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## An overview of land application of pig effluent-P using soil P chemistry and mass balance calculations

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### Abstract

Attention is directed at land application of piggery effluent (containing urine, faeces, water, and wasted feed) as a potential source of water resource contamination with phosphorus (P). This paper summarises P-related properties of soil from the 0–0.05 m depth at 11 piggery effluent application sites, in order to explore the impact that effluent application has had on the potential for runoff transport of P.

The sites investigated were situated on Alfisol, Mollisol, Vertisol, and Spodosol soils in areas that received effluent for 1.5–30 years (estimated effluent-P applications of 100–310 000 kg P/ha in total). Total ( $P_T$ ), bicarbonate extractable ( $P_B$ ), and soluble P forms were determined for the soil (0–0.05 m) at paired effluent and no-effluent sites, as well as texture, oxalate-extractable Fe and Al, organic carbon, and pH.

All forms of soil P at 0–0.05 m depth increased with effluent application ( $P_B$  at effluent sites was 1.7–15 times that at no-effluent sites) at 10 of the 11 sites. Increases in  $P_B$  were strongly related to net P applications (regression analysis of log values for 7 sites with complete data sets: 82.6% of variance accounted for,  $P < 0.01$ ). Effluent irrigation tended to increase the proportion of soil  $P_T$  in dilute  $CaCl_2$ -extractable forms ( $P_{TC}$ : effluent average 2.0%, no-effluent average 0.6%). The proportion of  $P_{TC}$  in non-molybdate reactive forms (centrifuged supernatant) decreased (no-effluent average 46.4%, effluent average 13.7%). Anaerobic lagoon effluent did not reliably acidify soil, since no consistent relationship was observed for pH with effluent application. Soil organic carbon was increased in most of the effluent areas relative to the no-effluent areas. The 4 effluent areas where organic carbon was reduced had undergone intensive cultivation and cropping.

Current effluent management at many of the piggeries failed to maximise the potential for waste P recapture. Ten of the case study effluent application areas have received effluent-P in excess of crop uptake. While this may not represent a significant risk of leaching where sorption retains P, it has increased the risk of transport of P by runoff. Where such sites are close to surface water, runoff P loads should be managed.

*Additional keywords:* runoff, phosphorus, swine, hog.

### Introduction

Agricultural industries are being closely scrutinised worldwide as sources of water and soil resource contamination (Sharpley 1995; Abbozzo *et al.* 1996; Burkholder *et al.* 1997). Researchers have focussed considerable attention on management techniques designed to minimise environmental impacts of agricultural waste nutrients (Abbozzo *et al.* 1996; Edwards *et al.* 1996a; Gburek and Sharpley 1997), methods of measuring the mobility of agricultural waste nutrients (Sharpley *et al.* 1996), and the effects of short-term applications of wastes and accidental spills (Daniel *et al.* 1992).

Inadequate supplies of nitrogen and phosphorus (P) often limit the development of nuisance algal growth in surface water bodies, and it is recognised that the most important

human activity related sources of elevated levels of these nutrients are from wastewater discharges and diffuse runoff (ANZECC 1992). Runoff is a particularly important pathway for P transport, as soils often have a large capacity to sorb leaching P. The sorption capacity of soils is well documented and has been studied for decades (e.g. Black 1942).

Export of P from agricultural land by runoff has been documented for various agricultural land uses, including inorganically fertilised pastures (Austin *et al.* 1996), animal manure fertilised pastures (Chaubey *et al.* 1995; Edwards *et al.* 1996b, 1997), and simulated cattle feedlot plots (Dillaha *et al.* 1988). Significant runoff transport of P was observed in each of these cases.

