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Long-Term Effects of Compost and Cover Crops on Soil Phosphorus in Two California Agroecosystems

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Inefficient P use in agriculture results in soil P accumulation and losses to surrounding ecosystems, highlighting the need to use P inputs more efficiently. Composts reduce the need for mineral fertilizers by recycling P from wastes at the regional scale, whereas cover crops reduce soil P losses and have the potential to increase internal soil P recycling by mobilizing soil P "fixed" from previous P applications. We studied the effects of compost and cover crops on soil P in two California experiments, using one to measure the effects of a single cover crop mixture and composted poultry manure across different management practices, and the second experiment to evaluate how different cover crops (pure grass, pure mustard, or grass-legumes) and yard compost affected soil P dynamics under organic management. We determined changes in soil P dynamics 8 to 18 yr after long-term experiments were established by measuring chloroform-extractable P, P sorption capacity, P saturation, and Hedley fractions. Cover crops generally increased microbial and organic P, whereas amendment with yard compost increased resin, microbial, and organic P, with no impact of cover crops and yard compost on other pools or on P sorption. In contrast, addition of composted poultry manure significantly increased all soil P pools (microbial, organic, and inorganic) and P saturation. Our results suggest a limited, moderate, and strong role of cover crops, yard compost, and composted poultry manure, respectively, in affecting soil P in California agroecosystems.

Abbreviations: L(O)CC, legume or legume-oat cover crop; P_{chl}, chloroform-extractable phosphorus; P_{sat}, degree of phosphorus sorption saturation; RR, Russell Ranch Sustainable Agricultural Facility; SOCS, Salinas Organic Cropping Systems; TP, total phosphorus.

Along with N, P is one of the most limiting nutrients in agriculture. Because P inputs from atmospheric deposition and weathering are small in most systems and insufficient to replace soil P removed by harvest, external P inputs are required to maintain soil fertility and avoid decreasing yields (Goulding et al., 2008). However, P use is inefficient in many agricultural systems, resulting in soil P accumulation and potential losses to surrounding ecosystems (Simpson et al., 2011). Developing management practices that increase P use efficiency is thus central to improve agricultural sustainability (Simpson et al., 2011).

Manure and yard compost that are applied on agricultural land reduce the need for mineral P fertilizers and recycle P-rich wastes at the regional scale. Manures are typically rich in both N and P, have low C/N and C/P ratios, and can thus provide sufficient labile nutrients for crops in intensive agriculture (Goulding et al., 2008). Nutrient concentrations and ratios vary among manure types and storage or composting conditions, with poultry and cattle manure respectively having the lowest and highest N/P ratios of commonly used manures (Nelson and Janke,

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2007). Composts, especially yard composts originating from municipal residues, are more variable in terms of nutrient availability (Frossard et al., 2002; Sinaj et al., 2002). In addition, they generally provide lower levels of plant-available nutrients shortly after application than manure due to lower nutrient content and higher C/N and C/P ratios (Hartz et al., 2000). Therefore, yard composts are often used to maintain or increase soil organic matter (Brennan and Boyd, 2012). While both amendments provide nutrients that can partially or fully replace mineral fertilizers, manures typically exhibit higher rates of nutrient release and should be more effective at providing adequate crop nutrition than composts.

Composts and manures contain a substantial fraction of their nutrients in forms that are not directly available to plants (Hartz et al., 2000; Frossard et al., 2002; Takahashi, 2013). In addition, manures and composts are generally enriched in P relative to N compared with plant requirements, which results in excessive P fertilization when manures or composts are applied to meet plant N needs (Sims et al., 2000; Eghball, 2002). Excess P is primarily retained within soils and accumulates over time in pools of lower plant availability, that is, P bound to Al or Fe oxides, Ca-rich minerals (e.g., apatite), or organic matter (Frossard et al., 1995; Rubaek et al., 2013). However, soil particles enriched in P can be lost via runoff, and excess P can be lost via leaching once the soil P sorption capacity is saturated—especially in coarse-textured soils—posing risks for groundwater and surrounding aquatic ecosystems (Pautler and Sims, 2000; Goulding et al., 2008). Therefore, management practices that increase internal soil P recycling and soil P retention may help reduce P input rates and soil P accumulation, resulting in lower risks of P loss to the environment.

One practice that could increase soil P retention and internal soil P recycling is the use of cover crops. Cover crops reduce P losses by minimizing erosion, runoff, and leaching rates via higher soil P retention (Sharpley and Smith, 1991; Bechmann et al., 2005). Cover crops can also convert low-availability P forms retained in soils into labile P by altering rhizosphere properties such as pH and enzyme activity (Horst et al., 2001). Grasses are typically used to reduce P losses and increase soil P retention (Sharpley and Smith, 1991) whereas legumes have a greater soil P mobilization capacity (Nuruzzaman et al., 2006). Evidence from low-P soils suggests that legumes are efficient at mobilizing soil P and providing P for subsequent crops in low-input agriculture on high-P fixing soils (Horst et al., 2001; Nuruzzaman et al., 2005). However, they may be less effective in mobilizing sufficient P when P availability is higher, as studies conducted in agricultural systems with higher soil P suggest limited benefits (Cavigelli and Thien, 2003; Kuo et al., 2005; Takeda et al., 2009). Furthermore, while P retention can be as high under mixtures of grasses and legumes as under pure grasses (Sharpley and Smith, 1991), it is unclear if these mixtures can mobilize soil P as effectively as pure legumes. Thus, understanding how different cover crops affect soil P dynamics across a diversity of management conditions is important to determine their potential to reduce P inputs in intensive agriculture.

Our main objective was to determine the effect of compost amendments and cover crops on soil microbial P, soil pools measured by Hedley fractionation, and P sorption capacity. We sought to determine whether compost or cover crops had the larger impact on soil P, how yard compost differed from composted poultry manure, and how the effects of cover crops varied among plant taxa (grasses, legumes, mustards) and management practices. We used two complementary California long-term agricultural experiments that allowed us to study (i) a single cover crop mixture used under conventional, organic (composted manure) or unfertilized management; and (ii) different cover crops (grass, legume-grass, mustards) and yard compost addition used under organic management.

