Biomass Decomposition and Phosphorus Release from Residues of Cover Crops under No-Tillage

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ABSTRACT

Under no-tillage field conditions, cover crop (CC) residues remain on the soil surface and decompose at a slow rate. We used three CC species (oat, *Avena sativa* L.; rye, *Secale cereale* L.; and ryegrass, *Lolium multiflorum* L.) to evaluate the residue biomass decomposition and P pools release–total phosphorus (Pt), inorganic phosphorus (Pi), and organic phosphorus (Po)–under no-tillage field conditions. The dynamics of biomass and P in CC residues was evaluated through the litterbag method during two annual periods in a long-term field experiment. Exponential decay models were fitted to each variable. Biomass decomposition and P release followed asymptotic decay models for the less labile residues (oat and rye) and single exponential decay models for the more labile ryegrass residues. Biomass and Pt showed rather equivalent dynamics, but Pi release was faster (Pi rates were three times higher and Pi plateaus were two to four times lower than biomass values). The release of Pt from CC residues reached agronomically significant quantities (2–16 kg ha^{-1}) during the cash crop growing season and most of this Pt was released as Pi (53–100%). A positive tradeoff was observed between the release of Pi, potentially available for subsequent crop, and the residue biomass remaining over the soil. This means that Pi was released from residues, but enough biomass remained on the soil surface for erosion control. These results can be incorporated into predictive models on the contribution of CC residues to soil coverage and P nutrition of subsequent crops.

Core Ideas

- Residue decomposition and P release was studied under no tillage field conditions.
- Cover crop residues releases significant quantities of P for the subsequent cash crop.
- Phosphorus release from residues is affected by precipitation regime and residue quality.
- A positive tradeoff was observed between crop residue P dynamics and soil coverage.

THE INTEGRATION OF CC into agricultural systems prompts ecosystem services such as increased nutrient cycling and soil coverage, reduced erosion and nitrate leaching, and improved soil physical conditions for root growth (Snapp et al., 2005; Schipanski et al., 2014). These benefits may be affected if CC are combined with no-tillage farming (Blanco-Canqui et al., 2011). The adoption of no-tillage is increasing worldwide, and in many countries is the predominant form of tillage (Kassam et al., 2009; Derpsch et al., 2010; Peiretti and Dumanski, 2014). Under no-tillage, crop residues remain on the soil surface as mulch and tend to be decomposed at a slow rate (Lupwayi et al., 2004; Curtin et al., 2008). However, if nutrients are retained within the undecomposed residues, their availability for the subsequent crop may be compromised (Schomberg et al., 2006; Noack et al., 2014; Varela et al., 2014). The inclusion of CC has been recommended in the Pampean Region (Argentina) as a way to moderate the adverse effects of monoculture soybean, such as poor soil coverage, low soil organic C storage and poor soil aggregation (Novelli et al., 2011; Varela et al., 2011). Around 80% of the agricultural land of this Region is cultivated following no-tillage practices (Peiretti and Dumanski, 2014), and monoculture soybean is the prevalent cultivation system (Integrated Agricultural Information System, http://www.siia.gov.ar). Winter monocots such as oat, rye, and ryegrass are among the CC most employed by the local farmers to ameliorate soil deterioration (Fernández et al., 2012; Restovich et al., 2012; Varela et al., 2014).

The decomposition of plant residues is regulated by their chemical composition, the nature of the soil-residue contact, and by environmental conditions such as temperature and soil moisture (Parr and Papendick, 1978; Heal et al., 1997; Stubbs et al., 2009). Most studies have used single exponential decay models for describing this process (Schomberg and Steiner, 1999; Kalburtji and Mamolos, 2000; Murungu et al., 2011), although some studies have used asymptotic ones (Ranells and Wagger, 1996; Lupwayi et al., 2004). The asymptotic models provide a better fit when decomposition stops at a certain point (Wider and Lang, 1982). Decay models provide an estimation of the decay constant (k_B) that characterizes the loss of biomass and allows comparisons between different residues, treatments, and environmental conditions. Additionally, asymptotic models

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Abbreviations: CC, cover crops; Pi, inorganic phosphorus; Po, organic phosphorus; Pt, total phosphorus.

provide a value for the plateau, which estimates the biomass that remains undecomposed. These models could apply to winter CC residues whose decomposition is very fast in the first stage but then is expected to slow down during the following winter.

Nutrient release from residues is governed by the residue decay constant and by the concentration and chemical speciation of the nutrient in inorganic and organic forms (Noack et al., 2012). Most studies addressing the dynamics of nutrients in CC have focused on N (e.g., Meisinger et al., 1991; Thorup-Kristensen et al., 2003; Poffenbarger et al., 2015), whereas relatively fewer studies have focused on other nutrients such as P (e.g., Shammas et al., 2003; Lupwayi et al., 2007; Noack et al., 2014). The inclusion of CC into cropping systems may involve a substantial intervention in the cycling of P because CC take Pi from the soil solution and return it later on the season within residues as Pi and a mix of Po pools (Noack et al., 2012, 2014; Damon et al., 2014). Speciation of P in residues varies with the physiological age of the plant (Noack et al., 2012). In general, CC are killed (either mechanically or chemically) during their vegetative phase, when usually a high proportion of Pi is present (Ha et al., 2008; Noack et al., 2012; Maltais-Landry and Frossard, 2015). In contrast to Pi, residue Po must undergo biochemical reactions before becoming available for plants (Bieleski 1973). In this sense, it is commonly accepted that Pi is released rapidly in the early stages of the residue decomposition (Salas et al., 2003; Ha et al., 2008; Malik et al., 2012). Nutrient dynamics in plant residues can be analyzed through single or asymptotic exponential decay models such as the ones explained for biomass decomposition (e.g., Njunie et al., 2004, Damon et al., 2014). The Pt, Pi, and Po decay constants (k_{Pt}, k_{Pi}, and k_{Po}) can be derived from these decay functions and then incorporated in models to predict the contribution of residue P to the soil P availability (Damon et al., 2014).

Under no-tillage conditions, plant residues remain on the soil surface where they have restricted contact to the microbial biomass and the soil matrix (Henriksen and Breland, 2002; Noack et al., 2014), but are subjected to more frequent wetting and drying cycles, that may favor the Pi release in the absence of a microbial biomass (Jones and Bromfield, 1969). Most studies on the dynamics of residue P pools during decomposition were performed with ground and buried residues under controlled conditions (Damon et al., 2014). In contrast, Pi and Po dynamics within intact residues in no-tillage systems has deserved little attention so far. If the rate of Pi release from CC residues is higher than the rate of biomass decomposition (k_B), the inclusion of CC will favor the maintenance of high levels of soil coverage while a large proportion of the absorbed P will remain available for subsequent crops. The objective of this research was to compare the dynamics of biomass and P pools after residues of CC are placed on the soil under no-tillage field conditions.

In this study we quantified Pt, Po, and Pi dynamics but we did not deal specifically with the dynamics of the microbial biomass within residues (Ladd et al., 1995; Grace et al., 2006). We adopted a single P pool for the residues and the microbial biomass within residues (as proposed by Damon et al. [2014], considering that P pools from microbial biomass and residues decay at the same rates). This approach assumes that, as the microbial biomass grows, the residue Po increases, whereas P is released when the microorganisms die.