Some research has been published on the longer term effects of livestock wastes on soil conditions and chemistry. Heckrath *et al.* (1995), as part of the Broadbalk Experiment at Rothamsted, investigated the impact of farmyard manure application (40 kg P/ha.year) over a period of >150 years on soil P characteristics. They observed significant concentrations of P in drainage water at a depth of 0.65 m. Of this P, 66–86% was in the form of dissolved molybdate-reactive P at the time of analysis. The authors concluded that leaching losses of P applied as manure can be closely related to soil bicarbonate-extractable P. Pote *et al.* (1999) found that a wide range of published soil test determinations of P were significantly ( $P < 0.01$ ) related to runoff-dissolved molybdate-reactive P, with most relationships exhibiting correlation coefficients >0.90. They investigated 3 Ultisols in their trial, and included the Olsen *et al.* (1954) bicarbonate extract amongst the soil tests studied. Other researchers have also indicated that soils containing high levels of P from excessive fertiliser application can become an important source of dissolved molybdate reactive P in runoff (Edwards *et al.* 1993). King *et al.* (1990) found that applications of piggery effluent over an 11-year period resulted in increases of soil test P to 10 times the level above which no response to P fertilisers would be expected. This elevation resulted from the application of effluent (equivalent to 6100 kg P/ha of total application over 11 years) to 'coastal' bermudagrass [*Cynodon dactylon* (L.)] growing in a Paleudult. In 2 case studies of areas that had received piggery effluent for 19 years (3700 kg net load of P/ha; Natrustalf soil; site P1) and 30 years (310 000 kg net load of P/ha; Haplustalf soil; site P2), Redding (2001) found that P had leached 0.7 m at site P1 and 1.5 m at site P2 (to the base of the soil profile). Increases in surface soil P concentrations (0–0.05 m depth) after effluent application at these sites were also much greater than was observed in a comparable laboratory sorption system (>25 times for P1; >57 times for P2).

Accepted methods for dealing with the quantities of faeces, urine, and wasted feed produced by piggeries in Australia have changed throughout the period that intensive pig rearing operations have existed. During the past 35 years, piggery operations have changed from semi-extensive to dominantly intensive production (Todd 1988), resulting in increases in average herd sizes from 4 sows to approximately 90 sows (Meo and Leary 1998). The 1960s in Australia saw the introduction of effluent ponds on piggeries (Todd 1988), which are today in general use. While some producers were already using piggery wastes as a source of low quality irrigation water prior to the introduction of environmental regulation for the pig industry (e.g. the Environmental Protection Act in Queensland in 1994, Anon. 1996), environmental regulation has accelerated the education of producers and necessitated the implementation of more environmentally sustainable practices. More producers now regard their wastes as nutrient resources, and are seeking innovations to reduce any adverse environmental impacts that may result from the collection, treatment, and reuse of the waste resource.

This paper provides a broader overview of waste P land application than previously published, through a selection of management case histories from Australia, and evaluates the impact of these practices through analysis of changes in surface soil P chemistry (0–0.05 m depth). The goal of these studies was to determine if current and past effluent application practices have increased the potential for contamination of surface water with P. In addition, soil oxalate extractable Fe (iron) and Al (aluminium), pH, and organic carbon were measured in paired effluent and no-effluent sites, allowing observation of the impact of effluent application on these characteristics.

## Materials and methods

### *Site selection, descriptions, and sampling*

Previous contact between Queensland Department of Primary Industries staff and pig producers provided the opportunity to approach producers who practice land application of effluent. All of the available industry contacts were screened for the study, which provided a range of land application sites representative of piggery enterprises and their distribution in Queensland. Where producers were unable to provide concise information on piggery production histories, they were rejected from the study.

From the 14 Australian piggeries and 15 effluent application areas to which access had been gained, 11 sites were selected for the case studies. The areas selected represent a range of Australian soil types, waste management practices (Tables 1 and 2), and herd sizes. The slopes of the effluent application areas selected varied from relatively flat lying (<1% slope, including sites 1, 2, 7, and 11) to sloping (<5%, sites 3, 4, 5, 6, 8, 9, 10). Slopes throughout the effluent application areas at each property, however, were subject to spatial variability. Additional information on sites 7 and 9 can be obtained from Redding (2001).

At each piggery, homogenous paired sites (based on soil type, topography, and management) representative of the soils that were effluent irrigated (EI) and soil that had not received effluent (NE, the unirrigated soils had not been used for crop production) were selected in close proximity to each other. Two composite surface samples (0–0.05 m) were collected at random from each of these sites. These composite surface samples were made by bulking together 25 randomly selected cores collected with a manual push tube sampler. The soil samples were air dried at 40°C within 12 h of collection.

