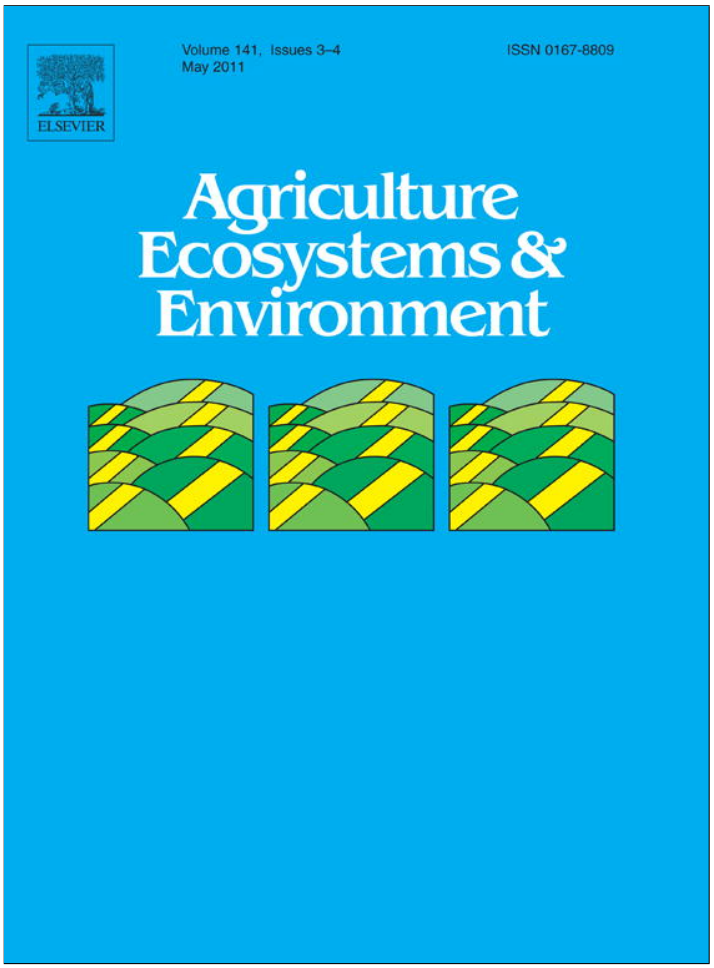
Provided for non-commercial research and education use.
Not for reproduction, distribution or commercial use.



This article appeared in a journal published by Elsevier. The attached copy is furnished to the author for internal non-commercial research and education use, including for instruction at the authors institution and sharing with colleagues.

Other uses, including reproduction and distribution, or selling or licensing copies, or posting to personal, institutional or third party websites are prohibited.

In most cases authors are permitted to post their version of the article (e.g. in Word or Tex form) to their personal website or institutional repository. Authors requiring further information regarding Elsevier's archiving and manuscript policies are encouraged to visit:

http://www.elsevier.com/copyright

Author's personal copy

Agriculture, Ecosystems and Environment 141 (2011) 461–465

Contents lists available at ScienceDirect

Agriculture, Ecosystems and Environment

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

Short communication

Potential carbon and nitrogen mineralization in soils from a perennial forage production system amended with class B biosolids

Virginia L. Jin[a,*], Mari-Vaughn V. Johnson[b], Richard L. Haney[c], Jeffrey G. Arnold[c]

[a] USDA – Agricultural Research Service, 137 Keim Hall, UNL – East Campus, Lincoln, NE 68583-0937, USA
[b] U.S. Department of State, Bureau of East Asian and Pacific Affairs, Office of Regional and Security Policy Affairs, 2201 C Street NW, Washington, DC 20520, USA
[c] USDA – Agricultural Research Service, Grassland, Soil & Water Research Laboratory, 808 E. Blackland Road, Temple, TX 76502 USA

ARTICLE INFO

Article history:
Received 13 December 2010
Received in revised form 23 March 2011
Accepted 29 March 2011
Available online 19 April 2011

Keywords:
Biosolids
Land application
C mineralization
N mineralization
Cynodon dactylon
C storage

ABSTRACT

The effects of biosolids application rate and history on soil potential C and N mineralization were measured over a 112-day laboratory incubation. Soils were collected from a large-scale biosolids recycling operation that surface-applies anaerobically digested Class B biosolids for commercial forage production. Five treatments were evaluated: unamended control; 22 Mg dry biosolids ha^{-1} y^{-1} applied for 25 years; 22, 45, and 67 Mg ha^{-1} y^{-1} applied for 8 years. Biosolids additions enhanced total soil organic C by 32–92% and total N by 30–157% compared to unamended soils. Total N increased with application rate and was dominated by nitrate-N. Potential C mineralization (cumulative CO_2 produced) was 11–62% greater in amended soils compared to controls and highest at 67 Mg ha^{-1} y^{-1}. Net N mineralization and immobilization were highest early in the incubation for 45 and 67 Mg ha^{-1} y^{-1} treatments. No significant differences in potential C and N mineralization between controls and soils amended at the lowest rate for 8 or 25 years suggests that biosolids applications at 22 Mg ha^{-1} y^{-1} are sustainable over the long-term. Higher potential N mineralization rates and soil nitrate concentrations under higher application rates may increase the risk of off-site nutrient transport and requires further evaluation.