MATERIALS AND METHODS
Study Site and Experimental Design

The experiment was conducted on a long-term field experiment located at the General Villegas Experimental Station of the National Agricultural Technology Institute (INTA) (Buenos Aires, Argentina; 34°52′27.47″ S, 62°45′31.95″ W) in the western Pampas Region (Varela et al., 2014). The experiment started in 2005 to quantify the effects of winter CC within a continuous no-tillage soybean [*Glycine max* (L.) Merr.] cropping system. The CC were planted annually in the same plot since the beginning of the experiment (2005), between successive soybean crops. The design of the experiment is a randomized complete block with four replicates per treatment. The average historic annual precipitation is 906 mm. The soil is a Thapto-Argic Hapludoll with the following characteristics in the top 20 cm: 119, 409, and 473 g kg^{-1} of clay, silt, and sand, respectively, pH (soil/water ratio 1:2.5) 5.8, available P (Bray and Kurtz I) of 26 mg kg^{-1} and organic C content of 14 g kg^{-1}. The present study focused on the comparison of residue decomposition between three CC species: oat, rye, and ryegrass. A control treatment without CC was also included. Cover crops were sown in plots 50 m long by 5 m wide on 17 Apr. 2009 and 4 May 2010, fertilized with 46 kg N ha^{-1} as urea at tillering and desiccated with herbicide (glyphosate, N-phosphonomethyl glycine) at the end of October. At this time, the three CC had reached different growth stages (approximately Zadoks decimal code 5.0, 6.0, and 6.5 for ryegrass, oat, and rye, respectively). Since the treatments were applied to the plots for five consecutive years, the results of this study show the cumulative effects of the whole sequence (i.e., 2009/2010 data and 2010/2011 data represent the fifth and sixth year accumulative effects). Biomass production (kg ha^{-1}) was determined before chemical desiccation. Then, soybean was sown and 50 kg ha^{-1} of diammonium phosphate (9 kg N ha^{-1}; 10 kg P ha^{-1}) was applied.

Daily precipitation and mean air temperature 1.5 m above the soil surface were recorded using a data-logger (weather station, SIAP+MICROS Supervis Versión 5.12.5, Argentina). Degree days were calculated for CC growth and decomposition periods using 0°C as base temperature (Quemada, 2004).

Initial Quality of Cover Crop Residues and Litterbag Preparation

Standing CC residues of each of the three species were collected immediately before the annual herbicide application. Residues were collected from plants selected at random until 1 kg of residues was obtained from each one of three plots for each CC species (three species × three plots = nine samples per year). Residues were separated into leaves, stems, and spikes. The proportion between these components was taken into account when residue samples of each CC were prepared for the chemical analyses and for preparing the litterbags (see next section). Fifty grams of residues of each CC were taken each year to determine their initial chemical quality. Samples were oven-dried at 60°C for 48 h and ground to pass a 0.5-mm sieve. Total C and N concentration were determined with an automatic combustion analyzer (LECO CN Analyzer, LECO, St. Joseph, MI). Lignin and cellulose concentration were determined by proximate analysis, using the detergent method of fiber fractionation (Goering and Van Soest, 1970). Total P was determined on dry

ashed residues (550°C) that were resuspended in HCl (2 M). Inorganic soluble P was extracted after shaking (200 rpm) 1 g of the residue material with 20 mL of cold (5°C) deionized water for 1 h and then filtered. Total and inorganic P content were quantified spectrophotometrically (Murphy and Riley, 1962). The Po content was calculated by difference between Pt and Pi (Jones and Bromfield, 1969).

Cover Crops Residue Decomposition and Phosphorus Release

The decomposition of CC residues was evaluated using the litterbag technique (Verhoef, 1995; Lupwayi et al., 2007). The size of the plastic litterbag was 0.15 by 0.20 m with a mesh size of 2 mm. For each CC species, residues from three plots (see previous section) were pooled and cut into 5 cm-long segments before filling the litterbags. In each litterbag, 12 to 14 g of ryegrass residues and 18 to 20 g of rye or oat residues were placed. The amounts of residue placed in litterbags of each treatment were proportional to the average dry matter production. Sixteen litterbags of each CC were randomly assigned to each plot and placed on the soil surface on 29 November in 2009 and on 1 December in 2010 (i.e., the decomposition experiment was conducted during two different years, on the same experimental plots). Macro-fauna can damage litterbags in the field and extra litterbags were placed in each plot to ensure that two undamaged bags could be collected at each sampling date. Gravimetric moisture and ash content were determined on additional litterbags to calculate the initial ash-free dry weight for each litterbag. In the 2009–2010 decomposition period, the bags were collected 31, 71, 103, 141, 171, 295, and 337 d after they were placed on the soil. In the 2010–2011 decomposition period, they were collected 77, 128, 219, 289, and 335 d after placed in the soil. Two litterbags were taken for each plot in each sampling date to individually determine the remaining biomass and P pools. These results were averaged to represent the value of each plot. Once collected, litterbags were stored at 5°C until laboratory analysis. Then, the residues were removed from the litterbag, shaken gently over a sieve (1 mm) to remove the soil and dried at 60°C for 48 h to determine the weight of the dry biomass. Samples were ground to pass a 0.5-mm sieve and subsamples (0.1 g) were ashed in a muffle furnace at 550°C for 16 h. The ash content was used to adjust the sample dry weights to an ash-free dry weight basis. This accounted for variable quantities of contaminating soil. Total P, Pi, and Po content of each litterbag was determined as described for initial residue assessments.

The accumulated residue biomass covering the soil surface (undecomposed soybean and CC residues from previous years, including some dying or dead CC biomass from the current year) was also determined at the beginning of the CC growth period (2009 and 2010) and at the end of the CC growth period (2010 and 2011). Two subsamples of lying residues were taken per plot through 625 cm² quadrants, shaken over a sieve (1 mm) to remove the soil and then dried at 60°C for 48 h to determine dry weight biomass.

Data and Statistical Analysis

Experimental data of remaining biomass or remaining P (Pt, Pi, and Po) vs. days were expressed as percentage of initial biomass or P and fitted to a (i) single exponential and (ii) asymptotic decay model for biomass, Pt, Pi, and Po for each period (Wider and Lang, 1982):

$$y = 100 \times \exp\{-\kappa\, x\} + \varepsilon \qquad [i]$$

$$y = \beta + (100 - \beta) \times \exp\{-\kappa\, x\} + \varepsilon \qquad [ii]$$

where y was the response variable, β represented a plateau (expressed as a percent), κ was the decay constant (d^{-1}), x was time expressed as days, and ε was the random error term. The estimations of these models were obtained by the least squares method and they were expressed as b (estimators of β, percent) and k (estimators of κ, d^{-1}). Decay constant were indicated as k_B, k_{Pt}, k_{Pi}, and k_{Po} for biomass, total P, Pi, and Po data, respectively.

To test whether remaining biomass or P data were most consistent with asymptotic model relative to single exponential decay model, an extra sum-of-squares F test was performed for each data set ($p < 0.05$) (Archontoulis and Miguez, 2015). In those cases where the Po content increased at the beginning of the decomposition process, indicating gross P immobilization, an exponential decay model could not be fitted since the start point of decomposition. Then, the exponential decay model was fitted starting from the highest value of each CC species.

To evaluate if a decomposition model presents differences due to treatment effect (oat, rye, and ryegrass), a full model including treatment effect was fitted (Schabenberger, 2009):

$$y_i = \beta_i + (100 - \beta_i) \times \exp\{-\kappa_i\, x\} + \varepsilon \qquad [iii]$$

where i identifies each treatment. Full (iii) and reduced (ii) models were compared by the extra sum-of-squares F test (Archontoulis and Miguez, 2015) to evaluate if there was a treatment effect. In this case, the model was re-parameterized by dummy variables to run the model, the value of the parameters for one treatment and the differences in the values of each parameter for other treatments were estimated. If these differences were not significant, no individual parameter was estimated for each treatment. If these differences were significant, individual parameters for treatments that differed were estimated. The model explaining the highest variability with the fewest parameters was selected (Schabenberger, 2009).

Analyses were performed with the nlme (Pinheiro et al., 2015) and nlstools (Baty et al., 2015) packages in R (version 3.2.2, 2015). Parameters of nonlinear models (k and plateau) were compared for remaining biomass and P pools data sets by the mean of the confidence intervals.

