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Agricultural Phosphorus Management for Environmental Protection: A Review

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Abstract

This review addresses research done over the past 40 years on different aspects of agricultural phosphorus (P) management for environmental protection. Inputs of P are essential for profitable crop production. Long-term application of P to agricultural soils has resulted in elevated levels of soil P. This accumulation may be desirable from an agronomic point of view but it represents a threat to freshwater quality. Indeed, P in runoff from agricultural land is an important component of non-point source pollution and can accelerate eutrophication of lakes and streams. Even very small amounts of P can raise the concentration above the critical value for eutrophication. Excessive eutrophication restricts water use for recreation, industry, and drinking due to the increased growth of undesirable algae and aquatic weeds. Current concerns facing the environmentally sound management of P in agriculture are similar worldwide and revolve around agricultural, economic, and environmental compromises associated with balancing productivity with environmental values. Agricultural P management strategies should be geared towards ensuring that P, a finite earth resource, is not wasted and those soils do not become so enriched with P that there is an unnecessary risk of too much P being mobilized to water from agricultural fields. Approaches, such as soil P testing and either P models or indices have been studied and implemented to mitigate agricultural P losses. Apparently, P indices are preferred instead of P models for this purpose because they are more flexible. A holistic management strategy is therefore suggested by using soil P testing in conjunction with either a P index or a P model.

Keywords

Mitigation Strategies, Non-Point Source Pollution, Phosphorus Indices, Phosphorus Models, Water Pollution

1. Introduction

The environmental significance of phosphorus (P) lies in its dominant role in accelerating eutrophication of aquatic ecosystems. Phosphorus is 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). However, the injudicious application of fertilizer P to soils for agricultural production could enhance eutrophication of freshwater when recalcitrant soil P is mobilized (Menezes-Blackburn et al., 2018). The most conspicuous impact of this cultural eutrophication is the creation of noxious, foul-smelling phytoplankton that reduces water clarity and harm water quality (Huang et al., 2017). In general, to maintain the quality of waters according to the United States Environmental Protection Agency, total P should not exceed 50 $\mu g \cdot L^{-1}$ in streams entering lakes/reservoirs, or 25 $\mu g \cdot L^{-1}$ within lakes/reservoirs (USEPA, 1986). 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 vast literature which exists on the functions and needs of P in agricultural production (Kleinman et al., 2015; Mardamootoo et al., 2021), a large amount of research is being done on every aspect of P in the environment, mostly related to water pollution and eutrophication (Osmond et al., 2019; Evans et al., 2020). The negative effects associated with eutrophication of surface waters are important from ecological, economic, and health perspectives.

In order to manage soil P to prevent eutrophication of waters, it is important to understand how and in which forms P is transported to water bodies so as to identify the best measures to mitigate agricultural P losses. Research has shown 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. In this review, we examined research findings on the sources of agricultural P, pathways for agricultural P loss to water, and speciation of P in runoff waters with a view to assessing the impacts of agricultural P on environmental pollution and identifying suitable mitigation measures.

The objective of this review is therefore firstly, to focus on the occurrence, movement, and associated problems of environmental P and secondly, to address best management practices of agricultural P ensuring optimum crop production with minimum environmental pollution. Aspects covered in the first part include inter alia eutrophication of water resources by P, acceptable P concentration in waters, movement of P in the landscape, a conceptual model for P transfer, and terminology for mobile P. The second part comprises soil P testing for crop production, management of fertilizer P, and estimation of P transport from agricultural land by either P models or P indices.

2. Methodology

The review originated from research that started almost simultaneously in Mauritius and South Africa, focussing on non-point P pollution of water by agricultural activities. In South Africa, the project was initiated and funded by the Water Research Commission. A multidisciplinary team from several institutions investigated the modelling of P dynamics in cropping systems at a field scale. The project in Mauritius was initiated by the Sugarcane Industry Research Institute and funded by the European Union under the African, Caribbean and Pacific Sugar Research Program. In this project, the developing of an index for soil P loss from sugarcane soils was investigated. A prerequisite for both projects was a thorough knowledge review.

Several search engines for academic research (e.g. Web of Science, Scielo, Agricola, Sciro, Science Direct, Google Scholar and Microsoft Academic Search) were used to compile a list of relevant scientific publications for each project. The searches were based on keywords where after abstracts were used to verify the relevancy of each publication for the independent knowledge reviews. For this review, the two knowledge reviews were integrated and then updated.

