digest of Agricultural Statistics, Central Statistics Office, Ministry of Finance and Economic Development, Mauritius, 1975 to 2009].

4.5.3 Phosphorus from sugarcane residues

Crop residue management has always been a primary concern in the sugarcane production system in Mauritius. Indeed on average, upon harvest as much as 12 t ha⁻¹ of trash (dried leaves) and green tops are left on the soil surface to act as a mulch, thus protecting the soil against the impact of rainfall and inhibits the propagation and development of the weeds. Other benefits of the mulch include soil moisture retention, particularly during the dry season, improvement of soil organic matter status and not least the supply of nutrients to the subsequent ratoon crop. As highlighted by Bakker (1999), on average 65 kg of P_2O_5 could be returned to the land by the crop residues for every hectare of sugarcane field through cane tops and trash.

Recent studies by Udeigwe *et al.* (2010) to evaluate the impacts of three sugarcane residue management techniques, namely postharvest burning of residue, shredding of residue and full post-harvest retention of residue on the quality of surface runoff waters showed that shredding and full harvest retention of residues yielded lower average total P concentration in the rainfall runoffs than where the residue was removed by burning. In the past, pre-harvest burning was also practised to eliminate leafy material, thus facilitating manual harvest of the cane stalks in

Mauritius but due to public concern about environmental and possible health problems that may arise from the particulate matter emission, burning is gradually being phased out, especially with the increased adoption of mechanized harvesting, green cane trash blanketing is fast becoming the norm in Mauritius.

While studies by Udeigwe *et al.* (2010) showed that keeping the crop residue as a blanket on the soil surface may reduce the movement of P, research carried out by Gilley *et al.* (2009) showed that the residue adsorption and desorption characteristics of the crop residues can differ significantly, particularly if more than one nutrient constituent is present in solution. As a result, the concentration of soluble nutrients including P, in the runoff following a rainfall event is dependent on the sorption characteristics of the crop residues. Thus, Cermak *et al.* (2004) showed that while certain plant residue (e.g. wheat residue) may sorb P especially as a result of a greater residue solution to contact time, other crop residues such as those of corn and soybean contribute to runoff nutrient load through leaching. In the same context, studies carried out by Boopathy *et al.* (2001) have shown that the natural bacteria and fungi present in the soil can cause extensive decomposition of post-harvest sugarcane residue, thus releasing nutrients such as P for mobilisation or for plant uptake.

A review of literature on the crop residues left on the soil surface after harvest thus indicates that the sugarcane mulch may represent both a benefit for reducing the risks of surface runoff and soil erosion as well as being a source of P for mobilisation through microbial decomposition or through adsorption/desorption. Additional research on sugarcane residue decomposition in soils and the amount of P release from the residue is therefore needed in Mauritius if a clearer role of the sugarcane mulch on P movement in soils is to be obtained. This additional research is particularly relevant for a sustainable sugarcane production system in Mauritius since the crop residues together with the mineral P fertilisers and the scums (filter muds) constitute the potential P sources which have consistently supplied P to sugarcane and which have enhanced the P status of the soils as presented in the section that follows.

4.6 Phosphorus status of sugarcane soils

The method adopted in Mauritius to assess soil P available to sugarcane utilizes as extractant, sulphuric acid (H_2SO_4) at a concentration of 0.1M (see Section 3.5.6). This method of Cavalot *et al.* (1988) had been validated by field experimental data which also provided the basis for formulating fertiliser recommendations at planting (Table 4.4). Fertiliser P is not recommended to sugarcane in soils with more than 80 mg P kg^{-1} extracted by the 0.1M H_2SO_4 and the field

experiments have shown that above the latter value, the residual effect of past P fertilization would be expected to last and thus maintain the maximum yield level during a sugarcane growth cycle of 6 to 7 years in Mauritius (Cavalot *et al.*, 1988).

Table 4.4: Phosphorus fertiliser recommendations to sugarcane in Mauritius based on soil P test values (Cavalot *et al.*, 1988).

Soil test value 0.1M H₂SO₄-P (mg kg⁻¹)	kg P₂O₅ ha⁻¹ to apply (to raise soil P to 80 mg kg⁻¹)
30	600
35	525
40	475
45	425
50	375
55	325
60	275
65	200
70	125
75	50
80	0

Using this recommendation to establish and then apply an agronomic threshold range of 80 to 100 mg kg⁻¹ to 2,700 soil tests data, it was found that approximately 32,100 hectares (52%) of sugarcane lands in Mauritius contained more P (i.e. P ≥ 100 ppm) than what is actually needed by the sugarcane crop (Figure 4.4). The majority (78%) of the sugarcane fields with an agronomically excessive P status were located on the latosolic soils.

But how many of the soils with an excess of P for sugarcane growth actually represent a threat to the quality of freshwaters in Mauritius remains unknown since no environmental soil P test is to-date available for their identification. As reviewed in Section 2.5.1, in shifting from the agronomic to the environmental concern it is reasoned that soil P in excess of crop requirements is vulnerable to removal by surface runoff or leaching. Soil P tests have consequently been used to indicate when P enrichment of runoff will become unacceptable (Sharpley *et al.*, 2001a). The most common approach has been to extend the interpretation of agronomic soil P tests. However it cannot be assumed that the calibration of the soil P test for crop response, as described above for sugarcane in Mauritius using 0.1M H₂SO₄ extractant, is equally valid for indicating P movement potential. Consequently if the agronomic soil P test adopted for sugarcane in Mauritius is to be effectively

extended into an environmental soil P test, further studies aimed at calibrating the 0.1M H₂SO₄ extractable P values to indicate the potential of P movement in the soils are needed. Those studies have been performed and the results are described in the chapter that follows.

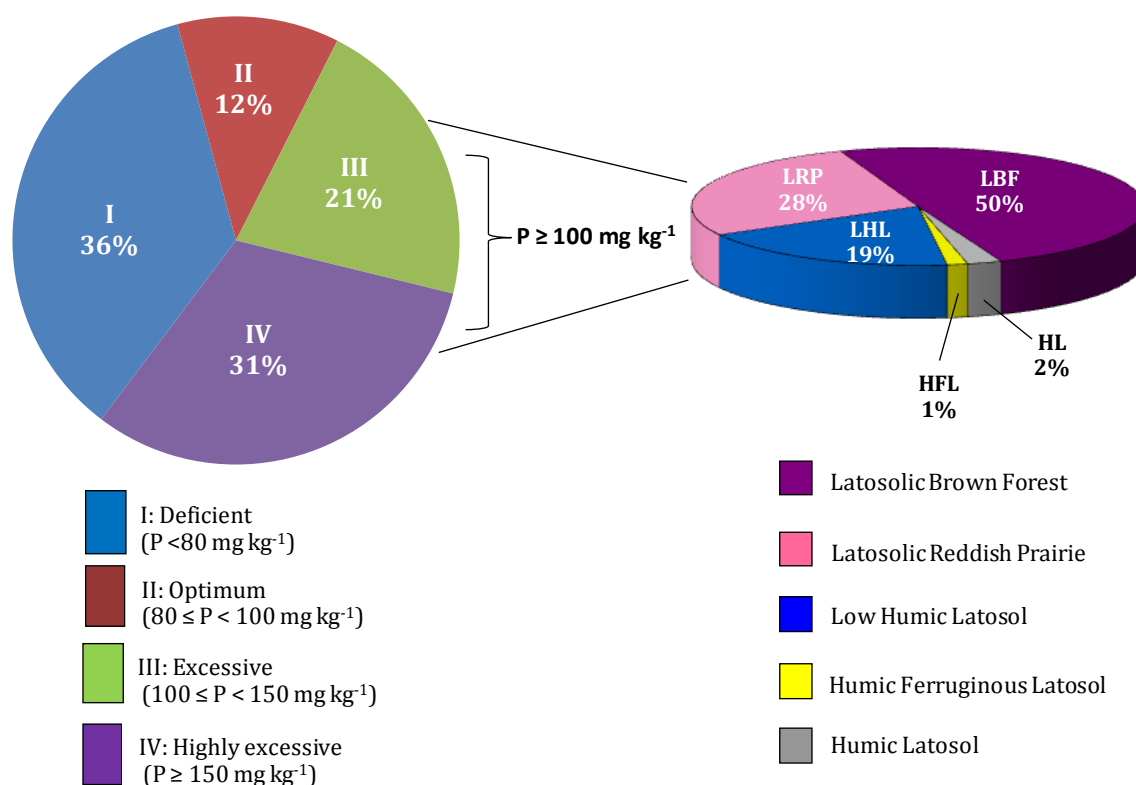


Figure 4.4: Agronomic P status of sugarcane soils and the distribution of those with an agronomically excessive P level ($P \geq 100 \text{ mg kg}^{-1}$) (Mardamootoo, 2009).

4.7 Conclusion

From the description of the soils under which sugarcane is grown in Mauritius and of the sources of P which have been applied to satisfy the requirements of the crop for optimum yield, it may be inferred that the soils have properties which, when considered overall, favour the retention of applied P. The low cation exchange capacity of the soils in Mauritius is compensated by the high clay content and by the predominant presence of the oxides of Al and Fe in the mineralogy of the clays. Therefore, the P in the soils of Mauritius would preferentially be mobilised in sediment forms during runoff events.

Moreover though many different sources of P have been applied to soils to meet the needs of the sugarcane crop, only two of them may be considered to have been significantly applied to cane fields and have thus contributed to improving the P status of the soils. In this context, the soils of

Mauritius have received P intensively and regularly in the form of mineral fertilisers and scums (filter muds). The impact of these P sources on the P status of the soils has been enormous with as much as 52% of the soils of Mauritius having an agronomically excessive P content.

However, as it cannot be inferred that the soils with agronomically excessive P status represent a threat to freshwater resources in Mauritius, extension of the current agronomic soil P test using 0.1M H₂SO₄ extractant into an environmental soil test still requires further studies that would accurately and reliably relate P movement in the soils to the levels of P extractable by 0.1M H₂SO₄. Many have claimed that neither an agronomic or environmental soil P test alone, are adequate indicators of P loss in runoff or its movement to surface waters and an estimate of the potential for transport is required (Sharpley *et al.*, 2012). These studies have been undertaken and are reported in the next chapter.

5 Assessing environmental phosphorus status of soils in Mauritius following long-term P fertilization of sugarcane

5.1 Introduction

The importance of optimising P nutrition has long been recognised in sugarcane cultivation in Mauritius. As reviewed in Chapter 4, different sources of P have been used to meet the needs of the sugarcane crop for that particular nutrient. One of these sources is mineral P fertiliser which the agricultural community has for more than 60 years, diligently applied on average 55 kg P₂O₅ ha⁻¹ every time the sugarcane is replanted. Through adsorption and precipitation processes (Prasad and Power, 1997), the P added is rapidly removed from the solution phase and is equally partitioned in the soils among the inorganic fractions of aluminium, iron and calcium phosphates (Anon, 1983). The latter fractions create a high soil P status, which can increase cane production by about 170 t ha⁻¹ over a whole crop cycle of seven to eight years (Cavalot *et al.*, 1986).

The above agronomic benefits of P fertilization on the productivity of cane fields are known in Mauritius as well as in most sugarcane producing countries in the world. The same cannot be said however on the significance of the high P status of the soils on surface water quality. As described in the literature review (Section 2.4), P loss from agricultural land is an increasing environmental concern due to the accelerated eutrophication of surface waters which leads to problems with their use for fisheries, recreation and drinking (Sharpley and Withers, 1994; Hodgkinson and Withers, 2007). Moreover, P (and not nitrogen) is the most important plant nutrient influencing eutrophication in surface waters most often at a P concentration that is tenfold lower than that required for plant growth (Daniel *et al.*, 1998). The challenge for soil scientists and the agricultural community at large is to be able to predict and prevent potential movement of P, particularly at very low concentrations (< 0.03 mg L⁻¹).

As reviewed by Moody (2011), the effort of the agricultural community led by soil scientists, have focused on the establishment of simple soil tests to identify soils which most likely represent a risk to water quality. McDowell and Sharpley (2001) found that extraction of surface soil with 0.01M CaCl₂ can be used to estimate P losses through overland flow or by leaching. Maguire and Sims (2002) have also shown that 0.01M CaCl₂ soil extraction is suitable for estimating dissolved reactive P concentrations in leachates from different soils varying in physical and chemical properties. Recent studies carried out by Schindler *et al.* (2009) have confirmed through simulated rainfall studies that P extracted in 0.01M CaCl₂ is indeed an effective estimator of total dissolved P in surface runoff.

As highlighted by Moody (2011), it is desirable from a practical viewpoint that any environmental risk assessment of soil P be based on the same soil test that is routinely utilized to formulate P fertiliser recommendations for crop nutrition. This infers that only one set of soil test need then be conducted and interpreted to meet the goals of both crop productivity and environmental risk assessment. In this perspective, soil P tests initially developed towards providing farmers with an indication of how much P is available to crops in a soil, are increasingly being extended for environmental P testing. This extraction is based on the assumption that the potential of the extracting solutions to desorb phosphate P from soil accurately reflects the amount of P loss from soil to runoff waters during rainfall events (Carmo Horta and Torrent, 2007).

Studies carried out by Schindler *et al.* (2009) have shown that agronomic soil P tests such as Olsen, Bray-Kurtz and Mehlich-3 are in fact all effective in estimating the total dissolved P in surface runoff waters. On this basis, there exists no valid scientific reason why the current agronomic soil P testing in Mauritius based on 0.1M H₂SO₄ extraction as outlined in Cavalot *et al.* (1988), cannot be extended to identify soils which represent a potential threat to surface water quality. As agricultural runoff has been identified as a potential threat to freshwater quality in Mauritius (Ministry of Environment and Sustainable Development, 2011), there is a compelling need to identify soils with a P status representing a risk to water quality degradation in Mauritius. Thus, a study using rainfall simulation to generate runoff was undertaken with the objective of enlarging the scope of the current agronomic soil P test (based on the 0.1M H₂SO₄ extraction), so as to also assess the potential risk of P loss from sugarcane soils to surface runoff waters.

5.2 Research procedure

Though simulated rainfall runoff studies require significant resources, besides being time-consuming and labour intensive, there exists however no better way for establishing and understanding the relationship between soil test P levels and runoff P concentrations (Kleinman *et al.*, 2002; Sharpley and Kleinman, 2003; Guidry *et al.*, 2006). In the present study, the relationship between P in surface soils of Mauritius as extracted by 0.01M CaCl₂ and the concentration of P in surface runoff waters was determined to ascertain firstly that the 0.01M CaCl₂-P from the soils can indeed be a surrogate for P in surface runoff waters. Then the critical soil 0.01M CaCl₂-P threshold above which an unacceptable enrichment of P in surface runoff waters (> 0.03 mg P L⁻¹) would occur was estimated. To be able to use the current agronomic soil P test (0.1M H₂SO₄ extraction) as an environmental P test as well, the relationship between the 0.01M CaCl₂-P and 0.1M H₂SO₄-P of surface soils was thereafter determined using regression and correlation techniques. The value of the 0.1M H₂SO₄-P in soils above which the risk of impairment of surface water quality is most

probable was thereafter derived and was used to assess the environmental P status of sugarcane fields in Mauritius.

5.2.1 Simulated rainfall surface runoff studies

Land in Mauritius as described in Section 3.2.2 has an undulating topography with slopes ranging from 0 to more than 30%, but for practical reasons the simulated rainfall surface runoff studies had focussed on fields with slopes not exceeding 20%. The tests were carried out just after planting since it is at this time, when plant cover is absent, that the risk of soil detachment or movement as a result of intense rainfall is most probable. The rainfall simulation tests were conducted as per the National P Research Project (2001) and with soils at field capacity to eliminate antecedent soil moisture as a variable. Besides, it is at field capacity that runoff and erosion would most likely occur. A detailed description of the rainfall simulator, the establishment of runoff plots and the procedures for collection of sediment and surface runoff samples is provided in Section 3.3.

5.2.2 Laboratory analysis

Following the rainfall simulation tests at the different sites, runoff samples were collected for P determination. Filtered (0.45 μ m) runoff samples were analysed for orthophosphate P by the method described by Murphy and Riley (1962). Unfiltered and filtered runoff samples were digested using perchloric acid as outlined in the standard methods for the examination of water and waste water for determination of total P and total dissolved P respectively (American Public Health Association, 1992).

Soils were sampled adjacent to runoff plots. The 0.01M CaCl₂-P in the air-dried soils (2 mm) was determined by shaking 10 g of the air-dried soil in 50 ml 0.01M CaCl₂ solution for 30 minutes as indicated by McDowell and Sharpley (2001). The agronomic soil P status was determined by shaking one gram of the air-dried soil with 50 ml 0.1M H₂SO₄ for one hour as outlined by Cavalot *et al.* (1988). More details on the laboratory methods used for the various measurements of P are given in Chapter 3.

5.2.3 Data analysis and interpretation

The flow weighted concentration of P in the runoff waters collected during the 30 minutes simulated rainfall event was used to evaluate the relationship between runoff P concentrations and the 0.01M CaCl₂-P level in the soils. Calculation of flow weighted concentration is outlined in Section 3.7.

Approximately 2,700 soil P test results for fields replanted with sugarcane in 2006 and 2007 were obtained from records available at the Mauritius Sugar Industry Research Institute, which offers a free soil testing service to sugarcane growers cultivating less than 10 ha of land, and also from records of sugar estates who perform their own soil P testing. Once the environmental soil P threshold based on 0.1M H₂SO₄-P extraction had been derived from the data in Section 5.3.1, the 2,700 soil P test results were evaluated to assess the effect of the long-term P fertilization on the environmental P status of sugarcane fields in Mauritius.

5.3 Results and Discussion

5.3.1 Relationship between surface runoff P concentrations and soil test P levels

Phosphorus in runoff from agricultural land is an important non-point source of pollution, accelerating eutrophication of lakes and streams (Daniel *et al.*, 1998). Natural concentrations of orthophosphate vary with different catchments depending upon factors such as geology and soil type but natural ranges lie between 0 to 0.01 mg L⁻¹ (European Environment Agency, 2004). Relatively undisturbed and uncontaminated lakes may however also have surface waters that contain total P concentrations ranging from 0.02 to 0.03 mg L⁻¹ (United States Environmental Protection Agency, 1986). As stated by Correll (1998), there exists in fact no clear generally accepted concentration of total P in surface waters that would prevent accelerated eutrophication. For many lakes, streams, reservoirs and estuaries even concentrations of 0.02 mg P L⁻¹ have been reported to cause accelerated eutrophication (Pierzynski *et al.*, 2000). However in general, surface waters that are maintained at a total P concentration of 0.01 to 0.03 mg L⁻¹ tend to remain uncontaminated of algal blooms (Osmond *et al.*, 1995).

In Mauritius, the accepted P level of inland surface waters to control eutrophication should not exceed 0.025 mg P L⁻¹, as there is a growing concern about its associated problems (e.g., increased sedimentation and turbidity of water bodies, ecological changes affecting human health). In this context and on the basis of existing data in the literature, it has been assumed for this study that to ensure no impairment of surface water quality in Mauritius, total P discharge from agricultural lands into the surface waters should not exceed 0.03 mg L⁻¹.

The results obtained in the present study showed a good correlation ($r^2 = 0.93$) between the total dissolved P and orthophosphate P concentrations in the runoff samples (Figure 5.1). This good correlation would avoid the need to analyse for total dissolved P concentration in the runoff samples. The linear relationship moreover indicates that the total dissolved P in the runoff

samples were mostly orthophosphates (approximately 95%) with minimal dissolved organic P, a finding which concurs with the results reported for example by Schindler *et al.* (2009).

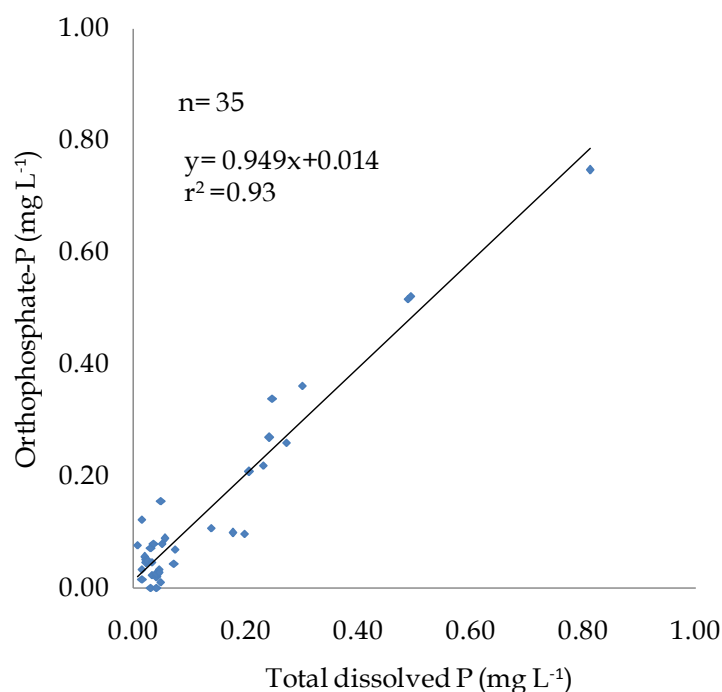


Figure 5.1: Relationship between the flow weighted concentration of orthophosphate P and the total dissolved P in surface runoffs collected over a period of 30 minutes in rainfall simulation studies in Mauritius.

A strong linear correlation ($r^2 = 0.92$) was also found between flow weighted concentration of orthophosphate P in the runoff waters and soil P extractable with 0.01M CaCl₂ (Figure 5.2). These results lend further credence to the conclusion drawn by Schindler *et al.* (2009) and Moody (2011), that the 0.01M CaCl₂-P is a reliable surrogate for estimating P in surface runoff and may be adopted in lieu of using an expensive and time consuming experimental setup in the field for collection of runoff from either natural or simulated rainfall. On the basis of the strong linear relationship shown in Figure 5.2, the 0.01M CaCl₂-P levels in soils which would correspond to P concentration in runoff waters ranging from 0.01 to 0.03 mg L⁻¹ would lie between 0.075 to 0.135 mg kg⁻¹.

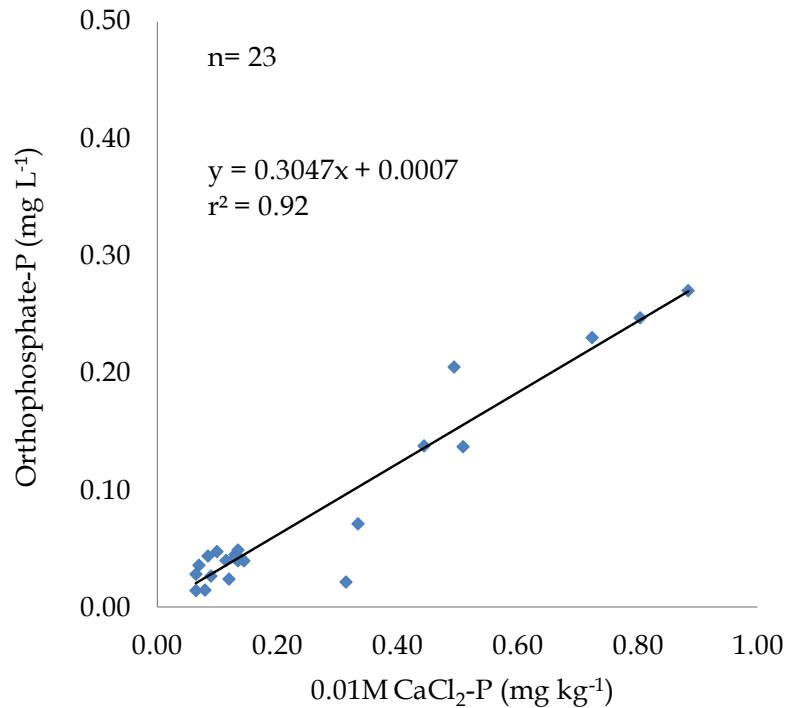


Figure 5.2: Relationship between the flow weighted concentrations of orthophosphate P in runoffs collected over 30 minutes during rainfall simulation studies and the 0.01M calcium chloride extractable P (0.01M CaCl₂-P) in surface soils of Mauritius.

Although the 0.01M CaCl₂ extraction, due to its strong relationship with runoff P concentrations would be a suitable method to assess the environmental P status of soils under sugarcane, it would be desirable from a practical viewpoint that assessment of the environmental P risk be based on the same soil test that is routinely used to formulate P fertiliser recommendations for sugarcane nutrition. Only one set of soil tests would then meet the goals of both agricultural productivity and environmental risk assessment. The 0.1M H₂SO₄ soil P test is already a well-established routine procedure in the P management of sugarcane fields in Mauritius, with no P fertilization recommended for soils having more than 80 mg kg⁻¹ 0.1M H₂SO₄-P as stated in Chapter 4 (Table 4.4)

On the basis of the above considerations, the relationship between the 0.01M CaCl₂-P and the 0.1M H₂SO₄-P of surface soils was examined using the different regression models shown in Table 5.1. As the quadratic model ($y = a + bx + cx^2$) was found to have the highest coefficient of determination ($r^2 = 0.87$), it was accordingly selected to derive the critical 0.1M H₂SO₄-P environmental threshold in the soils.

Table 5.1: Regression models used to describe the relationship between the 0.1M H₂SO₄ soil extractable P (x) and 0.01M CaCl₂ soil extractable P (y) in soils of Mauritius.

Regression models	Coefficient data	Coefficient of determination, r ²
$y=a+bx+cx^2$	$a=0.209, b=(-3.018 \times 10^{-3}), c=(1.59 \times 10^{-5})$	0.87
$y=a+bx+cx^2+dx^3+ex^4$	$a=-0.030, b=(5.76 \times 10^{-3}), c=(-8.59 \times 10^{-5}), d=4.533 \times 10^{-7}, e=(6.573 \times 10^{-10})$	0.85
$y=a+bx$	$a=-0.049, b=(1.843 \times 10^{-3})$	0.63
$y=a(b-e^{-cx})$	$a=0.134, b=0.996, c=(1.394 \times 10^{-4})$	0.62
$y=a(1-e^{-bx})$	$a=7.856, b=(1.934 \times 10^{-4})$	0.59

Based on the quadratic relationship between the 0.01M CaCl₂-P and 0.1M H₂SO₄-P (Figure 5.3) the critical range of 0.01M CaCl₂-P of 0.075 to 0.135 mg kg⁻¹, representing total P concentrations between 0.01 to 0.03 mg L⁻¹ in runoff waters, would correspond to a range of 120 to 160 mg P kg⁻¹ in the soils when 0.1M H₂SO₄ is used as the extractant. It could thus be inferred that impairment of the quality of surface waters in Mauritius would occur if 0.1M H₂SO₄-P in soil exceeds 160 mg kg⁻¹.

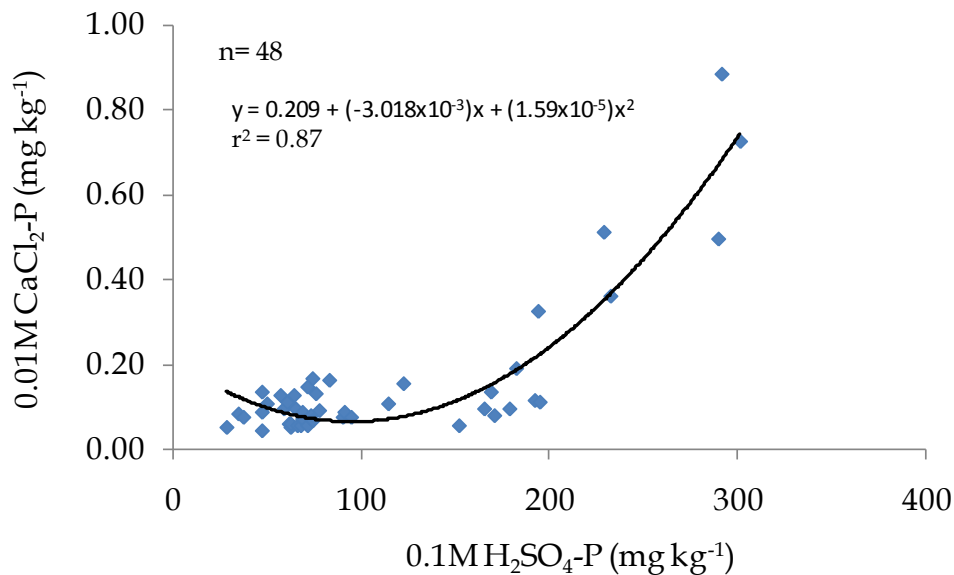


Figure 5.3: Relationship between the 0.01M calcium chloride extractable P (0.01M CaCl₂-P) and the 0.1M sulphuric acid extractable P (0.1M H₂SO₄-P) in surface soils of Mauritius.

5.3.2 Evaluation of the environmental P status of soils under sugarcane in Mauritius

As mentioned in Section 5.1, P fertilisation of sugarcane in Mauritius is practised at replanting only, i.e. after one plant cane plus six or seven ratoons and is based on the agronomic soil P test with no P fertilization recommended for values above 80 mg kg⁻¹ 0.1M H₂SO₄-P. The results obtained in this study have shown that a soil would not pose a threat to surface water quality in Mauritius if the 0.1M H₂SO₄-P is less than 120 mg kg⁻¹. Evaluation of the 2,700 soil P test data of fields replanted in 2006/2007 showed that approximately 72% of the sugarcane soils in Mauritius had a P status below that environmental threshold of 160 mg kg⁻¹ 0.1M H₂SO₄-P (Figure 5.4). Intensive and regular application of P to meet the P requirements of sugarcane has therefore not unduly lead to a soil P status that would represent a threat to water quality in Mauritius.

