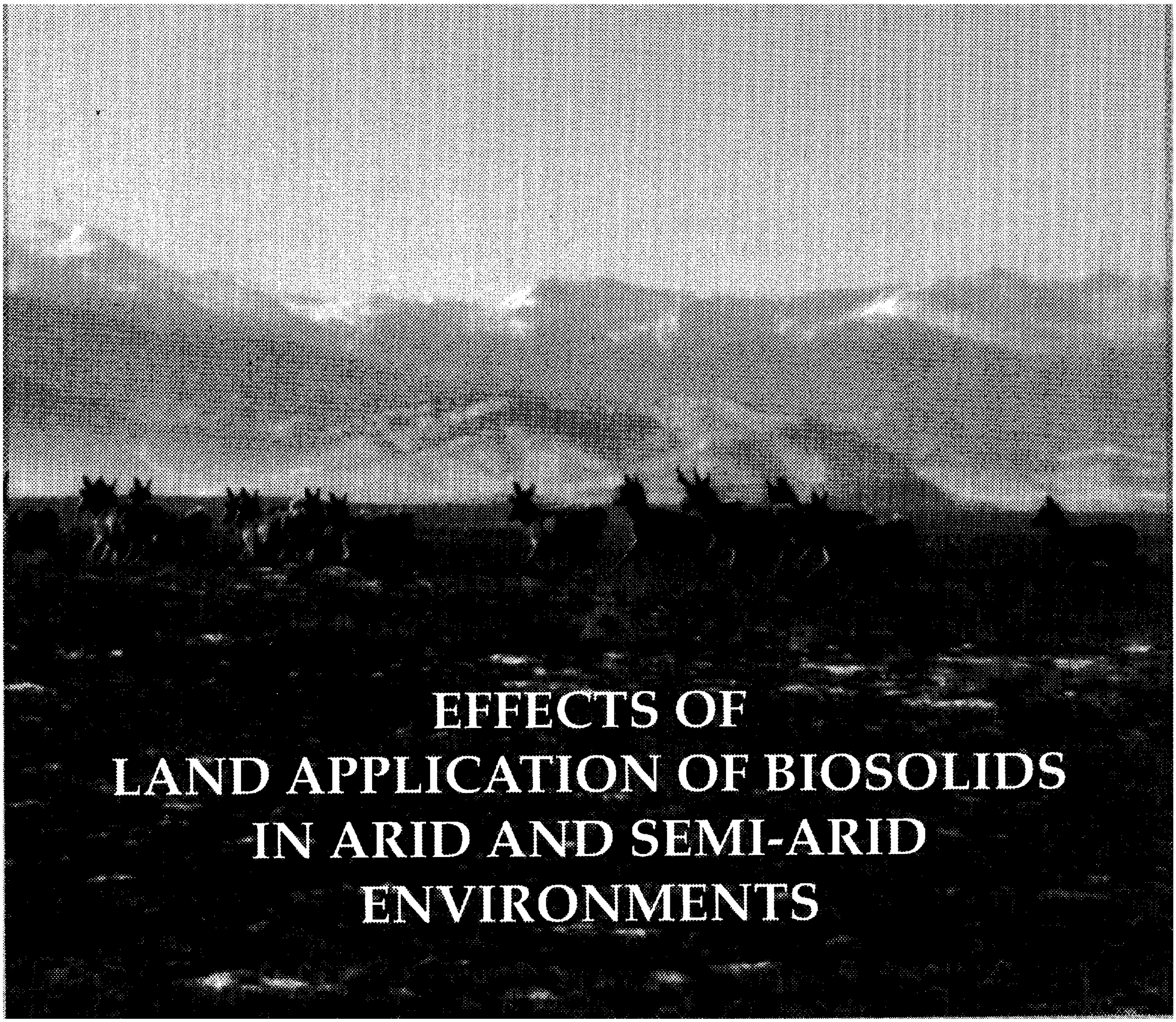
EFFECTS OF
LAND APPLICATION OF BIOSOLIDS
IN ARID AND SEMI-ARID
ENVIRONMENTS

EFFECTS OF LAND APPLICATION OF BIOSOLIDS IN ARID AND SEMI-ARID ENVIRONMENTS

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PROGRAM

Effects of Land Application of Biosolids in the Arid and Semi-Arid West

Fort Collins, Colorado
May 16-19, 1995

Tuesday May 16, 1995

8:00 - 8:15	Registration
8:15 - 8:30	*Welcome* by Bob Brobst

Session I - Vegetative Response

8:30 - 9:15	Rangeland Restoration with Surface-applied Biosolids: Effects on Soils and Vegetation of the Rio Puerco Watershed, Northcentral New Mexico *S.R. Loftin, R. Aguilar, R.R. Fresquez, and Francis*
915 -10:00	Tobosagrass and Alkali Sacaton Growth Responses to Topically Applied Biosolids in a Chihuahuan Desert Grassland *D.B. Wester, M.W. Benton, P. Jurado, R.G. Gatewood, and R.E. Sosebee*
10:00 - 10:15	Coffee Break
10:15 - 11:00	Evaluation of Sewage Sludge Products for use in Extensive Sheep Production Systems in Australia *D.L. Michalk, presented by G. King*
11:00 - 11:45	Sawmill Waste Utilization for Reclaiming Bentonite Mine Lands *G.E. Schuman, and E.M. Taylor, Jr.*
11:45 - 1:00	Lunch Break

Session II - Crop Benefits

1:00 - 1:45	11 Years of Biosolids Application to Dryland Winter Wheat *K.A. Barbarick, J.A. Ippolito, D.G. Westfall,* and R. Jepson
1:45 - 2:30	Biosolids Fertilization For Dryland Pacific Northwest Wheat Production *D.M. Sullivan, A.I. Bary, J.A. Kropf, and D.M. Granatstein*
2:30 - 3:15	Biosolids from an Agricultural Perspective *G. Wegner*
3:15 - 3:30	Coffee Break
3:30 - 4:15	N-viro Soil as a Gypsum Replacement in Cotton Production on a Sodic/Alkali Soil *B. McCullough-Sanden, H. McCuchin, R. Bailey, T. Logan, and B. Harrison*
4:15 - 5:00	A Comparison of Production of Dryland Wheat Between Use of Anhydrous Ammonia and Biosolids in Southeastern Colorado *R.B. Brobst and P. Hegeman*

Wednesday May 17, 1995

8:00 - 1:00	Attendees will take a field trip to the Fort Collins' *Meadow Springs Ranch*. A Box lunch will be provided.

Rangeland Restoration with Surface-applied Biosolids: Effects on Soils and Vegetation of the Rio Puerco Watershed, Northcentral New Mexico

Loftin, S.R., R. Aguilar, and P.R. Fresquez

Abstract

The Rocky Mountain Forest and Range Experiment Station began a project in 1985 to study the effects of surface applications of dried municipal wastewater biosolids on degraded rangeland in the Rio Puerco Watershed Research Area in Northcentral New Mexico. Albuquerque biosolids were applied at rates of 22.4, 45, and 90 Mg/ha, (10, 20, and 40 tons/acre) to replicated 3 X 20 m plots. Vegetation was monitored for changes in cover and nutrient (total nitrogen) and heavy metal content (cadmium, copper, lead, and zinc) of aboveground tissues. Soils were monitored for changes in nutrient (organic matter, nitrogen, and phosphorus) and heavy metal content. In general, vegetation cover and biomass increased with increasing biosolids application, as did soil nutrient and heavy metal content. Treatment effects on total nitrogen and heavy metal concentrations of aboveground plant tissues had diminished by the fifth growing season after biosolids application. Extractable Cu in soils receiving the highest biosolids application attained potentially phytotoxic levels, although no phytotoxic effects were observed. Therefore, we concluded that the intermediate rate of 45 Mg/ha was the optimum balance between nutrient input and heavy metal loading for this soil/plant system.

Keywords: Rangeland restoration, biosolids, sewage sludge, vegetation, soils, heavy metals.

Introduction

Rangeland degradation is a serious problem affecting the sustainability and productivity of rangelands throughout the western United States (Sheridan 1981). Although much of the damage occurred during the livestock heyday of the late nineteenth and early twentieth centuries, many areas have still not recovered their original productivity. Experience, observation, and research indicate that many areas will not recover even with removal of livestock. Traditional successional pathways appear to have been altered by the loss of soil resources and change in plant species composition to the point that grassland restoration may not be possible without some active land management intervention.

This alteration of traditional Clementsian successional processes is a common occurrence on arid and semiarid lands worldwide Westoby et al. 1989; Johnson and Mayeus 1992). In the traditional model of ecosystem succession, ecosystems are described as having one stable state, the climatic climax. Disturbance to the ecosystem would result in an altered state which would proceed along a predictable linear pathway back to the climax state. It is now generally recognized that the traditional linear model does not adequately represent the disturbance/recovery dynamics observed in many arid and semiarid regions. In arid regions, particularly grasslands, disturbance often leads to shrub invasion and redistribution of soil resources (Schlesinger et al., 1990; Grover and Musik, 1990). These post-disturbance ecosystems are often very stable and it is this observation that has lead to the emergence of a new model of arid ecosystem succession which is often referred to as the multiple stable states model (Westoby, 1989; Tausch et al. 1993). This model states that stable ecosystem states are separated by thresholds of disturbance. If a disturbance is great enough to force an ecosystem over this threshold the system will stabilize in a new state. If the disturbance is not great enough to cross the threshold then traditional Clementsian successional processes will return the system to its original state. For example, periodic drought or grazing may only temporarily alter the abundance and species composition of an ecosystem, whereas prolonged drought coupled with grazing and fire suppression can eventually convert a grassland to a shrubland. An ecosystem that has crossed a threshold of disturbance will theoretically require an equal or greater amount of energy to restore the system back to its original state. We believe that the combined fertilizer and mulching effects of surface-applied biosolids should provide the "force" necessary to "push" a degraded rangeland site over the threshold towards a productive semiarid grassland.

The research presented herein was initiated to test the effectiveness of different applications of surface-applied biosolids (treated municipal sewage sludge) on the recovery of a semiarid grassland in Northcentral New Mexico (Aguilar et al. 1994; Fresquez et al. 1990a; 1990b; 1991). One objective of the study was to determine the optimum application rate, balancing the beneficial mulching and nutrient amendment effects with the potentially detrimental effects of sludge-borne heavy metals.

Methods

Site description

The study site is located in the Rio Puerco Watershed Resource Area in Northcentral New Mexico, near the small town of San Luis. The Rio Puerco Watershed is recognized as one of the most degraded areas in the western United States (Sheridan, 1981). Vegetation on the study site was classified as a broom snakeweed (*Gutierresia sarothrae*)/blue grama (*Bouteloua gracilis*) - galleta (*Hilaria jamesii*) community, typical of degraded grasslands throughout the region (Francis 1986). The soil is classified as a deep, fine-silty, mixed, mesic Ustollic Camborthid (Los Lucas series). Mean annual precipitation ranges from 216 to 322 mm.

Experimental Design

A series of replicated 3 X 20 m experimental plots were arranged in a randomized block design. The four treatments include the control, and biosolids applications of 22.5, 45, and 90 Mg/ha (0, 10, 20, and 40 tons/acre). The plots were fenced off to exclude livestock. A <u>one-time</u> surface application of dried, anaerobically digested municipal biosolids from the City of Albuquerque Wastewater Treatment Plant was applied in June 1985.

Soils (0 - 15 cm depth) were sampled prior to the biosolids amendment (June 1985) and then annually from 1985 through 1989, and again in 1993. Soils were tested for organic matter (OM), total nitrogen (TKN), KCl extractable NH_4-N and NO_3-N, $NaHCO_3$ extractable phosphorus, pH, and DTPA (diethylenetriaminepentaacetic acid) extractable cadmium (Cd), copper (Cu), lead (Pb), and zinc (Zn). The specific analyses are described in Dennis and Fresquez (1989).

Vegetation cover was measured using the Community Structure Analysis (CSA) technique (Pase, 1981). Stems and leaves of blue grama, the most abundant grass, were analyzed for total nitrogen and heavy metal contents.

Analyses of data were conducted using standard ANOVA techniques followed by the Least Significant Difference (LSD) multiple comparison test when appropriate. All tests of significance used a type one error rate of 0.05.

Results and Discussion

Soil Organic Matter, Nitrogen, and Phosphorus

The municipal biosolids used in this treatment were an excellent source of organic matter and plant nutrients (Table 1). However, the success of the treatments can best be interpreted as a function of the ability of the soil/plant/ system to assimilate and store nutrients and increase above and belowground biomass.

Although we observed some significant treatment effects on soil OM in 1986 and 1989, no significant differences were detected in 1993 (Table 2). Surface application of biosolids did not appear to directly contribute to soil OM, instead the observed increases in organic matter apparently resulted from increased belowground productivity and/or surface soil incorporation of aboveground organic matter. The significant increases observed early in the study suggest a belowground response to increased nutrient availability, however, other biosolids application studies have reported an inhibition of root growth due to increased nutrient availability (Loftin and Aguilar 1994). At this time we are unable to explain the short-term soil organic matter response and it is too soon to evaluate the long-term response.

Table 1. Organic matter, plant nutrient, and heavy metal analysis of Albuquerque dried municipal biosolids, loading rates of the three biosolids applications, and U.S. Environmental Protection Agency (USEPA) limits.

Albuquerque Biosolids

	OM	TKN	P	Cd	Cu	Pb	Zn
	(%)			(mg/kg)			
	50	4.86	0.16	0.24	47.50	3.30	150
USEPA Limits (mg/ha)[1]				85	4,300	840	7,500

Rio Puerco Biosolids Loading Rates

Rate (Mg/ha)	OM	TKN	P	Cd	Cu	Pb	Zn
	(kg/ha)			(g/ha)			
22.5	11,250	1,094	36	5.4	1,069	74	3,375
45.0	22,500	2,188	72	10.8	2,138	148	6,750
90.0	45,000	4,376	144	21.6	4,276	296	13,500
USEPA Limits (kg/ha)[2]				39	1,500	300	2,800

[1]USEPA ceiling concentrations for municipal biosolids (USEPA, 1993).
[2]USEPA cumulative pollutant loading rates for land application of municipal biosolids (USEPA, 1993)

Table 2. Organic matter and plant nutrient levels in control and biosolids-treated soils. Means in the same column and year followed by the same letter are not significantly different (P>0.05).