### *Laboratory methods*

The following analytical techniques as set out in Rayment and Higginson (1992) were applied to the soil samples: pH in 1 : 5 soil : water suspension (Method 4A1; detection limit 0.1); organic carbon (OC) content by the method attributed to Walkley and Black (1934, method 6A1; detection limit 0.01%); oxalate-extractable Fe and Al (method 13A1; detection limit 0.001%).

Particle size analysis of the soil samples was carried out using the pipette method described in Gee and Bauder (1986), including sample pre-treatment for removal of soluble salts and organic matter (where necessary).

**Table 1. Application area soil types**  
Soil classification completed using Isbell (1996) and also Soil Survey Staff (1998)

Site	Australian Soil Classification	US Taxonomy
1	Mottled Eutrophic Grey Sodosol	Natrustalf
2	Epiacidic Self Mulching Black Vertosol	Haplustert
3	Endocalcareous Epipedal Brown Vertosol	Haplustert
4	Redoxic Hydrosol	Argiaquoll
5	Acidic Eutrophic Red Ferrosol	Haplustox
6	Massive Black Vertosol	Haplustert
7	Haplic Eutrophic Red Dermosol	Haplustalf
8	Humose Sesquic Semiaquic Podosol	Haplohumod
9	Eutrophic Mesonatric Grey Sodosol	Natrustalf
10	Episodic-Epicallcareous Massive Brown Vertosol	Haplustert
11	Endocalcareous-Endohypersodic Self Mulching Black Vertosol	Haplustert

**Table 2. Effluent application area management practices**  
Effluent irrigated areas are displayed in hectares

Site	Irrigation	Effluent irrigated area	Production
1	Flood	SI <sup>A</sup>	Grazed pasture
2	Mixed with clean water and irrigated from furrows	120	Cereal and cotton cultivation
3	Flood	0.4–32 <sup>B</sup>	One pumpkin crop produced
4	Flood	0.81–2.8 <sup>B</sup>	Grazed or cut pasture
5	Flood	24	Grazed pasture/cereal and legume cultivation
6	Flood	SI <sup>A</sup>	Grazed pasture
7	Flood	0.75	Grazed pasture
8	Solid-set spray	6.9–15 <sup>B</sup>	Horticulture
9	Solid-set spray	10–70 <sup>B</sup>	Horticulture
10	Flood	SI <sup>A</sup>	Legume and cereals
11	Flood irrigated	SI <sup>A</sup>	Broad area legume and cereal cropping

<sup>A</sup>Spatially irregular effluent application, precluding accurate estimation of application to a given area of soil.

<sup>B</sup>A range indicates changing application area throughout the period of piggery operation.

A range of P analyses and extractions were applied to the soils and these methods are listed in Table 3, along with the acronyms applied in the text. Bicarbonate-extractable P ( $P_B$ ) is related to the quantity of labile P (Dalal and Hallsworth 1977; Holford 1980), and was used in this study as an estimate of labile P. The non-molybdate-reactive P forms in the extractant ( $P_{OC}$ ) are used here as an estimate of organic P dissolved in extract solution and not readily hydrolysed by the acid Murphy and Riley (1962) colour reagent. Other authors (e.g. Chardon *et al.* 1997) have previously applied similar calculations. This estimation assumes that any P in the  $CaCl_2$  extracts following centrifuging is in solution. Such an assumption is based on the  $CaCl_2$  extractant's effect in reducing soil dispersion, and the employment of centrifuging appropriate to settle particles of  $>0.2$  mm (Stokes law, as described in Gee and Bauder 1986; extractants centrifuged for 20 min at 2100  $G$ , assuming a particle density of 2600  $kg/m^3$ ). Analyses of dilute  $CaCl_2$ -extractable P ( $P_C$ ) by Moody *et al.* (1988) indicated that this measurement was a reasonable estimate of P intensity, as it was found to be highly correlated with soil solution P (10 kPa matric suction).

