

# Available soil phosphorus, phosphorus buffering and soil cover determine most variation in phosphorus concentration in runoff from pastoral sites

M. R. Hart · P. S. Cornish

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**Abstract** Phosphorus (P) runoff from agricultural land contributes to elevated P concentrations in surface water. Although P concentrations in runoff are often higher when soil P is increased, attempts to quantify this relationship in field studies have been inconclusive, either because of confounding within experiments or methodological differences between experiments. The present study attempted to address these issues using simulated rainfall at 15 pastoral locations with varied lithology, land-use intensity, plant cover and soil-P properties in SE Australia, at a total of 38 on-farm sites, each with paired plots. At 27 sites, one of each pair was covered with 70 % shade-cloth to reduce rainfall energy. Although these were uncultivated pastoral sites and surface soil erosion rates were all relatively low, eroded surface soil had sufficiently high concentrations of P to make a major contribution to total P in runoff from some sites, even when plant-available soil P was low. We conclude that greater vegetative cover is required to manage P in runoff than to manage soil erosion. When soil was

adequately protected against erosion by either high vegetative cover or shade-cloth, greater than 90 % of the variation in both dissolved reactive P and total P was explained by two commonly measured soil variables: soil-test P (bicarbonate extract) and P-buffering index (where P was in the range where plants may respond to fertiliser). It appears that available soil P can be raised with fertiliser to improve agronomic production without fear of significantly increasing P losses in runoff provided soil test P remains below the agronomic critical concentration defined by the soil P-buffering index. These findings have broad relevance for environmental risk assessment and management where the data required for more complex procedures are unavailable.

**Keywords** Soil phosphorus · Phosphorus buffering index · Rainfall simulator · Surface runoff · Water quality · Pasture

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## Introduction

A range of responses in the mobilisation and transport of phosphorus (P) by rainfall-runoff has been reported for tilled and untilled conditions across varying agro-climatic environments and soil types (McDowell et al. 2001; Hart et al. 2004). Many studies have sought to establish a relationship between soil P and runoff P

concentrations, in the hope that a soil P test could be used for environmental management, and ideally for agronomic management also. No general relationship has yet been established. Although runoff P mostly increases with soil P concentration, such relationships tend to be specific to site, land-use, and experimental method including the soil P test (Wendt and Corey 1980; Schärer et al. 2006; Nash et al. 2007; Tunney et al. 2007). Confounding factors in addition to tillage include recent fertiliser application (Hart et al. 2004) and variation in soil P-buffering capacity (Dougherty et al. 2010).

Studies of P runoff from dairy pastures in Australia show at paddock scale (Baginska et al. 1998; Cornish et al. 2002) and plot scale (Cornish et al. 2002; Dougherty et al. 2008a) that total P concentrations in runoff can be high ( $>1 \text{ mg L}^{-1}$ ), predominantly in the dissolved form, and broadly related to soil P concentration. Other studies of pasture grazed by sheep at various locations in south-eastern Australia have also established rising relationships between the concentrations of P in soil and in runoff (McCaskill et al. 2003). However, methodological differences make it impossible to derive a single relationship from these studies that holds for all or even most of them, complicating the use of soil testing for environmental P management.

A further complication with managing P runoff from pastures is the lack of agreement about whether P is mobilised from pastoral land primarily by dissolution, or by particle detachment, as it is from cultivated land. Many studies at catchment scale indicate that particulate P dominates export from agricultural land, including pasture (e.g. McColl 1978; Smith 1987; Stevens et al. 1999; Douglas et al. 2007). However, whilst the majority of P measured in outflow from a catchment may occur as particulate P, much of that P may have been dissolved initially and adsorbed on to suspended particulate matter either in-stream or between collection and analysis of water samples (Sharpley et al. 1981, 2000; Lambert et al. 1992). In several Australian studies, up to 90 % of P in surface runoff from pastoral sites was in the form of dissolved reactive P (Nash and Murdoch 1997; Nash et al. 2000; Cornish et al. 2002). However, there was considerable variation in the proportions of total P as dissolved reactive P amongst these findings, which may be accounted for by variable sorption of P following mobilisation, or simply by differences in vegetative

cover and soil erosion. Given typical total soil P concentrations of 0.04–0.4 % (Whitehead 2000), even low sediment concentrations may contribute appreciable P to runoff.

The aims of our research are first to understand the relative importance of dissolution and particle detachment as P mobilisation processes in surface runoff from pasture, and second to evaluate whether a single relationship between soil properties and runoff P could be used for environmental management across soil types and varying land-use intensity, if confined to pastoral land and developed with a consistent methodology across sites.

This paper reports on-farm research using a rain-fall-simulator to investigate P loss in surface runoff from pastoral sites and its relationships with land use, groundcover, soil type, and soil P properties. The context was the Sydney Drinking Water Catchments in New South Wales, Australia, where the Sydney Catchment Authority provides guidance to land managers regarding fertiliser use, effluent disposal and other practices that may impact on water quality. This is a large catchment (15,700 km<sup>2</sup>) with low population density, so the detailed data required for complex water quality models or risk assessment tools are unavailable. The Sydney Catchment Authority requires broadly applicable guidelines that use readily obtainable inputs, to assist it to meet its water quality targets.

## Materials and methods

### Study area

The Sydney Drinking Water Catchments are situated in SE NSW, Australia. More than 60 % of the catchment area is non-agricultural land, mostly forest. Land use in the remaining 485,000 ha is mostly pastoral, ranging from intensive dairy on improved pasture through to fine wool production on unfertilised native pasture. Geology is varied, with sandstone, metamorphic rock, granite, basalt and shale comprising the most common lithological groups, making up 70 % of the catchment area (SCA/DWLC 2003). The soils derived from these parent materials cover a wide range of Australian Soil Classification orders, including Ferrosols derived from basalt, Chromosols and Kurosols from shale and metamorphic rock, and

Kandosols and Tenosols from sandstone. Average annual rainfall for the catchment ranges from around 1,500 mm in the eastern-most areas, to around 600 mm in the west.

#### Locations and sites

Fifteen locations were included in the study, spread across the middle section of the catchment between the towns of Robertson (34°35'S, 150°35'E) in the east and Goulburn (34°45'S, 149°44'E) in the west. At each location, from two to six sites were selected for rainfall simulation, within 30–40 m of each other, thus with the same general land-use and lithology but varying soil P and vegetative cover. This resulted in a total of 38 sites for rainfall simulations. With the design of the rainfall simulator, each site was comprised of two plots, each with its own soil P and runoff data, giving a total of 76 plots.

All locations fell into one of the following designations of current land use/pasture type: native/unimproved pasture; rural residential (cattle, sheep or horses); improved pasture (excluding dairying); and dairy pasture. Dairying is a relatively small and declining industry in the region, but as dairy pasture is often associated with high P in runoff, one-third of the locations were selected because they were used currently or in the recent past for dairy production. Based on information provided by the present land-owner on the fertiliser used in recent years, pasture improvement and stocking rates, we broadly considered the locations as being presently under 'low', 'moderate' or 'high' land use intensity.

The locations covered the most common soil lithological groups under agriculture in the catchment, namely metamorphic rock, basalt, and shale. Of the 76 plots, 18 were on basalt and 58 were on soils with non-basaltic lithology. Sites were avoided where fertiliser had been applied in the weeks prior to rainfall simulation and with obvious recent defecation, to minimise confounding by 'incidental' losses of P (Hart et al. 2004). All sites had a pasture cover, although of varying amount, and none showed obvious signs of soil erosion.

At each site, slope was measured using a spirit level clinometer, and the percentage vegetation cover, using a double layer grid point frame (50 points) of the same dimensions as the runoff plots, after Stanton (1960). Location descriptions are given in Table 1.

#### Soil sampling and analysis

For general characterisation of each location, two sets of soil samples were taken along transects in the vicinity of the runoff plot sites. One set of 25–30 core samples (2.5 cm diameter) was collected from the standard depth of 0–10 cm used for agronomic purposes. The other set of 25–30 samples (5 cm diameter) was collected from 0 to 2 cm, the depth commonly used for environmental purposes because surface runoff is thought to interact mainly with the top few mm of soil (Sharpley 1985). The cores were bulked, air dried at 40 °C, and passed through a 2 mm sieve.

Samples from both depths were analysed for bicarbonate-extractable (Colwell 1963) soil test P. The Colwell test is widely used in Australia for agronomic purposes, thus providing a large data base of potentially useful information for environmental-P management as well as agronomic management for which critical values are well established (Gourley et al. 2007). Briefly, 1.0 g soil was extracted with 100 mL 0.5 M NaHCO<sub>3</sub>, adjusted to pH 8.5, by shaking end-over-end for 16 h. Extracts were filtered (Adventec 5A) and extractable P was then measured, after colour development based on the molybdate blue method of Murphy and Riley (1962) (Rayment and Higginson 1992), using a Lachat Quickchem 8000 FIA+ ion flow injection analyser (LaChat Instruments, Milwaukee, WI).

The transect soil samples from the 0–10 cm depth were also analysed for phosphorus buffering index, the currently accepted commercial approach to measuring soil buffering capacity in Australia (Burkitt et al. 2002), as well as soil total P (TKP digestion in 18 M H<sub>2</sub>SO<sub>4</sub> in the presence of a Cu catalyst, measured by FIA—QuickChem method 13-115-01-1-B), Total C and N (LECO CNS2000 Analyser, Leco Corporation, St. Joseph, MI) and pH (1:5 water).

