American-Eurasian J. Agric. & Environ. Sci., 8 (6): 695-704, 2010
ISSN 1818-6769

A Preliminary Study on the Potential for Cycling Solids from Geotube Treatment of Dairy Lagoon Wastewater

[1]Ronnie W. Schnell, [1]Donald M. Vietor, [2]Clyde L. Munster,
[1]Richard H. White, [1]Tony L. Provin and [2]Saqib Mukhtar

[1]Soil and Crop Science Department, Texas A&M University,
370 Olsen Blvd. College Station, TX 77843-2474
[2]Biological and Agricultural Engineering Department, Texas A&M University,
2117 Scoates Hall, College Station, TX 77843-2474

Abstract: Geotube® residual solids (GRS) are a potential bioresource that could be used to improve soil physical and chemical properties on agricultural land. Yet, chemical properties of GRS could affect plant growth and repeated land application of GRS could elevate soil nutrient content and contribute transport and loss of nutrients in drainage water. The first objective was to evaluate the effects of increasing rates of GRS on physical, chemical and biological properties of contrasting soil types during turfgrass establishment. The second objective was to evaluate leaching losses of nutrients from contrasting soil types with and without incorporation of GRS during turfgrass establishment. Incorporation of GRS enhanced turfgrass growth and physical and chemical properties of contrasting soil types. However, following application of GRS, soil concentration of NO_3-N exceeded turfgrass uptake and excessive leaching loss of NO_3-N occurred. At high soil pH, residual Alum and associated hydrolysis products in GRS did not increase extractable soil Al, but did minimize leachate concentrations of dissolved reactive P forms. Composition, timing and application rate of GRS should be managed to optimize turfgrass establishment and prevent leaching loss of dissolved nutrients.

Key words: Geotube® · Phosphorus · Carbon · Nitrate · Turfgrass

INTRODUCTION

Geotube® Dewatering Systems (Miratech Division of Ten Cate Nicolon Corporation; Commerce, Ga.) collect solids (GRS) from waste liquids of municipal wastewater and septic systems, confined animal feeding operations and contaminated marine environments. Solids are retained during pumping of waste liquids into large porous tubes or socks (~ 300 m^3) constructed of synthetic fibers. In applications designed to separate solids from runoff and process wastewater of confined dairy feeding operations, Alum (Aluminum sulfate) and polyacrylamide are injected during pumping of waste liquid into the Geotube® [1]. Compared to untreated waste liquid, the metal salt and polymers enhanced flocculation of particulates and associated nutrients and increase water flow rate through the Geotube® wall [2]. Water draining from the Geotube® was returned to source lagoons or separate retention ponds. For liquid pumped from an agitated dairy wastewater lagoon, the Geotube® system reduced total solids 93.5%, soluble P 85% and total P 96% [1].

Although GRS solids separated from liquid wastes of industrial and municipal sources are often landfilled, GRS from wastewater of livestock operations is a potential bioresource for agricultural lands. Similar to manure, GRS could increase soil organic C (SOC) and provide essential mineral nutrients on fields proximate to livestock and composting operations [3-5]. In addition, the GRS sources of nutrients could produce and be exported through harvests of marketable crop products. Yet, repeated land application of GRS at rates exceeding crop nutrient requirements could exceed regulatory limits on soil nutrient concentrations [6]. The high soil concentrations contribute to nutrient transport and loss in water draining through soil [4, 7]. Sustainable systems for managing GRS will require maintenance of field, landscape and regional nutrient balances [8].

Corresponding Author: Ronnie Schnell, West Florida Research and Education Center, University of Florida, 4253 Experiment Road,HWY 182, Jay, FL 32565 (850) 995-3722, E-mail: rschnell@ufl.edu.

Turfgrass sod is among the marketable crops through which GRS sources of nutrients and organic C offer environmental and economic benefits [9]. Harvest of a shallow layer of GRS-amended soil with turfgrass sod, a non-food crop, exports more nutrients and organic C from fields on which GRS are applied than forage, grain, or fiber crops [10]. In addition, the GRS and associated polymer within the layer of harvested sod could reduce removal of native soil and enhance soil structure and water infiltration and storage compared to sod grown in mineral soil [11, 12]. Moreover, the GRS sources of nutrients within the harvested sod layer will reduce fertilizer requirements of the transplanted sod [9].

Although Alum injection enhances flocculation and separation of solids and P from waste liquid in the Geotube®, Alum hydrolysis products could reduce pH and plant growth in GRS-amended soil [5]. Yet, Alum mixed with composted biosolids reduced neither pH or grass establishment compared to soil amended with biosolids only [13]. Before volume-based GRS rates are used to amend soil for turfgrass establishment under field conditions, effects on plant growth and soil physical, chemical and biological properties need to be evaluated [14-16].

The first objective of this study was to evaluate effects of increasing rates of GRS on physical, chemical and biological properties of contrasting soil types and turfgrass establishment. The second objective was to evaluate leaching losses of nutrients from contrasting soil types with and without incorporation of GRS during turfgrass establishment.

MATERIALS AND METHODS

Mineralization Study: A complete randomized block design composed of four replications of the factorial combination of two soil types (Windthorst fine sandy loam; fine, mixed, thermic Udic Paleustalfs and Weswood silt loam; fine, mixed, thermic Fluventic Ustochrept) and three rates of GRS were implemented in incubation vessels (0.5 L) to evaluate C and N mineralization rates. A control and two volume-based GRS rates (0.125 and 0.25 $m^3 m^{-3}$) were mixed in 80 g dry soil, wetted to 60% of soil water holding capacity and incubated for 56 days at 25°C [17]. Alkali traps (0.25 N NaOH) absorbed CO_2 and were titrated with acid (0.25 N HCl) at 1, 2, 3, 7, 14, 21, 28, 35, 42, 49, 56 days to determine carbon mineralization rates [17]. Extractable (1 M KCl) inorganic N (NO_3-N + NH_4-N) concentrations in soil mixtures were determined colorimetrically before and after incubation to estimate N mineralization rates [18].