#### Estimation of P loads

A nutrient mass balance approach (relying on feed digestibility-related waste estimation) was used to estimate effluent-P loads, in order to avoid inaccuracies related to variations in effluent P concentrations and volumes. The principles applied were those described in Kruger *et al.* (1995). Subsequently, the model described by Casey (1995) was used to estimate the partitioning of P between anaerobic lagoon effluent and

**Table 3. Phosphorus extraction and analysis techniques**

Fraction	Method	Description
$P_T$	9A1 <sup>A</sup>	Total P by X-ray fluorescence; detection limit 0.001%
$P_B$	Colwell (1963)	1 : 100 soil : 0.5 M bicarbonate solution extraction, centrifuged supernatant analysed; detection limit 1 mg P/kg
$P_C$	9F2 <sup>A</sup> , Murphy and Riley (1962b) colorimetric analysis	1 : 5 soil : 0.005 M $CaCl_2$ extract, centrifuged supernatant analysed; detection limit 0.01 mg P/kg
$P_{TC}$	9F2 <sup>A</sup> , ICP analysis	As $P_C$ , except detection limit 0.05 mg P/kg using Thermo Jarell Ash Iris/Duo/HR Duo
$P_{OC}$	$P_{OC} = P_{TC} - P_C$	Calculation

<sup>A</sup> Method fully described in Rayment and Higginson (1992).

**Table 4. Phosphorus application history for each effluent application area and estimated net P Load**

The period over which the net P was applied is represented in years, while all other units are kg P/ha. Net P loads are estimated to 1 (site 2) or 2 significant figures

Site	Total effluent P applied	Total crop P removal	Total fertiliser P applications	Net P load	Period
1	SI <sup>A</sup>				
2	100	150	40	-10	10
3	4900	55	0	4800	14
4	1300	50	0	1300	24
5	380	140	60	300	8
6	SI <sup>A</sup>				
7	310 000	30	0	310000	30
8	120	<10	<10	120	1.5
9	3900	320	50	3600	19
10	SI <sup>A</sup>				
11	SI <sup>A</sup>				

<sup>A</sup>Spatially irregular effluent application, precluding accurate estimation of P application to a given area of soil.

sludge. Casey (1995) assumes that 90% of influent P partitions into the effluent pond sludge, with the remainder retained in the pond supernatant. This assumption was based on reviewed publications and has recently been confirmed by a survey of 15 effluent ponds in North Carolina (Bicudo *et al.* 1999). This composite methodology allowed the estimation of the total application of waste-P (from faeces, urine, and wasted feed) throughout the life of the effluent application areas at most sites (Table 4).

Information provided by producers on crop yields, inorganic fertiliser usage, and management practices, combined with mass balance estimates of waste production, enabled estimation of the net P loads applied to the effluent reuse areas. Crop P removal was estimated using P concentrations obtained from a variety of sources. Data on cereal crop P removal rates were taken from the average values stated in Kruger *et al.* (1995). P removal with cotton harvesting was estimated using the cottonseed lint P content listed in Skerman (2000). Huett *et al.* (1997) indicated that the P content of the horticultural crop produced at sites 8 and 9 varies between 0.1 and 0.5% dry weight basis, although no researcher appears to have measured the P content of the portion of the plant harvested as a vegetable. The concentration of the harvested portion was assumed to be 0.3% P (vegetable moisture content was assumed to be 570% dry basis; J. Olsen, Queensland Department of Primary Industries, pers. comm.). Inaccuracy in this value does not significantly influence the estimated net load, as crop removal is small compared with the effluent applications. Concentrations of P in pumpkin at peak harvest published by Huett *et al.* (1997, 0.50–0.60% P) were used to calculate P removal with pumpkin production.

Removal of P with cattle grazing was assumed to be negligible based on calculations using data from Noble (1996) and Agricultural Research Council (1980). For example, the upper limit of P removal from the irregularly irrigated grazed systems of the case study sites corresponds to maintaining a young herd (average size for the year, 300 kg) at a high stocking density of 0.66 head/ha. Under these circumstances, steer growth rates annually correspond to up to 180 kg/year. If this weight gain is 0.8% P (Agricultural Research Council 1980), then annual removal is likely to be approximately 1 kg P/ha, which is negligible compared with the application rates observed.

## Results and discussion

### *Effluent management*

The piggeries that took part in the study ranged from those employing effluent management practices that would have reflected best management practice prior to 1970 to those that

have kept pace with, or exceeded, industry best management practice throughout the period of operation of the piggery (Tables 1, 2, and 4). For example, site 7 had operated without ponds up until the time of the study and effluent was irrigated directly to land from a sump (management has since been reviewed). At other sites effluent is a well-managed resource used in crop production, and has been so since commencement of operations (e.g. site 8), or for several decades (e.g. site 9). Current industry best practice for the management of piggery wastes is reflected in the industry code of practice for Queensland (Streeten and McGahan 2000). Similar documents have been developed by other Australian States, or are under development. It should be noted, however, that soil samples from the sites are likely to represent soil characteristics developed throughout the period of piggery operations at these sites, and may not accurately represent recent improvements to management.