Soil properties determined from the transect sampling are summarised in Table 2. With respect to soil P properties at the 15 locations, total soil P ranged from 40 mg kg<sup>-1</sup> for low input pasture on soil of metamorphic origin to 2,400 mg kg<sup>-1</sup> for soil under intensive dairy management on soils derived from basalt. Colwell soil test P for the 0–10 cm depth (Colwell<sub>0–10</sub>) varied between locations from 7 to 307 mg kg<sup>-1</sup> reflecting land use intensity, with soil test P for low input locations ranging from 7 to

**Table 1** Location details (range and mean are for sites at each location)

Location	° Lat/Long	Soil lithology	Pasture type/ current land use	% Pasture cover (range, mean)	° Slope (range, mean)	Comments/nutrient inputs
1	−34.7711/ 149.6612	Metamorphic	Rural residential (horses)	46–56, 53	10–11, 10	Weedy, no recent nutrient inputs
2	−34.7613/ 149.6706	Metamorphic	Improved/beef cattle	46–54, 49	7–7, 7	Recently sown to pasture following crop, high inputs
3	−34.6032/ 149.7503	Metamorphic	Improved/sheep and cattle	52–86, 77	11–12, 11	Treading damage from grazing, moderate inputs
4	−34.4283/ 150.3832	Shale	Dairy	90–96, 94	7–8, 7	Very high inputs, both effluent and mineral fertiliser
5	−34.4388/ 150.3861	Shale	Rural residential (sheep)	96–100, 99	4–5, 5	Low inputs
6	−34.4843/ 150.3931	Basalt	Improved/Beef cattle	84–96, 88	11–13, 12	Moderate inputs
7	−34.4948/ 150.3835	Shale	Improved/cattle	92–100, 94	7–8, 8	Low inputs
8	−34.6104/ 150.6101	Shale	Unimproved/ cattle	100–100, 100	8–9, 8	Low inputs
9	−34.5772/ 150.4923	Shale	Rural residential (sheep)	100–100, 100	5–8, 7	No recent inputs
10	−34.6040/ 150.4805	Basalt	Improved/Beef cattle	95–100, 98	8–10, 9	Moderate inputs, ex-dairy
11	−34.5994/ 150.4741	Shale	Improved/Beef cattle	90–90, 90	7–9, 8	Moderate inputs, ex-dairy
12	−34.6053/ 150.4815	Basalt	Improved/Beef cattle	100–100, 100	10–10, 10	High inputs (effluent), ex-dairy
13	−34.5961/ 150.4927	Basalt	Improved/Beef cattle	80–80, 80	9–11, 10	High inputs (ex dairy)
14	−34.6007/ 150.4887	Shale	Improved/Beef cattle	90–90, 90	4–6, 5	Moderate inputs
15	−34.0392/ 150.3564	Shale	Dairy	100–100, 100	7–7, 7	High inputs

21 mg kg<sup>−1</sup> (mean 19 ± 1.9), moderate input locations from 15 to 60 mg kg<sup>−1</sup> (54 ± 8.8), and high input locations from 37 to 307 mg kg<sup>−1</sup> (159 ± 27.6). Colwell soil test P values for 10 of the 15 locations were in the range where pastures may respond to added P (<75 mg kg<sup>−1</sup>, refer to Table 2 footnotes and Havilah et al. 2005). The other 5 locations had soil test P values of between 60 and 307 mg kg<sup>−1</sup>. Four of these 5 locations were used now or in the past for dairy production. Phosphorus buffering index varied with lithology. Mean values for basalt (991, no units), shale (236) and metamorphic (61) locations fell within phosphorus buffering index categories of high, moderate and very low, respectively (Table 2).

To relate soil P to runoff P concentration, soils were also sampled from within the 76 rainfall-runoff simulator plots (10 cores per plot, bulked) and analysed for bicarbonate-extractable soil test P (0–2 and 0–10 cm) and phosphorus buffering index (0–10 cm) as described above.

#### Rainfall simulator and simulations

An oscillating boom rainfall simulator was used to generate surface runoff. It consisted of three Veejet 80100 flat fan nozzles attached to a hollow boom, mounted on an aluminium A-frame, 2.4 m above the ground, similar to Loch et al. (2001). The simulator

**Table 2** Soil properties at each location, rated low (L), medium (M) or high (H) for land-use intensity (0–10 cm depth, unless specified otherwise)

Location/ intensity	Colwell P (mg kg <sup>-1</sup> ) (0–2 cm)	Colwell P (mg kg <sup>-1</sup> ) (0–10 cm) <sup>a</sup>	Total P (mg kg <sup>-1</sup> )	Phosphorus buffering index (no units) <sup>a</sup>	Total C (%)	Total N (%)	pH <sub>w</sub>
1/L	16	7.5	40	51	1.8	0.15	5.3
2/H	68	37	90	36	1.7	0.15	4.9
3/M	95.5	60	100	94	3.7	0.27	5.3
4/H	310	211	1,340	104	5.1	0.40	5.4
5/L	40.5	21.5	320	160	4.7	0.22	5.2
6/M	104	40	1,190	169	4.2	0.36	5.6
7/L	35.5	21	630	96	4.3	0.34	5.4
8/L	26	18	1,690	452	4.5	0.27	5.5
9/L	21.5	13.5	310	367	5.9	0.32	5.3
10/M	63	30.5	1,080	526	5.9	0.39	5.7
11/M	28.5	15.5	710	165	6.1	0.37	5.2
12/H	380	307	2,400	361	6.8	0.42	5.0
13/H	230	132	810	2,908	5.8	0.39	5.4
14/M	55	41	450	446	4.9	0.30	5.3
15/H	180.5	79	<i>nd</i>	97	5.3	0.34	6.3

*nd* not determined

<sup>a</sup> According to a nation-wide study by Gourley et al. (2007), critical Colwell P for agronomic response varies with phosphorus buffering index, but in all soils is <50 mg kg<sup>-1</sup>. They reported mean phosphorus buffering index for soils derived from basalt (991), shale (236) and metamorphic (61) parent material, falling within phosphorus buffering index categories of high, moderate and very low, respectively

allows ‘rainfall’ to be applied to two adjacent plots, 2.0 × 0.75 m in size. The plot boundaries were defined by lengths of 3 mm sheet steel, 150 mm wide, bolted together to form open-ended rectangles. These were aligned parallel with the direction of slope, open at the down-slope end, with a gap of 10 cm between the two plots. Steel boundaries were driven into the ground to a depth of 75 mm. A shallow trench was dug at the open end of each plot into which a stainless steel trough was located to collect runoff. A thin band of bentonite clay applied to the front edge formed a watertight seal between the trough and the plot surface. Perspex sheets were placed at an angle over the troughs, to prevent water being sprayed directly into them. The surface runoff from the plots was collected in gravity-fed containers a few meters down slope of the plots.

The first set of 27 simulations (locations 1–9) was conducted in Oct–Nov 2005. One of the paired plots in each of these simulations was chosen randomly and covered with shadecloth (70 % mesh) supported 150 mm above the soil surface on a rectangular wooden frame placed on the metal boundary sheets.

Thus, the covered plots received the same rainfall intensity as the uncovered plots but with significantly reduced raindrop energy (Loch and Foley 1994). Vegetative cover on the paired plots varied from 46 to 100 %. Of the 27 pairs, 24 were on non-basalt soils. The pairs were matched well for groundcover (%), with the relationship between plots ‘with’ and ‘without’ shadecloth being:

$$y = 1.035x - 3.21 (R^2 = 0.99, P < 0.001) \quad (1)$$

In a second set of 11 simulations conducted in Nov. 2006, the shadecloth cover was not used. These provided a further 22 uncovered plots from locations 10–15. Twelve of these plots (6 pairs) were on basalt soils and the other 10 plots (5 pairs) were on non-basalt soils.

The total data-set comprised of 76 plots, each with its own soil P and runoff data; 27 plots with shadecloth from the first set, and 49 plots without shadecloth including 27 and 22 plots from the first and second set, respectively.

Water applied by the simulator was sourced from the Sydney Catchment Authority Pumping Station at

Burrawang, NSW. Previous laboratory testing showed that the P concentration of this water was close to detectable limits, typically  $<0.01 \text{ mg L}^{-1}$ . To determine rainfall rate prior to each rainfall-runoff simulation, plots were covered with plastic sheeting that directed 100 % of the water applied into the collection troughs. The volume of runoff collected from each plot over 5 min was then measured, and if necessary the simulator was re-positioned so that each of the paired plots received approximately the same volume of water during a simulation run. In this study, a rainfall intensity of  $60 \text{ mm h}^{-1}$  was applied until runoff commenced, after which rainfall application was continued for a further 15 min.

#### Runoff sample collection and analysis

Surface runoff collected over this period from individual plots was bulked before sampling for analysis. Samples were stored on ice in the field before being returned to the laboratory, where they were stored at  $-20 \text{ }^{\circ}\text{C}$  until analysis.

Sub-aliquots of the collected runoff were filtered in the field ( $0.45 \text{ }\mu\text{m}$ ) immediately after collection for dissolved reactive P analysis, based on the molybdate blue method of Murphy and Riley (1962), using FIA as described above. Bulk sub-samples of the collected runoff were taken for analysis of total P and total suspended solids. Total P was determined by alkaline persulphate oxidation, after Hosomi and Sudo (1986) and Maher et al. (2002). The percentage of total P present as dissolved reactive P (%DRP) was calculated to express the relative magnitude of the two mobilisation processes, dissolution and particle detachment.

Although alkaline persulphate oxidation is widely used for total P determination in turbid water, some of the total P was evidently not fully extracted from 6 of the runoff samples in which dissolved reactive P values substantially exceeded their total P values (3 sites, each covered and uncovered with shade cloth). These sites were all at Location 4, which had soils derived from shale and a long history of dairy production with high inputs (Table 1) resulting in concentrations of Colwell P ( $211 \text{ mg kg}^{-1}$ ) and total P ( $1,340 \text{ mg kg}^{-1}$ ) that are extreme for soils with this lithology in Australia. Incomplete extraction of total P may result from condensation, complexation or occlusion of unfiltered (high total suspended solids) high-P samples during storage (Maher and Woo 1998;

Jarvie et al. 2002). Repeating the analysis and the use of more aggressive extraction techniques (concentrated nitric + sulphuric acid; Pote and Daniel 2000) failed to resolve this issue. Data from Location 4 were therefore excluded from any statistical analyses requiring total P but retained in analyses of dissolved reactive P. We assume the problem was confined to total P in runoff samples from the shale soil with extreme P concentrations. Basalt soils are inherently higher in total P than non-basalt soils, but samples from basalt soils revealed no problem with total P extraction.