MATERIALS AND METHODS

Century Experiment at the Russell Ranch Sustainable Agricultural Facility—Davis (California, USA)

In the Century Experiment at Russell Ranch Sustainable Agricultural Facility (RR; established in 1993), we studied the effects of one legume or legume-oat cover crop [L(O)CC] mixture across different management systems. Cover crops were grown in the winter (October–March) and consisted of hairy vetch (*Vicia dasycarpa* Ten., 47.4 kg seed ha^{-1}) and 'Magnus' pea (*Pisum sativum* L., 87.0 kg seed ha^{-1}) in 1994 through 2005, or fava bean (*Vicia faba* L., 89.6 kg seed ha^{-1}), hairy vetch (22.4 kg seed ha^{-1}), and 'Montezuma' oat (*Avena sativa* L., 28.0 kg seed ha^{-1}) in 2006 through 2011 (Table 1). Therefore, the L(O)CC mixture consisted of 100% legume seeds until 2005 and a mixture of 80% legume seeds and 20% oat seeds in 2006 to 2011. We used 36 RR plots (0.4 ha), that were divided into two sub-experiments: Irrigated (referred to as RR-I) and Rainfed (referred to as RR-R). All plots were randomly allocated across two similar soil types: Yolo silt loam (fine-silty, mixed nonacid, thermic Typic Xerothents) and Rincon silty clay loam (fine, montmorillonitic, thermic Mollic Haploxeralfs).

The RR-I plots consisted of irrigated and fertilized 2-yr grain-tomato (*Solanum lycopersicum* L.) rotations subjected to organic, conventional, or mixed (conventional with cover crops) management. Tomato alternated with maize (1994–2007, *Zea mays* L.), a summer sorghum cover crop (2008–2009, *Sorghum bicolor* Moench) or winter wheat (2010–2011, *Triticum aestivum* L.). Because RR compared different farming systems rather than comparing specific management practices, fertilization types, and rates of application varied among systems (Table 1). Each system had six replicated plots, with three replicates under grain and three under tomato in any given year. Nutrient inputs were made primarily to supply sufficient crop N, and were reduced in the mixed system by applying mineral fertilizers only to the tomato phase in 1994 to 2008 vs. fertilization of both tomato and grain in 2009 to 2011. The organic system received intermediate N levels via composted poultry manure that was applied annually (7.2–21.9 Mg ha^{-1} yr^{-1}, 1.83% N, 1.37% P), with the exception of 2008 (no manure applied), and 2010 and 2011 (manure applied only to the tomato phase).

Table 1 Selected properties of the three experiments: Russell Ranch Irrigated (RR-I), Russell Ranch Rainfed [RR-R, where L(O)CC = legume or legume-oat cover crops] or Salinas Organic Cropping Systems (SOCS). In RR, inputs are mineral fertilizers (conventional, mixed, fertilized), composted poultry manure (organic) or no inputs [L(O)CC, unfertilized]. In SOCS, inputs are chicken manure, feather meal, and liquid fertilizers for all systems, with the addition of yard compost in all systems but control.

Experiment	Years	Crops†	System	Cover crops		Yields			Inputs ‡‡	
				Type	Frequency	Dry metric tons ha^{-1} yr^{-1}			N	P
						Wheat	Maize	Tomato	kg ha^{-1} yr^{-1}	
RR-I	1994–2011	Grain and tomato‡	Conventional	None		6.7 ± 1.8	11.5 ± 1.5	4.1 ± 1.2	184	23§§
			Mixed	Pea-vetch or vetch-oat-fava bean§	Biennially#	6.6 ± 1.2	6.2 ± 1.6	4.3 ± 1.4	75	14§§
			Organic	Pea-vetch or vetch-oat-fava bean§	Annually#	5.0 ± 1.0	6.8 ± 2.1	4.1 ± 0.6	135	101
						Wheat				
RR-R	1994–2011	Wheat and fallow	Unfertilized	None		3.4 ± 1.1			0	3§§
			Fertilized	None		4.9 ± 1.3			56	3§§
			L(O)CC	Pea-vetch or vetch-oat-fava bean§	Biennially	3.9 ± 1.1			0	3§§
						Lettuce ††	Broccoli ††			
SOCS	2003–2011	Lettuce and broccoli	Control	Pea-vetch-fava bean-rye	Every fourth year	0.66 ± 0.17	1.7 ± 0.3		204	30
			Compost	Pea-vetch-fava bean-rye	Every fourth year	0.75 ± 0.19	1.7 ± 0.3		435	67
			Legume-rye	Pea-vetch-fava bean-rye	Annually	0.91 ± 0.15	1.9 ± 0.1		435	67
			Mustard	Mustard mix¶	Annually	0.84 ± 0.14	1.8 ± 0.2		435	67
			Rye	Rye	Annually	0.85 ± 0.15	1.8 ± 0.2		435	67

† One crop per year in RR, two per year in SOCS.

‡ Grains: maize (1994–2007), sorghum summer cover crop (2008–2009) or wheat (2010–2011).

§ Pea-vetch for 1994–2005 (100% legume seeds) and vetch-oat-fava bean for 2006–2011 (80% legume seeds and 20% oat seeds).

¶ Mixture of *Sinapis alba* and *Brassica juncea*.

Every year until 2009 or only before tomato (2010–2011) for organic vs. only before maize (1994–2004) or tomato (2010–2011), or every year (2004–2009) for mixed.

†† Yields for lettuce and broccoli were computed from aboveground biomass and are therefore not marketable yields.

‡‡ Average for the whole study period, and only including fertilizers and composts, i.e. excluding inputs from N-fixation (75 kg N ha^{-1} yr^{-1} when cover crops are grown), seeds and transplants.

§§ Includes a one-time 49 kg P ha^{-1} application in 1999.

Nitrogen-fixation inputs of L(O)CC, estimated as 75 kg N ha^{-1} yr^{-1} in years when L(O)CC were grown (J. Six, personal communication, 2011), were not used to adjust N fertilization rates. The organic system had the highest and most variable P inputs: 50 to 114 kg P ha^{-1} yr^{-1} (tomato) or 163 to 300 kg P ha^{-1} yr^{-1} (maize) in 1994 to 1998, 78 to 115 kg P ha^{-1} yr^{-1} (tomato, maize, sorghum) in 1999 to 2007 and 2009 to 2011, or 0 kg P ha^{-1} yr^{-1} when composted poultry manure was not applied (wheat, 2008). In contrast, conventional and mixed systems respectively received 22 kg P ha^{-1} yr^{-1} (or 11 kg P ha^{-1} yr^{-1} in 2008, 2010, and 2011) and 11 kg P ha^{-1} yr^{-1} annually, except in 1999 when the conventional and mixed systems received an additional 49 kg P ha^{-1}. Cover cropping frequency varied among years and systems. All organic plots were cover-cropped every year until 2009 and only before tomato afterward, whereas cover crops were grown only before maize (1994–2004) or tomato (2010–2011), or every year (2004–2009) in the mixed plots. Conventional plots were never cover-cropped.

The RR-R plots consisted of rainfed 2-yr wheat-fallow rotations divided in three systems: fertilized with mineral fertilizer, unfertilized, or L(O)CC (Table 1). Each system had six replicated plots, with three replicates under wheat and three under fallow or L(O)CC in any given year. Nitrogen fertilization differed among systems—mineral fertilizer (fertilized), legume-derived N-fixation [L(O)CC], none added (unfertilized)—and no external P was added, except in 1999 when 49 kg P ha^{-1} were added to all systems (Table 1). L(O)CC—same mixtures as in RR-I—were grown during the fallow years of the L(O)CC system only.