Treatment (Mg/ha)	OM (%)	TKN	NH_4-N	NO_3-N	P
		Analysis (mg/kg)			
1985					
Control	1.2a	729b	3b	2b	5c
22.5	1.3a	817ab	9b	22ab	15bc
45.0	1.4a	845ab	20ab	42ab	20b
90.0	1.2a	924a	51a	61a	31a
1986					
Control	1.4ab	665b	3c	1c	4c
22.5	1.5a	828ab	10ab	10bc	20bc
45.0	1.5ab	843ab	22b	28b	44b
90.0	1.2b	987a	39a	54a	72a
1989					
Control	1.4b	682b	4b	7c	9b
22.5	1.8ab	890b	4b	14b	26ab
45.0	2.6a	1869a	39a	22a	42ab
90.0	2.3ab	1814a	42a	28a	57a
1993					
Control	1.3a	820c	8b	3a	6c
22.5	1.7a	1076b	9b	5a	25bc
45.0	1.7a	1158ab	15b	7a	43b
90.0	1.8a	1370a	47a	8a	77a

For the first five years of the study (1985 to 1989) soil mineral nitrogen (NH_4-N and NO_3-N) was rarely significantly greater than control levels for the 22.5 Mg/ha treatment, but the two highest biosolids treatments (45 and 90 Mg/ha) were often significantly greater than control levels (Table 2). However, by 1993 only the 90.0 Mg/ha treatment had significantly greater soil mineral nitrogen. In contrast, only the 90.0 Mg/ha application had significantly greater soil TKN than controls in 1985 and 1986. By 1993, all three treatments had significantly greater soil TKN. The increase in soil TKN reflects an increase in organic nitrogen, which is an indication of the success of the treatment. Organic nitrogen represents an important source of nitrogen which will become available for plant uptake following decomposition of the organic matter. The more nitrogen fixed into organic form, the greater the nitrogen reserve for future productivity.

Phosphorus, another important and often limiting plant nutrient, was significantly greater in the 45.0 and 90.0 Mg/ha treatments than control levels for most sampling dates, but soil P from the 22.5 Mg/ha treatment has never been significantly greater than controls.

Soil pH and Heavy Metals

Soil pH is an important factor in determining the availability of plant nutrients and, of more concern for this project, the availability of heavy metals. In general, heavy metal mobility in soils increases with increasing acidity. Acids produced during decomposition of organic biosolids and through nitrification of mineral nitrogen, can increase soil acidity. The 22.5 Mg/ha treatment had no significant effect on soil pH, however, soil pH from the 45.0 and 90.0 Mg/ha treatments was consistently significantly lower than controls (Table 3). Metals are relatively immobile and unavailable for biological assimilation in soils with a pH of 7.0 or above (Chang et al., 1984) and treated soil pH levels have remained above 7.0. Furthermore, "Cu, Zn, Ni, and Cr toxicities to even very sensitive plants have not been observed in any municipal sludge land application experiments with pH > 5.5" (Chang et al. 1992). Levels of heavy metals in the biosolids used for this project were well below USEPA (1993) maximum limits (Table 1). Heavy metal loading rates for all treatments were also well below established USEPA limits. No problems with heavy metal accumulation in soils or plant tissues is likely due to the combination of high soil pH and low heavy metal loading rates.

Soil heavy metal concentrations did not increase significantly above control levels at any sampling period for the 22.5 Mg/ha treatment and, therefore, should not present any problems (Table 3). Soil Cu levels were perhaps the most problematic, with levels significantly above controls for all sampling periods for the 45.0 and 90.0 treatments. In 1989 DTPA extractable Cu levels were above 20 mg/kg which is considered to be potentially phytotoxic (Tiedemann and Lopez, 1982) although no evidence of phytotoxicity was observed. Soil Zn responded in much the same way as Cu. However, Zn is not as phytotoxic as Cu and Zn concentrations in treated soils remained below 10.0 mg/kg. Soil Pb, which is relatively immobile in alkaline soils, did increase significantly in the 90.0 Mg/ha treatment but levels were below 2.0 mg/kg and should not be a problem.

Table 3. Soil pH and heavy metal levels from control and biosolids-treated plots. Means in the same column and year followed by the same letter are not significantly different (α=0.05).

Treatment (Mg/ha)	pH	Cd	Cu	Pb	Zn
		Analysis			
		(mg/kg)			
1985					
Control	7.8a	0.01a	1.04c	0.55a	0.29c
22.5	7.7ab	0.01a	1.19bc	0.60a	0.35bc
45.0	7.6b	0.01a	1.60ab	0.58a	0.61ab
90.0	7.5b	0.01a	2.10a	0.60a	0.76a
1986					
Control	7.8a	0.01a	0.92b	0.63a	0.27b
22.5	7.6ab	0.01a	2.21ab	0.60a	0.79ab
45.0	7.4b	0.02a	2.99a	0.65a	1.01ab
90.0	7.4b	0.02a	3.48a	0.68a	1.20a
1989					
Control	7.8a	0.01b	0.88b	0.76b	0.17c
22.5	7.7a	0.01b	2.40b	0.84b	1.06bc
45.0	7.4b	0.15a	23.52a	1.45ab	7.78ab
90.0	7.0b	0.20a	29.78a	1.61a	9.67a
1993					
Control	7.5a	0.00b	0.95c	0.70b	0.21c
22.5	7.5ab	0.00b	2.28bc	0.80b	1.03bc
45.0	7.3b	0.00b	6.08b	0.98b	2.35b
90.0	7.1c	0.06a	10.16a	1.13a	4.01a

Table 4. Total percent cover of all plant species. Means in the same column followed by the same letter are not significantly different (P>0.05).

Treatment (Mg/ha)	Percent Cover 1985	1986	1989	1993
0	30b	35b	30b	26b
22.5	37a	54a	51a	34a
45.0	35ab	52a	47ab	33ab
90.0	36ab	55a	45ab	35a

Table 5. Total nitrogen and heavy metal contents in blue grama grass aboveground tissue from control and biosolids-treated plots. Means in the same column and year followed by the same letter are not significantly different (α=0.05).

Treatment (Mg/ha)	Analysis: TKN (%)	Cu (mg/kg)	Zn (mg/kg)
1985			
Control	1.6c	3c	26b
22.5	2.2bc	5bc	83a
45.0	2.7ab	8ab	77a
90.0	2.8a	10a	73a
1986			
Control	0.8c	5a	22b
22.5	1.7b	7a	31a
45.0	2.1ab	7a	42a
90.0	2.1a	8a	47a
1989			
Control	2.0a	10a	26a
22.5	2.4a	11a	26a
45.0	2.5a	11a	23a
90.0	2.5a	13a	21a
1993			
Control	1.3a	6a	19a
22.5	1.3a	6a	15a
45.0	1.4a	6a	18a
90.0	1.5a	7a	25a-

Plant Cover

Plant cover from treated plots was significantly greater than controls in all post-treatment sampling periods, however, the response was not consistent from one sampling period to the next (Table 4). Also, the plant cover response was not linear with respect to the biosolids application rates. It is possible that all three biosolids applications provided excess mineral nutrients and the variability in response may have been due to differential water availability. Other studies have shown that the plant cover response is very dependent upon water availability (Loftin and Aguilar 1995), which is influenced by the interaction of precipitation and biosolids application rate. Given equal amounts of precipitation, variability in water availability can result from differential thickness of the biosolids layer which would influence the amount of precipitation to reach the mineral soil (interception) and the retention of water in the soil (mulching effect). A higher percentage of the water from a light rainfall might reach the mineral soil on the 22.5 Mg/ha plots because there would be less biosolids to intercept the rainfall. Conversely, the heavier biosolids applications might retain more soil water when rainfall is great enough to penetrate the biosolids layer and wet the soil.

Blue Grama Tissue N and Heavy Metals

Blue grama is the most abundant grass on the site and was, therefore, chosen as a representative species for aboveground tissue analyses to test the effects of the biosolids treatment on assimilation of nitrogen and heavy metals. Nitrogen assimilation is important in determining forage quality (protein content), however, from a restoration standpoint it is also an important index of restoration success. As previously stated in the discussion on soils, mineral nitrogen from the biosolids treatment is available for a relatively brief period of time and it is important that the vegetation assimilate and store as much of this available nitrogen as possible. In 1985 and 1986 most of the blue grama tissue from the treated plots had TKN levels significantly greater than tissue from controls (Table 5). However, by 1989 and again in 1993, TKN in blue grama tissue was not significantly different among treatments. Unfortunately, without productivity or aboveground biomass estimates these data do not allow us to make conclusive inferences about nitrogen storage within this plant/soil system.

Total heavy metal content of plant tissues was monitored to evaluate availability and the potential for transport through the food chain. In 1985, Cu and Zn levels in blue grama tissues from most of the biosolids treatments were significantly greater that controls (Table 5). By 1986, only Zn levels were significantly different from controls and after 1986 no significant differences were observed in either Cu or Zn. Cadmium levels in blue grama tissue were below the detection limit (0.5 mg/kg) and no significant differences in tissue Pb levels were recorded throughout the study. Heavy metal dynamics in this system would be better understood if data on uptake and storage in aboveground biomass were available. We can conclude that in this study at least, increased metal availability was limited to the first two to five years after biosolids application.

Conclusions

Eight years after biosolids application to study plots in the Rio Puerco Watershed Resource Area all three treatments had soil TKN values significantly greater than controls. The biosolids applications of 45.0 and 90.0 Mg/ha had significantly greater soil P, pH, Cu, and Zn, and the 90.0 Mg/ha treatment had significantly greater NH_4-N, Cd, and Pb. Plant cover increased significantly at all applications but the response was neither consistent nor linear. Analysis of aboveground blue grama grass tissue showed no significant differences in TKN, Cd, Cu, Pb, or Zn between treatments after eight years.

We believe that biosolids amendments can be used safely, without harming environmental quality. In order to balance the potential benefits of increased nutrient availability with the need to minimize the potential for heavy metal transport from the soil/plant system to higher levels of the food chain, we recommend the intermediate biosolids application of 45.0 Mg/ha for restoration of semiarid soil/plant systems similar to the one studied here.

Literature Cited

Aguilar, R., S.R. Loftin, and P.R. Fresquez. 1994. Rangeland restoration with treated municipal sewage sludge. *In*: Clapp, C.E., W.E. Larson, and R.H. Dowdy. *Sewage Sludge: Land Utilization and the Environment*. ASA-CSSA-SSSA, Madison, WI.

Chang, A.C., J.E. Warneke, A.L. Page and L.J. Lund. 1984. Accumulation of heavy metals in sewage sludge-treated soils. J. Environ. Qual. 13(1):87-91.

Chang, A.C., T.C. Granato, and A.L. Page. 1992. A methodology for establishing phytotoxicity criteria for chromium, copper, nickel, and zinc in agricultural land application of municipal sewage sludges. Journal of Environmental Quality. 21(4):521-536.

Dennis, G.L., and P.R. Fresquez. 1989. The soil microbial community in a sewage-amended semi-arid grassland. Biol. and Fert. Soils 7:310-317.

Francis, R.E. 1986. Phyto-edaphic Communities of the Upper Rio Puerco Watershed, New Mexico. US Dept. of Agric. Forest Service Res. Pap. RM-272, Rocky Mnt. Forest and Range Exp. St., Ft. Collins, CO.

Fresquez, P.R., R.E. Francis and G.L. Dennis. 1990a. Soil and vegetation responses to sewage sludge on a degraded semiarid broom snakeweed/blue grama plant community. Journal of Range Management 43:325-331.

Fresquez, P.R., R.E. Francis and G.L. Dennis. 1990b. Effects of sewage sludge on soil and plant quality in a degraded semiarid grassland. Journal of Environmental Quality 19:324-329.

Fresquez, P.R., R. Aguilar, R.E. Francis and E.F. Aldon. 1991. Heavy metal uptake by blue grama growing in a degraded semiarid soil amended with sewage sludge. Journal of Water, Air and Soil Pollution 57-58:903-912.

Grover, H.D. and H.B. Musik. 1990. Shrubland encroachment in southern New Mexico, U.S.A.: an analysis of desertification processes in the American Southwest. Climatic Change 17:305-330.

Loftin, S.R. and R. Aguilar. 1994. Semiarid rangeland response to municipal sewage sludge: plant growth and litter decomposition. pp 221-229. *In*: Clapp, C.E., W.E. Larson, and R.H. Dowdy. *Sewage Sludge: Land Utilization and the Environment*. ASA-CSSA-SSSA, Madison, WI.

Pase, C.P. 1981. Community structure analysis-a rapid, effective range condition estimator for semi-arid ranges. *In:* H.G. Lund, et at. (tech. coord.). *Arid land resource inventories: Developing cost-efficient methods*. USDA Forest Service Gen. Tech. Rep. WO-28, Washington D.C.

Schlesinger, W.H., J.F. Reynolds, G.L. Cunningham, L.F. Huenneke, W.M. Jarrell, R.A. Virginia and W.G. Whitford. 1990. Biological feedbacks in global desertification. Science 247:1043-1048.

Sheridan, D. 1981. Desertification of the United States. Council on Environmental Quality. U.S. Government Publishing Office. Washington D.C.

Tausch, R.J., P.E. Wigand and J.W. Burkhardt. 1993. Viewpoint: Plant community thresholds, multiple steady states, and multiple successional pathways: legacy of the Quaternary? Journal of Range Management 46(5):439-447.

Tiedemann, A.R. and C.F. Lopez. 1982. Soil nutrient assessments of mine spoils. *In*: Aldon, E.F. and W.R. Oaks. *Reclamation of Mined Lands in the Southwest Symposium Proceedings*. October 20-22, 1982. Albuquerque, NM.

U.S. Environmental Protection Agency. 1993. 40 CFR - Parts 257 and 503, Standards for the disposal of sewage sludge; final rule. Washington, DC.

W-170 Peer Review Committee. 1989. Peer review of standards for the disposal of sewage sludge (USEPA Proposed Rule 40 CFR Parts 257 & 503) USDA-CSRS W-170 Regional Research Committee. USDA, Washington, DC.

Westoby, M., B. Walker and I. Noy-Meir. 1989. Opportunistic management for rangelands not at equilibrium. J. Range Manage. 42(4):266-274.

Tobosagrass and Alkali Sacaton Growth Responses to Topically Applied Biosolids in a Chihuahuan Desert Grassland

David B. Wester, Mark W. Benton, Pedro Jurado,
Richard G. Gatewood, and Ronald E. Sosebee

Abstract

Objectives of this research were to investigate responses of tobosagrass (*Hilaria mutica*) and alkali sacaton (*Sporobolus airoides*) to biosolids topically applied during autumn, dormant, or growing seasons with and without supplemental irrigation; multiple yearly or seasonal applications also were studied. Application rates ranged from 0 to 89.7 Mg/ha. Standing crop at the end of the growing season was measured. Season and rate of application as well as irrigation effects were important factors affecting plant response. Plant response was species-specific. In general, plants provided with biosolids in the dormant season produced more standing crop than plants provided with biosolids immediately prior to the onset of the growing season. Additionally, standing crop generally increased at biosolids rates up to 17.9 Mg/ha.