Total suspended solids were determined after method 2540D, described in Standard Methods (APHA 2005).

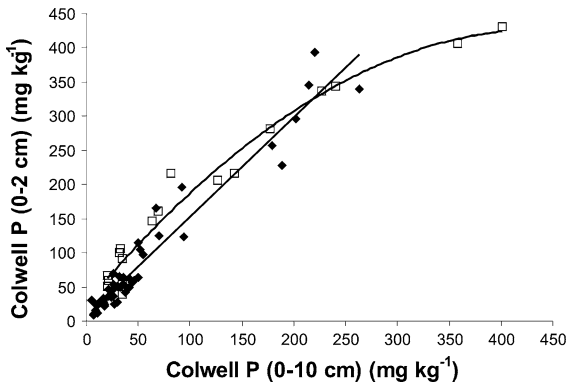
#### Statistical analysis

Statistical analyses were conducted using MS Excel 2003, Minitab 14.1 and S-Plus 6.1. Regression relationships between soil P and P in runoff for all of the soil types were compared statistically before dividing soil types into two groups for analysis, namely basalt and non-basalt soils. A paired *t* test was used for comparisons involving paired plots. The measure of dispersion given after a mean is the standard error of the mean. In linear regression, where the intercept was not significantly different from zero ( $P \geq 0.05$ ) the regression has been forced through the origin. Multiple regression was used to predict runoff P concentration from soil test P and phosphorus buffering index. The equations were developed with data from plots with shade cloth, to minimise confounding from soil erosion, and tested with an independent set of data from plots without shade cloth but with high vegetative cover and low surface erosion.

## Results

#### Soil test P at two sampling depths

Concentrations of soil test P in the 0–2 cm layer were higher than the 0–10 cm layer (Fig. 1). For basalt soils, the relationship between extractable P at the two depths was curvilinear, tending towards a 1:1 relationship as the concentration increased. For non-basalt soils, the relationship between depths was linear. Both regressions were highly significant ( $P < 0.001$ ).



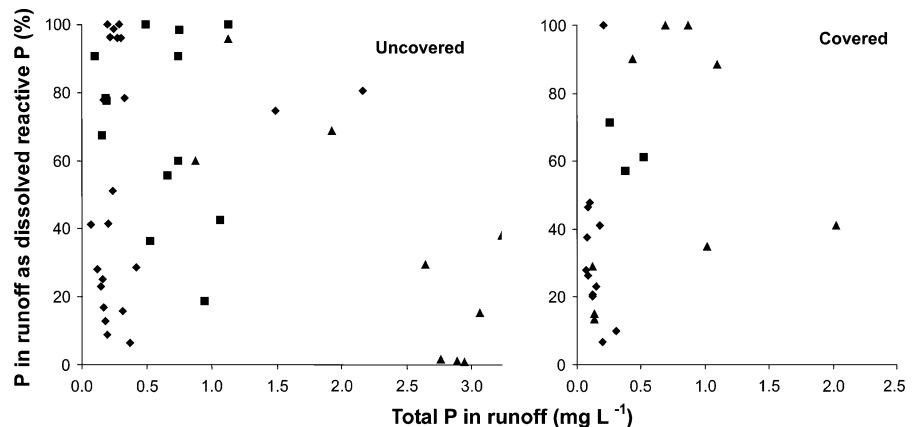
**Fig. 1** Bicarbonate-extractable (Colwell) soil P at two sampling depths in non-basalt soil (*diamond*),  $y = 1.46x + 7.3$  ( $R^2 = 0.95$ ,  $P < 0.001$ ) and basalt soil (*square*),  $y = -0.002x^2 + 1.82x + 26.5$  ( $R^2 = 0.97$ ,  $P < 0.001$ )

The percentage of total P present as dissolved reactive P in runoff (%DRP)

The %DRP varied widely, regardless of soil type and almost regardless of the absolute concentration of total P in runoff (Fig. 2). Three factors contributed to this variation.

First, %DRP was greater in plots with shade cloth (44 %, range 7–100 %) than paired-plots without shade cloth (28 %, range 1–96 %) ( $P < 0.01$ ). These 24 plot-pairs comprised all of the covered plots and a sub-set of the uncovered plots shown in Fig. 2 (Location 4 was not included, see “Materials and methods”). Greater %DRP with shade cloth was associated with lower concentrations of total suspended solids, viz. 167 mg L<sup>-1</sup> compared with 690 mg L<sup>-1</sup> from uncovered plots ( $P < 0.01$ ).

**Fig. 2** The percentage of total P present as dissolved reactive P in runoff (%DRP) from plots uncovered or covered with shade cloth on metamorphic (*triangle*), shale (*diamond*), and basalt (*square*) soils. A subset of 24 uncovered plots was paired with covered plots for paired *t* test comparison of %DRP (see text)



Second, for uncovered plots, %DRP was greater in basalt soils ( $73 \pm 8.5$  %) than shale ( $56 \pm 8.23$  %) and metamorphic ( $34 \pm 11.4$  %) soils ( $P < 0.05$ ), thus appearing to mirror the differences in phosphorus buffering index shown in Table 2.

Third, weak but significant positive relationships between %DRP (y) and soil test P(x) were observed within the non-basalt group of sites, whether shade cloth was used (Eq. 2) or not (Eq. 3):

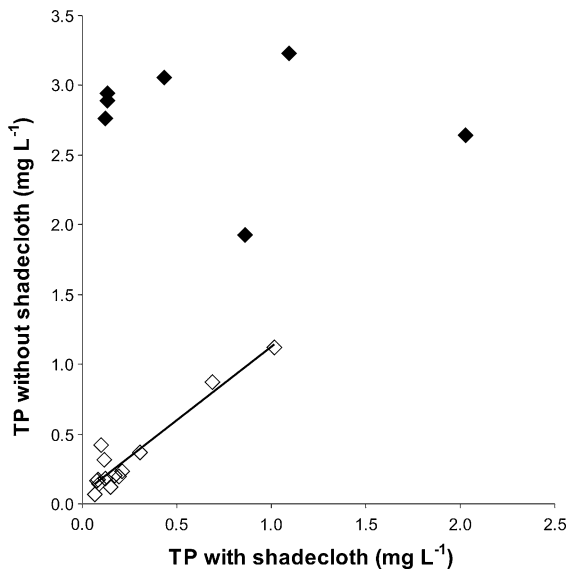
$$y = -72.7 + 38.7\text{Ln}(x) (R^2 = 0.53, P < 0.01) \quad (2)$$

$$y = -14.9 + 14.7\text{Ln}(x) (R^2 = 0.24, P < 0.05) \quad (3)$$

Effects of soil cover on concentrations of dissolved reactive P, total P and total suspended solids in runoff

For paired-plots (excluding Location 4), the mean dissolved reactive P concentration with shade cloth ( $0.23 \text{ mg L}^{-1}$ ) was not significantly different ( $P \geq 0.05$ ) from dissolved reactive P concentration without shade cloth ( $0.28 \text{ mg L}^{-1}$ ). When the plots with extreme soil P from Location 4 were included, shade cloth cover resulted in a small but significant ( $P < 0.05$ ) decrease in dissolved reactive P concentration, from  $0.76$  to  $0.64 \text{ mg L}^{-1}$ .

Shade cloth had potentially much greater effects on total P concentration (Fig. 3), but this was manifest only at the seven sites shown by closed symbols, for which total P averaged  $2.78 \text{ mg L}^{-1}$  when uncovered and  $0.69 \text{ mg L}^{-1}$  when covered ( $P < 0.001$ ). With these seven plot-pairs, the difference in total P reflected a difference in sediment (total suspended solids) concentration which averaged  $422 \text{ mg L}^{-1}$  from the covered plots and  $1,751 \text{ mg L}^{-1}$  from the



**Fig. 3** The effect of shade cloth on total P (TP) concentration ( $\text{mg L}^{-1}$ ) in runoff from non-basalt soils (excluding Location 4). The regression uses only data from plots with high vegetative cover and low total suspended solids (TSS) in runoff (open diamond),  $y = 0.043 + 1.09x$  ( $R^2 = 0.96$ ,  $P < 0.01$ )

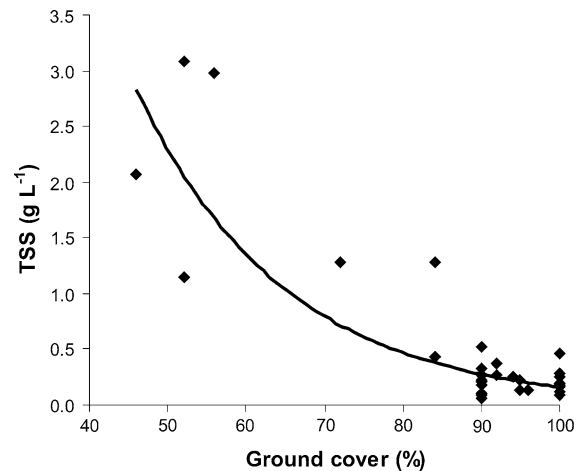
paired uncovered plots ( $P < 0.05$ ). These plots were at sites with relatively low pasture cover in Table 1. Shade cloth had little effect on total P in runoff from plots with high vegetative cover and low total suspended solids (open diamonds), as shown by the near 1:1 regression for these data (Fig. 3).