Published by Elsevier B.V.

1. Introduction

A common and economical option for managing municipal biosolids is land application of treated biosolids on agricultural fields. Land-applying biosolids as an alternative means of waste management provides soil conditioning and nutrient fertilization to agricultural systems (Hargreaves et al., 2008; Singh and Agrawal, 2008). The beneficial reuse of biosolids can improve both soil physical and chemical properties, including increased aggregate stability (Ojeda et al., 2003; Tejada and Gonzalez, 2007), soil water content (Nielsen et al., 2003; Kelly et al., 2007), soil organic C (SOC) (Klavidko and Nelson, 1979; Crecchio et al., 2001; Parat et al., 2005), and soil nitrogen (N) and phosphorus (P) (Kelly et al., 2007; Bayley et al., 2008). Currently, municipal biosolids applications are regulated both at the federal and state level by an agronomic N rate approach (USEPA, 1983, 1992, 1993; Cogger et al., 2006). Regulatory compliance minimizes potential nutrient transport from biosolids-applied areas by matching crop N requirements to biosolids N availability (Cogger et al., 2006). When applied at agronomically acceptable rates, biosolids-derived macronutrients and organic constituents have been linked to increased plant growth and greater agricultural productivity (Nielson et al., 1998; Speir et al., 2004; Sigua and Adjei, 2005).

The potential benefits of biosolids reuse, however, can be limited when crop requirements are exceeded. Such scenarios include (but are not limited to) cases when crop production is impacted by atypical weather conditions (i.e. prolonged drought, hail, and wind damage) or biological agents (i.e. pests and disease), or under very long-term applications where the availability of residual soil nutrients may be in excess of crop N need. The long-term sustainability of biosolids reuse systems, therefore, will depend on optimizing biosolids applications with agronomic practices that account for site-specific management history, soil characteristics, and weather conditions (Bastian, 2005; Cogger et al., 2006; Elliott and O'Connor, 2007).

Numerous studies have evaluated the soil impacts of short-term applications (<5 years) (Cogger et al., 1999; Barbarick and Ippolito, 2000; Withers et al., 2001; Fernandes et al., 2005; Sigua and Adjei, 2005; Gibbs et al., 2006; Oladeji et al., 2007). In contrast, studies evaluating long-term, repeated biosolids applications are limited (Mullen et al., 2005; Barbarick and Ippolito, 2007), particularly under large-scale field operations (Zerzghi et al., 2009). The effects of biosolids application rate and application history on soil quality can be assessed by measuring carbon mineralization (i.e. respiration) as an indicator of soil microbial activity (Franzluebbers et al., 2000; Haney et al., 2001; Haney and Franzluebbers, 2009). Further,

* Corresponding author. Tel.: +1 402 472 5137; fax: +1 402 472 0516.
E-mail address: Virginia.Jin@ars.usda.gov (V.L. Jin).

0167-8809/$ – see front matter. Published by Elsevier B.V.
doi:10.1016/j.agee.2011.03.016

Author's personal copy

the availability of C often controls microbial N process rates (i.e. mineralization and immobilization), resulting in strong positive correlations between microbial N transformations and soil respiration rates (Haney et al., 2001; Luxhøi et al., 2006). The objective of this study was to evaluate potential C and N mineralization in surface soils (0–10 cm) from a large-scale municipal biosolids recycling facility where large tracts (14–48 ha) have had different annual rates of biosolids applications for different lengths of time (8 years and 25 years). The site surface applies anaerobically digested, Class B municipal biosolids to perennial forage fields under no-till management for commercial hay production (coastal Bermudagrass, or *Cynodon dactylon* L.). Soil total and water-extractable C and N pools and C and N mineralization rates were quantified during a 112-day laboratory incubation study.

2. Materials and methods

2.1. Site description and soil collection

The study site is a municipally operated 485-ha, zero-discharge facility in Travis County, TX and is bounded by 5 km of the Colorado River. Approximately 220 ha of the facility are used in a year-round, continuous field rotation for hay production using coastal Bermudagrass (*Cynodon dactylon* L.). Currently, one-third of the total biosolids processed by the facility is land-applied as anaerobically digested, Class B biosolids. Following anaerobic digestion, biosolids are belt-pressed to reduce water content then loaded into manure-spreaders for surface application to a perennial no-tillage forage crop system. Annual mean characteristics of dry biosolids based on monthly analysis (November 2006–October 2007) were pH 8.46 (SM 4500-H B); 17.5% total solids (SM 2540 G); 10,745 mg NH_3-N kg^{-1} (SM 4500-NH_3 D); 14.7 mg $NO_2 + NO_3$-N kg^{-1} (EPA 353.2); 52,127 mg TKN kg^{-1} (EOA 351.2). Regulated biosolids elemental constituents for dry biosolids at all samplings fall well below the EPA 40 CRF Part 503 limits (USEPA, 1992, 1993).

