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Iron Man Gypsum treatment: reducing phosphate losses from soils used for dairy production

Yoongarillup Downs experiment

HEALTHY ESTUARIES WA

Revitalising Geographe Waterways

VASSE taskFORCE

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Summary

Many coastal sandy soils have poor phosphorus (P) retention capacity and are prone to becoming saturated with the nutrient. Loss of soluble P in runoff and leaching from P-saturated coastal sandy agricultural soils is a major cause of eutrophication in southern Western Australian estuaries. Incorporating high-P-adsorbing materials into these P saturated soils offers a management option to increase P retention and reduce P losses to waterways.

Healthy Estuaries WA (2020 to 2024) and Revitalising Geographe Waterways are programs funded by the Government of Western Australia’s Royalties for Regions, building on the work of the Regional Estuaries Initiative (2016 to 2020). These programs aim to improve the health of South West estuaries by reducing nutrient losses from their catchments. This report presents the first scientific findings from research into innovative remediation options to treat high-nutrient-loss soils in catchments – supported by the three programs.

Iron Man Gypsum (IMG) is a high-P-adsorbing material. It is a secondary product of titanium ore processing by Iluka Resources in Capel. At the time of publication, IMG was the only well-researched material with proven effects on P mobility that was available in bulk quantities for trials on farms near the Vasse region. During the past 20 years, IMG has been tested in a range of soil and water treatment situations by the CSIRO, ChemCentre and predecessors of the departments of Water and Environment, and Primary Industries and Regional Development.

While IMG prevents P leaching over the long-term when mixed into soils at high rates (>150 t/ha), these rates are impractical for grazing systems. Further research is needed to assess whether using IMG at lower rates can achieve similar benefits in different soil settings. We conducted a paddock-scale replicated plot experiment on a commercial dairy farm to investigate the benefits and risks of mixing lower rates of IMG (< 60 t/ha) into soils used for intensive pasture grazing.

IMG was mixed into the P-saturated topsoil at rates of 20 and 60 t/ha during pasture renovation in 2019. The soils were deep ploughed with a mouldboard plough before the IMG was spread and mixed into the topsoil (0–10 cm) ahead of reseeding with a mix of biennial grasses. Small amounts of P fertiliser were applied before and during the experiment, during which cattle grazed significant amounts of P and returned it unevenly as manure. Leachates from the soils were collected using pan lysimeters and analysed for nutrients and a broad range of other elements. This was accompanied by analyses of pasture tops during the growing season and soils in the summer dry period.

The IMG treatments immediately prevented losses of up to 24 kg P/ha from the topsoils, even at the lowest application rate of 20 t/ha, with no effects on pasture growth. Soil P adsorption capacity was slightly increased by both application rates – reducing the P in soil-water (indicated by $CaCl_2$-P) without effects on P available for plant growth. The reduced soil-water P in turn reduced the concentrations of P that could be leached from the soils by more than 90%. The longevity of these effects will

most likely depend on the balance of P removal by grazing and return as manure, which varies across the paddock. The IMG treatments also reduced leaching of organic nitrogen (N) and P but had no effect on leaching of the ammonium (NH_4) or nitrate (NO_3) that dominated total N losses of up to 270 kg N/ha/yr. The treatment also benefited soil quality: it increased soil pH and slightly increased its capacity to retain cations.

Treating soils with IMG increases manganese (Mn), cobalt (Co) and sulfate (SO_4) in leachates, but their toxicity risks are likely mitigated by increased leaching of calcium (Ca). Higher levels of such metal leaching have not been found in previous experiments in Western Australia and it appeared to be mediated by the lower initial pH of the soils. Concurrent leaching of Ca mitigated the risk of Mn affecting pasture plants and also likely protected aquatic organisms from Mn toxicity where leachates reached downstream environments. SO_4 also leached from the IMG-treated soils at much greater rates, but the concentrations were within the range seen in some untreated soils. Increased leaching of SO_4 is expected to wane after six years at the 60 t/ha application rate.

Mixing IMG into soils during pasture renovation can be part of a management strategy to immediately treat P-saturated soils in areas prone to surface runoff or leaching directly to waterways. In the Vasse region, a rate of 20 t/ha is sufficient to achieve immediate environmental benefits; that is, reduced P losses with minimal risks of increased leaching of Mn, Co and SO_4. Improvements to the soil properties did not yield short-term economic returns as increased pasture production, but the long-term effects are yet to be investigated.

IMG treatment faces cost hurdles associated with its purchase and freight. Reducing these to achieve greater on-farm retention of P would offer a more cost-effective approach for the society-at-large, compared with off-farm treatment of P in waterways and receiving environments. Further investigation of the effects of IMG on soil properties, P loss and patterns of Mn, Co and SO_4 leaching will be carried out under the Healthy Estuaries WA program, alongside additional trials in different soil and farm settings. These are aimed at building confidence among farmers and agronomists around the use of IMG as a soil amendment in pasture production systems.

1 Introduction

Iron Man Gypsum (IMG) is a high-P-adsorbing soil amendment. It has shown significant promise for improving retention of P as phosphate in sandy topsoils and reducing losses to waterways. The amendment is a dark brown, loamy material with 14% iron as iron oxide minerals in a gypsum matrix (Douglas et al. 2012; Degens & Shackleton 2016). IMG is produced by Iluka Resources at its Capel production facility as a secondary product of ilmenite ($FeTiO_3$) ore processing (Wendling et al. 2012). The material is the oxidised, dried, mixed and screened solids recovered from a sulfuric acid liquor that has been neutralised with $Ca(OH)_2$ – the liquor being the waste generated after washing of the ilmenite ore concentrate. At the time of publication, IMG was the only well-researched material with proven effects on P mobility that was available in bulk quantities in the lower South West for trials on farms.

Topdressing of IMG on pastures in Bassendean sands can increase surface-soil P retention and reduce P leaching. These sands are common on the Swan coastal plain and are widely used in agricultural production. Surface application of IMG at rates ranging from 10 to 50 t/ha on sandy soils north of Perth minimised leaching of P for the five years of monitoring (Sharma et al. 2018). However, in this study leaching losses were the result of P fertiliser applied during the experiment, with the soils initially having negligible plant-available P. Mixing with sandy soils at a higher rate of 150 t/ha at a turf farm achieved >97% reduction in leached P, as well as accumulation of organic matter (Douglas et al. 2010a, 2010b). The amendment has also been claimed to increase soil water-holding capacity (Douglas et al. 2010b, 2010a), but this has not been confirmed by direct measurement and has only been inferred from leachate measurements. IMG has been extensively investigated for other uses, such as treating soluble P in groundwater flowing to subsoil drains beneath urban developments (Degens & Shackleton 2016) and surface waters pumped from Ellen Brook (ChemCentre 2016; Department of Biodiversity and Attractions 2018).

Many sandy soils on the Swan coastal plain have an inherently poor capacity to retain P and prevent loss to leaching or runoff (Ritchie & Weaver 1993; Bolland et al. 2003). After many decades of fertiliser application for agricultural production, large areas of these soils are now saturated with P (Summers et al. 2020) that is leached to shallow groundwater or washed in runoff each winter. Overland flow can often be the largest direct pathway of P loss to waterways (Gerritse & Schofield 1989; Ruprecht & George 1993). This typically happens when low-lying sandy soils with a shallow watertable are saturated during rainfall events. The P loss from pasture soils is further magnified by high P accumulation and the saturated-P retention capacity common in the surface few centimetres of pasture soils (Ryan et al. 2017).

Applying and mixing IMG into sandy soils can improve P retention, reduce P leaching losses and – at higher rates – may yield the additional long-term benefit of improved carbon storage. This is particularly relevant for agricultural soils that have accumulated amounts of P many times more than plant requirements. It may take some decades for the P status of these soils to ‘run down’ (Gerritse & Schofield 1989). During this process, the soils will continue to generate high concentrations of P in runoff and leachates**.**

The optimal IMG application rate for minimising P loss from agricultural soils with a legacy of high P has not yet been determined. Surface-applied (top-dressed) IMG of 20 t/ha was sufficient to capture leaching losses of up to 15 kg P/ha/yr (Sharma et al. 2018), while mixing-in IMG at 150 t/ha captured up to 72 kg P/ha/yr (Douglas et al. 2010a). Mixing may yield greater benefits than top-dressed applications, particularly for soils where the whole surface soil-P adsorption capacity has been saturated with historic fertiliser or grazing practices.

This experiment investigated whether applying IMG at 20 t/ha could be as effective as 60 t/ha in terms of relative benefits and risks to the agricultural production system and the environment. We used a paddock-scale replicated plot design on pasture used for commercial dairy production that had a high risk of P losses to the local waterway. We measured the improvement in P retention and soil quality, as well as the agronomic benefits of IMG application during renovation of the pasture.

2 Background and design

The experiment was conducted at Garry and Tiffany Haddon's property: Yoongarillup Downs, 607 Yoongarillup Road, Sabina River, 13 km south-east of Busselton, Western Australia. The property is located in the upper part of the Vasse Diversion Drain's catchment.

2.1 Site description

The experiment site was in one of the main feed-production paddocks for the milking herd (1,000 to 1,100 head) where previous whole-paddock soil testing indicated a high risk of P loss in runoff. A shallow surface drain crosses the site's southern boundary with seasonally waterlogged flats either side of it. This drain flows via other channels to the regional drainage system that discharges into the Vasse Diversion Drain. The land gently slopes across the site with a fall of about 1.5 m from a rise in the south to the waterlogged flat in the north.

The soils are gradational sands – typically grading from grey sands to mottled yellow to reddish-yellow sandy clay subsoils. These vary from top-slope to foot-slope locations, particularly in the thickness of the overlying sand horizons (Figure 1). In the top-slope area of the paddock, the sands grade to yellow mottled sands at 10 to 30 cm depth (soil 3 being 35% of the experiment area in Figure 1), whereas in the mid-slope location (soil 2 being 30% of the experiment area in Figure 1) there are bleached sands at this depth with thinner mottled yellow sands above the sandy clays. Clay subsoils are shallower in the foot-slope locations (soil 1 being 34% of the experiment area in Figure 1) and topsoils are more frequently mottled, with bleached horizons being uncommon. These soils were consistent with the regional Abba sand (AB2) series, described as light grey sands with a grey organic stained surface overlying an apedal yellowish/greyish/brownish sandy clay loam or sandy clay at about 50 cm depth (Tille & Lantzke 1990).

Previous tests (in 2011 and 2013) of composite samples (diagonal sampling across the paddock) indicated the soils contained high concentrations of available P (48–114 mg/kg Colwell P) and generally low P retention (phosphorus buffer index or PBI 22–69). Follow-up pre-experiment testing stratified by soil type (March 2018) confirmed high but variable available P (13–86 mg/kg Colwell P) with low to moderate PBI (20–154) in the topsoils across the site. Subsoil PBI (10–20 cm) ranged from low (17) to very high (94), generally being lowest in the grey seasonally waterlogged sands (soil 1; Figure 1) and greatest in the top-slope yellow subsoils (soil 3; Figure 1). These layers also contained varying levels of available P ranging from 4 to 82 mg/kg. Mottled sands and clayey sands in some deeper horizons (30–70 cm) had PBI ranging 68 to 300. The history of high P application has resulted in the phosphorus retention index (PRI) being negative or very low in the topsoil of the grey sands, therefore indicating significant risk of P transport either by leaching or surface overland flow.

Soil P adsorption indicators: PBI and PRI

Phosphorus buffer index (PBI) and phosphorus retention index (PRI) are commonly used indicators of soil P adsorption properties. These are related but indicate different aspects of the status of soil P adsorption.

PBI is an index of the inherent, P adsorption capacity of soils prior to any fertiliser application (pre-land development). This does not vary with the current P fertility of the soil or history of P fertiliser. PBI was developed to assist with agronomic recommendations about soil P requirements for plant production, specifically indicating the tendency for a soil to tie up applied P in forms that are unavailable to plants.

In contrast, PRI is an index of the current state of the available P adsorption capacity (or residual capacity) and effectively varies depending on the P fertility history. The index was specifically developed for coastal WA sandy soils to indicate the residual P adsorption capacity remaining in sand soils following a history of land use to guide management of P leaching or runoff risks.

2.2 Experiment design

The experiment was a randomised replicated plot design covering 4.7 ha that compared untreated soils with those treated with 20 t/ha or 60 t/ha of IMG. This was carried out in conjunction with a routine pasture renovation program that involved liming to address surface and subsurface low pH, mouldboard ploughing and reseeding. While reseeding is a common practice, liming and mouldboard ploughing are not.

Before treatment, crushed limestone (ex Redgate Lime, Witchcliffe) was spread at 5 t/ha, then the surface 200 mm was inverted with a mouldboard plough to bury the topsoil weeds and surface organic matter. Untilled sub-plots were retained in the untreated areas to assess the effects of tillage (mouldboard ploughing followed by rotary harrowing). IMG was spread in a single pass with a custom-made twin spinner spreader with a load cell mounted on a 4WD truck. Each pass spread IMG in a 12.5 m strip with two adjacent strips applied in each plot. The site was treated with a further 5 t/ha of crushed limestone which, along with the IMG, was mixed into the surface 100 mm of soil with a single pass using a rotary power harrow. This also levelled the surface for seeding.

Using a double disc zero-till air-seeder, all plots were seeded each year with a mix of annual ryegrass (e.g. Winter Star II; *Lolium multiflorum* ssp. *westerwoldicum)* and biennial ryegrass (e.g. Concord II; *Lolium multiflorum*), along with a base of rose clover (*Trifolium hirtum*) and Persian clover (*Trifolium resupinatum*).

Each plot was 180 to 190 m in length and 24 m in width (including 2 m buffer zones) except for the untreated plots that were split into ~12 m plots with tillage treatments (Figure 1). Overall, 2.7 ha was treated with IMG.

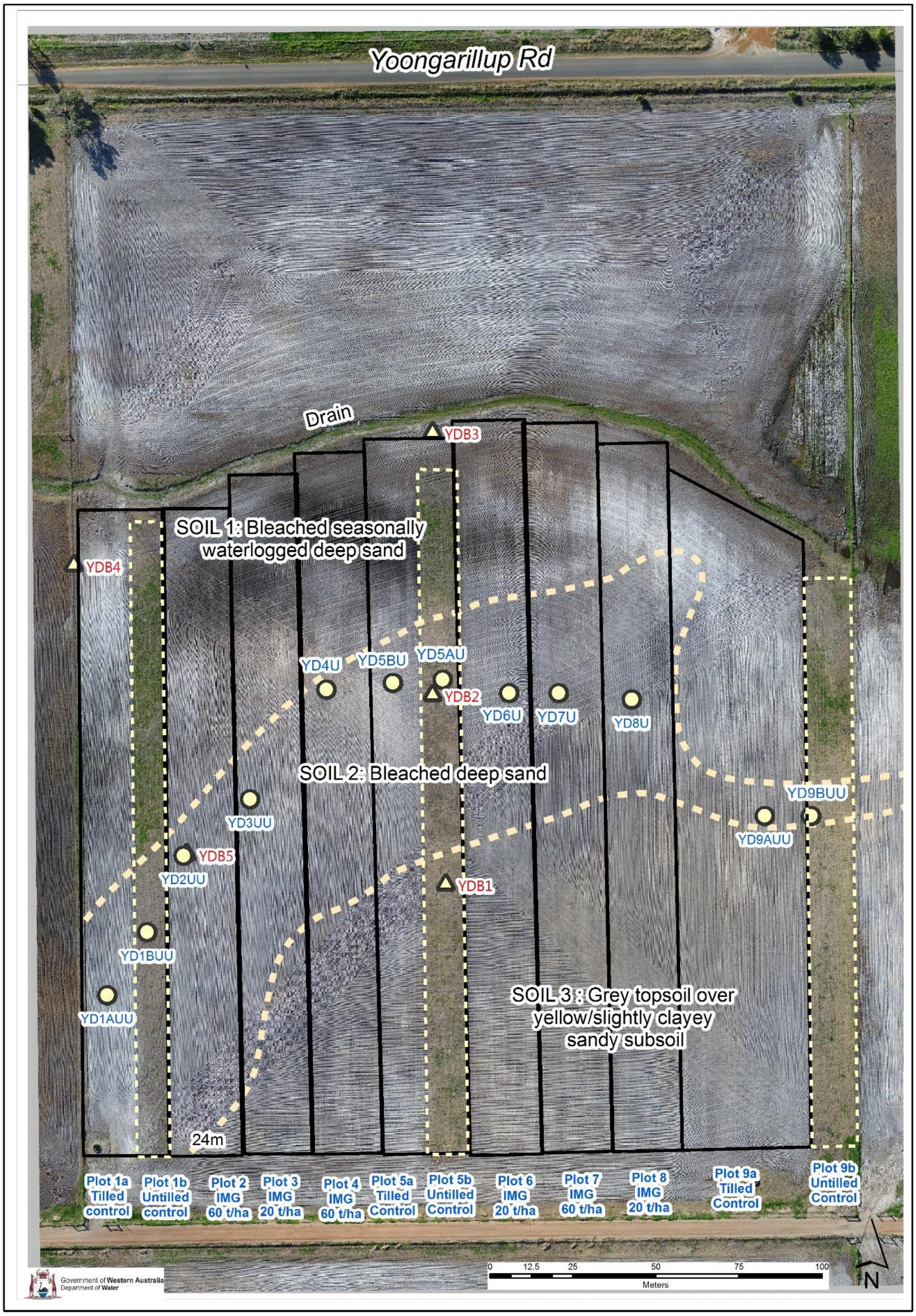

Figure 1: IMG experiment design on Yoongi Downs with monitoring sites (lysimeters labelled in blue, bores labelled in red).

The site was grazed in two strips (laterally across the plots) generally over one and a half days in three- to four-week cycles during winter and spring, then widened to the whole paddock being grazed every five weeks or so in summer and autumn. In winter and spring, the paddocks were fertilised after every grazing with mostly N- and K-based fertilisers. P was occasionally applied with the N and K, giving an annual rate of 6 kg P/ha/yr. Kieserite ($MgSO_4$) and trace elements (Cu, Zn, Mo) were included with one application of fertiliser in the first winter.