Keywords: Biosolids, standing crop, Chihuahuan desert, beneficial use.

Introduction

Land application of biosolids will be an increasingly common method of disposal as environmental concerns and costs associated with ocean disposal, incineration and landfilling grow. Although there has been much research on the effects of biosolids on agronomic crops, particularly in relatively humid areas, less information is available on biosolids effects on native rangeland in arid and semiarid environments.

Fresquez et al. (1990a,b; 1991) conducted extensive research on the effects of a one-time only (July) application of biosolids (at rates up to 90 Mg/ha) to semiarid rangeland dominated by blue grama (*Bouteloua gracilis*), galleta (*Hilaria jamesii*) and snakeweed (*Gutierrezia sarothrae*). Soil and vegetation responses were followed for several growing seasons after application (see Aguilar et al. (1994) for a summary). Blue grama production was increased by biosolids application for up to 5 growing seasons after application at 45 and 90 Mg/ha. Also, blue grama forage quality was improved by biosolids application without accompanying heavymetal uptake. Aguilar et al. (1994) also studied a one-time only(April) application of biosolids (45 Mg/ha) on semiarid rangeland. Aboveground plant cover was higher in treated plots only during one sampling period, and root growth was less in treated plots. Plant tissue in treated plots had higher TKN and Cu than in control plots.

Despite the extensive findings of Fresquez et al. (1990a, b; 1991) and the ongoing work of Aguilar et al. (1994), there are many aspects of land application of biosolids to semiarid rangelands that have not received adequate attention. The objectives of this research were to investigate effects of topically-applied biosolids on desert grassland vegetation. Tobosagrass (*Hilaria mutica*) and alkali sacaton (*Sporobolus airoides*), two widely distributed grasses common throughout the arid and semiarid southwestern United States, were studied. Factors investigated included rate and season of biosolids application, supplemental irrigation, and consecutive seasonal or yearly application.

Methods

Experiment I: (One-time only autumn application) Experimental units were 48, 0.5-m^2 plots in relatively homogeneous tobosograss vegetation on a Stellar sandy clay loam soil. Plots were established in August, 1992. Soils were sampled and analyzed for a suite of nutrients and metals prior to application of biosolids. Initial standing crop was harvested by hand clipping to a 5-cm stubble height; forage samples were oven dried and weighed to the nearest 0.01 g. Biosolids were applied in September, 1992. Treatments included 4 rates of biosolids (0, 6.7, 33.6, and 89.7 Mg/ha) and 2 levels of irrigation (supplemental irrigation provided or not provided) in a factorial arrangement of a completely randomized design with 6 replications. Supplemental irrigation was provided on 3 occasions during the fall of 1992, on 3 occasions during the 1993 growing season, and on 6 occasions during the 1994 growing season (1.25 cm/application).

Standing crop was harvested at the end of the 1993 and 1994 growing seasons by hand clipping to a 5-cm stubble height. Forage samples were oven dried and weighed to the nearest 0.01 g.

Experiment II: (Once per year seasonal application)
Experimental units were 960, 1-m^2 plots, established in tobosograss (n=480) or alkali sacaton (n=480) vegetation on a Stellar sandy clay loam soil. In each plot, biosolids were applied to a 0.85 m^2 area. Plots were established in winter 1993. Initial standing crop was harvested as in Experiment I.

Treatments included biosolids rates (0, 6.7, 17.9, and 89.7 Mg/ha), supplemental irrigation (as in Experiment I), season of application (dormant or growing season), and number of years of consecutive application (1, 2, 3, or 4 years). The experiment was setup as a split plot design, with season and years of application as main plot factors (in a factorial combination) and biosolids rate and irrigation as subplot factors (in a factorial combination) with 6 replications. Biosolids were applied in early March 1993 and late January 1994 in dormant season applications, and in early July 1993 and 1994 in growing season applications. Data from the first 2 years of the study are reported here. Standing crop was harvested in 1993 and 1994 as in Experiment I.

Experiment III: (Twice per year seasonal applications)
Experimental units were 384, 1-m^2 plots established in relatively homogeneous tobosograss vegetation in a Stellar sandy clay loam. Plots were established in winter 1994. Initial standing crop was harvested as in Experiment I.

Treatments included biosolids rate (0, 6.7, 17.9 or 33.7 Mg/ha), supplemental irrigation (as in Experiment I), season of application (dormant and growing season or spring and growing season), and number of years of consecutive application (1, 2, 3 or 4 years). The experiment was set up as a split plot design, with season and years of application as main plot factors (in a factorial combination) and biosolids rate and irrigation as subsplot factors (in a factorial combination) with 6 replications. Biosolids were applied in late January, 1994; mid-April, 1994; and early July 1994. Standing crop in 1994 was harvested as in Experiment I.

Statistical Analyses

Standing crop data from each experiment were analyzed with analysis of variance. Standing crop data for Experiments I and II were nonnormally distributed on the observed scale but normally distributed on a logarithmic scale (normality tested with the Shapino-Wilk (1965) Test). Homogeneity of variance was tested with Levene's (1960) test. Statistical analyses were completed on the log scale; however, back-transformed (geometric) means are presented. Fisher's LSD test was used to separate treatment means.

Results

Biosolids Analyses
Biosolids analyses used in this research are summarized in Table 1. Each biosolids application necessarily used a different batch of biosolids. In general, however, biosolids used in these experiments were similar in quality.

Rainfall Conditions
Monthly precipitation during 1993 equalled or exceeded long term averages throughout the year. In contrast, monthly rainfall during 1994 was below average throughout the year. Thus, 1993 and 1994 represent favorable and unfavorable conditions, respectively, with respect to soil moisture and plant growth.

Experiment I: One-Time-Only Autumn Application Initial Soil Analyses
At the beginning of the study, experimental units were homogeneous with respect to soil pH, EC, Na, Ca, Mg, SAR, exchangeable Na, organic matter, P, K, TKN, NH4-N, Cd, Pb, B, Mn, Cu, Ni and Zn in the surface horizon. In the B horizon, plots that were to receive 0 and 6.7 Mg/ha of biosolids had higher Ni, and plots that were to receive 0 Mg/ha had higher Zn. These few differences were present in the soil prior to biosolids application.

Table 1.	Biosolids quality analyses.					
		Biosolids Application Date				
Variable	Sept 1992	March 1993	July 1993	Jan 1994	April 1994	July 1994
TKN (%)	3.23	3.98	4.77	3.36	2.92	3.62
P (%)	2.95	2.36	2.46	1.47	1.932	1.41
K (%)	0.13	0.188	0.052	0.685	0.09	0.042
Ca (%)	2.25	1.85	3.02	1.29	2.55	1.904
Mg (%)	0.50	0.874	0.652	0.445	0.914	0.382
B (ppm)	155	44.2	50.2	34.5	32.6	32.6
Cu (ppm)	1000	969	1511	562	420	1033
Fe (ppm)	86050	21920	31980	23360	23092	19798
Mn (ppm)	585	707	563	544	1019	651
Zn (ppm)	1365	972	1543	812	1147	1100
Al (ppm)	13870	10994	10768	7531	7197	8279
Cd[1]	0.086	9.24	54.64	7.6	9.44	8.08
Pb[1]	0.22	273	262	212	321	187
Ni[1]	0.45	47	35	26	43	22

[1] Units are mg/L by ICP for Sept., 1992 and ppm for other dates.

Standing Crop, 1993

There was a weak ($P < 0.0826$) interaction between biosolids rate and irrigation effects: the proportional change in standing crop attributed to biosolids depended on irrigation (Fig. 1). In nonirrigated plots, standing crop in the 89.7 Mg/ha rate decreased relative to 0, 6.7 and 33.6 Mg/ha treatments. In irrigated plots, however, biosolids did not affect standing crop (Fig. 1). Additionally, there were no differences in standing crop between irrigated and nonirrigated plots that received 0, 6.7, or 33.6 Mg/ha of biosolids; however, standing crop was reduced at 89.7 Mg/ha if irrigation was not provided (Fig. 1).

Standing Crop, 1994

There was a strong ($P < 0.0123$) interaction between irrigation and biosolids rate in their effects on standing crop during the second growing season post-application (Fig. 2). Similar to results

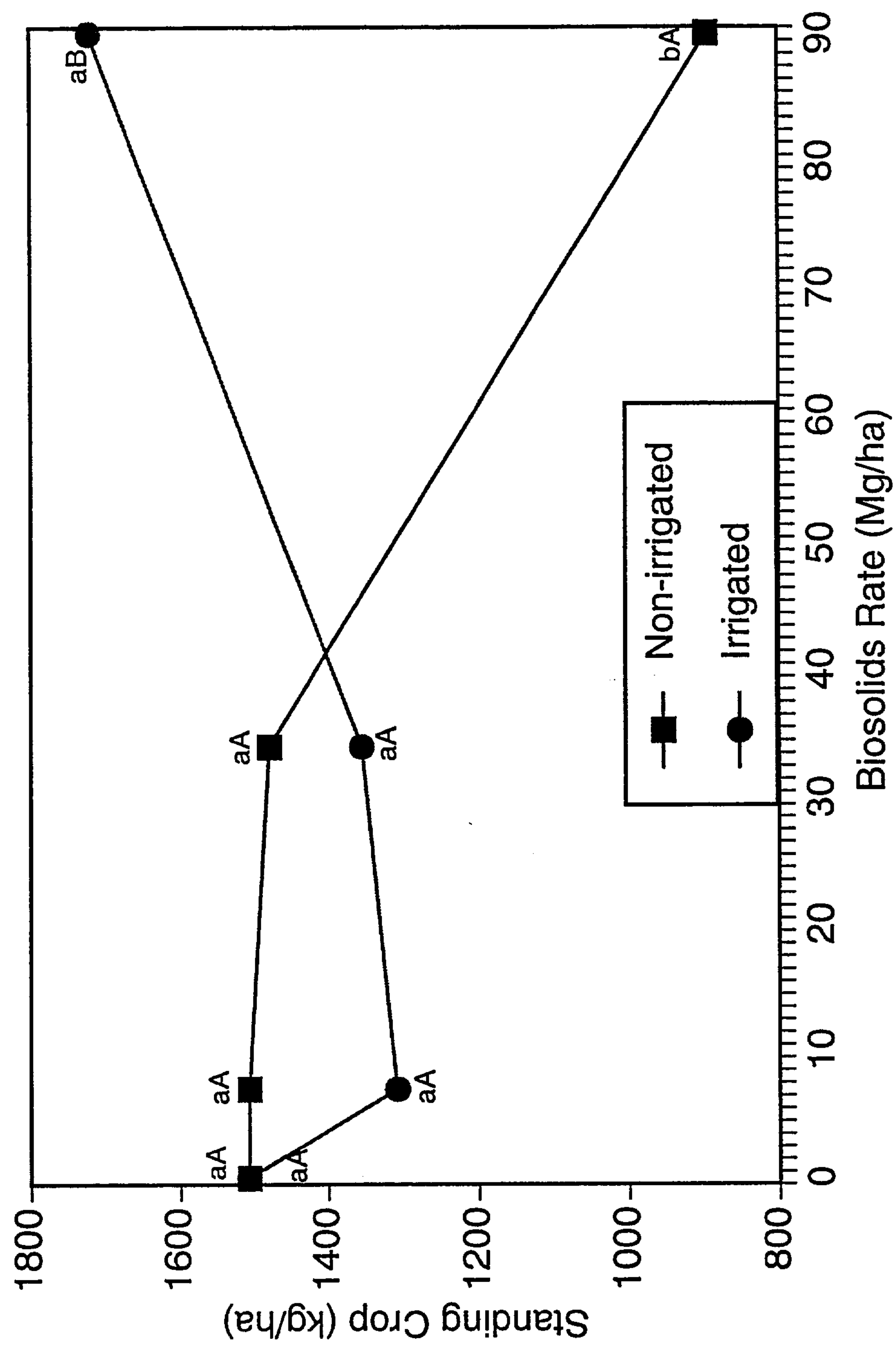

Figure 1. Tobosagrass standing crop, 1993, in plots that received biosolids in September, 1992. Rate means within an irrigation level followed by the same lowercase letter are not significantly different (P > 0.05). Irrigation means at the rate followed by the same uppercase letter are not significantly different (P > 0.05).

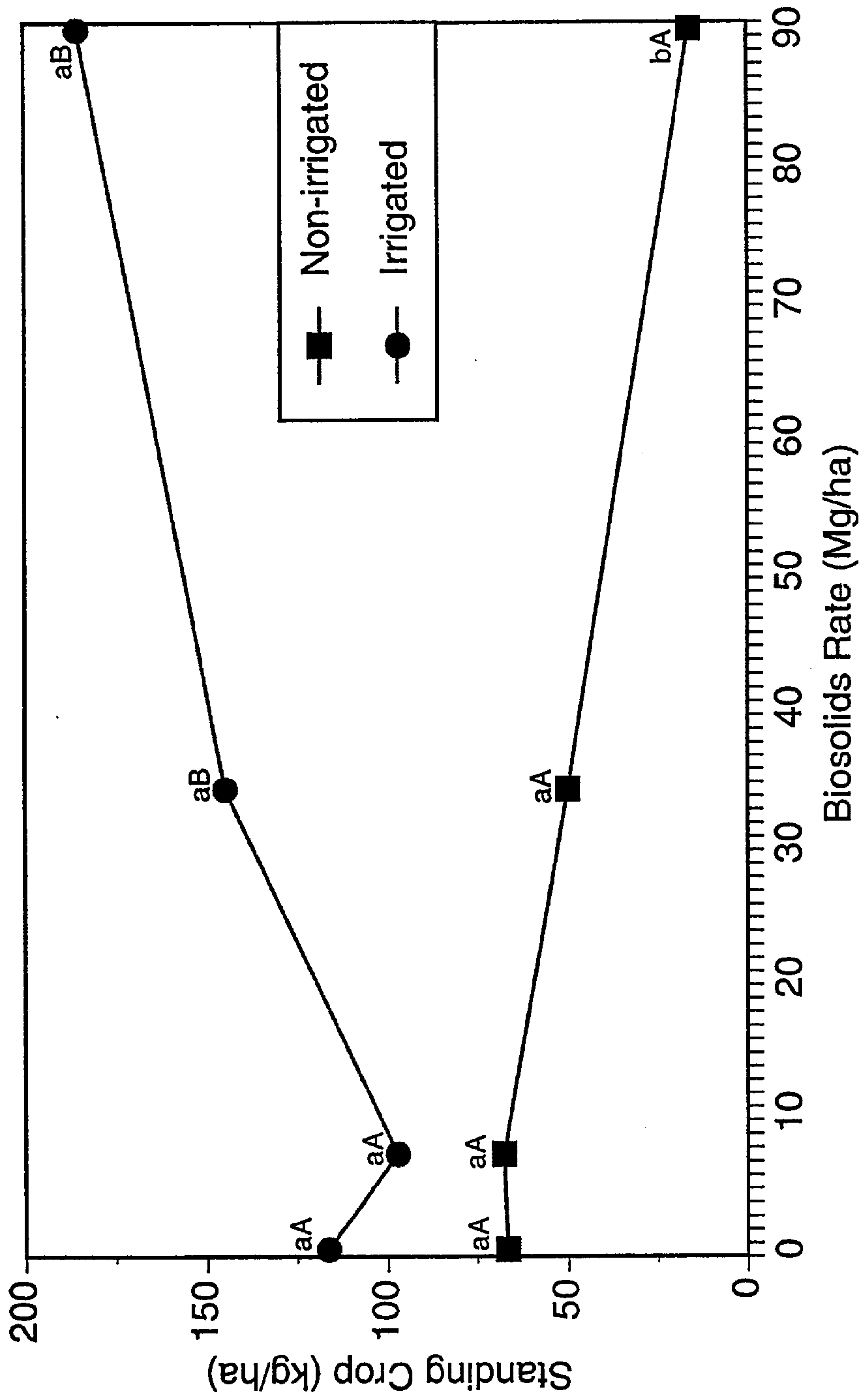

Figure 2. Tobosagrass standing crop, 1994, in plots that received biosolids in September, 1992. Rate means within an irrigation level followed by the same lowercase letter are not significantly different ($P > 0.05$). Irrigation means at the rate followed by the same uppercase letter are not significantly different ($P > 0.05$).

in 1993, standing crop was reduced in nonirrigated plots at the 89.7 Mg/ha rate compared to 0, 6.7 and 33.6 Mg/ha treatments. Also, biosolids did not affect standing crop in irrigated plots. Additionally, there were no differences in standing crop between irrigated and nonirrigated plots that received 0 or 6.7 Mg/ha of biosolids. However, standing crop was increased under irrigated conditions when 33.6 or 89.7 Mg/ha rates were applied compared to nonirrigated conditioned.