3. Phosphorus in the Environment

3.1. 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 1**), the trophic

Table 1. 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

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).

Over time (thousands of years), an initially clear (oligotrophic) lake's condition gradually deteriorates, with filling of sediment to eventually becoming a marsh (hypereutrophic), and then a dry land (Spiro & Stigliani, 2002). Low biological productivity is associated with a 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 & Beegle, 2001). Anthropogenic inputs of P may occur from sewage, agricultural runoff (especially where fertilizers 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 & 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 (N_2), 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 saline 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 (National Research Council, 2000). However when nutrient enrichment is sufficiently pronounced, the effects are invariably detrimental from an ecological, economic and animal/human health perspective. As highlighted by (Sharpley & Withers, 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 (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 water body 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 & 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 (Guidry et al., 2006). However, the economics of remediation remain a key limitation to substantial changes in agricultural management for environmental protection (Sharpley & 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 on 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 points and diffuse nutrient inputs to the water environment.

3.2. Acceptable Phosphorus 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 $\mu g \cdot L^{-1}$ to more than 200 $mg \cdot L^{-1}$ in some closed saline lakes but in most uncontaminated surface waters they vary from 10 to 50 $\mu g \cdot L^{-1}$ (Wetzel, 2001). As highlighted by the European Environment Agency, 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 μg P L^{-1} only (EEC, 2004). 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 μg P L^{-1} (USEPA, 1986). Most relatively pristine lakes in the USA are known to have surface waters that contain between 20 to 30 μg·L^{-1} total P. In general, surface waters that are maintained at 10 to 30 μg·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 μg·L^{-1} total P) and rivers/streams (varying between 8 to 40 μg·L^{-1} total P) in each ecoregion (USEPA, 2000a, 2000b). In general, however, the (USEPA, 2000a, 2000b) states that total P should not exceed 50 μg·L^{-1} in streams entering lakes/reservoirs, and 25 μg·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 μg·L^{-1} (Daniel et al., 1998).

The Australian Water Quality guidelines (ANZECC, 1992), which are comparable to the standards set by the (USEPA, 1988) recommend total P concentrations of 10 to 100 μg·L^{-1} for rivers and streams, 5 to 50 μg P L^{-1} for lakes and reservoirs and 1 to 10 μg P L^{-1} (as orthophosphate P) for coastal waters.

3.3. 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 & 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 or 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, 1985) 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 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 provide a pathway for the water and dissolved solutes to move through the soil (Sharpley & 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 clay 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 discussed later 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 & 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 or 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 water bodies will vary widely depending on the land from which it originated, on the transport route and on management practices (Haygarth & Jarvis, 1999). During transport, soluble and particulate P interacts 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).

3.4. Conceptual Model of Phosphorus 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 modelling P movement to surface waters and for developing risk assessment tools such as the P index, discussed in following sections. A conceptual model of P transfer as illustrated in **Figure 1** should incorporate the hydrological pathways, the P sources, the forms of P, the mobilisation processes and their inter-relationships in P transport.

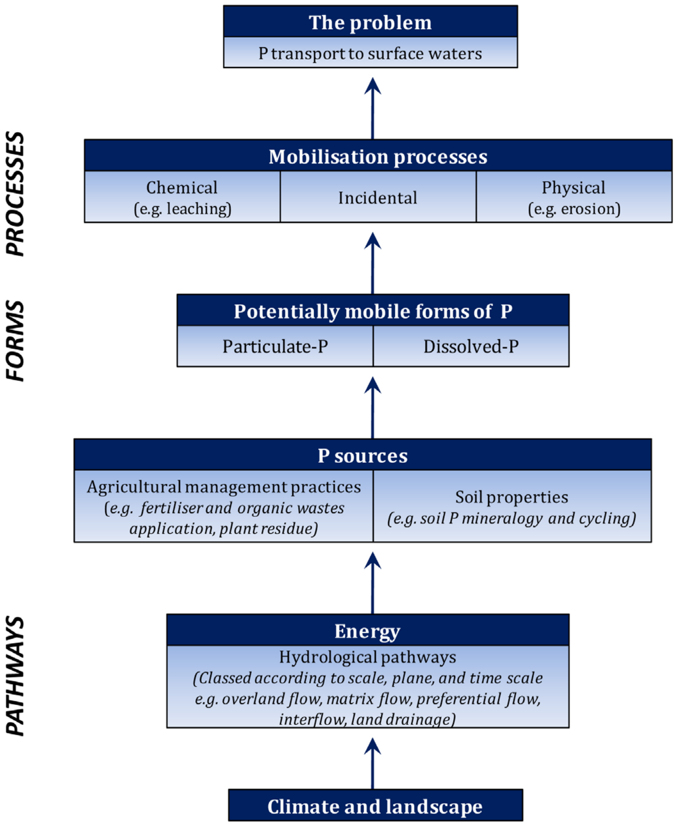

Figure 1. A conceptual model describing how non-point source P from agricultural land reaches surface waters, modified from (Haygarth & Sharpley, 2000).