2.3 Sampling and analysis methods

2.3.1 Soil and plant sampling

Plant-tissue testing of ryegrass was carried out annually in the winter–spring period three to four weeks after grazing. Samples were taken in mid-August and again in mid-October across all soil types in 2018. In 2019, samples were collected separately from each soil type in each plot in mid-September. Samples of leaf and stem tissue (representing whole plant tops) were collected by hand as a centre transect of 20 grab samples, with each grab sorted to exclude non-target plants. Samples were analysed as per the specifications in Appendix A. For nutrient comparison purposes, the ryegrass was assumed to be similar to *Lolium multiflorum*.

Pasture growth was measured three times during the first spring, at three- to four-week intervals from early September to late November immediately before the site was grazed. Biomass was estimated in each plot, across the three soil-hydrology zones, by taking a transect of 40 to 50 point measurements with a rising plate meter calibrated with plant oven-dry weights from 10 x 0.1 m^2 quadrants.

Sampling of topsoil (0–10 cm) was stratified in each plot by soil-hydrology zones and carried out in summer. Composite samples of 10 x 40 mm cores or 20 x 20 mm drill cores of topsoil per soil zone were collected and submitted for analysis (details in Appendix A). This included measurement of standard soil properties as well as P in soil-water P by analysis of phosphate-P in a 0.01 M $CaCl_2$ soil extract (termed $CaCl_2$-P). $CaCl_2$-P is widely used as an indicator of soil-water P (Moody 2011). P adsorption properties were assessed by analysis of the phosphorus buffer index or PBI (Burkitt et al. 2002) and the phosphorus retention index or PRI (Allen & Jeffery 1990). Both measures are indicators of soil-P buffer capacity with PBI effectively indicating inherent, pre-development P adsorption capacity and PRI indicating current residual capacity to adsorb further P (Bolland et al. 2003).

Soils (0–10 cm) at each lysimeter site were also sampled in summer each year. Composite samples of 20 cores were collected using a 20 mm drill corer from the same 2 m x 1 m plot to the south of each lysimeter site and submitted for analysis (see Appendix A for details).

2.3.2 Lysimeters

Zero-tension-pan lysimeters were used to collect leachates beneath the surface 10 cm in soil 2 – the soil with the lowest PBI where P was most likely to leach and be lost to runoff further downslope as groundwater seepage through the soil during winter. Ten lysimeters were installed on five plots on 31 May 2018, with site conditions being too wet to allow installation on the remaining plots. These were installed in January 2019. Some leachate was likely missed in 2018 as more than 100 mm of rainfall was recorded in the fortnight before the pans were installed.

Leachate collected in the drums was retrieved using a hand-vacuum extraction pump every two to four weeks depending on rainfall, with volumes measured to the nearest 50 mL. These were used to calculate total fluxes of nutrients and elements. See Appendix B for more details on the lysimeters.

Measurements of pH, EC and temperature were made in the field with samples taken for nutrients, anions and metals as per the specifications in Appendix A.

2.3.3 Data processing and statistical analyses

Results that were below analysis reporting limits (typically water quality analyses) were expressed as half the concentration of the reporting limit for the purposes of statistical analyses.

Phosphorus environmental risk index (after D. Weaver, pers. comm. 2019) or PERI was calculated from soil analyses as Colwell P/PBI (Moody 2011). P_{95} fertility index was calculated as the ratio: Colwell P/Colwell $P_{95\%prod}$ (Weaver & Wong 2011). $Colwell_{95\%prod}$ is the Colwell P required for 95% maximum pasture production given the PBI of each site from the formula $Colwell_{95\%prod} = 19.6 + 1.1*PBI^{0.55}$ (Gourley et al. 2019).

Volume-weighted concentrations of constituents in leachates were calculated for each winter–spring leaching period as:

$$C = \frac{\sum_{i=1}^{n} q_i c_i}{\sum_{i=1}^{n} q_i}$$

Where C = volume-weighted concentration (average annual concentration), q_i = volume of leachate for a specific period i, c_i = the concentration of a constituent in leachate collected for period i with n being the maximum number of leaching event samples for a year.

The statistical significance of differences between treatments were tested using one way (or two way) ANOVA following testing of normality using the Shapiro-Wilk test (SigmaPlot vers. 13). Transformations (log normal or square root) were applied when this was not met to allow fitting of the statistical models. Differences between means were deemed significant when $P<0.05$ and pairwise comparisons carried using the Holm-Sidak test (SigmaPlot vers. 13).

3 Results

3.1 Plot-scale soil properties (summer)

The three soil zones across the site were typically acidic fine-grained sands with moderate organic C contents. Untreated soils had $\mathrm{pH}_{\mathrm{CaCl2}}$ ranging from 4.8 to 5.1 with organic C ranging from 3.9 to 4.3%, which contributed to a CEC ranging from 6.8 to 7.7 c mole_c/kg (Table 1). These properties were similar between the soil zones.

P retention was generally very low in the untreated soils and was not significantly modified by tillage (mouldboard ploughing) despite the soil being inverted to 200 mm. The PBI of soil 1 was greater than the other two soils ($P<0.05$) and was highly variable for a sandy soil, ranging from 42 to 103 in untilled plots. There was a similar pattern with PRI, which ranged from 10 to 50. During summer sampling, iron oxides had precipitated along root channels in soil 1 and may explain the spatial variation in P adsorption properties. It was observed that tillage of soil 3 brought some of the deeper orange mottled sands to the surface, which doubled the PBI of some plots (in the site’s south-east) but had no measurable effect on PRI and PBI across the plots.

IMG treatment similarly increased the P adsorption capacity of soil 2 and 3 in the first year but there was no statistical difference between the rates. Across the two years, the PBI of soil 2 was increased by 14 to 23 and in soil 3 by 12 to 18 in both IMG treatments (Table 1; Table 2). There were similar patterns with PRI ,which was increased by 7 to 11 in soil 1 and 11 to 22 in soil 2 across both years. IMG treatment had no significant effects on PBI or PRI in soil 1. This was probably because the effect of the IMG treatment was swamped by the greater spatial variation in this soil (e.g. with PBI in the untreated but tilled plots varying from 26 to 74).

Tillage of the soils by mouldboarding had initial effects in reducing $CaCl_2$-P that depended on soil type. In the first year, tillage had reduced $CaCl_2$-P by half in soil 3 that persisted into the second year (Table 1; Table 2). The tillage effect in soil 2 was not clear in the first summer, but by the second year was similarly evident, with $CaCl_2$-P almost half that of the untilled soil. In contrast, there were no effects of tillage in soil 3 in either year (Table 1; Table 2). Across all soils over the two years, between 2 and 43% of plant-available P was present as $CaCl_2$-P in the untreated soils (Table 1; Table 2). This was consistently greatest in soil 2 (19–43%) where $CaCl_2$-P ranged from 7 to 15 mg/kg and the least in soil 1 (1–18%) where concentrations ranged from 0.6 to 1.2 mg/kg.

$CaCl_2$-P after IMG treatment was typically reduced to less than 1 mg/kg for all soils irrespective of the application rate, with the effect persisting over the two years (Table 1; Table 2). The reduction was greatest in soil 2, which had the highest concentrations at the start of the experiment.

Table 1: Summary soil P properties of topsoil (0–10 cm) stratified by soil-hydrology zones in the first summer after application.

Soil-hydrology zone and treatment		$CaCl_2$-P (mg/kg)	Colwell P (mg/kg)	PRI	PBI	P95 fertility index[2]	Colwell K (mg/kg)	SO_4-S (KCl extractable as mg/kg)	pH ($CaCl_2$)	Organic carbon (%)	CEC (c $mole_c$/kg)
Soil 1	No IMG (untilled)	1.2a[1]	52a	33a	81a	1.7	177a	259ab	5.07a	4.4a	6.8a
	No IMG (tilled)	1.2a	32a	15a	46a	0.9	145a	83a	5.40a	3.5a	4.3a
	20t/ha	0.1b	20a	21a	50a	0.7	93a	328b	5.80b	2.9a	4.8a
	60t/ha	0.1b	26a	42a	57a	0.9	81a	958c	5.70b	2.7a	7.1a
Soil 2	No IMG (untilled)	10.5a	38a	2a	19a	1.6	153a	285a	4.85a	3.9a	7.4b
	No IMG (tilled)	15.2a	44a	1a	17a	1.9	179a	44a	4.75a	3.6a	4.1a
	20t/ha	0.9b	28a	8b	31b	1.0	162a	574a	5.28b	3.7a	6.6b
	60t/ha	1.1b	40a	9b	31b	1.5	190a	1037b	5.33b	3.6a	7.8b
Soil 3	No IMG (untilled)	14.1a	85a	2a	37a	3.1	508a	90ab	4.95a	4.3a	7.7a
	No IMG (tilled)	7.3b	73a	6a	44a	2.6	254b	80a	5.10a	4.0a	7.7a
	20t/ha	0.5c	55a	20b	61b	1.8	137b	424b	5.43b	3.4a	6.1a
	60t/ha	0.2c	49a	28b	61b	1.6	139b	1410c	5.70b	3.7a	10.5a

[1] Mean values followed by the same letter in each soil group are not statistically different (P>0.05).

[2] P_{95} fertility index being the plant-available P relative to that required for a 95% production target (Weaver & Wong 2011).

Table 2: Summary soil P properties of topsoil (0–10 cm) stratified by soil-hydrology zones in the second summer after application.

Soil-hydrology zone and treatment		$CaCl_2$-P (mg/kg)	Colwell P (mg/kg)	PRI	PBI	Fertility index[2] (P_{95})	Colwell K (mg/kg)	SO_4-S (KCl extractable as mg/kg)	pH ($CaCl_2$)	Organic carbon (%)	CEC (c $mole_c$/kg)
Soil 1	No IMG (untilled)	0.6 a[1]	39 a	21 a	70 a	1.3	204 a	116 a	5.1 a	5.5 a	7.0 a
	No IMG (tilled)	0.6 a	22 a	14 a	43 a	0.8	122 a	76 a	5.5 a	2.7 b	3.8 c
	20t/ha	0.1 a	39 a	24 a	60 a	1.3	205 a	498 a	6.0 b	2.9 b	7.0 ab
	60t/ha	0.1 a	21 a	25 a	57 a	0.7	135 a	945 b	6.1 b	2.9 b	5.7 b
Soil 2	No IMG (untilled)	13.3 a	73 a	0.7 a	27 a	2.8	380 a	113 ab	5.2 a	4.5 a	7.3 a
	No IMG (tilled)	7.0 b	38 a	2.7 a	19 a	1.6	257 a	88 a	5.1 a	3.1 b	4.2 b
	20t/ha	0.4 c	29 a	11 b	41 b	1.0	165 a	517 b	5.7 b	3.2 b	5.3 a
	60t/ha	0.3 c	38 a	14 b	42 b	1.4	291 a	1596 c	5.8 b	3.3 b	7.4 b
Soil 3	No IMG (untilled)	12.1 a	71 a	1.8 a	36 a	2.6	421 a	85 a	5.1 a	5.0 a	8.3 a
	No IMG (tilled)	2.1 b	62 a	7.1 ab	46 a	2.2	259 a	65 a	5.4 a	3.7 b	5.1 c
	20t/ha	0.2 b	49 a	18 bc	64 b	1.6	211 a	499 a	5.7 b	3.5 b	6.3 bc
	60t/ha	0.4 b	52 a	25 c	58 ab	1.8	320 a	1569 b	5.8 b	3.0 b	7.4 ab

[1] Mean values followed by the same letter in each soil group are not statistically different (P>0.05).

[2] Fertility index being the plant-available P relative to that required for a 95% production target (Weaver & Wong 2011).

The range of $CaCl_2$-P induced by the treatments was weakly to well correlated (r = 0.28–0.61) with Cowell P each year but well correlated with PERI (r = 0.88–0.93; P<0.001) with a similar relationship each year. PRI was also correlated with $CaCl_2$-P, but as a negative exponential function (r = 0.63–0.71) that differed between the years for PRI<2.

Colwell P was not different between treatments after the first year, despite reduced $CaCl_2$-P and increased P retention capacity (Table 1). This effect also remained after the second year. There was variation in Colwell P of typically 20 to 30 mg/kg within treatments in each year and differences of up to 20 mg/kg in the same plots between the years that could have masked any slight trends. This variation in Colwell P contributed to a consistently high P_{95} fertility index: averaging 1.6 each year with a range of 0.7 to 3.1 across the treatments, depending on the soil type. The tilled (but untreated) and IMG-treated plots in soil 1 (in the waterlogged zone) had an index consistently less than 1 over the two years. This indicates that available P in these areas might be insufficient for optimum pasture production but only if other factors are not limiting growth. Frequent waterlogging – and the effect of this on N availability – is likely to be posing significant limitations to pasture growth in this soil.

IMG treatment increased plant-available S in most of the soils, and slightly increased pH and CEC. S concentrations were slightly decreased one year after tillage (mouldboard ploughing), although the effect was not statistically significant because of more than two-fold variation across the plots. These concentrations were more than 10 times that needed for pasture production (8 mg/kg, Gourley et al. 2019). With IMG treatment, available S increased by more than four-fold in the 20 t/ha rate and 10-fold in the 60 t/ha rate with the effect sustained over two years (Table 1). The IMG treatments increased soil pH by 0.3 to 0.5 in the first year, with the effect persisting into the second year. There were also slight increases in the CEC of soils treated with IMG: this was clearest in soil 2 (Table 1; Table 2).

The increased Ca in the IMG treatments had effects on exchangeable Mg (an indicator of plant-available Mg) but had no effects on plant-available K. Available K was above critical concentrations of 126 mg/kg (including the confidence interval 109–142 mg/kg) for sand soils (Gourley et al. 2019) and was similar across the IMG treatments in both summers (Table 1; Table 2). This was despite large variation in exchangeable Ca being expected to accelerate K leaching (Figure 2) and was probably because of regular K applications in fertiliser. In contrast, exchangeable Mg was generally lower in the IMG treatments (P<0.05) in both years but not different between the application rates (Figure 2). Exchangeable Mg concentrations were >0.2 meq/100 g, which was in the range considered sufficient for plant growth (Aitken & Scott 1999), while exchangeable Ca did not result in Ca:Mg ratios greater than 20 that might hinder Mg uptake (Fenton & Conyers 2002).

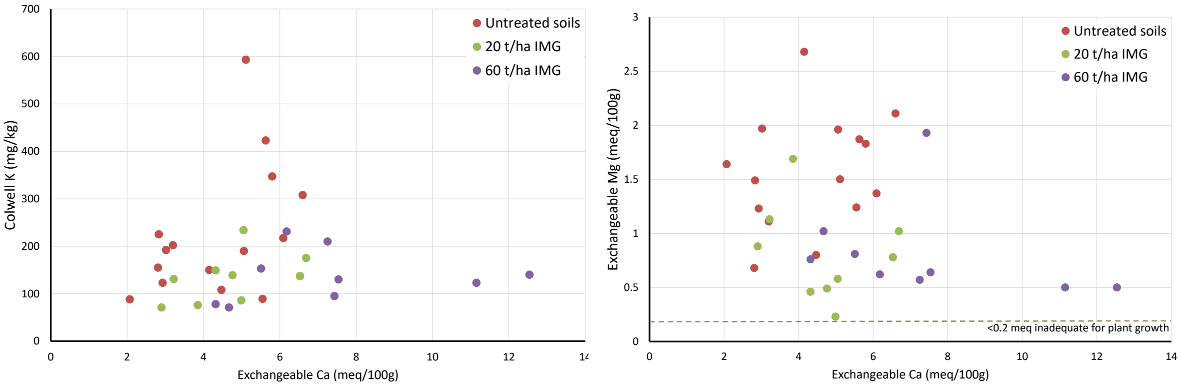

Figure 2: Colwell K and exchangeable Mg in relation to exchangeable Ca of the Yoongi Downs experiment in the first summer after establishment.

3.2 Properties of soils at the lysimeter sites

Soils at each lysimeter site were generally similar for P-adsorption characteristics, although there was some divergence in available P compared with the broader soil 2 area under each treatment.

Colwell P initially ranged from 23 to 100 mg/kg at each lysimeter and was typically more than 10 mg/kg greater than the soils elsewhere in each corresponding plot (Table 1; Table 2). The greatest concentrations were at the lysimeters in the untilled treatments with no IMG, with a large range (22–58 mg/kg) in the tilled soils (with and without IMG). There was an emerging trend of increasing Colwell P at most sites over the two years that averaged 2 mg/kg (Table 3) but this was confounded by significant spatial variation. Colwell P increased up to 38 mg/kg at some untreated sites but decreased up to 36 mg/kg at others. These contrasted with no trends for Colwell P in whole plot analyses of soil 2 because of the much greater variation at this scale.

$CaCl_2$-P at the start of the experiment ranged from less than 0.9 mg/kg in IMG-treated soils to 2 to 25 mg/kg in the untreated soils at each lysimeter site. These were similar to soil 2 elsewhere in each plot for each treatment. As with Colwell P, there was an emerging increasing trend in average $CaCl_2$-P (of 2 mg/kg) over the two years (Table 3). The changes over two years were characterised by increases of up to 14 mg P/kg in untilled soils with no IMG, but less than 1 mg P/kg in the IMG-treated soils, with no difference between the IMG treatment rates (Table 3).

PBI and PRI of soils at the lysimeter sites also showed similar patterns to those of soil 2 elsewhere in each plot over the two years (see section 3.1). Most of the untreated sites had a PBI ranging from 14 to 24 (Table 3), except the lysimeters in plot 9a and 9b on the site’s eastern side where the PBI was 44 to 60. Similarly, the PRI of the sites in the plots with no IMG (tilled and untilled) ranged from –2 to 0.4 (Table 3) except in plots 9a and 9b where PRI was 1 to 7.

Table 3: Average soil P properties at the lysimeter sites over two years.