A subset of fields within the 220-ha area was sampled for soils used in the present study. Since 1985, a 14 ha field has been treated with 22 dry Mg biosolids ha^{-1} y^{-1} (25-year treatment), an agronomic rate below the maximum annual metal loading rate regulated by the Texas Commission of Environmental Quality, TCEQ; 30 TAC §312.43(b)(4); TAC (2005). Annual applications in this field, however, were changed to a single application every 2-years (i.e. bi-annual applications) starting in 2006 to comply with nutrient management plan recommendations to address high soil N levels. In 2001, 105 ha of adjacent row-crop land was acquired and approved by the former Texas Natural Resource Conservation Commission (now TCEQ) as an experimental exemption area to evaluate the potential soil impacts of applying higher rates of biosolids amendments. The exemption area was split into 20 ha, 36, ha, and 49 ha fields for experimental applications of 22, 45, and 67 Mg ha^{-1} y^{-1}, respectively, starting in 2002 (8-year treatment). This newer area was converted to coastal Bermudagrass hay production in 2003. A 2 ha unamended control field is located next to the lowest application rate field. The control field has not received applications of biosolids or commercial fertilizer of any kind since 2000. Soils across the site are Bergstrom silt-loams (fine-silty, mixed, superactive, and thermic Cumulic Haplustolls) and are very deep, well-drained, and moderately permeable, with slopes <1%.

2.2. Incubation design and soil analyses

Surface soils (0–10 cm; 20 cm × 20 cm sampling area) were collected in January 2009 from each field (unamended control; 22 dry Mg biosolids ha^{-1} y^{-1} for 25 years; 22, 45, and 67 Mg ha^{-1} y^{-1} for 8 years) using a hand spade. No biosolids had been applied to any of the treatment fields for at least 3 months prior to soil sampling. Visible biosolids and plant material were removed from the soil surface prior to sampling. Soils were sampled from three random locations per treatment area and composited by treatment. Soils were sieved through a 2 mm mesh, large roots removed, and air-dried prior to laboratory incubations. Separate soil cores (4.2 cm diameter) also were collected for determining bulk density in each treatment field.

Triplicate soil samples from each treatment were moistened to 50% water-filled pore space. Each soil sample was incubated individually at 25 ± 1 °C in the dark. Potential C mineralization was measured as carbon dioxide (CO_2) evolution in all replicates. For each replicate, 40 g of soil was placed in a 50 mL disposable beaker and incubated in 1-quart canning jar containing vials with 10 mL of 1.0 *M* KOH to absorb CO_2 and sufficient water to maintain humidity. Alkali traps were replaced daily for the first 3 days of incubation, then weekly until the end of the incubation period. Blank incubation jars were used (i.e. all components except soil present) to calculate blank corrections in samples collected from incubation jars containing soils. Carbon dioxide evolved was determined by titration to neutral pH with 1.0 M HCl after the addition of 2 M $BaCl_2$ (Anderson, 1982). A total of 75 soil incubation jars were used in this study (5 treatments × 3 replicates × 5 destructive harvests, not including Day 0 soils; see below).

Soil concentrations of C and N and net N mineralization rates were measured over time using destructive sampling in triplicate for each treatment at Days 0, 7, 13, 28, 56, and 112. Initial total soil organic C (SOC) and total N (TN) in soils was determined by dry combustion at 680 °C (Vario Max CHN, Elementar, Hanau, Germany). Soils were extracted with water (10:1 deionizied water:dry soil ratio) to simulate available soil nutrients following a natural rain event, as would occur in a field setting. Water-extracts were measured for water-soluble organic C (WSOC) and organic N (WSON) to evaluate the potential in-field availability of labile C and N for microbial consumption (Apollo 9000, Teledyne Tekmar, Mason, OH). Inorganic N concentrations in water extracts were measured using continuous flow colorimetry (Flow Solution IV, OI Analytical, College Station, TX). Soil C and N concentrations in the 0–10 cm soil layer were converted to an area basis using soil bulk densities (Mg ha^{-1} for SOC, TN; kg ha^{-1} for water-soluble C and N). Net N mineralization (mg N kg^{-1} d^{-1}) over the incubation period was determined as the difference between water-extractable soil inorganic N concentrations (NH_4^+–N + NO_2^-–N + NO_3^-–N) between two consecutive destructive sampling days.

2.3. Statistical analyses

One-way analysis of variance (ANOVA) was used to test the fixed effect of biosolids treatment on cumulative CO_2 production and initial soil C and N concentrations. Two-way ANOVAs were used to test the fixed effects of biosolids treatment and sampling day on net N mineralization rates and changes in soil C and N concentrations over time. Data were tested for normality using the Shapiro–Wilk statistic and transformed when necessary. Post hoc multiple comparisons between significant treatment means were tested for differences using the Fisher's least significant difference (LSD) procedure. Linear regressions were used to examine correlations between soil potential C and N mineralization and total and water-soluble C and N concentrations. All statistical tests were performed using SAS 9.1 (SAS, Inc., Cary, NC).