Salinas Organic Cropping Systems Experiment—Salinas (California, USA)

The SOCS plots (established in 2003) allowed us to measure the effects of different cover cropping practices under certified organic management. All systems (plot size: 240 m^2) were under double-cropping production of lettuce (*Lactuca sativa* L.) and broccoli (*Brassica oleracea* L.), with or without a cover-cropping phase from November until February or March (Table 1). In 2004, spinach (*Spinacia oleracea* L.) was grown instead of broccoli, and in 2011, only lettuce was grown after the winter cover crop phase. The SOCS experiment was under tillage-intensive management that is typical for organic farms in this region. We used five systems from this experiment, with four replicate plots per system. All plots were on a common soil: Chualar loamy sand (fine-loamy, mixed, superactive, thermic Typic Argixerol). All systems received the same amounts of added N—chicken manure, feather meal, liquid fertilizers (e.g., plant or fish extracts)—during the production of vegetables: 22 kg N ha^{-1} yr^{-1} for spinach, 56–74 kg N ha^{-1} yr^{-1} for lettuce, 134–170 kg N ha^{-1} yr^{-1} for broccoli. Yard compost (15.2 Mg ha^{-1} yr^{-1}, C/N ~ 22, 1.5% N, 0.25% P) was added to four of the five systems before each cash crop to provide additional organic matter (see Brennan and Boyd, 2012 for details). Cover cropping frequency (every winter or every fourth winter) and species composition varied: pure 'Merced' rye (*Secale cereale* L., 90 kg seed ha^{-1}), a mustard mixture of 'Ida Gold' white mustard (*Sinapis alba* L., 6.7 kg seed ha^{-1}) and 'Pacific Gold' Indian mustard (*Brassica juncea* Czern., 4.3 kg seed ha^{-1}), or a mixture of rye (42 kg seed ha^{-1}), fava bean (147 kg seed ha^{-1}), 'Magnus' Pea (105 kg seed ha^{-1}), common vetch (*Vicia sativa* L., 63 kg seed ha^{-1}), and purple vetch (*Vicia benghalensis* L., 63 kg seed ha^{-1}) thereafter referred to as the legume-rye mixture. Seeds of the legume-rye mixture were mixed together with appropriate *Rhizobium* inoculants and planted simultaneously as described previously (Brennan and Boyd, 2012).

Sample Collection

Composited soil cores from the plow layer (0–30 cm) were taken in September 2011 from each plot in both experiments, at a time when all systems were fallow. Twenty soil cores per plot were pooled, sieved to 2 mm and kept moist at 4°C before processing. Because summers in California are very dry, field-moist soils were virtually air-dried at sampling and were thus used at field moisture levels for all analyses.

We also analyzed archived 1993 soil samples from RR plots (before the start of the experiment) for Hedley fractions and Olsen P content. These samples were collected at different depths (0 to 15 cm, 15 to 30 cm, 30 to 60 cm, 60 to 100 cm), air-dried, ground to 2 mm, and stored at room temperature. Only samples from 0- to 15-cm and 15- to 30-cm depths were used for Hedley fractions. In addition to 1993 samples, we measured Olsen P in soils from 2003 and 2012 processed in the same way as in 1993, with all four depths used. Similarly, we analyzed air-dried 2010 soil samples from SOCS (0–30 cm) for Hedley fractions to determine if results differed after 3 yr of winter fallow in the low cover-cropping frequency systems. Patterns in the SOCS plots were similar in 2010 and in 2011, hence we only report values for the 2011 samples so that chloroform-extractable P, Hedley fractions, and sorption data originate from the same samples.

Chloroform-Extractable Phosphorus

We measured chloroform-extractable P (P_{chl}), a proxy for microbial P, using a modified chloroform fumigation method after Brookes et al. (1982). For each soil sample, three subsamples were processed: fumigated, non-fumigated and spiked with P. Briefly, we fumigated triplicate 4-g subsamples for each soil with chloroform for 24 h, followed by a 1-h extraction with 40 mL of 0.5 M $NaHCO_3$ and filtration on Whatman 42 filter paper. Two milliliters of extract were then combined with 1 mL of 0.22 M $K_2S_2O_8$ and 0.9 M H_2SO_4 solution and digested for 90 min in an autoclave, and total P was analyzed by the ascorbic acid method (Tiessen and Moir, 2007). Non-fumigated and P-spiked (with a 25 μg P g^{-1} soil addition) subsamples were processed in the same way as fumigated subsamples, except that no chloroform was used during the 24-h incubation. We compared the P concentrations of fumigated, non-fumigated and non-fumigated spiked subsamples to determine P_{chl}, using the following equation:

$$P_{chl} = \frac{\text{Fumigated P} - \text{Non-fumigated P}}{\text{\% Recovery of spike}}$$

We did not include the fraction of biomass extractable by chloroform (Kp) as it was not measured in these conditions, thus we refer to P_{chl} rather than microbial P.

Hedley Fractions

We determined different soil P fractions (Hedley fractions) using a modified version of Tiessen and Moir (2007). Preliminary tests showed no differences between field-moist and air-dried samples or between 2-mm-sieved and ground samples. Hedley fractions were measured on 2-mm-sieved field-moist soils for 2011 samples and on air-dried ground samples for archived samples.

Samples of 0.5 g were sequentially processed with (i) distilled water and anion-exchange resins charged with 0.5 M $NaHCO_3$ (P_{resin}), (ii) 0.5 M $NaHCO_3$, (iii) 0.1 M NaOH, (iv) 1 M HCl, and (v) concentrated H_2SO_4 and H_2O_2. For the first four steps, we shook samples with 30 mL of solution (16 h, 15 rpm end-over-end), followed by centrifugation at 7000 × *g* (7000 rpm) and P analyses on the supernatant by the ascorbic acid method. We eluted resins with 0.5 M HCl for 2 h before measuring P. We used unfiltered supernatant, because we found no differences between filtered (0.45 μm) and unfiltered supernatants in preliminary tests. We digested $NaHCO_3$ and NaOH extracts with $K_2S_2O_8$ (see P_{chl} above), and computed organic P values as the difference between the digested and undigested samples after correcting for color in the undigested samples (Tiessen and Moir, 2007). For the final digest step, we decanted the soil sample with as little distilled water as necessary, mixed 5 mL of concentrated H_2SO_4 with samples, heated the mixture gradually to 360°C to evaporate the water, and made 0.5 mL H_2O_2 additions with heating at 360°C until the color in the sample disappeared. This technique yielded residual P ($P_{residual}$).

We estimated P recovery in Hedley fractions by comparing the sum of Hedley fractions to total P (TP) measurements obtained by X-ray fluorescence (Spectro XEPOS HE, Spectro Analytical Instruments GmbH, Germany) on a subset of seven samples from different systems in RR. Recovery of TP with Hedley fractions was high (>85%) and not affected by management, similar to previous studies (Ciampitti et al., 2011).