Treatment	**$CaCl_2$-P (mg/kg)**			**Colwell P (mg/kg)**			**PBI**			**PRI**		
	After setup	**1st summer**	**2nd summer**	**After setup**	**1st summer**	**2nd summer**	**After setup**	**1st summer**	**2nd summer**	**After setup**	**1st summer**	**2nd summer**
No IMG (untilled)	14.5	12.9	14.9	77	73	79	31	30	33	–0.7	–0.1	1.0
No IMG (tilled)	6.4	8.7	12.0	54	53	57	30	29	28	2.2	1.7	3.4
20t/ha IMG	0.3	0.5	1.0	32	29	36	34	33	33	6.8	6.5	8.7
60t/ha IMG	0.1	1.3	0.8	51	59	52	61	43	45	29.7	8.6	13.6
Two way ANOVA results	Treatment P < 0.001 Date P = 0.73 Interaction P = 0.98			Treatment P = 0.03 Date P = 0.97 Interaction P = 1.00			Treatment P = 0.04 Date P = 0.68 Interaction P = 0.93			Treatment P < 0.001 Date P = 0.009 Interaction P< 0.001		

Soils treated with IMG had a PBI ranging from 20 to 40 in the 20 t/ha treatment and 35 to 55 in the 60 t/ha treatment over the two years. This corresponded with PRI ranging from 4.9 to 10 in the 20 t/ha treatment and 6.7 to 16 in the 60 t/ha treatment (Table 3). These properties were similar over the two years, except in the 60 t/ha IMG treatment (Table 3) where the average PBI and PRI were initially 18 and 21 units greater ($P<0.05$) respectively than the following summers.

3.3 Plant uptake of nutrients and trace elements – whole plots

Most major nutritional elements (P, S and Ca) and essential trace elements (Fe, Mn and Mo) were within ranges adequate for the dominant ryegrass on the plots and well below levels that could be toxic to the plants (Table 4; Table 5). The exceptions to this were Cu, Zn and Mg. Cu and Zn were generally below concentrations adequate for growth in all tilled treatments over the two years (Table 4; Table 5). Lower concentrations of Mo were also found in the tilled treatments, but these were within the adequate range for grass growth. Mg concentrations in August of the first year were also marginally less than the adequate range, but only in the IMG treatments. Furthermore, ratios of Mg and Ca relative to K (as K:Ca+Mg with concentrations as meq/kg) were generally <2.2 across the site, indicating that low Mg concentrations did not pose a risk of grass tetany (metabolic Mg) deficiency in grazing cattle (Kemp & t'Hart 1957). With application of Mg in fertiliser after August 2018, concentrations were greater after this time and differences between the treatments were not significant in subsequent sampling.

N and K concentrations were less than the ranges considered adequate for ryegrass growth (i.e. marginal) in some plots in the first year but this was unrelated to the treatments (because the differences between treatments were not significant). Sampling stratified by soil type in each plot in the second year showed that nutrient concentrations were not greatly influenced by soil type (Table 5).

There were consistent significant increases in plant Mn and Ca concentrations in both years at the IMG application rate of 60 t/ha (Table 4; Table 5). There were similar effects in the 20 t/ha treatments, except the increases were slower to emerge; being statistically significant in the second year but not the first (Table 4; Table 5). The relative increases in plant Ca (but not Mn) concentrations appeared to be increasing each year; that is, from more than 1.3-fold in 2018 to more than 1.8-fold in 2019.

Other trace elements such as Cd, Th and U were mostly less than 0.02 mg/kg over the two years with no systematic differences between treatments. For the essential elements, Se was present at concentrations of <0.06 mg/kg whereas Ni concentrations were <3. 6 mg/kg, and Cr concentrations were <5 mg/kg (although mostly <1 mg/kg).

Table 4: Summary nutrient and element concentrations in whole Concord II ryegrass tops in the first growing season (2018).

Date	Treatment	N	P	K	S	Ca	Mg	Cu	Zn	Mn	Fe	Co	Mo
		%						mg/kg					
August 2018	No IMG (untilled)	3.5a	0.53a	2.87a	0.35a	0.42a	0.19ab	5.7a	37a	42a	137a	0.31a	3.5a
	No IMG (tilled)	3.5a	0.45b	2.03b	0.29a	0.40a	0.21b	3.5b	22b	32a	89a	0.10a	1.7b
	20t/ha	2.9a	0.41b	2.03b	0.31a	0.50a	0.15ab	3.6b	19b	67a	104a	0.19a	0.9b
	60t/ha	2.8a	0.36b	1.60b	0.33a	0.66b	0.13a	3.4b	16b	126b	233b	0.43a	0.5b
October 2018	No IMG (untilled)	3.0a	0.45a	2.83a	0.32a	0.56a	0.23a	6.1a	34a	84a	91a	0.17a	2.0a
	No IMG (tilled)	3.0a	0.50b	2.83a	0.31a	0.60ab	0.25a	5.8a	29a	54a	104ab	0.16a	2.5a
	20t/ha	2.8a	0.43a	2.50a	0.32a	0.82c	0.20a	5.3a	24a	127b	117b	0.23a	1.6a
	60t/ha	3.1a	0.46ab	2.67a	0.37a	0.79b	0.21a	5.8a	24a	150b	113b	0.30b	1.5a
	Adequate range[2]	3.0–4.2	0.35–0.5	2.0–3.2	0.24–0.41	0.25–0.3	0.16–0.5	6–15	25–150	25–250	50–60	NA	0.15–0.5
	Toxic range[2]	NA	NA	NA	NA	NA	NA	NA	>400	>1330	NA	NA	>60

[1] Within each sampling time, average values (of three replicates) followed by the same letters are not statistically different (P>0.05).

[2] Adequate and toxic concentration range of whole plant tops for *Lolium multiflorum* (Italian ryegrass) for Mn, K, Cu and Zn (McDonnell et al. 2018; Pinkerton et al. 1997), all other values estimated from *Lolium perenne* (perennial ryegrass) (Rosas et al. 2007; Pinkerton et al. 1997). NA = not known. Values below adequate range are shaded grey.

Table 5: Summary nutrient and element concentrations in whole ryegrass tops in the second growing season (Sept, 2019).

Soil-hydrology zone and treatment		**N**	**P**	**K**	**S**	**Ca**	**Mg**	**Cu**	**Zn**	**Mn**	**Fe**	**Co**	**Mo**
		%						mg/kg					
Soil 1	No IMG (untilled)	4.2bc	0.47b	3.1abc	0.44d	0.41a	0.31c	3.7ab	23	73a	158	0.12	0.74
	No IMG (tilled)	4.4c	0.50bc	2.8a	0.39bcd	0.44a	0.31c	2.9a	24	56a	163	0.14	0.59
	20t/ha	4.1abc	0.47b	3.0a	0.43d	0.56b	0.29c	3.4ab	23	175bc	173	0.23	0.84
	60t/ha	4.1abc	0.35a	2.8ab	0.43d	0.62bc	0.23b	3.2ab	21	270d	211	0.50	0.67
Soil 2	No IMG (untilled)	4.0abc	0.66d	3.9de	0.37abc	0.43a	0.21ab	5.6c	31	61a	114	0.12	2.83
	No IMG (tilled)	4.3bc	0.58c	3.8de	0.35ab	0.37a	0.22ab	4.4ab	24	48a	116	0.13	1.81
	20t/ha	3.8ab	0.50b	3.1abc	0.39abcd	0.69c	0.20ab	3.5ab	19	134b	135	0.20	0.59
	60t/ha	3.8ab	0.49b	3.4bcd	0.43d	0.72c	0.17a	3.8ab	20	177c	143	0.48	0.42
Soil 3	No IMG (untilled)	4.0abc	0.61d	4.3e	0.36abc	0.39a	0.21ab	6.3c	39	51a	106	0.15	2.30
	No IMG (tilled)	3.9bc	0.56cd	4.0de	0.33a	0.37a	0.23b	4.1ab	22	50a	105	0.14	1.64
	20t/ha	3.6a	0.47b	3.6cde	0.34ab	0.59b	0.19ab	3.5ab	19	137bc	125	0.27	1.03
	60t/ha	3.7ab	0.48b	3.7cde	0.41cd	0.61bc	0.20ab	3.5ab	20	196c	146	0.58	0.41
	Adequate range[2]	3.0–4.2	0.35–0.5	2.0–3.2	0.24–0.41	0.25–0.3	0.16–0.5	6–15	25–150	25–250	50–60	NA	0.15–0.5

[1] Average values (of the three replicates) followed by the same letters are not statistically different (P>0.05).

[2] Adequate concentration ranges as per Table 4.

There were no significant differences in dry matter production between the untreated tilled soils and any of the IMG treatments for the three measurements carried out in the first spring (Table 6). Initial differences between the untreated soils with and without tillage reflected residual biomass and weeds left from the previous grazing. Pasture biomass across the site ranged from 2.7 to 3.1 t dry matter/ha in September increasing to 6.1 to 6.9 t dry matter/ha in November.

Table 6: Standing pasture biomass (t dry matter/ha) before grazing.

Treatment	**Pre-grazing sampling date**		
	1 September 2018	**16 October 2018**	**7 November 2018**
No IMG (untilled)	4124 a[1]	3341 a	5769 a
No IMG (tilled)	3067 b	3445 a	6141 a
20 t/ha IMG	2805 b	2708 a	6985 a
60 t/ha IMG	2773 b	2856 a	6437 a

[1] Mean values followed by the same letter for each measurement date are not statistically different (P>0.05).

3.4 Leached nutrient concentrations

Concentrations of leached nutrients captured using lysimeters were less in the IMG-treated soils compared with the untreated soils, particularly for various forms of phosphorus.

More than 80% of leached P from the untreated soils was soluble P, which is considered to be mostly phosphate-P (Table 7). The balance was mostly organic P (comprising 10% of total P), with fine colloidal P being mostly less than 5%.

Average soluble-P concentrations (indicating phosphate-P) were greatest in the untreated soils and consistently least in the IMG-treated soils. Concentrations leached from tilled soils were slightly greater than from untilled soils in the first year, but similar and much greater in both in the second year when more soils were measured (Figure 3). Greater spatial variation in leachate concentrations was captured in the second year after two sets of additional lysimeters were installed in untreated plots.

Average soluble P concentrations leached from IMG-treated soils were far less variable across the years and consistently less than 0.35 mg P/L (Table 7). In contrast, soluble P concentrations leached from tilled soils with no IMG ranged from 0.6 to 22 mg/L over the two years, a similar range to that observed in untilled soils with no IMG. On average, soluble P concentrations leaching from the IMG-treated soils were > 90% less than those leaching from the soils with no IMG.

Similar patterns were also clear with other organic and colloidal components of leached P, with the greatest concentrations measured in the leachate from untreated soils. Concentrations leached from IMG-treated soils tended to remain similar during each year (Figure 3). On average they were 65 to 90% less than the untreated soils over the two years, with a smaller reduction in the second year (Table 7).

Concentrations of N in leachate varied widely between the lysimeters, irrespective of treatment, with the effect of the IMG treatments only on leaching of organic N. Annual average TN concentrations ranged from 19 to 135 mg/L (Table 8) and declined each year. During the first year, average TN concentrations ranged from 32 to 65 mg/L in June and decreased to less than 25 mg/L after July in most soils (Figure A11 in Appendix C). By contrast, in the following June, average TN concentrations were 433 mg/L in the first leachates which fell to less than 50 mg/L after July (Figure A11). Higher concentrations of TN tended to be dominated by oxidised forms of N such as nitrate (NOx; Table 8) that varied greatly during each year (Figure A12 in Appendix C) with no effect of the IMG treatments. In comparison, dissolved organic nitrogen (DON) concentrations typically varied little each year (Figure A11 in Appendix C) and were reduced by at least 60% in the IMG treatments (Table 8). There were similar effects on dissolved organic carbon (DOC) concentrations (Table 7).

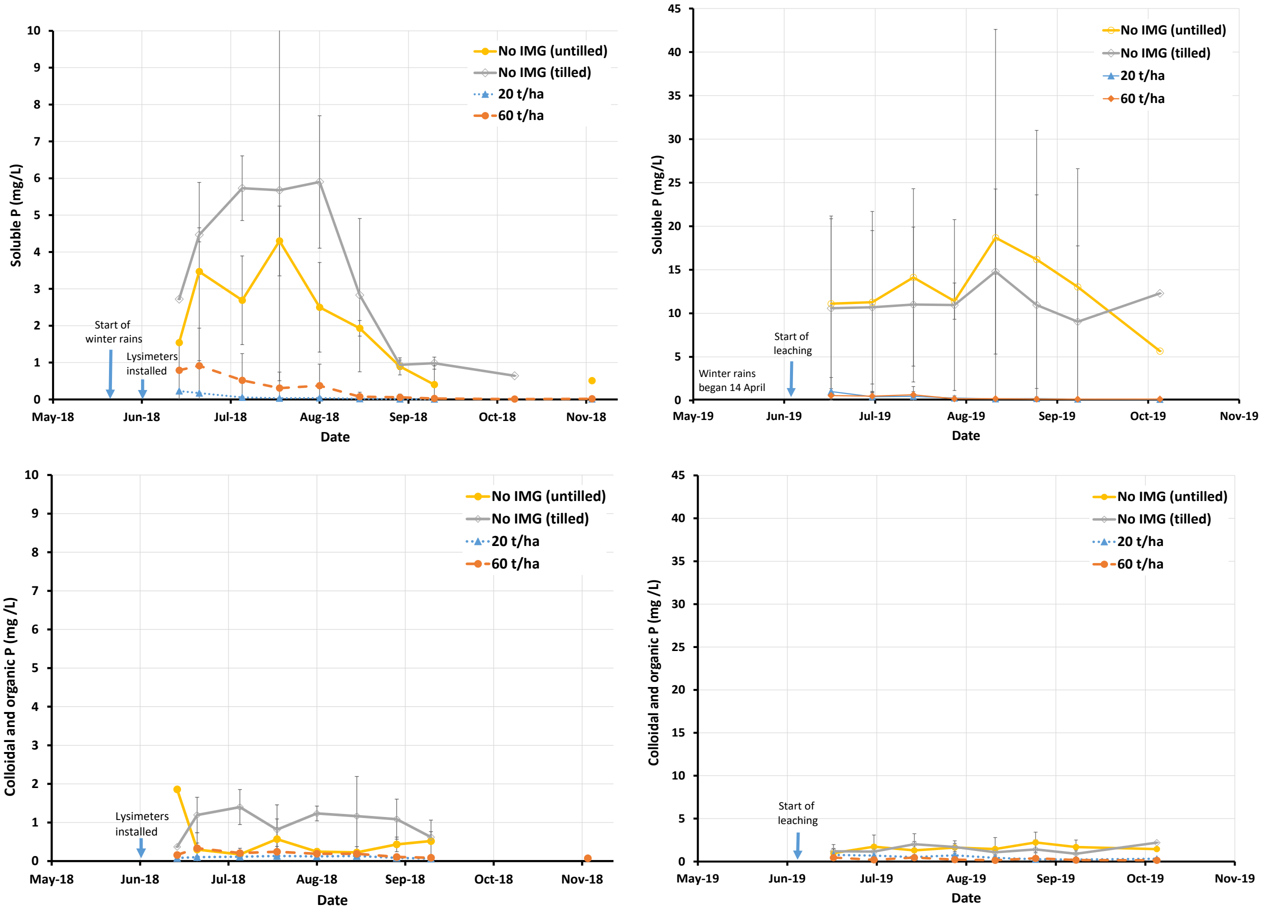

Figure 3: P in leachates from untreated and IMG-treated soils over two seasons.

Table 7: Annual average concentrations of P fractions and DOC (mean ± standard deviation) in leachates from the bleached sand topsoil (soil 2).

Treatment	**Soluble P (FRP)**				**Colloidal and organic P**[1]				**Organic P**[2]				**Dissolved organic C**			
	2018		**2019**		**2018**		**2019**		**2018**		**2019**		**2018**		**2019**	
No IMG (untilled)	2.15a[3]	±1.55	13.07a	±10.45	0.45b	±0.47	1.57a	±0.89	NM[4]	NM	0.94	±0.67	102a	±31	263a	±147
No IMG (tilled)	3.53a	±2.62	11.03a	±7.86	1.03a	±0.50	1.43ab	±0.76	NM	NM	0.97	±0.57	136a	±53	223a	±65
20 t/ha	0.05b	±0.07	0.35b	±0.61	0.11b	±0.03	0.49bc	±0.45	NM	NM	0.24	±0.12	38b	±14	101ab	±31
60 t/ha	0.32b	±0.57	0.32b	±0.42	0.19b	±0.17	0.28c	±0.22	NM	NM	0.15	±0.12	46b	±22	62b	±18

[1] Colloidal and organic P calculated as TP – FRP.

[2] Organic P calculated as total filterable P – FRP

[3] Where tested, values followed by the same letter in each year are not statistically different between treatments ($P<0.05$)

[4] NM = not measured

Table 8: Annual average concentrations of N fractions (mean ± standard deviation) in leachates from the bleached sand topsoil (soil 2).

Treatment	**Total N**				**Nitrate and nitrite (NO_x)**				**Ammonia (NH_4-N)**				**Dissolved organic N**			
	2018		**2019**		**2018**		**2019**		**2018**		**2019**		**2018**		**2019**	
No IMG (untilled)	19a[1]	±19	135a	±209	6.2a	±13.3	92a	±197	4.2a	±5.3	7.9a	±9.3	7.9a	±2.4	29.2a[1]	±31.7
No IMG (tilled)	27a	±13	45a	±50	3.4a	±6.5	24a	±49	12.8a	±14.2	1.5a	2.2	9.6a	±4.9	16.2a	±4.8
20 t/ha	17a	±12	32a	±57	6.4a	±9.6	21a	±50	5.6a	±6.1	3.5a	±6.1	3.2b	±1.5	6.0b	±1.7
60 t/ha	25a	±24	48a	±91	11.6a	±20.7	40a	±91	7.0a	±9.3	3.2a	±3.8	4.3b	±3.2	4.2b	±1.5

[1] Where tested, values followed by the same letter in each year are not statistically different between treatments ($P<0.05$).

3.5 Other leached element concentrations

Concentrations of SO_4, Ca and Mn in leachates were greater in the IMG-treated soils but for most trace metals were similar or greater in the untreated soils. Leachate pH ranged from 5.18 to 7.18 but was generally similar between the untreated and treated soils (see Table A14 in Appendix D). Similarly, salinity (indicated by Cl concentrations) varied widely between the lysimeters with no pattern across the treatments – other than being more variable in the untilled soils with no IMG.