3. Results and discussion

Initial SOC in biosolids-amended soils prior to incubation were 32–92% greater than unamended control soils, with highest SOC

Author's personal copy

Table 1
Mean initial soil pH, bulk densities, and total and water-soluble C and N in surface soils (0–10 cm) collected from control and biosolids-applied fields.[a]

Biosolids application rate (Mg dry biosolids ha^{-1} y^{-1})						
Constituent	Unit	0 Control	22 25-Year	22	45 8-Years	67
pH (1:10)		7.70a	7.35b	7.40b	7.37b	7.22c
Bulk density	g cm^{-3}	1.47a	1.47a	1.48a	1.47a	1.46a
Total SOC	Mg ha^{-1}	31.4a	45.7b	41.8b	48.1b	59.8c
Total N	Mg ha^{-1}	3.0a	4.6b	3.8c	5.6d	9.1e
Total C:N		10.4ab	9.9a	11.6b	8.7c	6.5d
WSOC	kg ha^{-1}	545ab	521ab	534ab	483a	555b
WSON	kg ha^{-1}	37.7a	45.9a	55.0a	143.5b	165.4b
Soluble org C:N		14.5a	11.6b	9.8b	3.4c	4.8c
NH_4^+-N	kg ha^{-1}	7.5a	7.9a	8.4a	10.0b	12.6c
NO_3^--N	kg ha^{-1}	11.1a	55.9b	71.4b	374.7c	573.6d
Inorganic N	kg ha^{-1}	18.6a	63.8b	79.8b	384.7c	586.2d
Total soluble N	kg ha^{-1}	56.3a	109.7a	134.8a	528.2b	751.7c
Total soluble C:N		9.7a	4.7b	4.1c	0.9d	0.7e

[a] $n = 3$ per treatment. Different letters across each row indicate significant differences ($P \leq 0.05$) between application treatments for the measured soil constituent.

in soils applied at the highest rate ($P \leq 0.05$) (Table 1). Increases suggested that dissolved organic C derived from increasing rates of surface-applied biosolids had infiltrated into surface soils, resulting in increased SOC concentrations. Initial differences in SOC were reflected in potential C mineralization patterns, with cumulative CO_2 production (mg CO_2-C kg^{-1} soil) increasing up to 62% relative to controls in the 67 Mg biosolids ha^{-1} y^{-1} treatment ($P \leq 0.05$) (Fig. 1). Our results are similar to other studies that have found increased C mineralization following land application of biosolids (Filcheva et al., 1996; Gibbs et al., 2006; Kao et al., 2006; Khan and Scullion, 2002).

Daily C mineralization rates (mg CO_2-C kg^{-1} soil d^{-1}) in all soils showed short-term, high respiration rates followed by steep declines as shown in previous studies (Chander et al., 1995; García-Gil et al., 2000; Pascual et al., 2004; Sánchez-Monedero et al., 2004). High C mineralization rates at the beginning of the incubation period likely corresponded to greater initial C availability originating from the turnover of microbial biomass following the re-wetting of initially dry soils (Fierer and Schimel, 2002). Further, daily C mineralization rates in soils treated at the highest rate were persistently greater than the other treatments throughout the 112-day incubation period (Fig. 2). The short-term CO_2 evolution from soils immediately following rewetting is positively correlated with both basal soil respiration and microbial biomass C (Franzluebbers et al., 2000; Haney et al., 2001). Repeated dry and rewetting events are representative of field conditions, particularly in the central Texas climate where long dry periods are punctuated by rainfall events throughout the year. Thus, significant and persistently higher C mineralization in soils treated with the highest application rate suggests that microbial activity levels and/or microbial biomass may have been greater due to increased C availability.

Alternatively, treatment effects on C mineralization may be attributed to shifts in microbial community composition/structure or to changes in microbial C use patterns, but such measurements were beyond the scope of the present study. In studies where these measurements have been made, soil microbial community

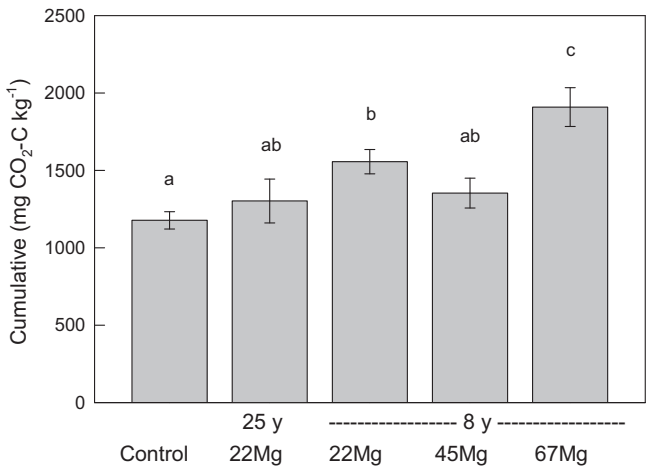


Fig. 1. Mean cumulative CO_2 production (mg CO_2-C kg^{-1} soil; ±se) over the 112-d incubation period. Different letters indicate significant treatment differences between application rates ($P \leq 0.05$).