The parameters describing the initial quality and quantity of residues were compared between treatments (species) and experimental years by linear models using generalized least squares. Cover crops species and years were considered fixed effects. Heterocedasticity between treatments and years was modeled using different options for the structure for the variance–covariance matrix of the residuals (Littell et al., 2000). The model with the smaller value of Akaike's information criterion (AIC) and Bayesian information criterion (BIC) was selected. Means of each level were compared using the least significant difference LSD test ($p < 0.05$). Analyses were performed using R and the InfoStat software (Di Rienzo et al., 2015).

RESULTS
Weather Conditions during Cover Crop Growing and Cover Crop Decomposition Periods

The precipitation regime during the CC growth period showed little variation between years: 261 and 217 mm in 2009 and 2010, respectively (Fig. 1). However, the two periods differed in the amount of precipitation during the summer prior to CC sowing (334 mm in 2009 and 539 mm in 2010) (Fig. 1). The decomposition of CC residues was evaluated annually, starting in December, approximately 1 mo after the chemical desiccation (end of October 2009 and 2010). Temperature conditions were rather similar during both decomposition periods (5409 and 5482 cumulated degree days in 2009–2010 and 2010–2011, respectively). In contrast, the precipitation regime had a great inter-annual variation: the 2009–2010 season accumulated 898 mm, while the 2010–2011 season accumulated only 501 mm (Fig. 1). These figures indicate that the weather conditions were more favorable for decomposition during 2009–2010.

Biomass Production and Initial Quality of Residues

The biomass production of the three CC was lower in 2009 than in 2010, but they ranked similarly in both years: rye = oat > ryegrass (Fig. 2a and 2b). Ryegrass only produced an equivalent to 31 and 64% of the average aerial biomass of rye and oat in 2009 and 2010, respectively (Fig. 2a and 2b). At the moment of chemical desiccation, rye was the species with the highest concentration of structural polymers (lignin and cellulose) and C/N ratio (Table 1). Ryegrass had the highest residue N and P concentration (Table 1) and showed a lower C/P ratio than oat, whereas differences with rye varied between years (Table 1).

Phosphorus in Cover Crop Residues

The amount of Pt in residues at the time of chemical desiccation varied between species and experimental years (Fig. 2c and 2d). In 2009, it averaged 7.8 kg Pt ha⁻¹ in oat and

Table 1. Chemical composition of three cover crop (CC) species (oat, rye, and ryegrass) at the end of two consecutive growing periods (2009 and 2010). Different letters in each row indicate significant differences at the 5% level according to LSD mean comparison. When significant interaction "species × time" was detected, mean comparison between species was performed for each year (uppercase letters). When the interaction term was not significant, LSD comparison between species averaged across years was performed (lowercase letters in the inter-row position).

| Residues properties	Year	CC species				
		Oat		Rye		Ryegrass
Total C, g kg⁻¹	2009	435		434		425
	2010	425		443		406
Lignin, g kg⁻¹	2009	31.67		44.00		35.67
	2010	38.67	b	48.67	a	28.33	b
Cellulose, g kg⁻¹	2009	242		279		230
	2010	287	ab	299	a	265	b
Total N, g kg⁻¹	2009	14.3B		12.42C		29.85A
	2010	18.33B		15.62C		20.9A
C/N ratio	2009	30.43B		34.94A		14.24C
	2010	23.26B		28.4A		19.62B
Total P, g kg⁻¹	2009	1.64B		2.08A		2.13A
	2010	3.1B		2.89B		4.79A
C/P ratio	2009	268A		209B		200B
	2010	137B		154A		85C
Pi, g kg⁻¹†	2009	1.12C		1.44A		1.28B
	2010	1.25B		1.3B		2.17A
Pi/Pt ratio	2009	0.69		0.69		0.60
	2010	0.40		0.45		0.45

† Pi, inorganic phosphorus; Pt, total phosphorus.
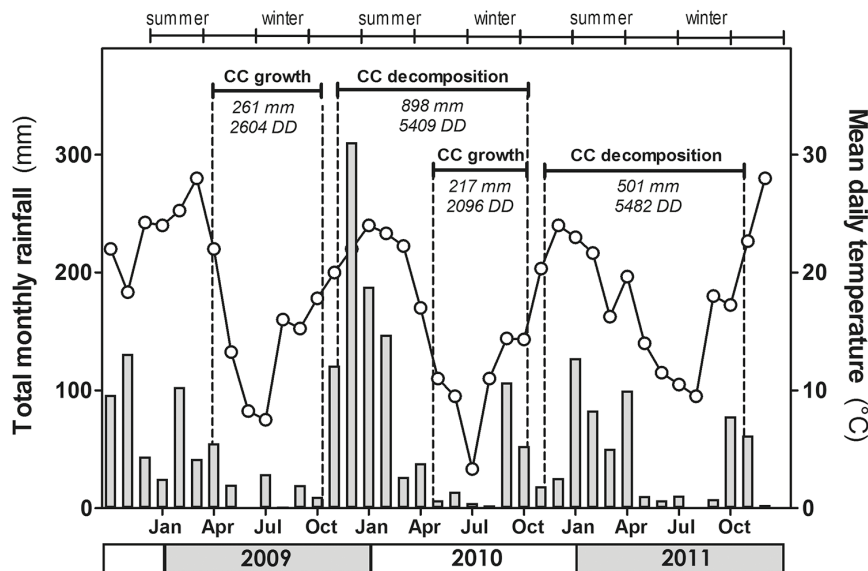
Fig. 1. Total monthly rainfall (bars) and daily average temperature (line + open circles) during the evaluated period. Cover crop (CC) growth periods (2009 and 2010) and residue CC decomposition (2009–2010 and 2010–2011) are indicated by horizontal intervals. Cumulative rainfall (mm) and degree days (DD) are included for each period.

rye, but only 3.0 kg Pt ha^{-1} in ryegrass (Fig. 2c). In 2010, the overall higher biomass production and higher P concentration (Table 1) determined a higher amount of P in the residues, averaging 32.2 kg Pt ha^{-1} without significant differences between species (Fig. 2d). The ratio Pi/Pt of the residues showed great inter-annual but no interspecific differences (Table 1). In 2009, it ranged 0.6 to 0.7 and dropped to 0.4 in 2010. However, despite a lower Pi/Pt ratio, Pi content of the standing biomass was higher in 2010 due to the increased biomass production and residue P concentration (14 kg Pi ha^{-1} average between CC; Fig. 2f).