Concentrations of SO_4 and Ca increased greatly with the higher rate of IMG treatment. Between years, annual average concentrations of SO_4 were similar in the treated soils (Table 9) but increased in the untreated soils in the second year (because of greater spatial variation captured across the experiment with the use of more lysimeters). Concentrations leached from IMG-treated soils ranged from 81 to 515 mg/L and were less variable than those leached from the treated soils. In the 20 t/ha treatment the concentrations ranged from 782 to 804 mg/L and were almost twice that of the 60 t/ha treatment at 1409 to 1535 mg/L (Table 9). Ca concentrations were also greater at the higher IMG rate and typically exceeded 230 mg/L throughout each year.

Concentrations of SO_4 and Ca each winter–spring leaching period were greatest in the first leachates and steadily decreased thereafter (Figure A10 in Appendix C). In the 60 t/ha treatment, annual concentrations of SO_4 and Ca decreased by 25 to 30% to an average of between 1400 and 1200 mg/L and 408 and 606 mg/L respectively. In the 20 t/ha treatment the annual concentrations decreased by more than 50% to an average of between 411 and 862 mg/L and 223 and 359 mg/L (Figure A10). It was notable that concentrations of Ca and SO_4 leached from some untreated soils were initially similar to those in the IMG treatments, at least in the second year. However, concentrations leached from untreated soils sites rapidly decreased through the year to less than 200 mg SO_4/L and less than 150 mg Ca/L (Figure A10 in Appendix C).

In contrast with Ca, concentrations of other cations such as Mg and K increased slightly – but not significantly – in the IMG treatments (Table A14 in Appendix D). As with Ca and SO_4, the greatest concentrations were associated with initial winter leaching: these were more than four times the concentrations in the following months.

Mn concentrations increased more than 30 times in the leachates from IMG-treated soils. More than 95% of leached manganese was in soluble form with a small component of particulate Mn (most likely washing of soil particles into the lysimeters). Mn concentrations steadily increased during the first year, reaching a maximum in late August, and then remained similar throughout the second year (Figure 4). These seasonal changes were generally poorly correlated ($r<0.2$; $P>0.1$) with other properties of the leachates (pH, alkalinity or Ca concentrations), except alkalinity in the first year ($r = 0.87$; $P<0.001$). Average annual total Mn concentrations ranged from less than 0.39 mg/L in untreated soils, increasing to 8.4 to 18 mg/L in the treated soils (Table 9) with no difference between the rates of applied IMG. Concentrations leached from the IMG-treated soils were also similar between the years, although exceeded 30 mg/L for a small number of events at the end of the first year. Average concentrations weighted by leachate volume are often more than double averages over time because of the effect of higher concentrations in early winter (see Table A15 in Appendix D).

The leaching of Ca and Mg resulted in average hardness being typically more than 600 mg $CaCO_3$/L in the IMG-treated soils and generally greater in the 60 t/ha treatment than the 20t/ha treatment (Table 9). Hardness can mitigate the plant and aquatic toxicity of many leached metals. Annual patterns of hardness concentrations reflected that Ca and Mg in the leachates were at their highest levels in early winter and then declined (Figure 4). Annual average hardness concentrations in the untreated soils ranged from 151 to 306 mg $CaCO_3$/L with large spatial variation (Table 9). Concentrations in the IMG treatments were also highly variable, but nevertheless more than four times greater than the untreated soils in the 20 t/ha treatment and more than 10 times greater in the 60 t/ha treatment (Table 9). Average annual concentrations weighted by leachate volumes from each lysimeter were more than double that of the unweighted average concentrations because of the particularly high concentrations and volumes leached in early winter (see Table A15 in Appendix D).

Concentrations of all trace elements except Co were generally similar or slightly less in leachates from untreated soils than the IMG-treated soils. There was a small but significant ($P<0.05$) increase in annual average cobalt concentrations in leachates in the 60 t/ha IMG-treated soils in the first year, of less than 19 µg/L (Table 10), with maximum volume-weighted concentrations for each site being less than 74 µg/L. This effect was not evident in the second year. In both years, Co concentrations in leachates were highly correlated with leached Mn concentrations ($r >84$; $P<0.001$). For other metals, however, there were slight reductions in concentrations of metals such as Bi, Cd, Cr, Cu, Th and U leached from IMG-treated soils when compared with tilled soils with no IMG (Table 10). Elements such as As, Fe, Pb and Zn leached in similar concentrations from untreated and IMG-treated soils.

Table 9: Annual average total Mn, SO_4 and hardness concentrations in leachates from the bleached sand (soil 2).

Treatment	**Total Mn (as mg/L)**				**SO_4 (as mg/L)**				**Hardness (as mg $CaCO_3$/L)**			
	2018		**2019**		**2018**		**2019**		**2018**		**2019**	
No IMG (untilled)	0.14	±0.08[a]	0.39	±0.38	94	±101	515	±660	232	±231	794	±882
No IMG (tilled)	0.19	±0.06	0.19	±0.18	81	±83	114	±225	151	±85	306	±531
20 t/ha	8.4	±5.0	12.1	±6.2	782	±271	804	±510	949	±362	1173	±788
60 t/ha	18.3	±16.7	12.2	±5.2	1535	±215	1409	±300	1790	±472	1727	±645

[a] Standard deviation

Table 10: Annual average total metal concentrations (µg/L) in leachates from the bleached sand (soil 2).

Treatment	**Year**	**As**	**Bi**	**Cd**	**Cr**	**Co**	**Cu**	**Fe**	**Pb**	**Ni**	**Th**	**U**	**Zn**
No IMG (untilled)	2018	0.4	0.04	0.5	4.7	4.2	353	883	20	531	0.1	0.4	991
	2019	47.2	0.03	2.2	5.0	9.5	251	884	18	64	21.1	0.6	2597
No IMG (tilled)	2018	0.5	0.04	0.6	12.9	5.8	274	2371	23	489	0.1	0.8	724
	2019	9.4	0.03	1.3	9.8	9.0	268	1783	12	25	82.1	1.0	838
20 t/ha	2018	0.3	0.04	0.2	1.5	15.3	5	1636	0	8	0.1	0.3	39
	2019	14.7	0.03	0.4	2.2	18.9	20	2145	2	14	6.1	0.5	315
60 t/ha	2018	0.3	0.04	0.3	1.5	23.6	62	789	9	476	0.1	1.2	2282
	2019	19.4	0.03	0.7	1.2	11.9	128	565	13	66	3.2	0.9	3477

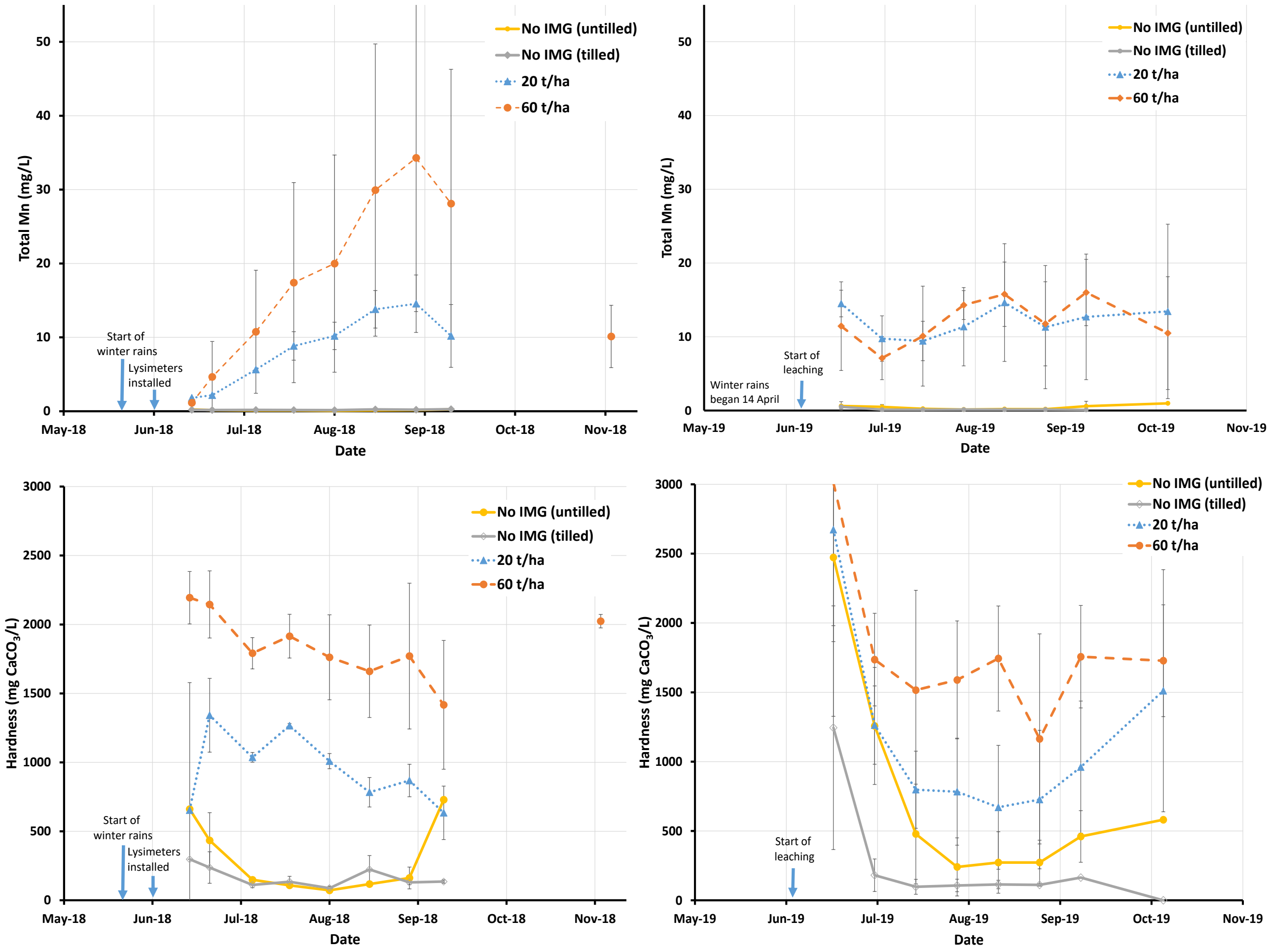

Figure 4: Total manganese (Mn) and hardness in leachates from untreated and IMG-treated soils over two seasons.

3.6 Leachate fluxes

Nutrient fluxes estimated from the lysimeters indicated a reduction in soluble P loss of up to 24 kg/ha/yr (more than 97% reduction) but only a slight effect on N loss. There were large variations in P losses across the experiment site: this was mostly due to variation in P concentrations more than leachate volumes (ranging from 220 to 307 mm in the first year and 64 to 160 mm in the second). Annual soluble P leached from untreated topsoils ranged from 4 to 43 kg P/ha/yr, with losses from the tilled soil being almost double that of the untilled soil in the first year (Table 11; Figure 5). By contrast, 0.3 to 2 kg/ha/yr of soluble P was leached from the IMG-treated soils with no difference between the rates of IMG applied (Table 11). Leaching of other forms of P (colloidal and organic) showed a similar effect of tillage in the untreated soils, although the loss was less than 5 kg P/ha/yr. These were reduced to less than 2 kg P/ha/yr in the IMG treatments.

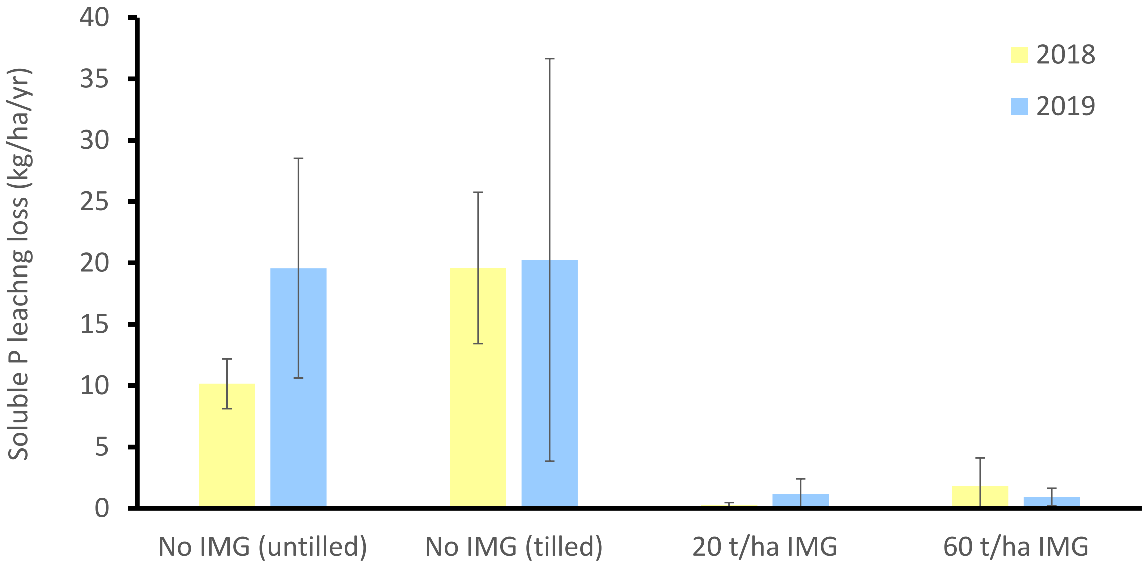

Figure 5: Average P loss by leaching from the topsoil of soils with and without IMG treatment.

Table 11: Summary P and N leachate fluxes (kg/ha/yr) estimated for the bleached sand (soil 2) using lysimeters.

Treatment	**Soluble P (FRP)**				**Colloidal and organic P[1]**				**Total N**				**DON**			
	2018		**2019**		**2018**		**2019**		**2018**		**2019**		**2018**		**2019**	
No IMG (untilled)	10.1b	±2.0[2]	19.6a	±11.6	2.0b	±1.8	2.3a	±0.9	82a	±46	244a	±27	34a	±12	42a	±9
No IMG (tilled)	19.6a	±6.2	20.3a	±6.3	4.9a	±0.3	2.4a	±0.9	128a	±52	125a	±41	41a	±0.3	29ab	±3
20 t/ha	0.3c	±0.2	1.1b	±8.8	0.5b	±0.0	1.4a	±1.1	80a	±24	145a	±85	15b	±3.1	14b	±1
60 t/ha	1.8c	±2.3	0.9b	±5.1	0.9b	±0.3	0.7a	±0.1	120a	±29	213a	±86	20b	±4.4	9b	±1

1 Colloidal and organic P calculated as TP – FRP.

2 Standard deviation of replicate lysimeters across replicate plots

Table 12: Summary oxidised N (NO_x), ammonia (NH_4), potassium (K) and magnesium (Mg) leachate fluxes (kg/ha/yr) estimated for the bleached sand (soil 2) using lysimeters.

Treatment	**NO_x**				**NH_4**				**K**				**Mg**			
	2018		**2019**		**2018**		**2019**		**2018**		**2019**		**2018**		**2019**	
No IMG (untilled)	29	±14[2]	183	±45	14.9	±11.8	14.0	±12.3	61a[1]	±36	245a	±259	57a	±37	169a	±35
No IMG (tilled)	16	±13	90	±39	4.2	±2.6	5.0	±2.8	105a	±21	158a	±24	42a	±15	81a	±63
20 t/ha	30	±18	113	±72	14.6	±10.3	10.4	±11.5	43a	±14	274a	±179	96a	±17	220a	±166
60 t/ha	58	±39	193	±87	39.7	±8.4	11.9	±8.4	367b	±87	482a	±107	315b	±31	238a	±96

[1] Where determined, values followed by the same letter in each year are not statistically different between treatments ($P<0.05$).

[2] Standard deviation of replicate lysimeters across replicate plots.

N loss by leaching ranged from 49 to 270 kg/ha/yr, being mostly dominated by nitrate (see Table 12). IMG treatment reduced losses of dissolved organic N, but had no effect on total N (Table 11) or losses of NH_4 and NO_3 (Table 12). Leaching of dissolved organic N from the IMG-treated soils was almost half that of the untreated tilled soils (>40 kg N/ha/yr) in the first year. This effect increased in the second year for the 60 t/ha treatment but not the 20 t/ha treatment.

Rates of K and Mg leaching were highly variable across the experiment site and between years, with slight effects from the IMG treatment. On average, K leached at 83 to 201 kg/ha/yr and Mg leached at 49 to 125 kg/ha/yr from the untreated soils, with no significant effect from tillage (see Table 12). Leaching from the 20 t/ha treatment was similar to the untilled soils without IMG in both years, but was more than double that observed in the 60 t/ha treatment for both elements in the first year. This difference was not evident in the second year due to much greater variation being detected with the additional monitoring sites in the untreated soils.

Leaching fluxes of gypsum (i.e. $CaSO_4$) were greater from soils with the higher rate of IMG treatment. An estimated 10 to 15 tonnes of gypsum/ha (as $CaSO_4$) was leached from the 60 t/ha IMG treatment over the two years, whereas 6 to 10 tonnes of $CaSO_4$/ha was leached from the 20 t/ha treatment. These losses represented 33 to 55% of the $CaSO_4$ initially applied in the 60t/ha IMG treatment (approx. 28 t/ha) but 67 to 100% of that applied in the 20 t/ha treatment (approx. 9.3 t/ha). In comparison, an estimated 71 to 133 kg/ha of manganese was leached from the IMG-treated soils over the two years (with similar rates for both IMG treatments), which is about 5 to 14% of the initial Mn applied in the IMG.

3.7 Leached nutrient concentrations in relation to soil properties

Average winter–spring concentrations of leached soluble P were broadly predictable from some soil properties each preceding summer. Leached P concentrations (weighted by leached volumes) were most strongly correlated with $CaCl_2$-P ($r = 0.96$; $P<0.001$): concentrations were approximately double that of soil concentrations (Figure 6). This was surprising given soil concentrations can vary depending on wetting events leading up to summer soil sampling. In summer 2020, $CaCl_2$-P increased by 0.7 to 6.6 mg/kg following a 50 mm summer rainfall event (unpublished data), although the relative differences between the soils remained similar. However, a single predictive function for any soil does not appear likely. Data from another similar experiment with IMG treatment of sandy soils at Keysbrook is presented below as a comparison. In that experiment the relationship between summer soil $CaCl_2$-P and average P concentrations in winter–spring leachates was significantly flatter than in the Yoongarillup experiment (Figure 6).

Leached P concentrations were less strongly, but still significantly correlated with PERI (Figure 7; $r = 0.77$ $P = 0.003$), but not significantly linearly correlated with Colwell P, PRI or PBI ($r<0.45$; $P>0.07$).