Native fertility varied among different plots of a given system, hence we computed a change (Δ) in P fractions between 2011 and 1993 in RR to account for among-plot variations within systems (no archived soils could be used for SOCS). Because the bulk density at 0 to 15 cm and 15 to 30 cm was not significantly different ($F = 3.7$, $p > 0.05$, average = 1.4 g cm^{-3}), samples from 1993 were simply averaged for the two depths and compared with 2011 samples (0–30 cm).

To simplify interpretation, we combined Hedley fractions and refer to the sum of inorganic $NaHCO_3$ and NaOH fractions as P bound to Al or Fe oxides (P_{AlFe}), to the sum of the organic fractions in the alkaline extracts as organic P (P_{org}), and to the HCl fraction as P bound to Ca minerals (P_{Ca}). The $NaHCO_3$ pool contributed 23% (range: 21–31%) of P_{AlFe} and 6% (range: 0–11%) of P_{org} in RR whereas it contributed 39% (33–43%) of P_{AlFe} and 11% (range 8–15%) of P_{org} in SOCS. The P_{resin} and $P_{residual}$ are considered to be the most and least plant-available Hedley fractions, respectively (Tiessen and Moir, 2007).

Phosphorus Sorption Isotherms

We determined sorption isotherms using a modified version of Sharpley et al. (2007). Briefly, we shook duplicate 1-g samples mixed with 25 mL of 30 mM KCl electrolyte buffered with 5 mM piperazine-N,N′-bis(2-ethanesulfonic acid) (PIPES). We added P as KH_2PO_4 at concentrations of 0, 4, 8, 12, 18, or 25 mg L^{-1} to each sample and shook samples end-over-end (24 h, 15 rpm) at 4°C to minimize microbial growth. We used KCl instead of $CaCl_2$ to prevent precipitation of P with Ca, and PIPES to maintain a fixed pH during the equilibration (6.4 was the average pH of all samples). After the equilibration was complete, samples were centrifuged at 795 × *g* (3750 rpm), filtered on Whatman 42 filter paper and P content in the extracts was determined by the ascorbic acid method.

Phosphorus content was fitted to Langmuir isotherms iteratively using nonlinear regression rather than commonly used linearized equations because nonlinear regression provides more accurate estimates of sorption parameters (Bolster and Hornberger, 2007). We used the following equation:

$$S = (S_{max} \times k \times C)/(1 + k \times C)$$

where S is the sorbed concentration (mg P kg^{-1} soil), S_{max} is the soil's maximum sorption capacity (mg P kg^{-1} soil), k is the binding strength coefficient (L mg^{-1}), and C is the equilibrium concentration (mg P L^{-1}). The difference between the P concentration at the beginning and at the end of the 24-h equilibration, in addition to previously sorbed P (S_0), yielded S. We determined S_0 by shaking duplicate 1-g samples with 25 mL of distilled water and an anion-exchange resin charged with 0.5 M $NaHCO_3$ (24 h, 4°C, 15 rpm end-over-end), elution with 0.5 M HCl for 2 h, and P determination by the ascorbic acid method. These extractions yielded slightly less P_{resin} than during Hedley fractions, likely because of a higher soil/solution ratio, but both measurements were highly correlated ($r > 0.89$ for systems taken separately, $r = 0.99$ for all data combined). We used these P_{resin} values to compute the degree of sorption saturation (P_{sat}) because extractions were made in conditions more similar to those used to determine sorption curves. We used the following equation to calculate P_{sat}, similar to the one used by Pautler and Sims (2000):

$$P_{sat} = S_0/S_{max}$$

Depth Profile of Olsen Phosphorus

Samples from 1993, 2003, and 2012 were analyzed for Olsen P down to 1 m (0–15 cm, 15–30 cm, 30–60 cm, 60–100 cm) in RR-I plots to determine the extent of any P accumulation below the plow layer in these systems. Samples (1 g) were extracted with 20 mL of 0.5 M $NaHCO_3$ for 30 min followed by filtration on Whatman 42 filter paper and P determination by the ascorbic acid method.

Phosphorus Budgets

We computed farm-gate budgets for each system. The inputs we quantified were fertilizers, composted manure, yard compost, and plant seeds (cover crops, wheat, maize) or transplants (lettuce, broccoli, tomato). We used manufacturer-specified P content for fertilizers, and determined P content in composts and plant material using H_2SO_4–H_2O_2 digestion (see "Hedley fractions" section above), where 50 mg of plant or composted material were used directly in the digestion procedure rather than at the end of a fractionation. For transplants, we determined P content in the aboveground plant biomass and in the soil of the transplant plug using H_2SO_4–H_2O_2 digestion.

We determined P removed in grain, fruit or leaf removal at harvest using H_2SO_4–H_2O_2 digestion. Yields in RR were determined by harvesting machinery whereas we had to convert total biomass into yields for SOCS, using harvest indices of 0.26 for lettuce (E. Brennan, unpublished data, 2011) and 0.24 for broccoli (R. Smith, unpublished data, 2014) that were representative of production in this region. We assumed that the harvest index for P was the same as for biomass.

We calculated cumulative P budgets for the whole study period and used these to determine how much P should have accumulated in the top 30 cm of soil, using site-specific bulk density values for RR (1.4 g cm^{-3}) and SOCS (1.22–1.3 g cm^{-3} depending on system).

Statistical Analyses

Unless otherwise specified, we present means and 95% confidence intervals so that differences among systems are visually explicit, that is, systems for which means and confidence intervals do not overlap are considered to be significantly different. Confidence intervals also highlight if a delta value is significantly different from zero, that is, if depletion or enrichment occurred.

For P_{chl} and Hedley P, additional statistical tests were used to determine the effects of different factors. We used nested ANOVAs for RR soils, with system (e.g., organic, conventional, or mixed) as the main fixed factor and cropping phase (e.g., grain or tomato) as the nested fixed factor. The data were pooled by system for visual representation but significant differences between cropping phases are discussed in the text. Hedley fractions and P_{chl} were analyzed using one-way ANOVAs for SOCS soils. We evaluated whether residuals were normally distributed and the data had homogeneous variances and applied transformations when necessary. Statistical tests were performed using R (Version 2.12.2, 2011).

RESULTS

Chloroform-Extractable Phosphorus

Composted manure, yard compost and cover crops generally increased P_{chl}. In RR-I, P_{chl} was highest with the combination of composted manure and cover crops in the organic system, intermediate with cover crops only in the mixed system, and lowest in the conventional system without composted manure and cover crops (Fig. 1A). The wheat phase had lower P_{chl} than the tomato phase for organic and conventional systems ($F = 12.9$, $p < 0.001$).