The percent of GRS applied C remaining was calculated for each sampling date as described previously [19]. The C respired from soil without GRS was subtracted from the amount respired from soil with GRS for each soil type before subtracting from the amount of C applied with GRS. The percent C remaining for each soil type and level of GRS rate was plotted versus time. The datum was fit to a four parameter double exponent decay model described by Berndt [20]. The model included parameters for fast and slow decomposing fractions of GRS. Fitting the model for percent GRS C remaining for each soil type and rate over time was done using SPSS 18.0 and the Levenberg-Marquardt algorithm to find the best fit.

Column Lysimeter Study: A complete randomized block design comprised four replications of a factorial arrangement of two soil types and three rates of GRS in column lysimeters (10-cm diameter x 30-cm depth) under greenhouse conditions. Soil for each a Windthorst fine sandy loam (fine, mixed, thermic Udic Paleustalfs) and Weswood silt loam (fine, mixed, thermic Fluventic Ustochrept) was air dried and sieved (< 10 mm). The GRS was collected after 1 yr of storage in a Geotube® that was used to remove sediment and P from wastewater of a confined dairy feeding operation (Mukhtar *et al.* 2007). The Geotubes® were used to treat 275,400 L of dairy wastewater and were injected with 19 L of Cytec #4512 polymer, 9 L of Cytec #4516 polymer and 454 L of Aluminum sulfate (Alum) [1]. The air-dried GRS was sieved through a 10-mm mesh screen before sampling, analysis and incorporation in soil. Soil with or without GRS was packed within 5-cm increments to achieve a consistent bulk density throughout a 30-cm depth over a layer of glass fiber cloth, which separated soil from a 5-cm depth of washed pea gravel. Two volume-based GRS rates (0.125 and 0.25 $m^3 m^{-3}$) were incorporated in soil packed within the top 10 cm of lysimeters as two 5-cm depth increments. Tifway bermudagrass (Cynodon dactylon L. Pers. X C. transvaalensis Burtt-Davey) was sprigged in all treatments after soil was firmed into columns. After sprigging, the hydrostatic pressure of a water column was used initially to wet soil from the bottom to soil surface within lysimeters. Excess water in each column, which drained through a fitting at the bottom, was collected for analysis. Well water was deionized through an in-line cartridge prior to use for irrigation of lysimeters.

One pore volume of leachate was displaced and collected from each column through surface irrigation with distilled water at 45 and 90 days after sprigging. Pore volumes were calculated as the lysimeter volume multiplied by [1- (bulk density/particle density)].

Leachate volumes were measured, sub-sampled and filtered (< 0.45 im). Dissolved reactive P (DRP), NH_4-N and NO_3-N in filtrate were analyzed colorimetrically within 24 hr after leachate was collected [18, 21]. Inductively coupled plasma optical emission spectroscopy (ICP) was used to measure total dissolved P (TDP) in filtrate [22]. Dissolved unreactive P (DUP) was calculated as the difference between TDP and DRP and is comprised . An Elementar Rapid Liquid TOC analyzer (Hanau, Germany) was used to analyze dissolved organic carbon (DOC) in filtrate.

Before packing in lysimeters, soil and GRS were sampled, dried, ground (< 2 mm) for analysis. Soil was digested using nitric acid and total P determined using ICP [23]. In addition, P was extracted using water (WEP) and the Mehlich-3 solution (M3P) [21, 24]. Soil concentration of total carbon and N was determined using combustion procedures [25]. In addition, lysimeters were cut into segments at 90 d after sprigging for sampling and analyses of soil from 0- to 10-cm and 10- to 30-cm depths. Soil water content was determined gravimetrically for oven-dried soil (60°C, 48 hr). Turfgrass was clipped to a 5-cm height when plant height exceeded 15 cm. Clippings were dried (60°C, 48 hr), composited over cutting dates, weighed, ground and digested for analysis of total N and P [23]. Total dry matter production over 90 days was used to compute average daily biomass production (g m^{-2} d^{-1}).

Analysis of variance was performed using SPSS 18.0 (Chicago, Illinios) to compare leaching losses and soil concentrations of nutrients and organic C among establishment treatments. Leaching events were analyzed separately. Soil and leaching data were analyzed as a complete randomized block design. If interaction was observed between soil type and GRS rate ($P < 0.05$), soil types were analyzed separately. Fisher's least significant difference test ($P = 0.05$) was used to compare treatment means. Regression analysis was used to evaluate relationships between soil type and leachate concentrations of P and organic C.

RESULTS AND DISCUSSION

Soil and GRS Properties: Total and extractable P and total N concentrations were similar between the Windthorst and Weswood soils before incorporation of GRS (Table 1). In contrast, pH and total organic carbon (TOC) concentration were greater for the Weswood than the Windthorst soil. In addition, extractable NO_3-N concentration of the Windthorst soil was 24-times greater than the Weswood soil. After air drying and storage in the Geotube® under field conditions, total P concentration in GRS was similar to values reported for semisolid dairy manure [26]. In contrast, WEP concentration was 98% lower in GRS than in the semisolid dairy manure (Table 1). The Alum and polymer injected during pumping of waste liquid into the Geotube® could have adsorbed dissolved P forms during dewatering of waste liquid and prevented P dissolution in water extracts of GRS [1]. Previous laboratory-scale studies indicated incorporation of Alum at an Al to total P molar ratio of 1.7 minimized water-soluble P release from anaerobically-digested biosolid [27].