(Haygarth & 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 also in terms of its potential for transport to water bodies. For example fields with either a high P status or a high erosive 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 & 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 1**). The conceptual model in **Figure 1** 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 fertilizers, 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 mobilized. The processes of mobilisation in the conceptual model, for the past, have 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 short-term P losses, for example when rainfall occurs shortly after application of P fertilizers and/or soil amendments resulting in high concentrations of P in runoff.

The hydrological transfer pathways are summarised in **Table 2** and illustrated in **Figure 2**. They are intended to cover a wide range of spatial scales in water

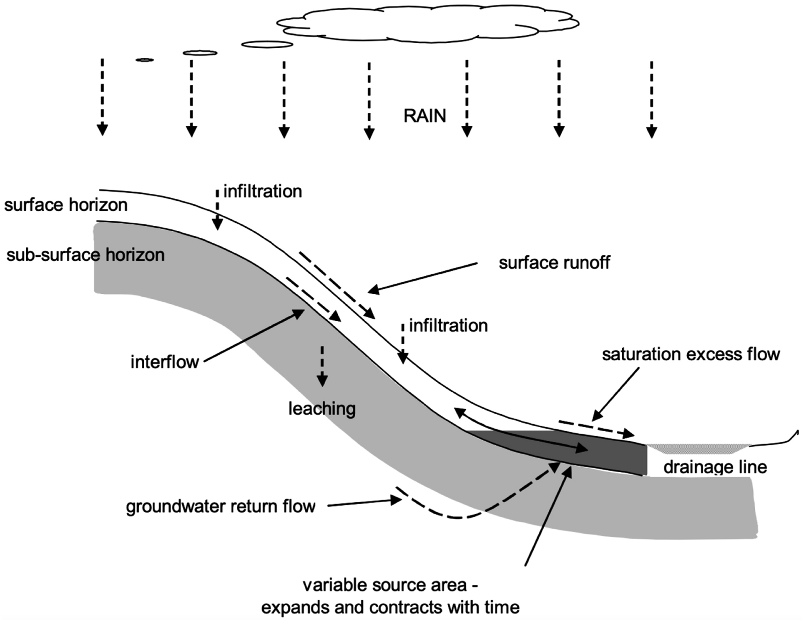

Figure 2. Basic components of hillslope hydrology (Dougherty, 2006).

Table 2. Hydrological processes, their approximate timeframes of occurrence and their variations in the plane of water movement, adapted from (Haygarth & Sharpley, 2000).

Term	Scale	Timescale	Definition
Interflow	Slope, field	Min, h[a]	*Lateral* flows below the soil surface.
Matrix flow (also by-pass flow)	Soil		*Vertical* 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 *lateral* 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]	*Lateral* 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.

flow, as well as the variations in plane and time scale of the flows (Haygarth & Sharpley, 2000). In their conceptual model, (Haygarth & 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** and **Figure 1**). 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).

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**). The variable source area concept when extended gives rise to the critical source area concept discussed in following sections.

3.5. Terminology for the Mobile Forms of Phosphorus

Research on P transfer from soils to waters, as can be seen from the conceptual P transfer model described in previous section, is multidisciplinary in nature. One consequence is that terminologies used for the different forms of P concerned with transfer to water bodies have varied according to research disciplines and have lead to misinterpretations and confusions. As a remedy, (Haygarth & 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 & Jarvis, 1999).

Although in previous sections, 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 0.45 µm membrane filtration is adopted to separate the "dissolved" from the "suspended" forms of P. However as reviewed by (Haygarth & 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 0.45 µm 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 & 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-Riley procedure has been shown to also determine loosely bound inorganic and organic forms of P, as a result of either molybdenum-enhanced hydrolysis or hydrous ferric oxide-orthophosphate reactions, or 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 & 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-Riley method 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-Riley method. 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 & 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 3**, has been proposed by (Haygarth & 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 & Sharpley, 2000) in relation to those commonly encountered in the literature is given in **Table 3**.

