Using a Simple Hydrologic Model to Link Management to Nutrient Concentrations and Loads in Runoff

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EXTENDED ABSTRACT

Soil phosphorus (P) poses a threat to aquatic and marine ecosystems. It is important that P concentrations are kept low to avoid eutrophication of waterways, as occurred in the spectacular and costly blue-green algal blooms in the Barwon and Darling River systems in 1990-1991. The management of water quality often requires comprehensive, catchment-wide approaches. Simulation models can play a role in alerting managers to the sources of P pollution, and ideally can help identify means of reducing that pollution.

There are many models of sediment and nutrient dynamics in catchments. Yet despite the importance of analysing relationships between nutrient exports and management, few models facilitate this analysis at a biophysical level, where the mechanisms can be explored and understood. To provide a predictive tool, we have added quantitative information about P export to an existing model of water balance and sediment export. Although P has been modelled in this case, other nutrients could be modelled with this approach as long as relevant literature is available for developing the necessary mathematical relationships. The input data include weather, soil, vegetation and land management. Runoff (mm), sediment concentration (g/L) and P export (kg/ha) are calculated on a daily basis.

We have used published data concerning P chemistry to estimate sub-classes of total P export, including bio-available particulate P, dissolved P, and bio-available P. Complex relationships sometimes arise from simple inputs and relationships through the effects of parameters such as soil P sorption and variables such as sediment concentration.

While the model is at an early stage of development, experts have indicated that it shows skill in reproducing expected relationships between forcing variables and the simulated concentrations and loads.

Analysis of land use and site effects at Clifton and Kingaroy in the northern grainbelt of Australia demonstrates the utility of this approach. For example, in scenarios where average annual runoff (mm/year) were similar, large differences were simulated in erosion (t/ha/year) and total P export (kg P/ha/year, Figure 1). Also, there was little correlation between average export of total P and average bio-available P (kg/ha/year) exported. There was, however, a modest correlation between average concentration (mg/L) and average export of bio-available P (kg/ha/year).

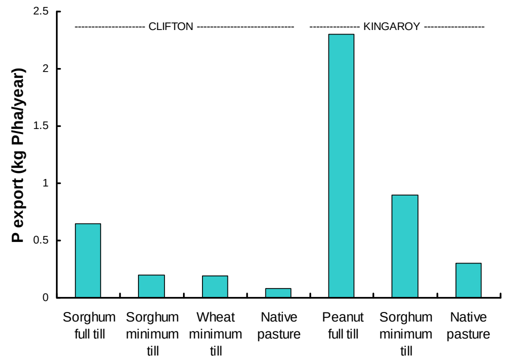

Figure 1. Average annual export of total P in runoff (kg P/ha/year) from various farming systems at Clifton and Kingaroy.

The input parameters for these types of simulation are now available for many situations. This, combined with the realism of these simulations (compared to simple factorial models), assures the continued use of this type of model.

1. INTRODUCTION

P movement into waterways has serious impacts on surface water quality (Sharpley *et al.* 1994). In Australia, P contamination of freshwater was associated with blooms of blue-green algae in the Murray-Darling River system in the 1980s and 1990s. One bloom in the Barwon and Darling River was more than 1000 km long (Davis and Koop 2006). In marine ecosystems, such as Moreton Bay, near Brisbane, and in the lagoon of the Great Barrier Reef, soil-derived P almost certainly plays a role in the development of blooms of toxic blue-green algae (Albert *et al.* 2005).

The challenges of maintaining low P concentrations in Australian waterways are likely to be quite different to those in Europe and the USA. In Australia, non-point sources from agriculture and other sources are more important (Sharpley 1995, Herath 1997). For example, McKee *et al.* (2000) found that more than 97 percent of P came from diffuse agricultural sources in the Richmond River estuary (draining a predominantly agricultural catchment). They also showed that the inter- and intra-annual variability of P inputs and exports from this sub-tropical estuary were higher than in temperate estuarine systems in the northern hemisphere.