Increasing vegetative cover in the absence of shade cloth substantially reduced sediment concentrations in runoff (Fig. 4). However, there was no such relationship between vegetative cover and total P in runoff because total P in soil (Table 2) varied between locations with similar groundcover (Table 1), resulting in P concentrations in the sediment in runoff ranging between 0.03 and 2.0 % (mean 0.34 %).

Effect of soil P concentration on P concentrations in runoff

#### Total P in runoff from plots without shade cloth

Despite a general trend for total P to rise with soil test P in the non-basalt group of soils, the data were variable and the relationships were not statistically significant for either the non-basalt or basalt soils, at either sampling depth (0–2 or 0–10 cm) (Fig. 5).



**Fig. 4** Relationship between total suspended solids (TSS) in runoff and ground cover, for sites with no shade cloth,  $y = 32.301e^{-0.0529x}$  ( $R^2 = 0.65$ )

#### Dissolved reactive P in runoff from plots without shade cloth

Dissolved reactive P concentration in runoff responded positively to soil P concentration amongst both basalt and non-basalt soils (Fig. 6). Differences in runoff dissolved reactive P concentration between soil types were small at low soil test P but increased as soil test P increased. The slopes of the regressions were significantly different ( $P < 0.001$ ). Consequently, dissolved reactive P concentrations in runoff were much greater in non-basalt than basalt soils at higher soil P concentrations.

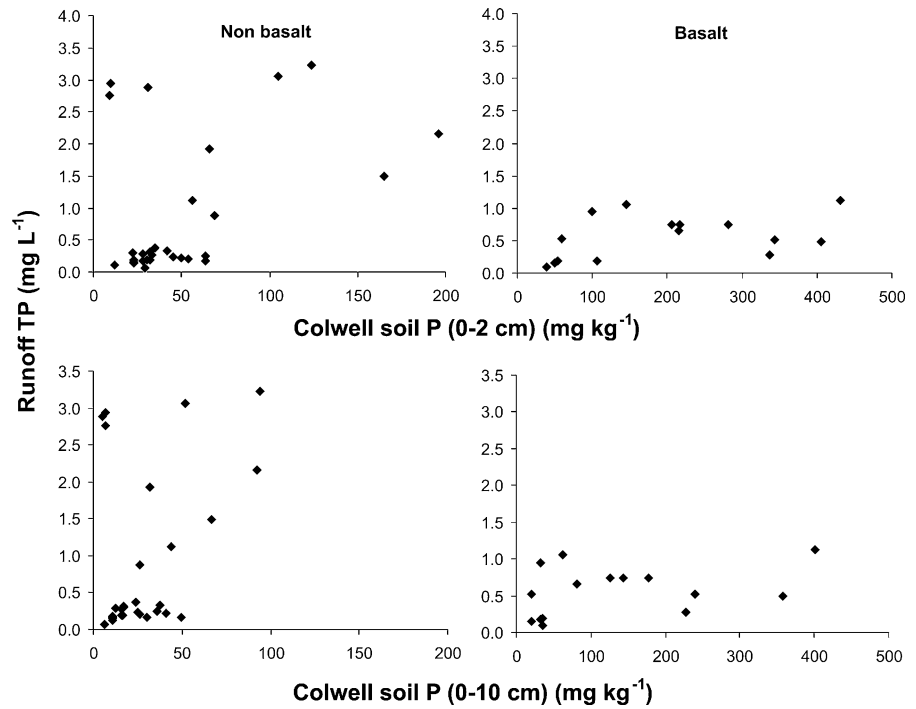
Soil sampling depth affected the slope of the regression within non-basalt soils ( $P < 0.05$ ), but the strength of the relationship was comparable for the two sampling depths ( $R^2 = 0.90$  and  $0.89$ ).

#### Effect of shade cloth on the relationship between soil test P and P in runoff

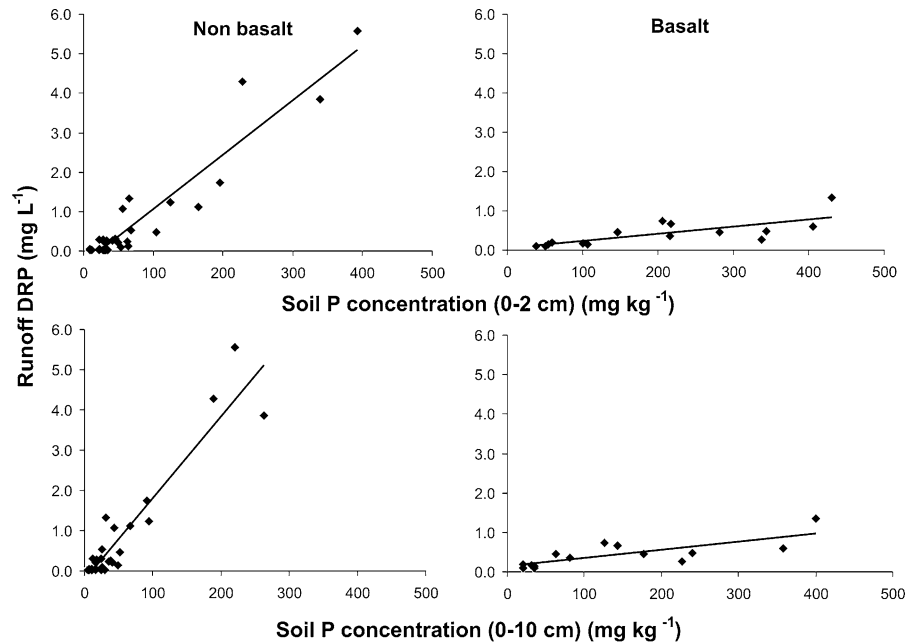
This analysis was restricted to non-basalt sites because of the small number of basalt sites with shade cloth. Data were from the 0–10 cm soil sampling depth. There was a positive relationship between soil test P and total P in runoff when plots were covered with shade cloth but no relationship without shade cloth (Fig. 7, top). Dissolved reactive P concentrations also rose with increasing soil test P, but this relationship was significant whether plots were covered with



**Fig. 5** Concentrations of total P (*TP*) in runoff in response to Colwell soil P, from basalt and non basalt soils and soil sampling from 0–2 to 0–10 cm depths. Sites with no shade cloth (excludes sites at location 4, see “Materials and methods” for explanation)



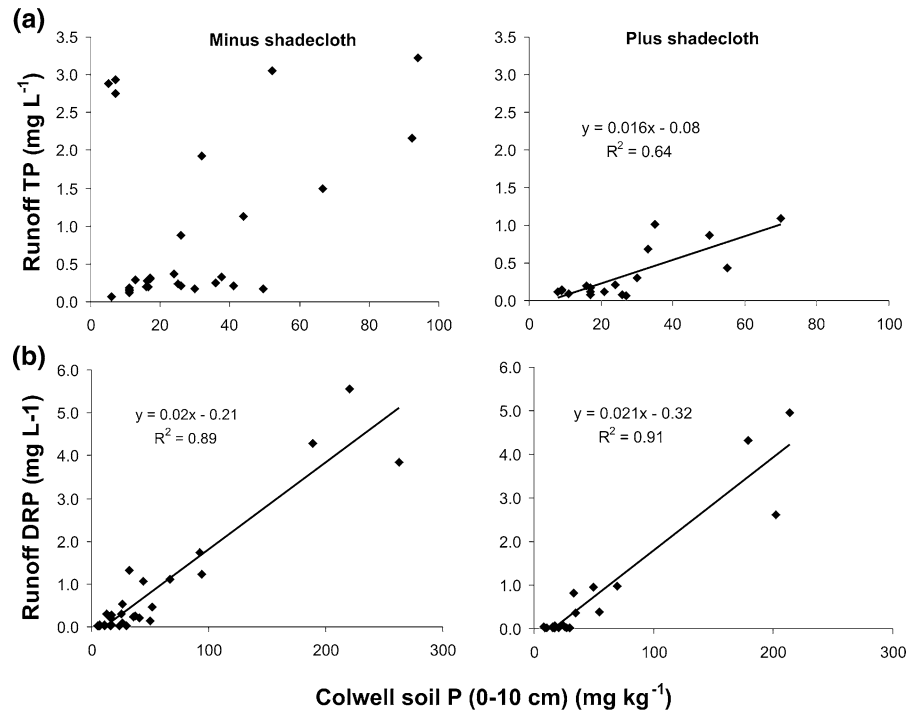
**Fig. 6** Concentrations of dissolved reactive P (*DRP*) in runoff in response to Colwell soil P, from non-basalt and basalt soils and soil sampling from 0–2 to 0–10 cm depth. Sites with no shade cloth, includes location 4. For non-basalt, 0–2 cm depth,  $y = -0.315 + 0.014x$  ( $R^2 = 0.90$ ) and for 0–10 cm depth,  $y = -0.21 + 0.02x$  ( $R^2 = 0.89$ ). For basalt soils, 0–2 cm depth,  $y = 0.047 + 0.002x$  ( $R^2 = 0.55$ ) and for 0–10 cm depth,  $y = 0.136 + 0.002x$  ( $R^2 = 0.61$ )



shade cloth or not (Fig. 7, bottom). Linear regressions were fitted to these data because there was no strong statistical argument to use a non linear model across all of the data, but it is important to note that

substantial increases in either total P or dissolved reactive P in the covered plots (and dissolved reactive P in uncovered plots) occurred only when Colwell<sub>0-10</sub> soil test P was greater than about 30 mg kg<sup>-1</sup>.