Comparison of Standing Crop Between 1993 and 1994
Standing crop in nonirrigated control plots was 22 times greater in 1993 than in 1994, a result attributed to the relatively poor precipitation conditions in 1994. During 1994 the proportional change in standing crop as biosolids increased was more strongly dependent on irrigation than in 1993. Also, the effect of irrigation in increasing standing crop was stronger ($P < 0.0001$) in 1994 than in 1993 ($P < 0.2638$).

Experiment II: Once-Per-Year Seasonal Applications First Growing Season Post Application
Tobosagrass
Biosolids rate ($P < 0.0001$), season of application ($P < 0.0013$), and irrigation ($P < 0.0001$) affected tobosagrass standing crop the first growing season after biosolids application; additionally, season and rate of application interacted in their effects on standing crop (Fig. 3). Irrigated plots produced 1,979 kg/ha of standing crop whereas nonirrigated plots produced only 1,736 kg/ha.

In plots that received biosolids in the summer of 1993, standing crop was similar in 0 and 6.7 Mg/ha treatment; additionally, standing crop at these rates was less than standing crop at 17.9, 33.6 or 89.7 Mg/ha (Fig. 3). In plots that received biosolids in the winter of 1993, however, standing crop was successively increased by 6.7, 17.9 and 33.6 Mg/ha of biosolids. Also, standing crop at 89.7 Mg/ha did not differ from standing crop at 17.9 or 33.6 Mg/ha rates.

Alkali Sacaton
Biosolids rate ($P < 0.0001$), season of application ($P < 0.0001$), and irrigation ($P < 0.0001$) affected alkali sacaton standing crop in the first growing season after biosolid application; also, season and rate of application interacted ($P < 0.0001$) in their effects on standing crop (Fig. 4). Irrigated plots produced 1,921 kg/ha whereas nonirrigated plots produced 1,748 kg/ha.

There was not a strong rate response to biosolids application in plots that were treated in the summer of 1993 (Fig. 4). In contrast, there was a strong rate response in plots that received biosolids in winter, 1993. Standing crop was successively increased by 6.7 and 17.9 Mg/ha; additionally, standing crop at the 89.7 Mg/ha rate did not differ from standing crop at the 33.6 Mg/ha rate.

Second Growing Season Post-Application
Tobosagrass
Tobosagrass standing crop in 1994 was affected by biosolids rate ($P < 0.0001$) and irrigation ($P < 0.0001$) effects; additionally, there was a rate of application by years of application interaction ($P < 0.0349$) (Fig. 5). Irrigated plots produced 393 kg/ha of standing crop compared to only 197 kg/ha in nonirrigated plots.

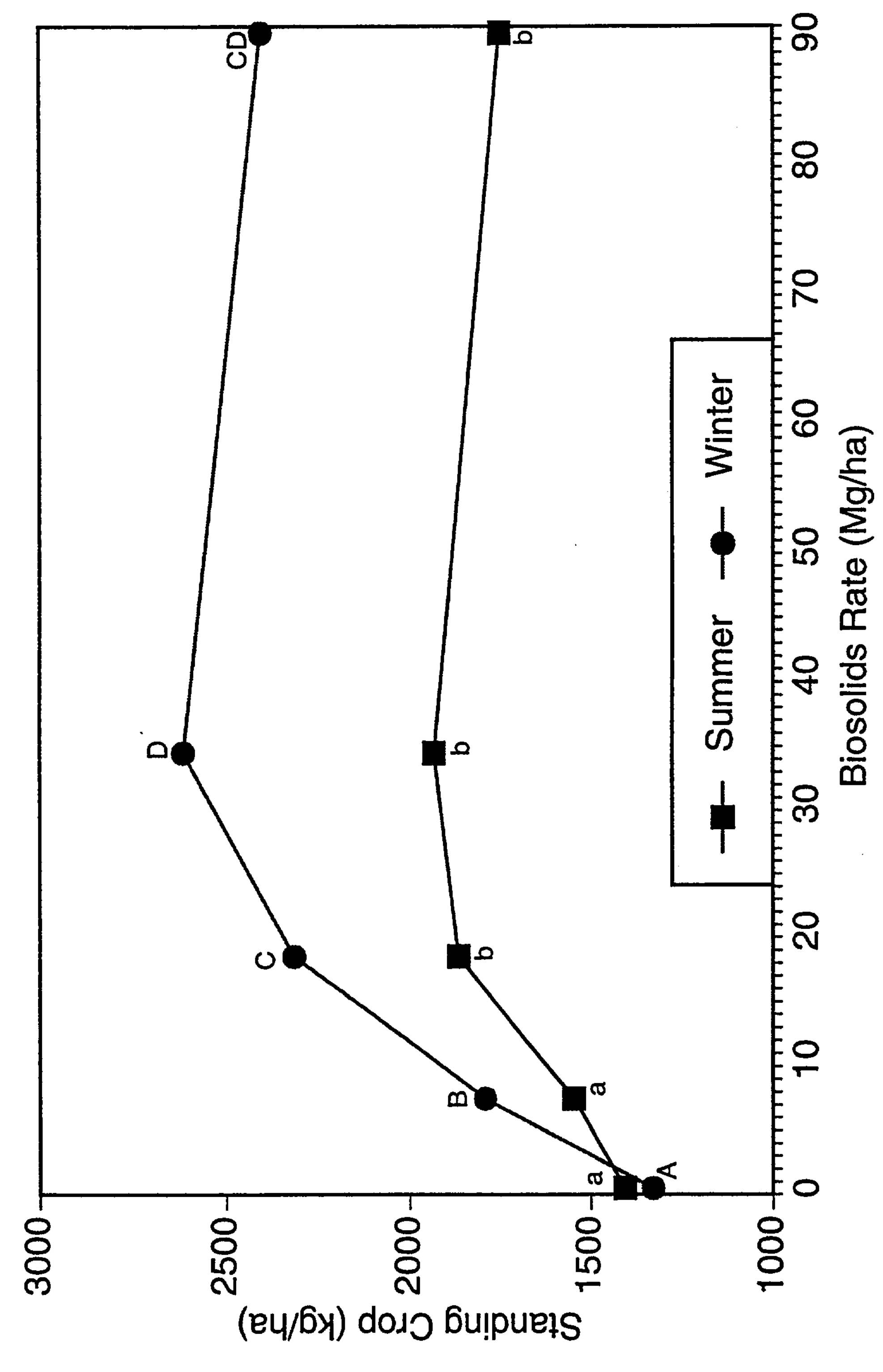

Figure 3. Tobosagrass standing crop, 1993, in plots that received biosolids in winter or summer, 1993. Rate means within a season of application followed by the same letter are not significantly different (P > 0.05).

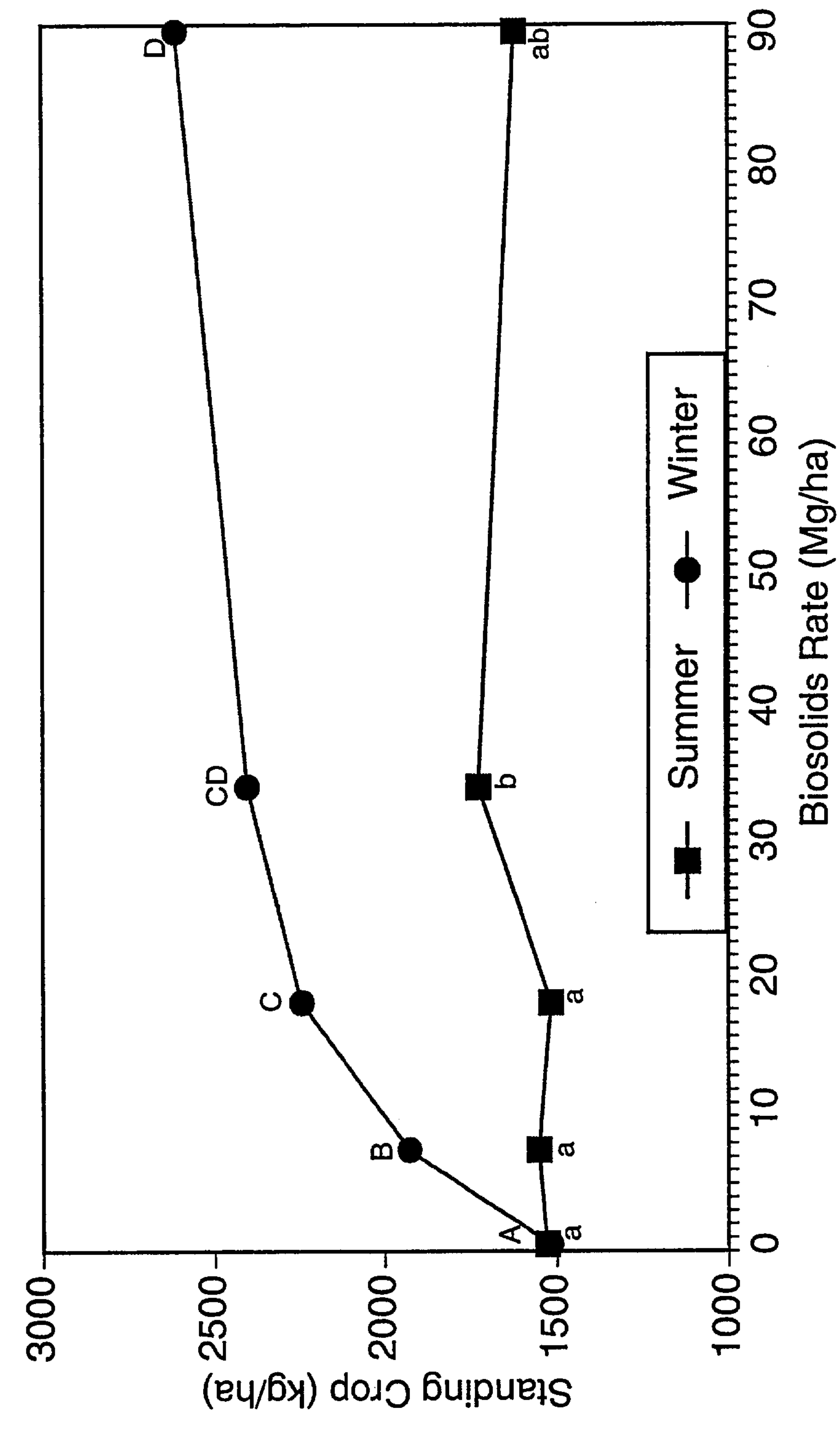

Figure 4. Alkali sacaton standing crop, 1993, in plots that received biosolids in winter or summer, 1993. Rate means within a season of application followed by the same letter are not significantly different (P > 0.05).