For 7 of the piggeries studied it was possible to estimate total application of waste-P (from faeces, urine, and wasted feed) throughout the life of the effluent application areas (Table 4). Four effluent application areas were removed from the original total of 15 due to difficulty in establishing what practices had been applied, or difficulties in defining appropriate paired sample sites for comparison to the effluent irrigated areas.

An important observation of the study was the proportion of piggeries that were applying their effluent in a spatially irregular fashion, which is unlikely to maximise effluent nutrient uptake by crops and may increase the potential for nutrient transport by leaching or runoff. Four of the piggery effluent application areas listed in Table 2 were receiving effluent in a spatially irregular fashion at the time of the study, as were 2 of the 4 excluded piggery effluent application areas. In addition, 4 of the effluent application areas were used solely or largely as cattle grazing areas, which is an ineffective method of P removal.

#### *Soil P*

For all soils and forms of P measured, average concentrations increased with effluent application except for those at site 2 (Table 5). At this piggery the comparatively low effluent P application rates and intensive crop production resulted in decreases in  $P_T$  and  $P_{OC}$ . The increases recorded for P forms in the surface soils (0–0.05 m depth) indicate increased potential for runoff transport of P (Pote *et al.* 1999). Increased soil  $P_B$  displayed a strong statistical relationship with net P applications (Table 4), although complete data were available for only 7 paired sites on a variety of soil types. Regression of the log  $P_B$  increase with effluent irrigation against the log net P applications produced a strong positive relationship (82.6% of variance accounted for;  $P < 0.01$ ; log data better approximated normality than untransformed data). While the data for most paired sites that have not received effluent indicate comparable  $P_B$  to published data for similar soils (Tables 1 and 4; e.g. Biggs *et al.* 1999), several of the sites exhibit very high  $P_B$  levels for virgin soils of their respective types. One example is site 7, where the EI area contains much higher  $P_B$  levels than the NE area, but concentrations in the unirrigated area are still higher than expected. This may be as a result of irrigation runoff, rainfall runoff, or irrigation overspray.

Only a small proportion of  $P_T$  was extractable with  $CaCl_2$  solution (0.1–10.1%, with an average of 1.3% for all soil samples; Table 5). With effluent irrigation, however, a number of trends were apparent. Firstly, effluent irrigation led to a greater proportion of  $P_T$  in  $CaCl_2$  solution extractable forms (EI sites, 0.2–10.1%, average 2.0%; NE sites, 0.1–2.8%, average 0.6%; Table 5). The higher proportion of  $P_T$  extractable as  $P_{TC}$  for the irrigated soils is probably indicative of higher P intensity (Moody *et al.* 1988). Secondly, magnitudes of  $P_{OC}$

**Table 5. Properties of the surface 0.05 m depth interval of the soils for the unirrigated and effluent irrigated soils at each of the sites**Data presented as mean  $\pm$  s.d. in mg/kg, except for  $P_T$  which is displayed as a percentage