Dynamics of Residue Decomposition

The decomposition of CC residues could be significantly explained by asymptotic decay models except for ryegrass in 2009–2010. This crop decomposed almost completely; therefore a single exponential decay model (i.e., with a zero plateau) was fitted (Fig. 2a, Table 2). The k_B was similar between CC species, but plateau values differed between them (Table 2). In the rainy 2009–2010 period, ryegrass residues decomposed almost completely (Fig. 2a, Table 2), while oat and rye reached a plateau at 15 and 38% of remaining biomass, respectively (Fig. 2a, Table 2). In contrast, in the drier 2010–2011 period, the three species showed lower decomposition percentages. In this second period ryegrass reached a plateau at 25% of the initial biomass, whereas oat and rye had a plateau at 50% (Fig. 2b, Table 2). These percentages mean that absolute values of residue biomass covering the soil after a year of decomposition were 0, 0.8 and 1.6 t ha^{-1} in 2009–2010 and 1.7, 5.4 and 5.3 t ha^{-1} in 2010–2011 for ryegrass, oat, and rye, respectively.
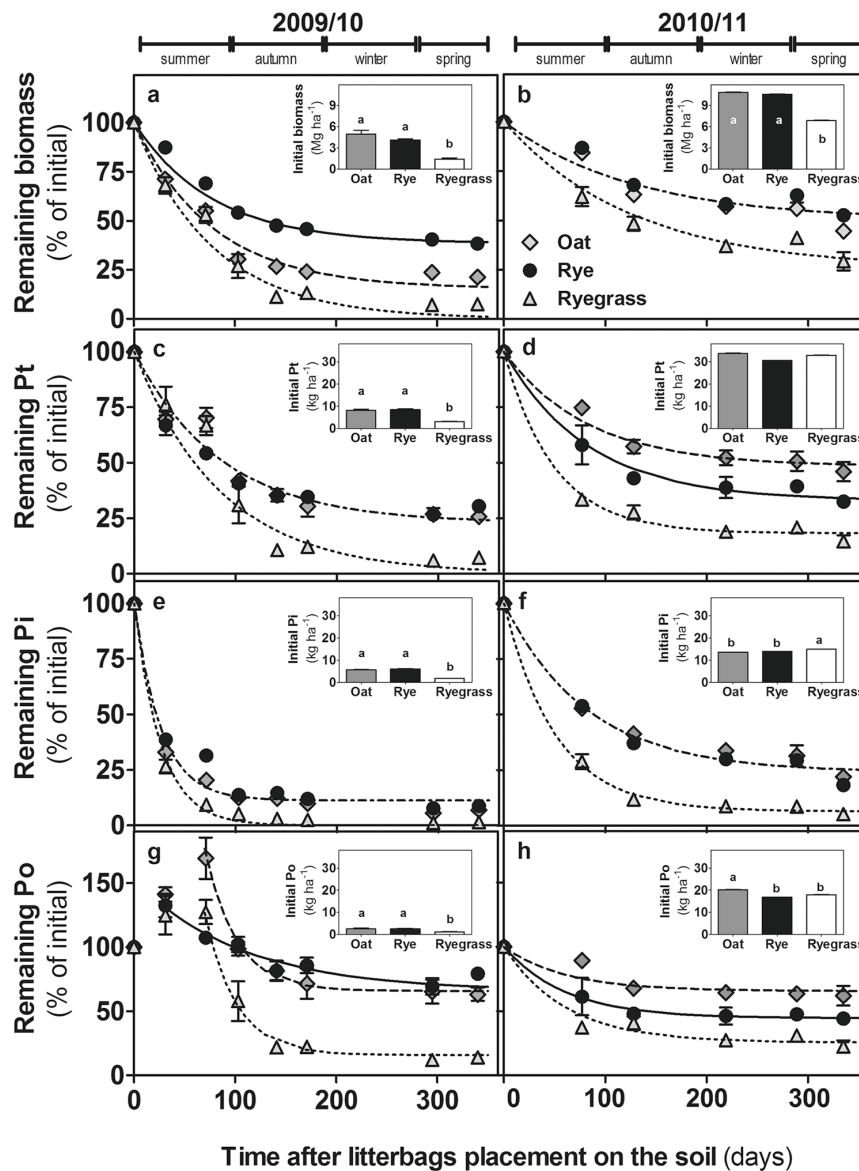

Fig. 2. (a and b) Remaining biomass, (c and d) total phosphorus (Pt), (e and f) inorganic phosphorus (Pi) and (g and h) organic phosphorus (Po) content of residues of three cover crops species (oat, rye, and ryegrass) after litterbag placement on the soil for both experimental periods (2009–2010 and 2010–2011). Data points represent the average of four replicates and vertical bars the standard error. Lines show decay models fitted for each variable (continuous lines for rye, dashed lines for oat, dotted lines for ryegrass and dash-dot lines for pooled oat and rye). Data for Po in 2009–2010 were fitted to decay model starting from the largest value. Scale of real seasons (summer, autumn, winter, and spring) is also provided. The small graphics inside each panel show the initial values of biomass, Pt, Pi, and Po for each species. In these graphs, different letters indicate significant differences at the 5% level according to LSD mean comparison.

Table 2. Parameters from the decay models estimated for remaining biomass, total phosphorus (Pt), inorganic phosphorus (Pi), and organic phosphorus (Po) in residues of oat, rye, and ryegrass in two experimental years. Parameter k is the relative rate of biomass decomposition (k_B) or of total P (k_{Pt}) or P pool (k_{Pi} or k_{Po}) release. Plateau is the amount of biomass or P that was not decomposed or released from residues after 1 yr in the soil. Graphical representations of k and plateau are shown in Fig. 2. Full model implies one parameter for each cover crop (CC) species and reduced model shows parameters shared between CC species after F test (α level: 0.05).

Year and parameters	CC species	Full model		Reduced model		
		k	Plateau	k	Plateau	R^2
2009–2010						
Biomass	Oat	0.0149	19.7		15.4	
	Rye	0.0105	35.8	0.0121	38.4	0.98
	Ryegrass	0.0126	3.2		0.0	
Pt	Oat	0.0116	23.0		22.9	
	Rye	0.0169	29.1	0.0117		0.94
	Ryegrass	0.0107	1.6		0.0	
Pi	Oat	0.0398	9.6		11.3	
	Rye	0.0309	11.3	0.0385		0.98
	Ryegrass	0.0438	2.7		0.0	
Po	Rye	0.0127	72.9	0.0095	65.8	
	Oat	0.0324	65.9	0.0325		0.99
	Ryegrass	0.0307	13.2		15.8	
2010–2011						
Biomass	Oat	0.0052	37.0		50.1	
	Rye	0.0057	46.8	0.0079		0.95
	Ryegrass	0.0108	31.9		25.3	
Pt	Oat	0.0095	44.9		48.4	
	Rye	0.0142	35.0	0.0121	32.9	0.99
	Ryegrass	0.0251	18.2	0.0203	18.2	
Pi	Oat	0.0125	26.1	0.0121	24.1	
	Rye	0.0118	22.1			0.99
	Ryegrass	0.0194	6.4	0.0196	6.4	
Po	Oat	0.0070	56.8		65.7	
	Rye	0.0173	45.0	0.0183	44.6	0.96
	Ryegrass	0.0218	28.2		25.6	

Differences between CC species were also evident in the quantification of residue biomass accumulated on the soil surface after 6 yr of CC cultivation. Oat and rye accumulated significantly higher biomass of residues than ryegrass (8155 and 9209 kg ha^{-1} for oat and rye, respectively vs. 5612 kg ha^{-1} for ryegrass) ($p < 0.001$).

Dynamics of Total Phosphorus during Residue Decomposition

Total P release from CC residues followed asymptotic models, except for ryegrass in 2009–2010, which followed a single exponential decay model (Fig. 2c and 2d, Table 2). Total P was released at the same rate (k_{Pt}) from residues of three CC in 2009–2010 but it was released completely in ryegrass, while oat and rye residues retained 23% of their initial Pt content (1.8 kg P ha^{-1}) (Fig. 2c, Table 2). In the 2010–2011 period, the k_{Pt} from ryegrass residues was higher than for oat and rye residues, and plateau ranking was oat > rye > ryegrass (Fig. 2d, Table 2).

Biomass decomposition and Pt release generally occurred at similar rates and reached similar plateau values (Fig. 3). The exceptions were ryegrass residues in 2010–2011, in which k_{Pt} was faster than k_B (Fig. 3b), and rye residues in both years, in which plateau for biomass was higher than for Pt (Fig. 3c and 3d).