In RR-R, P_{chl} was slightly higher with cover crops, although the difference between cover cropped and fertilized systems was not significant (Fig. 1B). The effect of cropping phase was not significant ($F = 0.6$, $p = 0.66$).

In SOCS, we found the highest P_{chl} with the combination of yard compost and annual cover-cropping of legume-rye, the lowest P_{chl} in the control without annual cover crops or yard compost, and no differences among other systems that all received compost but differed in cover crop type and frequency (Fig. 1C).

Hedley Fractions

In RR-I and RR-R, we could measure the effects of different systems on Hedley fractions over time, using archived samples. We found a substantial increase in P_{resin} and P_{AlFe} in the organic system, a modest increase in the conventional system, and a slight depletion or no change in the mixed system of RR-I (Fig. 2A). Furthermore, we found a strong increase in P_{Ca} in the organic system and depletion of this pool in the conventional and mixed systems. $P_{residual}$ increased significantly in the organic system, whereas it did not change or was slightly depleted in the other systems. In contrast, P_{org} increased in all systems, especially in the organic one, and the mixed system had higher P_{org} enrichment than the conventional system only if one conventional

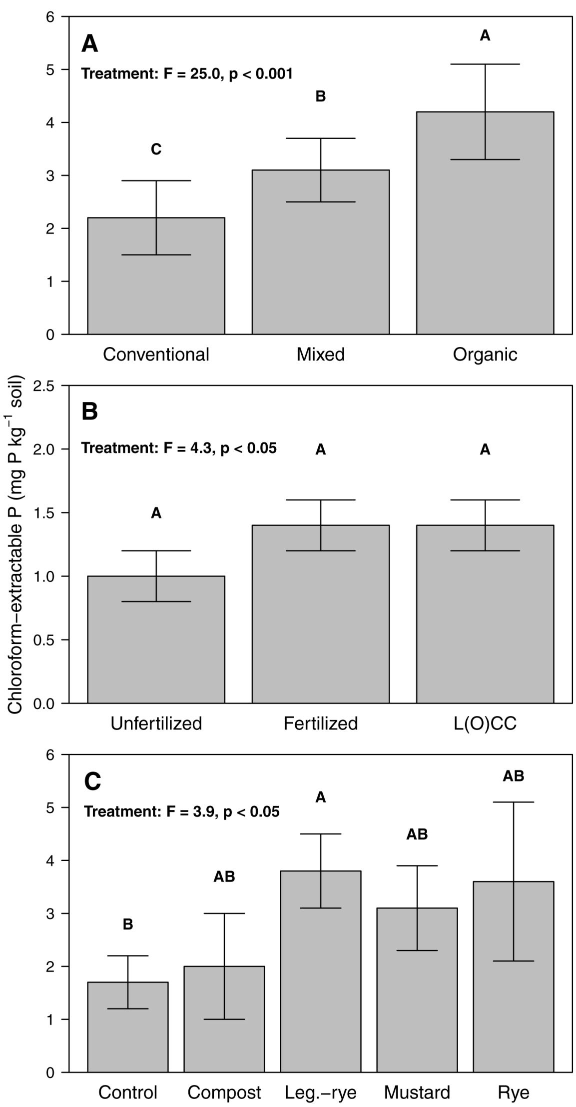

Fig. 1. Chloroform-extractable P in (A) Russell Ranch Irrigated, (B) Russell Ranch Rainfed [L(O)CC = legume or legume-oat cover crops], and (C) Salinas Organic Cropping Systems. Cover crops were grown annually or biennially in mixed, organic and L(O)CC systems for RR (100% legume seeds until 2005 vs. 80% legume seeds and 20% oat seeds in 2006–2011) and in all systems but control and compost for SOCS. Composted manure was added to the organic system for RR and yard compost was added to all systems but control for SOCS. Error bars are 95% confidence intervals of the mean. Different letters within a given experiment represent statistical differences according to a Tukey HSD test (α = 0.05) performed after a nested (RR) or one-way (SOCS) ANOVA.

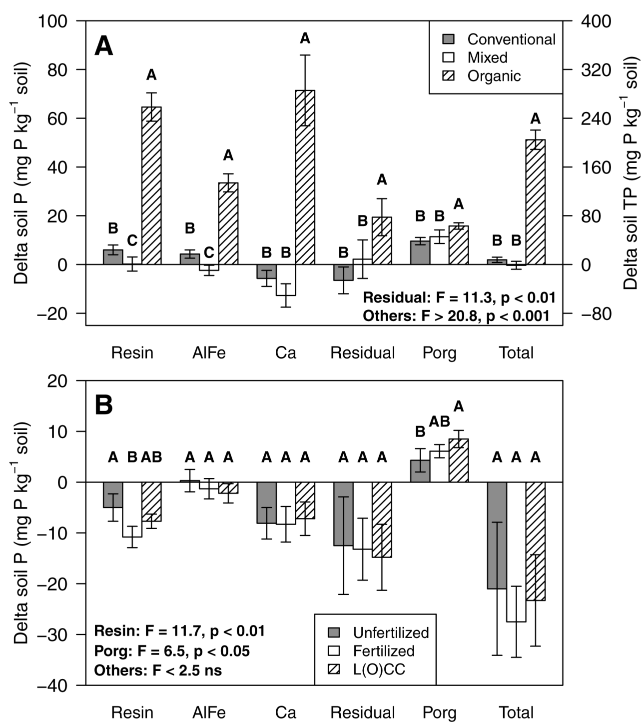

Fig. 2. Changes in Hedley fractions—resin P, P bound to Al or Fe oxides (AlFe), P bound to Ca minerals (Ca), residual P, organic P (P_{org}) and total P—in (A) Russell Ranch Irrigated and (B) Russell Ranch Rainfed [L(O)CC = legume or legume-oat cover crops] between 1994 and 2011. Cover crops were grown annually or biennially in mixed, organic and L(O)CC systems (100% legume seeds until 2005 vs. 80% legume seeds and 20% oat seeds in 2006–2011), and composted manure was added to the organic system only. Error bars are 95% confidence intervals of the mean. Different letters within a given experiment and soil P fraction represent statistical differences according to a Tukey HSD test (α = 0.05) performed after a nested ANOVA.

Table 2 Soil Hedley fractions in 2011– resin P (P_{resin}), P bound to Al or Fe oxides (P_{AlFe}), P bound to Ca minerals (P_{Ca}), residual P ($P_{residual}$), organic P (P_{org}), total P (TP)– in Russell Ranch Irrigated (RR-I), Russell Ranch Rainfed [RR-R, where L(O)CC = legume or legume-oat cover crops] or Salinas Organic Cropping Systems (SOCS). Cover crops were grown annually or biennially in mixed, organic and L(O)CC systems for RR (100% legume seeds until 2005 vs. 80% legume seeds and 20% oat seeds in 2006–2011) and in all systems but control and compost for SOCS. Composted manure was added to the organic system for RR and yard compost was added to all systems but control for SOCS. Mean and 95% confidence intervals of the mean are presented. Different letters within a given experiment and soil P fraction represent statistical differences according to a Tukey HSD test performed after a nested (RR) or one-way (SOCS) ANOVA.

