Agriculture, Ecosystems and Environment 224 (2016) 29–38

Contents lists available at ScienceDirect

Agriculture, Ecosystems and Environment

journal homepage: www.elsevier.com/locate/agee

# Managing the value of composts as organic amendments and fertilizers in sandy soils

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ARTICLE INFO

*Article history:*
Received 21 August 2015
Received in revised form 23 February 2016
Accepted 9 March 2016
Available online 24 March 2016

*Keywords:*
Biosolids
Animal manures
Municipal solid waste
Compost mixtures
Organic matter
Nutrients
Ca carbonates

ABSTRACT

Nutrient rich composts are employed at low rates to minimize risks of N and P losses; this limits their value as soil improvers through C addition and the build up of soil organic matter. Blending with nutrient-poor composts such as those from the organic fraction of municipal solid waste could reduce the risks of nutrient losses while maintaining the positive effects on soil organic matter. We conducted a 2-yr experiment with composts of diverse origin: organic fraction of municipal solid waste (MC), cattle feedlot manure (FC), poultry litter (PC) and biosolids (BC), alone or blended (FC-MC, PC-MC) in a sandy soil under the humid warm climatic conditions of NE Argentina. We studied the effects of a single application (40 Mg $ha^{-1}$) on the surface soil (0–10 cm) properties of a permanent subtropical pasture through annual chemical and biological analyses. On five dates, available N and P were also determined at 0–10 cm and 55–65 cm. Soil total C and N increased over time while potential N mineralization and $CO_2$ emission decreased. All amendments resulted in similar increments of soil C and N despite marked differences in quantity and quality of organic matter inputs. Because MC had substantial amounts of Ca carbonates, it contributed to a reduction of available P from manure composts through dilution and precipitation. The release of available P from biosolids composts (where P is bound to Fe and Al) was lower than from manure composts (where P is bound by Ca phosphates). The highest environmental risk from compost application would likely be the leaching of soluble N produced during the composting process and released immediately after field application. Blending with N-poor MC would contribute to nitrate dilution.

## 1. Introduction

Disposal of manure and biosolids is cause for increasing concern in many countries due to the constantly growing demand for meat products and the increased construction of wastewater treatment plants. Due to their low C/N ratio and high available-P content, manures and biosolids can lead to nitrate leaching into groundwater and P runoff into surface waters, making land application a major nonpoint source of water pollution (Pierzynski, 1994; Moore et al., 1995; Siddique and Robinson, 2003; Sims and Sharpley, 2005). Low application rates are therefore needed to minimize environmental problems, and this limits the value of manure and biosolids as organic amendments, i.e., as soil improvers through carbon addition and the build up of soil organic matter (SOM).

Composting nutrient-rich wastes with carbon-rich materials stabilizes organic matter, resulting in products of low N mineralization rates. This improves the amendment value of organic wastes with respect to N pollution (Chang and Janzen, 1996; Escudero et al., 2012), but has little or no effect on P availability (Sharpley and Moyer, 2000; Dao et al., 2001). Several treatments have been proposed to reduce easily available P of raw and composted organic wastes, including P precipitation as Fe and Al salts or P adsorption on amorphous Fe and Al oxides and biochar (Moore et al., 1995; Callahan et al., 2002; Bock et al., 2015). In fact, available P in biosolids is often reduced by coagulation-sedimentation during sewage treatment through addition of Fe and Al salts (McCoy et al., 1986; Wen et al., 1997; EPA, 2004). An alternative involves blending and co-utilization of different wastes to minimize negative environmental impacts and to take advantage of synergistic effects in soil (Cooperband, 2000). For example, N and P-rich

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http://dx.doi.org/10.1016/j.agee.2016.03.016
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composts from animal manures or biosolids could be blended with nutrient-poor sources, such as composts from the organic fraction of municipal solid waste, which have low N- and P contents but are often reported as rich in Ca carbonates (Beck-Friis et al., 2003; Kowaljow and Mazzarino, 2007). This could dilute the high N and P contents, and decrease P availability through the formation of Ca-P precipitates.

Risks of nutrient losses vary with soil type and climate. Higher losses are expected in sandy soils due to low nutrient retention and lack of SOM protection (Weber et al., 2007; Guo and Li, 2012), and in humid and warm climates where microbial activity and decomposition are enhanced (Mtambanengwe and Mapfumo, 2006; Tu et al., 2006). In northeastern Argentina, sandy soils are developed from alluvial deposits of two large rivers (Paraná and Uruguay) under a subtropical humid climate. Their texture makes them prone to losing nutrients and organic matter and poses a serious threat to water quality when used for agriculture; therefore, management practices are needed that increase the build-up of SOM and reduce the loss of nutrients.

The objectives of this work were to determine the extent to which composts of diverse origin: organic fraction of municipal solid waste (MC), cattle feedlot manure (FC), poultry litter (PC) and biosolids (BC), alone or blended (FC-MC, PC-MC), differed in their effects on the properties of a sandy soil in subtropical NE Argentina. The study addressed two broad aspects related to the patterns of compost effects on soil: (a) the amendment value, in terms of increase and persistence of SOM, and (b) the fertilizer value, in terms of the supply of available nutrients, and the associated risks of losses. We hypothesized that (a) even under the extreme conditions of coarse-textured soils and humid warm climates, the application of stabilized organic matter would increase SOM due to limited microbial activity, and (b) mixtures of nutrient-rich manure composts with nutrient-poor MC would reduce the risks of nutrient losses while maintaining positive effects on SOM.

## 2. Materials and methods

### 2.1. Study site

The study was conducted near Santa Ana (province of Corrientes) in northeastern Argentina (27° 27′ 25.70″ S and 58° 40′ 26.76″ W, 65 m a.s.l.). The climate is subtropical, with a mean annual temperature of 21–22° C and a mean annual precipitation of 1100–1200 mm. Soils are classified as argic Udipsamments (Arenosols according to ISSS-ISRIC-FAO, 1998), have a sandy (0–40 cm) to sandy loam texture (40–100 cm), with a buried argillic horizon below 130 cm (Escobar et al., 1996). They are characterized by very low fertility at the surface (0–10 cm): 2.2 g $kg^{-1}$ of organic C, 0.2 g $kg^{-1}$ of total N, pH of 6.3, 0.02 dS $m^{-1}$ electrical conductivity, 0.9 mg $kg^{-1}$ Olsen-P and 0.03 kg $kg^{-1}$ water content at 0.01 MPa. Soil textural composition at 0–10 cm is 91.0% fine sand (0.05–0.5 mm), 2.2% coarse sand (0.5–2.0 mm), 4.2% silt and 2.6% clay. The vegetation is a permanent grassland dominated by *Andropogon lateralis* and a mixture of the grasses *Sporobolus indicus* and *Paspalum notatum*, plus the forbs *Desmodium incanum*, *Bidens pilosa* and *Richardia* sp. During the study period (October 2011-December 2013), mean monthly temperatures varied from 13.6° C to 28.5° C, and monthly precipitation from 13 mm to 266 mm, June and July being the coldest and driest months (Fig. 1).