Dynamics of Inorganic Phosphorus during Residue Decomposition

The release of Pi from residues followed asymptotic models, except for ryegrass in 2009–2010, which followed a single exponential decay model (Fig. 2e and 2f, Table 2). As occurred with Pt, the release of Pi was described by the same k_{Pi} for the three CC species in 2009–2010 but this rate was higher for ryegrass than for oat and rye in 2010–2011 (Fig. 2e and 2f, Table 2). The plateau reached for Pi release was lower for ryegrass than for oat and rye. Ryegrass, the more labile species, released all Pi in the rainy 2009–2010 period but retained 6% of the initial Pi in the dry 2010–2011 period (Fig. 2e and 2f, Table 2). Oat and rye, however, retained 11 and 24% of the initial Pi content in 2009–2010 and 2010–2011, respectively.

In the rainy 2009–2010 season, the release of Pi occurred at a rate three times higher than the biomass decomposition and Pt release (Fig. 3a). In the dry 2010–2011 season, however, there were no differences between k_{Pi} and k_{Pt}, and only ryegrass showed $k_{Pi} > k_B$ (Fig. 3b). The plateaus for Pi were generally lower than for Pt and always lower than for biomass (Fig. 3c and 3d).
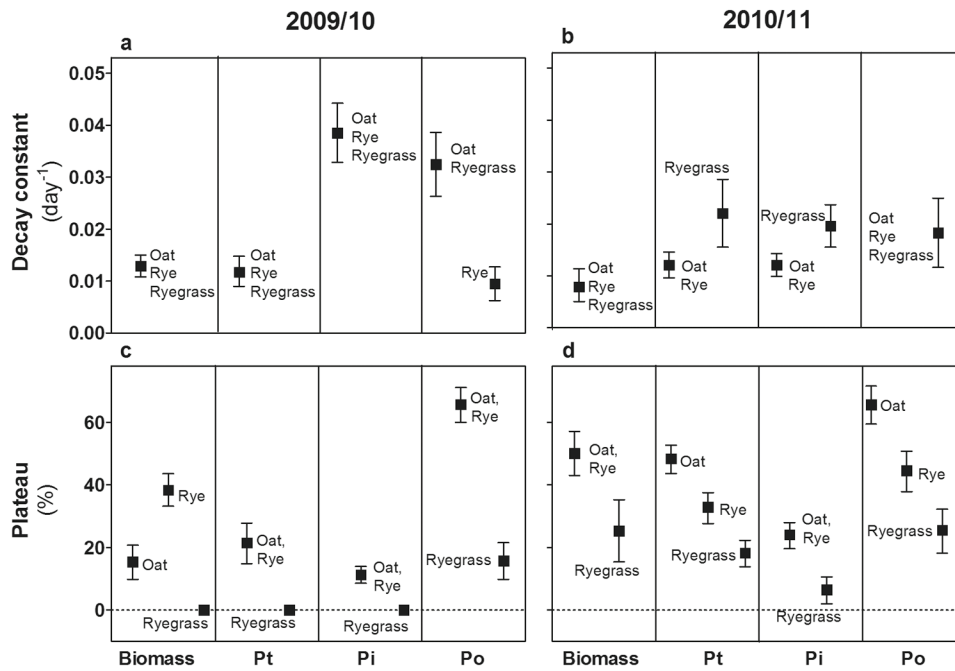
Fig. 3. (a and b) Decay constants (*k*) and (c and d) plateaus for the fitted decay models of biomass, total phosphorus (Pt), inorganic phosphorus (Pi) and organic phosphorus (Po) of CC residues during two decomposition periods (2009–2010 and 2010–2011). When *k* or plateau values did not differ between two or three species, they were pooled and represented by only one symbol. Vertical bars represent the 95% confidence interval.

Dynamics of Organic Phosphorus during Residue Decomposition

In 2009–2010, the three CC species showed an initial accumulation of the residue Po, reaching values of 127 to 169% of the initial Po content (Fig. 2g). This initial increase was followed by an asymptotic decay phase (Fig. 2g), which started after residue Po content reached a peak (31 d for rye and 71 d for oat and ryegrass). The k_{Po} for oat and ryegrass was more than three times higher than for rye in the 2009–2010 period, but the plateau value of oat and rye was four times higher than ryegrass (Fig. 2g, Table 2). In contrast, no initial peak in the residue Po was observed at the beginning of the 2010–2011 period. Instead, a continuous asymptotic function could be adjusted for this period (Fig. 2h, Table 2). During the 2010–2011 period, the release of Po occurred at the same rate (k_{Po}) for three CC species (Table 2). In this period, Po content reached a plateau at 66, 45, and 26% of the initial Po for oat, rye, and ryegrass, respectively. The amount of Po retained in CC residues after 1 yr in the soil ranged between 0.2 to 1.7 kg Po ha^{-1} and 4.6 to 13.3 kg Po ha^{-1} in 2009–2010 and 2010–2011, respectively.

In 2009–2010, k_{Po} values for oat and ryegrass were almost three times higher than k_B and k_{Pt} (Fig. 3a). For rye residues, in contrast, k_{Po} was lower than k_{Pi} and similar to k_B and k_{Pt} (Fig. 3a). Plateaus reached for Po in 2009–2010 were higher than those for biomass, Pt and Pi (for all species) (Fig. 3c). In 2010–2011, k_{Po} of the three CC was higher than k_B, but did not differ from k_{Pt} and k_{Pi} (Fig. 3b). The plateaus observed for Po were higher than the plateaus for Pi in all cases, but the relationship between Po and biomass or Pt differed between species (Fig. 3d).

DISCUSSION

In spite of the inter-annual weather differences, in both seasons (2009–2010 and 2010–2011) the CC species followed the same rank of biomass production and decomposition: rye and oat produced more biomass and their residues decomposed at a slower rate than ryegrass (Fig. 2a and 2b, Fig. 3c and 3d). In line with this rank, the percentage of soil covered by residues (which includes the accumulated effect of 6 yr of the CC–soybean sequence) was 52% higher in the oat and rye plots than in the ryegrass plots. Overall, these results suggest that oat and rye arise as better CC than ryegrass for soil conservation purposes in the western Pampas.

Residue biomass and Pt dynamics followed a rather similar decay pattern (Fig. 2a, 2b, 2c, and 2d), which is consistent with previous studies (Baggie et al., 2005; Bünemann et al., 2004; Jalali and Ranjbar, 2009). Our data of biomass and Pt residue dynamics generally fitted to asymptotic models, agreeing to previous works under no-tillage field conditions and long experimental periods as our study (e.g., 7 mo, Luna-Orea et al., 1996; 12 mo, Lupwayi et al., 2007). The plateaus of biomass and Pt (Fig. 2a, 2b, 2c, and 2d) were associated to the interruption of residue decomposition during the cold winter, approximately 7 mo after litterbags were placed on the soil (Fig. 1). It would be expected that decomposition resumes in the following warm and rainy season.

The magnitude of P release was affected by the type and amount of residues and by the environmental conditions during the decomposition period, as reflected by the interspecific and inter-annual differences in Pt release decay constant and plateaus (Fig. 3). Ryegrass was the less productive (Fig. 2a and 2b) and less mature species at desiccating time among the three evaluated CC and their residues had lower concentration of structural polymers (lignin and cellulose) and higher N and P contents compared

to oat and rye (Table 1). No differences were detected between oat and rye in biomass accumulation (Fig. 2a and 2b), but oat residues had higher N content and lower lignin content than rye (Table 2). The interspecific differences resulted in different plateaus in the fitted functions of biomass and Pt dynamics, but these differences were only large enough to modify the k_P value in the second growing season (Table 2, Fig. 3). These results indicate that ryegrass was the species with the highest biomass decomposition and P release rates and that the differences between oat and rye were not consistent. Similar relationships between residue chemical properties, biomass decomposition and nutrient release was observed by Kwabiah et al. (2003), Ha et al. (2008), and Zhang et al. (2008), who observed that residues with high lignin content, high C/N ratio and low total N and total P contents decompose slower and release less P than those with less lignin, less C/N ratio or high total N and total P content.