**Fig. 7** Effects of shadecloth on concentrations in runoff total P (TP, top) and dissolved reactive P (DRP, bottom) in response to Colwell soil P (0–10 cm), non basalt soils (total P excludes data from location 4, see text). For total P plus shadecloth,  $y = -0.08 + 0.016x$  ( $R^2 = 0.64$ ). For dissolved reactive P minus shadecloth,  $y = -0.21 + 0.02x$  ( $R^2 = 0.89$ ). For dissolved reactive P plus shadecloth,  $y = -0.32 + 0.021x$  ( $R^2 = 0.91$ )



Effect of soil P buffering index on P concentrations in runoff, with shadecloth

This analysis was restricted to plots with shadecloth cover to minimise the confounding effects of surface soil erosion, thus restricting the analysis to non-basalt soils. The response of P in runoff to soil phosphorus buffering index is given in Fig. 8, but because soil test P and phosphorus buffering index may be confounded, the soil test P associated with each value for phosphorus buffering index is shown in lieu of the customary symbols. P concentrations in runoff clearly rose as phosphorus buffering index fell, but all of the notably elevated values for total P ( $>0.3 \text{ mg L}^{-1}$ ) and dissolved reactive P ( $>0.1 \text{ mg L}^{-1}$ ) were also associated with soils with relatively high soil test P ( $>30 \text{ mg kg}^{-1}$ ). Nevertheless, the possible importance of phosphorus buffering index independently of soil test P is suggested by the sites falling within a narrow range of soil test P (26–35  $\text{mg kg}^{-1}$ ), shown in the inset graphs in Fig. 8.

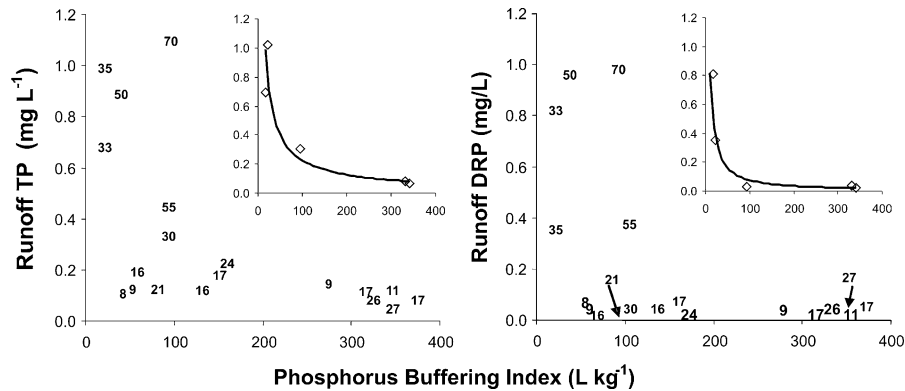
The interaction between phosphorus buffering index and soil test P implicit in Fig. 8 for the covered non-basalt soils parallels the significant interaction in Fig. 6 where runoff dissolved reactive P concentrations from the highly buffered basalt soils were similar to the more

weakly buffered non-basalt soils at low soil test P, but much lower at higher soil test P.

#### Predicting P concentrations in runoff

No statistical model was significant when fitted to the soil-P/runoff-P data for all of the sites. However, even without statistical analysis it was clear that high P concentrations in runoff could result from surface soil erosion (Fig. 3) and extreme soil P concentrations (Figs. 6, 7). The greater need is to predict runoff-P concentrations in the more uncertain situation where erosion is not a major factor and soil test P is within the range where farmers are most likely to be making decisions about fertiliser use and disposal of manure and effluent. Therefore, data were examined from plots with shadecloth (to manage erosion) and with  $<100 \text{ mg P kg}^{-1}$  to determine if P concentrations in runoff could be predicted satisfactorily from the measured soil variables. The resultant regression equations were then tested using an independent set of data from plots with good vegetative cover (or  $<500 \text{ mg total suspended solids L}^{-1}$ ).

The sites with shadecloth included 20 non-basalt and four basalt plots. The highest soil test P value in



**Fig. 8** The response of total P (*TP*) and dissolved reactive P (*DRP*) in runoff to phosphorus buffering index and soil test P for non-basalt soils with shadecloth cover. *Numbers* on the figure replacing symbols refer to soil test P ( $\text{mg kg}^{-1}$ ) for each plot.

these data was  $69 \text{ mg kg}^{-1}$ . The multiple regressions that best predicted total P and dissolved reactive P in runoff from soil Colwell<sub>0-10</sub> soil test P, phosphorus buffering index and their interaction were:

$$\begin{aligned} \text{Total P} = & 0.254 - 0.014 \text{ soil test P} + 0.0003 \text{ soil test} \\ & \text{P}^2 - 6.59(1/\text{phosphorus buffering index}) + 0.606 \\ & (\text{soil test P} \times 1/\text{phosphorus buffering index}) \\ & (R^2 = 0.91, P < 0.001) \end{aligned} \quad (4)$$

$$\begin{aligned} \text{Dissolved reactive P} = & 0.293 - 0.022 \text{ soil test} \\ & \text{P} + 0.0003 \text{ soil test P}^2 - 14.08 \\ & (1/\text{phosphorus buffering index}) \\ & + 0.813(\text{soil test P} \times 1/\text{phosphorus buffering index}) \\ & (R^2 = 0.94, P < 0.001) \end{aligned} \quad (5)$$

Regressions of the observed values for total P and dissolved reactive P and those predicted from Eqs. 4 and 5 passed through the origin with slopes of 1.08 (total P) and 0.78 (dissolved reactive P) (Fig. 9). The models systematically slightly under-predicted dissolved reactive P at high concentrations but there was no evident bias in the predictions of total P.

The independent data comprised 29 plots without shadecloth but with good vegetative cover. There were 23 plots on non-basalt soils and six on basalt. Plots had  $>90\%$  groundcover and the highest soil test P value was  $94 \text{ mg P kg}^{-1}$ . Some of these plots were pairs for the uncovered plots from the first set of simulations in 2005 and others were from the second set of simulations in 2006. Both regressions were significant

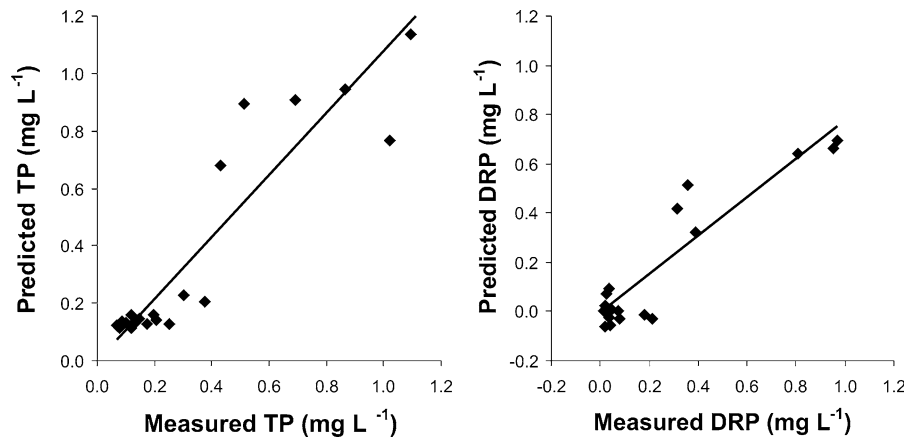
Insert graphs are for plots within a narrow range of soil test P ( $26\text{--}35 \text{ mg kg}^{-1}$ ) below the agronomic critical value (refer to text). For the inset graphs: total P =  $10.49x^{-0.84}$  ( $R^2 = 0.94$ ); dissolved reactive P =  $10.95x^{-1.08}$  ( $R^2 = 0.86$ )

( $P < 0.01$ ), with high  $R^2$ . However, Eq. 4 tended to under-predict total P at higher concentrations and Eq. 5 under-predicted dissolved reactive P at all concentrations.

## Discussion and conclusion

Numerous studies in south-eastern Australia have together established that relatively higher P concentrations in runoff from pastures are generally associated with higher P concentrations in soil (Costin 1980; Baginska et al. 1998; Cornish et al. 2002; McCaskill et al. 2003; Barlow et al. 2005; Dougherty et al. 2008a, b). However, they provide no guideline for soil P that may be used for environmental risk assessment and management because any quantitative relationships between the concentrations of soil P and runoff P are location and experiment-specific, as with other research internationally (Hart et al. 2004).

We aimed to determine if a generally applicable relationship between concentrations of soil P and runoff P could be developed for pastoral land by using a single methodology on fields that covered a range of soil types and land-use intensity, and if so to propose guidelines for soil P to use in environmental management. The sites covered a wide range of values for soil properties thought to influence P concentrations in runoff, but avoided sites with recent fertiliser or manure accretion to minimise confounding by ‘incidental’ losses of P (Hart et al. 2004). In some cases runoff plots were paired, with one of each pair being covered with



**Fig. 9** Measured and predicted total P (TP) and dissolved reactive P (DRP) in runoff from soils with soil test  $P < 70 \text{ mg kg}^{-1}$ , with shadecloth cover. For total P,  $y = 1.08x$  ( $R^2 = 0.83$ ). For dissolved reactive P,  $y = 0.78x$  ( $R^2 = 0.85$ )

shadecloth to reduce rainfall energy to help distinguish between dissolution and particle detachment (soil erosion) as primary mobilisation processes.

Before considering relationships between soil P and runoff P, the following discussion addresses the relative importance of the mobilisation processes. This is important, as the form in which P is mobilised determines the approach taken to managing P losses in runoff. For example, if particulate P is the predominant form mobilised, then it may be beneficial to mitigate soil erosion or construct vegetated buffer strips near to the source, but these measures may have little or no effect on P exports if dissolved reactive P predominates (Dorioz et al. 2006; Fiener and Auerswald 2009). Despite the fundamental importance of this principle, there is no agreement in the literature as to whether P is mobilised from pastures predominantly as dissolved reactive P or in the particulate form resulting from surface soil erosion (Hart et al. 2004; Bilotta et al. 2008). This uncertainty arises partly from the possibility of dissolved reactive P sorption to soil in subsequent overland flow (Sharpley et al. 1981, 2000; Lambert et al. 1992), which we minimised by collecting and filtering samples immediately after runoff.