### 2.2. Compost characteristics

#### 2.2.1. General characteristics

Biosolids compost (BC) and municipal compost (MC) were produced in NW Patagonia, in the cities of Bariloche (biosolids composting plant) and Villa La Angostura (municipal treatment plant), respectively. For the production of BC, biosolids were co-composted with wood shavings and yard trimmings, while MC was obtained from the organic fraction of municipal solid waste. The manure composts (FC and PC) were experimentally produced at the Universidad National de Córdoba (in central Argentina) and at the Universidad Nacional del Nordeste (in NE Argentina), respectively. FC was produced from beef cattle feedlot manure and sawdust, and PC from poultry manure, sawdust and rice hulls. Organic C, N and P were lower in MC than in BC, FC and PC; conversely, MC had the highest values of pH, total Ca and Ca carbonate (Table 1). Extractable P as percentage of total P was higher in manure composts (PC and FC), but the highest values of total P and N corresponded to PC and BC. All composts were stable and mature according to the following threshold values (Gómez-Brandón et al., 2008; Leconte et al., 2009): water soluble carbon (WSC) < 10 g $kg^{-1}$, WSC/total N < 0.7, $NH_4$-N/$NO_3$-N < 0.3, $NH_4$-N < 400 mg $kg^{-1}$, germination index (GI) > 60% (Table 1). Trace elements (Table S1) were below limits set in some European countries for composts (Houot et al., 2005; BOE, 2013). Differences

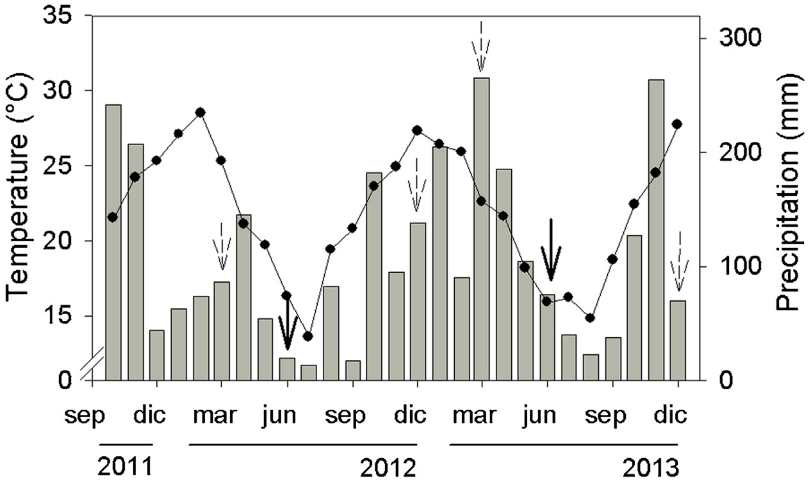

**Fig. 1.** Mean monthly temperatures (lines) and monthly precipitation (bars) during the study period. Arrows indicate soil sampling dates (full arrows for chemical and biological characterization at 0–10 cm; full- and dashed arrows for inorganic N and P analyses at 0–10 and 55–65 cm).

**Table 1**
Characteristics of the applied composts.

| | BC | MC | FC | PC |
|---|---|---|---|---|
| pH | 5.0 | 7.4 | 6.9 | 6.1 |
| EC[a] ($dS\,m^{-1}$) | 2.0 | 0.4 | 1.4 | 2.7 |
| Total N ($g\,kg^{-1}$) | 17.0 | 6.1 | 7.7 | 17.1 |
| Organic C ($g\,kg^{-1}$) | 280 | 100 | 160 | 190 |
| Lignin (% of OM[b]) | 26 | 37 | 31 | 31 |
| Water soluble C ($g\,kg^{-1}$) | 1.6 | 0.1 | 1.9 | 2.6 |
| Total P ($g\,kg^{-1}$) | 6.8 | 3.4 | 4.0 | 8.5 |
| Extract. P[c] ($mg\,kg^{-1}$) | 1120 | 96 | 1000 | 2890 |
| C/N | 19 | 17 | 23 | 12 |
| $NH_4^+$ –N ($mg\,kg^{-1}$) | 31 | 3 | 12 | 150 |
| $NO_3^-$ –N ($mg\,kg^{-1}$) | 1930 | 217 | 520 | 1510 |
| Ca ($g\,kg^{-1}$) | 10.7 | 41.5 | 11.3 | 17.6 |
| Mg ($g\,kg^{-1}$) | 5.0 | 5.1 | 4.7 | 3.7 |
| K ($g\,kg^{-1}$) | 3.1 | 1.1 | 4.9 | 4.5 |
| $CaCO_3$ (%) | – | 7.6 | – | – |
| GI[d] (%) | 66 | 97 | 86 | 79 |

Treatment code: BC, biosolids compost; MC, municipal solid wastes compost; FC, feedlot compost; PC, poultry litter compost.

[a] Electrical conductivity.

[b] Organic matter (OM = organic C × 1.8).

[c] Extracted in $NaHCO_3$ (1:100 ratio).

[d] Germination Index (*Lolium perenne* after 7 days).

in compost composition led to different inputs of organic matter and nutrients to the soil (Table 2).

*2.2.2. Compost P fractions*

A sequential fractionation procedure modified from Tiessen and Moir (1993) was employed to study P fractions of the composts. Samples were sequentially extracted with 0.5 M $NaHCO_3$ (labile-P), 0.1 M NaOH (P bound to Fe and Al) and 1 M HCl (P bound to Ca) using a sample-to-extractant ratio of 1:60, shaking for 16 h, and centrifuging at 2600*g*. Inorganic P (Pi) was determined in each extract by the molybdate ascorbic acid method. Total organic P (Po) was determined separately as the difference between acid extractable P (0.5 M $H_2SO_4$) in ashed samples (550° C) minus acid extractable P in non-ashed samples (Walker and Adams, 1958). Total P was determined in separate samples using $H_2SO_4/H_2O_2$ (Tiessen and Moir, 1993).