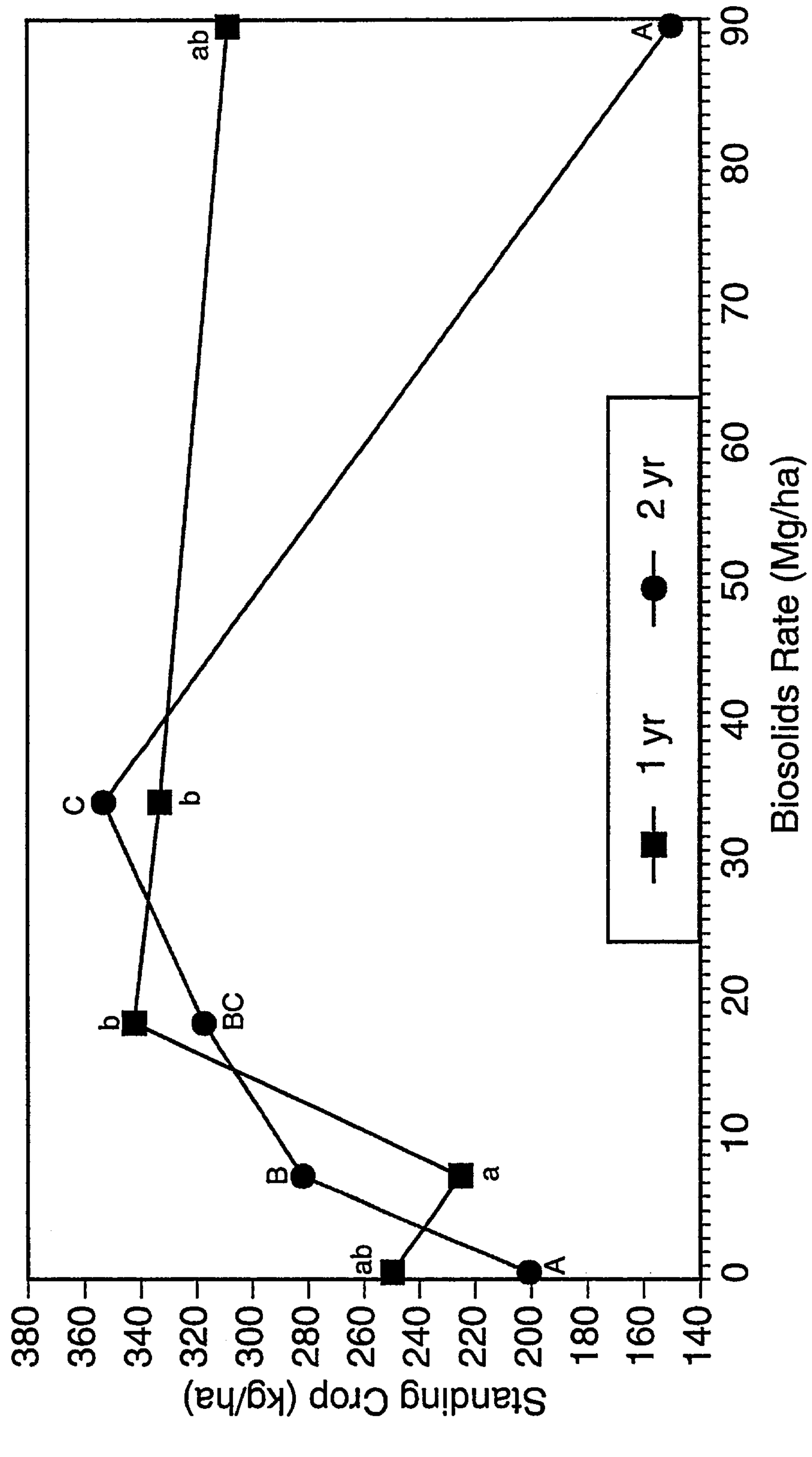

Figure 5. Tobosograss standing crop, 1994, in plots that received biosolids for one year only or for two consecutive years. Rate means for a given number of years of application followed by the same letter are not significantly different (P > 0.05).

In plots that were treated with biosolids in year 1 only, standing crop tended to increase up to the 17.9 Mg/ha rate; additionally, standing crop did not decline at the higher application rates (Fig. 5). Standing crop increased at rates up to 33.6 Mg/ha in plots that received biosolids for 2 consecutive years. However, in contrast to plots treated in year 1 only, standing crop decreased at the 89.7 Mg/ha rate when treated for 2 consecutive years.

Alkali Sacaton
Alkali sacaton standing crop was affected by biosolids rate ($P < 0.0001$), irrigation ($P < 0.0001$) and season of application (P 0.0280) as well as by a season by years of application interaction ($P < 0.0358$). Irrigated plots yielded 597 kg/ha compared to only 373 kg/ha in nonirrigated plots.

Standing crop increased up to the 17.9 Mg/ha biosolids rate. Standing crop decreased at the highest biosolids rate, and at this rate did not differ from standing crop at 0 and 6.7 Mg/ha rates.

Season and years of application interacted in their effects on alkali sacaton standing crop. In plots that received biosolids in year 1 only, winter-treated plots produced more standing crop than summer-treated plots. However, there was no difference in standing crop between winter- and summer-treated plots when biosolids were applied 2 consecutive years.

Experiment III: Twice-Per-Year Seasonal Applications
Tobosagrass standing crop was affected by biosolids rate ($P < 0.0001$), irrigation ($P < 0.0001$), season by rate interaction ($P < 0.0211$) and season by irrigation interaction ($P < 0.0297$). Standing crop responded more to biosolids applied in the winter/summer seasons than in the spring/summer seasons (Fig. 7), an effect especially apparent at the highest application rate.

Season of application and irrigation interacted in their effects on tobosagrass standing crop (Fig. 8). The irrigation effect was stronger in plots that received biosolids in winter/summer seasons than in spring/summer seasons.

Discussion

Plant production response to topically applied biosolids in a Chihuahuan desert grassland was species-specific and depended on season of application, growing conditions (i.e., soil moisture availability), and number of repeated applications.

Experiment I (one-time-only autumn application) yielded results different from other experiments in several ways. In Experiment I, the response of tobosagrass standing crop the first growing season after application differed in pattern as well as in degree when compared with dormant or growing season application. In essence, tobosagrass standing crop in this experiment was unaffected by biosolids (except for a decrease at the 89.7 Mg/ha rate in nonirrigated plots). Likewise, standing crop the second growing season post application was different in Experiment I than in other experiments.

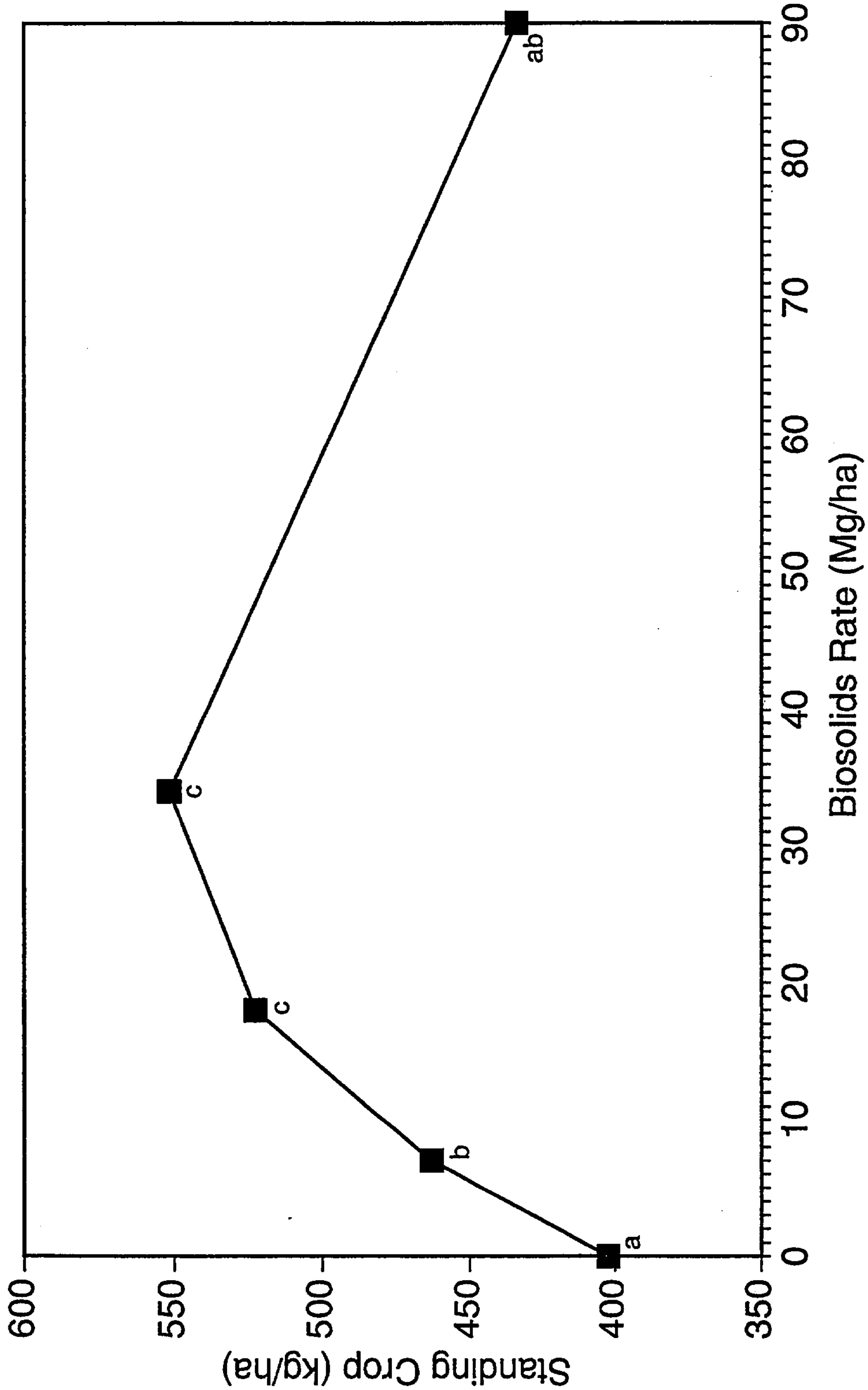

Figure 6. Alkali sacaton standing crop, 1994. Rate means followed by the same letter are not significantly different (P > 0.05).

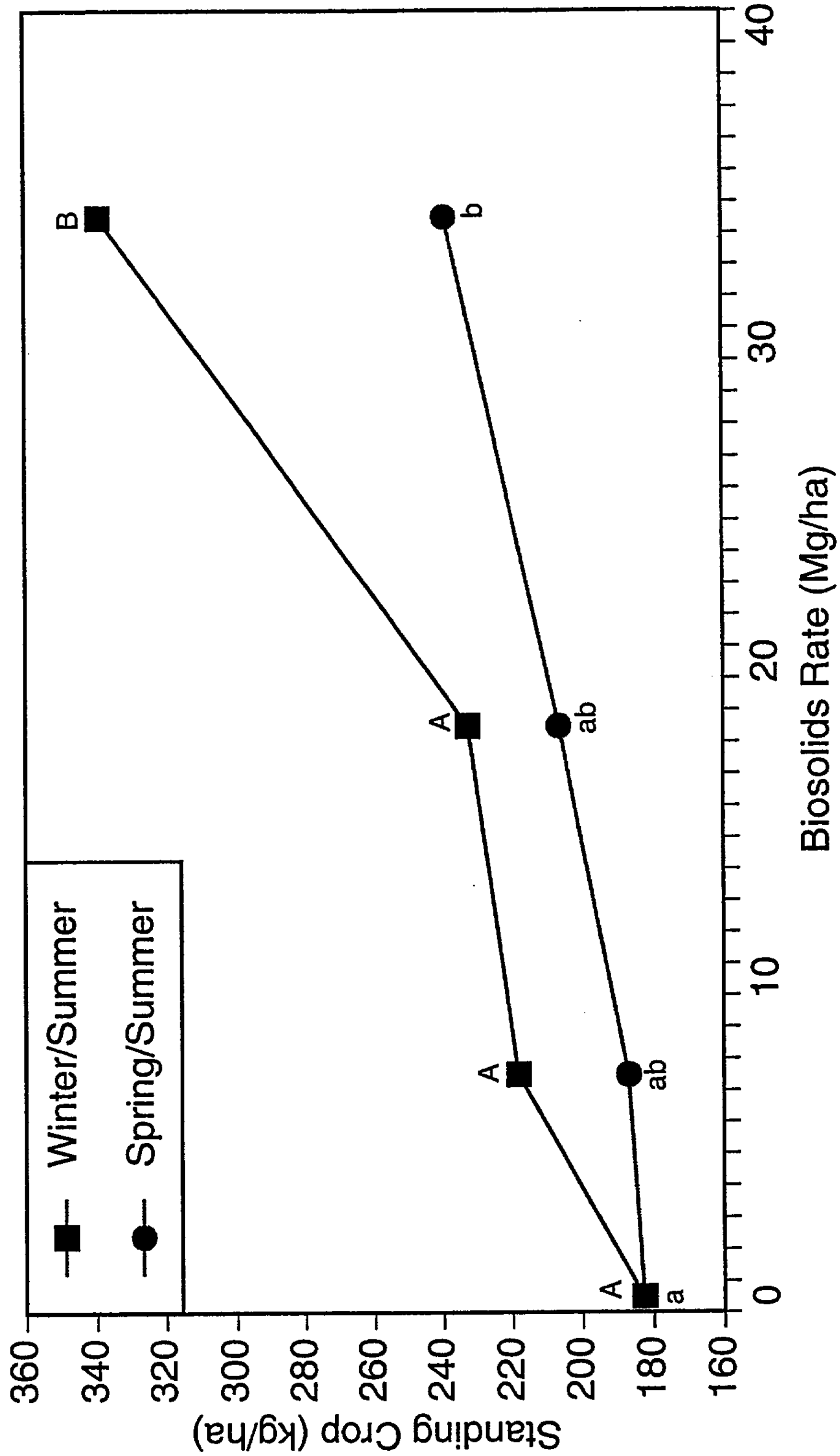

Figure 7. Tobosagrass standing crop, 1994, in plots that received biosolids in winter/summer or spring/summer seasons. Rate means within a seasonal application treatment followed by the same letter are not significantly different ($P > 0.05$).

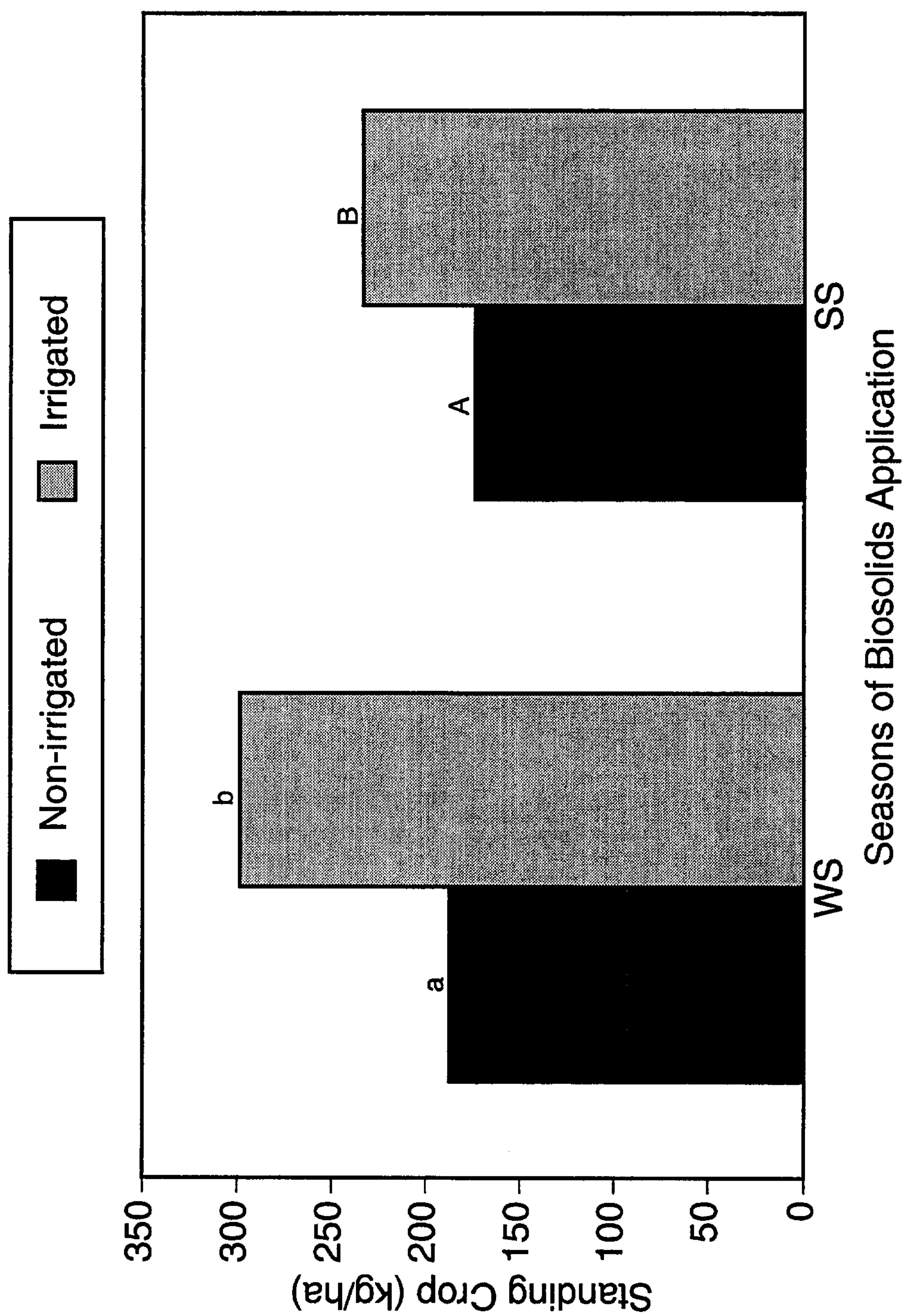

Figure 8. Tobosagrass standing crop, 1994. Irrigation means within a seasonal application treatment followed by the same letter are not significantly different (P > 0.05).