Site	Waste application	$P_T$	$P_C$	$P_{OC}$	$P_B$
1	No	0.021 $\pm$ 0.001	0.44 $\pm$ 0.01	0.59 $\pm$ 0.06	31.0 $\pm$ 2.8
	Yes	0.055 $\pm$ 0.002	6.20 $\pm$ 0.37	1.10 $\pm$ 0.12	239 $\pm$ 14
2	No	0.099 $\pm$ 0.004	1.87 $\pm$ 0.05	1.25 $\pm$ 0.03	168 $\pm$ 9
	Yes	0.087 $\pm$ 0.008	3.65 $\pm$ 0.67	0.89 $\pm$ 0.04	225 $\pm$ 14
3	No	0.198 $\pm$ 0.004	3.75 $\pm$ 0.88	1.21 $\pm$ 0.21	276 $\pm$ 7
	Yes	0.305 $\pm$ 0.021	16.8 $\pm$ 0.3	1.96 $\pm$ 0.23	879 $\pm$ 30
4	No	0.024 $\pm$ 0.006	3.87 $\pm$ 2.45	2.06 $\pm$ 0.08	79.5 $\pm$ 47.4
	Yes	0.062 $\pm$ 0.001	57.0 $\pm$ 3.2	5.21 $\pm$ 1.00	273 $\pm$ 25
5	No	0.199 $\pm$ 0.001	1.01 $\pm$ 0.05	0.99 $\pm$ 0.06	249 $\pm$ 16
	Yes	0.262 $\pm$ 0.018	3.63 $\pm$ 0.46	1.01 $\pm$ 0.33	424 $\pm$ 7
6	No	0.240 $\pm$ 0.004	0.94 $\pm$ 0.01	0.76 $\pm$ 0.01	194 $\pm$ 9
	Yes	0.632 $\pm$ 0.056	95.8 $\pm$ 8.9	5.57 $\pm$ 1.07	904 $\pm$ 1
7	No	0.172 $\pm$ 0.105	3.88 $\pm$ 4.53	1.55 $\pm$ 0.68	72 $\pm$ 48
	Yes	2.53 $\pm$ 0.49	399 $\pm$ 59	35.6 $\pm$ 4.9	3950 $\pm$ 1960
8	No	0.028 $\pm$ 0.002	0.12 $\pm$ 0.01	0.54 $\pm$ 0.02	49.0 $\pm$ 8.5
	Yes	0.089 $\pm$ 0.011	4.08 $\pm$ 0.12	1.06 $\pm$ 0.11	269 $\pm$ 3
9	No	0.019 $\pm$ 0.001	2.49 $\pm$ 0.25	0.96 $\pm$ 0.01	22.5 $\pm$ 0.7
	Yes	0.072 $\pm$ 0.007	23.9 $\pm$ 0.2	2.52 $\pm$ 0.06	290 $\pm$ 6
10	No	0.051 $\pm$ 0.004	0.35 $\pm$ 0.04	0.57 $\pm$ 0.00	49.5 $\pm$ 3.5
	Yes	0.138 $\pm$ 0.011	10.2 $\pm$ 0.7	1.35 $\pm$ 0.18	339 $\pm$ 5
11	No	0.057 $\pm$ 0.008	1.26 $\pm$ 0.52	0.92 $\pm$ 0.19	119 $\pm$ 67
	Yes	0.103 $\pm$ 0.008	5.35 $\pm$ 1.21	1.33 $\pm$ 0.10	296 $\pm$ 49

increased with effluent irrigation (except for site 2); however, in all EI areas a smaller proportion of  $P_{TC}$  was analysed as  $P_{OC}$  (EI, 5.1–23.9%, average 13.7%; NE, 22.3–82.5%, average 46.4%; Table 5). If the increased  $P_{OC}$  represents an increase in the soluble organic P intensity, then this may indicate greater potential for mobilisation of soluble organic P forms. This can be an important mode of transport of P in some soil systems that have received piggery wastes (Chardon *et al.* 1997).

#### *Changes to other soil characteristics*

Seven of the 11 EI sites displayed increased average OC in the 0–0.05 m depth interval compared with the NE sites (Table 6 and Fig. 1). On average, however, OC in EI sites was only slightly higher than the corresponding NE areas, with the first quartile of the box and whisker plot lying <100% (Fig. 1). Cultivation has been observed to be related to OC decline in soils (e.g. Dalal and Mayer 1986) and it is interesting to note that the 4 sites that displayed decreased OC in the EI areas relative to the NE areas were all cultivated (E areas 2, 5, 10, and 11; Table 2).

On average, effluent alkalinity (piggery effluent pH tends to be slightly alkaline in character; Kruger *et al.* 1995), and other soil or management processes, have tended to counteract any acidifying influence of the nitrification of effluent N (such acidification might be expected with the nitrification of other N sources; Bruce 1997, Fig. 1). Although some EI areas did display soil acidification (e.g. site 10), years of effluent irrigation need not lead to surface soil acidification (0–0.05 m).