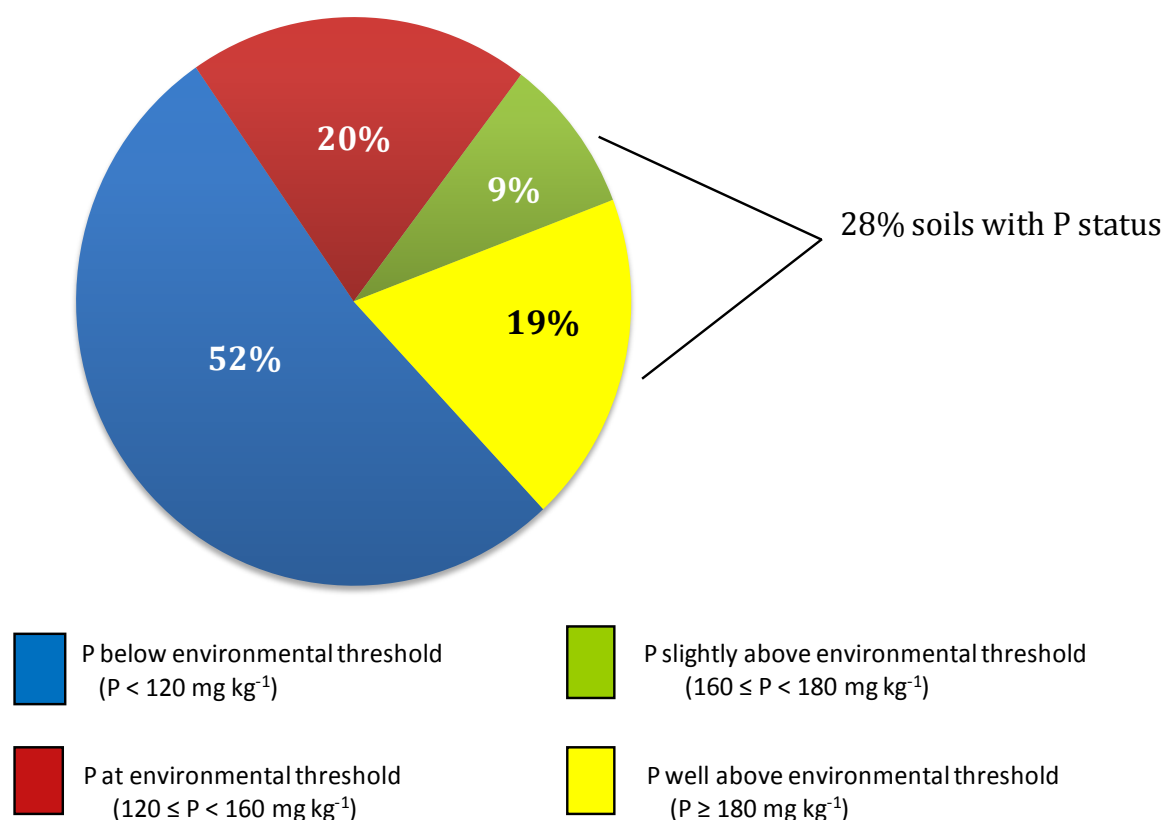


Figure 5.4: Environmental P status of sugarcane soils in Mauritius in 2006/2007.

Only some 28% of the soils had a P status above the environmental threshold of 160 mg P kg⁻¹. It is for those soils that measures such as the creation of riparian zones, the introduction of buffer crops (e.g. *vetiveria*) on the edges of sugarcane plantations or even terracing of sloping fields should be adopted to prevent their P from contaminating freshwater resources.

The results obtained in fact confirm that sugarcane production in Mauritius is predominantly environmentally clean as its P requirements are met in soils containing only 80 mg kg⁻¹ 0.1M H₂SO₄-P. At this soil P level, the P does not constitute a risk to surface water quality, as it is well below the environmental threshold range of 120 to 160 mg kg⁻¹ 0.1M H₂SO₄-P. Thus, replacing sugarcane by other crops in Mauritius may consequently increase the potential for P runoff and the risk of surface water degradation by accelerated eutrophication. With potato for instance, the P needs of this crop cannot be met unless the 0.1M H₂SO₄-P in the soils is above 175 mg kg⁻¹. Substituting sugarcane by potato will imply an even more intensive P fertilisation to raise the soil P status to 175 mg kg⁻¹, a soil P level which, as shown in this study, may be a potential risk to the quality of surface waters in Mauritius.

5.4 Conclusions

Based on field experimental data, the present study has shown that the 0.1M H₂SO₄ soil P test developed by Cavalot *et al.* (1988) to determine the availability of P to sugarcane in soils of Mauritius is also useful as a tool for assessing the environmental risk posed by the soil P in sugarcane fields. This study has therefore shown that with little additional resources in terms of infrastructure or manpower, it is possible to move a step ahead towards maintaining a sustainable sugarcane industry in sugarcane producing countries such as Mauritius while ascertaining that the environment remains clean. Indeed in revealing that the upper critical environmental P limit (160 mg kg⁻¹) is twice that of the agronomic P threshold (80 mg kg⁻¹) for sugarcane in Mauritius, the agronomic objectives of maintaining the soil P status for profitable sugarcane production can be easily met without any risk of raising the P status above the environmental soil P threshold. The interpretation from an agro-environmental perspective of the soil P test levels (using 0.1M H₂SO₄ extraction) in Mauritius as proposed by Mardamootoo (2009) was therefore reviewed and is illustrated in Figure 5.5.

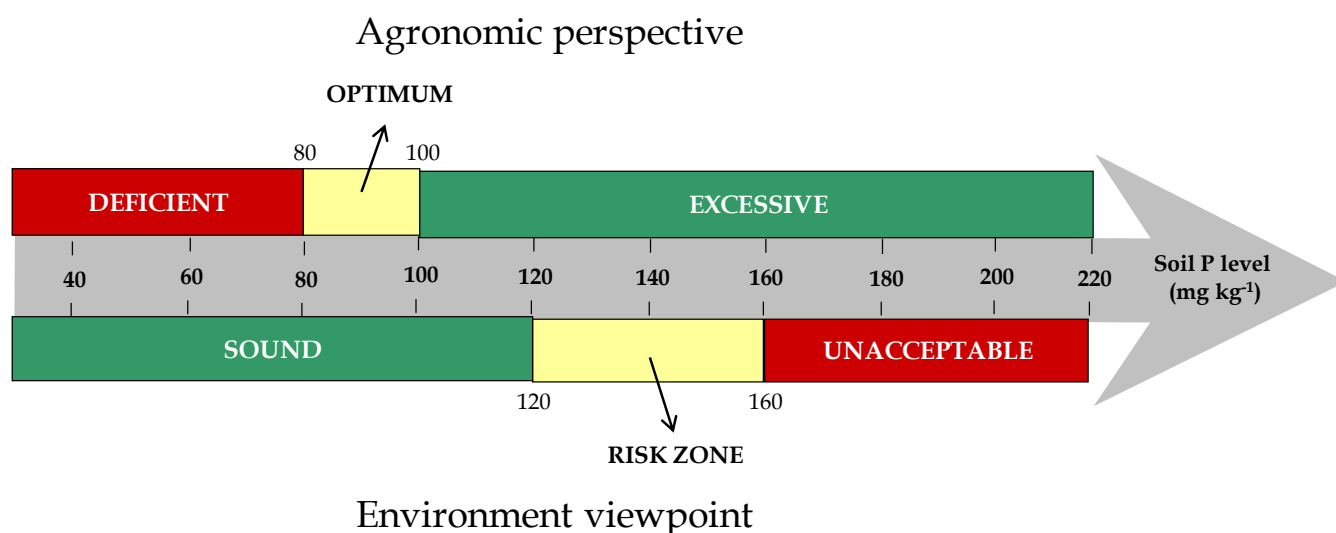


Figure 5.5: A review of the interpretation of the P status of sugarcane soils in Mauritius from an agronomic perspective and an environmental viewpoint using 0.1M sulphuric acid as extractant.

It should however not be inferred that the occurrence of soils with P levels above the environmental threshold is the sole or main reason for the apparent degradation of water quality in Mauritius, as noted by the presence of algal blooms in some regions of the country (Daby, 2001). As reviewed by Gburek *et al.* (2000) variables other than soil P contribute to P movement from fields and landscapes. For P to pose an environmental problem, apart from high soil P levels, the P must be susceptible to transport by leaching, runoff or by soil erosion. Soil P tests only measure soil P concentrations and they give no indication of the potential for the transport of the P in runoff and erosion, the two components known to play an important role in determining the environmental implications of excessive soil P (Gartley and Sims, 1994). Further studies are therefore needed and are reported in the chapters that follow to integrate soil P testing with site characteristics (e.g. erosion, runoff) and P nutrient management practices (P source, rate, method and timing of application) such that an accurate picture of the risks of agricultural P to freshwater eutrophication in Mauritius can be drawn.

6 Soil erosion as a transport factor for phosphorus movement from sugarcane soils in Mauritius

6.1 Introduction

Soil erosion is a dynamic and natural process where soil exists in its natural environment under native vegetation, but man's activities can accelerate the process. Soil erosion by water involves the process of detachment, transport and deposition of soil particles and occurs when soil is exposed to the power of rainfall energy and flowing water (Balousek *et al.*, 2000). Soil loss is dependent on a number of factors such as climatic conditions, soil type, topography and soil management practices. In agronomic terms, soil erosion is a major problem, resulting in the absolute loss of a fundamental resource or redistributing organic matter and nutrient-rich material at landscape scales, along slopes. Soil erosion also represents a substantial environmental threat through land degradation and the washing of sediments and agrochemicals into surface waters, where deposition can restrict navigational and recreational use of waters (Cheesman, 2004). Indeed, the detrimental impact of soil erosion and its associated pollutants on water quality is one of the major impairments of surface waters in most countries (Carpenter *et al.*, 2011).

As discussed in Section 2.4.3 of the literature review, the transport of particulate P from fields results from soil erosion, that is from the detachment and movement of soil particles in response to the intensity of either the falling rainfall or to the energy of flowing water resulting from the rainfall or snowmelt (runoff). It has been estimated that about 43% of P in water originates from agriculture, mainly due to soil erosion (Rickson, 2014). Finer and lighter soil constituents such as clays and organic matter are preferentially transported during runoff events and consequently particles transported in runoff are normally higher in P than in the soil from which they originated (Jin *et al.*, 2009). Actually Ekholm *et al.* (2005), estimated the loss of particulate P as a product of the erosion rate and the P concentration of eroded matter. Unlike dissolved P, particulate P represents a long-term source of P for aquatic biota as it first undergoes solubilisation reactions before becoming available to the aquatic biota and as explained in Section 2.4.5, the bioavailability of particulate P varies between 10 to 90% depending on the nature of the eroding soil and of the receiving lake (Fraser *et al.*, 1999).

Ekholm *et al.* (2005) indicated that soil P tests inadequately estimated total P losses first because it is a poor indicator of the total P concentration of soil and second it obviously does not reflect the erosion rate of soils. The erosion rate for a given site actually depends on the combination of a number of physical and management variables and can be predicted using models such as the

Revised Universal Soil Loss Equation (RUSLE), which includes factors such as rainfall-runoff erosivity, soil erodibility, slope length, slope steepness, cover-management and support practices (Renard *et al.*, 1997). Soil erosion is spatially and temporally variable and this makes it difficult to predict soil loss with any certainty (Moreira, 2007). Spatial variability is attributed to factors such as proximity to water bodies, slope, soil texture and hydrology, combined with management controls such as land use and fertiliser applications while temporal variation is due to factors such as rainfall intensity and duration (Moreira, 2007).

Few studies on soil erosion have been undertaken on sugarcane lands in Mauritius. Among those done was a study carried out at a sub-catchment scale at Valetta on a Humic Ferruginous Latosol soil over three sugarcane growth seasons from 1997 to 2001, to study the movement of soil and nutrients including P (Mauritius Sugar Industry Research Institute; Australian Centre for International Agricultural Research; Queensland Department of Natural Resources and Mines, 2001). As climate, soil type, topography, and other management practices vary across the island, data from that site were only valid for that soil type and climatic conditions prevailing during the years when the measurements were made. Another study to evaluate soil erosion on different soil types of Mauritius and to estimate values of the different parameters influencing soil erosion (e.g. soil erodibility and erosivity factors) was undertaken but without an attempt to measure particulate P losses (Seeruttun *et al.*, 2007). Hence, the objectives of this study were to assess soil erosion as a potential transport factor for P mobilisation by evaluating soil and associated P losses during runoff events.

To address this, simulated rainfall studies were undertaken at 20 sites located on the five major soil types of Mauritius to collect data on the magnitude of soil losses and P loads in the eroded material. Rainfall simulation on runoff plots was a preferred choice over natural rainfall as it is not only relatively cheaper but also provides a more rapid, efficient, and adaptable means of collecting data under a controlled cause-and-effect setting (Dougherty, 2006). In fact, rainfall simulation was originally developed with a view to assess soil erosion (e.g. Meyer, 1965) and was later adapted and found suitable to evaluate P mobilisation under a wide range of conditions and management practices (e.g. Sharpley *et al.*, 2001c; Miller *et al.*, 2006; Kleinman *et al.*, 2006; Schindler *et al.*, 2009).

6.2 Research procedure

Rainfall simulation runoff studies were performed at 20 sites (in duplicates) representing the five major sugarcane soils with slopes ranging between 8 to 20%. As detailed out in Section 3.3, rainfall was simulated at each site on duplicate runoff plots (0.75m by 2.1m) with a rainfall intensity of

100 mm hr⁻¹ was simulated for a total duration of 30 minutes such that runoff and soil erosion could be generated within a reasonable time period. The runoff plots were installed in such a way that they covered one cane row and one interrow, hence the experimental data could be extrapolated and considered representative of a sugarcane field area.

During the 30 minute simulated rainfalls, the volumes of runoff generated were measured and subsamples were collected for further laboratory analysis. Soil eroded during each runoff event was collected using a 2mm sieve as determined in Section 3.3.3 and Figure 3.4. The term 'bedload' is hereafter used to refer to the larger (> 2mm), coarser and more rapidly settling fraction of the eroded sediment which was carried from the runoff plots during the 30 minute rainfall event and deposited in the runoff gutters. The bedload was expressed in dry weight per unit area and was used as an estimate for the soil erosion potential at the different study sites.

The amount of sediment present in the runoff sample was determined by evaporating a sufficient aliquot volume, usually ranging between 50 to 100 ml of runoff collected (Section 3.4.4). The suspended sediment load represents the finer particles usually held in suspension in flowing water, which eventually settles out when water velocity decreases (Falken *et al.*, 1995). Ciesiolka *et al.* (1995) highlighted that differences in the nutrient concentration in the bedload and suspended load is to be expected due to varying P enrichment ratios. Thus, in this study, total P concentrations were determined separately for the bedload and suspended sediment load. The method of Tandon *et al.* (1968) was used to determine total P in the bedload and is described in Section 3.4.5. The total P concentration of runoff samples (i.e. in the suspended load) were determined in perchloric acid digested samples as explained in Section 3.4.2 and referenced in American Public Health Association (1992) for the standard methods for the examination of water and waste water. The agronomic soil P levels were also determined in soil samples which were collected adjacent to the runoff plots by the 0.1M H₂SO₄ extraction of Cavalot *et al.* (1988).

For data processing, the laboratory results obtained for total P in bedload (in mg kg⁻¹) and total P in runoff (in g L⁻¹) were divided by the runoff plot area and was expressed as weight per area (i.e. g ha⁻¹). To investigate the potential correlations between P losses, soil erosion and surface runoff, the relationships between total P in bedload and amount of bedload transported; total P in runoff and the agronomic soil P test levels; total P in runoff and runoff volume; and total P in runoff and the suspended sediment load were plotted in an X, Y co-ordinate plane for each rainfall event simulated.

6.3 Results and discussion

6.3.1 Vulnerability of soils to erosion in Mauritius

The bedload transported from the five soils varied considerably as observed in Table 6.1, reflecting clearly that soil erosion varies across the different soil groups. The highest mean amount of bedload transported was recorded on the Latosolic Reddish Prairie soils (32.67 kg ha⁻¹) followed by Humic Latosol, Humic Ferruginous Latosol, Latosolic Brown Forest and Low Humic Latosol with means of 9.34, 6.77, 3.87 and 2.14 kg ha⁻¹ respectively (Table 6.1).

Table 6.1: Bedload transport (expressed as oven-dried weight per unit area) during a 30 minutes simulated rainfall of 100 mm hr⁻¹ for the five main soil groups in Mauritius.

Soil type	Organic matter (%)	Clay (%)	Occurrence of bedload transport during runoff events (%)	Bedload (kg ha ⁻¹)	
				Range	Mean ±SE
Low Humic Latosol (LHL)	3.9	54.0	37	0.00 - 10.10	2.14 ±1.31
Humic Latosol (HL)	4.5	40.5	100	4.16 - 22.38	9.34 ±2.76
Humic Ferruginous Latosol (HFL)	4.4	34.0	62	0.00 - 25.84	6.77 ±3.79
Latosolic Reddish Prairie (LRP)	4.0	47.0	67	0.00 - 107.08	32.67 ±13.74
Latosolic Brown Forest (LBF)	6.5	30.2	67	0.00 - 12.54	3.87 ±1.93

Variation in erosion rates of the different soils has been attributed to differences in soil properties such as soil texture, aggregate stability, shear strength, infiltration capacity and organic matter content (Woodward and Foster, 1997; Sallaway *et al.*, 2001). Indeed the presence of organic matter in soils is known to act as a cementing agent which renders soils less erodible (Cheesman, 2004). Based on that principle, it would have been expected that the Low Humic Latosol soil which has the lowest OM content (3.9%) out of the five major soils would be the most erodible but that was not the case. In fact, the results of the study have shown that the Latosolic Reddish Prairie soils were the most erodible under the field and experimental conditions described in Section 6.2. A possible explanation could be the fact that removal of rocks and land grading to facilitate cultivation of the naturally rocky Latosolic Reddish Prairie soils prior to planting inevitably disturbed the stable soil structure causing more bedload to be transported during the simulated rainfall runoff events. Moreover, derocking and land grading during field preparation for planting

is known to reduce organic matter content of soil (Ng Cheong *et al.*, 2009) and in the long-term this may increase vulnerability of soils to erosion. In addition, the Latosolic Reddish Prairie soil which has a relatively high clay content (47%) represents a particular risk to P transport due to the preferential erosion of smaller size particles during runoff events as explained by Gburek *et al.* (2005).

Also of particular importance in Table 6.1 is the fact that the risks of bedload being detached from the soil surface and transported to the edge of the plot during runoff events is of the order of 62% and 67% on Humic Ferruginous Latosol and Latosolic Brown Forest soils, which are located in the high rainfall regions (> 2,600 mm annual rainfall). Even though the amount of soil loss from these soils is not as high as the Latosolic Reddish Prairie, the propensity of having runoff events following rainfall combined with small erosion events could account for a significant portion of P loss from fields as shown by the studies done by Quinton *et al.* (2001). Therefore soil conservation practices such as contour planting along hills and mountain slopes, planting vétiver or muguet as a cover crop or buffer strip on field borders on sloping lands and adoption of minimum tillage should not only be targeted to Latosolic Reddish Prairie soils, where the amount of soil loss is highest, as well as where there is a combination of high occurrence of bedload transport during runoffs (e.g. Humic Latosol, Humic Ferruginous Latosol, Latosolic Brown Forest) and relatively high bedload transport.

In fact, soil erosion from sugarcane fields is considered to be efficiently controlled due to the perennial characteristics of the crop and the cultivation practices such as those described further below (Anon, 2007). Actually in some countries (e.g. Guinope, Honduras) sugarcane itself has been employed as a 'live barrier' or in border planting to act as a soil conservation cover crop to reduce erosion (Cheesman, 2004). The relative value of sugarcane as a cover crop is enhanced since the crop tends to remain in the ground for a number of years (due to ratooning) thus producing an extensive root system and also the presence of a closed canopy during a major portion of the crop cycle protects the soil from the erosive effects of rain (Cheesman, 2004).

Sugarcane post-harvest management practices such as green cane trash blanketing following mechanised harvesting has also been found to reduce soil erosion as the mulch formed reduces the impact of raindrops on the soil surface decreasing surface runoff and also acts as a filter which retains the soil particles in suspension in the surface water (Ng Cheong *et al.*, 2003). Moreover, in the super-humid regions of Mauritius where trash blanketing is not recommended, the practice of lining trash in alternate interrows has been found to suffice to maintain soil losses well below tolerable limits (Ng Cheong *et al.*, 2001).

In contrast, the mechanisation of cultural practices which inevitably results in soil compaction due to in-field traffic from heavy machinery increases the risks of soil erosion. As explained by Cheesman (2004), compaction enhances surface sealing and crust formation, which occurs as a result of the breakdown of soil aggregates, with a decrease in soil aggregation, dispersion of fine clay particles can block pore spaces and create a relatively impermeable layer that can form a hardened crust on drying at the soil surface. The consequences of soil compaction on environment include reduction in infiltration rate, which exacerbates runoff and erosion and most probably enhanced P losses (Ng Cheong *et al.*, 2009). In order to alleviate soil compaction and its associated problems, inter-row decompaction in ratoon or subsoiling to increase soil porosity, minimum tillage and controlled traffic have been suggested (Anon, 2008; Ng Cheong *et al.*, 2009).

6.3.2 Total P transported by soil erosion

In an attempt to evaluate the risk of P mobility associated with soil erosion, the correlation between the amount of bedload transported during the simulated rainfall runoff event and the P attached to the bedload was established. A strong positive linear relationship ($y=1.433x$, $r^2=0.97$) was obtained showing that the amount of total P mobilised tended to increase with increasing rates of eroded material (Figure 6.1). The amount of P that could be mobilised during soil erosion from the runoff plots ranged between 0 to 163 g P ha⁻¹ for corresponding soil losses of 0 to 107 kg ha⁻¹ at simulated rainfall intensities of 100 mm hr⁻¹ for a 30 minute duration. As the loss of bedload enriched P is a function of soil erosion, conservation control measures that mitigate soil losses would concomitantly decrease the associated P losses. Therefore, soil conservation practices, as proposed by the Mauritius Sugar Industry Research Institute (2007), which include contour planting along hills and mountain slopes, planting of *vétiver* or *muguet* on field borders of sloping lands and adoption of minimum tillage would also be effective in controlling P losses. In addition, as highlighted in Section 6.3.1, sugarcane cultural practices, which reduce soil erosion (e.g. trash blanketing) will also help mitigate P losses while soil compaction due to mechanization of cultural operations may exacerbate P transport via soil erosion. Moreover a decrease in either soil P or P application rates can lead to decreases in P losses by erosion.

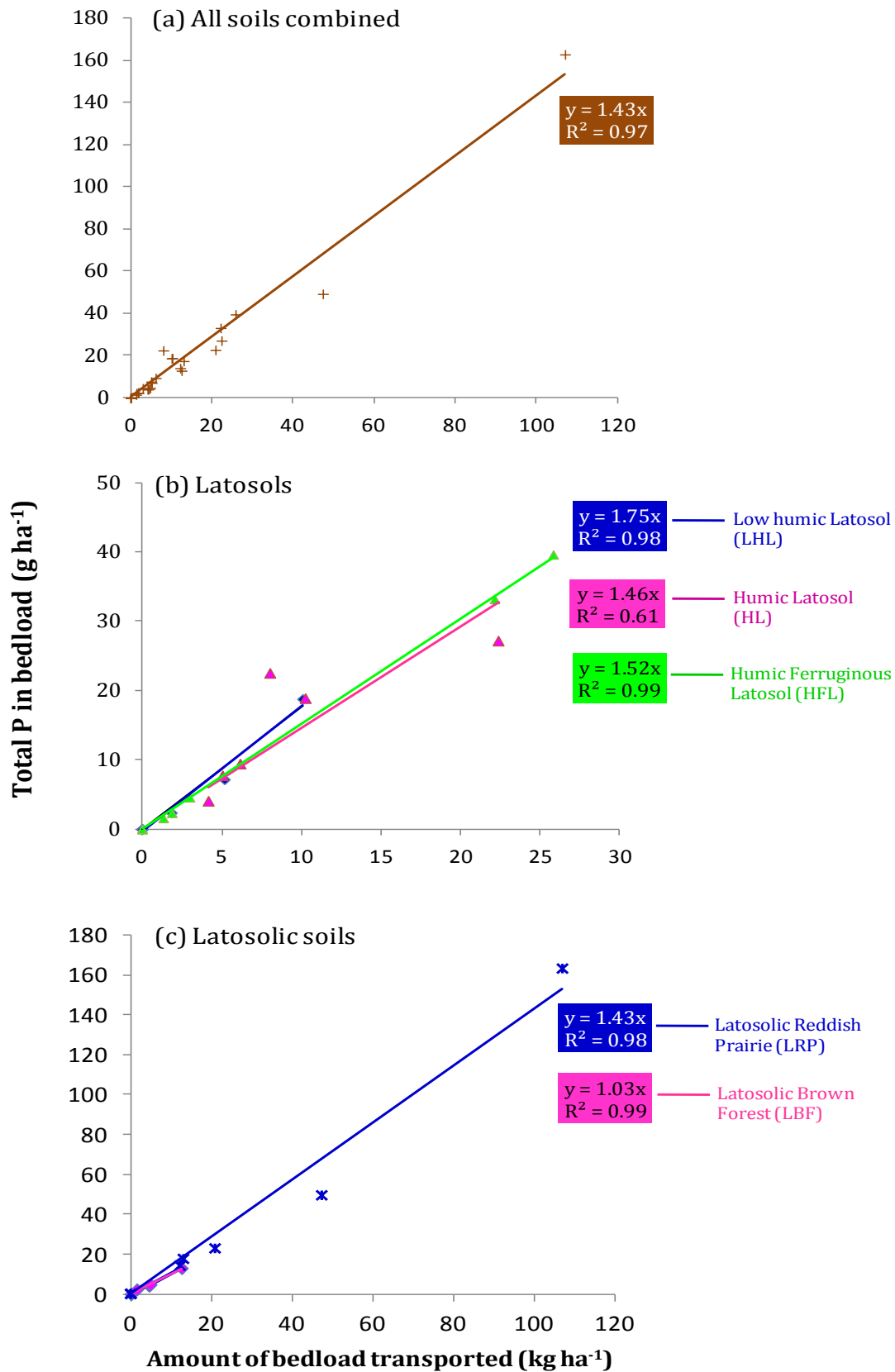


Figure 6.1: Relationship between the amount of bedload transported and bedload enriched P for (a) all soils combined, (b) latosols and (c) latosolic soils following simulated rainfall events of 100 mm hr⁻¹ for a total duration of 30 minutes.

Moreover, the regression slope of the linear relationship in Figure 6.1 actually represents the amount of P transported for every kilogram of soil that would be transported. It was observed that the regression slopes varied across the different soil types due to differences in the intrinsic soil properties. For example, the regression slopes for the Low Humic Latosol of the *Latosols* and Latosolic Reddish Prairie of the *Latosolic* soils were higher (Figures 6.1 b,c) which actually coincides with a comparatively higher clay content of 54.0% and 47.0% for the Low Humic Latosol and Latosolic Reddish Prairie respectively compared to the other soils of their respective groups (Table 6.1). Indeed Quinton *et al.* (2001) showed that there was a strong relationship between clay and P content. Penn *et al.* (2005) further explained that the high surface area and presence of various P sorbing minerals of high clay soils often absorb more P compared to other soils (e.g. coarse-textured soils). Therefore it would be expected that for similar erosion rates, more P will be mobilised from the high clay soil such as Low Humic Latosol and Latosolic Reddish Prairie. In addition, it is well established that erosion is a P transport mechanism that preferentially removes finer-sized soil particles and as a result the P content and reactivity of eroded material is usually greater than the soil from which it originated (Sharpley *et al.*, 2001b). Actually reference is often made to the term P-enrichment ratio which is a means of representing the integrated effective P loss from soils by accounting for the disproportionate effect of finer particles (Gburek *et al.*, 2005).

In order to effectively address water quality problems related to soil P losses during erosion, apart from erosion control measures, it is also imperative to take into consideration the sediment delivery ratio that is the proportion of eroded sediment that reaches the edge of the field or water body compared to the actual amount that leaves the detachment site. In reality, only a fraction of eroded sediment reaches receiving waters, whereas the other part could be deposited (e.g. at the toe of a slope, along field boundaries, in channels or depressions) before reaching the nearest stream or water-body (Lemunyon and Gilbert, 1993). Deposition of the eroded particles usually occurs when the velocity of the flowing water (i.e. the transporting agent) gets too low, for example when the ground becomes less sloping (Sallaway *et al.*, 2002). It is usually the heavier and coarser eroded soil particles which settle down during transport while the lighter and finer silt-sized soil aggregates tend to remain in suspension in runoff waters and are usually transported longer distances. In this context, the transport of P in surface runoff is discussed in the next section.

6.3.3 Phosphorus losses by surface runoff

In addition to soil erosion, surface runoff during rainfall events also represents a considerable pathway for P losses as explained in Section 2.4.3. In that context, the ability of the agronomic soil P test to also estimate total P losses was tested by establishing the relationship between the soil P

test levels (i.e. 0.1M H₂SO₄ extractable P) and the total P transported in runoff waters. From Figure 6.2, it is clear that the agronomic soil P test, which was initially designed to measure the amounts of P that would be released during the crop growing season did not show any correlation with total P losses in surface runoff during the simulated rainfall events.

As highlighted by Quinton *et al.* (2003), agronomic soil P tests such as the 0.1M H₂SO₄ extraction is related to the solubilisation of P and would therefore be expected to better predict dissolved P losses as previously shown in Section 5.3.1. Withers *et al.* (2007) suggested that soil P methodologies with some predictive value for assessing the degree to which surface runoff becomes entrained with eroding particles and enriched in P are potentially useful. Besides, the poor correlation between soil test P levels and total P losses indicates that there are other factors apart from soil P which are contributing to total P transfer in runoffs. Actually, none of the factors controlling surface runoff such as rainfall characteristics, soil detachment and soil permeability is directly related to soil P, and we would therefore not expect soil P tests to predict total P transfer in runoff with any confidence (Quinton *et al.*, 2003).