Varied and complex landscape processes generate P pollution. Material sources of P include fertilisers, native soil minerals and manures. P in runoff is often categorised as "particulate", "filterable" or "dissolved". These are also divided into fractions based on reactivity or bioavailability. For example, Nash and Murdoch (1997) found that P in runoff from dairy pastures was mostly soluble (rather than particulate), and reactive. Ninety three % of total P passed a typical filter (0.45 µm) and of that 91 % reacted with a mixture of ascorbic acid and molybdate. In another case, Nelson *et al.* (1996) found that only 14 % and 24 % of total P export from two grazed catchments was filterable. It is hoped that a dynamic model will reproduce and explain some of the variability observed between catchments and over time.

Unfortunately, the bioavailability of P in runoff is not simply defined or measured because there are many biotic and environmental factors that affect P availability and uptake. However, the amount of P extractable with 0.1 M NaOH is a useful indicator of bioavailability, as it corresponds closely to P uptake by common freshwater algae (Sharpley *et al.* 1991, 1993).

In agricultural systems with ample soil cover, such as grassed areas with minimal grazing, the bioavailable fraction of exported P is usually dominated by soluble P (Sharpley *et al.* 1992, Sharpley *et al.* 1993). This is because areas with high cover typically have low sediment concentrations in their runoff. The reverse is also true; that as cover is reduced and sediment concentrations rise, so does the contribution of particulate P to bio-available P.

To explore these relationships, and the potential utility of the model, several scenarios concerning P export from agricultural activities were analysed. Emphasis was given to linkages between soil and landscape characteristics, land management, erosion and the quantity, concentration and type of P exported.

2. METHODS

A daily timestep model (HowLeaky?, Rattray *et al.* 2004) was used to simulate crop growth and the components of the water balance. It is derived from PERFECT (Littleboy *et al.* 1996).

There are several differences between algorithms used to simulate water balance processes in PERFECT and HowLeaky? The main differences are that, in PERFECT: (i) crop cover is a linear function of leaf area index (LAI), up to 100% cover. In Howleaky?, cover has a non-linear relationship with LAI (cover = 100 * (1-$e^{-c.LAI}$) where c is 0.6) (ii) deep drainage uses the algorithm from the Gleams model (Leonard *et al.* 1987), whereas HowLeaky? drains all available water in a soil layer, up to a defined maximum (mm/day) (ii) potential soil evaporation is a function of LAI (Ritchie 1972), whereas HowLeaky? equates potential SE with the difference between potential evapotranspiration and transpiration (ie unsatisfied evaporative demand) (iv) dry matter accumulation is a non-linear function of water stress, whereas it is linear in HowLeaky? (v) leaf area accumulation is a non-linear function of water stress, whereas it is linear in HowLeaky?, and (vi) crop residue declines exponentially with time, whereas HowLeaky? uses a dynamic algorithm based on rainfall and temperature similar to that used in the SWAT model (Schomberg *et al.* 1994, Steiner *et al.* 1999).

HowLeaky? partitions rainfall into infiltration and runoff according to a modification of the runoff curve number method (Williams and LaSeur 1976). This approach was used in the PERFECT model (Littleboy *et al.* 1996).

2.1. The P submodel

The P model developed for HowLeaky? calculates total P on a daily basis from Equations 1 and 2.

$$TP\ (kg\ P/ha) = DP\ (kg\ P/ha) + PP\ (kg\ P/ha) \quad (1)$$

$$BAP\ (kg\ P/ha) = DP\ (kg\ P/ha) + BPP\ (kg\ P/ha) \quad (2)$$

where TP is total P exported (kg P/ha), DP is the dissolved P exported (kg P/ha) and PP is the particulate P exported per day (kg P/ha).

According to this model, P is transported in two forms: dissolved (DP) and particulate (PP). PP includes P sorbed by filterable soil particles and organic matter. PP is by far the larger amount of P transported from cultivated land under normal practices. While DP is immediately available for biological uptake, PP provides a large reservoir of P for aquatic biota. Together, DP and the fraction of PP that is bio-available (BPP) constitute total bioavailable P (BAP) (Tunney *et al.* 1997).