Similar to Alum acidulation of soil, pH in GRS collected after Alum injection in dairy waste liquid was relatively low compared to values ranging from 8.1 to 8.5 for composted or fresh dairy manures (Table 1) [28]. In addition, nitrification of mineralized N in GRS during storage for one year under field conditions could have lowered pH. The NO_3-N concentration in GRS was far greater than values reported for dairy manures and municipal organic waste [29, 30]. Large application rates of GRS could reduce soil pH and affect growth of acid-sensitive crops.

Analysis of soil with and without GRS following 90 d of turfgrass establishment revealed soil pH was lowest ($P < 0.001$) for the highest GRS rate, but pH of the Windthorst soil increased with or without GRS (Table 1, Table 2). Increases in both soil pH and Na concentration within the 10-cm depth at the 90-d sampling indicated Na bypassed the filtration cartridge and was applied in irrigation water. Before filtration of well water, Na concentration was 246 mg L^{-1} and pH was 8.06. Mehlich-3 extractable Na concentration of soil without GRS was an average of 4.2-fold greater than concentration in soil before treatments were imposed (Table 1, Table 2). Although irrigation water was a potential source of Na and dissolved salts, hydrolysis products of Alum in GRS and nitrification of mineralized N could have prevented increases in pH for soil mixed with GRS compared to controls for each soil (Table 2). Previous reports indicated increasing rates of compost or organic amendments increased or decreased soil pH, depending on composition [14]. Similar to the present study, increasing rates of sewage sludge treated with Alum reduced soil pH [16]. In contrast, incorporation of 0.25 m^3 m^{-3} of composted biosolids minimized Alum effects on pH of a sandy loam soil during establishment of Tifway bermudagrass [13].

Table 1: Total Organic Carbon (TOC), Total Nitrogen (TN), Total Phosphorus (TP), Mehlich-3 P (M3P), Water Extractable P (WEP) Mehlich-3 Na (M3Na) and extractable NO_3-N concentration of soil and Geotube® residual solids (GRS) before mixing and packing into columns.

	pH	TOC	TN	TP	M3P	WEP	M3Na	NO_3-N
		---------g kg^{-1}-------------		--mg kg^{-1}---				
Windthorst	6.4	3.0	0.5	211	12.5	4.1	138	45.8
Weswood	8.3	12.1	0.4	285	16.9	4.6	132	1.9
GRS	5.7	221	20.7	1586	-	18.6		740

Table 2: Mean soil pH, bulk density, water content, total organic carbon (SOC), N (TN), P (TP), Mehlich-3 P (M3P), water extractable P (WEP), NO_3-N and NH_4N Mehlich-3 Na (M3Na) concentration (0-10 cm depth) 90 d after incorporation of increasing rates of Geotube® residual solids (GRS). The standard error of the mean is given in parenthesis below mean.

Soil	GRS rate $m^3\ m^{-3}$	pH	Density g cm^{-3}	Soil water kg kg^{-1}	SOC g kg^{-1}	TN	TP	M3P	WEP	NO_3-N	NH_4-N	M3Na
						---mg kg^{-1}---------------------------------------						
Windthorst	0	8.6 (0.05)	1.32(0.11)	0.22(0.02)	3.3(0.1)	413(13)	43(2.9)	16(1.4)	1.9(0.3)	17.9(4.5)	12(2.5)	529(63)
Windthorst	0.125	8.3(0.06)	1.26(0.09)	0.21(0.02)	11.5(0.5)	1213(48)	239(8.9)	95(3.6)	13.7(1.6)	23.5(5.7)	21(5.0)	488(32)
Windthorst	0.25	8.1(0.06)	1.14(0.09)	0.12(0.02)	22.3(1.0)	2050(82)	594(108)	173(5.8)	19.7(3.1)	26.0(5.7)	35(9.3)	524(30)
Weswood	0	8.7(0.09)	1.31(0.09)	0.31(0.02)	4.3(0.2)	475(31)	272(10.6)	21(1.3)	2.5(0.3)	12.7(3.4)	9.9(2.3)	609(79)
Weswood	0.125	8.6(0.07)	1.20(0.08)	0.28(0.03)	11.5(0.2)	1150(19)	432(20.6)	79(3.3)	15.2(1.0)	23.3(4.3)	19(4.9)	713(79)
Weswood	0.25	8.3(0.11)	1.11(0.07)	0.26(0.02)	21.3(0.6)	1913(44)	683(25.9)	151(8.1)	16.4(1.5)	26.4(4.5)	33(8.2)	627(52)
Soil		**	ns	****	ns	ns	****	**	ns	ns	ns	ns
GRS rate		****	*	***	****	****	****	ns	****	****	ns	****
Soil*GRS rate		ns	ns	*	ns	ns	*	ns	ns	ns	ns	ns

* = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.005$; **** = $P < 0.001$

Similar to soil pH, irrigation water likely moderated effects of GRS sources of Alum hydrolysis products on extractable soil Al. Mehlich-3 Al concentration was greater ($P < 0.001$) for Windthorst (50 mg kg^{-1}) compared to Weswood (16.5 mg kg) soil, but similar with or without GRS for each soil. The GRS effects on soil pH and Mehlich-3 Al will depend on interactions among Alum injection rates into waste liquids and buffering effects of irrigation water, amended soil and organic matter in GRS.