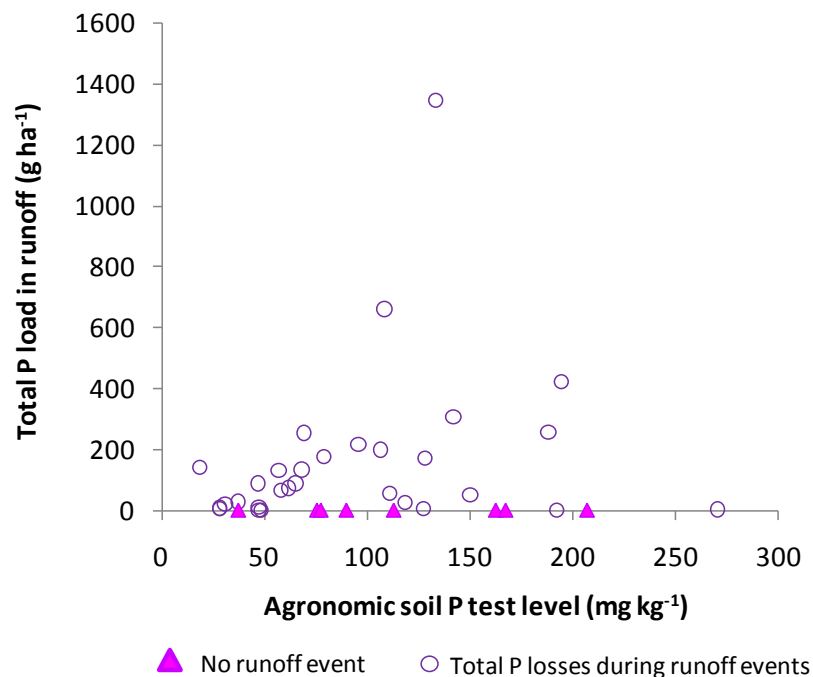


Figure 6.2: Relationship between the agronomic soil P test level (using the 0.1M H₂SO₄ extraction) and the total P loads in runoff waters following the 30 minutes simulated rainfall events of 100 mm hr⁻¹.

The relationship between runoff total P concentration and runoff volume was also established and a linear correlation ($y=7.29x-3.71$, $r^2=0.49$) was obtained (Figure 6.3a). This relationship was considerably improved when total P concentration in runoff volume was instead correlated to the suspended load in runoff rather than the runoff volume. Indeed, the total P concentration was found to be linearly correlated to the suspended load present in runoff waters with correlation coefficient of 0.92 (Figure 6.3b) indicating that a greater proportion of the P transported in runoff occurred as particulate P rather than dissolved P. It also demonstrates that P compounds in soils are sparingly soluble and that P is strongly adsorbed to soil particles (Wang *et al.*, 2010).

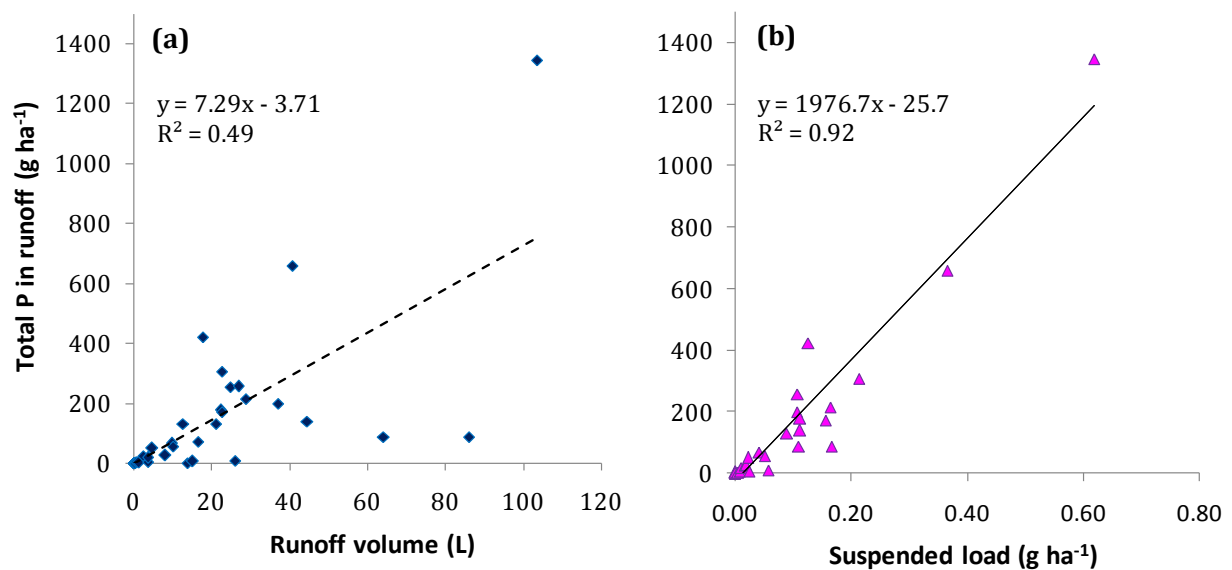


Figure 6.3: Relationship between total P in runoff waters and (a) runoff volume and (b) suspended load following simulated rainfall event of 100 mm hr^{-1} for a duration of 30 minutes.

Similar research findings have been obtained by Cox and Hendricks (2000) and Kleinman *et al.* (2002) showing that total P losses in runoff were highly influenced by suspended sediment load in runoff waters. These findings obtained suggest that management practices which reduce runoff volume (e.g. conservation tillage, cover crops) will to a certain extent reduce total P transport from fields. Moreover, reducing suspended sediments in runoff waters will also attenuate P losses and this can be achieved by having grassed waterways, buffers such as riparian buffers and buffer strips. In the next section, the different forms in which P is mobilised from the runoff plots and how suspended load influences the forms of P transported in runoff is discussed.

6.3.4 Forms of P mobilised

As highlighted in Section 2.4.3, soil P may be transported by surface runoff when water flowing across the soil surface either dissolves and transports soluble P or erodes and transports

particulate P. In Figure 6.4, it can be observed that irrespective of soil type, particulate P was the dominant form and ranged between 67 to 99% of the total P mobilised in runoff waters. These results actually concur with previous studies done at a watershed scale in Mauritius by Ng Kee Kwong *et al.* (2002) which showed that P transport in surface runoff was intimately linked to sediment.

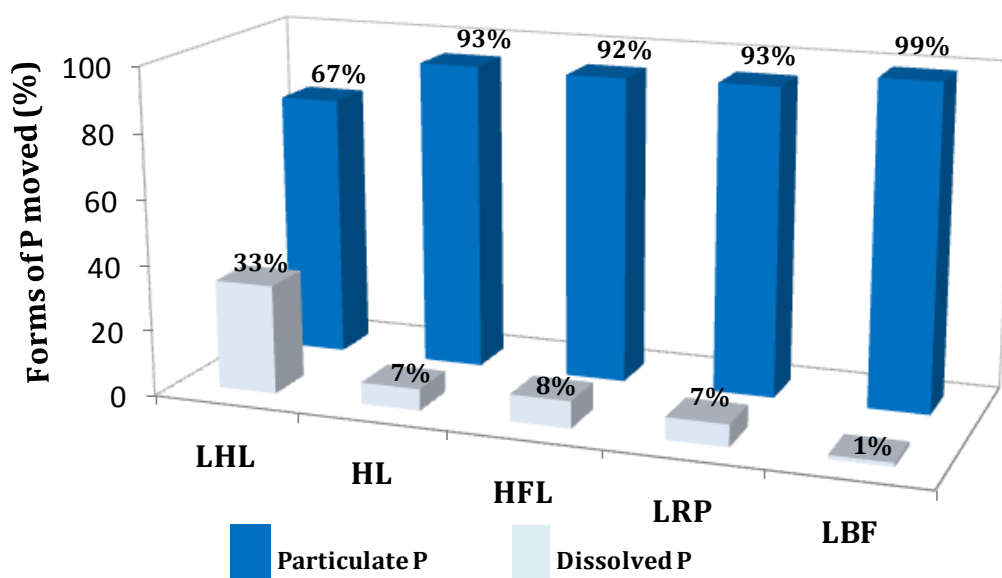


Figure 6.4: Average particulate and dissolved P losses (expressed as a % of total P) in runoff following 30 minutes simulated rainfall events of 100 mm hr⁻¹ on the different soil groups of Mauritius.

Although dissolved P constitutes only a minor portion (1 to 33%) of the total P mobilised in runoff waters, it is in a form which is immediately available for algal uptake causing accelerated eutrophication of freshwaters. In contrast, particulate P constitutes a long-term source of P once deposited in water bodies and its bioavailability as described in Section 2.4.5 and illustrated in Figure 2.8, depends on the nature of the sediment and also the surrounding environment (Truman *et al.*, 1993).

Results obtained in Figure 6.4, further clarify why the soil P test levels (i.e. 0.1M H₂SO₄-P) poorly correlated to total P losses (refer to Figure 6.2) in runoff because most of the P was lost in particulate forms which involved the detachment of soils and sediments during rainfall and runoff instead of a desorption-dissolution reaction controlling P release from surface soils in dissolved forms. The agronomic soil P test (0.1M H₂SO₄ extraction) utilised in Mauritius only extracts the

plant-available P fraction from the soil (i.e. soil P concentration) and gives no indication of the potential for particulate P losses in erosion or runoff. Quinton *et al.* (2003) emphasized that soil P tests in combination with sediment concentrations would provide a better means of assessing the mobilisation of total P in surface runoff.

In addition, it was found that the suspended sediment load present in the runoffs accounted for about 94% of the variation in the amount of particulate P measured in runoff samples collected during the simulated rainfall events (Figure 6.5).

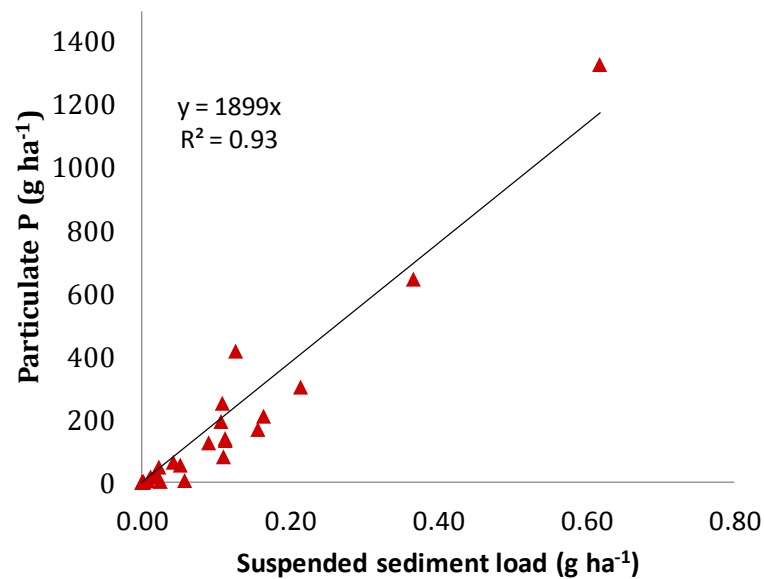


Figure 6.5: Relationship between suspended sediment load and particulate P measured in runoff samples collected during 30 minutes simulated rainfall of 100 mm hr⁻¹ at 20 sites in Mauritius.

Furthermore, the high correlation observed between the total P and suspended load (Figure 6.3b) was due to the apparent relationship between suspended sediment load and particulate P losses in runoffs (Figure 6.5). The suspended load obviously plays an important role as a carrier of particulate P and controlled runoff P losses as explained by Miller *et al.* (2006). This therefore confirms that the adoption of management practices such as cover crops, grassed waterways and conservation buffers which intercept runoff flow and trap sediments from agricultural land will be effective in reducing P losses.

6.4 Conclusion

Simulated rainfall runoff studies have shown that soil erosion does indeed represent a major mechanism for P transport from sugarcane soils in Mauritius. Owing to differences in soil

properties, soil erosion rates varied across the different soil types with the highest average erosion rates being observed from the Latosolic Reddish Prairie soils followed by Humic Latosol, Humic Ferruginous Latosol, Latosolic Brown Forest and Low Humic Latosol. Moreover, the amount of total P mobilised during erosion was found to be linearly correlated to the bedload transported during erosion, demonstrating clearly that P losses were strongly associated with eroded soil materials. The amount of P load mobilised during soil movement ranged between 0 to 162.91 g P ha⁻¹ at simulated rainfall intensities of 100 mm hr⁻¹ for 30 minutes.

Of particular importance is the fact that total P mobilised during runoff events was closely associated with suspended sediments present in runoff waters, indicating that soil detachment during soil erosion represents a major mechanism for P losses during rainfall events. It was also observed that irrespective of the soil type, the major portion (89% on average) of P losses occurred in particulate forms which were in turn strongly linked to suspended sediments present in runoff waters, which signifies that suspended soil particles controlled runoff particulate P.

Additionally, the soil P test used to formulate P fertiliser recommendations for sugarcane in Mauritius (Cavalot *et al.*, 1988) was as expected found to be inappropriate to predict total P losses during surface runoff and erosion. This is due to the fact that such soil P tests provide only an estimate of soil desorbable P without assessing the degree to which runoff waters becomes entrained with eroding particles and enriched in P. Knowledge about soil P transport via soil erosion and surface runoff in conjunction with soil P status (as determined by soil P tests) are key components required not only to predict P mobilisation but to also choose the best P management practices. Soil erosion control measures which seek primarily to protect the soil and its productivity can also be implemented to control the off-site impact of erosion notably those associated to P losses. Soil conservation measures include riparian buffers, grassed waterways, terraces, strip cropping and conservation tillage systems (Sims and Kleinman, 2005). Minimum tillage practices have indeed been adopted on sloping lands in Mauritius since 1980s for a better control of soil erosion (Ismael *et al.*, 2008). Despite the fact that erosion control has been shown to decrease particulate P losses, environmentally significant concentrations of dissolved P can occur in runoff from soils where erosion is kept at a minimum, especially from soils with a high P status or having recently received P applications (Penn *et al.*, 2005). Furthermore, the long-term adoption of conservation tillage practices as highlighted by Sims and Kleinman (2005) may actually exacerbate dissolved P losses in surface runoff due to stratification of P at the soil surface with an accumulation of P in the upper few centimetres, due to repeated surface applications of fertilisers or organic amendments without the mixing induced by plowing.

Even though sugarcane management practices implemented so far have been effective in controlling soil erosion, the increasing adoption of mechanised sugarcane cultivation practices have lead to a number of changes in the planning of fields which warrant particular attention with regard to soil erosion and its associated P losses. Such changes to facilitate the mechanisation of cultural practices include for example, longer cane rows on sloping (often in the direction of the slope) and soil surface derocking prior to plantation of sugarcane represent high risk zones for soil erosion. It is therefore imperative that improved P management practices be considered for those high particular areas. Also Latosolic Reddish Prairie soils necessitate particulate attention because in addition to being the most erodible soils, the Latosolic Reddish Prairie soils have a relatively high clay content (47%) which further accentuates the potential threat which this soil group represents to P mobilisation. Clayey soils are known to absorb more P and will act as a carrier for P during soil erosion and runoff events. The same also applies to Low Humic Latosol soils, which have on average a clay content of 54%.

It must be borne in mind that measurements of runoff and erosion from rainfall simulator tests on small plots represent edge-of-plots soil and P losses and the results are best restricted to comparisons. Sharpley and Kleinman (2003) emphasized that rainfall simulators and small plots cannot reproduce flow process occurring over a landscape or hillslope and as such must be limited to elucidating flow-soil P interdependencies. In fact, as we increase the scale, the system we are dealing with becomes more complex and the processes which are controlling the amount of P in surface waters change and shift in their relative dominance. The research findings actually provide a snapshot of cane fields when they are most prone to soil erosion and associated P losses. Indeed, the rainfall simulation runoff tests were done at a time in the crop cycle when there was no plant canopy to protect soil surface from rainfall impacts, fields were recently prepared and furrowed for plantation and also the soil was brought to field capacity prior to the tests. Hence, these results obtained from simulated rainfall over small runoff plots should be combined with research findings obtained at field or watershed scales for more elaborate interpretations.

In addition to information about transport factors, such as erosion and surface runoff, knowledge of factors governing surface runoff generation, proximity and connectivity of the critical source areas to surface waters, sensitivity of receiving waters to nutrient inputs is necessary to gain an appreciation of the potential of agricultural P losses to cause eutrophication. Topographical features of the agricultural landscapes influence P transfer by controlling water movement from the P source areas and the effect of topography on P transport is studied in the next chapter.

7 Role of topography on phosphorus movement from sugarcane soils in Mauritius

7.1 Introduction

The major physiographic features of Mauritius comprise a discontinuous ring of mountain ranges and isolated peaks, some 600 to 800m high, which enclose a central tableland rising from about 300m in the north to 600m in the southwest surrounded by low undulating, often hummocky, coastal plains (Arlidge and Wong You Cheong, 1975). Parish and Feillafé (1965) identified three distinct topographic patterns connected with the age of the parent lava. First, there are the comparatively small but often scenically dramatic mountain ranges of the oldest lavas. Then the gently rolling topography of the Intermediate Lavas of the Younger Volcanic Series with deeply incised rivers and some terraces and now stabilized gullies. Finally, there are the Late Lavas, with many extremely rocky areas of distinctly hummocky appearance and an almost complete absence of surface drainage, dominated by the vents which gave rise to this topographical region. The classification of landforms and their associated slope ranges has been previously described in Section 3.2.2.

Although cultivation on sloping lands is undesirable, socio-economic factors (e.g. population growth, high food demand) have contributed to the expansion of agricultural land on to slopes in various parts of the world (Cheesman, 2004). Actually in Mauritius, sugarcane is planted on about 5,000 hectares of land with slopes that sometimes exceed 20% (Ismael *et al.*, 2008). Figure 7.1 illustrates a topographic map of Mauritius highlighting the sugarcane areas in green. The closer the contour lines, the steeper the slopes and conversely the further apart the contour lines are, the gentler are the slopes. For example, it can be noted in Figure 7.1 that the southwestern area has a sloping topography compared to the northern area which is relatively flat. Based on the information available at the Land Index database of the Mauritius Sugarcane Industry Research Institute, it was noted that approximately 84% of sugarcane lands have been cultivated on flat to gently sloping lands (0 to 8% slope), 13% on moderately steep slopes (13 to 30% slope), and 4% on steep to very steep slopes (slope >30%).

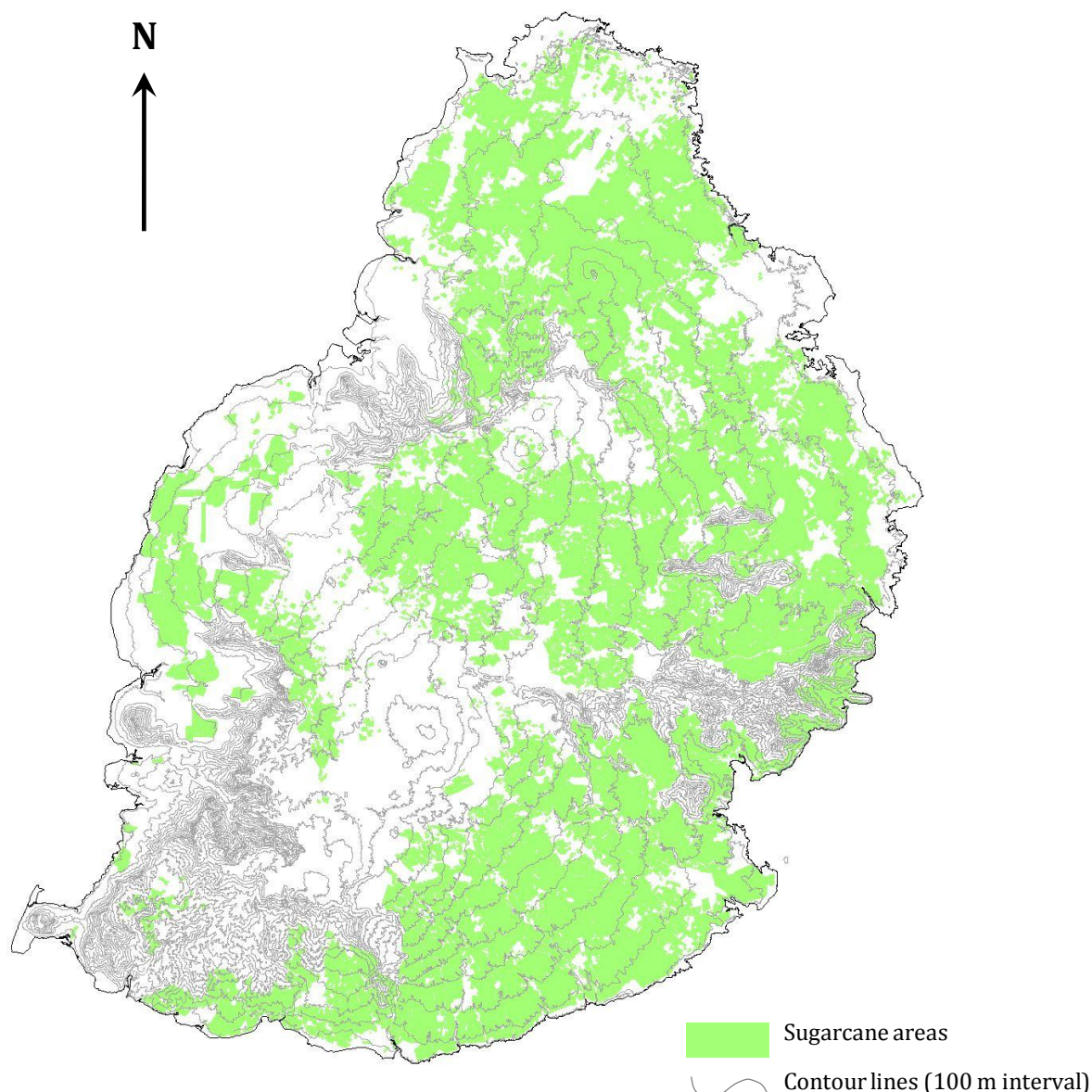


Figure 7.1: Topographic map of Mauritius including land areas under sugarcane cultivation (Map prepared by the GIS lab, Mauritius Sugarcane Industry Research Institute, October 2014).

The risks of surface runoff and soil erosion during rainfall are amplified with increasing field slopes and therefore cultivation on sloping land is not recommended. In fact, according to the laws of hydraulics, for each fourfold increase in the slope, the velocity of runoff water doubles (Cheesman, 2004) and as the runoff velocity increases, the power of flowing water to tear the soil surface loose and carry it along hillslopes increases rapidly (Gustafson, 1948). Moreover, Sharpley (1985b) showed that as soil slope increased, the effective depth of interaction of the surface soil to flowing water also increased and was found to be a function of runoff energy. This increase in the effective depth of interaction causes an increase in soil P desorption due to the turbulent mixing of

the thin soil surface layer with rainfall water (Ahuja *et al.*, 1982). Sharpley (1985b) stipulated that there are two physical processes involved in the development and maintenance of the effective depth of interaction during a runoff event, first there is the impact of raindrops on the soil surface which initiates turbulent mixing of rainfall water in a thin surface zone and second energy of the flowing runoff water as it moves downslope.

From the above description of hydrological processes, it is evident that landscape topography affects the processes of runoff and erosion and in consequence influences the amount of P transported from agricultural fields. In this perspective, field experiments were initiated to evaluate the influence of different slope categories on P mobilization from sugarcane lands in Mauritius at a small runoff plot scale. Rainfall was simulated on runoff plots (2.1m by 0.75m) at three different slope categories and the amount of P transported during runoff and erosion were investigated. The research findings are presented and discussed in this chapter.

7.2 Research procedure

7.2.1 Field experimental setup

Rainfall simulation tests as per the National P Research Project (2001) were performed at 20 sites representative of cane fields in Mauritius. Since sugarcane is rarely cultivated on land with more than 20% slope, this study focussed on fields with slopes not exceeding 20%. The field tests were carried out just after planting since it is at this time of the crop cycle, when plant cover is absent, that the risk of soil detachment or movement as a result of rainfall is most probable. As indicated by Haygarth *et al.* (1999), most P export occurs when the land is at or close to field capacity and therefore the runoff plots were brought to field capacity prior to the rainfall simulation tests. Moreover, by initiating rainfall simulation at field capacity conditions, hydrologic variability between plots related to spatial differences in antecedent soil moisture is reduced and in addition eliminates the need to apply high and infrequently occurring rainfall intensities (Srinivasan *et al.*, 2007).

A rainfall simulator similar to that described by Loch *et al.* (2001) was used to simulate rain at three different intensities (50, 100 and 150 mm hr⁻¹) with each simulated rainfall event lasting 30 minutes. The rationale behind the selection of those three rainfall intensities is discussed in Section 3.3.1 and details of the runoff plot establishment is given in Section 3.3.3. The runoffs generated during the 30 minutes simulated rainfall event were collected and the volumes measured. The soil eroded during that time period of 30 minutes, henceforth referred to as

bedload was also collected, and the dry weight used to estimate the weight of soil loss per unit area. Soil samples were also collected (as per procedure described in Section 3.5.1) to determine the P status of the fields studied.

7.2.2 Laboratory procedures

Total P in the runoff samples were determined as per the standard methods adopted for the examination of water and waste water by the American Public Health Association (1992). The method of Murphy and Riley (1962) was adopted to determine orthophosphate P concentration in the 0.45 µm filtered runoff samples (hereafter referred to as soluble or dissolved P). The sediment bound P was estimated as the difference between total P in the unfiltered runoff samples and orthophosphate P in the filtered runoffs (Shigaki *et al.*, 2007). The soil P status at the different sites was determined by the method of Cavalot *et al.* (1988) while the total bedload P level was determined by the method of Tandon *et al.* (1968). The technicalities of all the above-mentioned laboratory procedures are discussed in Sections 3.4 and 3.5.

7.2.3 Data processing and interpretation

Runoff P loads (g P ha⁻¹) were estimated from the flow weighted runoff P concentration (mg P L⁻¹) and runoff volumes generated for a known runoff plot area of 2.1m by 0.75m. In a similar manner, the total bedload P (g ha⁻¹) mobilised was estimated from the bedload P content (mg P kg⁻¹) and the total bedload transported during the 30 minutes of simulated rainfall. The calculations of means and standard deviations were done using Microsoft Excel while standard error was calculated using the equation below:

$$\text{Standard error} = \frac{\text{Standard deviation}}{\sqrt{(\text{number of observations})}}$$

7.3 Results and discussion

7.3.1 Effect of field slope on P transport by soil erosion

The potential for bedload transport and the associated P losses increases with increasing field slopes (Figure 7.2). With added steepness of field slope, the increasing velocity of flowing water has greater power to detach soil particles (i.e. to cause erosion) and carry it along the field slope (Gustafson, 1948). This increase in soil erosion with increasing field slopes causes more bedload P to be mobilised as illustrated in Figure 7.2. Research findings presented in Section 6.3.2 showed that there was a positive linear relationship between the total bedload P mobilised and soil erosion rates. Besides, the soils of Mauritius are generally acidic to near neutral and are known to fix P through adsorption with hydrous oxides of Fe and Al as discussed in Section 2.2.2.3, thus

explaining why eroded bedload contained significant amounts of P. It was also observed that the vulnerability of the soils to erode increased with increasing rainfall intensities and consequently the amount of bedload P transported during surface runoff events increased (Figure 7.2). This effect of increasing rainfall intensities on soil erosion and P transport has been discussed in Chapter 8.

The wide standard error bars in Figure 7.2 indicate that apart from field slope and rainfall intensity, there are other factors influencing soil erosion potential. For example, under the same cropping system and similar amount of rainfall, some soil types are known to erode more easily as compared to others. The soil intrinsic factors such as soil organic matter, clay and oxides content determine to a large extent the soil erodibility of different soil groups. Clay and organic matter content have cementing and binding abilities that play a crucial role in the formation and stability of soil aggregates, which eventually influence the vulnerability of soils to erosion (Wuddivira *et al.*, 2006).

As P is attached to eroded particles, application of erosion control measures will also serve to decrease P delivery from agricultural systems. For example, the adoption of minimum tillage as an erosion control measure on sloping sugarcane lands in Mauritius is a well-established practice (Seeruttun *et al.*, 2007), which will also be beneficial in addressing P loss from fields. Other examples of soil conservation practices which will also help reduce risks of P mobilisation from agricultural fields include contour farming along hills, appropriate crop residue management, installation of grassed waterways in concentrated flow areas and using buffers to intercept runoff (Sharpley *et al.*, 2006).

7.3.2 Effect of field slope on mobilisation of P in surface runoff

Studies done by Haggard *et al.* (2005) using the variable-slope box technique showed that surface runoff volume increased with increasing slope steepness while all other parameters (e.g. rainfall intensity, soil type) were maintained constant. The research findings illustrated in Figure 7.3 (a) confirm that runoff volumes were larger with increasing slopes irrespective of the rainfall intensity simulated. These larger runoff volumes were accompanied by simultaneous increases in total runoff P loads due to an increase in the effective mass of surface soil interacting with impacting rainfall and flowing runoff, thereby causing more P to be transported. Haggard *et al.* (2005) pointed out that *dissolved P concentrations* in surface runoff waters were not related to slope or runoff volume but it was rather the P loads that increased with slopes in a fashion similar to surface runoff suggesting that field topography influences P loss through increased surface

runoff. Moreover, studies done by Sporre-Moeny *et al.* (2004) showed that total P loads were most closely related to total runoff volumes than soil test P levels.