*2.2.3. Laboratory incubations*

To assess the quality of each compost in combination with the study-site soil, their potential capacity to release $CO_2$, to mineralize N and to release P was assessed in laboratory incubations prior to the field trial. The potential capacity of the composts to release $CO_2$ was determined using three representative soil samples of the site thoroughly mixed with composts and mixtures at a rate of 40 g amendments per kg soil. Samples (75 g) were incubated at 25° C and field capacity during 16 weeks in sealed, 1.5-L glass jars, which were distributed according to a completely randomized design. The evolved $CO_2$ was trapped in glass vials containing 0.2 M NaOH placed at an angle over the soil sample and titrated with 0.2 M HCl after $Ba_2Cl$ addition. At 2, 4, 8, 12 and 16 weeks the NaOH traps were replaced and analyzed, and soil moisture was replenished to field capacity. $CO_2$ emission was estimated as the cumulative $CO_2$-C evolved during the incubation period (Lerch et al., 1992; Tognetti et al., 2008).

The potential N mineralization and P release capacity of composts and mixtures was estimated using a non-leached, destructive sampling procedure (Lerch et al., 1992; Mazzarino et al., 1998). Samples consisted of 100 g of amended soil (40 g of amendment per kg soil) in 0.25-L plastic jars. Soil moisture was gravimetrically replenished to field capacity weekly. At each sampling date (0, 4, 8, 12 and 16 weeks), three jars per treatment were randomly selected for analysis of inorganic N and P, and discarded after sampling. Inorganic N was determined in 8 g-samples extracted with 2 M KCl (1:5, soil:solution ratio): $NO_3^-$-N by copperized Cd reduction and $NH_4^+$-N by the Berthelot reaction. Inorganic P was determined in 2-g samples extracted with 0.5 M $NaHCO_3$ at a soil:solution ratio of 1:20 (Olsen-P) by the molybdate ascorbic acid method (Sparks et al., 1996).

**Table 2**
Nutrients and carbon added to the soil at an amendment rate of 40 $t\,ha^{-1}$.

| | BC | MC | FC | PC | FC-MC | PC-MC |
|---|---|---|---|---|---|---|
| Dry weight basis ($kg\,ha^{-1}$) | | | | | | |
| Organic C | 11200 | 3950 | 6420 | 7520 | 5190 | 5740 |
| Total N | 680 | 244 | 308 | 684 | 276 | 464 |
| Total P | 271 | 135 | 159 | 339 | 147 | 237 |
| Soluble C | 62 | 4 | 75 | 103 | 40 | 54 |
| Inorganic N | 78 | 9 | 21 | 66 | 15 | 38 |

Treatment code: BC, biosolids compost; MC, municipal solid wastes compost; FC, feedlot compost; PC, poultry litter compost; FC-MC, 50% FC + 50% MC; PC-MC, 50% PC + 50% MC.

*2.3. Field trial*

The experiment was carried out in an enclosure of about half a hectare, where grasses were cut before installation. The following treatments were applied: non-treated control (C), inorganic fertilization (IF), biosolids compost (BC), municipal compost (MC), feedlot compost (FC), poultry manure compost (PC), a mixture of 50% PC + 50% MC (PC-MC), and a mixture of 50% FC + 50% MC (FC-MC). The experimental design was a randomized complete block design with three replications. Each plot was 2 m × 3 m with a 2 m buffer zone between plots and 3 m between blocks. Previous analyses of biosolids composts from the same composting facility showed that P is mainly bound to Fe and Al, so BC was not blended with MC. All composts were screened through 15 mm mesh, and the mixtures were done on a dry-weight basis. Composts and mixtures were applied at an equivalent rate of 40 $Mg\,ha^{-1}$ (dry weight), and the inorganic fertilizer at a rate of 50 $kg\,N\,ha^{-1}$ and 21 $kg\,P\,ha^{-1}$ (as urea and diammonic phosphate). Composts were surface-applied and distributed with rakes.

*2.4. Sampling and analyses of the field trial*

The experiment was started in October 2011 (spring) and sampled over two years until December 2013 (summer). Compost effects on SOM were assessed by analyzing organic C, total N and microbial activity, and effects on nutrient availability and risks of losses were assessed by analyzing inorganic N and P, exchangeable cations, and the residual capacity to mineralize N and to release/retain P. On the sampling dates residual composts remained on the soil surface and were removed before sampling. One composite sample consisting of two randomly selected cores was collected from each plot.

Two types of sampling were conducted: (a) five samplings at the surface (0–10 cm) and at depth (55–65 cm) to analyze inorganic N and P in March and June 2012, and March, June and December 2013 (5, 8, 17, 20 and 26 months after experiment installation). Olsen-P, $NO_3^-$-N and $NH_4^+$-N were determined in fresh field samples employing the methodologies described in 2.2.3. (b) a yearly sampling of the surface soil (0–10 cm) for chemical and biological characterization in June 2012 and June 2013 (8 and 20 months after experiment installation). A portion of the soil samples was air-dried for chemical analysis and the residual capacity to release $CO_2$, while another portion was maintained at field moisture to assess mineralizable N and the capacity to release/retain P.

Dried soils were sieved through a 2-mm mesh and analyzed for pH, electrical conductivity, Olsen-P and exchangeable cations (Ca, Mg, K and Na). The following analyses were carried out (Sparks et al., 1996): pH in water (1:2.5, soil:water ratio), electrical conductivity (1:5, soil:water ratio), Olsen-P as mentioned above and exchangeable cations in 1 M $NH_4OAc$ extracts by atomic absorption spectrometry (Perkin-Elmer AAnalyst 100). Total C and total N in samples ground to pass a 0.5-mm mesh were determined with an elemental analyzer (Thermo Electron Corporation Flash EA 1112). To test if the inorganic C from MC carbonates was a significant source of the soil total C evaluated with the C-N analyzer, two assays were conducted (SM Text of Supplementary Material). We concluded that the amount of Ca carbonates added with the composts was diluted in the soil rendering values not readily detectable with the C-N analyzer.

The residual capacity to release $CO_2$ was estimated from 7 d-incubations of rewetted soil (modified method of Franzluebbers et al., 1996). Dry soil samples (50 g) were rewetted to field capacity, incubated for 7 days at 25° C, and the evolved $CO_2$-C was trapped and analyzed as described in Section 2.2.3.