Incorporation of the respective GRS rates within the 10-cm soil depth increased concentrations of soil organic carbon (SOC) to 3 and 5 times that of soil alone (Table 2). In addition, incorporation of 0.25 m^3 GRS m^{-3} soil decreased ($P < 0.05$) soil bulk density 14.4% compared to soil without GRS. Previous reports corroborate this inverse relationship between soil bulk density and application rate of biosolids [31]. Similar to the present study, a volume-based rate (300 $m^3\ ha^{-1}$) of composted municipal biosolids reduced bulk density 19.7% after incorporation in a loamy soil [14].

In contrast to reports of increased soil gravimetric water content for soil amended with compost [14], water content was lowest ($P < 0.05$) for soil mixed with 0.25 $m^3\ m^{-3}$ of GRS (Table 2). In the present study, variation of turfgrass uptake and transpiration of water prior to soil sampling at 90 d could have accounted for differences in soil water content among treatments. Turfgrass biomass production was greater ($P = 0.05$) for turfgrass grown in Windthorst soil than in Weswood soil (Figure 1). In addition, biomass production increased ($P = 0.05$) for both soil types as GRS application rate increased (Figure 1). An inverse relationship ($r^2 = 0.55$, data not shown) was observed between soil water content and turfgrass biomass production. Increasing GRS rates could have increased soil N and mineral nutrient concentrations and contributed to greater turfgrass biomass production and water use [31].

Analyses of soil sampled 90 d after incorporation of GRS indicated increasing GRS rates increased ($P < 0.001$) plant-available nutrients in both soil types, including concentrations of Mehlich-3 and water extractable P (WEP) and extractable NH_4-N (Table 2). In addition, both total N and P increased ($P < 0.01$) as GRS rate increased (Table 2). Variation of extractable soil NO_3-N concentration did not differ ($P < 0.05$) between soil types or among GRS rates, but soil NO_3-N concentration was linearly related to GRS rate in both soil types (Table 3). The coefficients of determination for regression relationships between GRS application rate and soil extractable NO_3-N concentration were $r^2 = 0.95$ for Windthorst and $r^2 = 0.91$ for the Weswood soil (data not shown). The NO_3-N extracted from soil sampled at 90 d

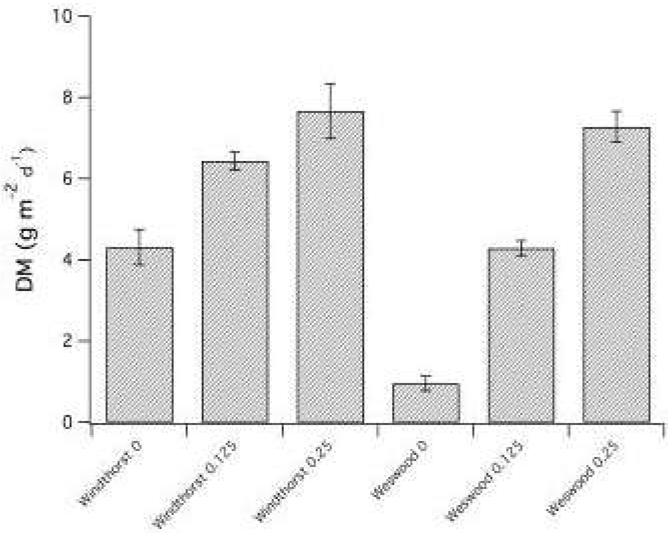

Fig. 1: Average daily dry matter (DM) production for turfgrass grown in Windthorst and Weswood soils amended with increasing rates of Geotube® residual solids (GRS). Error bars indicate the standard error of the mean

Table 3: Results of modeling percent C remaining over a 56-d incubation for Windthorst and Weswood soils with increasing rates of Geotube® residual solids (GRS) using a four-parameter double exponent model ($y = F*\exp^{(-kf * days)} + S*\exp^{(-ks * days)}$) . The intercepts for the fast (F) and slow (S) pool and decay rates for the respective pools are given. The rate constant (k) is in days.

		Fast Pool		Slow Pool	
Treatment	R^2	Intercept (F)	k_f	Intercept (S)	k_s
Windthorst 0.125	0.997	0.49	0.067	99.43	0.00013
Windthorst 0.25	0.999	0.28	0.089	99.62	0.00017
Weswood 0.125	0.992	0.38	0.093	99.43	0.00022
Weswood 0.25	0.987	0.28	0.103	99.57	0.00019

comprised NO_3-N available at incorporation of GRS and NO_3-N mineralized from TN during the ensuing 90 d. As variation of grass biomass among GRS rates indicates (Figure 1), the NO_3-N added in GRS and derived from mineralization of TN in GRS could have contributed to enhanced turfgrass establishment and growth (Table 2).

Carbon and Nitrogen Mineralization: Cumulative carbon mineralization was similar ($P > 0.05$) between Windthorst and Weswood soil types and was pooled to evaluate GRS rate effects. For all sampling dates over a 56-d period, increasing rates of incorporated GRS increased ($p < 0.001$) cumulative soil carbon mineralization in the order $0 < 0.125 < 0.25$ m m^{-3} of GRS (Figure 2). Although respective increases in GRS rate increased cumulative carbon mineralization 62% and 289% compared to soil without GRS, only 1.8 to 2.9% of GRS carbon was mineralized over a 56-day period (Figure 2). The relatively low mineralization rates for GRS were comparable to soil amended with 75 to 88% less carbon from straw-based compost [32]. The GRS sampled from Geotube® socks under field conditions could have comprised relatively recalcitrant forms of carbon that remained after storage for 1 yr under field conditions.