Several factors probably contribute to these differences. In Experiment I, experimental units were 0.5 m^2 whereas in Experiments II and III, plots 0.85 m^2 in area were treated with biosolids. Although application rates were similar (e.g., 6.7 Mg/ha were applied in both studies), the actual amount of surface area treated with biosolids was less in Experiment I. This "treated area effect" (which is currently being investigated in a separate experiment) may have influenced results. In addition, experimental units in Experiment I were in an area approximately 0.5 acre in size. As a precaution against accidental range fire, a fire line was bladed around this experimental area. This line effectively prevented any overland flow of water from natural rainfall events from reaching the experimental plots: in a sense, these plots were physically located in a "rain shadow".

Finally, Experiment I was a smaller experiment than Experiment II: a biosolids rate/irrigation treatment mean was based on n=6 observations in Experiment I. In contrast, in year 1 a biosolids rate/irrigation mean was based on n=24 observations in Experiment II (because this study was designed to investigate up to 4 consecutive years of application). Standing crop was lognormally distributed, and empirical observations suggested that means may be expected to increase as sample size increases. This effect, however, is likely to be complicated by the "treated area effect" discussed above.

Both tobosagrass and alkali sacaton responded more strongly the first growing season after dormant season application than after growing season application of biosolids. In fact, there was very little alkali sacaton response to summer-applied biosolids in the first year. Although irrigation effects were significant, irrigation was not involved in any interactions. Thus, the response of these species to biosolids was independent of irrigation effects.

Although the reasons underlying the more positive response to dormant season application are unclear, and this research was not designed to elucidate mechanisms involved in this phenomenon, it is likely that both physical and chemical (i.e. nutritional) effects are involved. From a physical standpoint, topically applied biosolids may act as a mulch, reducing soil moisture loss during the spring between dormant season application and the onset of the growing season. Additionally, with a C:N ratio of approximately 12:1, topically applied biosolids may act to stimulate soil microflora, the net effect of which is more available soil nutrients at the beginning of the growing season. This effect may be greater for plots treated with biosolids 6 months before the onset of the growing season than for plots treated 1-2 weeks before the onset of the growing season.

For rates up to 33.6 Mg/ha, there was little difference in second-year tobosagrass standing crop for plots treated one year only or for 2 consecutive years. However, second-year standing crop was significantly reduced by 2 consecutive years at the 89.7 Mg/ha rate. The physical barrier created by 2 consecutive years of biosolids at this rate may have prevented precipitation from reaching the soil surface, an effect especially detrimental during a dry year. Similar results were shown by alkali sacaton.

When biosolids were applied twice per year, tobosagrass standing crop was greater under dormant/growing season application than under spring/growing season application. In addition, there was a greater response to irrigation in plots that received biosolids in dormant/growing than in spring/growing season applications. These results are generally consistent with findings from the once/year applications (Experiment II).

Conclusions

Topically applied biosolids generally increased standing crop of tobosagrass and alkali sacaton. This effect was more pronounced when biosolids were applied in the dormant season than when they were applied just prior to the onset of growth. Two consecutive years of annual biosolids application at 89.7 Mg/ha reduced plant growth. This effect was probably related to physical consequences of 10 cm of biosolids laying on the soil surface where incoming rainfall may be intercepted and thereby prevented from reaching the soil surface. In plots that received biosolids twice per year, standing crop was greater with dormant/growing applications than with spring/growing season applications. In general, these results suggest that there is a positive production response for tobosagrass and alkali sacaton applied at rates up to 17.9 Mg/ha.

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Literature Cited

Aguilar, R., S.R. Loftin, T.J. Ward, K.A. Stevens, and J.R. Gosz. 1994. Sewage sludge application in semiarid grasslands: effects vegetation and water quality. New Mexico Water Resources Research Institute Tech. Completion Report, Acct. No. 013456999.

Fresquez, P.R., R. Aguilar, R.E. Francis, and E.F. Aldon. 1991. Heavy metal uptake by blue grama growing in degraded semiarid soil amended with sewage sludge. Water, Air, and Soil Pollution 57-58:903-912.

Fresquez, P.R., R.E. Francis, and G.L. Dennis. 1990a. Sewage sludge effects on soil and plant quality in a degraded, semiarid grassland. J. Environ. Qual. 19:423-429.

Fresquez, P.R., R.E. Francis, and G.L. Dennis. 1990b. Soil and vegetation responses to sewage sludge on a degraded semiarid broom snakeweed/blue grama plant community. J. Range Manage. 43:325-331.

Levene, H. 1960. Robust tests for equality of variance. In: Contributions to Probability and Statistics, I. Olkin (ed.), Stanford Univ. Press, Palo Alto.

Shapiro, S.S. and M.B. Wilk. 1965. An analysis of variance test for nonnormality (complete samples) Biometrika 52:591-611.

Evaluation of Sewage Sludge Products for use in sheep production in Australia: Some preliminary results

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Abstract

A grazing experiment was commenced in 1992 in southern New South Wales, Australia, to assess the benefits and risks associated with the application of sewage waste products (dewatered cake and N-Viro Soil) to pastures grown on poor soils which are grazed by sheep. Preliminary results indicate that N, P, Zn, Cu and Ni are the most mobile elements in these acid (pH[salt]<4.7) soils, and that the concentration of these are most likely to determine the commercial application rate of dewatered cake to pastoral land. Sewage waste equalled or improved the P, S, Ca and Mg content in plant tops relative to fertilizer, and also increased Zn and Ni concentrations, but not to phytotoxic levels. Biosolid contaminants have not been detected in ground water. Sheep production, product quality and livestock health have not been affected by these sewage-induced changes in soil and pasture after one year of exposure.

Key words: Biosolids, dewatered cake, N-Viro Soil, heavy metals, zinc, copper, sheep, pastures, acid soil, Australia.

Introduction

The disposal of sewage biosolids is becoming a major economic and environmental problem, and countries around the world are committing increasing resources to find effective long-term solutions. In New South Wales (NSW), Australia's most populous state, the production of residual products from the treatment of sewage and drinking water already exceeds 150 t (dry weight) per day, with Sydney accounting for almost half of this biosolids output. In 1990, Finney and Rawlinson reported that 60% of Sydney's wastewater solids was disposed into the ocean with the balance either incinerated (8%), used for coal dump rehabilitation (7%) or converted into products for agricultural use (25%). This level of agricultural use is low compared to some other countries. For example, 1970 statistics indicated that 40% of biosolids from inland sewage plants in the United Kingdom was applied to agricultural land (Sterritt and Lester 1980). However, this situation should change in Australia as disposal of biosolids into the ocean is now prohibited by new anti-

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pollution regulations which has forced wastewater treatment authorities to find alternative low-risk disposal methods.

Application of liquid, dewatered and alkali-treated biosolids to agricultural land is an attractive, cost effective disposal option which recycles organic matter and nutrients back to the soil in an environmentally acceptable manner. This method is widely practised overseas, but it is only recently that it has been considered as a feasible option for sewage waste disposal even though biosolids produced in NSW have an estimated fertilizer value of about $10 million/year (Ross *et al.* 1989). However, while applying sewage products to agricultural land provides a unique opportunity to improve the physical and chemical properties of poor soils, the concentrations of nutrients, trace elements, heavy metals and persistent organics contained in biosolids pose a severe hazard if applied injudiciously (Chaney *et al.* 1987; Lindsay 1983).

Heavy metals in large quantities may inhibit plant growth, adversely affect long-term productivity, and lead to accumulation of metals in the food chain. Some of these accumulations may be detrimental to human health with arsenic (As), cadmium (Cd), lead (Pb), mercury (Hg) and nickel (Ni) the most frequently identified as posing risk to human health (Seiler *et al.* 1988). Cd and Cu are of particular concern in Australia. A long history of applying superphosphate containing variable levels of Cd has increased soil Cd from 0.016 to 0.114 mg/kg in unfertilized soils to a range of 0.076 to 0.342 mg/kg in highly fertilized soils (Williams and David 1976). These elevated Cd levels have caused violations in residue limits in some livestock products (Langlands *et al.* 1988). This means that the scope for applying sewage products containing further Cd is limited and must be closely controlled (Ross *et al.* 1989). Cu poisoning is also a regular occurrence in Australian sheep and cattle, usually after incorrect supplementation for copper deficiency, indicating that the margin for safety for Cu is slim (Seawright 1989).

The bioavailability of metal contaminants can be reduced by applying lime to soil or pretreating dewatered solids with pH amendment agents. However, this may increase the cost of sewage waste and reduce availability of important plant nutrients. The latter may be advantageous in some cases as plant nutrients applied in excess of requirements also pose an important potential risk to the environment. Nitrogen (N) derived from sewage waste can contaminate ground and surface water supplies (Addiscott 1988), while excess levels of phosphorus (P) can result in the eutrophication of dams and waterways (Curran 1991). Significant downward movement of Zn and Cu to >1 m depth has also been reported on light acid soils with low cation exchange capacity (Sidle and Kardos 1977; Taylor *et al.* 1982; Lawrie *et al.* 1991).

The potential to pollute agricultural ecosystems by land application of biosolids is determined by the concentration of contaminants in the sewage product, the amount and frequency of application, the nutrient requirements of target crops and pastures, and the properties of the soils to which they are applied. For Sydney's wastewater solids, De Vries (1983) and Copeland (1990) reported relatively low levels of heavy metals in raw sewage from residential areas compared with other more highly industrialized parts of the city, but did note elevated Cu, Zn and Pb levels (Table 1). While biosolids produced from the

Table 1. Concentration of some elements in sewage from industrial and residential sectors of Sydney (de Vries 1983).

Nutrient/metal	Industrial (Malabar)	Residential (Warragamba)
Total N (%)	2.6	3.8
Total P (%)	0.9	0.6
Total Ca (%)	1.9	2.1
Total K (%)	0.1	0.1
Total Mg (%)	1.9	1.9
Total Na (%)	0.1	0.1
Cu (mg/kg)	872	396
Zn (mg/kg)	2190	1900
Mn (mg/kg)	214	114
Cd (mg/kg)	32	3
Ni (mg/kg)	156	25
Pb (mg/kg)	173	156

Sydney Water Corporation's treatment facilities in the non-industrial sectors of the city appear to be suitable for use in agricultural production on soils with pH >6.0, especially in regard to Cd, there is little local information available to frame regulations regarding rates and frequency of application to commercial crop and pasture systems grown on the majority of Australian soils which are highly degraded, infertile and acidic.

In fact, the initial guidelines specifying annual and maximum cumulative loadings of nutrients and contaminants (Ross *et al.* 1991) for sludge usage in Australia were based largely on overseas regulations and experiences. This was unsatisfactory as Australian soils are sufficiently different from those used for recycling sewage products in other parts of the world to preclude direct comparison and extrapolation from overseas research. To foster safe and sustainable use of biosolids in Australia, it is important that we develop guidelines appropriate to our environmental conditions and agricultural systems based on local research.

The development of such guidelines requires assessment of the benefits accrued from the nutrient and organic matter inputs and the risk of environmental pollution and product spoilage associated with heavy metal and organic contaminants resident in different biosolid products. The fourteen risk pathways identified by the United States Environmental Protection Agency (USEPA 1989a, 1989b) to assess the hazards of transferring sewage-applied contaminants to the environment and to the foodchains of livestock and man provide an excellent scientific framework (Logan and Page 1989) for evaluating the impact of biosolids usage in the Australian context. In 1991, the Sydney Water Corporation provided NSW Agriculture with an $8 million grant to establish a comprehensive research program to assess the risks associated with a range of potential uses of biosolids in NSW.

One area of potential use targeted in NSW is the 8 million hectares of highly degraded, in-

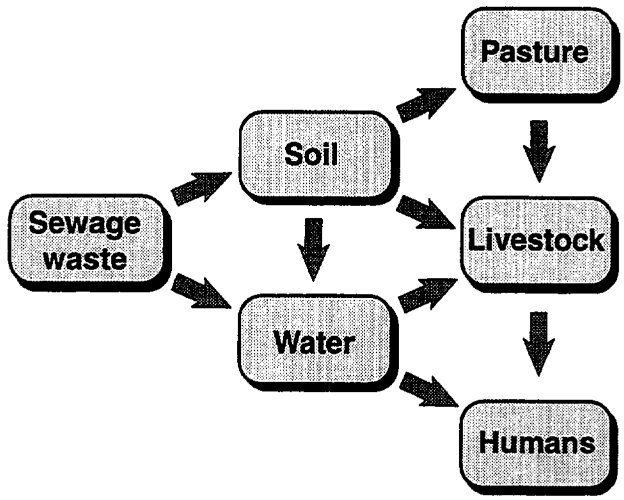

Figure 1. Risk pathways for contamination of a pasture based livestock production system by heavy metals and organics derived from sewage sludge.

fertile, acid ($pH_{[CaCl2]}$<5.0) soils (Helyar *et al.* 1990) which have low productivity in their present state. Extensive wool and sheep meat production which is often the only viable agricultural use possible on these soils, are significant agricultural enterprises. In the Southern Tablelands region where these soils account for the majority of the grazing lands, sheep enterprises generate an annual revenue exceeding $380 million which represents 76% of the total agricultural production. In general, these soils have low cation exchange (5-10 meq %), low organic matter (<2%), P and N deficiencies, and light-textured, shallow top-soil (35-50 cm) overlaying heavier clay subsoil which severely restrict rooting depth. The physical and chemical characteristics of these soils could be rapidly improved by the nutrients and organic matter contained in sewage products provided the risks of environmental pollution and possible contamination of pasture, meat and wool can be minimized by selecting appropriate sewage products and application rates.