Mineralizable N (pNmin) and the capacity to release/retain P were assessed in 16-wk incubations of field-fresh samples according to the methodologies described in 2.2.3. Inorganic N (Ni) was determined at 0, 4, 8, 12 and 16 weeks, and pNmin was calculated as the difference between Ni at $t_x$ minus Ni at $t_0$ (Lerch et al., 1992). Olsen-P was determined at $t_0$ and $t_{16}$, and the capacity to release/retain P was calculated as the difference between the two values.

Aboveground biomass was assessed 7 months after the initiation of the experiment (May, mild autumn, mean temperature 19.7° C, precipitation 54 mm) by manually cutting vegetation in two randomly selected quadrats (0.25 cm × 0.25 cm); vegetation samples were oven-dried at 60° C for 48 h and weighed.

### 2.5. Data analyses

The significance of differences in means of all variables among treatments was evaluated by ANOVA. Main effects were separated by tests of least significant differences (LSD). Null hypotheses were rejected at $p < 0.05$. In order to achieve homocedasticity, data of Olsen-P at three dates and pH, total N and pNmin at one date were $\log_{10}$-transformed, while Olsen-P at two dates was square root transformed. When data violated an ANOVA assumption, even after transformations, Friedman's rank test and Dunn's multiple comparison test were used ($p < 0.05$); this occurred with electrical conductivity and sum of exchangeable cations at one date. Statistica (StatSoft 1984–2003) and Infostat (2008) programs were used for analyses.

## 3. Results

### 3.1. Quality of organic amendments

During the 16-wk incubation assay, all amendments had significantly higher rates of $CO_2$ emission than the control (Fig. 2A). The highest cumulative $CO_2$ emissions corresponded to MC, alone or in mixtures (MC, FC-MC, PC-MC), which differed significantly from manure- and biosolids-composts (FC, PC, BC).

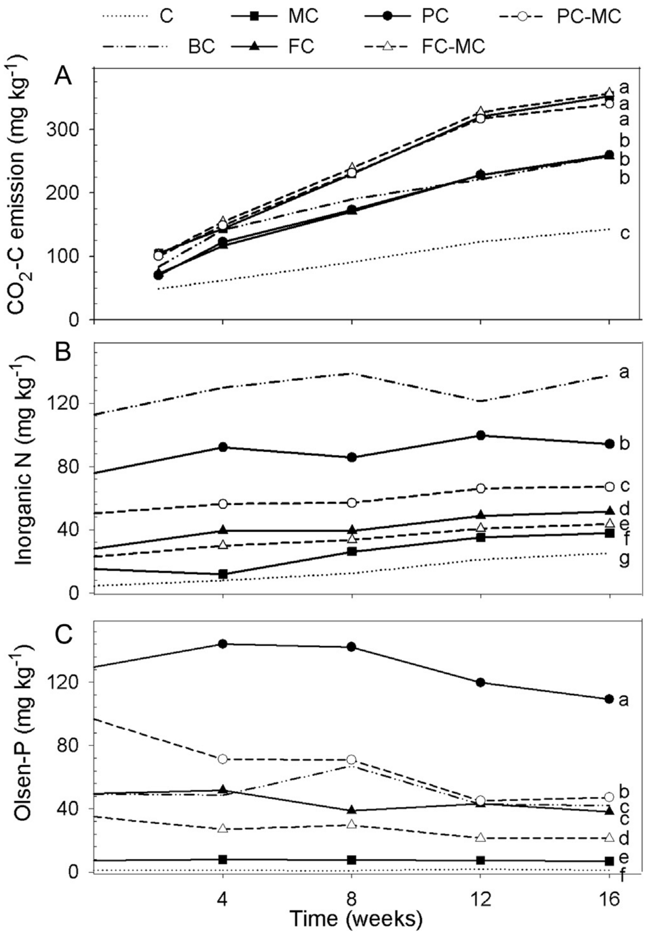

**Fig. 2.** Organic matter and nutrient quality of the applied amendments: (A) $CO_2$ emission, (B) soluble and mineralized N, and (C) phosphorus release/retention in 16-wk incubations of soil and composts. Different letters indicate significant differences among treatments for the last date ($p < 0.05$, $n = 3$). Treatment code: C, control; BC, biosolids compost; MC, municipal solid waste compost; FC, feedlot compost; PC, poultry litter compost; FC-MC, 50% FC + 50% MC; PC-MC, 50% PC + 50% MC.

Inorganic N was higher with the amendments (Fig. 2B), and much higher with BC and PC, reflecting the much higher inorganic N contents, especially nitrate, of these two amendments (Table 1). Blending PC with MC resulted in a 30% reduction of inorganic N through dilution. However, no differences were found between amendments and control in net N mineralization (inorganic N at $t_{16}$ minus $t_0$).

The amendments also showed different capacities to supply and release P over the incubation period (Fig. 2C), and this was related to their P fraction composition (Fig. 3). Thus, despite BC having higher total P than FC, both composts released similar amounts of Olsen-P, presumably because a high proportion of total P in BC was bound to Fe and Al (Pi-NaOH) and to more labile forms in FC. A large amount of Olsen-P (130 mg kg$^{-1}$) was supplied with PC, which was the compost with the highest labile-Pi (Pi-NaHCO$_3$). For this compost, a net P release was observed during the first 8 weeks of incubation, but this changed to P retention thereafter, implying different controlling factors over time. Phosphorus added with MC was very low, and mixtures containing MC reduced P release from FC and PC. This reduction was evident from the beginning of the incubation, with a marked negative slope during the first 4 weeks, especially in the case of PC-MC, indicating that MC not only exerts a dilution effect but also increases P retention. In all composts, the highest percentage of total P corresponded to inorganic P fractions, while organic P only represented 10–14% in FC, BC and MC, and less than 1% in PC (Fig. 3).

### 3.2. SOM increase and persistence in the field

Compost addition increased total soil C and N concentrations with respect to both the control and the inorganic fertilization treatment (Fig. 4A,B). Nevertheless, after the first 8 months total C and N were significantly higher only for FC and PC-MC, and after 20 months they were significantly higher in all compost and mixture treatments, with no significant differences among them. Only the PC-MC treatment showed similar high values of C and N in both years. There was a significant correlation between C and N concentrations for both dates pooled ($r^2 = 0.96$; $p < 0.001$). Consequently, the C/N ratio did not vary among treatments and ranged between 11 and 14.