Variation of cumulative carbon mineralization among GRS rates reflected the amount of organic matter added rather than differences in degradability. The percent carbon remaining plotted versus time was fitted to a four-parameter double exponent kinetic decay model. The decay constant of fast and slow decomposing C pools in GRS was greater for Weswood than for Windthorst soil (Table 3). In addition, the slow decomposing fraction comprised greater than 99% of carbon added with GRS (Table 3). Similarly, a previous study observed that greater than 98% of C from bermudagrass thatch was associated with the slow pool [20]. Compared to various other bioresources, the decay rate of GRS was relatively low [33], which was attributed to possible reductions in fresh organic matter before collection from Geotube® socks.

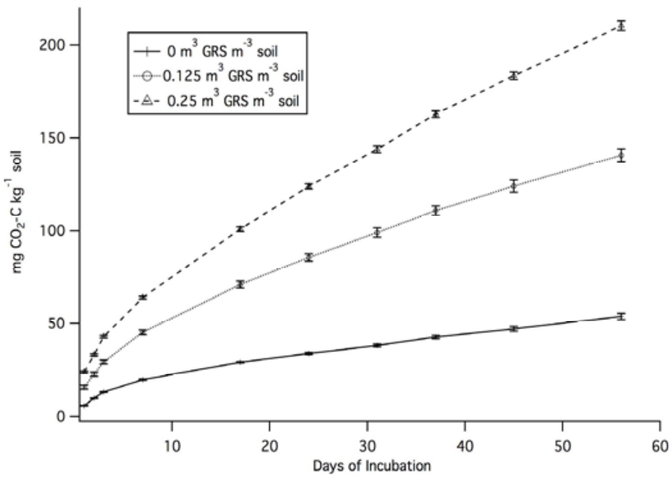

Fig. 2: Mean value of cumulative carbon evolved from Windthorst and Weswood soils with increasing rates of Geotube® residual solids (GRS) over a 56-d incubation. Error bars indicate the standard error of the mean

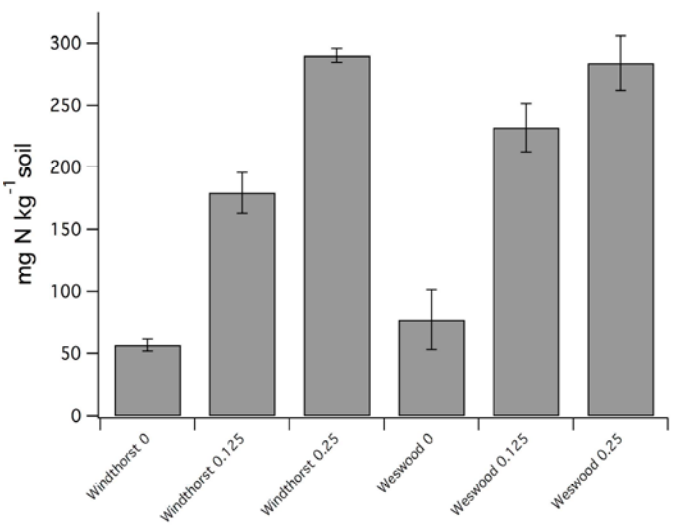

Fig. 3: Total nitrogen (NH4-N + NO3-N) mineralized for Windthorst and Weswood soils with and without Geotube® residual solids (GRS) over a 56-day incubation. Error bars indicate the standard error of the mean.

Similar to organic C, TN mineralization rate over 56 days was similar between Windthorst and Weswood soils. Averaged between soil types, the respective increases of GRS rate increased ($P < 0.001$) TN mineralization of NH_4-N and NO_3-N by 3- and 4-fold during the 56-day incubation period (Figure 3). The percentage of organic N applied with GRS (total N less NO_3-N) that was mineralized to NH_4-N and NO_3-N over 56 d was 8.0 to .4% for the low GRS rate (0.125 m^3 m^{-3}) and 5.5 to 6.6% for the high GRS rate (0.25 m^3 m^{-3}). The N mineralization rates were similar to values reported for compost [32], but the percentage of TN mineralized was slightly greater for GRS than for municipal organic waste [30]. High initial NO_3-N concentrations in GRS collected from Geotubes® could have contributed to greater mineralization of TN than for municipal wastes after both were mixed with soil. Concentrations and mineralization rates of initial N forms in GRS need to be quantified to optimize rates and timing of applications for turfgrass establishment. In addition, increases in soil concentrations of SOC, WEP and NO_3-N associated with large, volume-based GRS rates need to be monitored and managed to prevent leaching losses during turfgrass establishment [7].