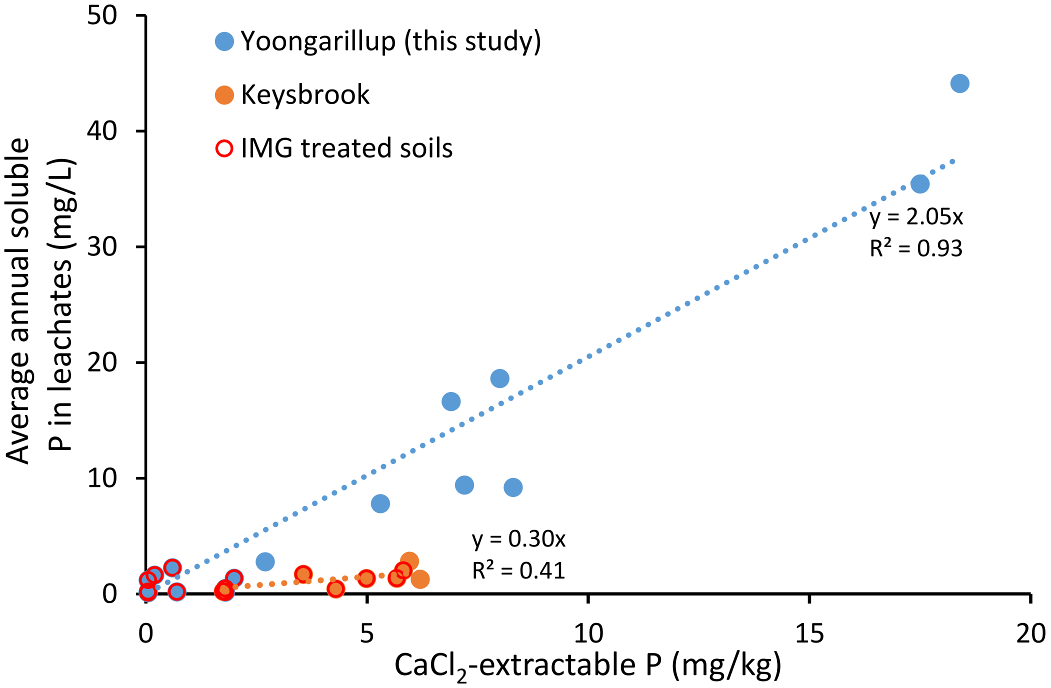

Figure 6: Average soluble P in leachates during winter–spring in relation to the previous summer soil-water P (as $CaCl_2$-extractable P) at two experiment sites with IMG treatments.

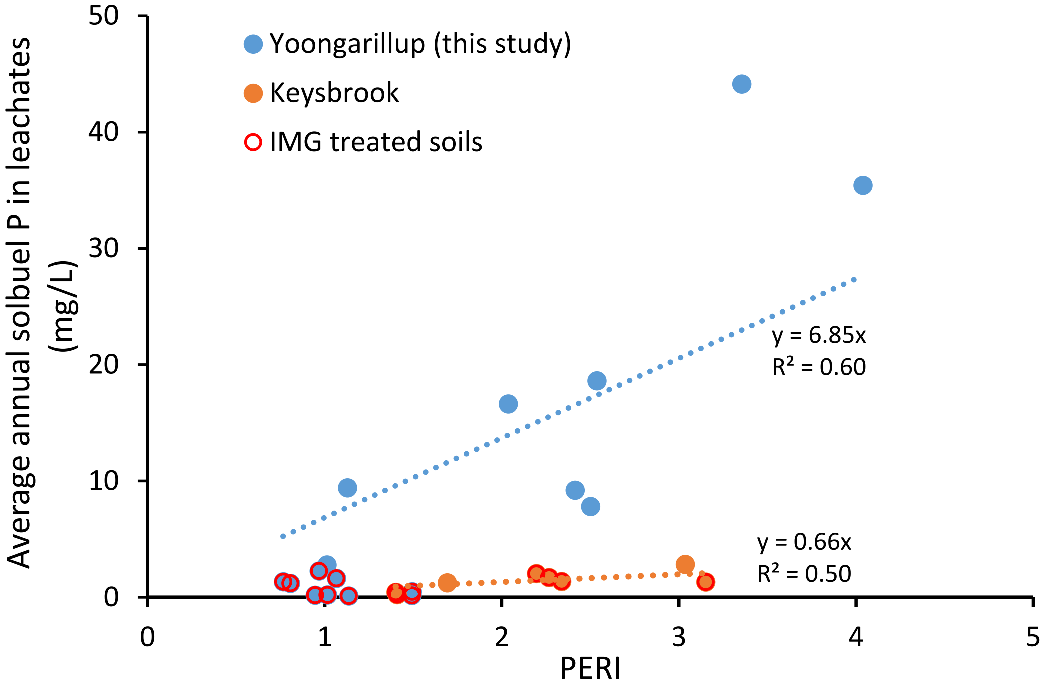

Figure 7: Average soluble P in leachates during winter–spring in relation to the previous summer PERI at two experiment sites with IMG treatments.

4 Discussion

4.1 Increased P retention and reduced P losses with IMG treatment

Mixing IMG into pasture soils saturated with high concentrations of plant-available P immediately limited the mobility of P from the soils. Most of the P lost from the soils by leaching was in a soluble form and the IMG treatments reduced this by more than 90% to <0.35 mg/L. This effect was due to IMG reducing the P in soil-water, indicated by $CaCl_2$-P, and was similar for the two rates of incorporation (20 t/ha and 60 t/ha).

Soluble P losses from soils with the greatest risk of P loss were reduced by an average of 24 kg/ha/yr with IMG treatment. These high-P-loss soils were the mid-slope bleached grey sands (soil 2) and represented 30% of the experiment area. Localised P losses ranging from 10 to 25 kg P/ha/yr (at each lysimeter) were reduced to less than 2 kg/ha/yr with IMG treatment. Although not measured, P leaching loss from the other soil types in the experiment area were likely to be much less because of the initially smaller $CaCl_2$-P concentrations and typically greater inherent P retention of these soils. The lifespan of the IMG in soil 2 is expected to be more than 10 years for the 20 t/ha application rate based on the annual rates of reduced P leaching and the maximum adsorption capacity of IMG of 21 kg P/tonne (Wendling et al. 2012). Further investigations are required to determine how concentrations of P in leachate change as the maximum adsorption capacity is approached, noting P adsorption isotherms for aged IMG in soils have not been published.

Topsoil P leaching represents loss of the nutrient from the production system to shallow groundwater, but gross export of P to local waterways is likely influenced by other soil and hydrological factors. Many factors influence the relationship between soil P losses locally and P lost at paddock or river-reach scales, with variability in source areas and transport pathways from these areas having a major role (Dougherty et al. 2004). At the experiment site, P loss from the untreated soils to the groundwater is large, yet P transport by shallow groundwater to the waterway crossing the site would be limited and slow. Factors such as the adsorption of P in higher PRI and PBI subsurface horizons, the comparatively slow rate of groundwater flow (likely to be < 0.2 m/day), and thinness of the shallow aquifer (<1.5 m) would limit the amounts of P transported as groundwater discharge to the waterway each year. The greatest effect of groundwater at the site would be to increase P transport as the water seeps through the P-saturated topsoils – effectively reverse leaching – to become surface runoff (Figure 8). This was frequently observed in the lower slopes of soil 2 (near the soil 1 boundary) for days after each rain event in winter with P concentrations probably like that of the leachates. The most direct and rapid pathway of loss from the experiment site, however, was likely through surface runoff. This was common from soil 1, which was persistently waterlogged in winter, and probably frequent across most of soil 2 after intense winter rainfall events (>10 mm/hour). Such events generated runoff by exceeding infiltration rates in soil 2, with water observed to pond rapidly on the surface despite the fine sand soils being unsaturated. The slow surface-soil infiltration rates were probably due to compaction and reduced topsoil porosity that often happens with periodic intensive grazing (McDowell et al. 2003).

The effect of IMG on leached P was primarily through a reduction in the P in soil-water (indicated as $CaCl_2$-P) with no measurable effects on the larger plant-available P (indicated by Colwell P). Every mg/kg less $CaCl_2$-P measured in the summer soils near the lysimeters broadly resulted in average leachate P concentrations being 2 mg/L lower the following winter. This type of relationship has been found in other soils where 0.01 M $CaCl_2$-P after leaching was a strong predictor of P concentrations in leachates (McDowell & Sharpley 2001; Hesketh & Brookes 2000). In the field, however, the predictive capacity of summer $CaCl_2$-P for winter leaching concentrations is likely to be influenced by within-season P cycling and inputs. Our experiment found evidence that $CaCl_2$-P varied seasonally in the untreated soils, particularly during winter where there was little to no P adsorption capacity. This aligns with a study that investigated sandy soils in the Peel (Weaver et al. 1988) where variation in soil-water P was driven by a combination of P mineralisation from organic matter and desorption from soil surfaces. Inputs of manure during each winter on intensively grazed paddocks (and mineralisation of P from this) would also contribute to variation in the soil-water P in soils. Annual summer monitoring showed variation of up to 8 mg/kg in $CaCl_2$-P across lysimeters from year to year, which further indicates significant annual variation in localised P inputs and hence leaching each winter thereafter. This is also made clear by the leaching relationship in this experiment being different to that measured at another site in the Peel.

Runoff P is also likely to have been reduced where IMG reduced the soil-water P in soils. Such effects on runoff P can be extrapolated from another experiment where IMG treatment of sandy soils in the Peel region found that amounts of soil-water P (measured as $CaCl_2$-P) directly influenced soluble P concentrations in runoff (Degens et al. unpublished). On the basis of this, the sustained more than 10-fold reduction in soluble P in soil 2 and soil 3 with IMG treatment is also likely to result in at least a 3-fold reduction in soluble P in any runoff. Most of these effects are probably limited to soil 2, which was the soil most likely to generate runoff during periods of intense rainfall (as discussed above). In contrast, the effect of the IMG on any runoff from soil 1 would be minimal because the untreated soils have inherently very low $CaCl_2$-P (<1.2 mg/kg). P loss from these soils is most likely as particulate P, which tends to be greatest after grazing when the soils are waterlogged. Cattle treading of waterlogged soils can disrupt the surface and increase surface transport of P as particulate matter (McDowell et al. 2003). Extended periods of waterlogging can also lead to increases in P solubility if sufficiently low redox conditions develop (Smith et al. 2021).

P adsorption properties of the whole plots were slightly increased with the IMG treatments but did not differ between the rates. PBI was increased by 14 to 17 with both IMG amendment rates whereas PRI was increased by 7 to 24. These were minor increases in P adsorption compared with the natural PBI range of <1 to >100 of soils in south-west Western Australia (Bolland & Windsor 2007) or the natural range of 16 to 103 in untreated soils across the paddock. Previous investigations reported smaller increases in PRI of less than 7 mL/g two years after mixing in 150 t/ha of IMG, despite an initial increase of up to 20 mL/g (Douglas et al. 2010a). This suggests there is some variability in how the conventional indicators of P adsorption respond to increases in P adsorption capacity following IMG treatment. It may also explain an initial effect of the IMG rates on P adsorption properties at the lysimeters that had diminished by the following summer.

The absence of a rate effect of IMG above 20 t/ha on P adsorption was attributed to variation across the site and the effects of high soluble Ca and SO_4. The latter may also explain the initially high PBI and PRI of the soils at the lysimeters in the 60 t/ha treatment that decreased by the first summer. High Ca concentrations in soil solutions during P adsorption measurements can result in precipitation of P as calcium phosphates (Allen & Jeffery 1990), thereby increasing the apparent PBI and PRI of the IMG-amended soils. Likewise, increases in the ionic strength during measurement can also increase P adsorption during measurement (Barrow 1984). This is likely with high Ca and SO_4 concentrations remaining in the soils or if residual gypsum dissolves during measurement. Both effects are probably variable across each treatment, depending on the gypsum remaining in the soils, and may have masked any increase in PBI with the higher rate of IMG treatment. Background variation in the PBI of soil 1 also masked the increased IMG treatment rate, with the PBI of the untilled soils typically ranging from less than 40 to more than 90 across the plots. This variation is attributed to seasonal waterlogging which affects the distribution of iron minerals in the soils, with iron precipitates frequently observed to precipitate along root channels in the soil.

Patterns of P fertility in the paddock were likely to have been dominated by the uneven return of P in manure by grazing cattle and will spatially determine the longevity of reduced P leaching with IMG treatment. Increased P retention in the IMG-treated soils would be expected to lead to a gradual increase in P storage (as Colwell P) compared with the untreated soils, where P would otherwise be lost by leaching or runoff. However, significant spatial variation in Colwell P across the site limited detection of any changes over the first two years and high manure-P return to the soils is probably the main reason for this. Fertiliser-P inputs were 6 kg P/ha/yr whereas manure-P inputs were estimated to return 16 to 103 kg P/ha/yr relative to 60 to 80 kg P/ha/yr removed in grass grazing (Appendix F). However, while grass grazing and fertiliser input is typically relatively even, manure return is likely to be highly uneven. With the paddock being progressively grazed in strips from the south (near the entry gate) to the north, ruminating cattle (i.e. not grazing) were observed to frequently congregate across the paddock even after grazing strips at the northern end. This effect was further exacerbated with summer feeding at the southern end of the paddock. While the resulting average P balance can range from net removal of 64 kg P/ha/yr to net accumulation of 43 kg P/ha/yr (Appendix F), within-paddock patterns of P fertility, and thus P leaching, will depend on the distribution of manure inputs.

The large annual inputs of manure P are also likely to accumulate available P in the surface soil layers with concomitant increases in P lost in runoff. Periodic power harrowing to mix the soil should minimise this risk, particularly where residual adsorption capacity remains in the topsoil beneath the surface few centimetres.

4.2 Effects of soil inversion on P retention and availability

Tillage influenced soil-water P in the soils but not P adsorption capacity, with the strength of the effect depending on the soil type. Inverting the soil by mouldboard ploughing more than halved $CaCl_2$-P in the topsoil of soil 3 with the effect persisting over two years, but had a lesser effect in the other soils. Most of the initial effect of tillage was probably due to burial of some of the high-P topsoil (which in the untilled soils remained at surface). However, for soil 3 the effect was also likely mediated by increases in PBI due to ploughing, bringing high-PBI subsoil materials to the surface. The subsoils beneath most of soil 3 were typically yellow and yellowish-red mottled sands (with some clayey sands) with a PBI >60 at 10–20 cm. This increased to a PBI of up to 300 in some deeper horizons (>30 cm) that were beneath most of the site.

Inverting and tilling soils with high-PBI subsoils may reduce P losses from soils with high P_{95} fertility without the need to apply soil amendments. Delving, spading and deep ploughing of soils used for cropping are practices used elsewhere in Western Australia to bring clays at depth to the surface with multiple benefits, including increased nutrient retention (Davenport et al. 2011). Further investigation is needed to assess if these practices could be applied to certain soils on the Swan coastal plain with high-PBI subsoils as an alternative practical measure to reduce P loss from high-risk soils. However, the availability of trace elements and other nutrients needs to be considered in using this practice as inversion of soils or mixing of subsoils can induce deficiencies in trace elements (e.g. Cu and Zn) in pastures, as was found for Cu in this experiment. This can be easily managed by applying trace elements.

4.3 Effect of IMG on loss of other forms of N and P

IMG treatment had the additional benefit of increasing retention of dissolved organic P and N by more than 60%. This effect was non-specific and apparent for all dissolved organic compounds, with DOC showing similar increased retention to that of organic N and P. DON retention by IMG has been found previously (Douglas et al. 2012; Wendling et al. 2012; Degens & Shackleton 2016) and is usually limited to a few years. Organic compounds may be retained by the adsorbing surfaces of iron oxides in IMG that would be progressively exposed as $CaSO_4$ dissolves, along with an effect caused by the increased Ca following IMG application. Adsorption of DOC from stream water has been correlated with the leaching of Ca, and appears to reduce once most of the Ca has been leached (Wendling et al. 2012). Similarly, more DOC is retained with increasing Ca concentrations, possibly because of increased cation bridging with reactive surfaces in soils (Kerr & Eimers 2012).

IMG does not reduce leaching losses of inorganic N (NO_3 and NH_4), which were up to 200 kg/ha/yr. Several previous investigations report retention of inorganic N by IMG but these contain some uncertainties. Retention of inorganic N in IMG beneath turf (Douglas et al. 2012) was not supported by consistent reductions in concentrations of NO_3 or NH_4 and relied on reductions in leachate volumes without analysis of the soil water-balance. Similarly, reduction of NO_3 and NH_4 concentrations in stream water flowing through IMG (Wendling et al. 2012) may have been due to effects of the material on N mineralisation and N transformations in the column rather than net N retention. In our investigation, the timing of most N loss as NO_3 in early winter reflected leaching of soil NO_3 that had accumulated from organic matter mineralisation and manure/urine inputs over summer and autumn.

4.4 Water quality risks of IMG treatment

IMG treatments resulted in large increases in Ca, SO_4 and Mn leaching that were not rate dependent. SO_4 concentrations increased to an average of 1,535 mg/L but were measured in similar concentrations in untreated soils. Up to 15 tonnes of gypsum/ha leached to groundwater from the IMG treatments in first two years, and it is likely that some Ca and SO_4 was also leached to surface runoff. This was common during periods of intense rainfall (when, as discussed previously, infiltration may be limited by compaction of the soil). Similar rates of leaching in each treatment indicated that rates of loss were limited by the rate of gypsum dissolution in the IMG rather than the amount of IMG mixed into the soils. The concentrations of Ca and SO_4 in leachates are expected to decrease with time as the $CaSO_4$ is depleted in the topsoil. Given initial rates of leaching, most of the gypsum is likely to have leached three to four years after the 20 t/ha treatment and four to six years after the 60 t/ha treatment. This can be considered an indicative period of risk for SO_4 leaching following IMG treatment under local rainfall conditions. In contrast, other reports indicate that Ca and SO_4 concentrations were slow to decrease over five years where IMG was applied to the surface of soil with no mixing (Sharma et al. 2018).