Average oxalate-extractable Fe and Al concentrations both appear to have increased slightly in the effluent receiving areas relative to the virgin soils, although this relationship is



**Table 6. Surface 0.05 m depth interval characteristics for the unirrigated and effluent irrigated soils, including organic carbon and oxalate-extractable Fe and Al**Data are presented as mean  $\pm$  s.d. or a single value where unreplicated, and all are reported as a percentage except pH

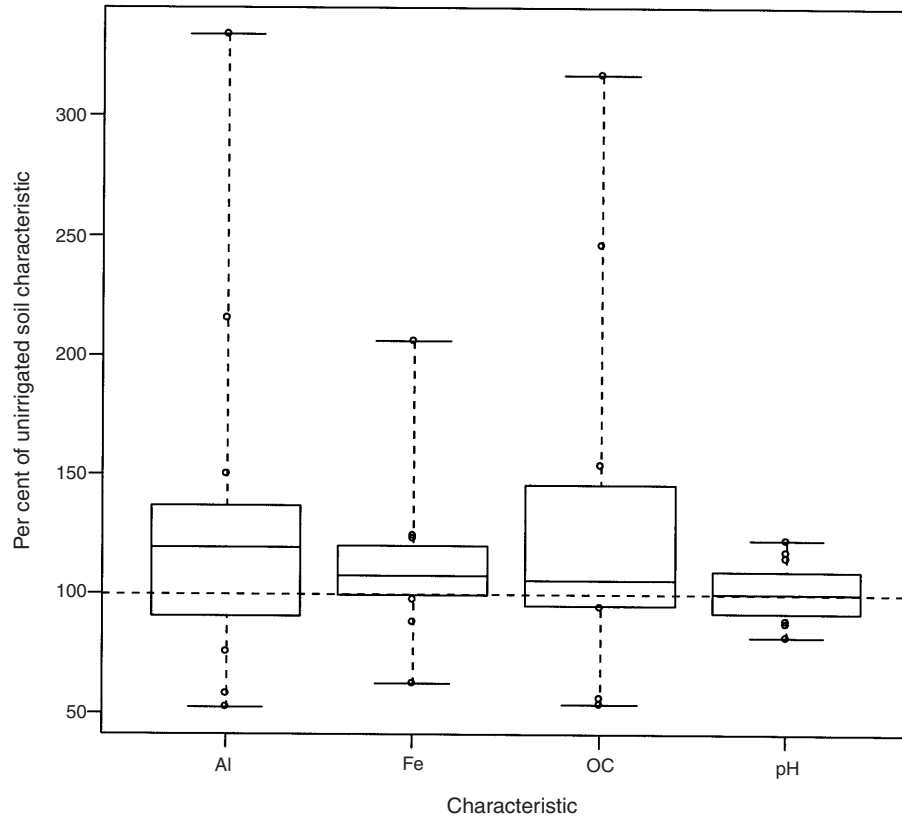
Site	Effluent	Al	Fe	OC	Silt	Clay	pH
1	No	0.027 $\pm$ 0.001	0.112 $\pm$ 0.000	0.60 $\pm$ 0.07	4	9	6.4 $\pm$ 0.1
	Yes	0.059 $\pm$ 0.003	0.230 $\pm$ 0.012	0.93 $\pm$ 0.18			6.7
2	No	0.230 $\pm$ 0.035	0.316 $\pm$ 0.030	2.38 $\pm$ 0.25	15	49	7.27 $\pm$ 0.06
	Yes	0.174 $\pm$ 0.006	0.279 $\pm$ 0.006	1.28 $\pm$ 0.04			6.36 $\pm$ 0.15
3	No	0.259 $\pm$ 0.034	1.11 $\pm$ 0.07	1.95 $\pm$ 1.20	26	52	6.71 $\pm$ 0.08
	Yes	0.323 $\pm$ 0.007	1.26 $\pm$ 0.03	2.48 $\pm$ 0.67			6.45 $\pm$ 0.16
4	No	0.065 $\pm$ 0.003	0.058 $\pm$ 0.014	1.85 $\pm$ 0.00	9	5	4.85 $\pm$ 0.03
	Yes	0.034 $\pm$ 0.000	0.036 $\pm$ 0.000	2.55 $\pm$ 0.35			5.72 $\pm$ 0.06
5	No	0.277 $\pm$ 0.007	1.44 $\pm$ 0.01	3.20 $\pm$ 0.21	22	42	5.02 $\pm$ 0.01
	Yes	0.329 $\pm$ 0.021	1.54 $\pm$ 0.06	3.05 $\pm$ 0.07			5.03 $\pm$ 0.01
6	No	0.274 $\pm$ 0.005	1.49 $\pm$ 0.03	2.10 $\pm$ 0.00	26	27	6.28 $\pm$ 0.08
	Yes	0.301 $\pm$ 0.014	1.52 $\pm$ 0.11	5.18 $\pm$ 0.04			5.57 $\pm$ 0.11
7	No	0.166 $\pm$ 0.001	0.827 $\pm$ 0.091	3.43 $\pm$ 0.88	22	34	6.14 $\pm$ 0.15
	Yes	0.096 $\pm$ 0.001	0.859 $\pm$ 0.045	10.9 $\pm$ 0.1			6.16 $\pm$ 0.01
8	No	0.140 $\pm$ 0.007	0.091 $\pm$ 0.003	1.90 $\pm$ 0.28	7	9	5.69 $\pm$ 0.09
	Yes	0.147 $\pm$ 0.033	0.088 $\pm$ 0.017	1.93 $\pm$ 0.32			5.42 $\pm$ 0.13
9	No	0.014 $\pm$ 0.003	0.061 $\pm$ 0.000	0.93 $\pm$ 0.11	4	5	5.94 $\pm$ 0.06
	Yes	0.018 $\pm$ 0.003	0.075 $\pm$ 0.007	0.98 $\pm$ 0.04			6.82 $\pm$ 0.09
10	No	0.133 $\pm$ 0.004	0.422 $\pm$ 0.019	1.70 $\pm$ 0.00	>1	50	8.11 $\pm$ 0.10
	Yes	0.199 $\pm$ 0.001	0.494 $\pm$ 0.002	1.60 $\pm$ 0.00			6.65 $\pm$ 0.24
11	No	0.105 $\pm$ 0.011	0.274 $\pm$ 0.040	2.38 $\pm$ 0.53	7	20	6.09 $\pm$ 0.01
	Yes	0.350 $\pm$ 0.002	0.341 $\pm$ 0.023	1.33 $\pm$ 0.04			7.47 $\pm$ 0.06