DP (kg P/ha) is the product of the concentration of dissolved P (DPconc, µg/L) and the amount of runoff (L/ha). DPconc is calculated from a function reported by Sharpley (1995) where DPconc (µg/L) is dependent on the degree of saturation (Psat, %) of the soil components that sorb P. We modified Sharpley's function slightly, so that there is no Y intercept, which eliminates the calculation of negative P concentrations at low Psat (Equations 3 and 4). These equations were tested and found to be valid for data collected by Davies *et al.* (2006) in dryland and irrigated catchments in dairy farmlets in South Australia.

$$DPconc = 10 \times Psat \qquad Psat < 5\% \quad (3)$$

$$DPconc = 50 + 30 \times Psat \qquad Psat > 5\ \% \quad (4)$$

where:

$$Psat = Extractable\ soil\ P / P\ sorption\ maximum \times 100 \quad (5)$$

$$DP\ (kg/ha) = DPconc\ (\mu g/L) \times Runoff\ (mm) \times 10^{-5} \quad (6)$$

The loss of PP is based on erosion-generated sediment concentrations (Sedconc, g/L) in runoff. Other factors are the total P in soil (Ptot, Kjehdahl P method, mg/kg) and a delivery ratio (DR) of the total hillslope erosion (Equations 7 and 8). This is the same approach used by Sharpley (1995). The delivery ratio (DR) is the ratio of sediment delivered to a waterway or basin to the total amount of sediment eroded from a slope, bay, or paddock. In the examples below, it is assumed to be 10%, which is a typical value for paddocks in the grainbelt of Queensland.

An enrichment ratio (PPer) is also required to account for differential deposition of P-rich and P-poor materials. In general, larger particles (e.g. sand grains) have low P status and high deposition rates, while small particles (e.g. clay particles) remain entrained, and increase the P status of sediment relative to eroded material. PPer is the ratio of PP concentration in the sediment delivered to the waterway (mg P/kg) to the bulk soil P concentration (mg P/kg).

$$PP\ (kg/ha) = Sedconc\ (g/L) \times Ptot\ (mg/kg) \times PPer \times Runoff\ (mm) / 10^5 \quad (7)$$

where

$$Sedconc\ (g/L) = Erosion\ (kg/ha) / runoff\ (mm) / 10 \times DR \quad (8)$$

The biological active fraction of PP (BPP) is calculated from Pa, the bio-available fraction of total P in the sediment (Equations 9 and 10). $P_{(0.1\ M\ NaOH)}$ is the test result for sediment extraction using 0.1 *M* NaOH to estimate bio-available P.

$$Pa = P_{(0.1\ M\ NaOH)} / Ptot \quad (9)$$

$$BPP = PP \times Pa \quad (10)$$

Where $P_{(0.1\ M\ NaOH)}$ has not been measured, it may be estimated by assuming that the bio-available P concentration is related to the concentration of P measured by the widely used Colwell (1963) method (Equation 11). Because the Colwell (1963) method uses a less aggressive extractant (0.5 *M* sodium bicarbonate) and process than the sodium hydroxide method, it seems likely that the constant shown in Equation 11, below, should be much greater than 1. However, Uusitalo and Ekholm (2007) indicated that algae extracts 120 to 150 percent of P extractable by the anion exchange resin (AER) method, which extracts considerably less P than the Colwell method (data from McBeath *et al.* 2007 for Australian soils, not shown). Therefore Pa may be less than the value of 1.2 assumed here.