While total C and N increased over time, the capacity to release CO$_2$ declined (Fig. 4C). This parameter, measured in 7-d incubations of rewetted soil, was significantly higher in all compost and mixture treatments after 8 months from the start of the experiment, but decreased after 20 months in all amended treatments and did not differ from the control. When the release of CO$_2$ was calculated per unit of C, a similar trend was observed after 8 months, while values were even lower than the control after 20 months (Table S2 of Supplementary Material).

### 3.3. Fertilizer value

Inorganic N (ammonium + nitrate) in the surface soil on five dates (5, 8, 17, 20 and 26 months after the start of the experiment) did not differ among treatments, except for PC and PC-MC which were significantly higher than the control after 8 months, and MC which was significantly lower after 20 months (Fig. 5A). Nearly all values were below 12 mg kg$^{-1}$, and ammonium was the main form of available N on most dates (60–80%). At depth 55–65 cm, inorganic N concentrations were generally lower than 3 mg kg$^{-1}$ (Fig. 5B).

Mineralizable N at the surface, estimated in 16-wk incubations 8 and 20 months after experiment installation, was significantly higher in most amended treatments during the first year, but differences disappeared after 20 months (Table 3). At the end of both incubations, the main form of inorganic N was nitrate. When mineralizable N was calculated per unit of N, a trend toward lower values in the amended plots was observed after 20 months (Table S2 of Supplementary Material).

Contrary to inorganic N, inorganic P (Olsen-P) of surface soils measured on the same five sampling dates was significantly higher in the amended treatments than the control and IF on all dates, except MC on one date (Fig. 6A). The highest values corresponded to treatments receiving PC (30–50 mg kg$^{-1}$ on the first three dates), and decreased after 20 and 26 months. On the last sampling dates, the highest values were around 10–13 mg kg$^{-1}$ with PC, PC-MC and BC. When blended with MC, inorganic P tended to decrease. After 5, 17 and 20 months values were significantly lower for PC-MC than PC alone, and after 8, 17 and 20 months values were significantly lower for FC-MC than FC alone (data not shown). Inorganic P concentrations decreased markedly with depth (55–65 cm), being lower than 1.5 mg kg$^{-1}$ for most dates and treatments (Fig. 6B).

The residual capacity to release/retain P, evaluated in 16-wk incubations (same incubations as pNmin), differed among dates and treatments (Table 3). After 8 months, P was retained during the incubation in PC, PC-MC, and FC-MC (16–21%) and was released in BC and MC (16–19%), while after 20 months a net release varying between 35 and 90% was observed in all treatments.

### 3.4. Basic cations and pH

The MC-treatment showed the highest values of pH and electrical conductivity; values of Ca were high and represented almost 90% of the exchangeable basic cations in this treatment

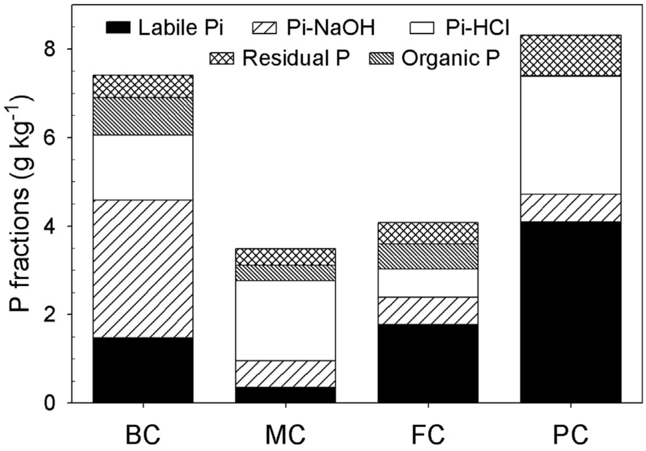

**Fig. 3.** Phosphorus fractions of biosolids compost (BC), municipal solid waste compost (MC), feedlot compost (FC) and poultry litter compost (PC).

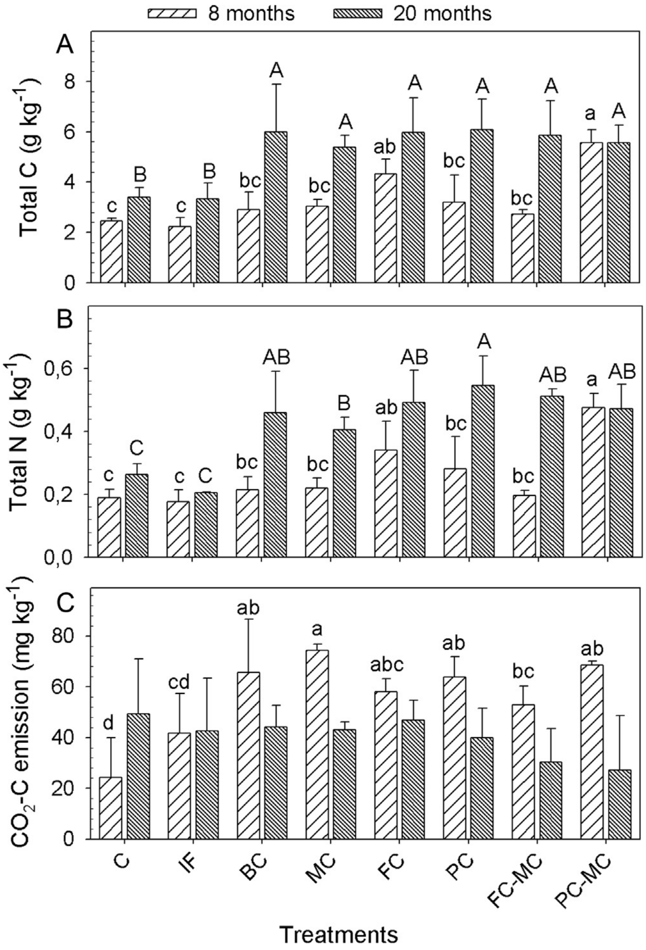

**Fig. 4.** Persistence of soil organic matter in the field over the experimental period: (A) total soil carbon, (B) total soil nitrogen and (C) residual $CO_2$ emission (7-d incubations of rewetted soil). Different lower case letters indicate significant differences among treatments after 8 months, and capital letters after 20 months, following experiment installation ($p < 0.05$, $n = 3$). Error bars represent ± SD. Treatment code: C, control; IF, inorganic fertilization; BC, biosolids compost; MC, municipal solid wastes compost; FC, feedlot compost; PC, poultry litter compost; FC-MC, 50% FC + 50% MC; PC-MC, 50% PC + 50% MC.