Obtained k_{Pi} values were generally higher than k_B values (Fig. 3a and 3b), whereas plateaus for Pi were generally lower than those found for biomass (Fig. 3c and 3d). This denotes a faster release of Pi than biomass decomposition, which implies a positive tradeoff between the amount of P released from the residues and potentially available to the subsequent crop and the amount of residue remaining for soil coverage. Differences between biomass and Pi dynamics were more pronounced in the 2009–2010 period, when residues had a higher proportion of Pi (Table 1) and the precipitation was higher (Fig. 1), suggesting that both initial residue quality and moisture conditions may regulate these processes under no-tillage field conditions.

At the beginning of the decomposition period, between 40 and 69% of residue Pt was Pi, without interspecific differences (Table 1). Even after 1 yr on the soil, a relevant proportion of P remained in the residues as Pi (16–34%) (Fig. 2c, 2d, 2e, and 2f). These findings contrast with previous laboratory studies that observed a rapid release of virtually all Pi present in residues (Kwabiah et al., 2003; Salas et al., 2003; Malik et al., 2012). These studies have generally identified leaching as the main factor for the rapid Pi release from residues and indicated that this process was independent of the utilization of residue C by microbial biomass (Kwabiah et al., 2003; Salas et al., 2003; Jalali and Ranjbar, 2009). Our results show that Pi release was higher in ryegrass, the more labile residue, than in oat and rye (plateau lower than oat and rye in 2009–2010 and k_{Pi} higher than oat and rye in 2010–2011, Table 2), suggesting that Pi release would require some previous rupture of C structures. In contrast to most previous reports performed with finely grounded and buried residues, our experiments were done with largely intact residues placed on top of the soil. Under these no-tillage conditions the interaction between residues, soil matrix, and soil microbiota is expected to occur slowly (Friesen and Blair, 1988; Guérif et al., 2001; Noack et al., 2014) which may delay residue degradation and thus, slow down Pi release. The relative slow rate of release and the high proportion of Pi in the residues at the end of the decomposition periods suggest that, under no-tillage field conditions, CC residues may potentially affect soil P availability for longer periods than those frequently employed in laboratory or greenhouse experiments. Undoubtedly, the extrapolation between experimental approaches (e.g., incubation/field conditions; buried/non buried residues; grounded/intact residues) should be made with caution because the different conditions may alter the factors that regulate residue decomposition and P release.

It is generally accepted that residue C/P ratios greater than 300 or P concentrations below 2 g P kg^{-1} are associated with net P immobilization in this residue (e.g., Iyamuremye and Dick, 1996; Kwabiah et al., 2003). Our results are consistent with these predictions since residue C/P ratios (range 85–268, Table 1) were mainly below and P concentrations (see range below) were mainly above threshold values and the fitted Pt exponential decay curves (Fig. 2c and 2d), showing that Pt content was not increased during the decomposition, indicated no net P immobilization. However, at the beginning of the 2009–2010 period, a sharp increase in residue Po was detected (Fig. 2g), which would be related to the microbial activity, taking soil or residue Pi and increasing the residue Po pool (Ladd et al., 1995; Grace et al., 2006). Since the observed peak in Po was lower than the release of Pi during the same period, it was not translated into net P immobilization. In this 2009–2010 period, residue P concentrations were close to the above mentioned threshold of 2 g P kg^{-1} (1.6, 2.1 and 2.1 g P kg^{-1} for oat, rye, and ryegrass, respectively) (Table 1), which may have triggered the observed increase in residue Po. In contrast, residues of the 2010–2011 period were rich enough in P (3.1, 2.9 and 4.8 g P kg^{-1} for oat, rye, and ryegrass, respectively; Table 1) which prevented any gross P immobilization.

Most studies quantifying P dynamics within crop residues have focused on total P without discriminating between different P pools (Luna-Orea et al., 1996; Salas et al., 2003; Lupwayi et al., 2007). Damon et al. (2014) proposed an empirical model to characterize the P release from residues and highlighted the key role of k_{Pi} and k_{Po} and the need for field data on these rates. Our k_{Pi} values, obtained in the field (0.038 and 0.016 d^{-1} on average between CC species for the humid 2009–2010 and the dry 2010–2011, respectively) are in the order of the empirical values proposed by Damon et al. (2014) (0.041 and 0.023 d^{-1} for optimum and suboptimal soil moisture conditions, respectively). However, our field k_{Po} values (0.021 and 0.0183 d^{-1} for the first and second year, respectively) were one order of magnitude higher than those proposed by the cited model (0.003 and vs. 0.001 d^{-1} for high and low soil moisture conditions, respectively). The discrepancy between the k_{Po} empirical values proposed by Damon et al. (2014) and our field ones may be related to: (i) the fact that many previous studies (reviewed by Damon et al., 2014) have measured residue Pt and then estimated k_{Pi} and k_{Po} from the dynamics of soluble and recalcitrant C (Jalali and Ranjbar, 2009; Damon et al., 2014); and (ii) most previous studies were performed with buried residues, in which Pi and Po dynamics may differ from what occurs under no-tillage conditions. On the other hand, the high k_{Po} observed for oat and ryegrass 2009–2010 (Fig. 3a) describes the rapid mineralization of the Po immobilized at the beginning of the decomposition period (Fig. 2g), which would be composed of labile Po forms within the microbial biomass.

In general terms, our findings suggest that both the amount of P annually released (up to 27 kg ha^{-1}) or sequestered (up to 16 kg ha^{-1}) by CC residues under no-tillage conditions may be agronomically considerable. Most P released from residues would have enriched the soil solution and eventually been taken up by growing microorganisms or plants (Damon et al., 2014; Maltais-Landry and Frossard, 2015). Following this assumption, the apparent maximum contribution of the CC aerial residue P to the subsequent soybean crop in the analyzed CC/soybean rotation can be calculated from the fitted decay curves. In

2009–2010, Pt released during the soybean critical period (0–90 d after sowing; Egli, 1998) was 4.5, 4.7, and 1.7 kg ha^{-1} for oat, rye, and ryegrass, respectively (Fig. 2c and 2e), and nearly 100% of this amount corresponded to Pi. In 2010–2011, Pt released from residues was larger compared to the first period (15.9 kg ha^{-1}) on average between CC species, and about 53% of this Pt was Pi. The amounts of released Pt were equivalent to 17% of soybean P demand, considering a P requirement (i.e., uptake) of 7 kg P t grain^{-1} (IPNI, 2012) and the soybean yield of this period (3 t ha^{-1}). In 2010–2011, released Pt was equivalent to 67% of soybean P demand (yield for this period was 3.4 t ha^{-1}). Future studies are needed to identify how much of released residue P is finally incorporated to the subsequent crop.

CONCLUSIONS

Obtained results cover a current knowledge gap by offering evidence on residue decomposition and P pools release under no-tillage field conditions. The release rates of Pt and biomass decomposition were similar among CC residues, and both processes were synchronic. In contrast, the release of Pi was partially decoupled from biomass decomposition either through a higher decay constant or a lower plateau than biomass. This indicates a positive tradeoff between the P forms potentially available for subsequent crops and the soil coverage provided by CC residues. The release of P from CC during the subsequent cash crop growing season may reach agronomically significant quantities (2–16 kg P ha^{-1} in our study). Our research highlights the importance of residue quality and environmental conditions on the dynamics of residue Pi under no-tillage field conditions. This evidence can be incorporated into predictive models on the contribution of CC residues to soil coverage and P nutrition of subsequent crops. The findings of this study also support the idea that the incorporation of CC into crop rotations should be analyzed as a soil conservation practice but also as a tool for P management. Cover crop residues would make a more relevant contribution than residues from grain crops, which retranslocate a larger proportion of absorbed P to the grain removed from the field.