Another term that is frequently encountered when discussing water quality issues related to P is bioavailable P (also algal-available P). The bioavailable P

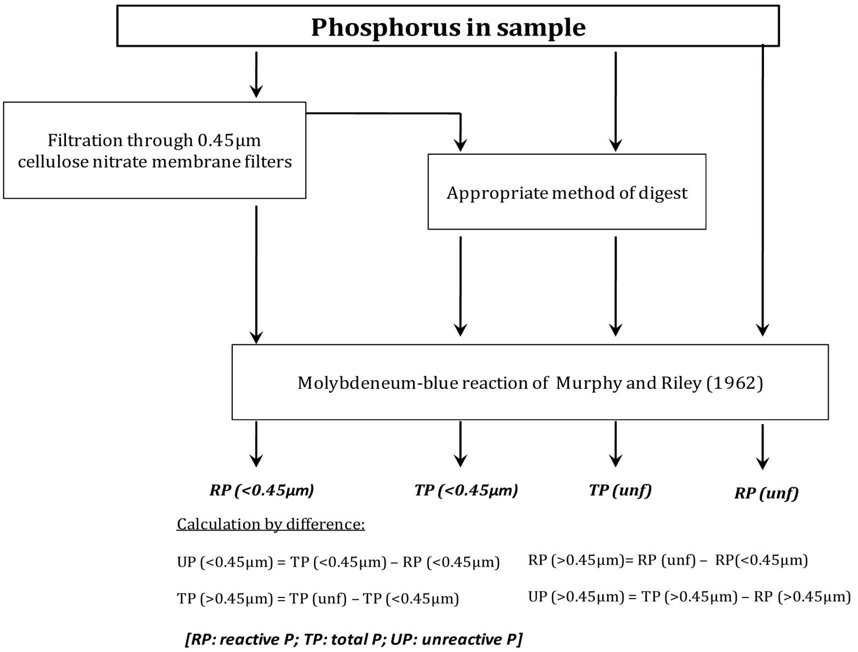

Figure 3. Nomenclature for the operationally defined forms of mobile P, adapted from (Haygarth & Sharpley, 2000). Samples are defined specifically according to filter size. The suffix in parentheses relates to the micron size of the filter used.

Table 3. Suggested methodologically defined terms of P forms in waters with their equivalence commonly found in the literature, adapted from (Haygarth & Sharpley, 2000).

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

[a][RP: reactive P; TP: total P; UP: unreactive P]; [b]May not necessarily be correct.

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 4**).

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, 1993).

4. Management of Agricultural Phosphorus for Environmental Protection

The continued input over the years of fertilizer 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 & Withers, 1994). Though the loss of agricultural P is not of economic importance to a farmer, it has often leaded to the deterioration of water quality due to accelerated eutrophication which has significant off-site economic impacts (Sharpley & Withers, 1994). Nutrient removal at