								ANOVA		
Experiment	System	P_{resin}	P_{AlFe}	P_{Ca}	$P_{residual}$	P_{org}	TP	Variable	F	Sig
		mg P kg^{-1} soil								
RR-I	Conventional	28 ± 2B	46 ± 4B	187 ± 9B	243 ± 15B	31 ± 2B	536 ± 13B	$P_{residual}$	5.8	$p < 0.05$
	Mixed	22 ± 4B	40 ± 6B	181 ± 8B	250 ± 10AB	33 ± 2B	526 ± 19B	P_{org}	9.3	$p < 0.01$
	Organic	88 ± 7A	79 ± 5A	267 ± 15A	273 ± 11A	37 ± 3A	744 ± 21A	Others	>75.2	$p < 0.001$
RR-R	Unfertilized	20 ± 3A	44 ± 5A	185 ± 10A	241 ± 8A	26 ± 3A	517 ± 14A			
	Fertilized	20 ± 2A	48 ± 5A	189 ± 6A	248 ± 12A	29 ± 1A	533 ± 19A	All	<2.0	ns
	L(O)CC	18 ± 2A	41 ± 4A	188 ± 8A	242 ± 5A	30 ± 3A	519 ± 14A			
SOCS	Control	69 ± 3B	58 ± 6A	158 ± 16A	127 ± 8A	43 ± 2B	456 ± 31B	P_{resin}	5.9	$p < 0.01$
	Compost	85 ± 7A	64 ± 7A	174 ± 22A	127 ± 8A	47 ± 4AB	497 ± 30AB	P_{org}	5.6	$p < 0.01$
	Legume-rye	79 ± 5AB	64 ± 4A	178 ± 13A	139 ± 9A	53 ± 2A	513 ± 25AB	TP	3.6	$p < 0.05$
	Mustard	82 ± 4A	63 ± 4A	182 ± 18A	139 ± 4A	50 ± 4A	516 ± 27AB	Others	<2.8	ns
	Rye	85 ± 7A	65 ± 8A	195 ± 14A	140 ± 9A	49 ± 1AB	535 ± 40A			

plot with higher P_{org} (13 mg P_{org} kg^{-1} soil) than the other five plots (8–10 mg P_{org} kg^{-1} soil) was removed from the comparison ($F = 29.1$, $p < 0.001$). We found a large increase in TP for the organic system and little if any change for the conventional and mixed systems. Overall, patterns among systems were similar when using soil P concentrations in 2011 instead of differences in P pools over time (Table 2).

For RR-R, we found the highest depletion of P_{resin} in the fertilized system and the lowest in the unfertilized system (Fig. 2B), although this effect was only observed during the fallow or cover-cropped phase ($F = 4.1$, $p < 0.05$). P_{AlFe}, P_{Ca}, and $P_{residual}$ were not significantly different among management systems, and were all depleted. In contrast, P_{org} increased in all systems, especially with cover crops. For this fraction, all systems were similar during the wheat phase ($F = 0.3$, $p = 0.72$) and the cover-cropped system had a higher increase than the others in the fallow phase ($F = 10.4$, $p < 0.05$). Total P was depleted in all systems, with no significant differences among systems. Overall, patterns among systems were similar but weaker when using soil P concentrations in 2011 instead of differences in P pools over time due to differences in native soil fertility (Table 2).

In contrast to RR, we were unable to observe the effects of different systems over time in SOCS due to the lack of archived samples, hence we report patterns of soil P concentrations observed in 2011. Systems amended with yard compost had higher soil P_{resin} than the control while cover crops had no effect on P_{resin} (Table 2). In contrast, yard compost and cover crops did not affect P_{AlFe}, P_{Ca} and $P_{residual}$. Organic P was higher with yard compost only when compost was combined with legume-rye or mustard cover crops. Total P was higher with yard compost combined with rye cover cropping than in the control system.

Phosphorus Sorption

In RR, sorption parameters (S_{max} and k), P_{resin} and P_{sat} were all higher with the combination of cover crops and composted manure in the organic system, but the increase compared with other systems was only significant for P_{resin} and P_{sat} (Table 3). In contrast, sorption parameters, P_{resin} and P_{sat} were little affected by cover crops and yard compost in SOCS.

Changes over Years in Olsen Phosphorus at Russell Ranch Sustainable Agricultural Facility

In both 2003 and 2012, we found a substantial Olsen P enrichment in the organic system for the plow layer (0–15 cm and 15–30 cm) and a slight P enrichment at 30 to 60 cm (Fig. 3). In contrast, Olsen P varied little in conventional and mixed systems: slight enrichment in the plow layer for mixed (15–30 cm) and conventional systems (0–15 cm and 15–30 cm) in 2003, and a small depletion at depth in the mixed (15–30 cm and 30–60 cm) and conventional systems (30–60 cm) in 2012.

Table 3 Resin P (P_{resin}), sorption maximum (S_{max}), binding coefficient (k) and sorption saturation (P_{sat}) in Russell Ranch Irrigated (RR-I), Russell Ranch Rainfed [RR-R, where L(O)CC = legume or legume-oat cover crops] or Salinas Organic Cropping Systems (SOCS). Cover crops were grown annually or biennially in mixed, organic and L(O)CC systems for RR (100% legume seeds until 2005 vs. 80% legume seeds and 20% oat seeds in 2006–2011) and in all systems but control and compost for SOCS. Composted manure was added to the organic system for RR and yard compost was added to all systems but control for SOCS. Mean and 95% confidence intervals of the mean are presented. Systems for which means and confidence intervals do not overlap are considered to be significantly different.

Experiment	System	P_{resin}	S_{max}	k	P_{sat}
		— mg P kg^{-1} soil —		L mg^{-1} P	%
RR-I	Conventional	21 ± 2	173 ± 17	0.26 ± 0.08	12 ± 3
	Mixed	15 ± 4	171 ± 13	0.24 ± 0.06	9 ± 3
	Organic	76 ± 6	193 ± 11	0.38 ± 0.09	39 ± 6
RR-R	Unfertilized	15 ± 2	167 ± 14	0.27 ± 0.08	9 ± 2
	Fertilized	15 ± 2	176 ± 15	0.25 ± 0.07	8 ± 2
	L(O)CC	12 ± 2	177 ± 14	0.23 ± 0.06	6 ± 1
SOCS	Control	61 ± 2	114 ± 9	0.32 ± 0.11	53 ± 9
	Compost	73 ± 5	125 ± 9	0.30 ± 0.09	58 ± 9
	Legume-rye	71 ± 6	126 ± 8	0.29 ± 0.07	56 ± 8
	Mustard	70 ± 3	120 ± 12	0.30 ± 0.12	58 ± 12
	Rye	71 ± 3	127 ± 12	0.26 ± 0.09	56 ± 10

erate deficit, with small differences (4–8 mg P kg^{-1} soil) between expected and observed soil P accumulation (conventional) or depletion (mixed). In RR-R, all systems had a moderate P deficit and expected soil P depletion, and observed soil P depletion was larger than expected by 11 to 12 mg P kg^{-1} soil.