REFERENCES

Archontoulis, S.V., and F.E. Miguez. 2015. Nonlinear regression models and applications in agricultural research. Agron. J. 107:786–798. doi:10.2134/agronj2012.0506

Baggie, I., D.L. Rowell, J.S. Robinson, and G.P. Warren. 2005. Decomposition and phosphorus release from organic residues as affected by residue quality and added inorganic phosphorus. Agrofor. Syst. 63(2):125–131. doi:10.1007/s10457-004-5131-5

Baty, F., C. Ritz, S.C. Martin Brutsche, J.P. Flandrois, and M.L. Delignette-Muller. 2015. A toolbox for nonlinear regression in R: The package nlstools. J. Stat. Softw. 66:1–21. doi:10.18637/jss.v066.i05

Bieleski, R.L. 1973. Phosphate pools, phosphate transport, and phosphate availability. Annu. Rev. Plant Physiol. 24:225–252. doi:10.1146/annurev.pp.24.060173.001301

Blanco-Canqui, H., M.M. Mikha, D.R. Presley, and M.M. Claassen. 2011. Addition of cover crops enhances no-till potential for improving soil physical properties. Soil Sci. Soc. Am. J. 75:1471–1482. doi:10.2136/sssaj2010.0430

Bünemann, E., D.A. Bossio, P.C. Smithson, E. Frossard, and A. Oberson. 2004. Microbial community composition and substrate use in a highly weathered soil as affected by crop rotation and P fertilization. Soil Biol. Biochem. 36:889–901. doi:10.1016/j.soilbio.2004.02.002

Curtin, D., G.S. Francis, and F.M. McCallum. 2008. Decomposition rate of cereal straw as affected by soil placement. Soil Res. 46:152–160. doi:10.1071/SR07085

Damon, P.M., B. Bowden, T. Rose, and Z. Rengel. 2014. Crop residue contributions to phosphorus pools in agricultural soils: A review. Soil Biol. Biochem. 74:127–137. doi:10.1016/j.soilbio.2014.03.003

Derpsch, R., T. Friedrich, A. Kassam, and H. Li. 2010. Current status of adoption of no-till farming in the world and some of its main benefits. Int. J. Agric. Biol. Eng. 3:1–26. doi:10.3965/j.issn.1934-6344.2010.01.0-0

Di Rienzo, J.A., F. Casanoves, M.G. Balzarini, L. Gonzalez, M. Tablada, and C.W. Robledo. 2015. InfoStat versión 2015. Grupo InfoStat, FCA, Universidad Nacional de Córdoba, Argentina. www.infostat.com.ar; http://www.infostat.com.ar (accessed 21 Mar. 2016).

Egli, D.B. 1998. Seed biology and the yield of grain crops. CAB Int., Wallingford, Oxfordshire, UK.

Fernández, R., A. Quiroga, and E. Noellemeyer. 2012. Cover crops: A viable alternative for the semiarid Pampa Region? Cienc. Suelo 30:137–150.

Friesen, D.K., and G.J. Blair. 1988. A dual radiotracer study of transformations of organic, inorganic and plant residue phosphorus in soil in the presence and absence of plants. Soil Res. 26:355–366. doi:10.1071/SR9880355

Goering, H.K., and P.J. Van Soest. 1970. Forage fiber analysis (apparatus, reagents, procedures, and some applications) Agriculture Handb. 379.U.S. Gov. Print. Office, Washington, DC.

Grace, P.R., J.N. Ladd, G.P. Robertson, and S.H. Gage. 2006. SOCRATES—A simple model for predicting long-term changes in soil organic carbon in terrestrial ecosystems. Soil Biol. Biochem. 38:1172–1176. doi:10.1016/j.soilbio.2005.09.013

Guérif, J., G. Richard, C. Dürr, and J. Machet. 2001. A review of tillage effects on crop residue management, seedbed conditions and seedling establishment. Soil Tillage Res. 61:13–32. doi:10.1016/S0167-1987(01)00187-8

Ha, K.V., P. Marschner, and E.K. Bünemann. 2008. Dynamics of C, N, P and microbial community composition in particulate soil organic matter during residue decomposition. Plant Soil 303:253–264. doi:10.1007/s11104-007-9504-1

Heal, O.W., J.M. Anderson, and M.J. Swift. 1997. Plant litter quality and decomposition: An historical overview. In: G. Cadish and K.E. Killer, editors, Driven by nature. CAB Int., Wallingford, Oxfordshire, UK. p. 3–30.

Henriksen, T.M., and T.A. Breland. 2002. Carbon mineralization, fungal and bacterial growth, and enzyme activities as affected by contact between crop residues and soil. Biol. Fertil. Soils 35:41–48. doi:10.1007/s00374-001-0438-0

IPNI. 2012. Estimates of nutrient uptake and removal. Int. Plant Nutrition Inst. http://www.ipni.net/article/IPNI-3296 (accessed 21 Mar. 2016).

Iyamuremye, F., and R.P. Dick. 1996. Organic amendments and phosphorus sorption by soils. Adv. Agron. 56:139–185. doi:10.1016/S0065-2113(08)60181-9

Jalali, M., and F. Ranjbar. 2009. Rates of decomposition and phosphorus release from organic residues related to residue composition. J. Plant Nutr. Soil Sci. 172:353–359. doi:10.1002/jpln.200800032

Jones, O.L., and M.S. Bromfield. 1969. Phosphorus changes during the leaching and decomposition of hayed-off pasture plants. Aust. J. Agric. Res. 20:653–663. doi:10.1071/AR9690653

Kalburtji, K.L., and A.P. Mamolos. 2000. Maize, soybean and sunflower litter dynamics in two physicochemically different soils. Nutr. Cycling Agroecosyst. 57:195–206. doi:10.1023/A:1009814218516

Kassam, A., T. Friedrich, F. Shaxson, and J. Pretty. 2009. The spread of conservation agriculture: Justification, sustainability and uptake. Int. J. Agric. Sustain. 7:292–320. doi:10.3763/ijas.2009.0477

Kwabiah, A.B., N.C. Stoskopf, C.A. Palm, and R.P. Voroney. 2003. Soil P availability as affected by the chemical composition of plant materials: Implications for P-limiting agriculture in tropical Africa. Agric. Ecosyst. Environ. 100:53–61. doi:10.1016/S0167-8809(03)00171-3

Ladd, J.N., M. Amato, P.R. Grace, and J.A. Van Veen. 1995. Simulation of 14C turnover through the microbial biomass in soils incubated with 14C-labelled plant residues. Soil Biol. Biochem. 27:777–783. doi:10.1016/0038-0717(94)00243-T

Littell, R.C., J. Pendergast, and R. Natarajan. 2000. Tutorial in biostatistics: Modelling covariance structure in the analysis of repeated measures data. Stat. Med. 19:1793–1819. doi: 10.1002/1097-0258(20000715)19:13<1793::AID-SIM482>3.0.CO;2-Q

Luna-Orea, P., M.G. Wagger, and M.L. Gumpertz. 1996. Decomposition and nutrient release dynamics of two tropical legume cover crops. Agron. J. 88:758–764. doi:10.2134/agronj1996.00021962008800050013x

Lupwayi, N.Z., G.W. Clayton, J.T. O'Donovan, K.N. Harker, T.K. Turkington, and W.A. Rice. 2004. Decomposition of crop residues under conventional and zero tillage. Can. J. Soil Sci. 84:403–410. doi:10.4141/S03-082

Lupwayi, N.Z., G.W. Clayton, J.T. O'Donovan, K.N. Harker, T.K. Turkington, and Y.K. Soon. 2007. Phosphorus release during decomposition of crop residues under conventional and zero tillage. Soil Tillage Res. 95:231–239. doi:10.1016/j.still.2007.01.007

Malik, M.A., P. Marschner, and K.S. Khan. 2012. Addition of organic and inorganic P sources to soil–Effects on P pools and microorganisms. Soil Biol. Biochem. 49:106–113. doi:10.1016/j.soilbio.2012.02.013

Maltais-Landry, G., and E. Frossard. 2015. Similar phosphorus transfer from cover crop residues and water-soluble mineral fertilizer to soils and a subsequent crop. Plant Soil 393:193–205. doi:10.1007/s11104-015-2477-6

Meisinger, J.J., W.L. Hargrove, R.L. Mikkelsen, J.R. Williams, and V.W. Benson. 1991. Effects of cover crops on groundwater quality. In: W.L. Hargrove, editor, Cover crops for clean water. Soil and Water Conserv. Soc., Ankeny, IA. p. 57–68.