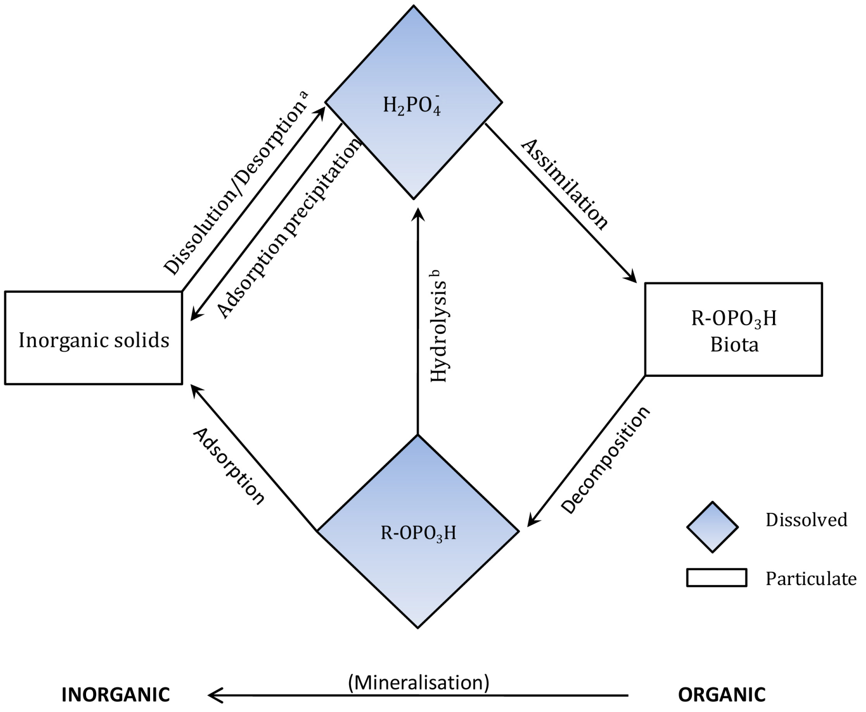

Figure 4. The biochemical P cycle in aquatic systems (Glennie et al., 2002). (^{a}reductive, photochemical, pH variability; ^{b}enzymatic, photochemical, pH variability).

the source, although expensive is an efficient method to reduce P concentrations in runoff from fields. As reviewed by (Herron et al., 2016), there exist several industrial by-products that have the potential to bind large amounts of P by a variety of chemical reactions, depending on the material used. Such materials include fly ash, steel slag, acid mine drainage residuals, drinking water treatment residuals and flue gas desulfurization gypsum.

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 take several years to be 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 fertilizer applications) and the P must be transported to a sensitive water body (Sharpley & Tunney, 2000). As summarized 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 fertilizer P applications with crop needs and yield goal potentials, the accumulation of P in soil will be minimized, 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 & 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., 2001). 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.

4.1. Soil Phosphorus Testing

As stated above, 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., 2001). 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, see (Bruulsem et al., 2019) for detail, and can therefore be considered polluting (**Figure 5**). 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.

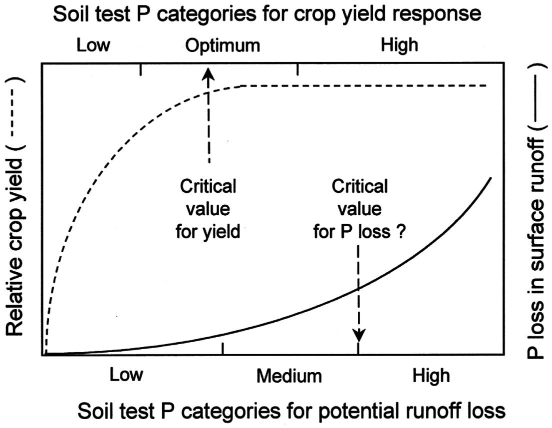

Figure 5. Interpretation of soil test P levels for agronomic and environmental purposes (Sharpley et al., 2001).

A variety of soil extractants, e.g., Mehlich-3 and Olsen have been evaluated as indicators of P loss potential, by relating the soil P extract to the P in surface runoff or subsurface leachate (Sims & Kleinman, 2005; Carmo Horta & Torrent, 2007; Mardamootoo, 2009). Numerous studies e.g. (Sharpley et al., 2001; 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 has recently been fertilized. This finding has led to extractions of surface soils with water or 0.01 M 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, 2000).

The split line model is one innovative approach proposed by (McDowell & 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 a 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 & Kleinman, 2005; Carmo Horta & Torrent, 2007; Mardamootoo et al., 2012). However, in other instances, as reported by (Moody, 2011), there is only a single linear relationship between the degree of soil P saturation and the P extracted in 0.01 M 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 recognized that they have limited application in environmental P risk assessment (Sharpley, 2000). 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 & Sharpley, 1998; Gburek et al., 2007). A more holistic approach that 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 (Sharpley, 2016). Such holistic approaches would include modelling and the P indexing system which are discussed in subsequent sections.

4.2. Best Management Practices

4.2.1. Management of Fertiliser Phosphorus Sources

The most common sources of P in agricultural runoff are mineral fertilizers 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), 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 fertilizer 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 fertilizer P was much higher than from fields where comparable rates were incorporated 5 cm below the soil surface. Fertilizers 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, fertilizers 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 & Schultz, 2002).

As mentioned, apart from the dose, form of P used and method of placement, timing of the fertilizer P applications to coincide with dry weather had been observed to be of fundamental importance in reducing incidental P loss from the fertilizers, 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., 2001). 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., 2001). (Zaimes & 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.