All SOCS systems had a P surplus after 8 yr and were expected to accumulate soil P, with the magnitude of the surplus and soil P accumulation being ~2.5 times larger when yard compost was used annually, that is, in all systems except the control (Table 4).

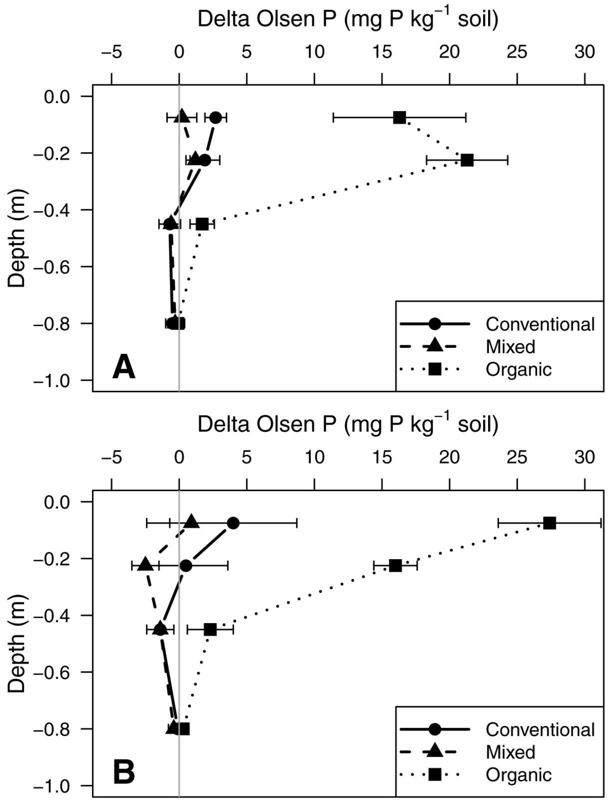

Fig. 3. Change in Olsen-P with soil depth in Russell Ranch Irrigated for (A) 1993–2003 and (B) 1993–2012. Cover crops were grown annually or biennially in mixed and organic systems (100% legume seeds until 2005 vs. 80% legume seeds and 20% oat seeds in 2006–2011), and composted manure was added to the organic system only. Error bars are 95% confidence intervals of the mean.

Phosphorus Budgets

The organic system had the highest P surplus and expected soil P accumulation in RR-I, but observed P accumulation was lower than expected by 148 mg P kg^{-1} soil (Table 4). The conventional system had a small surplus and the mixed system had a moderate deficit, with small differences (4–8 mg P kg^{-1} soil) between

Table 4 Cumulative P inputs (including seeds or transplants), crop removal and P balance, and expected and observed changes in soil P (ΔP) in the top 30 cm of soil for the study period in Russell Ranch Irrigated (RR-I), Russell Ranch Rainfed [RR-R, where L(O)CC = legume or legume-oat cover crops] or Salinas Organic Cropping Systems (SOCS). Cover crops were grown annually or biennially in mixed, organic and L(O)CC systems for RR (100% legume seeds until 2005 vs. 80% legume seeds and 20% oat seeds in 2006–2011) and in all systems but control and compost for SOCS. Composted manure was added to the organic system for RR and yard compost was added to all systems but control for SOCS.

Experiment	System	Years	Inputs	Crop removal	P balance	ΔP in top 30 cm Expected	ΔP in top 30 cm Observed
			kg P ha^{-1}			mg P kg^{-1} soil	
RR-I	Conventional	1994–2011	420	402	+19	+4	+8
	Mixed		262	300	-37	-9	-1
	Organic		1829	345	+1484	+353	+205
RR-R	Unfertilized	1994–2011	52	95	-43	-10	-21
	Fertilized		52	120	-68	-16	-28
	L(O)CC		58	104	-46	-11	-23
SOCS	Control	2003–2011	233	61	+172	+44	
	Compost		517	67	+449	+119	
	Legume-rye		528	80	+448	+122	
	Mustard		514	74	+440	+119	
	Rye		516	75	+441	+121	

DISCUSSION

Effects of Composted Poultry Manure on Phosphorus Dynamics

Composted poultry manure had a greater effect on soil P than other sources of fertility. The increase in soil P from poultry manure was not evenly distributed among pools: 32 to 35% of total soil P increase was found in each of P_{resin} and P_{Ca} vs. 16% in P_{AlFe} and <10% in each of $P_{residual}$, P_{org}, and P_{chl}. The modest increase in P_{org} and P_{chl} is consistent with Oehl et al. (2001), who compared two organic systems to a conventional one after 20 yr of cropping. A small P_{org} accumulation reflects the low abundance of P_{org} in manure, including poultry manure where P_{org} accounts for 18 to 40% of TP (Malik et al., 2013; Takahashi, 2013). We expected a higher P_{Ca} increase, however, given the high proportion of P_{Ca} in poultry manure (>66% of TP, Takahashi, 2013). The lack of increase in our study could be due to the dissolution of P_{Ca} into P_{resin}, as P_{resin} tripled in RR-I organic plots. Alternatively, high soil Mg levels and high Mg/Ca ratios in RR soils (E. Torbert, personal communication, 2013) suggest that Ca precipitation as dolomite [$CaMg(CO_3)_2$] could reduce soil solution Ca concentrations (Bui et al., 1990) and cause P to interact more strongly with Al and Fe compared with Ca, explaining the doubling in P_{AlFe} we observed. Ultimately, poultry manure led to high P accumulation rates in all fractions as a result of P application in excess of crop removal.

Using composted poultry manure to meet plant N requirements resulted in a surplus of 1484 kg P ha^{-1} relative to crop removal over 18 yr (82 kg P ha^{-1} yr^{-1}) in the organic system of RR-I (Table 4), consistent with what Pautler and Sims (2000) found when poultry manure was used to fertilize based on plant N demand. This excessive P application could be lowered by reducing poultry manure application rates and combining them with higher N/P nutrient sources, such as swine or cattle manure (Nelson and Janke, 2007), high N fertilizers (e.g., urea, feather meal) or N-fixation from legume cover crops (Tonitto et al., 2006). Otherwise, fertilizing exclusively with manure will increase soil P and increase risks of runoff and leaching (Eghball, 2002), especially when poultry manure is used (Sims et al., 2000).