Murphy, J., and J.P. Riley. 1962. A modified single solution method for the determination of phosphate in natural waters. Anal. Chim. Acta 27:31–36. doi:10.1016/S0003-2670(00)88444-5

Murungu, F.S., C. Chiduza, P. Muchaonyerwa, and P.N.S. Mnkeni. 2011. Decomposition, nitrogen and phosphorus mineralization from winter-grown cover crop residues and suitability for a smallholder farming system in South Africa. Nutr. Cycling Agroecosyst. 89:115–123. doi:10.1007/s10705-010-9381-5

Njunie, M.N., M.G. Wagger, and P. Luna-Orea. 2004. Residue decomposition and nutrient release dynamics from two tropical forage legumes in a Kenyan environment. Agron. J. 96:1073–1081. doi:10.2134/agronj2004.1073

Noack, S.R., T.M. McBeath, M.J. McLaughlin, R.J. Smernik, and R.D. Armstrong. 2014. Management of crop residues affects the transfer of phosphorus to plant and soil pools: Results from a dual-labelling experiment. Soil Biol. Biochem. 71:31–39. doi:10.1016/j.soilbio.2013.12.022

Noack, S.R., M.J. McLaughlin, R.J. Smernik, T.M. McBeath, and R.D. Armstrong. 2012. Crop residue phosphorus: Speciation and potential bio-availability. Plant Soil 359:375–385. doi:10.1007/s11104-012-1216-5

Novelli, L.E., O.P. Caviglia, and R.J.M. Melchiori. 2011. Impact of soybean cropping frequency on soil carbon storage in Mollisols and Vertisols. Geoderma 167-168:254–260. doi:10.1016/j.geoderma.2011.09.015

Parr, J.F., and R.I. Papendick. 1978. Factors affecting the decomposition of crop residues by microorganisms. In: W.R. Oschwald, editor, Crop residue management systems. ASA, Madison, WI. p. 101–129.

Peiretti, R., and J. Dumanski. 2014. The transformation of agriculture in Argentina through soil conservation. Int. Soil Water Conserv. Res. 1:14–20. doi:10.1016/S2095-6339(15)30010-1

Pinheiro, J., D. Bates. S. DebRoy, D. Sarkar, and R Core Team. 2015. nlme: Linear and nonlinear mixed effects models. R package version 3.1-121. http://CRAN.R-project.org/package=nlme (accessed 21 Mar. 2016)

Poffenbarger, H.J., S.B. Mirsky, R.R. Weil, M. Kramer, J.T. Spargo, and M.A. Cavigelli. 2015. Legume proportion, poultry litter, and tillage effects on cover crop decomposition. Agron. J. 107:2083–2096. doi:10.2134/agronj15.0065

Quemada, M. 2004. Predicting crop residue decomposition using moisture adjusted time scales. Nutr. Cycling Agroecosyst. 70:283–291. doi:10.1007/s10705-005-0533-y

Ranells, N.N., and M.G. Wagger. 1996. Nitrogen release from grass and legume cover crop monocultures and bicultures. Agron. J. 88:777–782. doi:10.2134/agronj1996.00021962008800050015x

Restovich, S.B., A.E. Andriulo, and S.I. Portela. 2012. Introduction of cover crops in a maize–soybean rotation of the Humid Pampas: Effect on nitrogen and water dynamics. Field Crops Res. 128:62–70. doi:10.1016/j.fcr.2011.12.012

Salas, A.M., E.T. Elliott, D.G. Westfall, C.V. Cole, and J. Six. 2003. The role of particulate organic matter in phosphorus cycling. Soil Sci. Soc. Am. J. 67:181–189. doi:10.2136/sssaj2003.0181

Schabenberger, O. 2009. Nonlinear regression in SAS. UCLA: Academic Technology Serv., Statistical Consulting Group, Los Angeles, CA. http://www.ats.ucla.edu/stat/sas/library/SASNLin_os.htm (accessed 21 Mar. 2016).

Schipanski, M.E., M. Barbercheck, M.R. Douglas, D.M. Finney, K. Haider, J.P. Kaye et al. 2014. A framework for evaluating ecosystem services provided by cover crops in agroecosystems. Agric. Syst. 125:12–22. doi:10.1016/j.agsy.2013.11.004

Schomberg, H.H., D.M. Endale, A. Calegari, R. Peixoto, M. Miyazawa, and M.L. Cabrera. 2006. Influence of cover crops on potential nitrogen availability to succeeding crops in a Southern Piedmont soil. Biol. Fertil. Soils 42(4):299–307. doi:10.1007/s00374-005-0027-8

Schomberg, H.H., and J.L. Steiner. 1999. Nutrient dynamics of crop residues decomposing on a fallow no-till soil surface. Soil Sci. Soc. Am. J. 63:607–613. doi:10.2136/sssaj1999.03615995006300030025x

Shammas, K., A.M. O'Connell, T.S. Grove, R. McMurtrie, P. Damon, and S.J. Rance. 2003. Contribution of decomposing harvest residues to nutrient cycling in a second rotation Eucalyptus globulus plantation in south-western Australia. Biol. Fertil. Soils 38:228–235. doi:10.1007/s00374-003-0654-x

Snapp, S.S., S.M. Swinton, R. Labarta, D. Mutch, J.R. Black, R. Leep et al. 2005. Evaluating cover crops for benefits, costs and performance within cropping system niches. Agron. J. 97:322–332. doi:10.2134/agronj2005.0322

Stubbs, T.L., A.C. Kennedy, P.E. Reisenauer, and J.W. Burns. 2009. Chemical composition of residue from cereal crops and cultivars in dryland ecosystems. Agron. J. 101:538–545. doi:10.2134/agronj2008.0107x

Thorup-Kristensen, K., J. Magid, and L.S. Jensen. 2003. Catch crops and green manures as biological tools in nitrogen management in temperate zones. Adv. Agron. 79:227–302. doi:10.1016/S0065-2113(02)79005-6

Varela, M.F., P.L. Fernandez, G. Rubio, and M.A. Taboada. 2011. Cover crops: Effects on soil macroporosity and soil structural stability in a silt loam soil. (In Spanish, with English abstract.) Cienc. Suelo 29:99–106.

Varela, M.F., C.M. Scianca, M.A. Taboada, and G. Rubio. 2014. Cover crop effects on soybean residue decomposition and P release in no-tillage systems of Argentina. Soil Tillage Res. 143:59–66. doi:10.1016/j.still.2014.05.005

Verhoef, H.A. 1995. Litterbag method. In: K. Alef and P. Nannipieri, editors, Methods in applied soil microbiology and biochemistry. Academic Press, London. p. 485–487.

Wider, R.K., and G.E. Lang. 1982. A critique of the analytical methods used in examining decomposition data obtained from litter bags. Ecology 63:1636–1642. doi:10.2307/1940104

Zhang, D., D. Hui, Y. Luo, and G. Zhou. 2008. Rates of litter decomposition in terrestrial ecosystems: Global patterns and controlling factors. J. Plant Ecol. 1:85–93. doi:10.1093/jpe/rtn002