Quantitative use of this model will require further study and calibration or replacement of this relationship. Ideally, adsorption-desorption relationships would be used to describe the bio-availability of P in the sediment, but the necessary parameters are not yet available for a range of Australian soils and sediments.

$$Pa = P_{(Colwell)} \times 1.2 / Ptot \quad (11)$$

2.2. Land management analysis

Some effects of land management on P concentrations and export were assessed in hypothetical scenarios based on cropping systems in southern Queensland. Four scenarios were simulated concerning cropping on a Vertosol (Isbell 2002) at Clifton, on the eastern Darling Downs: (i) sorghum with traditional tillage (a summer crop with low stubble cover), (ii) sorghum with zero tillage (a summer crop with high stubble cover), (iii) wheat with zero tillage (a winter crop with high stubble cover), and (iv) native pasture (perennial vegetation with high ground cover).

At Kingaroy, three scenarios were simulated on a Ferrosol (Isbell 2002): (i) peanut (a summer crop with low stubble cover), (ii) sorghum with zero tillage (a summer crop with high stubble cover), and (iii) native pasture (perennial vegetation with high ground cover). Peanut cropping involves much more soil disturbance and stubble removal than sorghum cropping. Full till is simulated by regularly removing residue biomass. The timing of tillage is rainfall-dependent, except for tillage at planting. Minimum till only reduces residues at planting.

Soil parameters used in the model that relate to the water balance and P model are shown in Table 1 and Table 2, respectively. Parameters relating to erosion (such as slope) were measured in a basaltic upland at "Greenwood" near Acland (approximately 60 km north of Clifton) and on a Ferrosol at the Redvale Research Station near Kingaroy.

Table 1. Depth and volumetric moisture content (%) of soil layers simulated in the two soils.

Parameter	Vertosol				
Bottom of layer (cm)	15	30	60	90	120
Saturation (%)	51	55	53	51	47
Drained upper limit (%)	58	49	48	47	43
Lower limit (%)	26	26	30	32	34

	Ferrosol		
Bottom of layer (cm)	15	35	180
Saturation (%)	50	50	45
Drained upper limit (%)	33	37	37.5
Lower limit (%)	21	24	23

Table 2. Soil parameters for the P model. These are for the surface layer of the soil. Enrichment ratio data from M. Silburn (pers. comm.) for vertisols on the Darling Downs, and D. Rattray (pers. comm.) for the Ferrosol at Kingaroy.

Parameter	Soil type	
	Vertosol 1	Ferrosol
Ptot (mg/kg)	500	2500
$P_{Colwell}$ (mg/kg)	15	40
Psorption (mg/kg)	250	1000
Delivery ratio (DR, %)	10	10
Enrichment ratio (PPer)	1	2

3. RESULTS AND DISCUSSION

Average annual runoff (mm/year) was broadly similar for all combinations of site and land use except the native pasture at Clifton where runoff was less than the other sites (Figure 2). This is because the expected increase in transpiration due to green, perennial ground cover are maximised at Clifton where the soil is deep and has high PAWC, while at Kingaroy, the soil has low PAWC. This leads to higher infiltration (through being drier on average), higher deep drainage and less transpiration per unit of rainfall (data not shown). The simulated water balances are consistent with landholder observations that all farming systems, including pastures and agroforestry systems, have low rates of runoff and high rates of deep drainage.

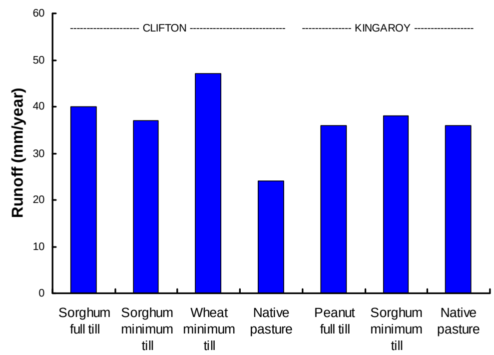

Figure 2. Average annual runoff (mm/year) from the various farming systems at Clifton and Kingaroy.