Leaching Loss of Nutrients and Carbon: Consistent with observed increases in soil WEP, both GRS rates (0.125 and 0.25 m^3 m^{-3}) contributed to greater ($P < 0.01$) concentrations of dissolved P forms in leachate compared to controls without GRS (Figure 4). Dissolved un-reactive

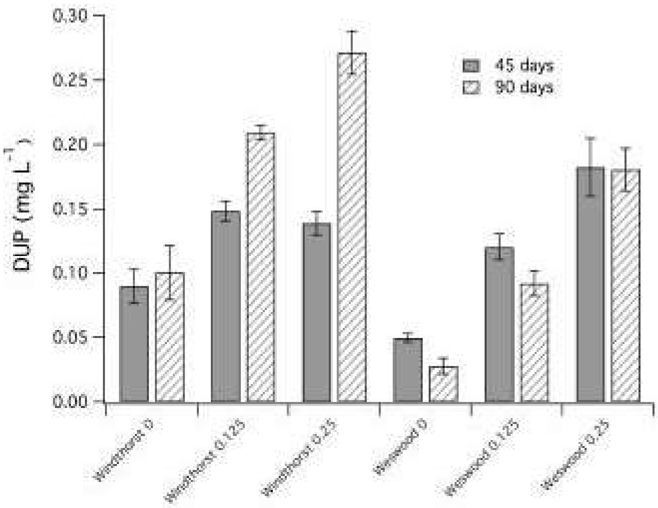

Fig. 4: Concentration of dissolved un-reactive P (DUP) in leachate collected at 45 and 90 days for Windthorst and Weswood soils amended with increasing rates of Geotube® residual solids (GRS). Error bars indicate the standard error of the mean.

P (DUP) concentration in leachate, calculated as concentrations of TDP-DRP, made up a large proportion of TDP at 45 and 90 d after GRS incorporation. The DUP is operationally determined through TDP and DRP analyses, but could comprise organic P and small amounts of inorganic P, including polyphosphates and pyrophosphates [34]. At 90 d, the percentage of TDP that was DUP ranged from 60 to 90% for soils without GRS and from 87 to 100% for soils with GRS. Similarly, Chardon *et al.* [35, 36] reported that 90% of TP in leachate from soil columns amended with manure solids or slurries was DUP. Previous studies evaluating P species in animal manure indicated 30% of the total P in manure is organic P [37].

In contrast to large proportions of DUP in TDP for the current study, Kleinman *et al.* [38] observed that surface applied poultry manure increased the fraction of DRP in leachate TDP from 7 to 72%. In the present study, injection of Alum and polymers during dewatering of waste liquid could have adsorbed or precipitated DRP in GRS and minimized concentrations and transport of DRP in leachate from soils amended with GRS. In addition, soil sorption and turfgrass uptake could have reduced soil concentration of DRP compared to DUP concentration in leachate. Low concentrations of DRP could limit the portion of TDP that is bioavailable, yet hydrolysis of DUP could serve as a long-term source of P in aquatic ecosystems [39].

Similar to GRS effects on soil and leachate concentrations of P forms, increasing GRS rates increased SOC concentration and dissolved organic carbon (DOC) concentration in leachate at 45 days after planting. Incorporating 0.125 m^3 m^{-3} of GRS increased leachate concentration of DOC 59% and incorporating 0.25 m^3 m^{-3} increased leachate concentration of DOC 76% compared to soil without GRS (data not shown). A previous evaluation of leaching losses from transplanted sod indicated DOC loss was two times greater for sod grown with composted biosolids than for sod grown with inorganic fertilizer [40]. After turfgrass was established in lysimeters for the present study (90-d sampling), DOC concentration in leachate was similar with or without incorporation of GRS. Leaching loss of DOC from turfgrass biomass at 90 days could have masked contributions of GRS to DOC leaching loss and variation of DOC leaching loss among treatments [40, 41]. For sods transplanted from turfgrass grown with composted biosolids, the proportion of SOC attributed to turfgrass sources increased during a 10-month period of establishment [40].

The concentration of DOC in leachate at 45 days after planting was linearly related to the concentration of DUP (Figure 5). The relationship between leaching loss of DUP and DOC is not unexpected considering DUP comprised organic P and both DOC and DUP concentrations reflected the application rates of GRS. Yet, it does not eliminate the possibility of DOC contributing to transport of inorganic P forms in soil [42]. Injection of Alum during dewatering of waste liquid could have formed phosphate-metal-humic complexes less than 0.45 ìm in diameter, which were recovered in leachate at 45 d, before being masked by turfgrass sources of DOC at 90 d [43].

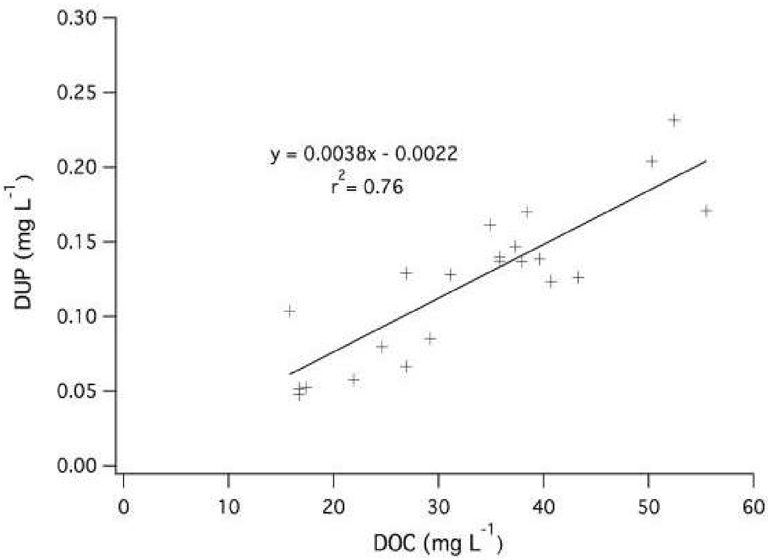

Fig. 5: Relationship between concentration of DOC and DUP in leachate at 45 days after planting turfgrass in Windthorst and Weswood soils amended with increasing rates of Geotube® residual solids (GRS)

Table 4: Concentration of NO_3-N in leachate at 45 and 90 days after planting of turfgrass in Windthorst and Weswood soils amended with increasing rates of Geotube® residual solids (GRS).