not consistent across all sites (Table 6, Fig. 1). Australian piggery effluent is known to contain Fe (0.23–2.4 mg Fe/L in the 0.45 mm filtered fraction; K. D. Casey, unpublished data), although concentrations of Fe and Al were not measured in the effluents of this study. The oxalate reagent extracts poorly crystalline Fe and Al phases (Rayment and Higginson 1992), and amorphous forms of both Fe and Al are linked to reducing soluble P in soils (Singh and Gilkes 1991; Holford 1997). Increases in these constituents may increase soil P sorption capacity.

### Conclusions

Past effluent reuse practices at these piggeries have established large increases in surface soil P forms for most of the effluent application areas, increasing the potential for transport of P in runoff water. The effluent application areas studied have received effluent for up to 30 years (Table 4), and to some extent the increased potential for runoff transport at the older application sites reflects application practices that are no longer in effect. Some of the piggeries at the time of the study, however, were applying their waste in a manner that precluded the calculation of application rates or did not maximise crop production (6 out of 15 effluent application sites initially screened). These practices prevent effective reuse of the waste resource, and demonstrate a lack of effective environmental management of these sites.

For the 11 different soil management systems studied, no consistent changes to soil oxalate-extractable Fe, oxalate-extractable Al, or pH were observed with effluent irrigation in the medium (1.5 years) to long term (30 years). While consistent increases in OC were



**Fig. 1.** Boxplots of soil oxalate extractable Al, oxalate extractable Fe, organic carbon, and pH for the effluent irrigated areas as a percentage of the average value of the corresponding unirrigated paired sites.

not observed across all sites, it was found that those sites that exhibited decreased soil OC with effluent irrigation were all intensively cultivated and cropped (4 sites out of the 11).

Given these observations of P accumulation in effluent application areas and runoff transport potential there is a strong case for runoff treatment, capture, or other management to reduce P loads as a precaution where effluent application areas are in close proximity to surface water (such as the use of vegetated filter strips; Dillaha *et al.* 1988; Edwards *et al.* 1996a).

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