(Table 4). A similar trend was observed for treatments with MC in the mixtures that showed significantly higher values of electrical conductivity, basic cations and Ca but not pH. This marked effect of MC was clearly related to its elevated content of Ca carbonates (Table 1).

### 3.5. Grassland biomass

Aboveground biomass 7 months after amendment application varied between 50 and 130 $g\,m^{-2}$ (average of $n = 3$) with the highest values corresponding to BC (data not shown). However, values exhibited high variability, so BC was only marginally higher than all other treatments ($p < 0.10$).

## 4. Discussion

### 4.1. SOM increase and persistence

Compost addition increased total soil C and N over time through a slow incorporation that involved easily degradable material in a first phase, and much more stabilized material during the second year. This was reflected in higher potential N mineralization and $CO_2$ emission after 8 months than after 20 months following compost application.

It is generally considered that soil C stabilization is associated with physical protection in soil aggregates and the formation of complexes with clay and silt fractions. In recent work in temperate grasslands, Ryals et al. (2014) found an increase of C and N in the free and occluded light fractions three years after the application of organic amendments, which were negatively correlated with sand content and positively correlated with clay content. In sandy soils, such as our case, the formation of aggregates and fine-fraction complexes are strongly limited. Here, the persistence of added C appears to be more dependent on its quantity and quality in terms of degradability (Weber et al., 2007). In sandy soils of semiarid grasslands amended with MC and BC, we found a more persistent effect with BC (Kowaljow et al., 2010; González-Polo et al., 2015). In the present study, we expected a more rapid decomposition and incorporation of all amendments due to the humid, warm climate. However, rapid plant growth and the presence of a dense, permanent root system might have slowed incorporation, while the highly stable state of all composts prior to application would explain the limited decomposition (Griffin and Hutchinson, 2007; Fabrizio et al., 2009; Kätterer et al., 2014).

All amendments resulted in similar soil C and N increments despite marked differences in their C and N contents. Thus, MC supplied almost three times less C and N than BC and two times less C and N than PC to the soil (Table 2), but had comparable effects on soil C and N. Some indicators of compost quality also suggest that MC quality was low: contents of lignin as percentage of organic matter were higher and those of water-soluble C lower than in the other composts (Table 1). However, pre-treatment

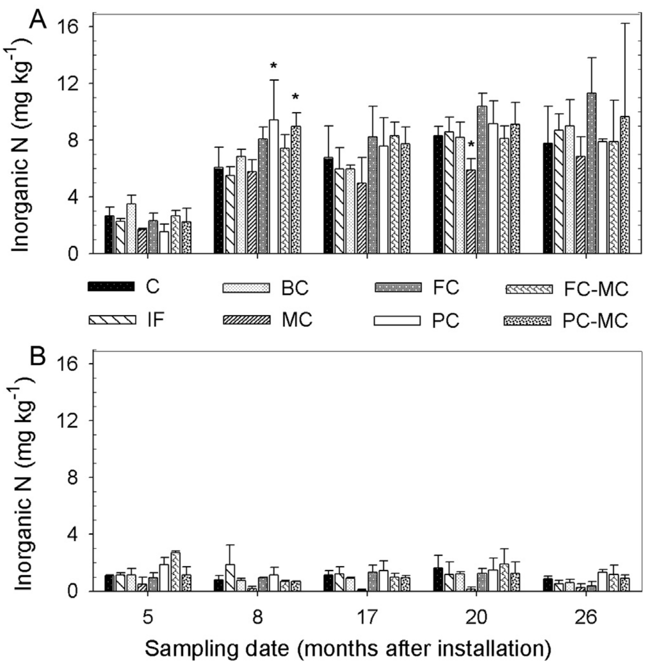

**Fig. 5.** Changes in soil inorganic N ($NH_4^-$-N + $NO_3^-$-N) at five dates during the experimental period: (A) at the surface (0–10 cm) and (B) at depth 55–65 cm. An asterisk indicates significant differences between treatments and the control for the same date ($p < 0.05$, n = 3). Error bars represent ± SD. Treatment code: C, control; IF, inorganic fertilization; BC, biosolids compost; MC, municipal solid wastes compost; FC, feedlot compost; PC, poultry litter compost; FC-MC, 50% FC + 50% MC; PC-MC, 50% PC + 50% MC.

**Table 3**

Mineralizable N (pNmin in mg $kg^{-1}$), residual P after 16 wk-incubation (mg $kg^{-1}$) and percentage of P retention/release capacity 8 and 20 months after the start of the experiment.

| | C | IF | BC | MC | FC | PC | FC-MC | PC-MC |
|---|---|---|---|---|---|---|---|---|
| pNmin[a] after 8 | 19.7c | 19.8c | 26.0ab | 28.7a | 27.0ab | 31.0a | 22.4bc | 26.9ab |
| months | (1.5) | (1.4) | (6.7) | (2.6) | (1.6) | (6.4) | (0.6) | (4.9) |
| pNmin after 20 | 19.0 | 21.1 | 24.0 | 21.4 | 21.8 | 27.5 | 20.8 | 21.0 |
| months | (4.6) | (6.2) | (2.7) | (2.0) | (2.2) | (6.0) | (2.8) | (2.5) |
| Pi t16[b] after 8 | 1.2[e] | 2.4e | 16.5c | 5.3de | 19.2bc | 31.6a | 7.7cd | 27.3ab |
| months | (0.4) | (1.1) | (5.2) | (2.0) | (2.6) | (8.2) | (3.8) | (3.3) |
| P retention/release[c] | – | – | 19 | 16 | – | −18 | −16 | −21 |
| Pi t16 after 20 | 1.3e | 1.3e | 15.5b | 4.3d | 13.7bc | 23.2a | 9.9c | 13.8bc |
| months | (0.6) | (0.5) | (3.9) | (0.5) | (1.5) | (2.1) | (4.1) | (1.2) |
| P retention/release | – | – | 50 | 90 | 63 | 45 | 81 | 35 |

Treatment code: C, control; IF, inorganic fertilization; BC, biosolids compost; MC, municipal solid wastes compost; FC, feedlot compost; PC, poultry litter compost; FC-MC, 50% FC + 50% MC; PC-MC, 50% PC + 50% MC. Different letters indicate significant differences among treatments for the same variable and date. Standard errors of means are given in parentheses.

[a] Inorganic N at t16 minus inorganic N at t0 of 16-wk incubations.

[b] Inorganic P at t16 of 16-wk incubations.