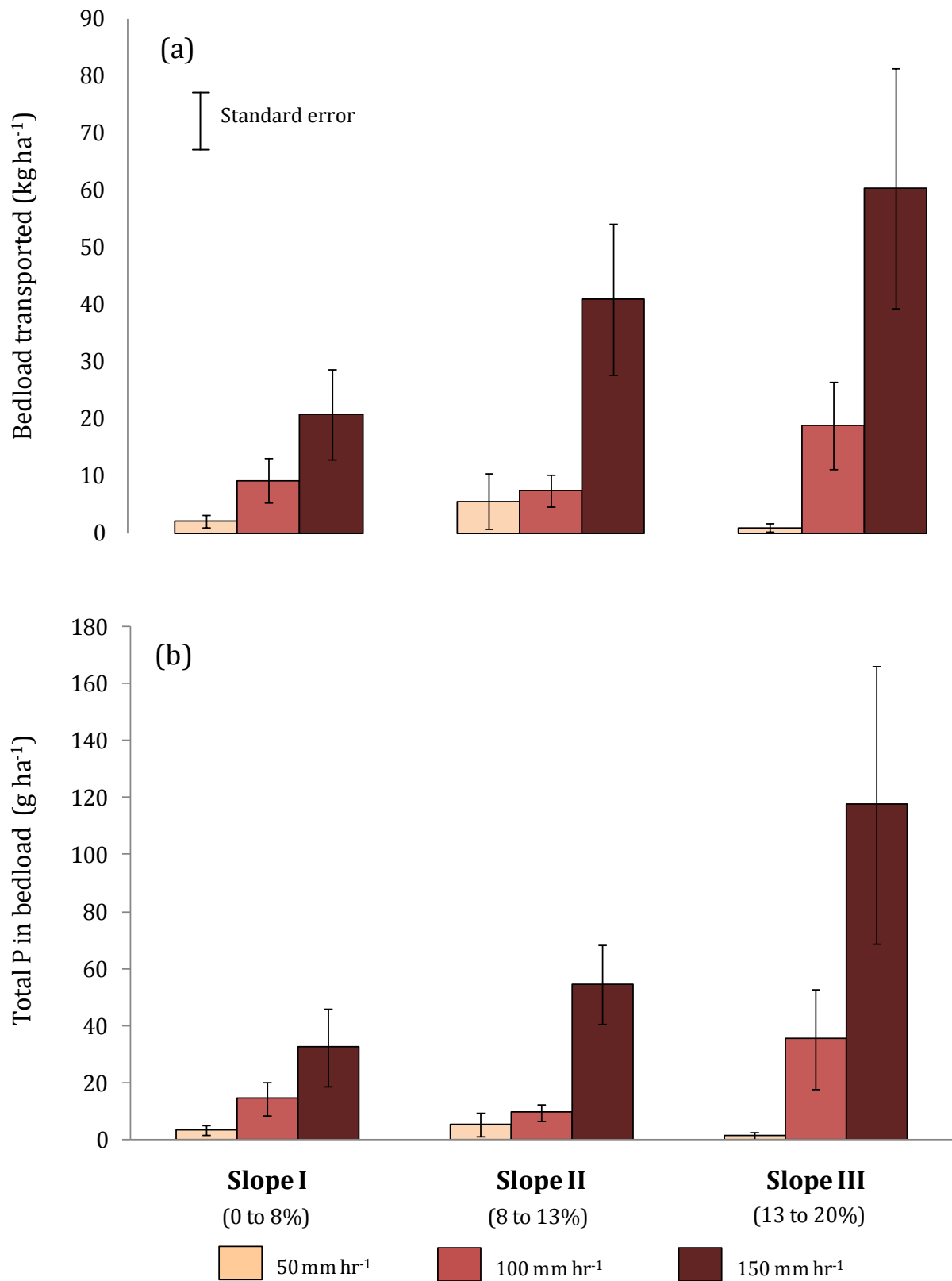


Figure 7.2: Effect of slope on (a) amount of bedload transported and (b) associated P lost at rainfall intensities of 50, 100 and 150 mm hr^{-1} .

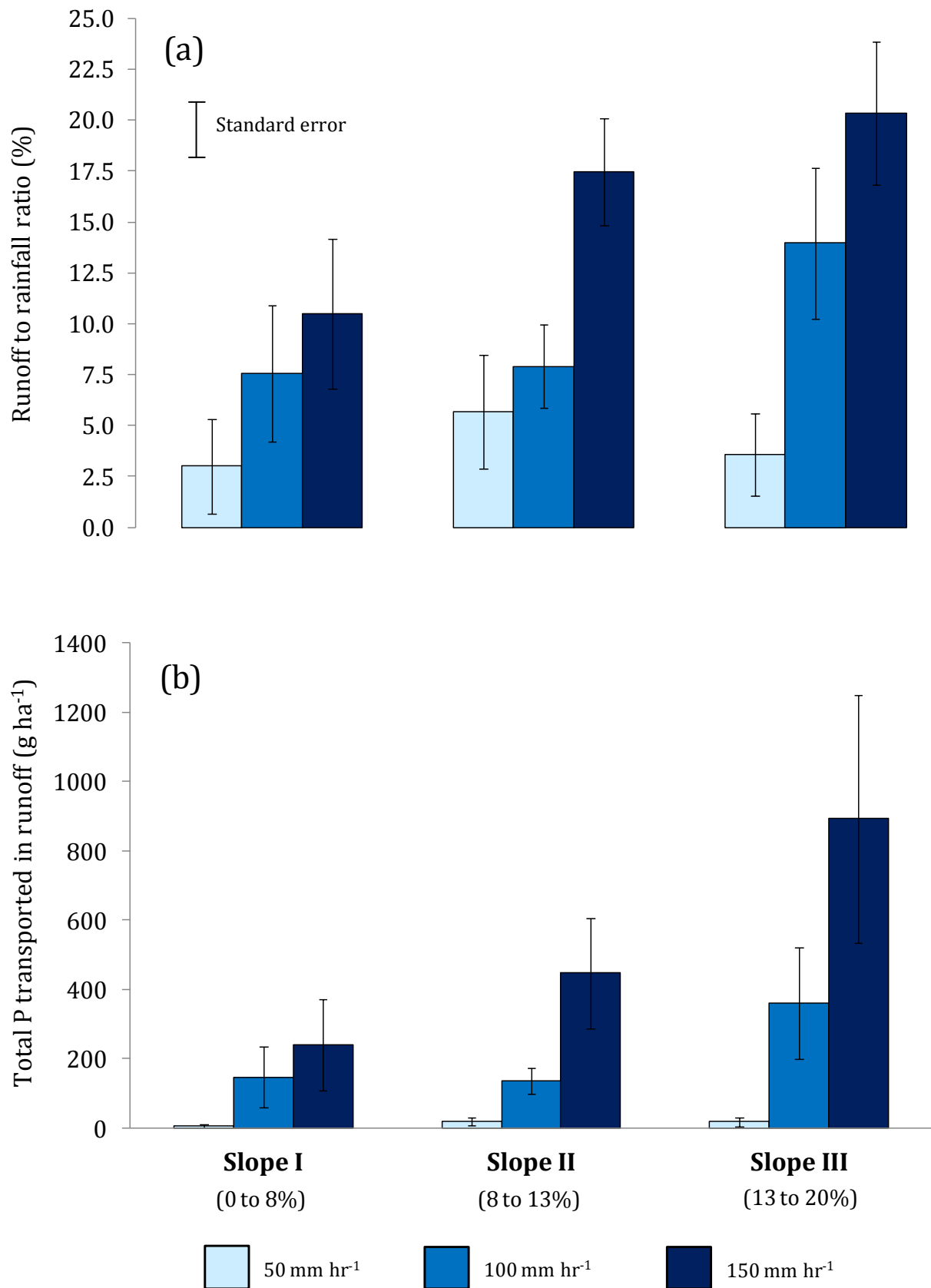


Figure 7.3: Effect of slope on (a) the percentage of runoff to rainfall and (b) on the amount of total P transported in runoff waters at rainfall intensities of 50, 100 and 150 mm hr⁻¹.

From Figure 7.3, it can also be observed that both the amount of runoff generated and the runoff P loads increased with increasing rainfall intensities. Generally, soil nutrient losses are greater under high intensity rainfall due to larger runoff volumes (Kleinman *et al.*, 2006). The effect of rainfall intensities on P transport from fields is further discussed in Chapter 8. Furthermore, a high variation in the results for the different replicates, as indicated by the standard error bars in Figure 7.3, implies that apart from field topography, there are other variables (e.g. soil properties, soil P levels) influencing runoff generation and related P losses.

The major abstraction from rainfall during surface runoff generation is infiltration and depends upon several soil properties (e.g. soil bulk density, saturated hydraulic conductivity) and site characteristics (e.g. field topography, landscape hydrology) which evidently affects surface runoff production from a landscape (Haggard *et al.*, 2005). Kleinman *et al.* (2006) pointed out that most rainfall simulation studies simulate *infiltration excess runoff*, which occurs when rainfall rate exceeds soil infiltration rate. Therefore assuming that *infiltration excess runoff* is the main mechanism for surface runoff, the permeability classes of the different soil types at three slope categories were estimated from the runoff volumes (i.e. the difference between the volume of rainfall applied and total runoff volume was assumed to correspond to the volume of infiltrating water). The results are presented in Section 9.4.2 and from Table 9.6, it can be noted that soil permeability decreased with increasing slopes. As explained by Ghildyal and Tripathi (1987), on steep slopes the fast movement of water allows little time for infiltration while on gentle slopes, , infiltration is more readily occurs due to the slow movement of water.

The research findings reflect the importance of implementing best management practices on sloping lands so as to minimize runoff P mobilization. For example, measures that slow down flow velocity, such as grassed waterways, vegetated buffer zones can be implemented on sloping lands so as to reduce the risks of surface runoff, but such measures may exacerbate subsurface losses of P (Bechmann, 2005). In fact, vegetation intercepts the erosive nature of falling raindrops retarding the amount and velocity of surface runoff and thereby favours water infiltration into the soil profile. Cover crops and mulching with crop residues protect the soil surface from raindrop impact and also enhances water infiltration through the soil profile and hence decreases runoff volume (Singh and Lal, 2005).

7.3.3 Effect of slope on forms of P transported during surface runoff

The proportion of sediment-bound and soluble P forms in runoff waters is illustrated in Figure 7.4 and it is evident that sediment-bound P is the predominant form of P transported irrespective of field slope and soil P levels. As pointed out by Pierzynski *et al.* (2000), the fraction of sediment P

loss depends not only on rainfall characteristics but also on the topography of field landscape. While Hart *et al.* (2004) stipulated that the literature contains many conflicting generalisations about predominant forms in runoff and that the major pathway for P transport from agricultural soils is through soil particulate loss during runoff events, Dougherty *et al.* (2008) reported that small runoff plots (similar to those utilised for this study) have hydrological characteristics different from those of larger landscape, and may therefore alter the relative importance of particulate transport. However, studies done under natural rainfall at a watershed scale in Mauritius have also concluded that particulate P is indeed the predominant form of P transported in runoff waters (Ng Kee Kwong *et al.*, 2002).

In Figure 7.4, it is noted that there was a slight increase in the proportion of sediment-bound P transported as the slope class increased, that is the sediment-bound P increased from 85% at slope class I to 90% for slope class II and to 92% for slope class III. A possible explanation could be that as soil slope increases, runoff velocity increases causing a more turbulent mixing in the zone of interaction and therefore resulting in more soil sediments being detached from the soil surface and carried in the fast flowing runoff waters along the slope. Although sediment-bound P may account for up to 90% of total amount of P transported in runoff, it is recognised that only a portion of this is available for biological uptake in water systems (Sharpley, 1985a). If 20 to 30% of this sediment P becomes available in receiving waters, losses of bioavailable sediment P may be at a similar level to, or even exceed those of the readily available dissolved reactive P from cultivated fields (Ekholm and Lehtoranta, 2012). While sediment-bound P may not be immediately bioavailable, it nonetheless represents a long-term P pool if deposited at the bottom of water bodies and therefore it is important to adopt appropriate measures to mitigate not only soluble P losses but particulate P losses as well. The implementation of management practices such as grassed waterways or riparian buffers at the edge of fields as well as the presence of crop residues in fields are efficient in trapping sediments from runoff waters and hence in reducing sediment concentration along the entire field slope.

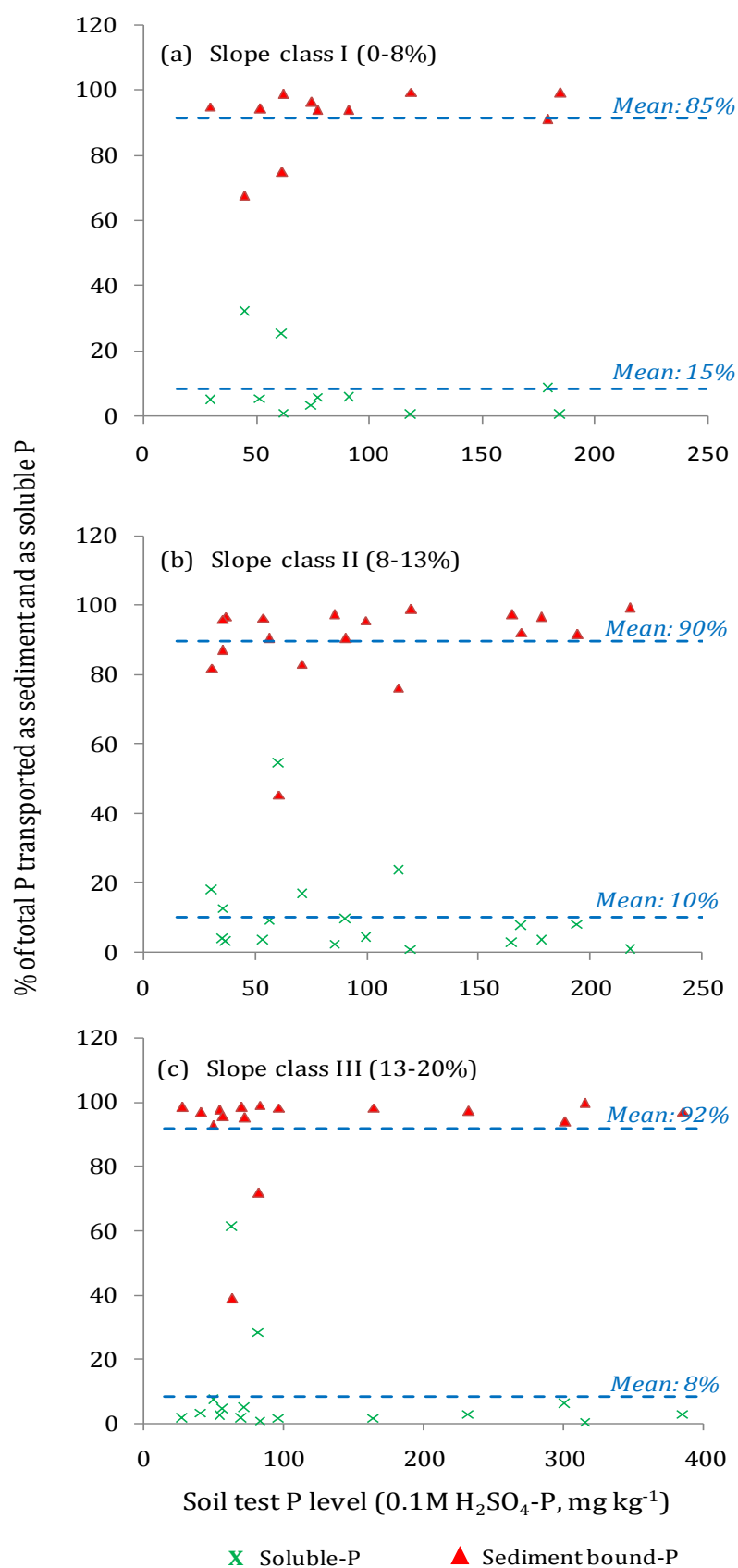


Figure 7.4: Proportion of soluble and sediment-bound P mobilised in runoffs during 30 minutes simulated rainfall intensity of 150 mm hr⁻¹ from runoff plots with slope classes (a) 0 to 8%, (b) 8 to 13% and (c) 13 to 20%.

7.4 Conclusion

The research findings emerging from this study have clearly shown that field topography plays an influential role in runoff generation, erosion and consequently P mobilisation from field landscapes. Indeed, transport of P from agricultural fields is enhanced with increasing field slopes on account of larger runoff volumes and a greater amount of eroded soil and sediments. It is therefore imperative to implement suitable management practices so as to control agricultural P transport from sloping landscapes. Such management practices as discussed in this chapter could include riparian buffers, grassed waterways, mulching with crop residues and cover crops. Moreover, it has been observed that apart from topography, there are other factors which affect the extent of P mobilisation and it is thus important that an integrated approach be adopted for any P loss assessment tool to adequately estimate P losses from fields.

It should also be borne in mind that small runoff plot studies such as those done in this study cannot fully reproduce flow processes occurring over a landscape or hillslope and are usually limited to elucidating flow-soil P interdependencies (Sharpley and Kleinman, 2003). Indeed, small plots have hydrological characteristics different to those of larger landscape and therefore further work is needed on longer slopes at watershed scales so to consolidate these results obtained at small plot scales.

8 Influence of rainfall on phosphorus movement from soils cropped with sugarcane in Mauritius

8.1 Introduction

The mean annual rainfall over the island as a whole is in the order of about 2,000 mm (National Water Research Group, 2012). Although Mauritius occupies an area of 1,860 km², there is owing to its topography a high variation in rainfall over very short distances. Indeed, Mauritius can be divided into three agro-climatic rainfall zones, namely the sub-humid, humid and super-humid. The subhumid zone covers the low rainfall (< 1,600 mm annually), high temperature region confined to the coastal belt which is more important on the leeward side, it covers 19.4% of the island. The humid zone is an intermediate rainfall region (1,600 to 2,600 mm annually) between the coastal belt and the centre of the island, representing 34.7% of the island. Finally, the super-humid zone is a high rainfall (> 2,600 mm annual rainfall) and relatively low temperature region, in the higher parts of the island's centre and covers 45.6% of the island. Concerning the distribution of the rainfall over the year, about 70% of the total rainfall occurs in the summer months of November to April. October is the driest, with only 3.5% of the total, while March is the wettest with 15% of the total. The monthly average increases from November through March, decreases sharply in April and May, and then decreases more gradually until October.

Surface runoff, as explained by Kleinman *et al.* (2006) may be generated by two non-exclusive mechanisms, namely by infiltration excess surface runoff which occurs when the rainfall rate exceeds soil infiltration rate and by saturation excess surface runoff happening when the water table rises to the soil surface and the soil water storage capacity is exceeded. The occurrence of infiltration excess surface runoff is limited to rainfall periods whereas saturation excess surface runoff may occur during and immediately after rainfall events (Srinivasan *et al.*, 2002). Rainfall intensity undoubtedly affects surface runoff generation and soil erosion which eventually influences transport of P from fields (Kleinman *et al.*, 2006). Actually, studies done in Mauritius at a catchment scale showed that differences in nutrient transport were caused by differences in rainfall distribution and intensity indicating that rainfall is the primary parameter that determines whether or not there will be nutrient mobilisation from the sugarcane fields (Sallaway *et al.*, 2001). Sharpley (1985b) showed that rainfall intensity is positively related to the effective depth of interaction between soil and runoff water, thus affecting desorption of P from surface soils to runoff waters, hence the necessity of studying the impact of different rainfall intensities on P mobilisation.

The utilisation of rainfall simulators on small plots to generate surface runoffs is widely used for studying the processes and quantifying the mobilisation of P from different soil-plant systems into surface runoff since it provides an accurate and efficient means of collecting data and testing hypotheses. In addition, it allows greater control of the experimental conditions and also negates the need for expensive and often lengthy monitoring exercises under natural rainfall on catchment scales (Dougherty *et al.*, 2008). The use of rainfall simulator is further motivated by the fact that precipitation is the main driver of transfer processes for nutrients and soil particles in the agricultural landscape (Bechmann *et al.*, 2005). Hence, in view of the significant influence which rainfall intensity can have on the magnitude and the potential risk of soil P movement from sugarcane fields particularly those cultivated on the sloping lands, a study was conducted using simulated rainfall, with the objective of evaluating P losses by surface runoff and soil erosion at three different rainfall intensities (50, 100 and 150 mm hr⁻¹) in Mauritius. In so doing, the need for improved P management practices in sugarcane fields particularly those located on the sloping lands in the high rainfall regions of Mauritius can be assessed.

8.2 Research procedure

In order to study the effect of rainfall intensity on P mobilisation, simulated rainfall runoff studies were carried out on 20 sites having slopes ranging between 8 to 13% on sugarcane fields in Mauritius following the protocol of the National P Research Project (2001). In addition, the studies were conducted with the soils at field capacity to eliminate antecedent soil moisture as a variable and it is at field capacity that the soil would be most prone to mobilisation. The soils at the 20 study sites were not too dissimilar in properties being silty clay or silty clay loam in texture at the top 15 cm layer with an organic matter content of 3 to 6%.

A rainfall simulator similar to that described by Loch *et al.* (2001) was used to simulate rain at three different intensities (50, 100, 150 mm hr⁻¹) with each simulated rainfall event lasting 30 minutes. As pointed out in Section 3.3.1, rainfall intensities of 50 and 100 mm hr⁻¹ were chosen because they are representative of rainfall intensities generally encountered over the island (Seeruttun, 2006). Rainfall intensity of the order of 150 mm hr⁻¹ was also studied since such high intensities may be measured during cyclonic periods and torrential rains. The simulated rainfall was applied to 2.1m by 0.75m runoff plots which were established in duplicate with their longer side in the direction of the slope and along the cane rows. The runoffs generated during the 30 minutes simulated rainfall event were collected and the volumes measured. The soil eroded during that time period of 30 minutes (henceforth referred to as bedload) was also collected, dried in an

oven at 105°C and the resulting dry weight was computed to estimate the weight of soil loss per unit area.

Total P in the runoff samples were determined as per the standard methods adopted for the examination of water and waste water by the American Public Health Association (1992). The orthophosphate P concentration was determined in the 0.45 µm filtered runoff samples by the method of Murphy and Riley (1962). The agronomic P levels of the soils sampled at a depth of 0 to 15 cm were determined using 0.1M H₂SO₄ as extractant by shaking 1 g of the air-dried soil (< 2 mm) with 50 ml of 0.1M H₂SO₄ for one hour (Cavalot *et al.*, 1988). The sediment bound P was calculated as in Shigaki *et al.* (2007) by the difference between total P in the unfiltered runoff samples and orthophosphate P in the filtered runoffs. More precision about the technical details of the analytical procedures are given in Sections 3.4 and 3.5.

8.3 Results and discussion

8.3.1 Effect of rainfall intensity on soil P transport

The influence of rainfall intensity on the field scale movement of P at the 20 sites differing in soil P status in Mauritius is shown in Figure 8.1. As expected, increasing the rainfall intensity generally enhanced the total amount of P moved. This increase in P lost, as explained by Sharpley (1985b) can be attributed to an increase in the effective depth of interaction between surface soil and rainfall-runoff which occurs as a result of more turbulent mixing in the zone of interaction. The occurrence of 'no runoff events' and by corollary the absence of soil movement was highest at the lowest rainfall intensity of 50 mm hr⁻¹ studied. Indeed on the 20 field sites, 14 instances of 'no runoff events' were recorded with 50 mm hr⁻¹ rainfall intensity while only three and zero instances of 'no runoff event' were found at 100 mm hr⁻¹ and 150 mm hr⁻¹ respectively. It must be emphasised that rainfall intensity of the magnitude of 150 mm hr⁻¹ is only encountered on certain occasions, namely during a cyclonic event whereas, under the generally prevailing rainfall events, the rainfall intensities do not exceed 100 mm hr⁻¹ (Seeruttun, 2006). The data obtained thus showed that mobilisation of P under the rainfall conditions existing in Mauritius would barely exceed 1.3 kg ha⁻¹ and would be limited to 2.3 kg ha⁻¹ with the occurrence of cyclonic events (Figure 8.1).

As also seen in Figure 8.1, for any of the three rainfall intensities studied, the soil test P level showed no direct correlation with the amount of total P moved from the runoff plots. This result could be explained by the fact that recent P additions (in this case mineral P fertilisers) to soils

have eliminated the relationship between soil P levels and runoff P concentrations as explained by Vadas *et al.* (2005). There is nonetheless a growing interest in using agronomic soil P tests as an indicator of the potential for P loss from agricultural fields, and methods such as the 0.1M H₂SO₄ extraction used in Mauritius developed to assess the availability of the soil P to sugarcane could provide an indication of the potentially mobile P from the fields. Indeed as pointed out by Gartley and Sims (1994), concern about the relationship between soil P and degradation of surface water quality has created a demand for several soil testing laboratories in the United States to expand beyond their traditional scope to eventually offer an environmental interpretation of the analytical results. The rationale is that soils with higher test P levels represent a greater risk of P loss during erosion and surface/subsurface runoffs. However the present research findings point towards the caution which must be exercised in the adoption of that rationale, for example a field with a soil test P level of 41 mg kg⁻¹ 0.1M H₂SO₄ extractable P had been shown to have more P moved than one with a soil test P of 296 mg P kg⁻¹ at the rainfall intensity of 150 mm hr⁻¹ (Figure 8.1). This shows that the effect of soil test P level could be masked by other factors such as soil erosion susceptibility, field topography and soil permeability and therefore soil test P levels should not be considered in isolation from those other factors identifying fields with a potential for P movement.

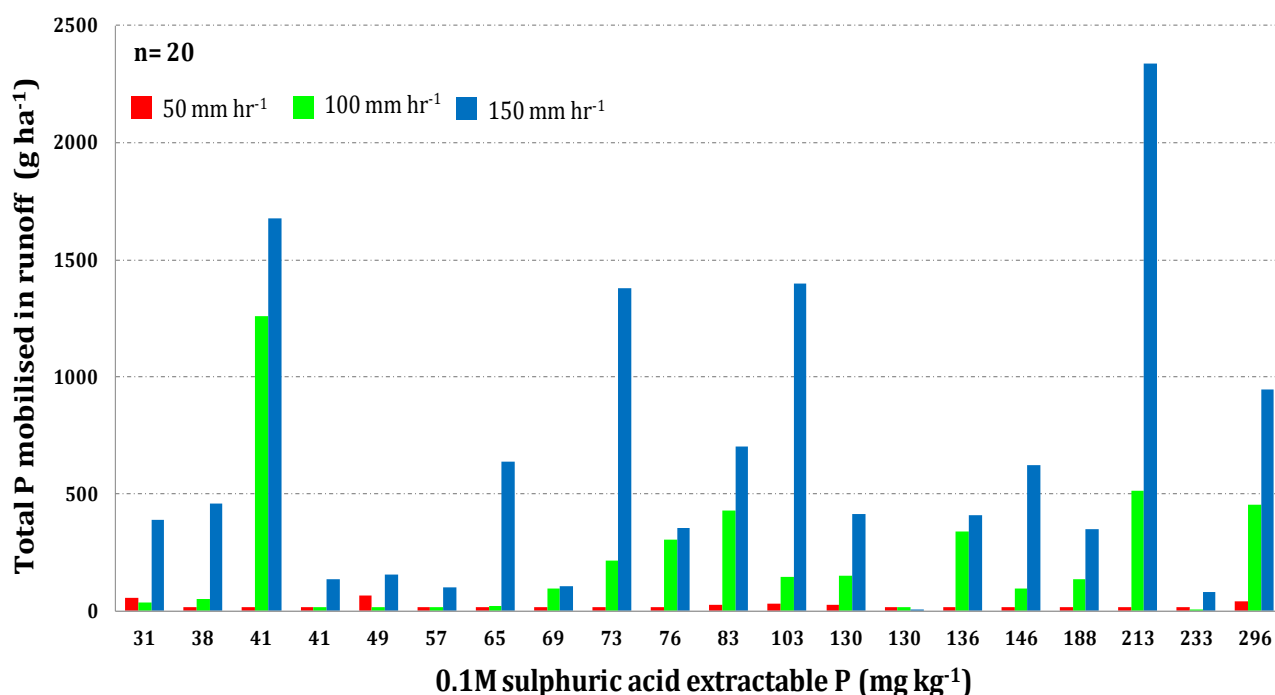


Figure 8.1: Influence of rainfall intensity on the movement of P from soils cropped with sugarcane on field slopes ranging between 8 to 13% in Mauritius.

8.3.2 Effect of rainfall intensity on forms of P moved

As illustrated in Figure 8.2, the P was moved mostly not as soluble P but attached to the sediment in the runoff and on average 95% of the total P transported in the runoff after the rainfall events of 30 minutes occurred as sediment bound P. Moreover, it was also observed that rainfall intensity did not influence the forms of P mobilised during runoff and irrespective of the rainfall intensity, the major portion of P that was mobilised from the sugarcane soils of Mauritius occurred mostly as sediment bound P. In contrast, Sporre-Moeny *et al.* (2004) found that high intensity rainfall resulted in a greater portion of total P in runoff to be particulate P than low-intensity rainfall. In this study, the soluble P transported in the runoff ranged between 1.3 to 70.4 g ha⁻¹ as compared to 8.0 to 2339.2 g ha⁻¹ sediment P moved during the 30 minutes rainfall event for the highest rainfall intensity simulated (i.e. 150 mm hr⁻¹). As soluble P in runoffs is determined by soil P desorption which increases as sorbed P (as reflected by the soil test P level) accumulates in the soil following P additions, the present study thus tends to show the relatively minor importance of this desorption mechanism compared to the physical movement of sediment in the transport of P from fields in Mauritius.

These research findings give further credence to previous results obtained by Ng Kee Kwong *et al.* (2002) which showed that soil P movement to water-bodies in Mauritius occurred mainly attached to soil particles during runoff. They also showed that sugarcane planted during the months of January to March (long season plant cane) in Mauritius when cyclonic events can be expected, the presence of a surface mulch such as legume crop residues is of prime importance to limit the movement of agricultural P from fields during that short period when the foliar canopy of the sugarcane has not yet developed to minimise the impact of the raindrops on the soil surface. In ratoon cane, there will be little risk of P movement during the cyclonic events which may occur during January to March since, with green cane harvest, a trash mulch would already be present on the soil surface.