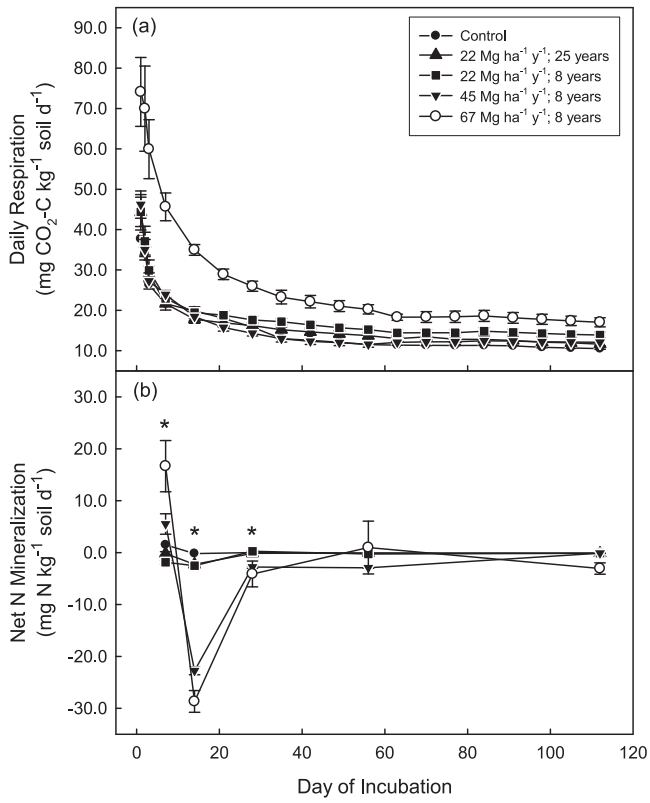


Fig. 2. (a) Mean daily respiration rates (mg CO_2-C kg^{-1} soil d^{-1}; ±se) for soils from different biosolids application treatments. (b) Changes in mean net N mineralization rates (mg N kg^{-1} soil d^{-1}; ±se) over time in soils from different biosolids treatments. Positive values indicate net N mineralization (i.e. production), and negative values indicate net immobilization. Asterisks indicate significant treatment differences ($P \leq 0.05$).

Author's personal copy

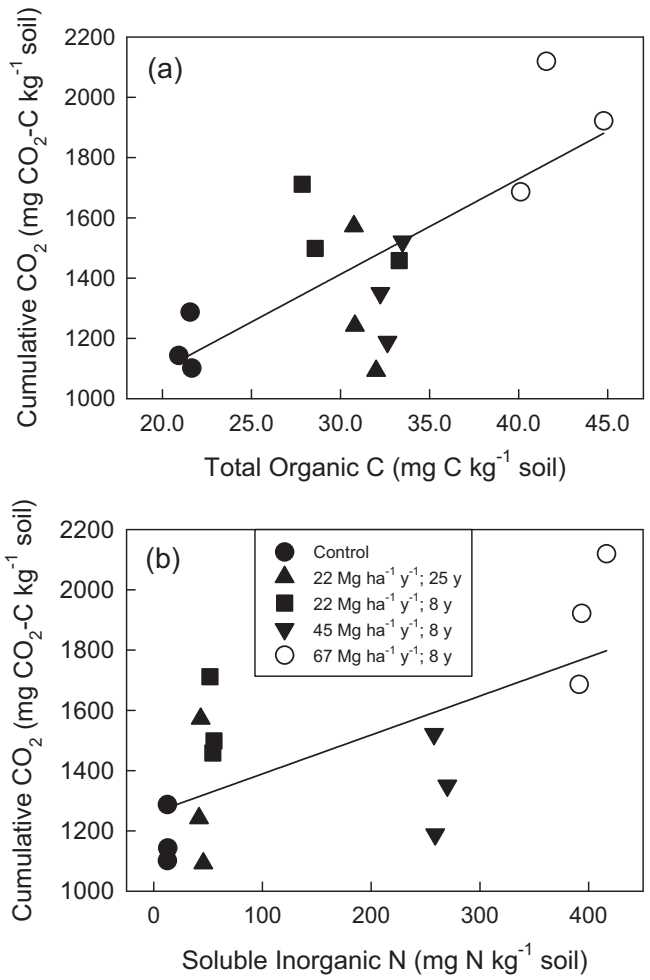


Fig. 3. Correlations between soil cumulative CO_2 production (mg CO_2-C kg^{-1} soil) and soil concentrations (mg kg^{-1} soil) of (a) total organic C ($y = 31.7x + 463$; $R^2 = 0.53$; $P = 0.002$); (b) soluble inorganic N ($y = 1.3x + 1260$; $R^2 = 0.44$; $P = 0.007$).

diversity and composition differ in biosolids-amended agricultural soils compared to control soils, primarily in response to increased SOC (Anderson et al., 2008; Macdonald et al., 2007; Sullivan et al., 2006b). However, other studies on alkaline soils such as those used in the present study indicate that biosolids applications can alter microbial C use patterns (Lawlor et al., 2000; Sullivan et al., 2006a), but do not change soil microbial diversity or shift the composition of specific microbial taxonomic groups involved in soil fertility (Crecchio et al., 2004).