The increase in P_{sat} from 6 to 12% in all RR systems that did not receive poultry manure to 39%

in the organic system of RR-I suggests that this system is above the P leaching threshold of 25 to 40% proposed by Pautler and Sims (2000) and that P leaching below the plow layer is likely in these soils. This is supported by a smaller accumulation of soil P in the top 30 cm than expected based on P budgets, and by Olsen P accumulation in the 30- to 60-cm layer, similar to the P accumulation observed at 30 to 60 cm by Eghball et al. (1996) or at 25 to 50 cm by Rubaek et al. (2013). If P surpluses are not reduced, P accumulation could continue and propagate deeper in the soil, increasing risks of P leaching to groundwater (Oehl et al., 2002). However, the smaller increase in 2003 to 2012 relative to 1993 to 2003 suggests that high manure application rates before 1999 (see *Material and Methods* above) may be the main driver of P accumulation below the plow layer. If so, lower manure application rates implemented in 1999 should reduce the rate of P accumulation and associated leaching risks.

Effects of Yard Compost on Phosphorus Dynamics

In contrast to composted poultry manure, yard compost had subtler effects on soil P pools, despite dry matter inputs that were twice as high as composted poultry manure inputs. P_{resin} was the only pool showing a significant increase, 16 mg P_{resin} kg^{-1} soil after 8 yr (Table 2), consistent with previous studies reporting limited changes in soil P with yard compost vs. manure (Hartz et al., 2000; Sinaj et al., 2002; Malik et al., 2013). We also found non-significant increases in P_{org} and P_{chl}, consistent with the low P_{org} content of yard compost (30% of TP on average, Frossard et al., 2002). However, we expected to see increases in P_{AlFe}, P_{Ca}, and $P_{residual}$ given the dominance of these P pools in yard composts (49–70% of TP, Frossard et al., 2002), and a substantial increase in TP given the P surpluses computed (Table 4). The lack of change we observed suggests that the soil P that should have accumulated in P_{AlFe}, P_{Ca}, or $P_{residual}$ might have been lost to leaching, although this would likely be influenced by cover crops (see "Effects of cover crops on Phosphorus dynamics" section below). Yard compost did not alter P sorption but P_{sat} was high everywhere (>50%) and above the 25 to 40% threshold defined by Pautler and Sims (2000). Given this high P_{sat} on sandy soils with a low P sorption capacity, significant P leaching could occur—Rubaek et al. (2013) estimate that up to 30 kg P ha^{-1} yr^{-1} can be leached in favorable conditions. Therefore, it is possible that we found weak effects of yard compost because of conditions favorable to P leaching in these P-rich, coarse-textured soils.

Effects of Cover Crops on Phosphorus Dynamics

Cover crops, regardless of system or mixture, had no effect on soil P dynamics, except for a slight increase in P_{chl} and P_{org} with legume-grass mixtures. This small increase is consistent with laboratory incubations (Malik et al., 2012) and field results following several years of cover cropping (Kuo et al., 2005; Takeda et al., 2009) where P_{chl} and P_{org} were the pools most affected by cover crops. While P_{chl} is considered labile due to its fast turnover (Oehl et al., 2001), P_{org} must be desorbed and mineralized before uptake and is therefore less available to plants (Frossard et al., 1995). Given that the increase in P_{org} was always several fold larger than the increase in P_{chl}, it is unlikely that P availability increased substantially with cover crops in these systems.

We found no evidence of P mobilization driven by cover crops, with no depletion in pools of lower plant availability (P_{Ca}, P_{AlFe}, $P_{residual}$) and no concomitant increase in P_{resin}. This differed from systems with low P inputs (mostly tropical) where legumes but not grasses mobilized soil P pools of lower plant availability and benefitted subsequent cash crops (Horst et al., 2001; Nuruzzaman et al., 2005). In contrast, cover crops had subtler effects in systems where P was more abundant (mostly temperate): Kuo et al. (2005) found no significant change in P_{Ca}, P_{AlFe}, or $P_{residual}$ after several years of cover cropping and Cavigelli and Thien (2003) found no improvement of sorghum P uptake after a legume cover crop. Hence, P mobilization driven by cover crops might be less important in systems where P is more abundant and P limitation is weaker.

In contrast to this lack of cover crop effect on soil P dynamics, cover-cropping with rye increased total P in the sandy SOCS soils, despite similar P budgets among systems receiving yard compost (Tables 2 and 4). Rye likely increased P retention capacity and potentially reduced P leaching in these coarse-textured, high P_{sat} soils, consistent with other studies reporting that rye and other grasses are often more efficient at reducing soil P losses than legumes or mustards (Kuo et al., 2005; Takeda et al., 2009). Mixtures of legumes and grasses might be as efficient at reducing P leaching as pure grasses (Sharpley and Smith, 1991), although this would likely vary as a function of grass and legume abundance, soil type, or other factors. Therefore, targeting cover crop species for P leaching rather than P mobilization might be more beneficial in systems with high soil P, such as in California, which could favor grasses over other cover crops.

Implications for Management

Using composted poultry manure with crop N requirement as the primary criterion for determining application rates resulted in P over-application. Farmers could reduce P over-fertilization and simultaneously meet plant P requirements by using composts and manures with higher N/P ratios, increasing N-fixation rates and using N-rich inputs. Otherwise, we would expect high soil P, P_{sat}, and leaching risks, and low P-use efficiency as observed here and in other studies (Sims et al., 2000).

In contrast, yard compost only increased P_{resin} significantly and had no measurable effect on P sorption, likely because of its lower P content. Stronger effects may be observed in systems that are less P saturated however (e.g., Sinaj et al., 2002), whereas high P leaching in coarse-textured soils may offset other benefits of yard compost, for example, higher C inputs to soils (Brennan and Boyd, 2012). Ultimately, the high P surplus and leaching risks resulting from yard compost use could be reduced with similar changes to management as those suggested for manure.

Our results suggest that the capacity to mobilize P is not important as a criterion for cover crop species choice in high P agricultural systems when compared with other factors (e.g., weed suppression).

In contrast, the greater potential of grasses to increase soil P retention and reduce leaching could trigger trade-offs with other cover crop benefits, for example, N-fixation inputs from legumes.

Finally, using inputs where N and P can be decoupled (e.g., mineral fertilizers) instead of inputs where N and P cannot be separated (e.g., composts) resulted in smaller P imbalances, lower leaching risk and higher P-use efficiency, similar to what Oehl et al. (2002) found after 21 yr of an organic vs. conventional comparison. However, P inputs with mineral fertilizers were often insufficient in RR, resulting in moderate P deficits and soil P depletion that threaten long-term soil fertility if they are not addressed via higher fertilizer addition rates. Overall, our results highlight the challenges in reducing or eliminating P inputs from mineral fertilizers while providing sufficient N and P during the growing season to maintain soil fertility and maximize yields while avoiding long-term overfertilization in systems with abundant soil P, such as in California.

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