The high concentrations of Mn leached from the IMG-treated areas has not been found in other IMG experiments and are likely due to the lower pH of the soil. Average concentrations of Mn of up to 18 mg/L are an order of magnitude greater than the <0.5 mg/L reported for experiments with IMG mixed into soil at 150 t/ha (Douglas et al. 2012) or surface-applied IMG at up to 50 t/ha (Sharma et al. 2018). The contrasting results are attributed to the Yoongi Downs soil being more acidic (pH_{CaCl2} 4.7–5.4) than where IMG was mixed with the turf soil (pH_{CaCl2} 5.9 to 6.5) (Douglas et al. 2012). Where IMG was top-dressed (Sharma et al. 2018), the soil pH_{CaCl2} was 4.2 to 4.8, but contact between the IMG and the soil was limited (livestock was excluded and IMG remained on the surface). The mineralogy of the Mn in IMG is unknown (Sharma et al. 2019; Douglas et al. 2012) and could not be resolved by XRD analysis (Degens 2020, unpublished data). However, Mn is likely to be present as either a hydroxide or carbonate salt, therefore solubility would increase when IMG is mixed into acidic soils. This inferred mineralogy is based on tests showing that Mn in IMG is more soluble (>2% by mass) in batch pH extracts less than pH 6.5 (Sharma et al. 2019) and because the material is a product of $Ca(OH)_2$ neutralisation of an oxidised sulfuric acid leachate. There was a positive correlation between Mn and alkalinity in leachates in the first year, but not the second, which indicates initial leaching might be mediated by dissolution reactions in the soil.

Mn leached in very hard waters (average > 949 mg $CaCO_3$/L) during the first two years of this experiment. As increasing hardness reduces the toxicity of Mn to aquatic organisms (Peters et al. 2011), this mitigates potential aquatic toxicity of Mn discharging to waterways from shallow groundwater or surface runoff. The average concentration of up to 17.2 mg/L is much greater than the surface water quality guideline limit of 1.9 mg/L for a 99% probability of there being no impacts on aquatic organisms for waters with hardness < 50 mg $CaCO_3$/L (ANZECC & ARMCANZ 2000). If the leachates were conservatively diluted 10-fold naturally by rainfall or streamflow (with no hardness) the average concentration would be 1.7 mg Mn/L with a hardness of at least 100 mg $CaCO_3$/L. This indicates the Mn concentrations pose no risks to aquatic organisms, should the waters discharge directly to streams undiluted. Preliminary results of aquatic toxicity testing using south-west waters with 100 mg $CaCO_3$/L also indicate there is no risk of toxicity with Mn being non-toxic at concentrations below 4 mg/L (Nice & Degens, unpublished data).

Concentrations of all trace elements except Co were reduced in the leachates from soils treated with IMG. The maximum estimated flux of Co leached from the lysimeters over the two years (14 mg/m^2) represented less than 1% of the Co applied with the IMG (Appendix E), which suggests most Co is retained in the amended soils – although the mechanisms controlling the leaching require further investigation. Concentrations of Co in leachate of up to 19 µg/L from IMG-treated soils are much greater than the < 5 µg/L previously reported in an experiment with IMG applied at 150 t/ha (Douglas et al. 2012), but similar to leachates from IMG top-dressed at up to 50 t/ha (Sharma et al. 2019). The correlation of Co with Mn leaching suggests linked mechanisms controlling mobility. Co may be co-precipitated with Mn minerals – of which the reactivity with the soil solution depends on dissolution of the gypsum matrix that probably encases these minerals.

The toxicity risk posed by the increased Co concentrations – should leachates reach streams – is likely to be minimal. As with Mn, acute toxicity may be mitigated by hardness (Diamond et al. 1992; Pourkhabbaz et al. 2011), although this has been considered to have no effect on chronic toxicity (Nagpal 2004). Leachate concentrations were up to 19 µg/L from the IMG-treated soils and exceeded low-reliability trigger limits for 90% protection levels of 2.8 µg/L, but were less than a moderate reliability trigger of 90 µg/L (ANZECC & ARMCANZ 2000). Further investigation is required to verify whether Co concentrations remain high over multiple years and if this presents a potential risk to aquatic ecosystems where shallow groundwater discharges to drains in treated paddocks.

4.5 Agronomic benefits and risks

The absence of any effects of IMG treatment on plant production indicated the improvements in soil properties are not critically important for plant growth in the short-term and that other factors are more limiting (such as waterlogging). The reduction in $CaCl_2$-P with IMG treatment had no influence on plant P uptake or plant growth, with much of the site still containing excessive concentrations of available P following treatment. Leaf P concentrations were consistently >0.35% and averaged

0.5% across the experiment, indicating that P remained non-limiting for plant growth. There was no observable difference in plant growth which corresponded to a P_{95} fertility index >1 across most of the experiment site (indicating P was non-limiting). Where the P_{95} fertility index was less than 1 and as low as 0.5 in the waterlogged soil (soil 1), P availability was still more than sufficient for plant growth with adequate to high (>0.5% P) concentrations of P in leaf tissue. Extended periods of waterlogging possibly constituted the most limiting factor to plant growth in this soil.

Average P concentrations in leachates from the IMG-treated soils dipped below levels considered optimal for grasses but did not result in reduced plant P uptake. Concentrations in leachates frequently reduced during each growing season and by late spring were often less than 0.02 mg/L in the IMG treatments. This is equivalent to a solution P of <0.04 mgL (using the relationship derived by Moody 2011) which is below 0.15 mg/L; considered optimal for most grasses derived from solution cultures (Asher & Loneragan 1967). However, while plant-leaf P concentrations were slightly reduced in the IMG treatments, these remained in a range adequate for maximum growth. This indicates that the grasses on the experiment site were most likely accessing soil P in pore-waters that were not reflected by concentrations in leachates during wet periods. It is also likely that solution-culture-derived estimates of adequate pore-water P concentrations do not account for additional mechanisms of plant P uptake from soils, such as mycorrhizal fungi and chelating exudates.

IMG treatment had additional positive effects on soils other than improved P retention. Soil pH was increased by 0.3 to 0.5 units at both treatment rates with a slight increase in cation exchange capacity in the 60 t/ha treatment. The increase in pH is attributed to the weak liming effect of IMG which has an equivalent neutralising capacity equivalent to less than 6.6 % pure $CaCO_3$ (Appendix I). The 60 t/ha of IMG treatment was equivalent to applying 2.8 t/ha of pure limesand (with 100% neutralising value), whereas incorporating 20 t/ha was similar to adding 0.9 t/ha of pure limesand. The increases were smaller than expected and might be due to a gradual neutralising effect as the gypsum dissolves to expose what is probably minor carbonate minerals. Mixing the IMG in with the Aglime (applied after the IMG was spread) may have also increased the effectiveness of this applied lime, with Ca from the gypsum increasing the dissolution of the limesand (McLay et al. 1994). The initial benefits of the increase in soil pH_{CaCl2} from approximately 5 to 5.5 were estimated to be equivalent to applying about 1 t Aglime/ha given the organic C > 3% (Moore et al. 2001). The increase in CEC is minor and not expected to have any significant effect on the behaviour of retained cations such as K, Ca and Mg. Other experiments with IMG report slight increases in soil organic C over four years after mixing at 150 t/ha into soil under turf production (Douglas et al. 2010a). However, it was unclear whether the site had uniform organic C at the beginning of this experiment.

There was little increase in plant uptake of Mn from the IMG-treated soils despite greater concentrations leaching through the root zone. Phytotoxic concentrations of Mn in solution culture are generally > 0.5 mg/L with Ca concentrations of about 10 to 20 mg/L (Millaleo et al. 2010; Kopittke et al. 2010). The much greater Ca in the soil water indicated by leachate concentrations of >230 mg Ca/L probably competitively inhibited Mn uptake, as it is common for increasing Ca to reduce Mn toxicity in low pH soils (Horst 1988). Further work is needed to determine whether Mn remains in IMG-treated soils after many years of leaching and if there is sufficient Ca to mitigate any potentially toxic effects on plants. The initial results of the experiment indicate that Ca is leaching faster than Mn with more than 67% of the applied $CaSO_4$ having leached in the 20 t/ha treatment during the first two years compared with less than 14% of the Mn.

High rates of Ca leaching in the IMG-treated soils increased the concentrations of leached K and Mg, but these effects were largely confined to early winter. These effects were probably due to Ca displacement of K and Mg from exchangeable cation storage in the soil. Initially high Ca and SO_4 concentrations in the early winter leaching were most likely due to high concentrations accumulating with extended dissolution of $CaSO_4$ following summer rainfall events and as the soils wet up in autumn. The similar concentrations of Ca leaching contributed to similar rates of K and Mg leaching per hectare with both rates of IMG. Annual summer soil analyses also indicated the loss of Mg by leaching with lower exchangeable Mg but plant available K (as Colwell K) remained similar. This was most likely because of regular K applications during each growing season. Plant-tissue concentrations of Mg and K also remained sufficient over the two years.

4.6 Cost and benefits

Prospects for the adoption of IMG as a soil amendment to reduce P losses will depend on the financial cost-benefit ratio as well as environmental risks (costs) and benefits. Financially, the costs of IMG treatment hinge on supply, transport and application costs that need to be considered in relation to the benefits of improved P retention and the longevity of this. As discussed above, while IMG treatment can improve several soil properties, including P retention, these may not result in increases in plant production because the soil properties were not factors limiting plant growth, at least in the short-term.

Current costs to apply IMG in the Vasse region when compared with the value of the P retained is not likely to be immediately attractive to farmers from a business perspective. The financial benefit of retaining P on the farm, when projected over 10 years, ranges from $206 for paddocks losing 5 kg P/ha/yr to $826 per ha for paddocks losing 20 kg P/ha (Table 13). Except for soils with large P losses, the cost of IMG application for the simplest amendment regime (20 t/ha top-dressed) is typically more than double the benefit of P retention (Table 13). These estimates assume that IMG is effective for 10 years. The benefits marginally increase to a maximum of $150/ha if IMG is effective out to 15 years.

The longevity of the IMG benefits is unclear and remains a significant uncertainty for costing benefits. As discussed in Section 4.1, the laboratory-measured maximum adsorption capacity of IMG might be 21 kg P/tonne which for 20 t/ha carries a maximum P adsorption of 410 kg P/ha. However, this capacity is when P would begin to leach at concentrations like unamended soils. Use of a 10-year lifespan assumes a reasonable overall P retention of between 50 and 200 kg P/ha, in scenarios where there are on-going annual losses of between 5 to 20 kg P/ha. Investigations in the Peel show the longevity of reduced runoff losses with top-dressed IMG may be limited in areas of paddocks receiving with higher annual P loads (Degens et al. unpublished).

Reducing the costs of IMG application by up to $600/tonne is necessary for IMG to be immediately economically attractive to many landholders with average losses of < 10 kgP/ha. This is the expected loss from soils on many farms. Application costs include purchasing, transporting and spreading IMG: these totalled $780/ha for IMG top-dressed at 20 t/ha for the experimental site, but could be up to $900/ha if tillage costs are included. The costs are naturally more than three times this where IMG is applied at 60 t/ha. These calculations assumed an IMG price of $20/tonne, which is conservatively below the current price of $50/tonne for small loads from Iluka Resources in Capel. Transport comprises most of the cost, which is $13 to $16/tonne from Capel, depending on the carrier. Spreading costs are based on contractor rates of up to $9/tonne, including mobilisation and loading. Subsidising application costs by removing supply and freight costs (leaving only on-farm spreading costs) brings the costs into balance with the benefits. Where greater losses of P from soils occur there is less need to reduce costs; however, realising the benefits of the saved P in terms of greater pasture production rates may determine the use of IMG in these situations.

Reducing P in waterways by improving P retention on farms is likely to be significantly more cost-effective than off-farm treatment of P in waterways. IMG offers an immediate at-source reduction of P losses from farms with the costs of intervention by way of subsidy or incentive ranging from $3.00/kg P to $12.00/kg P retained (Table 13). In comparison, the costs of effective in-stream treatment for P removal are estimated to range from $50 to more than $1,000 for every kg of P removed. The upper estimate is the minimum cost estimate for treating water using high-P-adsorbing clays. There is significant uncertainty around the lower cost estimate, being in-drain treatment, which depends on the longevity of P removal of any P-adsorbing media. Many in-drain treatments are typically smothered or become clogged and are not effective after a few years. These costs indicate that funding on-farm amendment costs strategically focused on high-P-loss areas is cheaper than treating P in drains and waterways.

Table 13: Estimated costs and benefits of improving on-farm retention of P.

Scenario	Annual soil P loss via leaching and/or runoff (kg P/ha)	Benefits of retained P in soils over 10 years ($/ha)[1]	IMG treatment cost ($/ha)	Cost:benefit ratio	Cost of reduced P loss based on subsidy ($/kg)
IMG applied at 20 t/ha, no subsidy	5	$ 206	$ 780[a]	3.8	
	10	$ 413	$ 780	1.9	
	20	$ 826	$ 780	0.9	
IMG applied at 20 t/ha with no supply cost	5	$ 206	$ 180[b]	0.9	$ 12.00
	10	$ 413	$ 180	0.4	$ 6.00
	20	$ 826	$ 180	0.2	$ 3.00

[1] Present value of future saved P in the year that the IMG is applied, estimated using an annual discount rate of 2% (i.e. equivalent return on the money if invested elsewhere) with a 10-year return period with a P price of $4.60/kg (estimated from the value of P in single superphosphate priced at $424/tonne including GST in 2020).

[a] Treatment cost includes purchase, freight from Capel to the mid-part of the Peel region and spreading.

[b] Treatment cost for spreading only.

5 Concluding comments

Mixing IMG into soils at rates of 20 t/ha or 60 t/ha was immediately effective in preventing losses of up to 24 kg P/ha/yr from soils under intensive dairy grazing with no effects on pasture production. IMG slightly increased the adsorption capacity of the soils, reducing the amounts of P in soil-water that could be leached (measured as $CaCl_2$-P). The net effect of these soil changes was a greater than 90% reduction in P concentrations in leaches. These effects were achieved while maintaining the P available for plant uptake in the soils and with no effects on plant growth. The influence of IMG on P losses via different pathways from the paddock is conceptualised in Figure 8. While both rates of IMG achieved the same initial effect, it is likely that the P adsorption capacity at the lower rate (20 t/ha) will be exhausted sooner than the higher rate. Further monitoring of the site will be needed to confirm this.

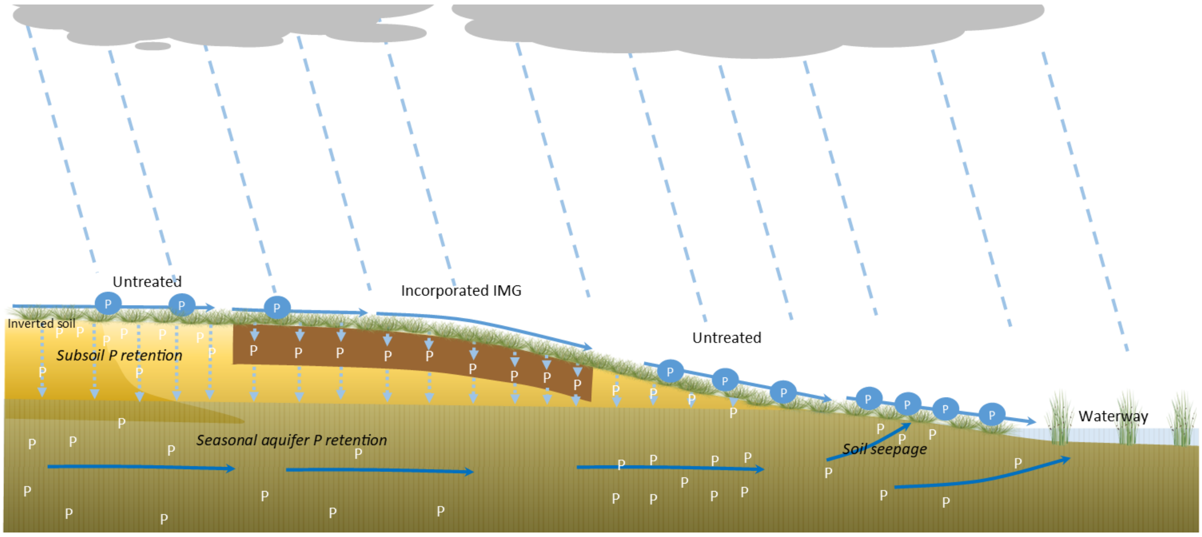

Figure 8: *Conceptual model of P transport pathways and influence of IMG on these at the Yoongi Downs site.*

Mixing IMG with soils during renovation of pastures provides an option to immediately treat areas in catchments that have P-saturated soils prone to leaching and surface runoff. The effects on soil-water P also directly reduces P lost in runoff, as runoff assays on IMG-treated soils in the Peel region have found. Some of the reduction in runoff loss will also be due to reduced P in groundwater seepage which would gain most P as water moves through the topsoil to the surface – like leaching but in reverse. Areas at risk of P runoff or seepage are mostly where sandy soils are prone to saturation because of a high watertable or have limited infiltration following rain events.

Broad-level indications of paddocks with risk of high P losses might be identified using conventional Colwell P and PBI analyses in standard farm soil tests. Phosphorus Environmental Risk Index (PERI) calculated using these measurements potentially provides an indicator of P leaching risk and priority areas for treatment.

The PERI’s effectiveness in indicating risk at the scale of on-farm sampling needs to be confirmed.

IMG treatment had other immediate benefits for soil quality, including raising soil pH by up to 0.5 units and slightly increasing the capacity to retain cations. Water quality also benefitted from increased retention of organic N and P, but there were no effects on leaching of ammonia and nitrate. Most of the high rates of N leaching (up to 270 kg N/ha/yr) were due to NO_3 losses that were magnified by mineralisation of N in soils over summer–autumn.

Inversion of soils using deep ploughing offers an additional option to manage soils with high P losses, at least for some soil types. Burying high-P topsoils with mouldboard ploughing can immediately reduce some risks to runoff, although P leaching losses are likely to continue. The greatest benefits are likely to be achieved where adsorption capacity associated with clays or iron oxide minerals in subsoils are brought to the surface and mixed with sandy topsoils. The application of this management option is likely to be site specific, depending on soil type and the P retention of the deep soil horizons, and is the subject of further investigation.