Potential C mineralization correlated positively with both total SOC and extractable inorganic N concentrations (Fig. 3a and b), which also has been observed in other studies (García-Gil et al., 2000; Gibbs et al., 2006; Pascual et al., 1999). Carbon mineralization, however, was not related to initial labile C availability in the form of water-soluble organic C (WSOC). The absence of a relationship with WSOC coupled with the positive correlation with inorganic N suggests an N-limitation rather than a C-limitation to microbial mineralization of carbon from soil organic matter in these biosolids-amended soils.

Similar to treatment patterns in initial SOC, initial TN in amended soils was significantly greater than unamended control soils and increased by 30–157% with increasing application rate (Table 1). Although the majority of labile N (i.e. water-soluble N) was made up of organic forms in unamended soils, the water-soluble N pool became dominated increasingly by inorganic N, specifically nitrate, as application rate increased ($P \leq 0.05$) (Table 1). Further, increases in soluble inorganic N pools in soils from the experimental exemption area were not directly proportional to increases in application rates, with 5–8-fold increases in nitrate concentrations associated with doubling and tripling of the lowest application rate, respectively.

In addition to significant changes in inorganic pool size, rates of net N mineralization (i.e. positive values) and immobilization (i.e. negative values) were greatest in soils amended with the two highest application rates (Fig. 3b). The greatest fluxes in net N mineralization and immobilization rates tended to co-occur with high daily respiration rates early in the incubation period, consistent with a resource pulse response (i.e. rainfall event) that would trigger greater microbial activity associated with higher C substrate availability (Hernández et al., 2002; Sullivan et al., 2006a). Significant treatment differences in net N transformation rates on Days 7, 14, and 28 did not persist to the end of the incubation period. The quantity of organic N mineralized within 7 days of adding water to soils treated at the two higher application rates almost doubled the extractable concentration of soil inorganic N, which was quickly immobilized by Day 14. Rapid initial net N mineralization, along with overall increases in soluble inorganic N, emphasize the potential for increased N movement off-site via leaching or run-off (Cogger et al., 2006; Zhang et al., 2006), particularly if multiple rain events occurred within days of each other before soils could immobilize N. While repeated rain events within a short period of time can occur, single rainfall events with long dry periods in between are much more common in central Texas. Under the single rainfall scenario, it is likely that soils would dry out within a few days following rewetting. Soil dessication would constrain microbial activity, including the immobilization of nitrate, and lead to the subsequent build-up of nitrate in the soil. This build-up was consistent with both the prevailing climate history at this site as well as the high overall nitrate concentrations in field soils collected for this incubation study.

4. Conclusions

Beneficial reuse of municipal biosolids via land application increased potential C mineralization and net N mineralization and immobilization in soils amended at the highest application rates. Long-term increases in SOC in this no-till forage production system using surface-applied biosolids indicate the long-term potential of C storage, even at the lowest application rate. At this municipal biosolids recycling site, surface applications of biosolids at the lowest rate appear to be sustainable in the long-term. Higher potential N mineralization rates and nitrate concentrations in soils amended with 45 and 67 Mg dry biosolids ha^{-1} y^{-1}, however, may limit long-term use of these application rates, as the resulting excess soil N levels increase the risk of off-site nutrient transport via run-off or leaching and requires further evaluation.

Acknowledgments

This research was supported by a grant from the USDA-CSREES National Research Initiative Managed Ecosystems Program (2008-35101-04440 to JGA). The authors gratefully acknowledge E. Witherington for her role in the laboratory incubations. We also thank K. Anderson, D. Greene, J. Slagle, and D. White from the study site for their cooperation, E. Janacek and C. Kolodziejczyk for field assistance, A. Gibson for laboratory assistance, and S. Duke for statistical consulting.

References

Anderson, I.C., Parkin, P.I., Campbell, C.D., 2008. DNA- and RNA-derived assessments of fungal community composition in soil amended with sewage sludge rich in cadmium, copper and zinc. Soil Biol. Biochem. 40, 2358–2365.