Soil	GRS rate	45 d	90 d
	m^3 m^{-3}	-----------mgL^{-1}----------	
Windthorst	0	86	0.3
Windthorst	0.125	161	1.2
Windthorst	0.25	232	64.5
Weswood	0	0.02	0.1
Weswood	0.125	78	0.1
Weswood	0.25	73	1.0
Soil		**	*
GRS rate		*	*
Soil*GRS rate		ns	*

* = P < 0.05; ** = P < 0.01; *** = P < 0.005; **** = P < 0.001

The effects of soil NO_3-N on leachate NO_3-N concentrations, whether applied with GRS or released through mineralization of TN, differed ($P < 0.05$) between leachate collection dates (Table 2, Table 4). Similar to soil NO_3-N concentrations, NO_3-N concentrations in leachate collected at 90 d were similar with and without GRS for the Weswood soil and greater for 0.25 m^3 m^{-3} of GRS in the Windthorst soil. Similarly, leachate concentrations of NO_3-N were greatest ($P < 0.05$) at 45 d for 0.25 m^3 m^{-3} of GRS mixed with the Windthorst soil (Table 4). For the Windthorst soil, NO_3-N concentrations in leachate were 87% and 170% greater for respective increases in GRS rate than without GRS. Leachate concentrations of NO_3-N were greater with than without GRS for the Weswood soil, but were more than 50% less than the Windthorst soil at respective GRS rates at 45 d. Reductions in leachate NO_3-N concentrations from 45 to 90 d after planting indicated slow mineralization rates of TN in GRS were not sufficient to replace root uptake and leaching losses of NO_3-N during turfgrass establishment (Figure 2, Table 4). For both soils, the high NO_3-N concentration in leachate collected 45 d after planting exceeded USEPA regulatory limits for drinking water. In previous studies, incorporation of composted manures increased leachate NO_3-N concentrations less than observed for GRS in the present study, but concentrations similarly exceeded regulatory limits [31].

CONCLUSION

Incorporation of GRS enhanced turfgrass growth and physical and chemical properties of contrasting soil types, but leaching loss of dissolved nutrients could be problematic during turfgrass establishment. Although volume-based GRS rates provided plant-available N and P and mineralizable C and N forms in soil, NO_3-N and dissolved P forms exceeded turfgrass uptake and increased leachate concentrations during turfgrass establishment. Low decay rates after mixing with soil and high NO_3-N concentration in GRS indicated substantial mineralization of organic N occurred during 1 yr of storage in Geotube® socks before collection and incorporation in soil. At high soil pH, residual Alum and associated hydrolysis products in GRS did not increase extractable soil Al, but did minimize leachate concentrations of dissolved reactive P forms. Composition, timing and application rate of GRS need to be managed to optimize turfgrass establishment while preventing leaching loss of dissolved nutrients.

REFERENCES

1. Mukhtar, S., L. Lazenby and S. Rahman, 2007. Evaluation of a synthetic tube dewatering system for animal waste pollution control. Applied Engineering in Agriculture, 23(5): 669-675.
2. Worley, J., T. Bass and P. Vendrell, 2008. Use of geotextile tubes with chemical amendments to dewater dairy lagoon solids. Bioresource Technol., 99(10): 4451-4459.
3. Gregorich, E. *et al.* Soluble organic carbon and carbon dioxide fluxes in maize fields receiving spring-applied manure. Journal of Environmental Quality, 1998. 27(1): p. 209-214.
4. Hay, F., *et al.* 2007. Leaching loss of NO 3-N and dissolved P from manure and fertilizer during turfgrass establishment. Plant and Soil., 296(1): 1-17.
5. Tisdale, S., *et al.* 1993. Micronutrients and other beneficial elements in soils and fertilizers. Soil Fertility and Fertilizers, 1: 1-624.
6. O'Connor, G., *et al.* 2005. Sustainable land application: An overview. J. Environmental Quality, 34(1): 7-17.
7. Maguire, R. and J. Sims, 2002. Soil testing to predict phosphorus leaching. Journal of Environmental Quality, 31(5): 1601-1609.
8. Bergstrom, L., B. Bowman and J. Sims, 2006. Definition of sustainable and unsustainable issues in nutrient management of modern agriculture. Soil Use and Manage., 21(s1): 76-81.
9. Munster, C., *et al.* 2004. Assessment of manure phosphorus export through turfgrass sod production in Erath County, Texas. J. Environmental Manage., 73(2): 111-116.
10. Vietor, D.M., *et al.* 2002. Export of Manure Phosphorus and Nitrogen in Turfgrass Sod. J. Environmental Quality, 31(5): 1731-1738.
11. Green, V. and D. Stott, Polyacrylamide: A review of the use, effectiveness and cost of a soil erosion control amendment. in 10th International Soil Conservation Organization. 2001. Purdue University and the USDA-ARS National Soil Erosion Research Laboratory.
12. Schnell, R.W., *et al.* 2009. Effects of Composted Biosolids and Nitrogen on Turfgrass Establishment, Sod Properties and Nutrient Export at Harvest. Hort Sci., 44(6): 1746-1750.
13. Vietor, D.M., *et al.* 2010. Biosolid and Alum effects on runoff losses during turfgrass establishment. Bioresource Technol., 101(9): 3246-3252.
14. Aggelides, S. and P. Londra, 2000. Effects of compost produced from town wastes and sewage sludge on the physical properties of a loamy and a clay soil. Bioresource Technol., 71(3): 253-259.
15. Malecki-Brown, L., J. White and K. Reddy, 2007. Soil biogeochemical characteristics influenced by alum application in a municipal wastewater treatment wetland. J. Environmental Quality, 36(6): 1904-1913.
16. Wang, F., *et al.* 1998. Effects of alum-treated waste water sludge on barley growth. Water, Air and Soil Pollution, 108(1): 33-49.
17. Haney, R., *et al.* 2004. Soil carbon and nitrogen mineralization: Influence of drying temperature. Soil Science Society of America J., 68(2): 489-492.
18. Sims, G., T. Ellsworth and R. Mulvaney, 1995. Microscale determination of inorganic nitrogen in water and soil extracts. Communications in Soil Science and Plant Analysis, 26(1): 303-316.
19. Kaboneka, S., W.E. Sabbe and A. Mauromoustakos, 1997. Carbon decomposition kinetics and nitrogen mineralization from corn, soybean and wheat residues. Communications in Soil Science and Plant Analysis, 28(15): 359-1373.
20. Berndt, W., 2008. Double Exponential Model Describes Decay of Hybrid Bermudagrass Thatch. Crop. Sci., 48(6): 2437-2446.
21. Pierzynski, G., 2000. Methods of phosphorus analysis for soils, sediments, residuals and waters. Southern Cooperative Series Bulletin, 396: 1-110.
22. Eaton, A. and M. Franson, 2005. Standard methods for the examination of water and wastewater. Washington, D.C. Amer Public Health Assn.
23. Havlin, J. and P. Soltanpour, 1980. A nitric acid plant tissue digest method for use with inductively coupled plasma spectrometry. Communications in Soil Science and Plant Analysis, 11(10): 969-980.
24. Mehlich, A., 1984. Mehlich 3 soil test extractant: A modification of Mehlich 2 extractant. Communications in Soil Science and Plant Analysis, 15(12): 1409-1416.