[c] ((Pi t16 − Pi t0)/Pi t0) × 100. Negative values indicate net P retention, positive values net P release.

incubations indicated that the highest rates of $CO_2$ emission corresponded to MC, alone or in mixtures, while FC, PC and BC showed intermediate values with respect to the control soil (Fig. 2A). It was evident that the additional source of $CO_2$ was Ca carbonates, which were present in considerable amounts in MC (Table 1). Several studies indicate the presence of carbonates in municipal solid waste composts (Beck-Friis et al., 2003; Spaccini and Piccolo, 2008) and a significant liming effect when applied to the soil (Amlinger et al., 2007; Kowaljow and Mazzarino, 2007; Haber et al., 2008). Apparently Ca carbonates are formed during composting of municipal solid waste associated with low nitrification rates, resulting in a marked alkalinization, up to a pH of 8.5 (Beck-Friis et al., 2003; Tognetti et al., 2007). On the contrary, N and P rich manures and biosolids usually show a striking decrease of pH during composting, associated with high nitrification rates (Huang et al., 2004; Banegas et al., 2007; Leconte et al., 2009) and the formation of neutral Ca phosphates (Cooperband and Good, 2002).

There is increasing evidence that $CO_2$ can be lost from calcareous or limed soils as a result of organic matter additions that enhance microbial respiration and nitrification, leading to acid dissolution of $CaCO_3$ (Emmerich, 2003; Bertrand et al., 2007; Tamir et al., 2013). A similar reaction might have taken place in this study, when carbonate-rich MC was incubated with the soil: a liming effect and a concurrent increase of organic matter that temporarily enhanced acidification, carbonate dissolution and $CO_2$ emission. This implies that while in FC-, PC-, and BC-treatments $CO_2$ emissions were related to labile C and microbial activity, a portion of the $CO_2$ evolved in the MC treatment was not derived from microbial respiration. We hypothesize that during the incubations to estimate compost C quality, the applied rate of all composts resulted in the saturation of the C storage capacity of the soil

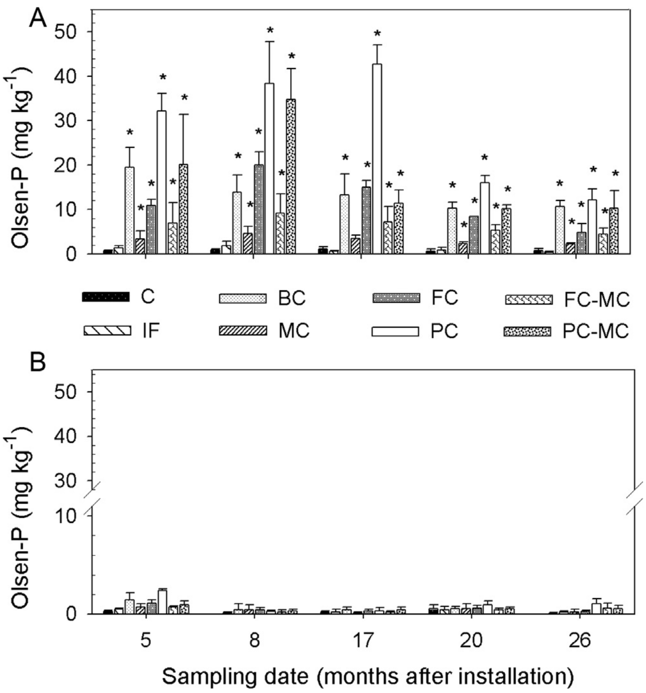

**Fig. 6.** Changes in Olsen-P at five dates during the experimental period: (A) at the surface (0–10 cm) and (B) at depth (55–65 cm). An asterisk indicates significant differences between treatments and the control for the same date ($p < 0.05$, n = 3). Error bars represent ± SD. Treatment code: C, control; IF, inorganic fertilization; BC, biosolids compost; MC, municipal solid wastes compost; FC, feedlot compost; PC, poultry litter compost; FC-MC, 50% FC + 50% MC; PC-MC, 50% PC + 50% MC.

**Table 4**
Soil pH, electrical conductivity (EC), sum of exchangeable cations and percentage of exchangeable Ca 8 and 20 months after the start of the experiment.

| | C | IF | BC | MC | FC | PC | FC-MC | PC-MC |
|---|---|---|---|---|---|---|---|---|
| 8 months | | | | | | | | |
| pH | 6.3bc | 6.0c | 6.2bc | 7.1a | 6.6abc | 6.6abc | 6.7ab | 6.6abc |
| | (0.2) | (0.2) | (1.0) | (0.5) | (0.2) | (0.2) | (0.3) | (0.2) |
| EC | 0.019cd | 0.011d | 0.021bcd | 0.046a | 0.029abc | 0.043a | 0.042ab | 0.044a |
| (dS $m^{-1}$) | (0.002) | (0.002) | (0.003) | (0.021) | (0.016) | (0.010) | (0.001) | (0.003) |
| Sum of cations | 0.84de | 0.55e | 1.27bcd | 3.41a | 1.27cd | 1.93abc | 2.37a | 3.08a |
| (cmol $kg^{-1}$) | (0.03) | (0.10) | (0.27) | (2.38) | (0.50) | (0.28) | (0.08) | (0.22) |
| Ca[a] | 63c | 64c | 52d | 87a | 59cd | 63c | 77ab | 76b |
| (%) | (2.9) | (1.5) | (3.5) | (4.2) | (6.0) | (1.2) | (0.5) | (2.5) |
| 20 months | | | | | | | | |
| pH | 6.6cd | 6.2e | 6.4de | 7.2a | 6.7bcd | 7.0ab | 6.8bc | 6.4de |
| | (0.3) | (0.4) | (0.1) | (0.2) | (0.2) | (0.3) | (0.2) | (0.1) |
| EC | 0.034cd | 0.031d | 0.035bcd | 0.058a | 0.047abcd | 0.055ab | 0.052abc | 0.065a |
| (dS $m^{-1}$) | (0.002) | (0.010) | (0.002) | (0.010) | (0.005) | (0.007) | (0.008) | (0.004) |
| Sum of cations | 0.97d | 0.97d | 1.37cd | 2.78a | 1.92bc | 2.54ab | 2.57ab | 2.43ab |
| (cmol $kg^{-1}$) | (0.08) | (0.11) | (0.10) | (0.34) | (0.09) | (0.44) | (0.54) | (0.35) |
| Ca[a] | 67cd | 69c | 61d | 89a | 68c | 74c | 82b | 81b |
| (%) | (3.7) | (5.3) | (4.1) | (1.1) | (1.2) | (2.5) | (1.4) | (1.2) |

Treatment code: C, control; IF, inorganic fertilization; BC, biosolids compost; MC, municipal solid wastes compost; FC, feedlot compost; PC, poultry litter compost; FC-MC, 50% FC + 50% MC; PC-MC, 50% PC + 50% MC. Different letters indicate significant differences among treatments for the same variable and date. Standard errors of means are given in parentheses.