IMG treatment results in increased leaching of Mn and, to a lesser extent, Co. However, risks of these metals to plants and aquatic organisms are likely mitigated by the high concentrations of Ca also leaching with the metals. The magnitude of Mn leaching has not been found in previous Western Australian IMG experiments and appears to be linked with pH-mediated reactions of the IMG in the mildly acidic soils. Leaching of SO_4 from IMG-treated soils also greatly increased but was similar to some untreated soils. Increased leaching of Ca and SO_4 is expected to persist for no more than six years, even at the highest rate of IMG. The long-term trends and magnitudes of the leaching risks will be investigated in ongoing work.

The adoption of IMG as a soil amendment in the Vasse region faces cost hurdles, noting that on-farm retention of P is more cost-effective than off-farm water treatment. The initial cost-benefit analysis indicates that costs to the farmer outweigh direct benefits by more than double for a minimum application scenario of 20 t/ha of top-dressed IMG. IMG application costs are largely determined by transport but also depend on the sale price of IMG. Currently, on-farm benefits include increased retention of P on farms that would otherwise be sourced as fertiliser P and improved soil pH, but with no pasture growth benefits measured to date. Other benefits may emerge in future years if soil organic C retention improves. Off-farm benefits of P retention in terms of reduced risk of algal blooms and avoiding treatment costs to remove P are likely to greatly outweigh the on-farm benefits. The cost of removing P from drains and downstream waterways can be considered an externalised cost of farm production that is carried by the environment and society as a whole.

Further investigations into the effects of IMG on soil properties, P loss and patterns of Mn, Co and SO_4 leaching will be carried out under the Healthy Estuaries WA program. These will be complemented with additional trials in different soil and farm settings to build confidence among farmers and agronomists to use IMG as a soil amendment in pasture production systems.

Appendices

Appendix A — Analysis specifications

Leachate water – analysis suite			
Property	**Method**	**Limit of resolution**	**Reference**
Physical properties	pH, electrical conductivity (EC), dissolved oxygen (DO) and temperature, measured with a YSI probe	pH 0.01 EC 0.001 ms/cm DO 0.1% Temp 0.1°C	(APHA 1998)
Total P (TP)	Persulphate digestion and ascorbic acid colorimetric method (APHA method 4500-P, current)	0.005 mg/L	(APHA 1998)
Total N (TN)	Persulphate digestion and cadmium reduction method (APHA method 4500-N, current)	0.025 mg/L	(APHA 1998)
NH_3-N	Phenate colorimetric method (APHA method 4500-NH_3, current)	0.01 mg/L	(APHA 1998)
NO_x-N	Cadmium reduction and colorimetric method (APHA method 4500-NO_3-, current version)	0.01 mg/L	(APHA 1998)
Filterable reactive P (FRP = soluble P)	Ascorbic acid colorimetric method (APHA method 4500-P, current)	0.005 mg/L	(APHA 1998)
Dissolved organic N (DON)	By calculation from TN analysis on filtered sample	0.05 mg/L	(APHA 1998)
Total P – filtered (to calculated organic P)	Persulphate digestion and ascorbic acid colorimetric method (APHA method 4500-P, current)	0.005 mg/L	(APHA 1998)
Dissolved organic C (DOC)	APHA method 5310: inorganic carbon must be purged before analysis, hence volatile organic species will be lost. Report as non-purgeable organic carbon.	1 mg/L	(APHA 1998)
Anions (Cl, SO_4)	Ion chromatography or turbidimetric methods	Cl 1mg/L SO_4 1 mg/L	(APHA 1998)
Alkalinity	Titration	1 mg/L	(APHA 1998)

Leachate water – analysis suite			
Property	**Method**	**Limit of resolution**	**Reference**
2qDissolved metals (Ca, Mg, Sr, Fe, Mn, K)	Digestion in nitric/hydrochloric mix with peroxide finish Analysis of digest by ICPAES – generally metals >1 mg/L Analysis of digest by ICPMS – metals <1 mg/L	Mg, Ca, K, Na – 1 mg/L	(APHA 1998)
Total metals (As, Ba, Bi, Co, Cr, Cd, Cu, Fe, Ga, Mn, Ni, Pb, Th, Ti, U, Zn)	Digestion in nitric/hydrochloric mix with peroxide finish. Analysis of digest by ICPMS – metals <1 mg/L (anhydride generation for As)	Fe, Mn, Ga, Sr, – 1 µg/L As, Ba, Bi, Cd, Cr, Co, Cu, Pb, Zn – 0.5 µg/L U, Th – 0.1 µg/L Bi, Cd – 0.05 µg/L	(APHA 1998)

* Note: Soluble P is determined as FRP

Soil sample – analysis suite			
Property	**Method**	**Limit of reporting**	**Reference**
pH	pH in 0.01 M $CaCl_2$ extract of air dried soil (<35ºC) with 1:5 soil:solution – method 4B41 pH in water with 1:5 soil:solution – method 4A1	Not applicable	(Rayment & Lyons 2011)
EC	ED in 1:5 deionised water extract of air dried soil with 1:5 soil:solution – method 3A1	0.01 dS/m	(Rayment & Lyons 2011)
PBI	Equilibration of air dried (<35ºC) soil with 100 mg/L P solution (1:10) with automated reactive P measurement – method 9I2a/b	1	(Rayment & Lyons 2011)
PRI	10 mg P/L solution in 0.02M KCl in a 1:20 soil:solution ratio with air dried (<35ºC) soil for 16 hours with analysis of P remaining in solution	Not applicable (units of mL/g)	(Rayment & Lyons 2011)
Colwell P (plant available P)	0.5 M $NaHCO_3$ extraction (1:100) of air dried (<35ºC) soil with automated reactive P measurement – method 9B2	15 mg/kg	(Rayment & Lyons 2011)
$CaCl_2$-P	Method 9F2 – 0.01 M $CaCl_2$ extraction (1:5) with automated reactive P measurement	0.005 mg/kg	(Rayment & Lyons 2011)
KCl extractable S	Method 10D1 – 0.25 M KCl extraction (1:7) at 40ºC with SO4 measurement by ICPAES	0.5 mg/kg	(Rayment & Lyons 2011)
Colwell K	Method 18A1 – 0.5 M $NaHCO_3$ extraction (1:100) for 16 hours with automated reactive P measurement	1 mg/kg	(Rayment & Lyons 2011)
NH_4 and NO_3	Method 7C2b – 2M KCl extraction (1:10) at 25ºC for 1 hour with N measurement by automated colourimetric measurement	1 mg/kg	(Rayment & Lyons 2011)

Soil sample – analysis suite			
Property	**Method**	**Limit of reporting**	**Reference**
CEC with prewash	Method 15E1 – alcohol and glycerol prewash to remove soluble salts with 1 M NH_4Cl extraction (1:10) and cation determination by ICPMS	0.1 c mole$_c$/kg	(Rayment & Lyons 2011)
Total organic carbon	Combustion and infrared analysis, persulphate UV oxidation – method 6B3 OR Dichromate digestion and titration – method 6A1	0.05%	(Rayment & Lyons 2011)
Total P	Strong acid digestion (either Kjeldahl concentrated sulfuric or nitric) with analysis by ascorbic acid – method 9A3a	2 mg/kg	(Rayment & Lyons 2011)
Total S	High temperature combustion (by LECO or other analyser) of dried and finely ground soil – method 10A2	0.01 % (100 mg/kg)	(Rayment & Lyons 2011)
Total metals (Ca, Mn, Fe)	Strong acid digestion (aqua-regia – HNO_3 and HCl) of oven dried and finely ground soil with analysis of extracts by ICPMS (for high % elements) – method 17B1 (reverse Aqua regia) or method 17C1 (Aqua regia)	Ca, Fe – 50 mg/kg Mn – 5 mg/kg	(Rayment & Lyons 2011)
Total metals (Sr, Pb, Cd etc)	Strong acid digestion (aqua-regia – HNO_3 and HCl) of oven dried and finely ground soil with analysis of extracts by ICPAES (for lower % elements) – method 17B1 (reverse Aqua regia) or method 17C1 (Aqua regia)	Sr – 0.1 mg/kg Cd – 0.05 mg/kg	(Rayment & Lyons 2011)

Plant sample – analysis suite			
Property	**Method**	**Limit of reporting**	**Reference**
Total C, N	High temperature combustion (by LECO or other analyser)	0.01 %	(McQuaker et al. 1979)
Total P, K, S, Cu, Zn, Mg, Mn, Ca, Mn, Na, Fe, B	Digestion by H_2O_2 and nitric acid of dried ground samples with digestate analysed by ICP spectroscopy (ICPOES)	0.01 %	(McQuaker et al. 1979)
Total Cd, Mo, Co, Se, Pb, Cu, Ar, Ni, U, Th + other trace elements	Digestion by H_2O_2 and nitric acid of dried ground samples with digestate analysed by ICP-MS	0.01 mg/kg	(McQuaker et al. 1979)

Appendix B — Lysimeter design and sampling

Zero tension lysimeters were used to collect water percolating from topsoils during winter and spring in two experiments. While this design can sometimes fail to capture all recharge, research shows that this does not influence the concentrations of leachates (Peters & Durner 2009). Furthermore, installation of the pan immediately beneath the soil surface with increased surface area can minimise these losses (Jemison & Fox 1992).

Each lysimeter was a shallow circular PVC collection pan with a conical base draining via plastic tubing to a 10 L or 20 L polycarbonate drum beneath the pan (Figure A9). The drums were fitted with sampling tubes (held in the base of the drums with a Teflon-coated weight) and an air return tube, that allowed air to exit as water entered the inlet tube. Small collection and air return tubes (2 mm ID polythene) were used on the 10 L drums, whereas the 20 L drums were fitted with 4 mm ID black polythene sampling tubes for faster pumping.

Previous research suggests that lysimeters are prone to lateral losses of unsaturated flows, but that side walls can minimise this effect (Gee et al. 2002). For annual leachate volumes of >100 mm/yr, side walls of at least 5 cm on the collection pan can reduce losses to <10% (Gee et al. 2002). At Yoongi Downs, 24-cm-diameter pans with an 8 cm side wall were used, creating a catch area of 452 cm^2. Each collection pan was filled with medium-grained washed white sand to facilitate free drainage of water from the soil to the drain tube. A fine nylon mesh was used to stop the sand washing into the tubing.

Lysimeters were installed by digging a ~0.6 m pit with an adjacent bench of ~0.4 m. The topsoil of interest was removed as a single intact layer and placed to the side during construction. The top of each pan was specifically set below the soil layer of interest and seated on bricks with soil firmly packed around these to prevent settling. Once the pan was in place, the collection drum was seated in a deeper hole offset from the pan to ensure water would flow freely into the drum from the pan. The drum was set at an angle with the sampling tubes (weighed down with a Teflon-coated sleeve) placed inside at the lowest corner of the drum. Soil was placed back around the drum (in the same layers as the surrounding profile) and compacted.

The sample and air tubes were fed to the surface, housed in 90 mm diameter PVC tubes and protected from cattle or machine damage using a cast steel well cover (265 mm OD FB series). To prevent insect contamination, the air tubes were capped with aquarium bubbler stones and the sample tubes were capped with coarse syringe filters.

Leachate collected in the drums was pumped using a hand-vacuum extraction pump (6.5L Alemlube pump) every two to four weeks depending on rainfall. The first part of each sample was used to rinse the pump with all volume collected and measured to the nearest 0.1 L. The collection pump was rinsed with deionised water between each sampling.

Leachate volume is measured as mL and recorded as millimetres leachate (on an area basis) using the conversion *0.0221 (being volume/45.2).

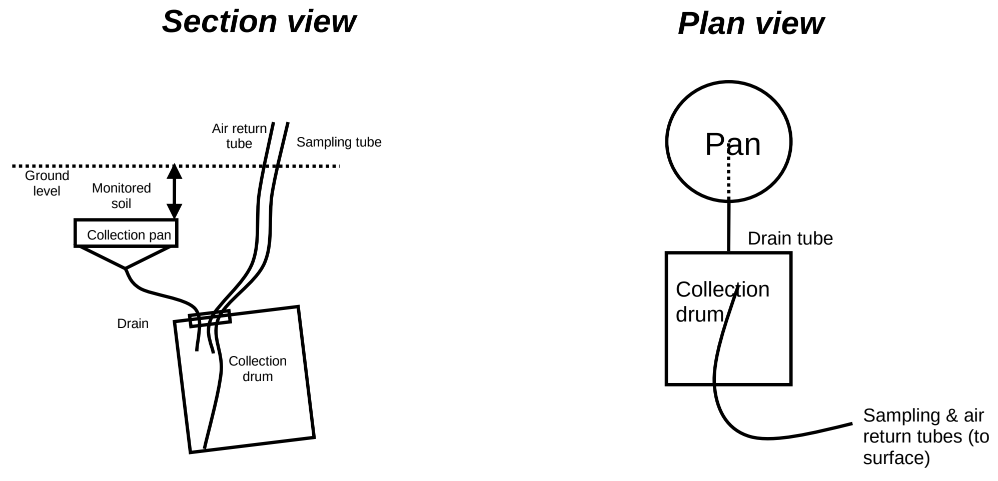

Figure A9: Generic lysimeter design and installation plan

Appendix C — Temporal patterns of nutrients and selected element concentrations in leachates from the Yoongi Downs experiment

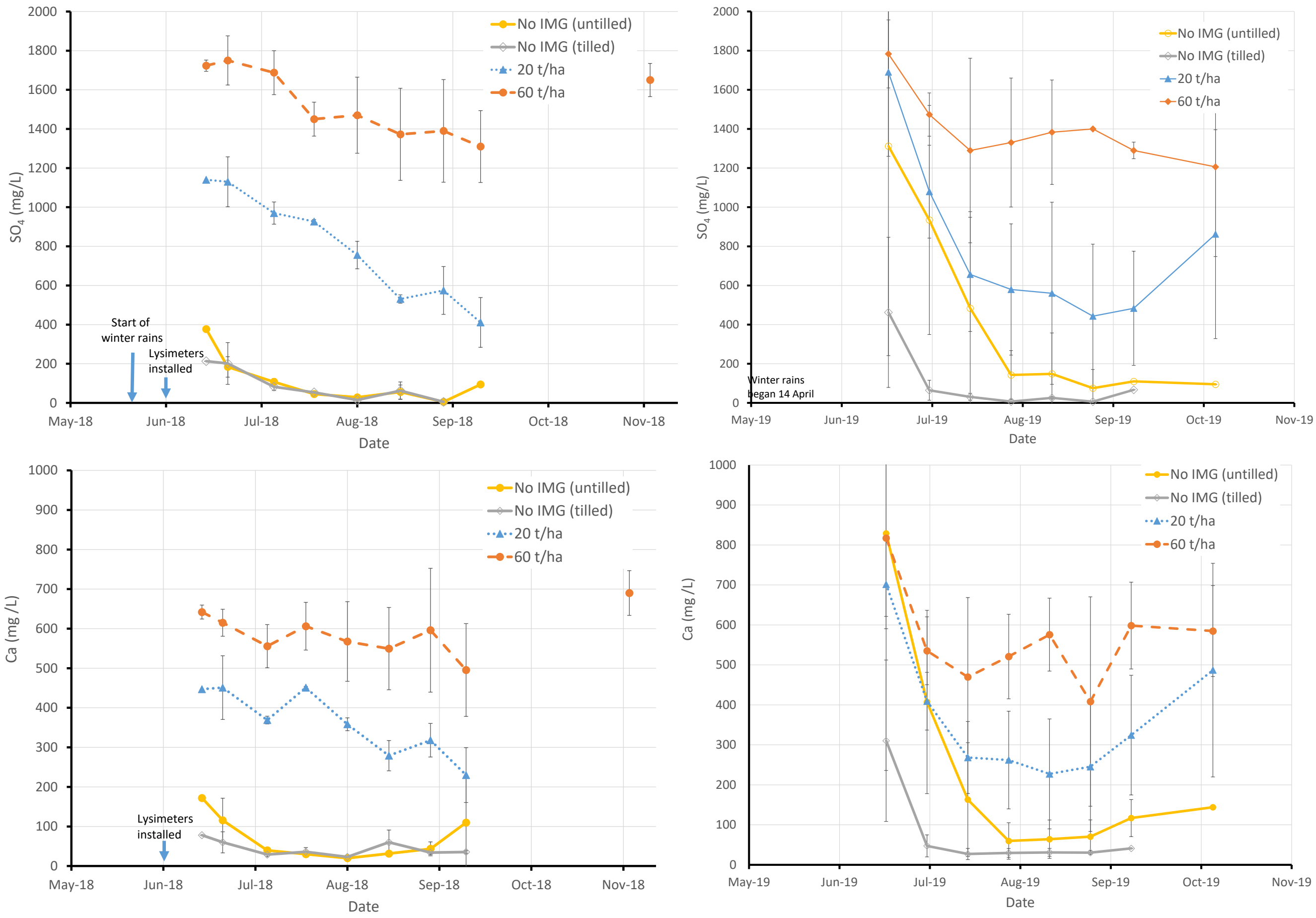

Figure A10: Sulfate (SO_4) and total calcium (Ca) in leachates from untreated and IMG-treated soils over two seasons (error bars represent standard deviation of three field replicates).