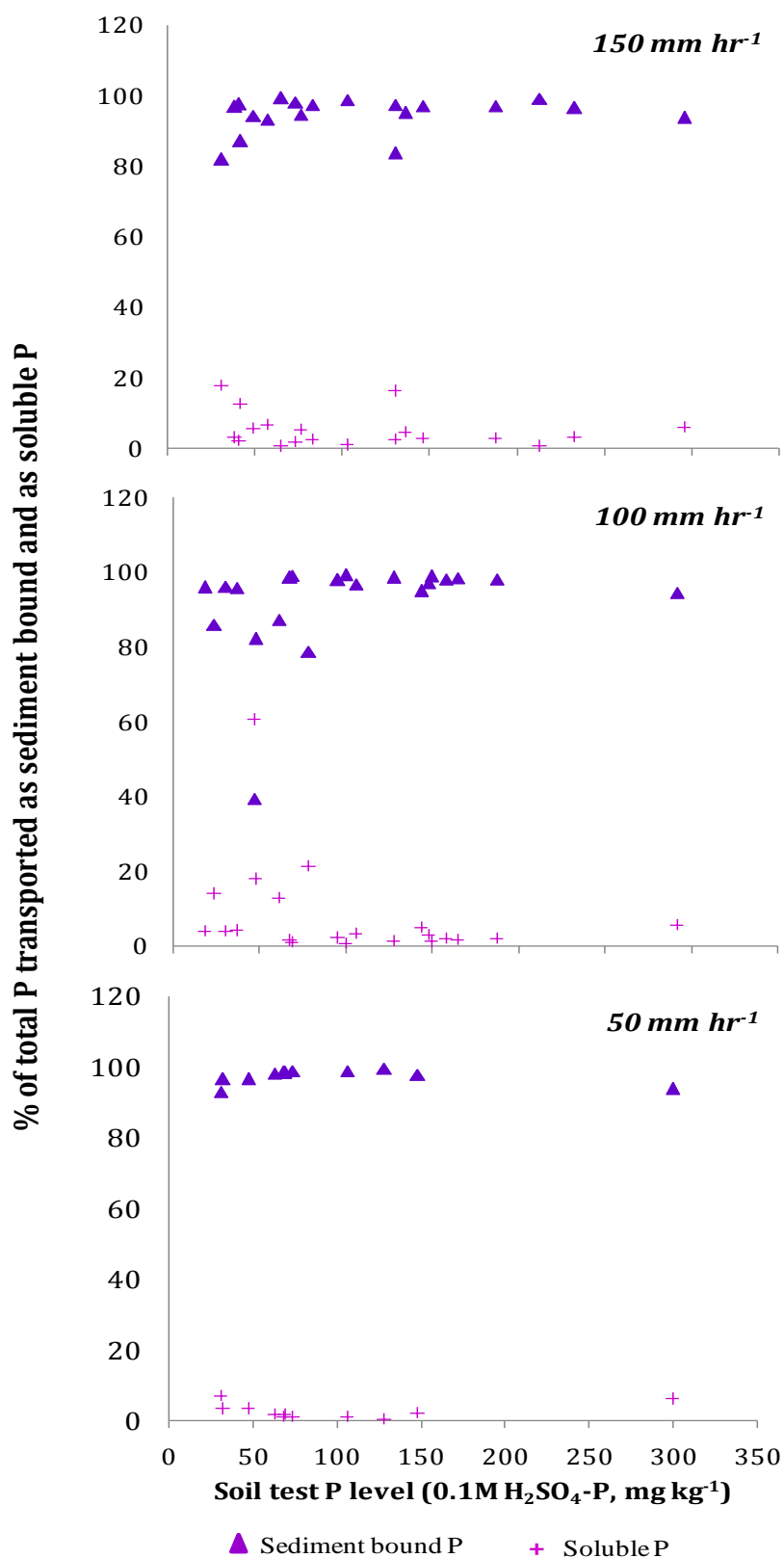


Figure 8.2: Forms of P (soluble or sediment bound) mobilised in runoff during 30 minutes simulated rainfall intensities of 50, 100 and 150 mm hr⁻¹ from runoff plots with slopes ranging between 8 to 13% in Mauritius.

8.3.3 Effect of rainfall intensity on runoff generated

As mentioned in Section 8.1, rainfall intensity is known to affect surface runoff generation as well as concentration of P in runoff (Kleinman *et al.*, 2006). The influence of rainfall intensity and the necessity of minimising the impact of raindrops on the soil surface to reduce P transport from sugarcane fields in Mauritius is shown in Figure 8.3. As pointed out by Dougherty (2006), increasing rainfall intensity generally leads to an increase in runoff rate as also the case in Figure 8.3.

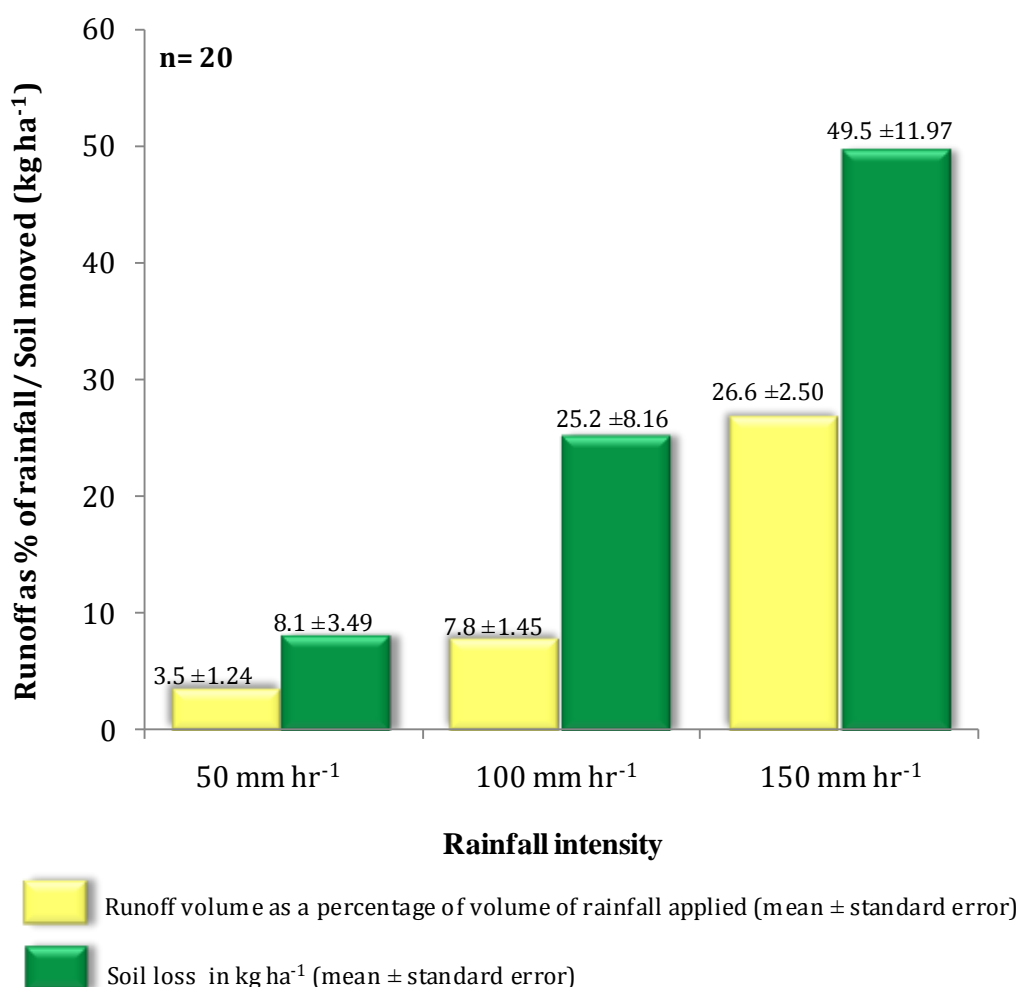


Figure 8.3: Influence of rainfall intensity on the amount of soil moved (kg ha⁻¹) and on the volume of runoff (expressed as a % of the volume of simulated rainfall applied) generated from soils on field slopes ranging between 8 to 13% in Mauritius.

Runoff volume as a percentage of rainfall applied was enhanced nearly eight-fold when the rainfall intensity was increased from 50 to 150 mm hr⁻¹. The amount of soil loss also increased nearly six-fold when rainfall intensity was raised from 50 to 150 mm hr⁻¹. Though only approximately 10% of the total P moved as soluble P in the runoff, the data in Figure 8.3 moreover showed that the

control of runoff volume is critical to limit sediment P transport. In the present study, at high rainfall intensity the volume of water infiltrating into the soil decreased to generate more surface runoff. Adoption of cultural practices such as mulch cover should not only be aimed at reducing sediment loads in runoff waters but must also favour water infiltration at the expense of surface runoff if tangible effect has to be obtained in the effort to reduce P movement from fields. Other transport-oriented best management practices which centre on improving rainfall infiltration into soils, reducing effective slope length to control runoff energy, increasing soil cover and roughness to minimize raindrop impacts and intercepting runoff waters such as conservation tillage practices, setting up of artificial drainage systems, riparian buffers and grassed waterways could also be adopted (Sims and Kleinman, 2005).

8.4 Conclusion

In providing information on the effect of varying simulated rainfall intensity on the processes of P mobilisation in soils cropped with sugarcane in Mauritius, this study has confirmed that the agronomic soil test P which is increasingly being adopted to indicate potential P movement in soil is on its own an inaccurate pointer of the threat which soil P represents to freshwaters. Results obtained also showed that on average 95% of the total P was transported as sediment-bound forms and was not affected when rainfall intensity was increased. In contrast, runoff volumes as well as the amount of soil loss increased when the rainfall intensity was raised from 50 to 150 mm hr⁻¹. The amount of P transported in the runoff waters at the highest rainfall intensity studied (150 mm hr⁻¹) ranged between 1.3 to 70.4 g ha⁻¹ for dissolved P while sediment-bound P varied between 8.0 to 2339.2 g ha⁻¹.

Moreover, in showing that the amount of P moved increases with rainfall intensity and that the P is mobilised mostly attached to soil particles irrespective of the rainfall intensity, the results of the study also indicate that to prevent the risks of surface water impairment due to sediment losses, management practices such as trash cover should be targeted at reducing the impact of the raindrops reaching the soil surface. Those management practices should not only reduce the load of sediment in the runoff but also diminish runoff volume by favouring drainage and infiltration into the soil. The results obtained in this chapter point towards the importance of including a precipitation factor in the P index management tool so as to reflect the bearing of varying rainfall intensities on P mobilization from sugarcane fields.

9 Integrating source and transport factors to derive an index for assessing risks of phosphorus mobilisation from sugarcane fields in Mauritius

9.1 Introduction

The P index, as discussed in Section 2.5.4 is a simple approach developed to rank the potential for P losses from agricultural fields by identifying areas where sources of P coincide with high risks of P transfer to freshwaters. As reviewed by Berzina and Sudars (2010), there is no standard P index and as a result there exists different versions of the P index which have been tailored to account for specific regional conditions (e.g. Gburek *et al.*, 2000; Weld *et al.*, 2001; Sharpley *et al.*, 2003; Bechmann *et al.*, 2005). In comparison to the original P index developed by Lemunyon and Gilbert (1993), subsequent versions of the P indices separately evaluate the P source and transport characteristics of a field and combine them in a multiplicative manner. Berzina and Sudars (2010) divided the process design of the P index into four broad steps, where the first step involved the identification of factors to be included in the P index framework, followed by assigning factor values and subsequently assigning risk level of each value with a numerical value (e.g. very low (0), low (2), medium (4), high (6) and very high (8)) for each factor before eventually defining the structure of the P index (Figure 9.1).

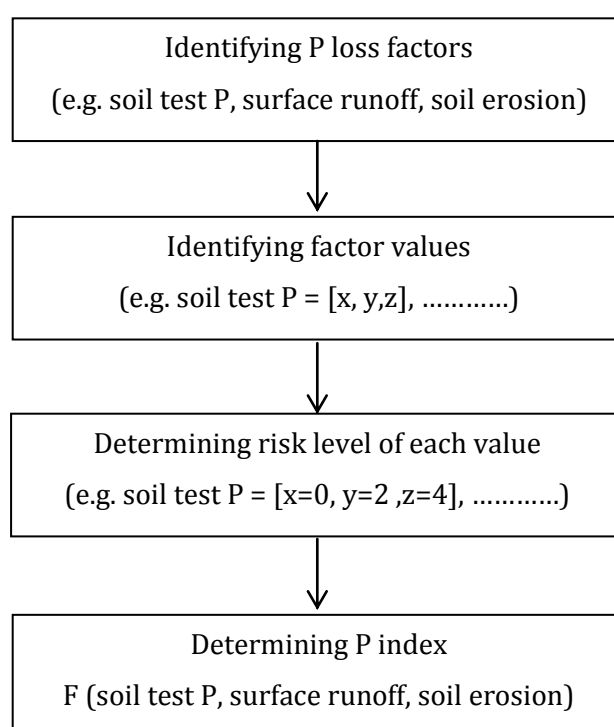


Figure 9.1: Process of P index design (adapted from Berzina and Sudars, 2010).

This chapter integrates the research findings obtained from the rainfall simulation tests which have been discussed in preceding chapters into an index for assessing P mobilisation from sugarcane soils in Mauritius. In addition, the source and transport factors considered in the P index are discussed in relation to local climatic conditions and sugarcane cultivation in an attempt to justify their inclusion in the proposed P index.

9.2 Description of the proposed P index for sugarcane fields of Mauritius

As stated above, the proposed P index incorporates both source and transport factors. Source factor accounts for the amount of P in the soil which is available for transport. It is thus a chemical entity, while the transport factor on the other hand describes the hydrological processes which transform potential P sources into actual P losses from the field (Gburek *et al.*, 2000). Five source variables have been identified for the P index for sugarcane fields in Mauritius and include dissolved P (estimated from soil test P), particulate P (estimated from soil erosion), P application rate, the method of application and application timing. The transport variables in the P index are soil erosion, surface runoff potential and best management practices have been included to account for the benefit which the latter would have on reducing P losses.

The source and transport factors of the proposed P index and the calculations to quantify each of them are discussed in Sections 9.3 and 9.4 respectively. The P source and transport variables will then be combined in an additive or a multiplicative manner as suggested by Gburek *et al.* (2000) to obtain the overall P index for a field, which will then be interpreted based on a site vulnerability chart as described in Section 9.6.

9.3 Source factors

9.3.1 Dissolved P

Studies carried out by Sharpley (1985b) have shown that the loss of dissolved P in surface runoff is highly dependent on the P content of surface soil (about 0 to 5cm) and therefore the inclusion of soil P status in the proposed P index is justified. Although environmental soil P methods such as water extractable P, 0.01M calcium chloride extractable P, bioavailable P and soil P saturation, more accurately quantify soluble P transport, the use of agronomic soil P tests are preferred in P indices because agronomic soil P tests are well established procedures within nutrient management plans and are common to extension officers and farmers. However, it should be borne in mind that the existing agronomic soil test was developed to predict crop performance

rather than for water quality protection purposes and must therefore be re-interpreted so as to address those environmental concerns (Sharpley *et al.*, 2003).

Indeed many studies (e.g. Sims *et al.*, 2002; Carmo Horta and Torrent, 2007) have shown that traditional agronomic soil tests such as Mehlich-3P, Olsen-P and Bray-1P are highly correlated with P in runoff, particularly dissolved P, hence there does not seem to be a need to add a new and often more time-consuming soil P test (Sims and Kleinmann, 2005). To that end, as documented in Chapter 5, the scope of the agronomic soil P test adopted for sugarcane in Mauritius was extended such that it can also be adopted to provide an indication of the environmental soil P status. The standard agronomic method adopted in Mauritius to formulate P fertiliser recommendations is based on the 0.1M sulphuric acid extraction as indicated in Section 4.6.

Soil testing for fertiliser recommendations is performed every time the sugarcane fields are replanted, that is after a crop cycle of six to seven years and no P fertilisers is recommended for soils having a P status greater than 80 mg kg⁻¹. The research findings presented in Chapter 5 have shown that soils with P concentrations above 160 mg kg⁻¹ may impair runoff water quality. In the proposed P index (Table 9.2), dissolved P losses will be estimated using the exponential relationship shown in Figure 9.2.

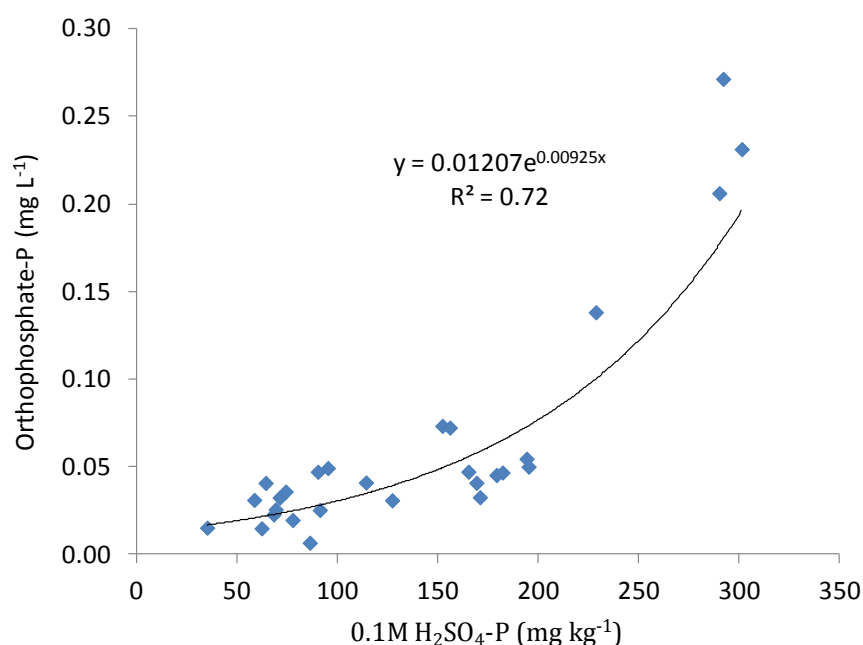


Figure 9.2: Relationship between soil test P (i.e. the 0.1M H₂SO₄-P) and the orthophosphate-P concentrations in runoff.

Another important consideration during soil testing is the soil sampling protocol especially with respect to soil sampling depth. For agronomic purposes, the root zone must be sampled and sugarcane roots may reach depths of more than 90cm but as a greater volume of the roots is concentrated in the first 45cm of the soil profile, it is this zone which is usually sampled (STASM, 2003). On the contrary, the zone of interaction of runoff waters with most soils is normally less than 5cm and Sharpley (1985b) found that such shallower depths more accurately estimated potential surface runoff P enrichment. However, for practical reasons, it is preferable to use the current agronomic sampling depth rather than to incur additional workload and costs associated with different soil sampling depths (Heathwaite *et al.*, 2003). Nonetheless, caution must be taken during the interpretation of the risks of soil P losses because the soil P status over the standard soil sampling depth for fertiliser recommendations (e.g. 0 to 45 cm soil) may underestimate the potential for P losses in runoff waters (e.g. 0 to 5 cm soil; Withers *et al.*, 2007).

9.3.2 Particulate P

The rainfall simulation studies undertaken in Mauritius have shown that particulate P accounts for a significant portion (on average 90%) of total P in runoff waters and therefore a particulate P component in the P index framework is warranted. As discussed in Section 6.3.3, contrarily to dissolved P losses, particulate P losses during runoff are inaccurately predicted from soil P tests and are best estimated from erosion rates (Ekholm *et al.*, 2005). Therefore in the proposed P index, particulate P, as shown in Table 9.2 is represented as a function of erosion using the relationship documented in Figure 9.3 (i.e. slope of linear relationship between soil loss and particulate P in runoff), whereby the loss of particulate P will be estimated as a product of the erosion rate and the particulate P coefficient of 5.5 (corrected for the units of erosion as tonnes ha⁻¹). The typical erosion rates for the different soil types will be obtained from erosion prediction models as discussed in Section 9.4.1.

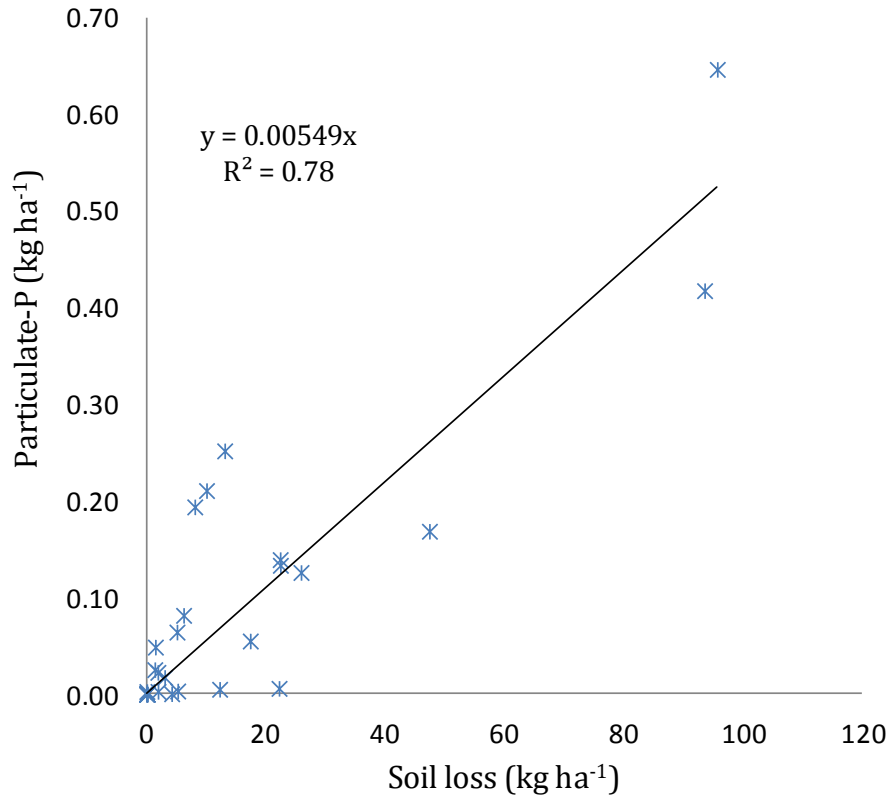


Figure 9.3: Relationship between amount of soil loss and particulate P transported in runoff waters.

9.3.3 Phosphorus application to sugarcane fields

Soil P, as discussed in the previous section is important in determining potential P losses in surface runoff but the fresh applications of P sources (e.g. fertilisers, manures, poultry litter) can override the effect of soil P concentration on surface runoff P (Sharpley *et al.*, 2001b). Those fresh applications of P may cause as mentioned in Section 2.4.4, ‘incidental’ runoff P losses when rainfall interacts directly with those P sources when applied onto the soil surface. As pointed out by Withers *et al.* (2003), the rates of P losses depend on the amount of P applied, method of P source placement and the timing of storm events after application. Therefore these factors are included in the proposed P index framework for sugarcane soils of Mauritius (Table 9.2).

The rates of P fertiliser application as shown in Table 4.4, are based on soil analysis of samples taken just after land preparation. For sugarcane cultivation in Mauritius, P fertilisation at planting is an important practice as all P requirements are applied for the whole crop cycle of six to seven years. Annual supplementation of P in the ratoons on the soil surface is only (and rarely) envisaged as an emergency approach (Parish, 1964). In addition to mineral fertilisers, it is a common practice to apply scums in the furrows before planting at a minimum amount of 12 t ha⁻¹

for a homogenous coverage in the furrow and to have a good crop yield response (STASM, 2003). The different sources of P available for application to sugarcane fields in Mauritius have been discussed in detail in Section 4.5.2. In the suggested P index, the application rate refers to the amount of P ($\text{kg P}_2\text{O}_5 \text{ ha}^{-1} \text{ yr}^{-1}$) that will be applied to sugarcane fields. The application method will include four categories, namely buried in furrows at a depth greater than 20cm, buried in furrows at a depth less than 20cm, surface applied with cane trash and surface applied on bare soil. The risk of associated P losses is lower with the placement of P fertiliser in the furrow at plantation compared to surface application as reflected by the P loss rating values in Table 9.2. In fact studies done by Mueller *et al.* (1984) have shown that significant P losses occurred during runoff events when P sources (e.g. manure) were surface applied in fields.

Timing of P application to soils relative to when runoff occurs is a key consideration for preventing incidental P losses. In fact, in tropical environments such as Mauritius, one of the most critical drivers of offsite movement of nutrients is the onset of seasonal rainfall (Sallaway *et al.*, 2001). The potential for P loss is greater immediately following P application then declines over time, as the added P gradually interacts with soils and is converted to increasingly recalcitrant forms (Sharpley *et al.*, 2002). Indeed, watershed studies have demonstrated that runoff P loss was greatest during the planting season, a time of intense rains, high P application and minimum crop cover (Burwell *et al.*, 1975). In view of the pertinence of the timing of P applications during P loss-risk assessment, it is included in most P indices and since the timing of a specific rainfall-runoff event is usually not predictable, factors for timing of application are in most P indices related in a more general way to the seasons of the year and merged with factors for application (Buczko and Kuchenbuch, 2007).

According to Halais and Davy (1969), Mauritius can be categorised into three agro-climatic regions, namely the sub-humid (< 1,500 mm annually), humid (1,500 to 2,500 mm annually) and super-humid (> 2,500 mm annually). The mean monthly rainfall distribution for the three agro-climatic regions for the last 10 years shows that the highest rainfall period is from January to March followed by April to August and September to December (Figure 9.4).

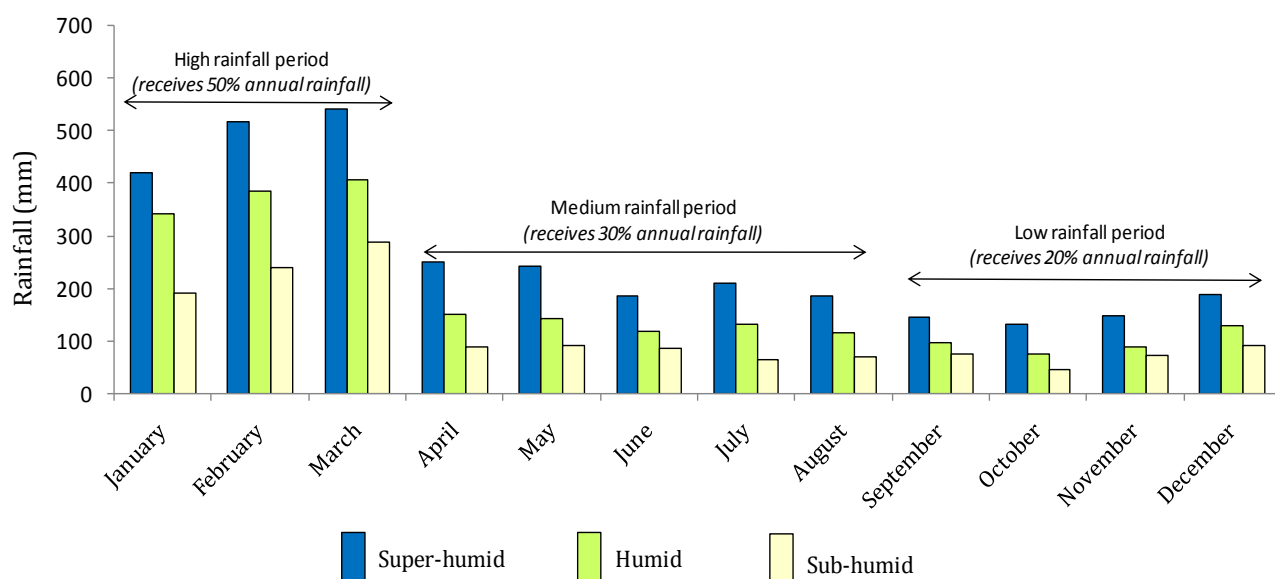


Figure 9.4: Mean monthly rainfall distribution over the three agro-climatic regions of Mauritius (from 2004 to 2013).

In consequence, the highest risks of P losses will occur during the months of January to March, especially if P fertilisers are to be applied during that period and therefore the highest P loss rating value is assigned to that particular period and the relative P loss rating values is based on the percentage annual rainfall during each period (Table 9.1 and Appendix 2).

Table 9.1: Rainfall distribution as a percentage of annual rainfall occurring during each period for the three agro-climatic regions of Mauritius and assigning the relative risk of P application timing on potential for P loss in the P index.

Period	Percentage of annual precipitation				Loss rating value
	Super-humid	Humid	Sub-humid	Mean	
January to March	47	52	52	50	0.5
April to August	34	30	28	31	0.3
September to December	19	18	20	19	0.2

The P application rating which will be obtained by multiplying the P application rate by loss rating values for application method and application timing and the overall P source potential will be the sum of the dissolved P loss rating, particulate loss rating and P application rating as indicated in Table 9.2.

Table 9.2: Phosphorus loss potential due to source factors.

Proposed P source factors	Proposed P loss category	Proposed loss rating value
Dissolved P		$0.01207 e^{0.00925 \times \text{Soil test P (mg P kg}^{-1}\text{)}}$
Particulate P		$5.5 \times \text{erosion (tonnes ha}^{-1} \text{ year}^{-1}\text{)}$
P application		kg P ₂ O ₅ applied per hectare per year
Application method	Buried (depth > 20cm) in furrows	0.2
	Buried (depth < 20cm) in furrows	0.4
	Surface application with cane trash	0.6
	Surface application on bare soil	1.0
Application timing	September to December	0.2
	April to August	0.3
	January to March	0.5
P application rating = (P application rate x method x timing)		
SOURCE POTENTIAL = Dissolved P loss rating + Particulate P loss rating + P application rating		

9.4 Transport factors

9.4.1 Soil erosion

The simulated rainfall-runoff studies undertaken in Mauritius as discussed in Chapter 6 have shown that soil erosion represents a significant mechanism for P transport especially at plant cane, that is at a time in the crop cycle when the field has recently been ploughed and when there is no plant cover. As in most existing P indices, erosion prediction models such as the Revised Universal Soil Loss Equation (RUSLE) described by Hudson (1992) have been utilised to estimate average annual erosion (tonnes hectare⁻¹ year⁻¹). In the proposed P index, the annual erosion will be estimated by RUSLE2 developed by Foster *et al.* (2003) and is calculated as simplified in the equation that follows:

$$A = R \times K \times L \times S \times C \times P,$$

where A is the estimated soil loss per unit area averaged over time and space for a given sugarcane field and is expressed in tonnes per hectare per year.

R is the rainfall-runoff erosivity factor accounting for the rainfall characteristics and the runoff characteristics generated by the rainfall. Studies undertaken in Mauritius by

Seeruttun *et al.* (2006) suggested that a mean value of 300 can be used for R to predict soil losses.

K is the soil erodibility factor which accounts for soil properties that influence soil loss and is thus soil specific. Studies done in Mauritius on erosion plots (22.1m long by 10m wide) under natural rainfall for a period of four years have shown that soil erodibility factors of 0.14, 0.05, 0.08 and 0.01 may be used for soil loss prediction in the Low Humic Latosol (LHL), Humic Ferruginous Latosol (HFL), Latosolic Brown Forest (LBF) and Humic Latosol (HL) soil groups respectively (Seeruttun *et al.*, 2006). For the Latosolic Reddish Prairie soils (LRP), a K value of 0.01 was assumed since those soils usually have high infiltration rates due to the presence of gravels of unweathered basalts (Parish and Feillafé, 1965).

L is the length factor and is a ratio which compares the soil loss with that from a field of specified length of 22.1m.

S is the slope steepness factor that is determined on the field for a given slope length.

C is the cover or cropping-management factor and is used to reflect the effect of cropping and management practices on erosion rates. For sugarcane in Mauritius the C factor is considered as 0.07 (Seeruttun *et al.*, 2006).

P is the conservation support practice that accounts for those practices (e.g. contouring, strip-cropping or terracing) that affect soil erosion by modifying the flow pattern, grade or direction of surface runoff. Values for the P factor were obtained from the Arkansas nutrient management planner's guide (Daniels *et al.*, 2005) and are a function of rainfall energy and intensity factor (EI). The EI values of 100, 110 and 130 were assumed for the sub-humid, humid and super-humid regions respectively.