[a] Exchangeable Ca as percentage of the sum of basic cations.

(Six et al., 2002) or microbial saturation (Poirier et al., 2013), but those treatments receiving MC, alone or in mixtures, had an additional $CO_2$ pulse resulting from carbonate dissolution. A similar effect might be occurring in the field, though over a longer time period (20 months) than that observed in the incubation, where composts were thoroughly mixed with the soil and incubated at optimal temperature and moisture conditions.

### 4.2. Fertilizer value and risk of nutrient losses

The liming effect of MC would be responsible for the retention of P in the soils amended with mixtures of P-rich composts and MC, as observed during the incubation to estimate compost quality (Fig. 2C). Although several authors found that addition of MC decreased soil P fixation, they refer mostly to sesquioxide-rich soils (Giusquiani et al., 1988; Jimenez et al., 1993; Mkhabela and Warman, 2005), where composts can increase pH causing precipitation of Al and Fe, form organic complexes with Al and Fe, or block P sorption sites with organic compounds. In our sandy soil, where sorption sites are negligible, main consequences of MC addition seem to be a dilution effect on P concentrations and an increase of P retention in the form of Ca phosphates, as often observed in limed soils (Bolan et al., 2003); this was especially evident in the case of PC, the compost with the highest concentrations of labile Pi. On the contrary, when PC was added alone, Olsen-P values were higher and tended to increase during the first 8 weeks of incubation. Values decreased afterwards, probably because increased P concentration in the soil solution led to supersaturation and precipitation of crystalline forms of P (e.g., hydroxyapatite). In contrast, manure Pi is associated with amorphous Ca and Mg phosphates which are more soluble (Pagliari and Laboski, 2013).

Something similar might be occurring in the field, though over a longer time period than observed in the laboratory incubations, because surface-applied composts were slowly incorporated. The highest Olsen-P values at 0–10 cm corresponded to the PC treatment and remained high at least until 17 months after the start of the experiment; when mixed with MC, values were lower in three of the five analyzed dates and tended to decrease earlier than in the PC treatment (Fig. 6). After 8 months, several treatments (PC, PC-MC, FC-MC) showed a residual capacity to retain P, i.e., when soils were incubated at optimum temperature and moisture conditions, they retained about 16–20% of initial Olsen-P (Table 3), which probably resulted from the precipitation of less soluble P minerals (as mentioned above for PC) and the liming effect of MC (the latter evidenced in the increase of exchangeable Ca; Table 4). After 20 months a new equilibrium was reached and a residual capacity to release P was observed in all treatments, although with a trend toward lower absolute values of Olsen-P (Fig. 6). The decrease of P concentration in the soil solution (due to plant uptake and/or precipitation of less soluble P minerals) would tend to maintain a constant dissolution of manure Pi until all Ca and Mg phosphates were dissolved (Pagliari and Laboski, 2013). Thus, inorganic P in the field tended to decrease at the surface after 20 and 26 months, but values did not increase with depth, suggesting an exhaustion of soluble P minerals. In conclusion, blending PC or FC with MC contributed to decrease Olsen-P at the surface, which reduces the risk of P loss through runoff and leaching.

The case of BC was quite different to manure composts. Despite having high total P concentrations, its P release was most likely regulated by P bound to Fe and Al (Fig. 3). This fraction, which represented the highest percentage of total P in BC, is usually considered a limited source of available P (McCoy et al., 1986; Wen et al., 1997). Consequently, the BC treatment showed lower P release and a continuous decrease over time in the field. These traits combined to reduce the risk of P loss compared to the PC treatment.

The main environmental risk of BC was related to the high loss of inorganic N shortly after compost application. (It was also high for PC and the fertilizer treatment.) Despite marked differences among composts in soluble N (Table 2), values in the surface soil and at 55–65 cm were very low after 5 months, and the aboveground biomass measured after 7 months showed only marginal differences for BC, the compost with the highest amount of mineral N. Possibly most nitrate was lost before 5 months during the heavy rains after application (Fig. 1), while ammonium was partially retained in the soil exchangeable complex, which in any case was very low. Soluble N is produced during the composting process, and differences in soluble N concentrations among composts are due to the quality of the original substrate and to composting treatments. Immediately after compost application, there is a pulse of soluble N, but the high stability of the composts (reflected in low rates of potential N mineralization) restrains additional N losses over time. Thus, soluble N concentration becomes an important limitation to compost application in sandy soils, and mixtures with N-poor materials, such as MC, would help to minimize losses through dilution (Fig. 2B).

## 5. Conclusions

Despite favorable conditions for decomposition (sandy soil and humid warm climate), a single application of highly stable composts resulted in an increase of soil organic C and N two years later. Increments of soil C and N were similar for all amendments despite marked differences in the quantity and quality of organic matter inputs.

Blending N- and P-rich composts from animal manures with composts from the organic fraction of MC, which have low N- and P contents but are rich in Ca carbonates, reduced the risk of P losses through dilution and liming effects. Biosolids compost differed markedly from manure composts because, despite having high total P concentrations, the P release in soil was regulated by P bound to Fe and Al. The highest environmental risk of compost application is likely the leaching of nitrates accumulated during the composting process, while high compost stability restrains additional N losses over time. Blending with N-poor MC would contribute to nitrate dilution, reducing the risk of losses.

Further research should focus on blending different types of composts to maximize amendment and fertilizer value under different environmental scenarios (soil and climate) that regulate incorporation, decomposition and losses.

## Acknowledgments

This research was funded by the Agencia Nacional de Promoción Científica y Tecnológica (PICT-2008-1027). Composts were provided by Cátedra de Microbiología (UNNE), Facultad Ciencias Agropecuarias (UNC), Cooperativa de Electricidad Bariloche and Municipalidad de Villa La Angostura. We gratefully acknowledge A. Romero for providing the study site and J.J. Ewel and the reviewers for comments improving the final version of the manuscript. E. Castán was supported by a fellowship for graduate students of ANPCyT and CONICET.

## Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.agee.2016.03.016.

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