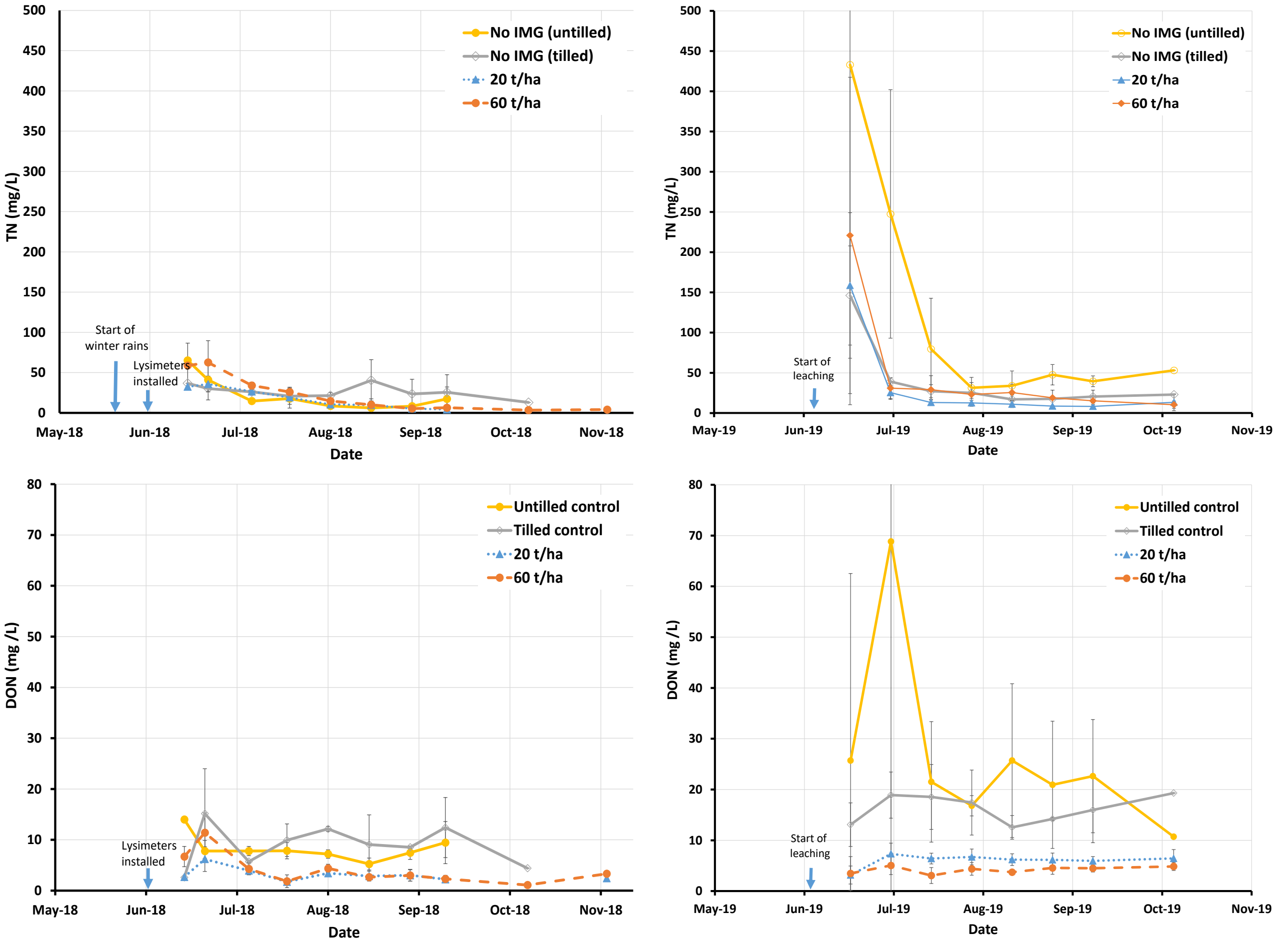

Figure A11: Total nitrogen (TN) and dissolved organic N (DON) in leachates from untreated and IMG-treated soils over two seasons (error bars represent standard deviation of three field replicates).

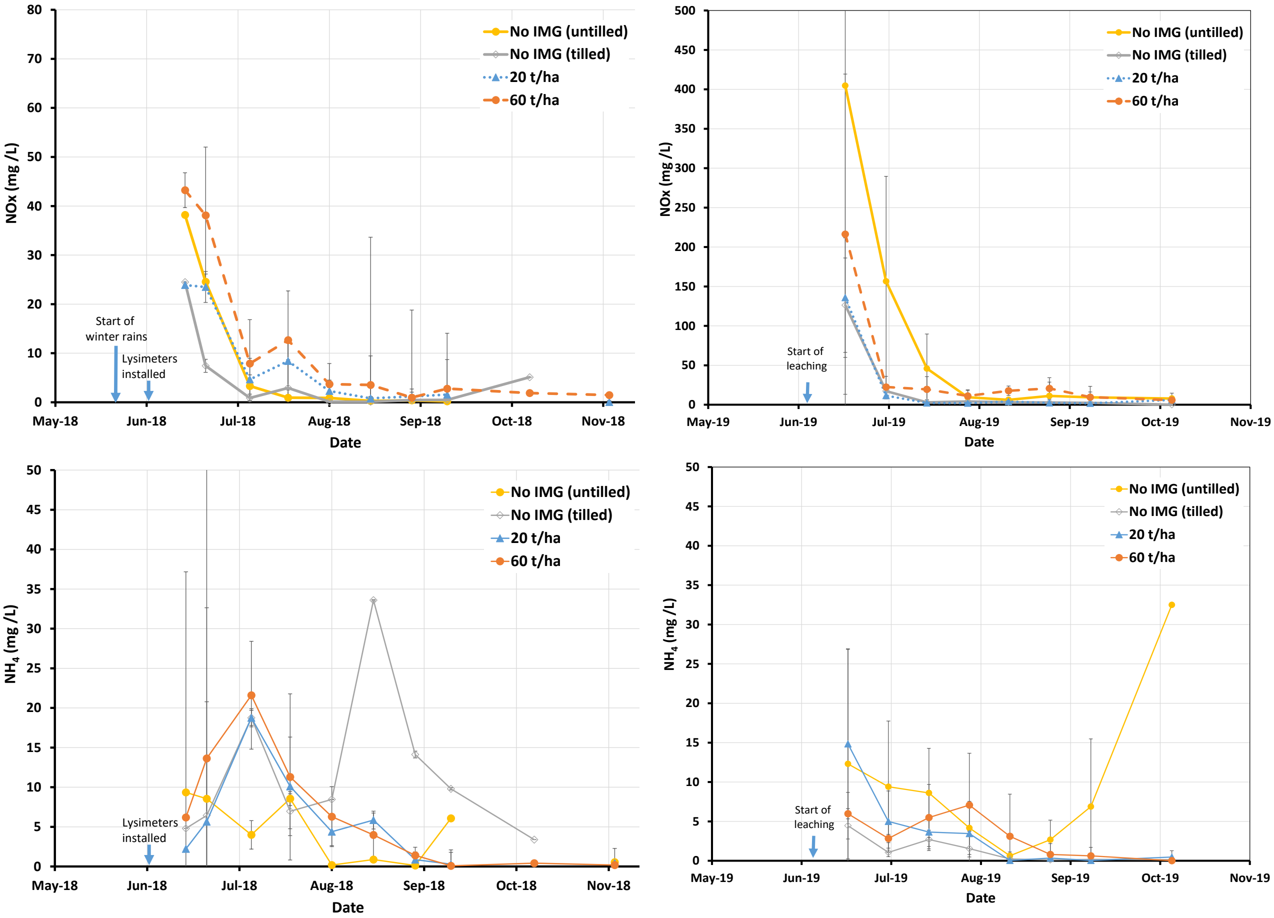

Figure A12: Oxidised N (as NOx) and ammonia (NH_4-N) in leachates from untreated and IMG-treated soils over two seasons (error bars represent standard deviation of three field replicates).

Appendix D — Concentrations of selected elements leached from soils in the Yoongi Downs experiment

Table A14: Annual average pH, Cl, K and Mg concentrations in leachates for the bleached sand (soil 2).

Treatment	**pH**				**Cl (mg/L)**				**K (mg/L)**				**Mg (mg/L)**			
	2018		**2019**		**2018**		**2019**		**2018**		**2019**		**2018**		**2019**	
No IMG (untilled)	5.96	±0.65	6.28	±0.43	138	±169	806	±1177	14	±15	131	±93	20	±19	49	±44
No IMG (tilled)	6.39	±0.51	6.45	±0.42	96	±68	190	±336	30	±21	79	±49	14	±7	29	±52
20 t/ha	6.83	±0.59	6.64	±0.33	44	±50	289	±441	9	±9	69	±99	29	±14	63	±78
60 t/ha	6.84	±0.32	6.54	±0.26	205	±280	242	±355	73	±71	154	±178	93	±39	86	±60

Table A15: Summary average volume weighted Mn, SO_4 and hardness concentrations (mg/L) in leachates for the bleached sand (soil 2).

Treatment	**Total Mn (as mg/L)**				**SO_4 (as mg/L)**				**Hardness (as mg $CaCO_3$/L)**			
	2018		**2019**		**2018**		**2019**		**2018**		**2019**	
No IMG (untilled)	0.21	±0.07	0.8	±0.5	166	±49	1036	±289	363	±156	1961	±199
No IMG (tilled)	0.31	±0.07	0.4	±0.3	125	±6	278	±146	269	±45	747	±333
20 t/ha	15.89	±4.80	21.4	±7.7	1341	±16	1357	±469	1726	±71	2715	±1314
60 t/ha	30.07	±18.80	19.3	±2.7	2621	±191	2446	±302	3161	±236	3182	±1018

Appendix E — Element and bulk property analysis of Iron Man Gypsum

Table A16: Major element analysis (corrected to 39% moisture content) and major properties of IMG used in this experiment with summary analysis of collated independent samples from multiple bulk analyses and published sources.

Element[1]/ property	**Yoongi Downs Bulk IMG[2]**	**Average**	**Minimum**	**Maximum**	**Number samples/ reports[3]**	**Analysis methods**
Ca	9.4	11.3	9.3	13.0	8	XRF, Digestion & ICPMS
Fe	7.4	10.0	7.0	14.1	8	XRF, Digestion & ICPMS
S	8	9.1	8.0	10.2	8	XRF, Digestion & ICPMS
Mn	2.0	2.0	1.5	2.4	8	XRF, Digestion & ICPMS
Mg	0.6	0.6	0.5	0.8	8	XRF, Digestion & ICPMS
Ti	0.1	0.4	0.1	1.3	8	XRF, Digestion & ICPMS
pH	8.1	8.1	7.7	8.4	6	1:5 in 0.01M $CaCl_2$
Neutralising value (bulk)[4]	4.7	4.8	3.5	7.0	5	HCl digestion
PBI	828	879	828	970	3	PBI-Colwell correction
PRI	>1000	>1000	>1000	>1000	3	

[1] Element contents are % by weight corrected to median moisture content of 39% for received bulk samples.

[2] Composite sample of 20 cores taken from bulk IMG delivered to experiment site.

[3] Combined published and DWER analysis data. Published data from Degens & Shackleton (2016), Douglas et al. (2012), Sharma et al. (2018) and Wendling et al. (2012). DWER analysis data from four truckload batches used for field experiments over four years.

[4] Bulk neutralising value (as % equivalent $CaCO_3$) based on analysis of an unsieved whole sample. A particle size weighted neutralising value is less than this with >47% of dry sieved IMG being >1 mm.

Table A17: Minor element analysis (as ppm corrected to 39% moisture content) of IMG used in this experiment with summary analysis of collated independent samples from multiple bulk analyses and published sources.

Element[1]	Yoongi Downs Bulk IMG[2]	Average	Minimum	Maximum	Number samples/ reports[3]	Analysis methods
Ag	NA	< 2	< 2	< 2	2	XRF
As	5.0	5.8	3.6	7.2	9	XRF, Digestion & ICPMS
B	NA[4]	11.5	–	–	1	Digest
Ba	15	26	14	47	9	XRF, Digestion & ICPMS
Bi	<0.1	1.7	< 0.1	5.0	6	XRF, Digestion & ICPMS
Br	<0.1	1.5	< 1	8.6	3	XRF
Cd	0.04	0.5	< 0.04	0.7	7	XRF, Digestion & ICPMS
Ce	NA	103	68	116	4	XRF
Co	225	156	112	262	9	XRF, Digestion & ICPMS
Cr	61	67	56	87	9	XRF, Digestion & ICPMS
Cs	NA	4.0	< 1	9.3	4	XRF
Cu	15	19	14	32	9	XRF, Digestion & ICPMS
Ga	14	15	12	19	8	XRF, Digestion & ICPMS
Ge	NA	< 1	–	–	2	XRF
Hf	NA	< 7	–	–	2	XRF
I	NA	< 7	–	–	2	XRF
K	NA	551	420	775	4	XRF
La	NA	34	24	45	5	XRF, Digestion & ICPMS
Mo	NA	1.8	<0.5	1.4	3	XRF, Digestion & ICPMS
Na	NA	1140	539	2210	4	XRF
Nb	NA	19	12	38	4	XRF
Nd	NA	34	24	52	3	XRF
Ni	61	47	34	65	9	XRF, Digestion & ICPMS
P	NA	976	270	3700	4	XRF
Pb	27	25	5.7	58	9	XRF, Digestion & ICPMS
Rb	NA	7	7	7	2	XRF

Element[1]	Yoongi Downs Bulk IMG[2]	Average	Minimum	Maximum	Number samples/ reports[3]	Analysis methods
Sb	NA	8	2.9	11	3	XRF, Digestion & ICPMS
Sc	NA	< 4	< 4	< 4	2	XRF
Se	0.2	0.9	< 0.2	3.6	7	XRF, Digestion & ICPMS
Sm	NA	< 10	< 10	16	2	XRF
Sn	NA	2.6	< 0.5	5.0	3	XRF, Digestion & ICPMS
Sr	690	644	521	754	8	XRF, Digestion & ICPMS
Ta	NA	< 6	< 6	< 6	2	XRF
Te	NA	< 6	< 6	< 6	2	XRF
Th	80	84	70	110	2	XRF
Tl	NA	7.1	5.7	8.6	2	XRF
U	4.0	7.8	3.7	11.5	8	XRF, Digestion & ICPMS
V	NA	58	36	75	3	XRF, Digestion & ICPMS
Y	NA	13	12	14	2	XRF
Yb	NA	< 9	< 9	< 9	2	XRF
Zn	24	29	20	49	9	XRF, Digestion & ICPMS
Zr	NA	36	11	106	4	XRF

[1] Element contents are % by weight corrected to median moisture content of 39% for received bulk samples.

[2] Composite sample of 20 cores taken from bulk IMG delivered to experiment site.

[3] Combined published and DWER analysis data. Published data from Degens & Shackleton (2016), Douglas et al. (2012), Sharma et al. (2018) and Wendling et al. (2012). DWER analysis data from four truckload batches used for field experiments over four years.

[4] NA = not analysed.

Appendix F — Estimating paddock P balance with grazing cattle

Manure P inputs per grazing event was calculated as:

Cattle number / area grazed x % daily time on paddock x manure P excreted/day

Manure P input was estimated to range from 1.8 to 11.1 kg P/ha/grazing cycle (Table A18) based on dairy cattle excreting 21 to132 g P/cow/day (Aarons et al. 2020). This wide range is a function of the milk production per animal relative to the P in supplemental feeds. P intake in supplemental feeds to dairy cattle ranges from 2 to 155 g/cow/day with an average of 42 g/cow/day (Aarons et al. 2020). In Western Australia, surveys found the P in supplemental feeds ranged from 7 to 53 g/cow/day with an average of 27 g/cow/day (Sharon Aarons pers.comm. 2021).

The calculation also assumed 500 cows grazing 2.5 ha strips for 10 hours each cycle (the minimum time on the paddock between 12-hourly milking). Cattle were grazed in 2.5 ha strips on the paddocks eight times per year during the main grass growing seasons (winter to early summer). Manure from these were calculated to add between 14 and 89 kg/ha/yr to the paddock (Table A18). There were an additional minimum of four nights per year in summer when the cattle were fed supplementary feed and allowed to roam the whole 7.8 ha. Collectively, the manure P input from grazing could average 48 kg/P/ha/yr, but range from 16 to 103 kg P/ha/yr mostly depending on the P content of supplemental feeds (Table A18).

During the main grass growth period, P removed in grazing was estimated to range from 7.5 to 10 kg P/ha per grazing cycle, assuming consumption of 1.5 to 2 t/ha each time and 0.5% P in the grazed pasture.

Table A18: Estimates of manure P inputs from summer and winter grazing.

Manure inputs	Manure P excreted (g P/cow/day)	Manure input per winter-spring night/day feed (kg P/ha)	Winter-spring grazing manure input per year (kg P/ha)	Summer grazing manure input per year (kg P/ha)	Total manure P input (kg/ha/yr)
Min	21	1.8	14	2	16
Avg	61	5.1	41	7	48
Max	132	11.1	89	14	103

Shortened forms

ANOVA	Analysis of variance
$CaCl_2$-P	Calcium chloride extractable phosphorus
CEC	Cation exchange capacity
FRP	Filterable reactive phosphorus
IMG	Iron Man Gypsum
meq	Milli-equivalent
PBI	Phosphorus buffer index
PERI	Phosphorus environmental risk index
PRI	Phosphorus retention index
TN	Total nitrogen
TP	Total phosphorus
XRD	X-ray diffraction
XRF	X-ray fluorescence

Glossary

ANOVA	Analysis of variance, being a statistical approach to partitioning variation due to different causes (factors) and testing whether these are by chance.
$CaCl_2$-P	Measurement of P in soil-water (generally ortho-phosphate P), obtained using a weak salt extract (0.01 M $CaCl_2$).
Cation exchange capacity	Cation exchange capacity, being a measure of the amounts of common positive ions (usually calcium, sodium, potassium and magnesium) that a soil can hold on clays, organic matter and iron oxides.
Flux	Mass movement (as leachates in this report) per unit area per unit time (generally kg/ha/yr in this report).
FRP	Filterable reactive P which is the concentration of ortho-phosphate measured after filtering through a 0.45 um membrane. This is an indicator of phosphate-P concentration.
Iron Man Gypsum	The product name for a material rich in gypsum and iron oxide which is a by-product of titanium refining at Iluka’s plant in Capel.
meq	Milli-equivalent being the equivalent moles of positive or negative charge
Soluble P	Plain language term for phosphorus technically known as filterable reactive phosphorus (FRP). This is phosphorus determined as the reactive phosphorus (orthophosphate-P) in water after filtering through a 0.45 μm filter membrane.
Phosphorus buffer index	A measure of the inherent P adsorption capacity of soils.
pH_{CaCl2}	Soil pH measured in a solution of 0.01M$CaCl_2$ extracted from soil.
Phosphorus environmental risk index	An indicator of potential risk of soil P loss (under leaching or runoff conditions) derived as Colwell P/PBI.
Phosphorus retention index	A measure of the capacity of soil to adsorb further P reflecting the residual P adsorption capacity of soils. Values < 0 indicate no capacity to retain P, values > 0 indicate capacity to retain P.
Total nitrogen	The mass concentration of all forms of elemental N (organic and inorganic) in soil or water.
Total phosphorus	The mass concentration of all forms of elemental P (organic and inorganic) in soil or water.

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