The soil erosion estimates for the different slopes, soil types and climatic regions can be obtained from Tables 9.3, 9.4 and 9.5 (also see Appendix 3). The estimated erosion obtained from RUSLE2 is used in the P transport potential component to account for P loss with eroded particulate material (Table 9.9).

Table 9.3: Estimated annual soil loss as a function of soil type and slope for the sub-humid region of Mauritius.

Slope (%)	Estimated annual soil loss (tonnes ha ⁻¹ yr ⁻¹)				
	LHL	LBF	HFL	HL	LRP
1	1.08	0.62	0.38	0.08	0.08
2	1.92	1.10	0.68	0.14	0.14
3	2.76	1.58	0.98	0.20	0.20
4	3.62	2.07	1.29	0.26	0.26
5	4.53	2.59	1.62	0.32	0.32
6	5.59	3.20	2.00	0.40	0.40
7	6.71	3.83	2.40	0.48	0.48
8	7.98	4.56	2.85	0.57	0.57
9	9.45	5.40	3.38	0.68	0.68
10	11.03	6.30	3.94	0.79	0.79
11	12.71	7.26	4.54	0.91	0.91
12	14.81	8.46	5.29	1.06	1.06
13	16.55	9.46	5.91	1.18	1.18
14	18.38	10.50	6.56	1.31	1.31
15	19.69	11.25	7.03	1.41	1.41
16	21.00	12.00	7.50	1.50	1.50
17	22.31	12.75	7.97	1.59	1.59
18	23.63	13.50	8.44	1.69	1.69
19	24.94	14.25	8.91	1.78	1.78
20	26.25	15.00	9.38	1.88	1.88

Table 9.4: Estimated annual soil loss as a function of soil type and slope for the humid region of Mauritius.

Slope (%)	Estimated annual soil loss (tonnes ha ⁻¹ yr ⁻¹)				
	LHL	LBF	HFL	HL	LRP
1	1.26	0.72	0.45	0.09	0.09
2	2.44	1.40	0.87	0.17	0.17
3	3.66	2.09	1.31	0.26	0.26
4	4.83	2.76	1.73	0.35	0.35
5	6.04	3.45	2.16	0.43	0.43
6	7.32	4.19	2.62	0.52	0.52
7	8.64	4.94	3.08	0.62	0.62
8	10.08	5.76	3.60	0.72	0.72
9	11.46	6.55	4.09	0.82	0.82
10	12.99	7.43	4.64	0.93	0.93
11	14.44	8.25	5.16	1.03	1.03
12	15.75	9.00	5.63	1.13	1.13
13	17.06	9.75	6.09	1.22	1.22
14	18.38	10.50	6.56	1.31	1.31
15	19.69	11.25	7.03	1.41	1.41
16	21.00	12.00	7.50	1.50	1.50
17	22.31	12.75	7.97	1.59	1.59
18	23.63	13.50	8.44	1.69	1.69
19	24.94	14.25	8.91	1.78	1.78
20	26.25	15.00	9.38	1.88	1.88

Table 9.5: Estimated annual soil loss as a function of soil type and slope for the super-humid region of Mauritius.

Slope (%)	Estimated annual soil loss (tonnes ha ⁻¹ yr ⁻¹)				
	LHL	LBF	HFL	HL	LRP
1	1.31	0.75	0.47	0.09	0.09
2	2.63	1.50	0.94	0.19	0.19
3	3.94	2.25	1.41	0.28	0.28
4	5.25	3.00	1.88	0.38	0.38
5	6.56	3.75	2.34	0.47	0.47
6	7.88	4.50	2.81	0.56	0.56
7	9.19	5.25	3.28	0.66	0.66
8	10.50	6.00	3.75	0.75	0.75
9	11.81	6.75	4.22	0.84	0.84
10	13.13	7.50	4.69	0.94	0.94
11	14.44	8.25	5.16	1.03	1.03
12	15.75	9.00	5.63	1.13	1.13
13	17.06	9.75	6.09	1.22	1.22
14	18.38	10.50	6.56	1.31	1.31
15	19.69	11.25	7.03	1.41	1.41
16	21.00	12.00	7.50	1.50	1.50
17	22.31	12.75	7.97	1.59	1.59
18	23.63	13.50	8.44	1.69	1.69
19	24.94	14.25	8.91	1.78	1.78
20	26.25	15.00	9.38	1.88	1.88

9.4.2 Surface runoff

Surface runoff class is included in the P index to estimate the potential for surface runoff to occur from a field and is based on soil type and field slopes. The common techniques employed in P indices to account for runoff potential; include soil saturated hydraulic conductivity, hydrologic soil groups and runoff curve numbers which describe response of runoff to rainfall (Heathwaite *et al.*, 2005). Field experimental data (i.e. runoff to rainfall ratio) gathered from the rainfall simulation tests to study the effect of topography on P mobilisation (discussed in Section 7.3.2) were utilised to determine the permeability of the five main soils in Mauritius (Table 9.6).

Table 9.6: Soil permeability at three different slope categories for the five main soils of Mauritius.

Soil type	Slope (%)	Permeability (mm hr ⁻¹)	
		Range	Mean± SE
Low Humic Latosol (LHL)	0-8	131 - 148	139± 4
	8-13	84 - 146	119± 13
	13-20	99 - 145	116± 10
Humic Latosol (HL)	0-8	64 - 149	123± 30
	8-13	102 - 121	110± 6
	13-20	66 - 128	88± 20
Humic Ferruginous Latosol (HFL)	0-8	129 - (>150) ^a	-
	8-13	97 - 124	110± 6
	13-20	114 - 121	116± 2
Latosolic Reddish Prairie (LRP)	0-8	85 - (>150) ^a	-
	8-13	94 - 141	127± 8
	13-20	101 - 116	122± 10
Latosolic Brown Forest (LBF)	0-8	125 - 139	125± 10
	8-13	124 - 148	132± 8
	13-20	127 - 147	135± 6

SE: standard error, ^a: Soil permeability greater than the amount of simulated rainfall applied.

Based on these results, the soils were grouped into three permeability classes of slow, moderate and high (Table 9.7). A similar approach to that of the Maryland P index (Anon, 2008) was adopted to estimate the surface runoff potential in the P index for Mauritius and takes into account soil permeability and slope categories, where low permeability coupled with high slope represents the highest risk to surface runoff and P mobilisation (Table 9.7).

Table 9.7: Surface runoff potential as a function of soil type and slope.

Soil type	LRP	HFL	LBF	LHL	HL
Relative permeability class	Rapid	Rapid	Moderate	Moderate	Slow
Slope (%)	Relative runoff risk				
0-8	VL	VL	L	L	M
8-13	L	L	M	M	H
13-20	M	M	H	H	VH
>20	H	H	VH	VH	VH

VL: Very low, L: Low, M: Medium, H: High, VH: Very high

9.4.3 Precipitation

In Mauritius the mean annual rainfall changes abruptly from 800mm on the west coast of the island to over 4,000mm in the central plateau over a distance of only 20km. Since rainfall is appreciably greater in some areas than in others, the risk of runoff and erosion would also be expected to vary and it was therefore imperative to include a precipitation factor in the P index. The precipitation factor for the three different agro-climatic zones (illustrated in Appendix 4), namely sub-humid, humid and super-humid regions is indicated in Table 9.8. The precipitation factor was obtained by normalising the annual rainfall to that occurring in the humid region and thus the precipitation factor is 0.64 for the sub-humid region, 1.00 for the humid region and 1.46 for the super-humid region.

Table 9.8: Precipitation factor for the climatic regions of Mauritius.

Agro-climatic zone	Annual precipitation, mm	Precipitation factor
Sub-humid	1,390	0.64
Humid	2,170	1.00
Super-humid	3,160	1.46

Table 9.9: Phosphorus loss potential due to transport factors.

Proposed P transport factors	Category	Proposed loss rating value
Soil erosion		tonnes hectare ⁻¹ year ⁻¹
Runoff potential	Very Low	0.2
	Low	0.4
	Medium	0.6
	High	0.8
	Very high	1.0
Precipitation factor	Sub-humid region	0.64
	Humid region	1.00
	Super-humid region	1.46
TRANSPORT POTENTIAL = (Erosion + Runoff potential) x Precipitation factor		

9.5 Best management practices multiplier

The adoption of best management practices (BMPs) in reducing P losses will be taken into account by the inclusion of the BMPs multiplier, an approach similar to the one adopted in the Arkansas P index (Sharpley *et al.*, 2010). The multiplier associated with each BMP is calculated as one minus the effectiveness of the BMP implemented and the multiplier for all the BMPs implemented will then be computed as follows:

$$\text{BMPs multiplier} = (1 - \text{Effectiveness}_1) \times (1 - \text{Effectiveness}_2) \times (1 - \text{Effectiveness}_n)$$

If no additional BMPs are implemented, the BMPs multiplier will be 1 and if BMPs are adopted, then the BMP multiplier will have a value less than 1. In addition to the management practices considered in the source and transport components of the P index, other possible BMPs and their effectiveness in decreasing runoff P losses are presented in Table 9.10.

Table 9.10: Credit given for potential best management practices (BMPs) for use in the P index to assess P losses from sugarcane fields in Mauritius (adapted from Sharpley *et al.*, 2010).

Best management practice (BMP)	Credit (%)
Diversion	5
Pond	20
Field border	10
Herbaceous buffer	20
Forest buffer	20
Filter strip	20
Grassed waterway	10
Structure for runoff water control	30
Terrace	10
Wetland creation	20

9.6 Phosphorus index calculation and risk interpretation

The overall P index for a given site will be calculated by multiplying the source potential by the transport potential as outlined below:

$$\text{Overall P index} = \text{Source potential} \times \text{Transport potential} \times \text{BMPs multiplier}$$

Once the P index value is calculated, fields are assigned a P loss risk of either low, medium, high or very high with each of the four classes being associated with an interpretation and guided management practice as shown in Table 9.11. Recommendations range from cautions regarding build-up of soil P levels for the low risk class to no additional P application until soil P levels and P index values are reduced for the very high class.

Table 9.11: Interpretation and recommendations for the P indices of sugarcane soils in Mauritius
(adapted from Sharpley *et al.*, 2001b; DeLaune *et al.*, 2006 and Weld *et al.*, 2001).

P index value	P loss risk	General interpretation and recommendations
< 33	Low	<i>Low potential for P loss.</i> If current farming practices are maintained, there is a low risk of adverse impacts on surface waters.
33 to 65	Medium	<i>Moderate potential for P loss.</i> The chance for adverse impacts on surface waters exists, and remediation measures should be taken to minimise P loss. Use of the P index to identify specific field areas that could represent long-term concerns and the implementation of BMPs are to be considered in those areas.
66 to 100	High	<i>High potential for P loss</i> and adverse impact on surface waters. To determine using the P index if one factor is disproportionately affecting risk of P losses. Inclusion of an appropriate BMP and/or reducing P application to bring P loss risk to medium (<65) is warranted. A conservation P management plan is required with the long-term goal of shifting the P loss risk to the medium or lower category.
> 100	Very High	<i>Very high potential for P loss</i> and adverse impact on surface waters. The adoption of BMPs to decrease this value below the very high category in the short term and a conservation management plan that would reduce the P loss risk to a lower risk category, with the long-term goal of having a P index in the medium or lower category.

9.7 Conclusion

As discussed in this chapter, the P index proposed to evaluate the risk of P losses from sugarcane fields in Mauritius is a qualitative assessment tool ranking field site's vulnerability to P loss. The P index will be part of the planning process before the sugarcane fields are planted and will be applicable for the whole sugarcane crop cycle (i.e. usually one plant cane and six to seven ratoons) as long as all other factors in the P index remain the same for all years in the planning cycle. When management changes occur during the planning cycle, it will be necessary to evaluate if these changes impact the P index and risk of P losses. Moreover, such a planning process would help to select alternative management practices that would reduce the risk of P losses from fields where the potential of P movement is initially high.

The factors proposed in the P index for sugarcane soils of Mauritius have been based on both scientific data (e.g. source factors) as well as past experience (e.g. precipitation factor). Although an accurate watershed model might be more satisfying to the research scientific community, such a model will prove to be too complicated and data intensive for routine application by extension officers, field staff and nutrient management planners (Gburek *et al.*, 2006). Therefore, to ensure an effective and successful adoption of a nutrient management tool, a balance between scientific rigor and end-use practicality was sought during the development of the P index for Mauritius. Evaluation and validation of the proposed P index is necessary to ensure an accurate and efficient nutrient management tool and is discussed in the following chapter.

10 Evaluation of the derived index for assessing risks of phosphorus mobilisation from sugarcane fields in Mauritius

10.1 Introduction

An important prerequisite for the utilisation of the P index as a management tool is to ensure that it reasonably reflects the relative differences in risks of P transfer. Even though the P index established in Chapter 9 is based on field experimental data, some factors (e.g. application method, application timing and precipitation) have been based on empirical knowledge and past historical data and it is therefore important to test the P index. Buczko and Kuchenbuch (2007) pointed out that as the P indexing procedure yields semi-quantitative estimates for the risk of P loss from fields, validation or calibration of calculated PI values is not straight forward. Ideally, a P index should be evaluated against observed P loss data under various scenarios and measured at the point where the runoff from a field reaches a field edge, stream, tile inlet, or other water source (Sharpley *et al.*, 2012). Nelson and Shober (2012) identified six general approaches that have been used to evaluate P indices in the literature which included comparison with other P indices, the use of sensitivity analysis, simulated rainfall runoff from small plots and simulation model data. Although comparisons of P loss ratings between indices are good for identifying differences among P indices, they do not provide an indication of the accuracy of the P index as a risk assessment tool and therefore evaluation of the accuracy of a P index requires validation of P index ratings against a separate independent and valid assessment of P loss (Nelson and Shober, 2012).

Indeed several evaluations of P indices using measured (or simulated) P losses have been conducted at different scales. Eghball and Gilley (2001) and Sharpley *et al.* (2001c) found relatively good correlations between total P losses from simulated rainfall runoff plots and P index ratings. DeLaune *et al.* (2001) carried out field-scale studies under natural rainfall and a strong correlation ($r^2=0.83$) between measured annual P loads and P index ratings was obtained. Long-term water quality monitoring data of different watersheds have also been utilized to validate P indices (e.g. Birr and Mulla, 2001; Bechmann *et al.*, 2007). For instance, Birr and Mulla (2001) obtained a good correlation ($r^2=0.70$) between P index rating and total P stream monitoring data in watersheds.

In the absence of field experimental data, Veith *et al.* (2005) assessed the reliability of the P index to categorize the potential risk of P loss from fields within a watershed by comparing with predictions from the Soil & Water Assessment Tool (SWAT) model which had originally been validated. Furthermore, to avoid extensive field-scale validation of the P index model, the

sensitivity and scenario analyses provide a useful and efficient technique to evaluate which factors of a P index have the greatest influence on the P index rating. It also helps to identify factors which warrant further research so as to improve the accuracy of the P index to identify *critical source areas* in a field. Beaulieu *et al.* (2006) performed sensitivity analysis on the P index model of Québec to identify the site characteristics and input variables impacting most on the calculated P index rating, thereby isolating those P index factors which would require further investigation so as to improve the estimates of P delivery risk to surface waters.

In this chapter, the P index developed to assess P losses from sugarcane soils of Mauritius was evaluated using first the sensitivity analysis, followed by plot-scale field measurements under simulated rainfall.

10.2 Research procedure

10.2.1 Sensitivity and scenario analyses

The sensitivity and scenario analysis was employed to study the behaviour and assess which factors of the established P index have the greatest influence on the calculated P index. This entailed creating a baseline scenario (i.e. typical field characteristics) and varying each P index variable in turn while keeping others constant, so as to isolate the effect of each model component on the P index rating. However, it should be noted that the hypothetical scenarios may at times not be realistic. By quantifying the change in the final site rating from an incremental perturbation in a single input variable (while holding others constant), sensitivity analysis will help identify input variables that have a dominant impact on the P index (Brandt and Elliot, 2005).

10.2.2 Edge-of-plot field testing under simulated rainfall

Simulated rainfall surface runoff studies (as described in Section 3.3) were carried out at ten sites at three different slope categories (i.e. 0 to 8, 8 to 13 and 13 to 20%) with three different rainfall intensities (i.e. 50, 100 and 150 mm hr⁻¹). The field experimental data gathered from those runoff plots were used to evaluate the potential of the established P index as a risk assessment tool at an edge-of-plot scale. This was done by comparing the measured P concentrations in simulated rainfall runoff waters with the corresponding P index ratings of the respective field sites (details of field sites are given in Appendix 5).

10.3 Results and discussion

10.3.1 Sensitivity and scenario analyses

The behaviour of the source component of the P index was examined using the sensitivity analysis and the results are illustrated in Figure 10.1. The P index correctly reflects that with increasing soil P levels, the risks of P losses during runoff events increases (Figure 10.1a). The sensitivity analysis also revealed that the P index ratings would increase with increasing P application rates. Unlike the application rate which is a continuous variable in the P index, the application method and timing have been included as discrete variables as there is a clear-cut distinction among the different P loss categories (refer to Table 9.2). In Figure 10.1b, it is observed that the rate of change in the P index rating is highest when P sources (i.e. fertilisers or organic materials) are surface applied on bare soil and is lowest when buried in furrows at a depth greater or equal to 20cm. From an environmental point of view, it is therefore advisable that P sources be applied during the months of September to December when the risks of P loss is lowest (Figure 10.1c).

Running the sensitivity analysis on the transport component of the P index is more complicated compared to the source component as the transport variables are often interrelated. For example, a variation in field slope will influence both runoff potential and erosion rates and it is therefore difficult to isolate the effect of each variable on the P index rating. Figure 10.2 illustrates how the P index responds to changes in field slopes across the five main soils under sugarcane in Mauritius. The sensitivity analysis shows that irrespective of the soil type, the P index rating increases as the field slopes become steeper. This increase in the P index rating is due to a higher vulnerability of the fields to surface runoff and soil erosion owing to increasing field slopes. The highest erosion rates were obtained for the Low Humic Latosols followed by the Latosolic Brown Forest soils, Humic Ferruginous Latosols, Humic Latosols and Latosolic Reddish Prairie soils. Hence, soil conservation measures should generally be targeted to sugarcane fields under the Low Humic Latosols especially those located on steep slopes. Moreover, it was observed in Figure 10.2 that a small change in the field slope results in a relatively large increase in the P index rating especially for the Low Humic Latosols, suggesting that the erosion factor is a highly weighted variable in the P index. Hence future research efforts should be oriented towards improving the accuracy of soil erosion rates estimates. Several research reports have pointed out that the reliability of output of any model, including the P index is dependent on the accuracy of the input values (Sharpley, 1995; Silberstein, 2006; Harmel *et al.*, 2010). Although in this case better estimates of erosion would improve the ability of P indices to provide relative risk assessments, the added complexity may not be totally warranted for the intended users (Harmel *et al.*, 2005).

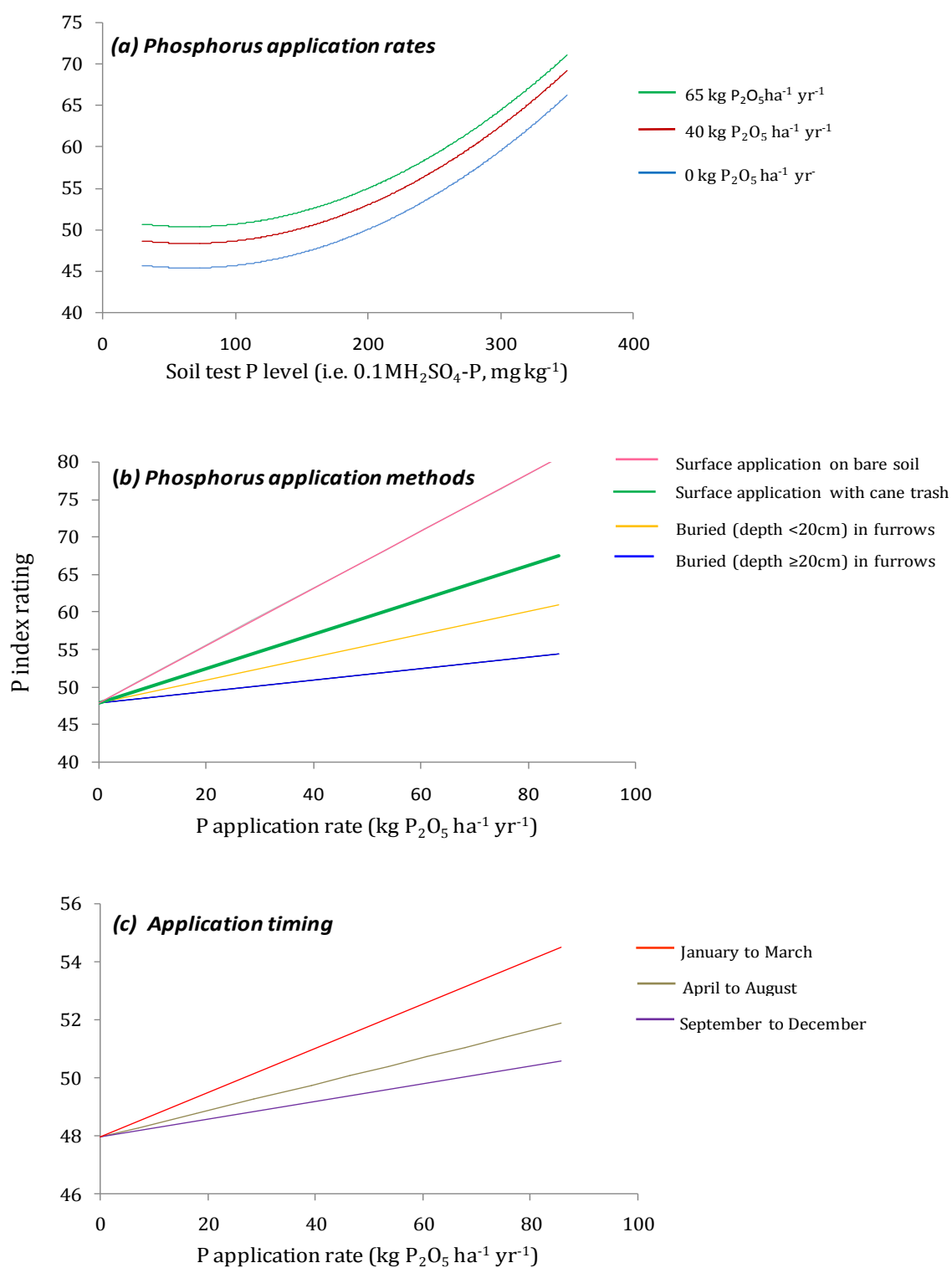


Figure 10.1: Sensitivity of the P index to (a) P application rates, (b) P application methods, and (c) application timing. (Baseline conditions unless indicated otherwise in figure: $160\text{ mg kg}^{-1}\text{ }0.1\text{M H}_2\text{SO}_4\text{-P}$, $10.50\text{ t ha}^{-1}\text{ yr}^{-1}$ erosion rate, 8% field slope, fertilisers applied in furrows at a depth of 20cm in February).

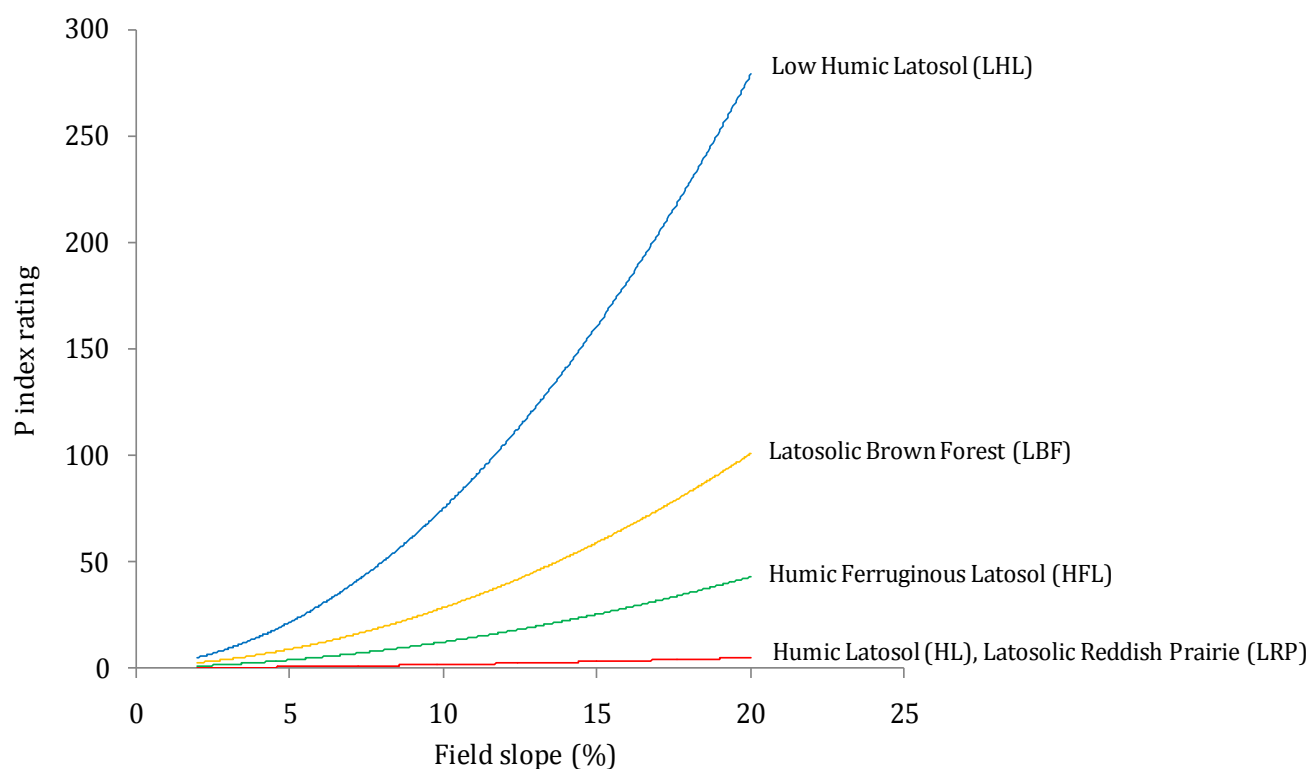


Figure 10.2: Sensitivity of the transport component of the P index across different soil types with varying field slopes. (Baseline conditions: 160 mg kg⁻¹ 0.1M H₂SO₄-P, fertilisers applied at a rate of 45 mg kg⁻¹ P₂O₅ in furrows at a depth of 20cm in February).

10.3.2 Edge-of-plot field testing

The total P concentrations in the runoffs were poorly correlated ($r^2 = 0.15$) to soil test P levels determined by the 0.1M sulphuric acid extraction (Figure 10.3). This trend is similar to that observed in the simulated rainfall runoff studies during the development of the P index. This confirms that soil testing is a poor predictor of runoff P concentrations and that apart from soil P status, transport mechanisms such as surface runoff and soil erosion should be taken into consideration when assessing P mobilisation from fields.

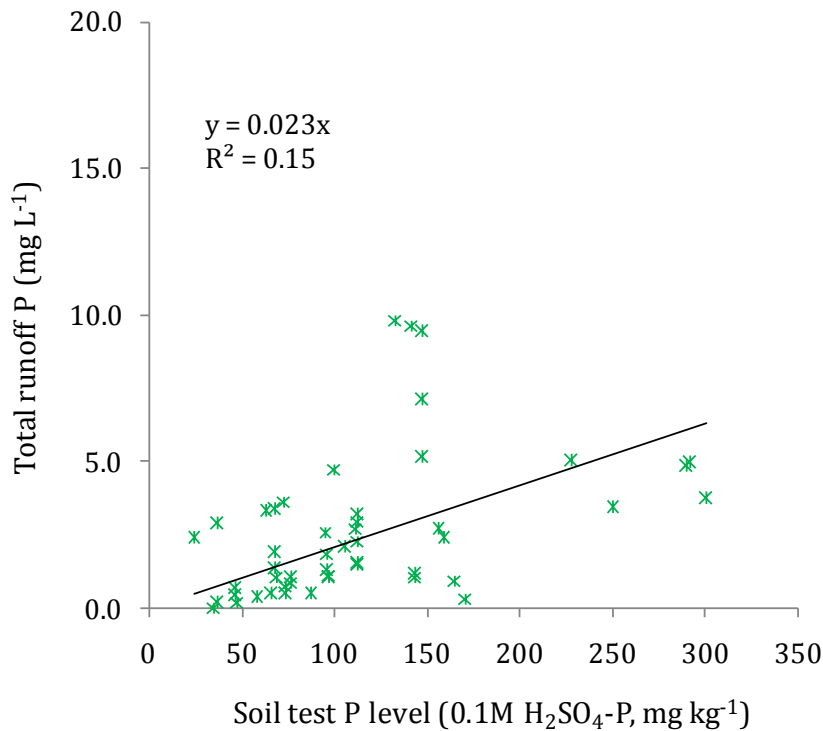


Figure 10.3: Relationship between soil test P level and total runoff P concentrations under simulated rainfall conducted on runoff plots (2.1m by 0.75m).

As indicated by Buczko and Kuchenbuch (2007), a drawback of the evaluation of the P index at runoff plot scales is that factors operating at larger scales (mostly transport factors) are not taken into account. Indeed this could explain the poor relationship between P index rating and total P concentrations in runoffs following simulated rainfall (Figure 10.4a). This relationship was improved when instead of the overall P index, the source potential rating was correlated with total runoff P concentrations (Figure 10.4b), thus showing that transport factors could not be adequately validated using small runoff plots. Therefore, in addition to this plot-scale assessment of the P index, a watershed-scale validation is also required to complement these edge-of-plot field results (Sharpley *et al.*, 2001c). Until more validation data is obtained, it is of critical importance that along with the calculated field P index ratings, clear definition of what the risk assessment tool is useful for and what it is not designed to do should be provided to users. It is emphasized that the P index is not a P loss *quantification* tool but rather a P loss *risk assessment* tool meant for utilization by extension or field officers such that best P management practices be adopted in the critical source areas as explained in Section 2.5.4.