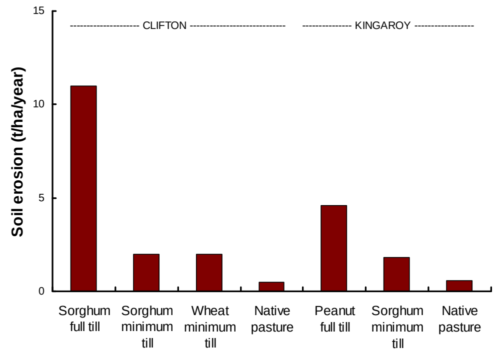

Figure 3. Average annual soil erosion (t/ha/year) from the various farming systems at Clifton and Kingaroy.

Estimates of soil erosion (t/ha) from the various scenarios are shown in Figure 3. Land use has a large effect on the amount of erosion, as does the location. The difference due to land use is mainly associated with differences in crop and residue cover, as affected by the system of tillage, while the site differences are mainly due intrinsic site properties, including slope (5% at Clifton and 2% at Kingaroy). The rates of erosion at Clifton for minimum till wheat and full tillage sorghum were similar to those measured in a similar landscape and with a similar rainfall pattern at "Greenwood" by David Freebairn (pers. comm.).

P exports at Clifton are lower because the soil has much less total P (500 mg/kg), and no enrichment of P in the sediment (PPer = 1, Table 2), while the soil at Kingaroy is high in total P (2500 mg/kg) and is enriched in the sediment (PPer = 2, Table 2).

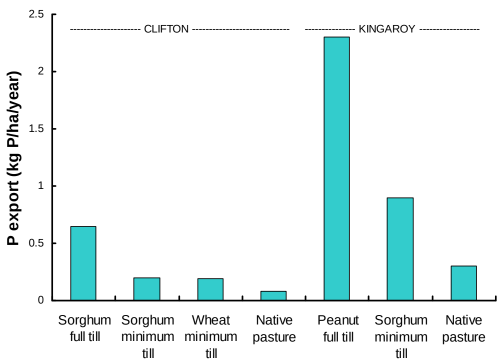

Figure 4. Average annual export of total P in runoff (kg P/ha/year) from the various farming systems at Clifton and Kingaroy.

Two additional factors were calculated for the scenario of Clifton with minimum tillage and wheat cropping: (i) the variability of dissolved and particulate P exports over time, and (ii) the relationship between of export amount on export composition (dissolved and particulate P). The results are shown in Figure 5 and Figure 6, respectively.

Figure 5 shows that P export is highly variable from year to year, and that P export involves both dissolved and particulate P in most years. Only rarely was there a dominance of one type or the other. Exceptional years in Figure 5 include 1996, where the dissolved amount was only 33% of the total, and 1999, 2000 and 2003, where the dissolved amount was more than 75% of the total.

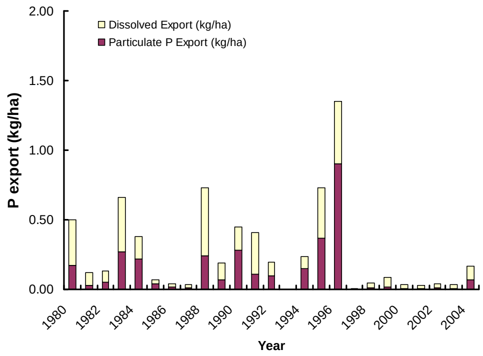

Figure 5. The quantity of P exported in solution and in particulate matter for the scenario of wheat cropping with minimum tillage at Clifton.

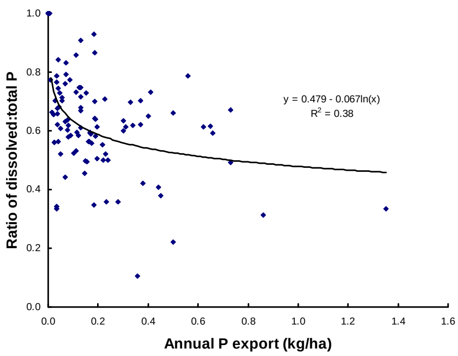

Figure 6. The ratio of dissolved to total P export and annual P export from the scenario of wheat cropping with minimum tillage at Clifton.