Author's personal copy

Anderson, J.P.E., 1982. Soil respiration. In: Page, A.L., Miller, R.H., Keeney, D.R. (Eds.), Methods of Soil Analysis. Part 2. Chemical and Microbiological Properties. Soil Science Society of America, Madison, WI, pp. 831–871.
Barbarick, K.A., Ippolito, J.A., 2000. Nitrogen fertilizer equivalency of sewage biosolids applied to dryland winter wheat. J. Environ. Qual. 29, 1345–1351.
Barbarick, K.A., Ippolito, J.A., 2007. Nutrient assessment of a dryland wheat agroecosystem after 12 years of biosolids applications. Agron. J. 99, 715–722.
Bastian, R.K., 2005. Interpreting science in the real world for sustainable land application. J. Environ. Qual. 34, 174–183.
Bayley, R.M., Ippolito, J.A., Stromberger, M.E., Barbarick, K.A., Paschke, M.W., 2008. Water treatment residuals and biosolids coapplications affect semiarid rangeland phosphorus cycling. Soil Sci. Soc. Am. J. 72, 711–719.
Chander, K., Brookes, P.C., Harding, S.A., 1995. Microbial biomass dynamics following addition of metal-enriched sewage sludges to a sandy loam. Soil Biol. Biochem. 27, 1409–1421.
Cogger, C.G., Sullivan, D.M., Bary, A.I., Fransen, S.C., 1999. Nitrogen recovery from heat-dried and dewaterered biosolids applied to forage grasses. J. Environ. Qual. 28, 754–759.
Cogger, C.G., Forge, T.A., Neilsen, G.H., 2006. Biosolids recycling Nitrogen management and soil ecology. Can. J. Soil Sci. 86, 613–620.
Crecchio, C., Curci, M., Mininni, R., Ricciuti, P., Ruggiero, P., 2001. Short-term effects of municipal solid waste compost amendments on soil carbon and nitrogen content, some enzyme activities and genetic diversity. Biol. Fertil. Soils 34, 311–318.
Crecchio, C., Curci, M., Pizzigallo, M.D.R., Ricciuti, P., Ruggiero, P., 2004. Effects of municipal solid waste compost amendments on soil enzyme activities and bacterial genetic diversity. Soil Biol. Biochem. 36, 1595–1605.
Elliott, H.A., O'Connor, G.A., 2007. Phosphorus management for sustainable biosolids recycling in the United States. Soil Biol. Biochem. 39, 1318–1327.
Fernandes, S.A.P., Bettiol, W., Cerri, C.C., 2005. Effect of sewage sludge on microbial biomass, basal respiration, metabolic quotient and soil enzymatic activity. Appl. Soil Ecol. 30, 65–77.
Fierer, N., Schimel, J.P., 2002. Effects of drying-rewetting frequency on soil carbon and nitrogen transformations. Soil Biol. Biochem. 34, 777–787.
Filcheva, E., Cheshire, M.V., Campbell, C.D., McPhail, D.B., 1996. Effect of heavy metal contamination on the rate of decomposition of sewage sludge and microbial activity. Appl. Geochem. 11, 331–333.
Franzluebbers, A.J., Haney, R.L., Honeycutt, C.W., Schomberg, H.H., Hons, F.M., 2000. Flush of carbon dioxide following rewetting of dried soil relates to active organic pools. Soil Sci. Soc. Am. J. 64, 613–623.
García-Gil, J.C., Plaza, C., Soler-Rovira, P., Polo, A., 2000. Long-term effects of municipal solid waste compost application on soil enzyme activities and microbial biomass. Soil Biol. Biochem. 32, 1907–1913.
Gibbs, P.A., Chambers, B.J., Chaudri, A.M., McGrath, S.P., Carlton-Smith, C.H., Bacon, J.R., Campbell, C.D., Aitken, M.N., 2006. Initial results from a long-term, multi-site field study of the effects on soil fertility and microbial activity of sludge cakes containing heavy metals. Soil Use Manage. 22, 11–21.
Haney, R.L., Franzlubbers, A.J., 2009. Soil CO_2 evolution: response from arginine additions. Appl. Soil Ecol. 42, 324–327.
Haney, R.L., Hons, F.M., Sanderson, M.A., Franzleubbers, A.J., 2001. A rapid procedure for prediction of N mineralization in manured soils. Biol. Fertil. Soils 33, 100–104.
Hargreaves, J.C., Adl, M.S., Warman, P.R., 2008. A review of the use of composted municipal solid waste in agriculture. Agric. Ecosyst. Environ. 123, 1–14.
Hernández, T., Moral, R., Perez-Espinosa, A., Moreno-Caselles, J., Perez-Murcia, M.D., García, C., 2002. Nitrogen mineralization potential in calcareous soils amended with sewage sludge. Bioresour. Technol. 83, 213–219.
Kao, P.H., Huang, C.C., Hseu, Z.Y., 2006. Response of microbial activities to heavy metals in a neutral loamy soil treated with biosolid. Chemosphere 64, 63–70.
Kelly, J.J., Favila, E., Hundal, L.S., Marlin, J.C., 2007. Assessment of soil microbial communities in surface applied mixtures of Illinois River sediments and biosolids. Appl. Soil Ecol. 36, 176–183.
Khan, M., Scullion, J., 2002. Effects of metal Cd, Cu, Ni, Pb or Zn enrichment of sewage–sludge on soil micro-organisms and their activities. Appl. Soil Ecol. 20, 145–155.
Klavidko, E.J., Nelson, D.W., 1979. Changes in soil properties from application of anaerobic sludge. J. Water Pollut. Control Fed. 51, 325–332.