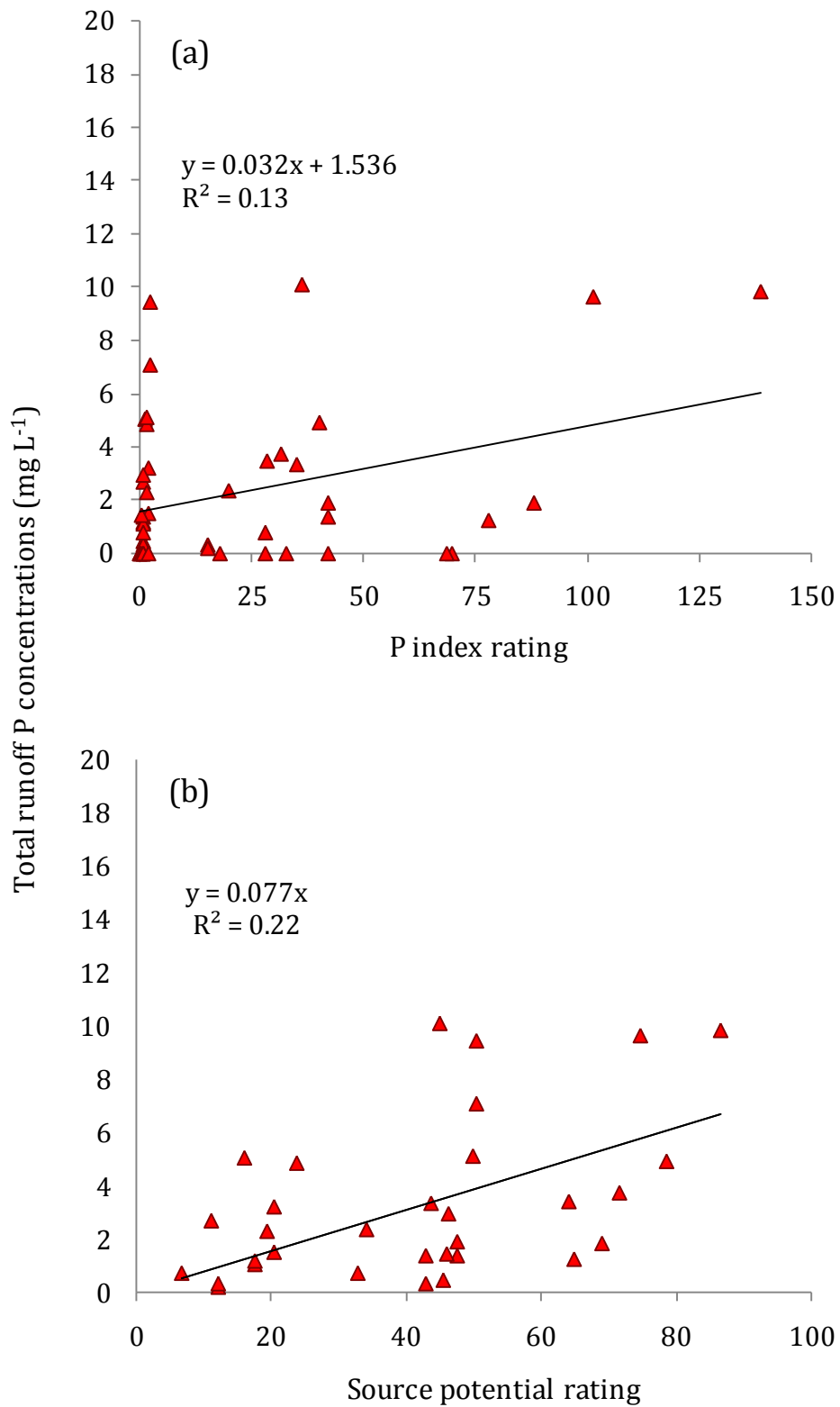


Figure 10.4: Relationship between (a) P index ratings (b) source potential ratings and total runoff P concentrations following 30 minutes simulated rainfall (50, 100 and 150 mm hr^{-1}) over runoff plots (2.1m by 0.75m).

10.4 Conclusion

The sensitivity analysis, as also indicated by Brandt and Elliot (2005) has proved to be a relatively straightforward and computationally simple method for evaluating the behaviour of a P index model. Indeed, the sensitivity analysis results have provided useful information prior to embarking on more elaborate, time consuming and expensive watershed validation studies. For example, soil erosion appears to be a highly weighted factor in the P index matrix and therefore improving soil erosion estimates should be the priority for future work.

Although, the evaluation of the P index using the data from small runoff plots could not account for transport factors, it has nonetheless provided useful preliminary field data. The results have shown clearly that soil testing alone is not an adequate method to estimate the extent of P transport from fields and therefore an integrated approach (which includes soil testing) will be more effective. Furthermore, it should be borne in mind that the rainfall simulation tests were conducted to represent the worst case scenario, that is at field capacity and just after sugarcane plantation when fertilisers have recently been applied and the risks of surface runoff and soil erosion is highest. Obviously, the P index should be tested using differing scenarios (e.g. adoption of best management practices) so as to gain further confidence in its potential as a P risk assessment tool. Buczko and Kuchenbuch (2007) further indicated that the testing of the P index using P loss measurements covering a short time interval might be problematical as a risk cannot be validated by one single measurement. Therefore, in addition to these edge-of-plot testing, it would be necessary to validate the P index using long-term water quality monitoring data consisting of total P concentrations collected from watersheds, such as the study done by Birr and Mulla (2001) in an attempt to compare P index values with stream and lake P monitoring in Minnesota.

Heathwaite *et al.* (2003) suggested that as a P index is used more widely and more fields are evaluated, further information will be gained on the strengths and weaknesses of the approach such that the index may be refined in line with the new knowledge and user-feedback. Also, information obtained from the evaluation of P indices would lead to improvements and advancements in the P index concept (Nelson and Shober, 2012). Indeed, if a P index is deemed to perform inadequately the same model output may be used to guide index revisions such as modifying how the index is calculated, what factors are included, and how each factor is weighted (Sharpley *et al.*, 2012).

11 Summary, synthesis and recommendations

11.1 Summary

11.1.1 Purpose of the study

Sugarcane is the major crop cultivated in Mauritius and in 2013, sugarcane fields occupied some 53,500 hectares of land, that is approximately 29% of the island or 86% of agricultural land with an average cane productivity of 74 t ha⁻¹ over the last five years. The adoption of good management and fertilization practices has played an important role in maintaining this sugarcane productivity (Ng Kee Kwong and Deville, 1992). Phosphorus fertilization for its part is a well-established practice in Mauritius and since the late 1980s, P fertilizer application rates are based on the agronomic soil P test using 0.1M H₂SO₄ as extractant (Cavalot *et al.*, 1988). This soil P test has been based on field experiments carried out in Mauritius and it was found that above a soil P level of 80 mg kg⁻¹, sugarcane yield responses to P fertilization was not obtained (Cavalot *et al.*, 1988). Furthermore, studies carried out by Cavalot *et al.* (1986), have shown that application of P fertilisers is most effective when applied as a starter fertilizer at recommended rates during plantation, as fertilizer P applied later in the sugarcane crop cycle is not taken up and thus, does not translate to sugarcane yield increase.

While the application of P to agricultural land is essential in maintaining crop productivity, P runoff from agricultural landscapes can accelerate eutrophication of surface waters, impairing it for fisheries, recreation and drinking purposes. In fact, agricultural runoff has been identified as one of the potential threats to water quality in Mauritius (Ministry of Environment and Sustainable Development, 2011). Any mismanagement of agricultural P can increase the risk of incidental P losses during runoff and impair freshwater quality. Although these losses may not always be of economic importance to farmers, only small amounts of P (about 10.5 kg P ha⁻¹ in lake Erie, USA; Smith *et al.*, 2015) can trigger eutrophic conditions in freshwaters. Besides, previous studies undertaken in Mauritius have shown that runoff P concentrations of limnological significance (> 0.1 mg P L⁻¹) can occur, particularly during high flow events (Ng Kee Kwong *et al.*, 2002).

Hence, the ultimate purpose of this research project was to prepare a cost effective decision support tool to encourage a judicious use of fertilisers in sugarcane production and to minimize P runoff to freshwaters in Mauritius. As an essential element of the project is to maintain both a profitable sugarcane industry and a clean environment, the decision support tool to minimize offsite P movement was developed within a fully integrated framework to ensure that it not only

has long term positive impact on water quality but is also practical in terms of efficient sugarcane production.

11.1.2 Results derived from study

Field experiments evaluated the factors influencing the vulnerability to movement of P from sugarcane soils in Mauritius and to develop a management tool. This tool combines source and transport factors so as to identify areas within a field that are most prone to P losses. In accomplishing this, rainfall-surface runoff simulations were conducted as per the National P Research Project (2001) at 30 sites representative of the five main soils under which sugarcane is cultivated in Mauritius. Rainfall was simulated on runoff plots (2.1 m by 0.75 m in duplicates) at three different slope categories (0 to 8%, 8 to 13% and 13 to 20%) and under three different rainfall intensities (50, 100 and 150 mm hr⁻¹) for a duration of 30 minutes each. The tests were carried out just after planting the cane, that is at a stage in the crop cycle when the soil is bare and most susceptible to erosion and surface runoff. Furthermore, the rainfall simulations were undertaken when the soils were at field capacity conditions so as to eliminate antecedent soil moisture as a variable and also to avoid the utilization of impractically large volumes of water to generate runoff. Runoff and soil eroded during each rainfall event was measured, collected and each sample analysed for P. In addition, soils were characterized for basic soil properties in terms of pH, particle size distribution, organic matter, exchangeable bases, cation exchange capacity and P status.

Preliminary results obtained from the rainfall simulation studies showed a strong linear relationship between 0.01M CaCl₂ soil extractable P and dissolved runoff P concentrations, indicating that simple laboratory soil extractions using CaCl₂ can in the absence of expensive and labour intensive field experimental setups provide accurate estimates of dissolved runoff P concentrations. However for practical reasons, it was desirable that the assessment of environmental risk associated to soil P status be based on the same soil test utilised to formulate fertiliser recommendations for sugarcane, such that only one set of soil test results will meet the goals of both crop productivity and environmental risk assessment. In that vein, the relationship between the 0.01M CaCl₂-P and the agronomic soil P levels using the 0.1M H₂SO₄ extraction of Cavalot *et al.* (1988) was determined so as to expand the scope of the traditional soil P test into an agro-environmental soil P test. An environmental soil P threshold of 160 mg kg⁻¹ was determined, which is actually twice that of the existing agronomic threshold of 80 mg kg⁻¹. This established environmental threshold has provided a rapid and useful environment risk indicator for the soil P status, however it cannot be inferred that the occurrence of soils with P levels above the environmental threshold is the sole reason for impairment of water quality. The corollary is also

true that soils with a P concentration below the threshold can be a risk to P runoff and water impairment, if P is improperly applied and runoff and erosion occur (Sharpley *et al.*, 2012). In fact, no direct correlation between soil P status and the total amount P transported in runoff waters was obtained which can be explained by the fact that soil P tests provide an indication of desorbable P in soils without assessing the degree to which runoff waters become entrained with eroding particles and enriched in P.

Further results emerging from the rainfall simulation experiments revealed that increasing rainfall intensities enhanced the total amount of P mobilised during runoff events. A similar trend was observed when slope steepness was increased. The enhancement in P transport with increasing rainfall intensities and slope steepness was attributed to larger runoff volumes as well as more soil and sediment being detached and transported during runoffs. On gentle slopes the relatively slow movement of water favoured infiltration into the soil profile while on steep slopes surface runoff predominated over infiltration. With larger runoff volumes, the effective mass of surface soil interacting with rainfall and flowing runoff increased causing more P to be desorbed and particulate P to be detached from the soil surface.

Total P concentrations and sediment loads in runoffs were found to be strongly correlated indicating that a greater proportion of the P transported in runoff occurred as particulate P rather than dissolved P. Irrespective of soil type, rainfall intensity and slope steepness, particulate P was the predominant form of P transported during simulated rainfall events and accounted for about 90% of the total P transported in runoff waters. While particulate P may not be immediately bioavailable, it nonetheless represents a long-term P pool once deposited at the bottom of water bodies and it is therefore important to adopt appropriate measures to mitigate not only soluble P losses but particulate P transport as well (Sharpley, 1993b).

The research findings also showed that soil erosion rates varied considerably across the different soil groups due to differences in the intrinsic soil properties notably soil texture, organic matter content, aggregate stability and infiltration capacity. In addition, irrespective of soil type a strong positive correlation between the amount of bedload transported during simulated runoff events and the P attached to the bedload was observed, showing that P mobility in the landscape is closely associated with soil erosion. Hence, soil conservation management measures that mitigate soil loss such as contour planting, field borders and minimum tillage practices would concomitantly decrease associated P loss.

To sum up, it was evident that apart from P sources, susceptibility of fields to runoff and erosion during rainfall events were important factors to be taken into consideration when assessing P mobilisation from sugarcane fields. The research work presented a risk assessment tool for P losses from sugarcane soils of Mauritius. This tool is based on the P index concept originally developed in the United States and thereafter tailored and adopted in European countries. The proposed P index integrated all factors influencing P losses based not only on the research findings obtained from the simulated rainfall experiments undertaken in Mauritius but also on historical data. The P index incorporated source and transport factors so as to identify critical source areas, that is areas in the field which are most vulnerable to P losses. The source factors accounted for the amount of P in the soil or any other form of P (fertilisers, organic amendments) available for transport. Source factors were thus chemical entities, while transport factors described the hydrological processes intrinsic to a site, which transform potential P sources into actual P losses from the field. As indicated in Table 11.1, five source variables were identified for the P index for sugarcane in Mauritius and included dissolved P (estimated from soil test P), particulate P (estimated from soil erosion), P application rate, the method of application and application timing. The transport variables in the P index were soil erosion, surface runoff potential and a precipitation factor to account for the varying rainfall distribution over the island was also included (Table 11.1). When estimating soil erosion and runoff rates, variations due to field location with respect to the three agro-climatic regions (i.e. sub-humid, humid and super-humid) were considered, thus making the P index adaptable across the highly variable eco-climatic regions of the island.

The established P index was tested using scenario and sensitivity analysis and was also evaluated at a plot scale using rainfall simulation. The sensitivity analysis suggested that the erosion factor was the most highly weighted variable in the index. Hence improvements in estimating soil erosion rates will positively influence the accuracy of the P index output. Additionally, the edge-of-plot testing revealed that the total runoff P concentrations were better correlated to the ratings of the source component of the P index than the overall P index ratings. Transport factors are in fact best evaluated at much larger scales and therefore in addition to the edge-of-plot testing, some validation at watershed scales will also be necessary to gain further confidence on the potential of the P index to rank fields according to their risks of P transfer.

Table 11.1: Source and transport factors in the P index developed for sugarcane soils of Mauritius.

Source/Transport factors	P loss category	P loss rating value
Dissolved P		$0.01207 e^{0.00925 \times \text{Soil test P (mg P kg}^{-1}\text{)}}$
Particulate P		5.5 x erosion (tonnes ha ⁻¹ year ⁻¹)
P application		kg P ₂ O ₅ applied per hectare per year
Application method	Buried (depth > 20cm) in furrows	0.2
	Buried (depth < 20cm) in furrows	0.4
	Surface application with cane trash	0.6
	Surface application on bare soil	1.0
Application timing	September to December	0.2
	April to August	0.3
	January to March	0.5
P application rating = (P application rate x method x timing)		
Source potential = Dissolved P loss rating + Particulate P loss rating + P application rating		
Soil erosion		tonnes hectare ⁻¹ year ⁻¹
Runoff potential	Very low	0.2
	Low	0.4
	Medium	0.6
	High	0.8
	Very high	1.0
Precipitation factor	Sub-humid region	0.64
	Humid region	1.00
	Super-humid region	1.46
Transport potential = (Erosion + Runoff potential) x Precipitation factor		
OVERALL P INDEX = Source potential x Transport potential		

11.2 Implications and recommendations for the sugar industry of Mauritius

11.2.1 Application of research results

The overall outcome of this study is in line with the efforts of the sugarcane industry in Mauritius to search for better means of managing agrochemicals such as fertilisers so that both the profitability from sugarcane production is sustained and the quality of the environment is protected. The pertinence of such commitment should not be overlooked because as the standard of living improves, health, safety and environment simultaneously grow into very sensitive public issues, leading to agrochemicals being perceived with disfavor by the public. National authorities may respond by introducing even stricter regulations or even mandatory reductions in the usage of fertilisers. As it is virtually impossible to produce profitable sugarcane yields without the necessary fertilizer supplements, the aim of research and development in environment protection has been focused on identification or improvement of management practices to limit mobilization

of fertilizer nutrients from the site of application thereby mitigating off-site environmental impacts.

It should be borne in mind that the endeavor to reduce the risk of P contamination of surface water from the outset include the involvement of not only the farmers, who directly face the problems of changing P management practices, but also the public and environment interest groups so as to better understand the complexity of the issue, the costs involved to effect change and the time frame required to reduce risks of P to water quality. It is expected that the adoption of the P index as nutrient management planning tool will receive an initial resistance from the part of farmers and to make a transition in agriculture, it will be necessary for planters to realize the environmental bottom line of P mobilization from sugarcane fields. It is thus imperative to convey to farmers the idea that nutrient management plans such as the P index may not always save money on every sugarcane field and may in some cases cost money. In fact, agronomic decisions regarding P management have positive economic return to the farmer through improved yields or reduced fertilizer cost, but however P management decisions regarding water quality may at often times have a negative economic effect on the planters.

This study has provided a decision support tool to determine environmentally sustainable use of P fertilisers in sugarcane production by indicating to extension officers and farmers in Mauritius where improved management practices should be targeted to reduce the incidence of accelerated eutrophication of lakes and rivers. The ultimate outcome is better management practices that are both agronomically and environmentally sound for producing sugarcane. The developed P index is a field evaluation tool ranking site vulnerability of P loss from sugarcane fields to identify and prioritize P management options. In fact, when the factors of the index are analysed, it becomes apparent that individual parameters could be influencing the index disproportionately. These identified parameters should be the basis for choosing the best and most suited management measures. A selection of best management practices has been provided in the developed P index framework and include terracing, construction of diversions in field areas where terracing is not possible, field borders, field strips, grassed waterways, forest buffers and herbaceous cover. This list is not exhaustive and as the P index is used more widely, more management measures especially those related to sugarcane cultivation practices could be identified with extension officers and farmer input and included in the list.

It is emphasized that the developed P index is not a quantitative predictor of P loss from particular fields and has not been intended to be an evaluation tool for determining whether planters are complying with the local water quality standards. Furthermore, the adoption of the P index tool

surpasses the use of simulations models (SWAT) because they provide a balance between scientific rigor and end-use practicality. 'Perfect' watershed models can provide accurate estimates of P losses from a watershed which is therefore satisfying to the research scientific community. However, such models tend to be more data-demanding, time and expertise expensive and are often too complicated for application by extension officers, field staff and nutrient management planners, thus having minimal practical and real-world impact. By virtue of its simplicity, flexibility and user-friendliness the P index is more suitable as a routine nutrient management tool over models such as SWAT.

The developed P index will be made accessible to end-users through extension meetings, seminars, technical and user documentation describing how to utilize the tool. In the near future the adoption of the P index may be facilitated through the development of simple spreadsheets or any other suitable software for calculating P index ratings for different cane fields. It is planned that user feedback during field use of the index, can provide ideas to improve the accuracy and usability of the P index. As with any successful field tool, its development, implementation, and adoption results from an open collaboration among researchers, extension officers, and farmers.

11.2.2 Suggestions for future research

The research approach of rainfall simulation on small plots has the advantage of providing rapid results, is an efficient way to collect data, provides control of many landscape variables that potentially confound watershed research and is adaptable to a range of research approaches (Dougherty *et al.*, 2008). However, all research approaches have their limitations and the utilization of rainfall simulators to estimate edge-of-plot nutrient and soil losses is no exception. Sharpley and Kleinman (2003) stipulated that rainfall simulators and small runoff plots cannot reproduce flow processes occurring over a landscape and are most appropriate for elucidating flow-soil P interdependencies. Small plots have hydrological characteristics different to those of larger landscape and therefore further work is needed on longer slopes at watershed scales so as to support the results already obtained at small plot scales.

Watershed monitoring actually represents the most direct evaluation of soil and nutrient management impacts on water quality because watershed export of P is the ultimate concern to eutrophication (Kleinman *et al.*, 2004). Watershed scale evaluations integrate all processes controlling P losses and assess land use changes on downgradient water quality impacts. Further, efficiencies of the best P management practices proposed in the established P index were based on studies done in the United States under crops other than sugarcane. It is therefore recommended that those selected management practices, as well as other management measures specific to

sugarcane be tested at a field or watershed scale, so as to refine the credit given to each management practice. Furthermore, examination of P export from watersheds versus the actual P index ratings will provide a basis for selection of science-base breakpoints associated with P index interpretations. Indeed, the ranking of site vulnerability to P losses for P index interpretations and the breakpoints between the different categories (low, medium, high and very high) of the current P index was adapted from Sharpley *et al.* (2001c) and was obtained by normalizing the P index values by calculating the P index rating with all source and transport variables set as high as explained in Sharpley *et al.* (2003).

The edge-of-plot testing was unable to fully assess the transport factors and therefore testing of the P index at a watershed scale will provide more insight into the ability of the P index as a risk assessment tool. Moreover, the results of the sensitivity analysis showed that the output of the developed P index is greatly influenced by the soil erosion factor and should therefore be more thoroughly investigated and validated on a field or watershed scale. It should be remembered that this P index was established with the realization that as it is used more widely and more cane fields are evaluated, further information will be gained on the strengths and weaknesses of the tool which will eventually lead to modifications and further developments of the risk assessment tool. In summary, the P index as a nutrient management tool will continue to evolve as more information is gained on P transformation and mobilization processes, as well as incorporating farmer knowledge and experience. This will ensure that updates and improvements in the index will translate to more accurate and reliable estimates of the impacts of agricultural P management at field or watershed scale on downstream water quality.

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APPENDICES

APPENDIX 1(a): Characteristics of soils representative of the Low Humic Latosol (*Humic Nitosol*).

Study site	Soil sample code	Soil depth (cm)	pH	OM (%)	Particle-size (oven-dried basis)					Ca	Mg	K	Na	CEC
			(H ₂ O)	Oven-dried basis	% Clay	% Silt	% Total sand	% Coarse sand	% Fine sand	cmol+ kg ⁻¹				
Highlands	2011/04	0-15	5.3	4.2	49.4	24.3	26.2	8.1	18.1	3.40	0.73	0.68	0.31	5.1
La Mecque	2011/08		7.0	3.4	53.5	23.6	22.9	8.3	14.5	14.84	5.84	7.48	0.61	28.8
Olivia	2011/16		5.5	4.3	65.6	20.3	14.1	2.6	11.4	3.82	1.08	0.53	0.36	5.8
Savinia	SAV 5		6.8	3.0	49.2	21.9	28.9	7.8	21.1	6.38	3.52	0.66	2.36	12.9
Olivia	2011/18		5.6	4.3	72.2	17.8	10.0	2.0	8.0	4.94	1.09	0.37	0.31	6.7
Henrietta	HRT 5		6.9	4.3	34.2	32.2	33.6	11.3	22.3	11.06	1.09	0.58	0.23	13.0
Mean value			6.2	3.9	54.0	23.4	22.6	6.7	15.9	7.40	2.23	1.72	0.70	12.0
Standard deviation			0.8	0.6	13.4	4.9	9.0	3.6	5.6	4.58	2.04	2.83	0.82	8.9
Standard error			0.3	0.2	5.5	2.0	3.7	1.5	2.3	1.87	0.83	1.15	0.34	3.6
Highlands	2011/05	15-30	5.0	3.8	49.8	24.0	26.2	7.5	18.6	2.75	0.71	0.46	0.24	4.2
La Mecque	2011/09		7.0	3.0	56.1	23.3	20.6	7.7	12.9	14.24	6.15	5.12	0.55	26.1
Olivia	2011/17		5.6	3.7	66.2	19.2	14.6	2.5	12.1	4.26	0.88	0.28	0.31	5.7
Savinia	SAV 10		6.4	4.2	63.2	19.6	17.2	4.8	12.4	9.62	3.97	0.86	1.17	15.6
Olivia-I	2011/19		5.6	4.4	73.6	16.8	9.5	2.1	7.4	5.45	1.10	0.42	0.35	7.3
Henrietta	HRT 6		6.6	3.9	39.6	27.8	32.6	12.2	20.4	7.96	0.65	0.66	0.26	9.5
Mean value			6.0	3.8	58.1	21.8	20.1	6.2	14.0	7.38	2.24	1.30	0.48	11.4
Standard deviation			0.7	0.5	12.2	4.0	8.3	3.8	4.8	4.18	2.29	1.88	0.36	8.2
Standard error			0.3	0.2	5.0	1.6	3.4	1.5	1.9	1.71	0.94	0.77	0.15	3.4

APPENDIX 1(b): Characteristics of soils representative of the Humic Latosol (*Humic Nitosol*).

Study site	Soil sample code	Soil depth (cm)	pH	OM (%)	Particle-size (oven-dried basis)					Ca	Mg	K	Na	CEC
			(H ₂ O)	Oven-dried basis	% Clay	% Silt	% Total sand	% Coarse sand	% Fine sand	cmol ⁺ kg ⁻¹				
Etoile	2011/14	0-15	5.5	4.3	33.9	32.4	33.7	10.0	23.7	3.51	0.77	0.27	0.27	4.8
St- Pierre	MDA 1		6.0	4.6	34.4	31.0	34.6	12.6	22.0	6.29	0.92	0.31	0.30	7.8
Belle Rive	BRIV 4		5.7	4.4	37.6	38.7	23.7	11.8	11.9	2.24	0.83	0.29	0.30	3.7
Belle Rive	BRBC 3		5.3	4.6	52.8	24.1	23.1	8.7	14.4	4.53	1.39	0.49	0.36	6.8
Belle Rive	B-R 9		5.2	4.6	43.7	28.6	27.7	11.0	16.7	1.53	0.63	0.20	0.20	2.6
Mean value			5.5	4.5	40.5	31.0	28.6	10.8	17.7	3.62	0.91	0.31	0.28	5.1
Standard deviation			0.3	0.1	7.9	5.4	5.4	1.5	5.0	1.89	0.29	0.11	0.06	2.2
Standard error			0.2	0.1	3.5	2.4	2.4	0.7	2.2	0.84	0.13	0.05	0.03	1.0
Etoile	2011/15	15-30	5.5	4.0	51.5	23.6	24.9	7.0	17.9	2.29	0.74	0.16	0.20	3.4
St- Pierre	MDA 3		5.8	4.2	32.4	34.5	33.1	9.7	23.4	5.95	0.89	0.38	0.21	7.4
Belle Rive	BRIV 5		5.6	3.9	38.9	35.2	25.9	13.1	12.8	1.48	0.68	0.29	0.23	2.7
Belle Rive	BRBC 9		5.0	4.0	53.7	14.6	31.7	17.0	14.7	2.31	0.67	0.12	0.17	3.3
Belle Rive	B-R 2		5.8	5.2	35.4	37.3	27.4	12.1	15.3	4.86	0.94	0.19	0.33	6.3
Mean value			5.5	4.3	42.4	29.0	28.6	11.8	16.8	3.38	0.78	0.23	0.23	4.6
Standard deviation			0.3	0.5	9.7	9.7	3.6	3.7	4.1	1.92	0.13	0.11	0.06	2.1
Standard error			0.2	0.2	4.3	4.3	1.6	1.7	1.8	0.86	0.06	0.05	0.03	0.9

APPENDIX 1(c): Characteristics of soils representative of the Humic Ferruginous Latosol (*Humic Acrisol*).

Study site	Soil sample code	Soil depth (cm)	pH	OM (%)	Particle-size (oven-dried basis)					Ca	Mg	K	Na	CEC
			(H ₂ O)	Oven-dried basis	% Clay	% Silt	% Total sand	% Coarse sand	% Fine sand	cmol+ kg ⁻¹				
Valetta	2011/02	0-15	6.6	4.8	33.9	34.1	32.0	13.7	18.4	13.21	1.34	0.56	0.32	15.4
Belle Rive	2011/12		5.2	3.8	44.5	31.8	23.6	12.2	11.4	1.38	0.53	0.21	0.25	2.4
Rose Belle	2011/20		5.3	6.0	35.3	24.0	40.7	18.8	21.9	2.57	0.91	0.23	0.31	4.0
Riche en Eau	R-E 1		5.8	2.1	33.4	34.7	31.9	16.1	15.9	4.05	0.68	0.14	0.24	5.1
Riche en Eau	REE 2		5.8	4.8	31.5	36.4	32.1	14.1	18.0	3.24	0.71	0.38	0.27	4.6
Cent Gaulettes	CENT 1		6.5	4.8	36.1	25.6	38.3	13.6	24.8	6.61	1.01	0.72	0.35	8.7
Mean value			5.9	4.4	34.0	30.2	35.8	15.6	20.1	4.12	0.83	0.37	0.29	5.6
Standard deviation			0.6	1.3	4.6	5.1	6.0	2.3	4.7	4.31	0.29	0.23	0.05	4.8
Standard error			0.2	0.5	1.9	2.1	2.4	1.0	1.9	1.76	0.12	0.09	0.02	1.9
Valetta	2011/03	15-30	6.6	4.4	33.7	35.1	31.3	12.7	18.6	11.82	1.28	0.52	0.28	13.9
Belle Rive	2011/13		5.2	4.0	45.2	31.2	23.6	12.9	10.6	1.33	0.47	0.20	0.17	2.2
Rose Belle	2011/21		5.4	4.4	25.8	38.1	36.1	13.6	22.5	1.06	0.55	0.16	0.17	1.9
Riche en Eau	R-E 7		5.8	2.0	31.5	34.4	34.1	14.3	19.8	4.18	0.71	0.12	0.17	5.2
Riche en Eau	R-E 8		5.3	3.9	25.5	33.9	40.6	15.4	25.2	1.00	0.39	0.06	0.13	1.6
Cent Gaulettes	CENT 4		5.9	4.6	42.7	23.7	33.6	13.4	20.2	6.16	0.72	0.38	0.30	7.6
Mean value			5.6	3.7	31.4	32.5	36.1	14.2	21.9	3.10	0.59	0.18	0.20	4.1
Standard deviation			0.5	1.0	8.3	4.9	5.7	1.0	4.9	4.25	0.32	0.17	0.07	4.8
Standard error			0.2	0.4	3.4	2.0	2.3	0.4	2.0	1.73	0.13	0.07	0.03	1.9

APPENDIX 1(d): Characteristics of soils representative of the Latosolic Reddish Prairie (*Eutric Cambisol*).