The mobilisation process in pastures

#### *The relative importance of dissolution and particle detachment*

Despite the lack of cultivation and the absence of obvious soil erosion at the pastoral study sites, dissolution did not always dominate P mobilisation.

Three factors contributed to variation in the *relative* magnitude of P mobilisation as dissolved P, expressed as the percentage of total P as dissolved reactive P (%DRP).

First, the amount of vegetative cover was not always adequate to protect against particle detachment. Shadecloth reduced the mean concentrations of sediment in runoff from  $690$  to  $167 \text{ mg L}^{-1}$ , thereby decreasing particulate P concentrations and increasing the mean %DRP from 28 % in uncovered plots to 44 % from the paired covered plots. The other two factors, soil type and soil test P, affected the %DRP through their effects on the concentration of dissolved reactive P. Soils derived from basalt had higher average %DRP (73 %) than soils derived from either shale (56 %) or metamorphic parent material (34 %). Whilst this appeared to reflect their differences in soil P buffering capacity, the relationship may not be a causal as the red Ferrosols derived from basalt tend to be structurally more stable than the other soils (mostly Kurosols). High soil test P also contributed to higher %DRP, which was most evident at four sites with non-basalt soils and a history of dairy production. Other research in Australia has found a high proportion of total P as dissolved reactive P in runoff from high-input dairy pastures (e.g. Nash and Murdoch 1997; Barlow et al. 2005; Dougherty et al. 2008a, b).

#### *Surface soil erosion and the loss of P in runoff from pasture*

Total P concentration in runoff was related to soil test P, but this was evident only when the confounding

effect of soil erosion was reduced by protecting soils with shade cloth (Fig. 7, top). The importance of soil erosion was at first surprising, given that there was no obvious surface erosion at any of the sites. However, amongst the uncovered plots, sediment concentrations rose sharply as ground cover fell below 90 % (Fig. 4), with the highest total suspended solids concentrations ( $\sim 3 \text{ g L}^{-1}$ ) observed when vegetative groundcover was lowest ( $\sim 50 \%$ ). The effect of ground cover on soil erosion is well known (Costin 1980; Castillo et al. 1997; Loch 2000) and there are published standards for the amount of cover recommended to protect against erosion. In arable farming the amount of residue recommended may be as low as 30 % (Freebairn et al. 2006), whilst for managing surface erosion in permanent pasture the New South Wales Department of Primary Industries and the Hawkesbury-Nepean Catchment Management Authority advise  $>70 \%$  cover (Anon. 2006).

Managing surface-soil erosion from pastures is thus an obvious first line of attack on P losses in runoff, but our data suggest that targets for managing soil erosion are too low for managing P losses in runoff. This is because erosion rates that are negligible from the perspective of soil conservation may nevertheless result in significant loss of sediment-P in runoff. The following estimates illustrate this point.

With total suspended solids of  $3 \text{ g L}^{-1}$ , we estimate an average annual rate of soil erosion of  $<2.0 \text{ t ha}^{-1}$  assuming 600 mm annual rainfall (typical for the west of the study region) and an annual runoff coefficient of 0.1 (Young et al. 2001). This also assumes that total suspended solids concentrations in rainfall-runoff are the same as with the rainfall simulator which, with its high rainfall rate of  $60 \text{ mm h}^{-1}$ , may over-estimate average annual erosion. Whilst  $2 \text{ t ha}^{-1}$  exceeds the “tolerable” rate of soil loss of  $<0.5 \text{ t ha}^{-1} \text{ year}^{-1}$  espoused by Loughran et al. (2004), it is comparable to other studies on well-managed grazed pasture in SE NSW, for example  $2.2 \text{ t ha}^{-1} \text{ year}^{-1}$  in Mahmoudzadeh et al. (2002) and  $3.3 \text{ t ha}^{-1} \text{ year}^{-1}$  in Erskine et al. (2002). It is much less than may occur from tillage-based agriculture, e.g.  $19 \text{ t ha}^{-1} \text{ year}^{-1}$  from vegetable farms near Sydney (Hollinger et al. 2001).

Despite sediment losses from the sites with 50 % groundcover being low from an erosion perspective, the concentrations of total P in runoff were high from an environmental perspective, with  $3 \text{ mg P L}^{-1}$  from sites with greatest total suspended solids being 150

times the default trigger value ( $0.02 \text{ mg L}^{-1}$ ) requiring risk assessment in upland streams in slightly disturbed ecosystems in SE Australia (ANZECC/ARMCANZ 2000). Thus, a relatively low erosion rate ( $<2 \text{ t ha}^{-1} \text{ year}^{-1}$ ) and low soil P (Fig. 5) may still lead to runoff P concentrations that concern environmental regulators.

Exports of P or ‘P loading’ are just as important as P concentrations, as the following estimates of annual export coefficient illustrate for sites generating runoff with  $3 \text{ g L}^{-1}$  total suspended solids. These calculations assume an average total P of  $3 \text{ mg L}^{-1}$  (Fig. 3), rainfall of 600 mm, and runoff coefficient of 0.1 (as above). The estimated export coefficient of  $1.8 \text{ kg P ha}^{-1} \text{ year}^{-1}$  is comparable with farm-scale estimates in the region that have highlighted concerns with P runoff from pasture land (Baginska et al. 1998; Cornish et al. 2002). Thus, P losses of environmental concern may occur despite the rates of soil erosion being low from the perspective of soil conservation.

#### *The groundcover required to manage P losses in runoff*

Although surface erosion rates were low with around 50 % vegetative cover, as shown above, increasing cover to  $>90\text{--}95 \%$  further reduced erosion (Fig. 2). Effective cover reduced total P concentrations in runoff to an order of magnitude less than occurred with 50 % ground cover, whether the cover came from vegetation (Fig. 3) or from shade cloth (Fig. 7). The exceptions to this were sites with extreme soil test P concentrations arising from a history of dairy production. We conclude that maintaining groundcover above the thresholds recommended for erosion management has the potential to significantly reduce total P losses from pasture sites without a history of dairy production.

In our study, low ground cover was sometimes associated with low soil P (Tables 1, 2), so consideration should be given to applying P fertiliser where soil infertility is likely to have contributed to the low ground cover (Robinson and Lazenby 1976; McGufficke 2003).

With respect to environmental water quality guidelines, it is notable that even with good ground cover and low soil test P, total P in runoff at the point of generation will be up to  $0.2 \text{ mg L}^{-1}$  (Fig. 7 top, right), which is well above the environmental ‘trigger’ value

(0.02 mg L<sup>-1</sup>) for the streams to which field runoff discharges. Clearly it is inappropriate for environmental regulation to apply these trigger values to in-field or edge-of-field measurements.

#### Runoff P responses to soil test P and P buffering index

Where surface erosion was controlled through high ground cover or shade cloth, much of the remaining variation in P concentrations in runoff across the diverse soil types and land-use intensity was accounted for by variation in soil test P and phosphorus buffering index. Dissolved reactive P concentrations in runoff increased with soil test P, but in linear regressions the increase was much greater from non-basalt soils with low or moderate phosphorus buffering index than from basalt soils with high phosphorus buffering index (Fig. 6). Consequently, soil test P alone was unrelated to P concentration in runoff for the combined soil types, whereas multiple regression including both soil test P and phosphorus buffering index predicted P concentration in runoff reasonably well (Figs. 9, 10).

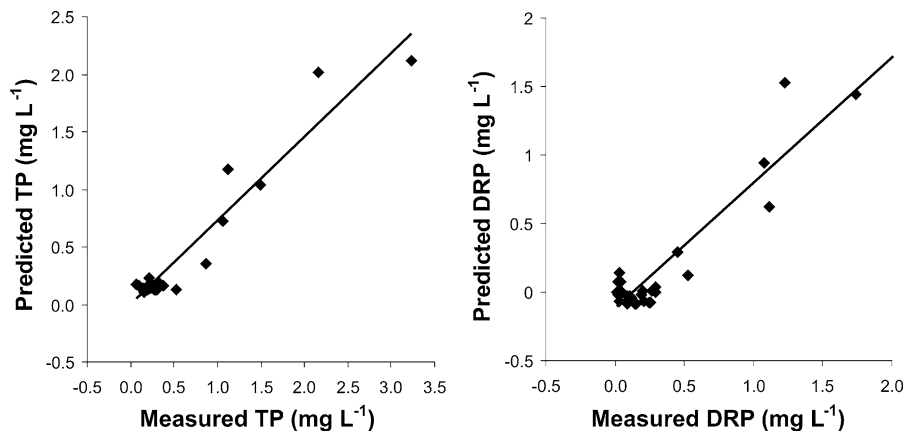
Whilst a range of runoff-P responses to soil test P has been reported in the literature (McDowell et al. 2001; Hart et al. 2004), the interacting effect of phosphorus buffering index is less well documented under field conditions. However, the chemistry behind the effect has been discussed (Ehlert et al. 2003; Moody 2007) and there is supporting evidence from controlled field experiments. In one laboratory study, Olsen soil test P (also bicarbonate extraction)/P sorption index (PSI) was a good predictor of ‘readily

desorbable’ P (Hughes et al. 2006). Under controlled field conditions, dissolved reactive P concentration in overland flow from pasture was related, in one study, to Olsen P and P sorption index (PSI) (McDowell and Condron (2004) and, in another, to degree of P saturation which is a function of soil test P and PSI (Allen et al. 2006). In Australia, Dougherty et al. (2010) found that 80 % of the variation in dissolved reactive P concentration in runoff from pastures growing in reconstructed soils in runoff trays was explained by bicarbonate-extractable soil test P (Colwell) and phosphorus buffering index, which measure soil P quantity and buffering, respectively.