25. McGeehan, S. and D. Naylor, 1988. Automated instrumental analysis of carbon and nitrogen in plant and soil samples. Communications in Soil Science and Plant Analysis, 19(4): 493-505.
26. Leytem, A., J. Sims and F. Coale, 2004. Determination of phosphorus source coefficients for organic phosphorus sources: Laboratory studies. Journal of environmental Quality, 33(1): 380-388.
27. Huang, X. and M. Shenker, 2004. Water-soluble and solid-state speciation of phosphorus in stabilized sewage sludge. J. Environmental Quality, 33(5): 1895-1903.
28. Sharpley, A. and B. Moyer, 2000. Phosphorus forms in manure and compost and their release during simulated rainfall. J. Environmental Quality, 29(5): 1462-1469.
29. Johnson, G., *et al.* 2006. Topdressing turf with composted manure improves soil quality and protects water quality. Soil Science Society of America J., 70(6): 2114-2121.
30. Tognetti, C., M. Mazzarino and F. Laos, 2008. Compost of municipal organic waste: Effects of different management practices on degradability and nutrient release capacity. Soil Biology and Biochemis., 40(9): 2290-2296.
31. Evanylo, G., *et al.* 2008. Soil and water environmental effects of fertilizer-, manure- and compost-based fertility practices in an organic vegetable cropping system. A griculture, Ecosystems and Environ., 127(1-2): 50-58.
32. Flavel, T. and D. Murphy, 2006. Carbon and nitrogen mineralization rates after application of organic amendments to soil. J. Environmental Quality, 35(1): 183-193.
33. Gale, E., *et al.* 2006. Estimating plant-available nitrogen release from manures, composts and specialty products. J. Environmental Quality, 35(6): 2321-2332.
34. Broberg, O. and G. Persson, 1988. Particulate and dissolved phosphorus forms in freshwater: composition and analysis. Hydrobiologia, 170(1): 61-90.
35. Chardon, W., G. Aalderink and C. Van der Salm, 2007. Phosphorus leaching from cow manure patches on soil columns. J. Environmental Quality, 36(1): 17-22.
36. Chardon, W., *et al.* 1997. Organic phosphorus in solutions and leachates from soils treated with animal slurries. J. Environmental Quality, 26(2): 372-378.
37. Hansen, J., B. Cade-Menun and D. Strawn, 2004. Phosphorus speciation in manure-amended alkaline soils. J. Environmental Quality, 33(4): 521-1527.
38. Kleinman, P., *et al.* 2005. Phosphorus leaching through intact soil columns before and after poultry manure application. Soil Sci., 170(3): 153-166.
39. Toor, G., *et al.* 2003. Characterization of organic phosphorus in leachate from a grassland soil. Soil Biology and Biochemis., 35(10): 1317-1323.
40. Dai, X., *et al.* 2009. Effect of Composted Biosolids on Soil Organic Carbon Storage During Establishment of Transplanted Sod. Hortscience, 44(2): 503-507.
41. Wright, A., *et al.* 2008. Compost impacts on dissolved organic carbon and available nitrogen and phosphorus In turfgrass soil. Waste Manage., 28(6): 1057-1063.
42. Donald, R., D. Anderson and J. Stewart, 1993. Potential role of dissolved organic carbon in phosphorus transport in forested soils. Soil Science Society of America J., 57(6): 1611-1618.
43. Guardado, I., O. Urrutia and J. Garcia-Mina, 2007. Size Distribution, Complexing Capacity and Stability of Phosphate- Metal- Humic Complexes. J. Agric. Food Chem., 55(2): 408-413.