Formulation of regulations to achieve these objectives for land application of low metal biosolids to such soils which are used for livestock production is a formidable task which requires well-designed, long-term field experiments. In 1992, a large-scale grazing experiment was established at Goulburn (34°47'S; 149°43'E) on the Southern Tablelands of NSW to investigate the impacts of sewage waste products on a pasture-based sheep grazing ecosystem. The two main aims of the program were: (1) to assess the potential of biosolids as an economic and sustainable soil ameliorant and fertilizer for sheep production systems; and (2) to measure the effects of metal and organic contaminants of sewage products on the water, soil, plant and livestock components of the pasture ecosystem using the USEPA risk pathway model. The risk pathways addressed in this project are shown in Figure 1.

Materials and Methods

Site characteristics. The 100 ha experimental site selected is typical of the land class used for low-input sheep grazing enterprises in NSW. The site covers a series of three hill formations with altitude ranging from 640 to 680 m. Hill A which is located at the southern end of the site is undulating (slopes <<10 %) with a maximum relief of 35 m. Hill B is an undulating low rise with a maximum relief of less than 10 m. Hill C is located at the northern end of the site and is the steepest area with maximum relief of more than 40 m and small areas exceeding 10%.

Ten plots of approximately 1.1 ha each were laid out on each of the three major soil types identified by an extensive preparatory soil survey taken at fixed 100 m intervals over the entire area. These soils which represent 70% of the soils of the region and which roughly correspond with the hill formations differ in texture and clay content, but have similar chemical properties to those of the sandy loam duplex soil (Aeric Kandiaqualfs in *Soil Taxonomy*) described in Table 3 found on Hill A.

The site was an unimproved pastoral area dominated by red grass (*Bothriochloa macra)* with a small percentage of spear grass (*Stipa* spp.), wallaby grass (*Danthonia* spp.) and weeping grass (*Microlaena* spp.). Some subterranean clover (*Trifolium subterraneum*) (cv. Mt Barker) was present, but was not a major pasture component. Prior to clearing, the native vegetation was a savanna woodland of yellow box (*Eucalyptus melliodora*) and red gum (*E. camaldulensis*).

Pre-treatment earth works. The Soil Conservation Service of NSW designed and implemented a series of detention structures to minimise the movement of suspended sediment off the experimental site. These structures included a series of banks, dams, sediment traps and "silt stop" fencing. Since these structures were installed in January 1993, no runoff has escaped from the experimental site.

Treatments. A control and four biosolid treatments were duplicated on each soil type. The control was based on best agricultural practice for pasture development recommended by local agronomists for these soil types. This consisted of the broadcast application and incorporation of 2.5 t/ha of lime in summer prior to pasture sowing in autumn. Lime plus superphosphate (50:50) was applied at 400 kg/ha at sowing with annual maintenance input of 250 kg/ha superphosphate.

N-Viro Soil (alkali-treated sludge designed to reduce availability of heavy metals) was applied at 7.5 t/ha which was the equivalent neutralizing rate of lime applied in the control. Dewatered sludge cake (28% solid) obtained from the Malabar Sewage Treatment Plant (STP) was applied at 30, 60 and 120 t/ha as a one-time application and incorporated with offset discs within 24 hours of application as specified in the "Interim Guidelines for Land Application of Sewage Waste Products in NSW" (Ross *et al.* 1991). Malabar STP sewage waste which was also the biosolid source used in the manufacture of N-Viro Soil, was chosen to ensure that the maximum cumulative loading (MCL) for at least some heavy metals was approached at the highest application rate (Table 2).

A total of 1827 dry tonne of dewatered sewage waste was applied to the Goulburn site during the period December 15, 1992 and April 8, 1993. Samples were randomly collected during the application phase and were analysed to determine the loadings of nutrients, heavy metals and organochlorines on each treatment (Table 2).

Table 2. Loading rates (kg/ha) of nutrients,heavy metals and organics contained in fertilizers and sludge products applied to Hill A at Goulburn.

Loading rate of nutrients (kg/ha)	Control	Treatments N-Viro Soil	30t/ha	Dewatered cake 60 t/ha	120 t/ha	Maximum cumulative loading[1]
Total Nitrogen	NM	90[4]	744[4]	1715[4]	3528[4]	400[2]
Total Phosphorus	21	58[4]	478[4]	966[4]	1903[4]	NR
Total Calcium	1099	1503[4]	915	1729	3984	NR
Total Potassium	NM	255[4]	37[4]	76[4]	126[4]	NR
Total Magnesium	10	31[4]	93[4]	197[4]	416[4]	NR
Total Sulphur	37	86[4]	452[4]	907[4]	1656[4]	NR
Heavy metal or organics loading rate (kg/ha)						
Arsenic	0.01	0.08	0.11	0.35	0.54	20[3]
Cadmium	0.01	0.05	0.42	0.79	1.56	3[3]
Chromium	0.02	1.34	9.72	18.70	36.48	100[3]
Copper	0.01	4.32	56.79	74.55	145.11	100[3]
Manganese	0.41	2.89	6.01	12.32	23.28	NR
Mercury	0.01	0.01	0.17	0.35	0.66	1[3]
Nickel	0.02	0.62	5.24	10.10	19.56	60[3]
Lead	0.05	1.99	10.15	20.35	38.88	150[3]
Selenium	0.02	0.14	0.65	1.37	2.55	5[3]
Zinc	0.14	7.0	84.94	177.42	332.04	200[3]
Dieldrin	NM	BD	0.01	0.02	0.05	0.02[3]
Chlordane	NM	BD	0.02	0.03	0.06	0.02[3]
DDT/DDD/DDE	NM	BD	BD	BD	0.03	0.5[3]

[1]Levels specified in "Interim Code of Practice for Use and Disposal of Biosolid Products", Draft, June 1994, NSW Environmetal Protection Authority; [2]Crop nitrogen requirement given kg total N/ha/year; [3]MCL given as kg/ha assuming incorporation to 0.075 m in soil with bulk density of 1.33 dry t/m^3; [4]Nutrient loadings for N-Viro Soil and sludge includes basal fertilizer applied at sowing; NM = not measured; NR = no restriction; BD = below detection.

ents, heavy metals and organochlorines on each treatment (Table 2).

Assessment of water quality. Background samples of water from appropriate areas downstream from the sludge application site were analysed for bacteria, chemicals and metals which are common contaminants of sewage waste to identify ambient levels in the watershed before sewage products were applied. Following application of biosolids, water quality was monitored regularly over a 12 month period at 24 fixed locations in the two catchments. Also, in accordance with EPA requirements, piezometers were installed at 15 and 30 m to assess and monitor quality of deep groundwater.

Pastures and sheep. In autumn 1993, all plots were sown with a pasture mixture which included subterranean clover (*Trifolium subterraneum*), white clover (*T. repens*),

Table 3. Chemical characteristics of a duplex soil (0-30 cm) at Goulburn before and after application of fertilizer and lime (Control), N-Viro Soil and dewatered cake at three rates (After Michalk *et al.* in press).

Treatment/ soil depth (cm)	pH[1]	Bray P (mg/kg)	Total C (%)	Total N (%)	Exchangeable cations[2] Ca	Mg	K	Na	Al	Total CEC
Prior to treatment										
Depth: 0-10	4.7	2	2.1	0.1	2.8	2.4	0.3	0.2	0.3	6.0
10-20	4.9	1	1.1	0.1	2.2	2.9	0.3	0.2	0.1	6.7
20-30	4.9	0	1.0	0.1	2.4	6.6	0.3	0.6	0.1	10.0
After treatment - Spring 1993										
Control										
Depth: 0-10	6.0	24	2.9	0.1	5.0	1.0	0.3	0.1	0.0	6.4
10-20	5.0	3	0.7	0.1	2.0	1.7	0.3	0.2	0.1	4.3
20-30	5.2	<3	0.5	0.1	2.1	4.6	0.2	0.7	0.0	7.6
N-Viro Soil										
Depth: 0-10	5.7	16	1.5	0.1	4.1	0.7	0.6	0.1	0.0	5.5
10-20	5.1	3	0.9	0.1	2.3	1.3	0.3	0.1	0.0	4.0
20-30	5.1	<3	0.5	0.1	1.8	3.0	0.2	0.1	0.0	5.1
Dewatered cake										
30 t/ha cake										
Depth: 0-10	4.7	69	2.1	0.2	4.0	1.3	0.4	0.1	0.1	5.9
10-20	4.7	21	1.0	0.1	2.8	2.3	0.4	0.0	0.1	5.6
20-30	5.0	23	0.9	0.1	3.0	4.4	0.3	0.2	0.0	7.9
60 t/ha cake										
Depth: 0-10	4.8	126	2.6	0.2	5.3	1.0	0.4	0.1	0.1	6.9
10-20	4.6	23	0.8	0.1	2.2	1.0	0.2	0.0	0.1	3.5
20-30	4.7	13	0.7	0.1	2.3	4.5	0.3	0.3	0.2	7.6
120 t/ha cake										
Depth: 0-10	5.4	304	5.9	0.4	13.0	2.1	0.4	0.2	0.0	15.7
10-20	5.0	81	1.3	0.2	4.3	1.6	0.3	0.0	0.0	6.2
20-30	5.1	31	0.7	0.1	3.0	5.2	0.3	0.1	0.0	8.6

[1]pH measured in $CaCl_2$; [2]Exchangeable cations expressed in meq %.

cocksfoot (*Dactylis glomerata*), phalaris (*Phalaris aquatica*), perennial ryegrass (*Lolium perenne*) and ryecorn (*Secale cereale*). A basal dressing of lime/super (250 kg/ha) was applied at sowing to all plots.

Pasture yield and botanical composition are monitored at six weekly periods along fixed transects using the dry weight rank and estimation procedures in BOTANAL (Tothill *et al.* 1992). At each sampling, forage tops are analysed for P and N. In autumn and spring each species present is sampled and analysed for heavy metals and organic pollutants.

Fine wool (<19 micron fibre diameter) Merino ewes commenced grazing the plots in spring 1993 initially at 5 ewes/plot, but stocking rate was increased to 10/plot in spring 1994. Ewes are joined in February and commence lambing in July. Lambs are weaned in

November and relocated to another replicate of the same treatment to replace ewes slaughtered for tissue assays.

Liveweight gain of ewes and lambs, and ewe condition are measured at six weekly intervals. Wool production and quality is measured annually. Fresh tissue samples from 10 ewes purchased in 1993 and 5 ewes purchased in 1994 were analysed to provide baseline levels of heavy metals and organics present in the liver, kidney, muscle and fat of sheep prior to exposure to sewage contaminants. One ewe per plot and one wether lamb per treatment are slaughtered in December each year to monitor changes in heavy metal and pesticide levels in tissues. Wool grease and fibres are analysed for heavy metals and organic pollutants. Milk samples are taken in spring and tested for heavy metals.

Supplementary feed is provided to sheep during times of feed deficit. Sheep nuts or grain is fed on the ground which is the feeding method used by local producers. This provides maximum opportunity for sheep to ingest soil (and thereby sewage contaminants) along with supplementary feed supplied. However, the amount of sewage product ingested directly by sheep was not measured in this study. Uptake of pollutants via water was eliminated by providing stock water in a trough system sourced from dams isolated from possible contamination by sewage products.

Monitoring movement of nutrients and sewage contaminants in soil and water. Runoff and exclusion plots established on each treatment at the start of the program provide the means to monitor downward and lateral (both surface and sub-surface) movement of nutrients and heavy metals in soil and water. Runoff after rainfall events is measured from standard 42 x 2 m plots located in a down-slope position. To prevent inflow from outside sources, the runoff plots are surrounded by a 20 cm timber barrier partly embedded in the soil. Runoff (including both water and solids) is collected in plastic repositories and measured after each rainfall event. Samples are analysed for nitrogen and heavy metal content.

The 2 x 2 m exclusion plots, which have received no sludge or fertiliser treatment, are also located in each treatment. Similar barriers isolate these exclosures from the surrounding pasture. However, on the down-slope side the barrier has an opening through which surface runoff from within the plot can pass to avoid waterlogging effects. Collection cups have been installed just above the clay layer (30 to 40 cm) within these exclusion plots to monitor lateral movement of N. To monitor downward and lateral movement of nutrients and sludge contaminants, soil samples are taken on the up-slope and down-slope sides within the exclusion plots as well as in adjacent positions outside. Stock are excluded from the exclosures but have access to the runoff plots. A general soil sampling is also undertaken in each treatment plot in autumn and spring each year.

Results

This paper reports some preliminary results of the effects of the applied sewage products on the water, soil, pasture and livestock components measured on the sandy loam duplex soil (Hill A) described in Table 3. It is anticipated that the monitoring program will continue for the next 5 to 8 years to enable the impact of sewage to be assessed for two full cycles of the sheep production system.

Loadings of nutrients, metals and organics in treatments. The biosolid products and chemical fertilizers contained a range of plant nutrients, but were also contaminated with heavy metals and pesticides (Table 2). All rates of dewatered cake contained N in excess of the 400 kg total N/ha/year currently allowable for use on crops and pastures. Of the metals present, only Cu and Zn exceeded the maximum cumulative loadings (MCL) permissible under the interim guidelines (EPANSW, 1994) at the 120 t/ha rate. However, all metals except As, Mn, and Se, exceeded the maximum annual loading (MAL) (Awad *et al.* 1989) at the 60 t/ha rate. This provided a good range of metal loadings to test the risk of accumulation in grazing sheep. Loadings of dieldrin and chlordane also exceeded the MCL (Table 2). No nutrients and contaminants in the control fertilizer or N-Viro Soil excceded the MAL or MCL (Table 2).

Water quality monitoring. Testing of water quality undertaken prior to application of sewage products showed that the levels of N and P in farm dams around the Boxers Creek and Murray's Flat catchments were sufficiently high to stimulate aquatic growth. One sample site located downstream from a piggery had the highest concentration of N (0.7 mg/L), fluoride and selenium, as well as high levels of faecal coliforms and *E.coli.* In general, metals and pesticides were present only in trace quantities throughout the catchment. However, there were some sites where elevated concentrations of Cd, Cr and Pb were recorded although the sources of these 'spikes' is not known.