Lawlor, K., Knight, B.P., Barbosa-Jefferson, V.L., Lane, P.W., Lilley, A.K., Paton, G.I., McGrath, S.P., O'Flaherty, S.M., Hirsch, P.R., 2000. Comparison of methods to investigate microbial populations in soils under different agricultural management. FEMS Microbiol. Ecol. 33, 129–137.
Luxhøi, J., Bruun, S., Stenberg, B., Breland, T., Jensen, L.S., 2006. Prediction of gross and net nitrogen mineralization-immobilization-turnover from respiration. Soil Sci. Soc. Am. J. 70, 1121–1128.
Macdonald, C.A., Singh, B.K., Peck, J.A., van Schaik, A.P., Hunter, L.C., Horswell, J., Campbell, C.D., Speir, T.W., 2007. Long-term exposure to Zn-spiked sewage sludge alters soil community structure. Soil Biol. Biochem. 39, 2576–2586.
Mullen, R.W., Raun, W.R., Basta, N.T., Schroder, J.F., Freeman, K.W., 2005. Effect of long-term application of biosolids on molybdenum content and quality of winter wheat forage. J. Plant Nutr. 28, 405–420.
Nielsen, G.H., Hogue, E.J., Forge, T., Nielsen, D., 2003. Surface application of mulches and biosolids affect orchard soil properties after 7 years. Can. J. Soil Sci. 83, 131–137.
Nielson, G.H., Hogue, E.J., Nielson, D., Zebarth, B.J., 1998. Evaluation of organic wastes as soil amendments for cultivation of carrot and chard on irrigated sandy soils. Can J. Soil Sci. 78, 217–225.
Oladeji, O.O., O'Connor, G.A., Sartain, J.B., Nair, V.D., 2007. Controlled application rate of water treatment residual for agronomic and environmental benefits. J. Environ. Qual. 36, 1715–1724.
Ojeda, G., Alcaniz, J.M., Ortiz, O., 2003. Runoff and loses by erosion in soils amended with sewage sludge. Land Degrad. Dev. 14, 563–573.
Parat, C., Chaussod, R., Leveque, J., Andreux, F., 2005. Long-term effects of metal-containing farmyard manure and sewage sludge on soil organic matter in a fluvisol. Soil Biol. Biochem. 37, 673–679.
Pascual, I., Antolín, M.C., García, C., Polo, A., Sánchez-Díaz, M., 2004. Plant availability of heavy metals in a soil amended with a high dose of sewage sludge under drought conditions. Biol. Fertil. Soils 40, 291–299.
Pascual, J.A., García, C., Hernandez, T., 1999. Lasting microbiological and biochemical effects of the addition of municipal solid waste to an arid soil. Biol. Fertil. Soils 30, 1–6.
Sánchez-Monedero, M.A., Mondini, C., de Nobili, M., Leita, L., Roig, A., 2004. Land application of biosolids. Soil response to different stabilization degree of the treated organic matter. Waste Manage. 24, 325–332.
Sigua, G.C., Adjei, M.B., 2005. Cumulative and residual effects of repeated sewage sludge applications: forage productivity and soil quality implications in South Florida, USA. Environ. Sci. Pollut. Res. 12, 80–88.
Singh, R.P., Agrawal, M., 2008. Potential benefits and risks of land application of sewage sludge. Waste Manage. 28, 347–358.
Speir, T.W., Horswell, J., van Schaik, A.P., McLaren, R.G., Fietje, G., 2004. Composted biosolids enhance fertility of a sandy loam soil under dairy pasture. Biol. Fertil. Soils 40, 349–358.
Sullivan, T.S., Stromberger, M.E., Paschke, M.W., 2006a. Long term impacts of infrequent biosolids applications on chemical and microbiological properties of a semi-arid rangeland soil. Biol. Fertil. Soils 42, 258–266.
Sullivan, T.S., Stromberger, M.E., Paschke, M.W., 2006b. Parallel shifts in plant and soil microbial communities in response to biosolids in a semi-arid grassland. Soil Biol. Biochem. 38, 449–459.
Tejada, M., Gonzalez, J.L., 2007. Application of different organic wastes on soil properties and wheat yield. Agron. J. 99, 1597–1606.
Texas Administrative Code, 2005. Land Application for Beneficial Use and Storage at Beneficial Use Sites; 30 TAC §312B. Texas Administrative Code, Austin, TX.
US EPA, 1983. Process Design Manual: Land Application of Municipal Sludge. EPA-625/1-83-016. US EPA, Washington, DC.
US EPA, 1992. Technical Support Document for Land Application of Sewage Sludge, Vol. I. EPA/822/R-93/001A. US EPA, Washington, DC.
US EPA, 1993. Standards for the Disposal of Sewage Sludge; Final Rule. 40 CRR-Part 257 and 503. US EPA, Washington, DC.
Withers, P.J.A., Clay, S.D., Breeze, V.G., 2001. Phosphorus transfer in runoff following application of fertilizer, manure, and sewage sludge. J. Environ. Qual. 30, 180–188.
Zerzghi, H., Gerba, C.P., Brooks, J.P., Pepper, I.L., 2009. Long-term effects of land application for class B biosolids on the soil microbial populations, pathogen, and activity. J. Environ. Qual. 39, 402–408.
Zhang, M., Heaney, D., Henriquez, B., Solberg, E., Bittner, E., 2006. A four-year study on influence of biosolids/MSW co-compost application in less productive soils in Alberta, nutrient dynamics. Compost. Sci. Util. 14, 68–80.