The ultimate goal when managing P for environmental protection is to prevent the sources of P from being transported. From this perspective, the strategies that have been developed by (Sharpley et al., 2000) to reduce P transport can be categorized 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 through flow (Sharpley, 2000). 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 & Withers, 1994; Sims et al., 2002; Sims & 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 & 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 centimeters due to the repeated surface applications of fertilizer or manure P without the mixing associated with ploughing (Sims et al., 2002; Sims & Kleinman, 2005). To avoid an accumulation of P in the upper few centimeters of soil, the intermittent use of conventional tillage to mix the topsoil with the subsoil has been recommended (Sims et al., 2002; Sims & 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 et al., 2002; Sims & Kleinman, 2005). The practicing of conservation tillage may have therefore unintended consequences and implications for sustainable P management as detailed by (Dodd & Sharpley, 2016).

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 & 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 (Singh & Lal, 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 (Singh & Lal, 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 & 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 immobilization of the P by the same vegetation composing the buffers (Sharpley et al., 1994; Sims et al., 2002; Sims & Kleinman, 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, 2000; Zaimes & Schultz, 2002; Sheppard et al., 2006).

4.2.2. Modelling Phosphorus 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 as illustrated by the conceptual model (**Figure 1**) and hillslope hydrology (**Figure 2**). Tools that 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). Models of this nature are very diverse and hence a generalized schematic diagram of their operation is not presented.

In fact, many of the basic approaches of modelling P transport were developed in the 1980s and despite other advancements in modelling, such as graphic user interfaces and the use of geographic information system (GIS) layers, the basic approaches to modelling P movement have changed very little (Radcliffe & Cabrera, 2007). Modelling 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 modelling 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 and leaching, organic and inorganic fertilizer 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 (Sharpley et al., 1994). Recently, a P module was developed and evaluated for the Root Zone Water Quality Model (RZWQM2) by (Sadhuklan et al., 2019).

Export coefficient models, for their part, rely on simple and empirical formulations to represent transport processes, for example, the use of a runoff coefficient for generation of runoff from rainfall. One example of the export coefficient models is the Generalized 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:

Runoff dissolved P in $\text{mg}\cdot\text{L}^{-1}$ = (available fertilizer P in mg) × (% fertilizer P released) × [(P distribution factor)/(total rain during the event in L)].

The P distribution factor is a factor without units which is obtained in the following way:

P distribution factor = 0.034 exp [(3.4) (runoff to rain ratio)].

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

The use of models such as SWAT to identify critical source areas has 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 modelling 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 emphasing 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. However, these models are of importance because most of the management decisions to optimize agricultural P use for cropping and hence protecting the environment happen at field-scale level (Annandale & Du Preez, 2005). For example in South Africa, the local Soil Water Balance (SWB) model was adapted for P modelling in soil-plant systems (Van der Laan, 2009; Van der Laan et al., 2012). Models of this kind require in many instances specific inputs such as the susceptibility of soil types to agricultural P losses. Parameters were formulated by (Van der Laan et al., 2009) for the SWB model, using the South African Soil Classification System (Soil Classification Working Group, 1991). These parameters could also be of value for catchment-scale models such as the South African agro-hydrologically model ACRU (Schultze, 2007), using the hydropedology grouping of South African soil forms as proposed by (Van Tol and Le Roux, 2019).

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 modelling 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 these is the P index approach which is reviewed the next section.

4.2.3. Phosphorus 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 are those agricultural fields or part of them which are most vulnerable to P loss in the watershed (Berzina & Sudars, 2010). 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 (Kort et al., 2007)). 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 adopted the P index to improve the management of agricultural P. Recently, (Mardamootoo, 2015) developed also an index for P loss from sugarcane fields in Mauritius.

The original P index was developed by (Lemunyon & Gilbert, 1993) with the initial goals of: 1) Assessing the risk of P transport from a field to a water body; 2) Identifying the critical factors that influence P loss; 3) Helping to select management practices that would decrease P loss from a site. In the following two decades since it was introduced by (Lemunyon & 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 & Shober, 2012). The original P index comprised only eight characteristics namely soil erosion, irrigation erosion, runoff class, soil P test, P fertilizer application rate, P fertilizer application method, organic P source application rate and organic P source application method and the five value categories of negligible, low, medium, high and very high (**Table 4**).

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 5**). 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.

The initial P index as proposed by (Lemunyon & 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 & Gilbert, 1993) was 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 as return period/distance among the site transport characteristics. A return period describes the recurrence interval of

Table 4. Framework of the original P index as proposed by (Lemunyon & Gilbert, 1993).