The present study included a heterogeneous set of field sites. In basalt soil, the runoff-P response to soil test P was clearly linear, as well as being smaller than in the non-basalt soils as discussed above. Statistically, the non-basalt soils behaved as a group, although contrasting responses may have been revealed with more data, given that phosphorus buffering index ranged from low to moderate (Table 2) and the response to soil test P appears much more likely to be non-linear in soils with low phosphorus buffering index (Dougherty et al. 2010).

For the non-basalt soils, the data for runoff P over the full range of soil test P were fitted to linear regression rather than non-linear regression, for statistical reasons, but inspection of the data suggests the response may be non-linear, at least at lower values of soil test P. There was no systematic rise in runoff P concentration up to about 50 mg Colwell<sub>0-2</sub> P kg<sup>-1</sup> (or 30 mg Colwell<sub>0-10</sub> P kg<sup>-1</sup>) (Fig. 7). Some authors have fitted ‘bent stick’ statistical models to their data to capture this non-linearity, using the ‘break point’ to

**Fig. 10** Measured and predicted total P (TP) and dissolved reactive P (DRP) for plots without shade cloth but with > 90 % groundcover (or < 500 mg L<sup>-1</sup> total suspended solids) and soil test P < 100 mg kg<sup>-1</sup>. For total P,  $y = 0.74x$  ( $R^2 = 0.92$ ). For dissolved reactive P,  $y = -0.12 + 0.94x$  ( $R^2 = 0.86$ )



define a critical P concentration for environmental purposes (McDowell et al. 2003).

In the absence of data for P buffering capacity, 30 mg Colwell<sub>0–10</sub> P kg<sup>-1</sup> could be considered as a single ‘critical P’ concentration for environmental management of non-basalt pasture soils. However, there was clearly an effect of phosphorus buffering index even if the non-basalt soils were statistically homogeneous. For example, for a selection of non-basalt sites with soil test P of 30 (±5) mg kg<sup>-1</sup>, both total P and dissolved reactive P increased by an order of magnitude as phosphorus buffering index fell from 400 (moderate) to 25 (very low) (Fig. 8). This undoubtedly explains some of the variation in the relationship between soil test P and runoff P evident in Fig. 7.

#### Environmental risk prediction from soil test P and P buffering index

Multiple regression was used to predict total P and dissolved reactive P from soil test P and phosphorus buffering index, first with the data from plots covered with shade cloth to minimise confounding from soil erosion. The resultant equation was then used without re-parameterisation on independent data from plots without shade cloth but high vegetative cover, which we took to represent good pasture management. In the model providing the best fit, the runoff P responses to both soil test P and phosphorus buffering index were curvilinear (Eqs. 4, 5), which makes it risky to apply the model to data beyond the range used for parameterisation.

The model gave satisfactory predictions of P concentrations in runoff when applied to the independent data with soil test P < 100 mg kg<sup>-1</sup> and vegetative cover >90 % and no shade cloth (Fig. 10).

The coefficient of determination for total P ( $R^2 = 0.91$ ) using Eq. 4 and for dissolved reactive P ( $R^2 = 0.94$ ) using Eq. 5 were superior to those using the model published by Dougherty et al. (2010) when parameterised with the present data, but the relationships were nevertheless highly significant:

$$\text{Log dissolved reactive P} = -4.71 + 1.038 \cdot (\log \text{Colwell P} + 0.15 \cdot (\log \text{Colwell P} \cdot \log \text{phosphorus buffering index})) \quad (R^2 = 0.76, P < 0.001) \quad (6)$$

$$\begin{aligned} \text{Log total P} = & -3.90 + 1.625 \cdot \log \text{Colwell P} \\ & - 0.1869 (\log \text{Colwell P} \cdot \log \text{phosphorus} \\ & \text{buffering index}) \quad (R^2 = 0.85, P < 0.001) \end{aligned} \quad (7)$$

These results confirm that soil P guidelines for *environmental* management need to reflect both soil P capacity and buffering. This is the conclusion reached by Gourley et al. (2007) for *agronomic* management in Australia, where current practice is to measure Colwell soil test P (a capacity test) and phosphorus buffering index and from these determine a critical concentration using Eq. 8:

$$\begin{aligned} \text{Critical P (Colwell)} = & 19.6 \\ & + 1.1 (\text{phosphorus buffering index}^{0.55}) \end{aligned} \quad (8)$$

Calculations using Eq. 8 result in critical concentrations of Colwell P for agronomic management ranging from 29 mg kg<sup>-1</sup> for soils from very low phosphorus buffering index (36) through to 40 mg kg<sup>-1</sup> for moderately buffered soils (141–280), and 55 mg kg<sup>-1</sup> for soils with high phosphorus buffering index (281–840).

#### Management guidelines

The first step to reducing P concentrations in runoff is to maintain greater than 90–95 % ground cover, which is greater than recommended for managing soil erosion. Where ground cover is low because of P-deficiency, pasture improvement should be undertaken including the use of P fertiliser.

Where cover is high, fertiliser use should be guided by measurements of both soil P capacity and buffering. In Australia, where the Colwell soil test is common, we suggest that Eq. 8 (Gourley et al. 2007) can be used to provide critical P concentrations for environmental risk assessment that have been normalised for phosphorus buffering index.

Available soil P can be raised with fertiliser to improve agronomic production without fear of significantly increasing P losses in runoff provided soil test P remains below the agronomic critical concentration defined by the soil phosphorus buffering index. This supposition needs to be verified experimentally, but in the meantime it provides a defensible approach to environmental risk management.

The Colwell test uses a sampling depth of 10 cm for agronomic application. Although surface runoff is thought to interact mainly with the top few mm of soil (Sharpley 1985), values for soil test P in the 0–2 and 0–10 cm depths are highly correlated (Fig. 1), so the vast amount of soil P data collected for agronomic management should also be useful for environmental management.

Where P-fertiliser can be applied to pasture with little risk of significantly increasing P in runoff, care is still needed to avoid the ‘incidental losses’ associated with recently applied fertiliser (or manure). Once fertiliser P has been sorbed and entered into the various mineral and organic pools, it is much less likely to be mobilised than freshly applied fertiliser (Hart et al. 2004; Davis et al. 2005). Thus, it is management of the fertiliser that matters for environmental protection of water quality, not the concentration of P in the soil *provided* it remains below the critical value and soil erosion is managed by retaining good vegetative cover.

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## References

- Allen BL, Mallarino AP, Klatt JG, Baker JL, Camara M (2006) Soil and surface runoff phosphorus relationships for five typical USA Midwest soils. *J Environ Qual* 35:599–610
- Anon (2006) Best management practices for temperate perennial pastures in New South Wales. New South Wales Department of Primary Industries, ISBN 0 7347 17261, pp 36
- ANZECC/ARMCANZ (2000) Australian and New Zealand guidelines for fresh and marine water quality. NWQMS Paper No. 4. Australian and New Zealand Environment and Conservation Council & Agriculture and Resource Management Council of Australia and New Zealand, Canberra, Australia
- APHA (2005) Standard methods for the examination of water and waste water, 21st edn. American Public Health Association, Washington, DC
- Baginska B, Cornish PS, Hollinger E, Kuczera G, Jones D (1998) Nutrient export from rural land in the Hawkesbury-Nepean catchment. In ‘Proceedings of the 9th Australian Agronomy Conference’. Wagga Wagga. (The Australian Society of Agronomy: Wagga Wagga, NSW.) pp 753–756
- Barlow K, Nash D, Grayson RB (2005) Phosphorus export at the paddock, farm-section, and whole farm scale on an irrigated dairy farm in south-eastern Australia. *Aust J Soil Res* 56:1–9
- Bilotta GS, Brazier RE, Haygarth PM, Macleod CJA, Butler P, Granger S, Krueger T, Freer J, Quinton J (2008) Rethinking the contribution of drained and undrained grasslands to sediment-related water quality problems. *J Environ Qual* 37:906–914
- Burkitt LL, Moody PW, Gourley CJP, Hannah MC (2002) A simple phosphorus buffering index for Australian soils. *Aust J Soil Res* 40:497–513
- Castillo VM, Martinez-Mena M, Albaladejo J (1997) Runoff and soil loss response to vegetation removal in a semiarid environment. *Soil Sci Soc Am J* 61:1116–1121
- Colwell JD (1963) The estimation of the phosphorus fertilizer requirements of wheat in southern New South Wales by soil analysis. *Aust J Exp Agric Anim Husband* 3:190–197
- Cornish PS, Hallissey R, Hollinger E (2002) Is a rainfall simulator useful for estimating phosphorus runoff from dairy pasture? A question of scale-dependency. *Aust J Exp Agric* 42:953–959
- Costin AB (1980) Runoff and soil and nutrient losses from an improved pasture at Ginninderra, Southern Tablelands, New South Wales. *Aust J Agric Res* 31:533–546
- Davis RL, Zhang H, Schroder JL, Wang JJ, Payton ME, Zazulak A (2005) Soil characteristics and phosphorus level effect on phosphorus loss in runoff. *J Environ Qual* 34:1640–1650
- Dorioz JM, Wang D, Poulencard J, Trévisan D (2006) The effect of grass buffer strips on phosphorus dynamics—a critical review and synthesis as a basis for application in agricultural landscapes in France. *Agric Ecosyst Environ* 117:4–21
- Dougherty WJ, Nash DM, Cox JW, Chittleborough DJ, Fleming NK (2008a) Small-scale, high-intensity rainfall simulation under-estimates natural runoff P concentrations from pastures on hill-slopes. *Aust J Soil Res* 46:694–702
- Dougherty WJ, Nicholls PJ, Milham PJ, Havilah EJ, Lawrie RA (2008b) Phosphorus fertilizer and grazing management effects on phosphorus in runoff from dairy pastures. *J Environ Qual* 37:417–428
- Dougherty W, Burkitt L, Milham P, Harvey D (2010) The effect of soil available P and P buffering on runoff P concentration from pastures. In: Gilkes RJ, Prakongkep N (eds) Proceedings of the 19th world congress of soil science; soil solutions for a changing world; ISBN 978-0-646-53783-2; Published on DVD; <http://www.iuss.org>; Division Symposium 3.2; Nutrient best management practices; 2010 Aug 1–6. Brisbane, Australia: IUSS; 2010, pp 272–275
- Douglas RW, Menary W, Jordan P (2007) Phosphorus and sediment transfers in a grassland river catchment. *Nutr Cycl Agroecosyst* 77:199–212
- Ehlert P, Morel C, Fotyama M, Destain J-P (2003) Potential role of phosphate buffering capacity of soils in fertilizer management strategies fitted to environmental goals. *J Plant Nutr Soil Sci* 166:409–415
- Erskine WD, Mahmoudzadeh A, Myers C (2002) Land use effects on sediment yields and soil loss rates in small basins of Triassic sandstone near Sydney, NSW, Australia. *Catena* 49:271–287
- Fiener P, Auerswald K (2009) Effects of hydrodynamically rough grassed waterways on dissolved reactive phosphorus