Study site	Soil sample code	Soil depth (cm)	pH	OM (%)	Particle-size (oven-dried basis)					Ca	Mg	K	Na	CEC
			(H ₂ O)	Oven-dried basis	% Clay	% Silt	% Total sand	% Coarse sand	% Fine sand	cmol ⁺ kg ⁻¹				
Maison Blanche	2011/06	0-15	6.0	3.8	53.0	21.3	25.6	11.9	13.7	8.31	3.58	2.87	0.56	15.3
La Sourdine	SOU 2		5.7	4.0	54.8	23.6	21.5	0.8	20.7	7.32	5.34	0.83	0.55	14.0
Esperance	ESP 2		6.2	5.0	58.3	21.9	19.8	8.5	11.4	12.60	6.18	2.33	0.59	21.7
Pierrefonds	P-R 1		6.7	4.8	44.2	19.0	36.8	18.1	18.7	10.14	4.48	1.11	0.78	16.5
Maison Blanche	2011/22		6.1	4.1	41.5	21.2	37.3	1.9	35.3	9.50	4.03	2.09	0.67	16.3
Saviniah	LBSV 9		6.2	4.7	45.8	20.4	33.8	16.9	16.9	9.09	5.16	0.79	0.66	15.7
Beau Songes	BS 2		6.6	3.8	50.8	22.6	26.6	10.4	16.2	13.23	6.00	4.53	0.71	24.5
Pierrefonds	PRF 1		7.1	2.1	27.4	29.6	43.0	21.2	21.8	15.29	4.62	1.04	0.84	21.8
Mean value			6.3	4.0	47.0	22.5	30.6	11.2	19.3	10.68	4.92	1.95	0.67	18.2
Standard deviation			0.4	0.9	9.7	3.2	8.3	7.4	7.3	2.74	0.92	1.30	0.10	3.8
Standard error			0.2	0.3	3.4	1.1	2.9	2.6	2.6	0.97	0.32	0.46	0.04	1.4
Maison Blanche	2011/07	15-30	6.0	3.5	52.1	21.9	26.0	11.7	14.3	8.88	4.42	4.12	0.58	18.0
La Sourdine	SOU 3		5.9	3.9	49.2	29.8	20.9	0.8	20.1	8.24	5.30	0.56	0.61	14.7
Esperance	ESP 7		6.3	4.9	56.7	24.9	18.4	7.9	10.4	12.24	7.88	1.74	0.40	22.3
Pierrefonds	P-R 3		6.7	4.5	46.9	20.5	32.7	16.6	16.1	10.49	4.34	1.18	0.77	16.8
Pierrefonds	P-R 5		6.4	4.0	49.8	26.9	23.3	11.3	12.0	12.36	6.84	2.90	0.37	22.5
Maison Blanche	2011/23		6.2	3.9	42.7	23.9	33.4	16.5	16.8	9.71	4.07	2.31	0.69	16.8
Pierrefonds	PRF 3		6.6	2.8	42.1	23.0	34.9	17.2	17.7	13.42	4.04	0.95	1.01	19.4
Mean value			6.3	3.9	48.5	24.4	27.1	11.7	15.3	10.76	5.27	1.97	0.63	18.6
Standard deviation			0.3	0.7	5.2	3.2	6.6	5.9	3.3	1.95	1.52	1.24	0.22	2.9
Standard error			0.1	0.3	2.0	1.2	2.5	2.2	1.3	0.7	0.6	0.5	0.1	1.1

APPENDIX 1(e): Characteristics of soils representative of the Latosolic Brown Forest (*Dystric Cambisol*).

Study site	Soil sample code	Soil depth (cm)	pH	OM (%)	Particle-size (oven-dried basis)					Ca	Mg	K	Na	CEC
			(H ₂ O)	Oven-dried basis	Clay (%)	Silt (%)	Total sand (%)	Coarse sand (%)	Fine sand (%)	cmol+ kg ⁻¹				
Rose Belle	2011/10	0-15	5.4	7.2	29.4	30.2	40.3	2.0	38.3	1.55	1.17	0.77	0.33	3.8
Mare Chicose	MC 4		5.9	5.1	13.0	39.6	47.4	17.5	29.9	2.59	3.01	0.57	0.21	6.4
Joli Bois	JB 5		5.3	8.7	35.4	30.8	33.8	15.3	18.5	2.74	1.03	0.27	0.34	4.4
Le Val	VAL 1		5.2	5.0	52.7	29.7	17.6	10.2	7.4	3.40	3.23	0.69	0.41	7.7
Joli Bois	JBOIS 2		5.8	6.8	20.5	40.2	39.4	17.7	21.7	7.87	1.11	0.28	0.37	9.6
Mean value			5.5	6.5	30.2	34.1	35.7	12.6	23.1	3.63	1.91	0.52	0.33	6.4
Standard deviation			0.3	1.6	15.2	5.3	11.2	6.6	11.7	2.46	1.11	0.23	0.08	2.4
Standard error			0.1	0.7	6.8	2.4	5.0	3.0	5.2	1.10	0.50	0.10	0.03	1.1
Rose Belle	2011/11	15-30	5.1	6.5	30.3	28.9	40.8	20.1	20.7	1.51	1.29	4.92	0.33	8.1
Joli Bois	JBOIS 10		5.4	6.1	33.9	31.4	34.8	16.1	18.7	3.01	1.04	0.24	0.21	4.5
Joli Bois	JBOIS 10		5.4	6.1	33.9	31.4	34.8	16.1	18.7	3.01	1.04	0.24	0.21	4.5
Mean value			5.3	6.3	32.7	30.5	36.8	17.4	19.4	2.51	1.13	1.80	0.25	5.7
Standard deviation			0.2	0.2	2.1	1.4	3.5	2.3	1.2	0.86	0.14	2.70	0.07	2.0
Standard error			0.1	0.1	1.2	0.8	2.0	1.3	0.7	0.50	0.08	1.56	0.04	1.2

Appendix 2: Monthly rainfall distribution for the three climatic regions of Mauritius based on 10 years rainfall data, i.e. from 2004 to 2013.

Month	Sub-humid	Humid	Super-humid
	mm year⁻¹		
January	189	342	419
February	240	385	516
March	287	406	541
April	88	149	250
May	92	141	243
June	85	117	185
July	63	130	209
August	68	116	185
September	73	96	145
October	44	75	132
November	72	87	148
December	89	128	187
Annual total	1,390	2,170	3,160

Appendix 3(a): Calculation of erosion using RUSLE for Low Humic Latosol (LHL).

Sub-humid (EI = 100)							Humid (EI = 110)							Super-humid (EI = 130)						
A	R	L	P	C	S	K	A	R	L	P	C	S	K	A	R	L	P	C	S	K
t ⁻¹ ha ⁻¹ yr ⁻¹					%		t ⁻¹ ha ⁻¹ yr ⁻¹					%		t ⁻¹ ha ⁻¹ yr ⁻¹					%	
1.08	300	1	0.82	0.07	1	0.14	1.26	300	1	0.96	0.07	1	0.14	1.31	300	1	1.00	0.07	1	0.14
1.92	300	1	0.73	0.07	2	0.14	2.44	300	1	0.93	0.07	2	0.14	2.63	300	1	1.00	0.07	2	0.14
2.76	300	1	0.70	0.07	3	0.14	3.66	300	1	0.93	0.07	3	0.14	3.94	300	1	1.00	0.07	3	0.14
3.62	300	1	0.69	0.07	4	0.14	4.83	300	1	0.92	0.07	4	0.14	5.25	300	1	1.00	0.07	4	0.14
4.53	300	1	0.69	0.07	5	0.14	6.04	300	1	0.92	0.07	5	0.14	6.56	300	1	1.00	0.07	5	0.14
5.59	300	1	0.71	0.07	6	0.14	7.32	300	1	0.93	0.07	6	0.14	7.88	300	1	1.00	0.07	6	0.14
6.71	300	1	0.73	0.07	7	0.14	8.64	300	1	0.94	0.07	7	0.14	9.19	300	1	1.00	0.07	7	0.14
7.98	300	1	0.76	0.07	8	0.14	10.08	300	1	0.96	0.07	8	0.14	10.50	300	1	1.00	0.07	8	0.14
9.45	300	1	0.80	0.07	9	0.14	11.46	300	1	0.97	0.07	9	0.14	11.81	300	1	1.00	0.07	9	0.14
11.03	300	1	0.84	0.07	10	0.14	12.99	300	1	0.99	0.07	10	0.14	13.13	300	1	1.00	0.07	10	0.14
12.71	300	1	0.88	0.07	11	0.14	14.44	300	1	1.00	0.07	11	0.14	14.44	300	1	1.00	0.07	11	0.14
14.81	300	1	0.94	0.07	12	0.14	15.75	300	1	1.00	0.07	12	0.14	15.75	300	1	1.00	0.07	12	0.14
16.55	300	1	0.97	0.07	13	0.14	17.06	300	1	1.00	0.07	13	0.14	17.06	300	1	1.00	0.07	13	0.14
18.38	300	1	1.00	0.07	14	0.14	18.38	300	1	1.00	0.07	14	0.14	18.38	300	1	1.00	0.07	14	0.14
19.69	300	1	1.00	0.07	15	0.14	19.69	300	1	1.00	0.07	15	0.14	19.69	300	1	1.00	0.07	15	0.14
21.00	300	1	1.00	0.07	16	0.14	21.00	300	1	1.00	0.07	16	0.14	21.00	300	1	1.00	0.07	16	0.14
22.31	300	1	1.00	0.07	17	0.14	22.31	300	1	1.00	0.07	17	0.14	22.31	300	1	1.00	0.07	17	0.14
23.63	300	1	1.00	0.07	18	0.14	23.63	300	1	1.00	0.07	18	0.14	23.63	300	1	1.00	0.07	18	0.14
24.94	300	1	1.00	0.07	19	0.14	24.94	300	1	1.00	0.07	19	0.14	24.94	300	1	1.00	0.07	19	0.14
26.25	300	1	1.00	0.07	20	0.14	26.25	300	1	1.00	0.07	20	0.14	26.25	300	1	1.00	0.07	20	0.14

Appendix 3(b): Calculation of erosion using RUSLE for Humic Ferruginous Latosol (HFL).

Sub-humid (EI = 100)							Humid (EI = 110)							Super-humid (EI = 130)						
A	R	L	P	C	S	K	A	R	L	P	C	S	K	A	R	L	P	C	S	K
t ⁻¹ ha ⁻¹ yr ⁻¹					%		t ⁻¹ ha ⁻¹ yr ⁻¹					%		t ⁻¹ ha ⁻¹ yr ⁻¹					%	
0.38	300	1	0.82	0.07	1	0.05	0.45	300	1	0.96	0.07	1	0.05	0.47	300	1	1.00	0.07	1	0.05
0.68	300	1	0.73	0.07	2	0.05	0.87	300	1	0.93	0.07	2	0.05	0.94	300	1	1.00	0.07	2	0.05
0.98	300	1	0.70	0.07	3	0.05	1.31	300	1	0.93	0.07	3	0.05	1.41	300	1	1.00	0.07	3	0.05
1.29	300	1	0.69	0.07	4	0.05	1.73	300	1	0.92	0.07	4	0.05	1.88	300	1	1.00	0.07	4	0.05
1.62	300	1	0.69	0.07	5	0.05	2.16	300	1	0.92	0.07	5	0.05	2.34	300	1	1.00	0.07	5	0.05
2.00	300	1	0.71	0.07	6	0.05	2.62	300	1	0.93	0.07	6	0.05	2.81	300	1	1.00	0.07	6	0.05
2.40	300	1	0.73	0.07	7	0.05	3.08	300	1	0.94	0.07	7	0.05	3.28	300	1	1.00	0.07	7	0.05
2.85	300	1	0.76	0.07	8	0.05	3.60	300	1	0.96	0.07	8	0.05	3.75	300	1	1.00	0.07	8	0.05
3.38	300	1	0.80	0.07	9	0.05	4.09	300	1	0.97	0.07	9	0.05	4.22	300	1	1.00	0.07	9	0.05
3.94	300	1	0.84	0.07	10	0.05	4.64	300	1	0.99	0.07	10	0.05	4.69	300	1	1.00	0.07	10	0.05
4.54	300	1	0.88	0.07	11	0.05	5.16	300	1	1.00	0.07	11	0.05	5.16	300	1	1.00	0.07	11	0.05
5.29	300	1	0.94	0.07	12	0.05	5.63	300	1	1.00	0.07	12	0.05	5.63	300	1	1.00	0.07	12	0.05
5.91	300	1	0.97	0.07	13	0.05	6.09	300	1	1.00	0.07	13	0.05	6.09	300	1	1.00	0.07	13	0.05
6.56	300	1	1.00	0.07	14	0.05	6.56	300	1	1.00	0.07	14	0.05	6.56	300	1	1.00	0.07	14	0.05
7.03	300	1	1.00	0.07	15	0.05	7.03	300	1	1.00	0.07	15	0.05	7.03	300	1	1.00	0.07	15	0.05
7.50	300	1	1.00	0.07	16	0.05	7.50	300	1	1.00	0.07	16	0.05	7.50	300	1	1.00	0.07	16	0.05
7.97	300	1	1.00	0.07	17	0.05	7.97	300	1	1.00	0.07	17	0.05	7.97	300	1	1.00	0.07	17	0.05
8.44	300	1	1.00	0.07	18	0.05	8.44	300	1	1.00	0.07	18	0.05	8.44	300	1	1.00	0.07	18	0.05
8.91	300	1	1.00	0.07	19	0.05	8.91	300	1	1.00	0.07	19	0.05	8.91	300	1	1.00	0.07	19	0.05
9.38	300	1	1.00	0.07	20	0.05	9.38	300	1	1.00	0.07	20	0.05	9.38	300	1	1.00	0.07	20	0.05

Appendix 3(c): Calculation of erosion using RUSLE for Latosolic Brown Forest (LBF).

Sub-humid (EI = 100)							Humid (EI = 110)							Super-humid (EI = 130)						
A	R	L	P	C	S	K	A	R	L	P	C	S	K	A	R	L	P	C	S	K
t ⁻¹ ha ⁻¹ yr ⁻¹					%		t ⁻¹ ha ⁻¹ yr ⁻¹					%		t ⁻¹ ha ⁻¹ yr ⁻¹					%	
0.62	300	1	0.82	0.07	1	0.08	0.72	300	1	0.96	0.07	1	0.08	0.75	300	1	1.00	0.07	1	0.08
1.10	300	1	0.73	0.07	2	0.08	1.40	300	1	0.93	0.07	2	0.08	1.50	300	1	1.00	0.07	2	0.08
1.58	300	1	0.70	0.07	3	0.08	2.09	300	1	0.93	0.07	3	0.08	2.25	300	1	1.00	0.07	3	0.08
2.07	300	1	0.69	0.07	4	0.08	2.76	300	1	0.92	0.07	4	0.08	3.00	300	1	1.00	0.07	4	0.08
2.59	300	1	0.69	0.07	5	0.08	3.45	300	1	0.92	0.07	5	0.08	3.75	300	1	1.00	0.07	5	0.08
3.20	300	1	0.71	0.07	6	0.08	4.19	300	1	0.93	0.07	6	0.08	4.50	300	1	1.00	0.07	6	0.08
3.83	300	1	0.73	0.07	7	0.08	4.94	300	1	0.94	0.07	7	0.08	5.25	300	1	1.00	0.07	7	0.08
4.56	300	1	0.76	0.07	8	0.08	5.76	300	1	0.96	0.07	8	0.08	6.00	300	1	1.00	0.07	8	0.08
5.40	300	1	0.80	0.07	9	0.08	6.55	300	1	0.97	0.07	9	0.08	6.75	300	1	1.00	0.07	9	0.08
6.30	300	1	0.84	0.07	10	0.08	7.43	300	1	0.99	0.07	10	0.08	7.50	300	1	1.00	0.07	10	0.08
7.26	300	1	0.88	0.07	11	0.08	8.25	300	1	1.00	0.07	11	0.08	8.25	300	1	1.00	0.07	11	0.08
8.46	300	1	0.94	0.07	12	0.08	9.00	300	1	1.00	0.07	12	0.08	9.00	300	1	1.00	0.07	12	0.08
9.46	300	1	0.97	0.07	13	0.08	9.75	300	1	1.00	0.07	13	0.08	9.75	300	1	1.00	0.07	13	0.08
10.50	300	1	1.00	0.07	14	0.08	10.50	300	1	1.00	0.07	14	0.08	10.50	300	1	1.00	0.07	14	0.08
11.25	300	1	1.00	0.07	15	0.08	11.25	300	1	1.00	0.07	15	0.08	11.25	300	1	1.00	0.07	15	0.08
12.00	300	1	1.00	0.07	16	0.08	12.00	300	1	1.00	0.07	16	0.08	12.00	300	1	1.00	0.07	16	0.08
12.75	300	1	1.00	0.07	17	0.08	12.75	300	1	1.00	0.07	17	0.08	12.75	300	1	1.00	0.07	17	0.08
13.50	300	1	1.00	0.07	18	0.08	13.50	300	1	1.00	0.07	18	0.08	13.50	300	1	1.00	0.07	18	0.08
14.25	300	1	1.00	0.07	19	0.08	14.25	300	1	1.00	0.07	19	0.08	14.25	300	1	1.00	0.07	19	0.08
15.00	300	1	1.00	0.07	20	0.08	15.00	300	1	1.00	0.07	20	0.08	15.00	300	1	1.00	0.07	20	0.08

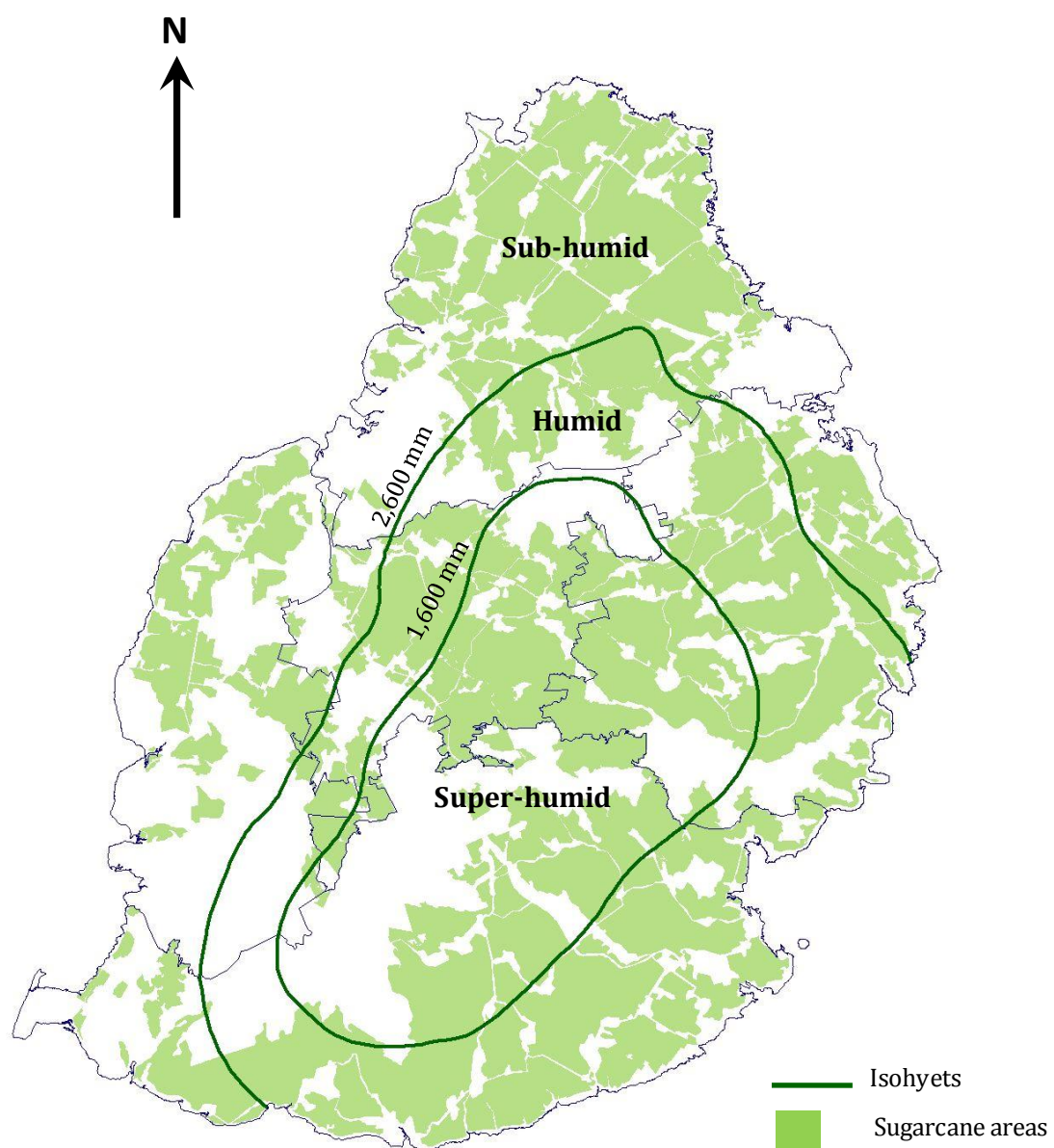
Appendix 3(d): Calculation of erosion using RUSLE for Humic Latosols (HL).

Sub-humid (EI = 100)							Humid (EI = 110)							Super-humid (EI = 130)						
A	R	L	P	C	S	K	A	R	L	P	C	S	K	A	R	L	P	C	S	K
t ⁻¹ ha ⁻¹ yr ⁻¹					%		t ⁻¹ ha ⁻¹ yr ⁻¹					%		t ⁻¹ ha ⁻¹ yr ⁻¹					%	
0.08	300	1	0.82	0.07	1	0.01	0.09	300	1	0.96	0.07	1	0.01	0.09	300	1	1.00	0.07	1	0.01
0.14	300	1	0.73	0.07	2	0.01	0.17	300	1	0.93	0.07	2	0.01	0.19	300	1	1.00	0.07	2	0.01
0.20	300	1	0.70	0.07	3	0.01	0.26	300	1	0.93	0.07	3	0.01	0.28	300	1	1.00	0.07	3	0.01
0.26	300	1	0.69	0.07	4	0.01	0.35	300	1	0.92	0.07	4	0.01	0.38	300	1	1.00	0.07	4	0.01
0.32	300	1	0.69	0.07	5	0.01	0.43	300	1	0.92	0.07	5	0.01	0.47	300	1	1.00	0.07	5	0.01
0.40	300	1	0.71	0.07	6	0.01	0.52	300	1	0.93	0.07	6	0.01	0.56	300	1	1.00	0.07	6	0.01
0.48	300	1	0.73	0.07	7	0.01	0.62	300	1	0.94	0.07	7	0.01	0.66	300	1	1.00	0.07	7	0.01
0.57	300	1	0.76	0.07	8	0.01	0.72	300	1	0.96	0.07	8	0.01	0.75	300	1	1.00	0.07	8	0.01
0.68	300	1	0.80	0.07	9	0.01	0.82	300	1	0.97	0.07	9	0.01	0.84	300	1	1.00	0.07	9	0.01
0.79	300	1	0.84	0.07	10	0.01	0.93	300	1	0.99	0.07	10	0.01	0.94	300	1	1.00	0.07	10	0.01
0.91	300	1	0.88	0.07	11	0.01	1.03	300	1	1.00	0.07	11	0.01	1.03	300	1	1.00	0.07	11	0.01
1.06	300	1	0.94	0.07	12	0.01	1.13	300	1	1.00	0.07	12	0.01	1.13	300	1	1.00	0.07	12	0.01
1.18	300	1	0.97	0.07	13	0.01	1.22	300	1	1.00	0.07	13	0.01	1.22	300	1	1.00	0.07	13	0.01
1.31	300	1	1.00	0.07	14	0.01	1.31	300	1	1.00	0.07	14	0.01	1.31	300	1	1.00	0.07	14	0.01
1.41	300	1	1.00	0.07	15	0.01	1.41	300	1	1.00	0.07	15	0.01	1.41	300	1	1.00	0.07	15	0.01
1.50	300	1	1.00	0.07	16	0.01	1.50	300	1	1.00	0.07	16	0.01	1.50	300	1	1.00	0.07	16	0.01
1.59	300	1	1.00	0.07	17	0.01	1.59	300	1	1.00	0.07	17	0.01	1.59	300	1	1.00	0.07	17	0.01
1.69	300	1	1.00	0.07	18	0.01	1.69	300	1	1.00	0.07	18	0.01	1.69	300	1	1.00	0.07	18	0.01
1.78	300	1	1.00	0.07	19	0.01	1.78	300	1	1.00	0.07	19	0.01	1.78	300	1	1.00	0.07	19	0.01
1.88	300	1	1.00	0.07	20	0.01	1.88	300	1	1.00	0.07	20	0.01	26.25	300	1	1.00	0.07	20	0.14

Appendix 3(e): Calculation of erosion using RUSLE for Latosolic Reddish Prairie (LRP).

Sub-humid (EI = 100)							Humid (EI = 110)							Super-humid (EI = 130)						
A	R	L	P	C	S	K	A	R	L	P	C	S	K	A	R	L	P	C	S	K
t ¹ ha ⁻¹ yr ⁻¹					%		t ¹ ha ⁻¹ yr ⁻¹					%		t ¹ ha ⁻¹ yr ⁻¹					%	
0.08	300	1	0.82	0.07	1	0.01	0.09	300	1	0.96	0.07	1	0.01	0.09	300	1	1.00	0.07	1	0.01
0.14	300	1	0.73	0.07	2	0.01	0.17	300	1	0.93	0.07	2	0.01	0.19	300	1	1.00	0.07	2	0.01
0.20	300	1	0.70	0.07	3	0.01	0.26	300	1	0.93	0.07	3	0.01	0.28	300	1	1.00	0.07	3	0.01
0.26	300	1	0.69	0.07	4	0.01	0.35	300	1	0.92	0.07	4	0.01	0.38	300	1	1.00	0.07	4	0.01
0.32	300	1	0.69	0.07	5	0.01	0.43	300	1	0.92	0.07	5	0.01	0.47	300	1	1.00	0.07	5	0.01
0.40	300	1	0.71	0.07	6	0.01	0.52	300	1	0.93	0.07	6	0.01	0.56	300	1	1.00	0.07	6	0.01
0.48	300	1	0.73	0.07	7	0.01	0.62	300	1	0.94	0.07	7	0.01	0.66	300	1	1.00	0.07	7	0.01
0.57	300	1	0.76	0.07	8	0.01	0.72	300	1	0.96	0.07	8	0.01	0.75	300	1	1.00	0.07	8	0.01
0.68	300	1	0.80	0.07	9	0.01	0.82	300	1	0.97	0.07	9	0.01	0.84	300	1	1.00	0.07	9	0.01
0.79	300	1	0.84	0.07	10	0.01	0.93	300	1	0.99	0.07	10	0.01	0.94	300	1	1.00	0.07	10	0.01
0.91	300	1	0.88	0.07	11	0.01	1.03	300	1	1.00	0.07	11	0.01	1.03	300	1	1.00	0.07	11	0.01
1.06	300	1	0.94	0.07	12	0.01	1.13	300	1	1.00	0.07	12	0.01	1.13	300	1	1.00	0.07	12	0.01
1.18	300	1	0.97	0.07	13	0.01	1.22	300	1	1.00	0.07	13	0.01	1.22	300	1	1.00	0.07	13	0.01
1.31	300	1	1.00	0.07	14	0.01	1.31	300	1	1.00	0.07	14	0.01	1.31	300	1	1.00	0.07	14	0.01
1.41	300	1	1.00	0.07	15	0.01	1.41	300	1	1.00	0.07	15	0.01	1.41	300	1	1.00	0.07	15	0.01
1.50	300	1	1.00	0.07	16	0.01	1.50	300	1	1.00	0.07	16	0.01	1.50	300	1	1.00	0.07	16	0.01
1.59	300	1	1.00	0.07	17	0.01	1.59	300	1	1.00	0.07	17	0.01	1.59	300	1	1.00	0.07	17	0.01
1.69	300	1	1.00	0.07	18	0.01	1.69	300	1	1.00	0.07	18	0.01	1.69	300	1	1.00	0.07	18	0.01
1.78	300	1	1.00	0.07	19	0.01	1.78	300	1	1.00	0.07	19	0.01	1.78	300	1	1.00	0.07	19	0.01
1.88	300	1	1.00	0.07	20	0.01	1.88	300	1	1.00	0.07	20	0.01	1.88	300	1	1.00	0.07	20	0.01

Appendix 4: The agro-climatic and sugarcane regions of Mauritius.



Appendix 5: Details of study sites earmarked for edge-of-plot field testing of the index developed for assessing P losses from sugarcane soils of Mauritius.

Field site	Agro-climatic region	Soil group	Date planted and fertilised	Fertilizer application rate (kg P ₂ O ₅ ha ⁻¹)	Fertiliser application method	Organic amendment application rate	Organic amendment application method
La Mecque	Sub-humid	Low Humic Latosol	August 2010	0	Buried in furrows	None	Not applicable
Belle Rive	Super-humid	Humic Ferruginous Latosol	August 2010	Not available	Not available	Not available	Not available
Domaine de L'Etoile	Super-humid	Humic Latosol	September 2010	Not available	Not available	Not available	Not available
Henrietta	Humid	Low Humic Latosol	March 2011	Not available	Not available	Not available	Not available
St Pierre	Humid	Humic Latosol	April 2011	Not available	Not available	Not available	Not available
Joli Bois	Super-humid	Latosolic Brown Forest	June 2011	0	Buried in furrows	None	Not applicable
Cent Gaulettes	Super-humid	Humic Ferruginous Latosol	October 2011	184	Buried in furrows	None	Not applicable
Beau Songes	Sub-humid	Latosolic Reddish Prairie	November 2011	215	Buried in furrows	None	Not applicable
Pierrefonds	Sub-humid	Latosolic Reddish Prairie	February 2012	420	Buried in furrows	None	Not applicable
Le Val	Super-humid	Latosolic Brown Forest	May 2012	0	Buried in furrows	None	Not applicable