	P loss rating (value)				
Site characteristic (*Weighing factor*)	**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 fertilizer application rate, lb P acre^{-1} (*0.75*)	None	<15	16 - 40	41 to 65	>65
P fertilizer 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

Table 5. Site vulnerability chart for interpretation of the original P index (Lemunyon & Gilbert, 1993).

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

*P index = ∑(Site characteristic P loss rating value × weight).

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 & 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 & 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 & 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 6**. 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 (Haygarth et al., 1998; Reid et al., 2012).

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 modelling the three major pathways of P movement from fields to water, namely erosion, rainfall runoff and snowmelt (**Figure 6**).

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/fertilizer 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

Table 6. Phosphorus source and transport factors (Reid et al., 2012).

P form	P source	P transport
Particulate P	Soil P	Surface runoff
Dissolved P	Fertilizer 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

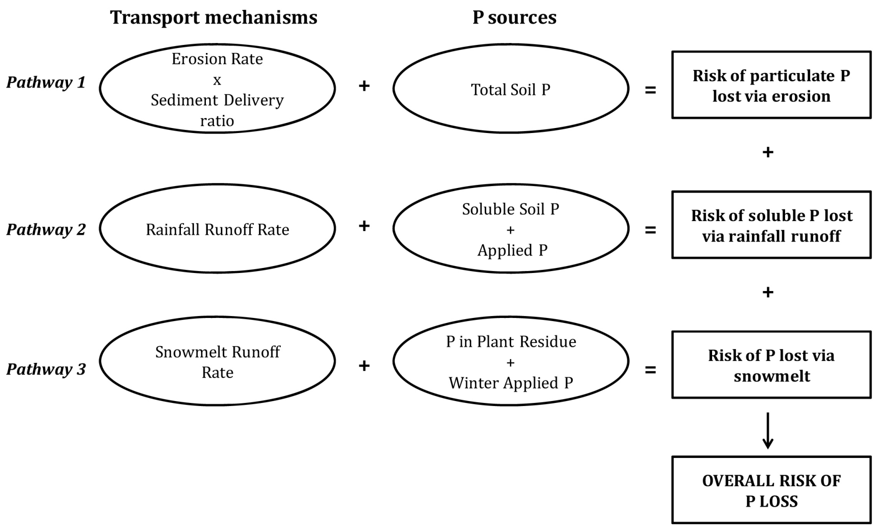

Figure 6. The Minnesota P index model (Lewandowski et al., 2006).

(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 & 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.

The P index for Mauritius is based on historical data (Mardamootoo et al., 2010, 2012) and data generated by rainfall simulation (Mardamootoo et al., 2013, 2015). For the latter, effects of rainfall intensity, field slope and water erosion on P movement were quantified at 20 diverse sugarcane fields. The results showed that runoff and hence erosion varied across soil types. Increasing rainfall intensity and field slope enhanced P mobilization. Total runoff P was stronger correlated with suspended sediments ($r^2 = 0.92$) present in runoff waters than with runoff volume ($r^2 = 0.49$), indicating that about 90% of total P loss was in the particulate form regardless of soil type, rainfall intensity and field slope. (Mardamootoo, 2015) based this P index for Mauritius on sites' vulnerability to P loss by accounting for source (dissolved P, particulate P, P application rate, method of application and application timing) and transport (precipitation, surface runoff and soil erosion) factors.

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 & Shober, 2012). (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). (Thomas et al., 2016) suggested that the use of high resolution data could be considered to identify critical source areas with a P index.

5. 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 water body its removal becomes tedious and very expensive. Accordingly, much of the research done has focussed on a better understanding of P movement from the agricultural fields to water bodies to prevent accelerated eutrophication by the P loading. This review has shown that though regular soil testing, selection of appropriate P application rates to meet reasonable crop yield expectations, and prescriptive application of mineral fertilizer using methods that maximize 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 may overwhelm source factors (e.g. high P soil) in determining P losses.

Two holistic approaches have been proposed and adopted for the elaboration of improved management practices in agricultural fields, namely P modelling, and the P index concept. In view of the shortcomings of P modelling 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 into 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 such as soil type, climate, and land management. It is evident that future research strategies should be geared towards more interdisciplinary studies which involve soil scientists, hydrologists, agronomists, limnologists, animal scientists, economists, and social scientists.

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Conflicts of Interest

The authors declare no conflicts of interest regarding the publication of this paper.

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