- loads coming from agricultural watersheds. *J Environ Qual* 38:548–559
- Freebairn DM, Cornish PS, Anderson WK, Walker SR, Robinson JB, Beswick AR (2006) Management Systems in Climate Regions of the World—Australia. In: ‘Dryland Agriculture’. Agronomy Monograph 23. American Society of Agronomy, Crop Science Society of America, Soil Science Society of America, Madison, 2nd edn, Chap. 20, pp 837–878
- Gourley CJP, Melland AR, Waller RA, Awty IM, Smith AP, Peverill KI, Hannah MC (2007) Making better fertiliser decisions for grazed pastures in Australia. Victorian Government Department of Primary Industries, Melbourne
- Hart MR, Quin BF, Nguyen ML (2004) Phosphorus runoff from agricultural land and direct fertilizer effects: a review. *J Environ Qual* 33:1954–1972
- Havilah E, Warren H, Lawrie R, Senn A, Milham P (2005) Fertilizers for pastures. New South Wales Department of Primary Industries, Sydney
- Hollinger E, Cornish PS, Baginska B, Mann R, Kuczera G (2001) Farm-scale stormwater losses of sediment and nutrients from a market garden near Sydney, Australia. *Agric Water Manage* 47:227–241
- Hosomi M, Sudo R (1986) Simultaneous determination of total nitrogen and total phosphorus in freshwater samples using persulphate digestion. *Int J Environ Stud* 27:267–275
- Hughes S, Reynolds B, Bell SA, Gardner C (2006) Simple phosphorus saturation index to estimate risk of dissolved P in runoff from arable soils. *Soil Use Manage* 16:206–210
- Jarvie HP, Withers PJA, Neal C (2002) Review of robust measurement of phosphorus in river water: sampling, storage, fractionation and sensitivity. *Hydrol Earth Syst Sci* 6:113–132
- Lambert D, Maher W, Hogg I (1992) Changes in phosphorus fractions during storage of lake water. *Water Res* 26:645–648
- Loch RJ (2000) Effects of vegetation cover on runoff and erosion under simulated rain and overland flow on a rehabilitated site on the Meandu Mine, Tarong, Queensland. *Aust J Soil Res* 38:299–312
- Loch RJ, Foley JL (1994) Measurement of aggregate breakdown under rain: comparison with tests of water stability and relationships with field measurements of infiltration. *Aust J Soil Res* 32:701–720
- Loch RJ, Robotham BG, Zeller L, Masterman N, Orange DN, Bridge BJ, Sheridan G, Bourke JJ (2001) A multi-purpose rainfall simulator for field infiltration and erosion studies. *Aust J Soil Res* 39:599–610
- Loughran RJ, Elliott GL, McFarlane DJ, Campbell BL (2004) A survey of soil erosion in Australia using caesium-137. *Aust Geog Stud* 42:221–233
- Maher W, Woo L (1998) Procedures for the storage and digestion of natural waters for the determination of filterable reactive phosphorus, total filterable phosphorus and total phosphorus. *Anal Chim Acta* 375:5–47
- Maher W, Krikowa F, Wruck D, Louie H, Nguyen T, Huang WY (2002) Determination of total phosphorus and nitrogen in turbid waters by oxidation with alkaline potassium peroxodisulfate and low pressure microwave digestion, autoclave heating or the use of closed vessels in a hot water bath comparison with Kjeldahl digestion. *Anal Chim Acta* 463:283–293
- Mahmoudzadeh A, Erskine WD, Myers C (2002) Sediment yields and soil loss rates from native forest, pasture and cultivated land in the Bathurst area, New South Wales. *Aust Forest* 65:73–80
- McCaskill MR, Ridley AM, Okam A, White RE, Michalk DL, Melland A, Johnston WH, Murphy SR, Andrew MH (2003) SGS nutrient theme: environmental assessment of nutrient application to extensive pastures in the high rainfall zone of southern Australia. *Aust J Exp Agric* 43:927–944
- McCull R (1978) Chemical runoff from pasture: the influence of fertiliser and riparian zones. *N Z J Mar Freshw Res* 12:371–380
- McDowell RW, Condron LM (2004) Estimating phosphorus loss from New Zealand grassland soils. *N Z J Agric Res* 47:137–145
- McDowell RW, Sharpley AN, Condron LM, Haygarth PM, Brookes PC (2001) Processes controlling soil phosphorus release to runoff and implications for agricultural management. *Nutr Cycl Agroecosyst* 59:269–284
- McDowell RW, Monaghan RM, Morton J (2003) Soil phosphorus concentrations to minimise potential P loss to surface waters in Southland. *N Z J Agric Res* 46:239–253
- McGufficke BR (2003) Native grassland management: a botanical study of two native grassland management options on a commercial cattle property. *Rangel J* 25:37–46
- Moody PW (2007) Interpretation of a single-point P buffering index for adjusting critical levels of the Colwell soil P test. *Aust J Soil Res* 45:55–62
- Murphy J, Riley JP (1962) A modified single solution method for the determination of phosphate in natural waters. *Anal Chim Acta* 27:31–36
- Nash D, Murdoch C (1997) Phosphorus in runoff from a fertile dairy pasture. *Aust J Soil Res* 35:419–429
- Nash D, Hannah M, Halliwell D, Murdoch C (2000) Factors affecting phosphorus export from a pasture-based grazing system. *J Environ Qual* 29:1160–1166
- Nash D, Hannah M, Barlow K, Robertson F, Mathers N, Butler C, Horton J (2007) A comparison of some surface soil phosphorus tests that could be used to assess P export potential. *Aust J Soil Res* 45:397–400
- Pote DH, Daniel TC (2000) Analyzing for total phosphorus and total dissolved phosphorus in water samples. In: Pierzynski GM (ed) Methods of phosphorus analysis for soils, sediments, residuals, and waters. Southern Cooperative Series Bulletin No. # 396. North Carolina State University, Raleigh, pp 94–97
- Rayment GE, Higginson FR (1992) Australian laboratory handbook of soil and water chemical methods. Inkata Press, Melbourne
- Robinson GG, Lazenby A (1976) Effect of superphosphate, white clover and stocking rate on the productivity of natural pastures, Northern Tablelands, New South Wales. *Aust J Exp Agric Anim Husb* 16:209–217
- Roth CH (2004) A framework relating soil surface condition to infiltration and sediment and nutrient mobilization in grazed rangelands of northeastern Queensland, Australia. *Earth Surf Process Landforms* 29:1093–1104

- SCA/DLWC (2003) Sydney catchment authority soil landscape database. Sydney-South Coast Regional, Soil Survey Unit. NSW DLWC, Parramatta
- Schärer M, Vollmer T, Frossard E, Stamm C, Flühler H, Sinaj S (2006) Effect of water composition on phosphorus concentration in runoff and water-soluble phosphate in two grassland soils. *Eur J Soil Sci* 57:228–234
- Sharpley AN (1985) Depth of surface soil-runoff interaction as affected by rainfall, soil slope, and management. *Soil Sci Soc Am J* 49:1010–1015
- Sharpley AN, Ahuja LR, Menzel RG, Smith SJ, Rhoades ED, Olness AE (1981) The sorption of soluble phosphorus by soil material during transport in runoff from cropped and grassed watersheds. *J Environ Qual* 10:211–215
- Sharpley AN, Foy BH, Withers PJA (2000) Practical and innovative measures for the control of agricultural phosphorus losses to water: an overview. *J Environ Qual* 29:1–9
- Smith CM (1987) Sediment, phosphorus, and nitrogen in channelised surface runoff from a New Zealand pastoral catchment. *N Z J Mar Freshw Res* 21:627–639
- Stanton FW (1960) Ocular point frame. *J Range Manage* 13:153
- Stevens DP, Cox FW, Chittleborough DJ (1999) Pathways of phosphorus, nitrogen, and carbon movement over and through texturally differentiated soils, South Australia. *Aust J Soil Res* 37:679–693
- Tunney H, Kiely G, Morgan G, Moles R, Byrne P, Jordan P, Daly K, Doody D, Kurz I, Bourke D, O'Reilly C, Ryan D, Holden N, Jennings E, Irvine K, Carton O (2007) Pathways for nutrient loss to water with emphasis on phosphorus: final integrated report. End of Project Report RMIS 5021. Johnstown Castle Research Centre, Wexford
- Wendt RC, Corey RB (1980) Phosphorus variations in surface runoff from agricultural lands as a function of land use. *J Environ Qual* 9:130–136
- Whitehead DC (2000) Nutrient elements in grassland: soil-plant-animal relationships. CABI Publishing, Wallingford
- Young WJ, Rustomji AO, Hughes D, Wilkins D (2001) Regionalisation of flow variables used in modelling riverine material transport in the National Land and Water Resources Audit. CSIRO Land and Water, Canberra. Technical Report 36/01, August 2001