Figure 6 shows that the relationship between export and P type (dissolved or particulate) is not strong (R^2 = 0.38). The confounding influence of runoff volume masks any such relationship that might exist if P export and the fractions were expressed as concentrations. Figure 6 shows that the ratio appears to be restricted in range only when P exports are above approximately 0.6 kg/ha/year.

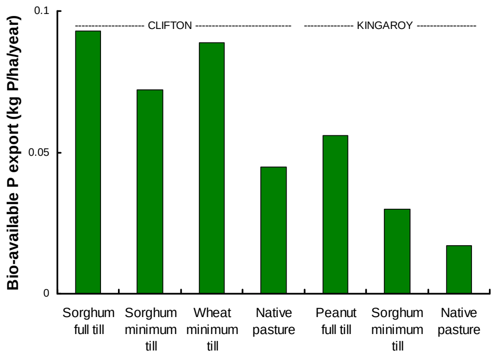

Figure 7. Average annual export of bio-available P in runoff (kg P/ha/year) from the scenarios at Clifton and Kingaroy.

At Kingaroy, the annual export amounts and concentrations of bio-available P were less than at Clifton (Figure 7 and Figure 8). For example, exports of bio-available P for a system of sorghum with minimum tillage were 0.03 and 0.072 kg/ha for Kingaroy and Clifton, respectively. A parameter that plays a key role in the differences is P sorption maximum (mg/kg), which is very high at Kingaroy (1000 mg/kg), due to the presence of P-sorbing constituents such as sesquioxides. Soils that have low sorption capacity will have correspondingly high bio-availability of P per unit of total P content.

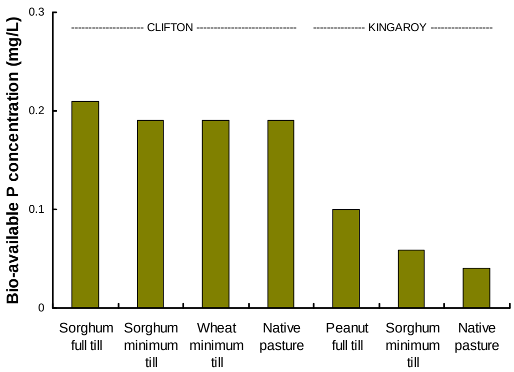

Figure 8. Average annual bio-available P concentrations (mg/L) for a variety of simulated farming systems at Clifton and Kingaroy.

At Kingaroy, there is likely to be a major response to land use change, while at Clifton it is likely that factors other than land use would need to be targeted to achieve reductions in the export of bio-available P (Figure 7). Factors, other than those associated with vegetation type, that can be examined with this model include slope length, soil P saturation (%), and deliver ratio.

A feature of the results concerning bio-available P is the lack of correspondence with simple measures, such as runoff, erosion and total P export. Although each equation in the P sub-model is simple, and there are relatively few of them, they create complex interactions that can obscure relationships between biophysical parameters, management choices and outcomes. There is clearly a need for models that emulate the complex emergent properties of real systems. Simpler models, including factorial models, will be at a relative disadvantage in terms of realism due to a lack of underlying functional relationships.

4. CONCLUSIONS

Overall, the results show that erosion, P export and bio-available P export vary from site to site, with soil type, and from year to year. These effects parallel the types of responses that have been observed in the field, including the sites specifically simulated.

Linking a daily time-step model of water balance and erosion to P transport has enabled the prediction of P export from climate, soils and land management information. This is a significant advance over simple models of sediment and nutrient transport that use “book” values or “hindcast” P export from existing land use. Furthermore, empirical research and models can only track land use effects after implementation. Prediction of outcomes requires a mechanistic model such as the one described here.

However, simple models have persisted for a variety of reasons. One reason has been that applications of models such as this have often been time-consuming or infeasible because of a lack of readily-available data, or a lack of skilled modellers. Improvements in data availability and the ease of using these models make routine applications, such as those shown above, readily achievable, even for novice modellers.

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