The post-treatment monitoring of surface water during 1993 indicated that the water quality in the catchment surrounding the experiment was not affected by the application of sewage waste. There was no difference between nutrient and heavy metal levels in the Boxers Creek and Murrays Flat catchment and those taken from the Gundarry catchment, a nearby area with similar land uses, but to which no biosolids had been applied.

Due to drought conditions, no water has been collected from the shallow piezometer (15 m), but groundwater samples from the deep piezometer (30 m) showed that selenium (Se), molybdenum (Mo) and manganese (Mn) exceeded guidelines for stock water. However, it is unlikely that sewage waste is the contaminating source as the total N and P levels in the groundwater were low. Given the high N loading of the 120 t/ha treatment and the mobility of N in light textured soil, one would expect elevated N levels in the shallow and deep piezometers to accompany any increase in the groundwater of Contaminants derived from biosolid application.

Movement of nutrients and sewage contaminants in soil.
Nutrients: Fertilizers and sewage products both produced changes to soil chemistry measured in the spring following treatment. Lime and lime/super increased pH in the 0-10 cm layer of control plots, doubled exchangeable Ca and increased available P (Bray test) by 22 mg/kg (Table 3) from the 21 kg/ha of P applied (Table 2). N-Viro Soil increased pH by 1 unit, and increased available P to 16 mg/kg in topsoil. Due to the composition of the kiln dust used in its manufacture, N-Viro Soil was more effective than fertilizer or dewatered cake in increasing exchangeable K.

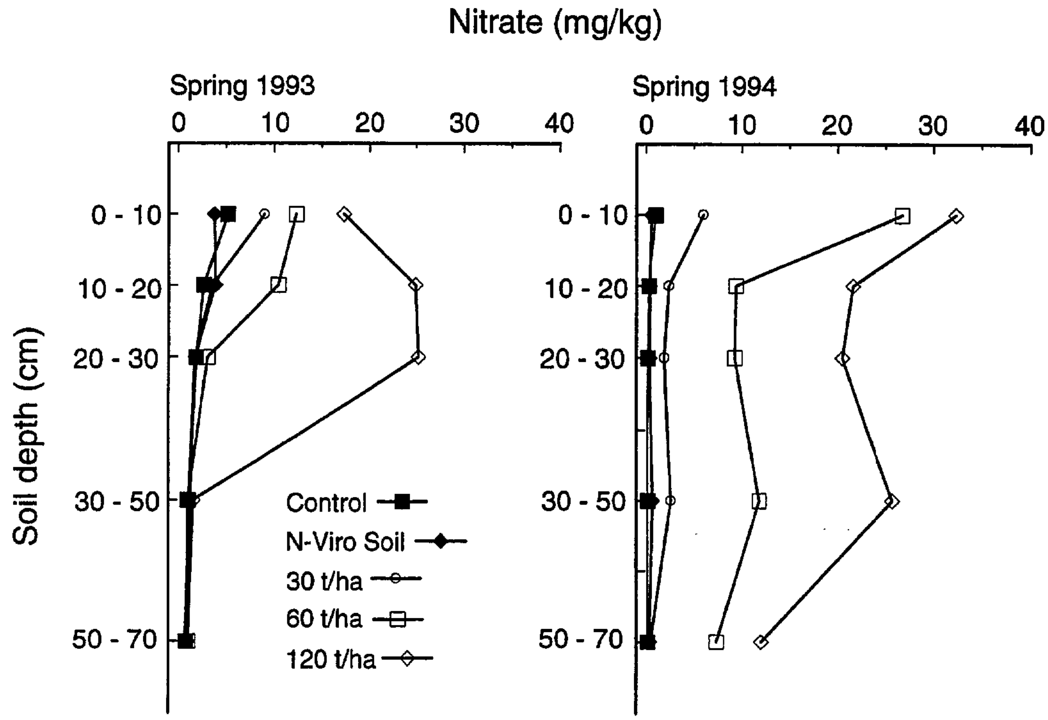

Figure 2. Effect of sewage products on downward movement of nitrate on a duplex soil (Hill A) at Goulburn.

The large amount of Ca supplied by dewatered cake (0.9 to 4 t/ha, depending on application rate of cake) was not as effective as Ca applied in lime in increasing soil pH, but it did increase exchangeable Ca (Table 3). Total C, CEC and total N were doubled by cake application at the 120 t/ha rate. The increase in available P was linearly correlated with the total P loading of each cake treatment ($R^2 = 0.96$) with the gradient suggesting that about 22% of the total P applied became available in the first spring following incorporation of dewatered cake.

Since only a small amount of the total N (<10%) and P (<2%) supplied was taken up by the pasture, downward and lateral movement of these nutrients may constitute an important source of surface and groundwater pollution over time. Mineralization and movement of N occurred reasonably quickly after treatment with sewage waste. Soil tests undertaken in spring 1993 and 1994 detected downward movement in plots treated with dewatered cake with elevated nitrate levels measured at 30-50 cm in 1993 and 50-70 cm in 1994 (Figure 2). Nitrate concentration was proportional to the rate of sewage waste applied. The 70% increase in nitrate measured in the 0-10 cm layer between the two sampling dates is consistent with the mineralisation rate of N in soils of this type. There was no significant movement of the N supplied by N-Viro Soil (Figure 2).

Downward movement of P was also detected in the loamy topsoil with available P in the 10-20 and 20-30 cm depths being significantly higher in plots treated with dewatered cake than the control and N-Viro Soil plots (Table 3). Like N, P concentration at depth was proportional to the rate of sewage waste applied.

Table 4. Concentration and movement of heavy metals in a duplex soil at Goulburn before and after application of fertilizer and lime (Control), N-Viro Soil and three rates of dewatered cake.

Treatment/	Heavy metal concentration (mg/kg)										
Soil depth (cm)	As	Cd	Cr	Cu	Mn	Mo	Hg	Ni	Pb	Se	Zn
Prior to treatment											
Depth: 0-10	10	0.6	35	5	290	<3	<2	8	33	<7	25
After treatment - Spring 1993											
Control											
Depth: 0-10	10	0.7	39	5	254	<3	<2	6	30	<7	20
10-20	13	0.7	51	7	210	<3	<2	10	42	<7	22
20-30	12	0.6	51	8	140	<3	<2	9	41	<7	21
N-Viro Soil											
Depth: 0-10	14	0.6	42	8	280	<3	<2	8	36	<7	23
10-20	15	0.7	51	5	234	<3	<2	9	39	<7	20
20-30	15	0.7	58	6	162	<3	<2	10	44	<7	22
Dewatered cake											
30 t/ha cake											
Depth: 0-10	14	0.9	49	33	251	<3	<2	15	50	<7	145
10-20	17	0.6	48	17	330	<3	<2	11	44	<7	52
20-30	13	<0.6	40	14	276	<3	<2	10	38	<7	40
60 t/ha cake											
Depth: 0-10	14	0.7	47	48	213	<3	<2	13	47	<7	121
10-20	18	0.8	42	9	201	<3	<2	9	41	<7	34
20-30	15	<0.6	42	9	120	<3	<2	9	37	<7	28
120 t/ha cake											
Depth: 0-10	12	2.2	74	178	242	<3	<2	28	69	<7	477
10-20	13	1.0	57	47	205	<3	<2	14	46	<7	135
20-30	11	0.7	51	27	143	<3	<2	11	42	<7	78

Further leaching of N and P into the B-horizon is not expected as the change in clay content (12 to 58%) provides a significant physical barrier to downward movement. However, this creates conditions conducive to lateral movement. While soil tests undertaken inside and outside exclusion plots in spring 1993 and 1994 did not detect any lateral movement, saturation of the 30-50 cm soil layer has not occurred due to dry seasonal condition during 1994. Further monitoring will determine the importance of lateral movement in this soil type.

Surface runoff collected from runoff plots in autumn 1993 detected elevated levels of N, P, K, Ca, Mg, Na, and S in plots treated with dewatered cake compared to the controls. However, once full ground cover was achieved in spring 1993, solutes and sediments in runoff were reduced to levels unlikely to pose any problems of contamination. Overall, lower runoff volumes were recorded where 60 and 120 t/ha of dewatered cake was applied. For example, a 50 mm rainfall event produced a runoff yield of 1566 L/ha on control plots, but only 219 L/ha in the 120 t/ha treatment.

Sewage contaminants: Post-treatent soil analyses detected high concentrations of Zn, Cr, Cd, Ni, Pb and Cu in the topsoil of plots treated with sewage cake compared to N-Viro Soil which caused little change in metal content relative to the control (Table 4). Downward movement of Cu and Zn was detected in cake treated plots where their concentrations at 30 cm in the 120 t/ha plots were 3 times those of the control. No lateral movement of Zn or Cu was detected in preliminary comparisons from outside and inside exclosure on the 120 t/ha plots.

Effects of sewage products on pasture parameters.

Production and composition: Pasture production has responded significantly to the nutrients provided either by conventional ferilizers or sewage products. Overall, plots treated with dewatered cake produced slightly more dry matter, especially during droughty periods when pasture benefited from the improved infiltration and water holding capacity of soil where >60 t/ha of cake was applied (Table 5). No ill effects of sewage contaminants on the yield or composition observed over the 18 months since the pasture was sown.

Botanical composition has changed significantly since sowing (Figure 3). Ryecorn, an annual cereal, accounted for >80% of total biomass during the establishment phase in 1993. By June 1994, the composition of ryecorn declined to about 30% and perennial grasses increased to 40 to 60% of standing biomass. The large proportion of ryecorn measured in November 1994 and February 1995 was dry residue which had little feed value for grazing sheep.

At present cocksfoot is the most abundant of the grasses sown accounting for 50 to 70% of the perennial component with the higher proportions present in the control and N-Viro Soil plots. Phalaris content increased with sewage application from <11% in the control to >20% in the 120 t/ha treatment. Ryegrass accounted for <5% of production in all treatments.

Legumes accounted for 27 to 35% of pasture yield in the control and N-Viro Soil, but were a minor component of pasture where dewatered cake was applied (Figure 3). White clover was the dominant legume until spring 1994, but declined over the summer due to the dry conditions. This may caused high stolon mortality that will limit the ability of white clover to respond to autumn rains.

Table 5. Effects of sewage products and fertilizer on available pasture yield (t/ha) on Hill A at Goulburn.

Treatment	Selected sampling dates					
	June 1993	Sept 1993	June 1994	Aug 1994	Nov 1994	Feb 1995
Control	3.44	9.45	3.23	3.12	2.87	2.32
N-Viro Soil	3.09	NM	3.54	4.27	3.08	3.01
30 t/ha cake	4.55	11.4	4.14	3.05	2.47	2.37
60 t/ha cake	5.55	9.1	4.48	2.76	2.91	2.56
120 t/ha cake	5.65	9.9	4.62	3.04	3.01	2.76

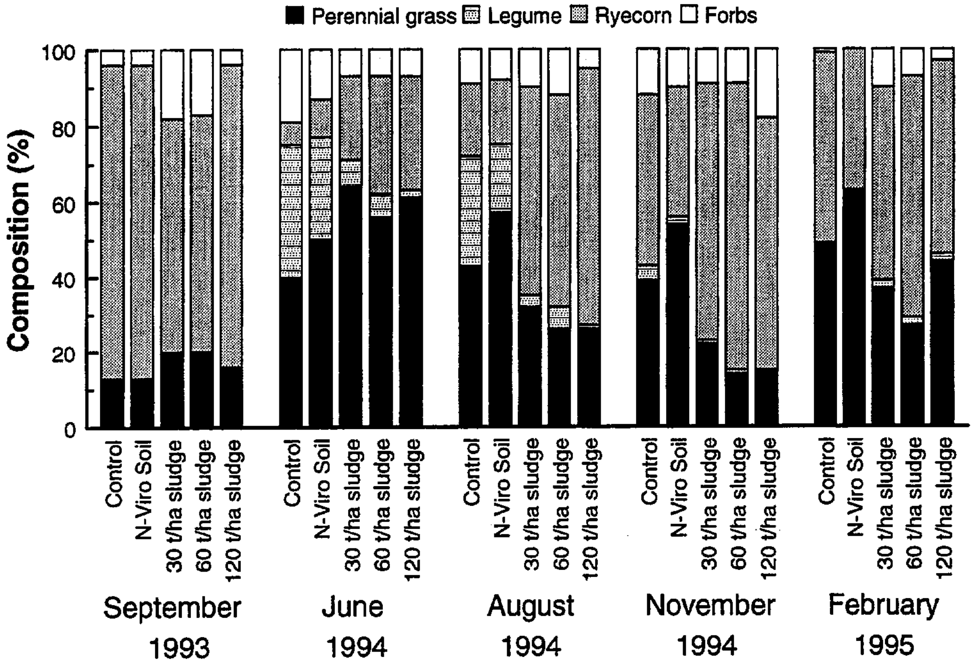

Figure 3. Effects of sewage products and fertilizers on composition of pastures grown on duplex soil at Goulburn

Broadleaf weeds such as capeweed (*Arctotheca calendula*) and volunteer tomatoes (in dewatered cake plots only) comprise 5 to 20% of pasture on offer (Figure 3). These species are regarded as weeds of sown perennial grass based pastures that are unimportant for livestock production, but they compete vigorously with desirable species.

Nutrient content of pastures: Application of dewatered cake significantly increased the concentration of N, P, S, Ca and Mg in all pasture species when compared to the control and N-Viro Soil treatments. This is illustrated by samples collected in August 1994 (Table 6). Boron was below the critical levels for ryecorn (5 mg/kg) in all treatments, but was not deficient in other pasture components. Zinc levels were high in all pasture species in the dewatered cake treated plots with the concentration increasing with application rate (Figure 4). Copper levels were only high in capeweed (Figure 4).

Content of contaminating metals: In general, the heavy metal contents of sown grasses, legumes and ryecorn were below the maximum tolerable levels (MTLs) for livestock production (Figure 4). In contrast, the 'other species' category which consisted mainly of capeweed accumulated all heavy metals at extreme levels (Figure 4), and exceeded the MTL (given in brackets) for Cd (0.5 mg/kg), Cu (15 mg/kg) and Fe (500 mg/kg). Nickel content was elevated in all pasture species (except ryecorn) grown on plots treated with dewatered cake, but the level was still below the MTL for sheep (50 mg/kg).

Effects of sewage products on sheep production.

Production parameters: There has been no significant difference in ewe condition between treatments, but ewes grazing the control and N-Viro Soil plots were at least 4 kg

Table 6. Effect of sewage waste products on the macro-nutrient content of pasture species at Goulburn (August, 1994).

Treatments	Pasture component	Nutrient (%)				
		P	S	Ca	K	Mg
Control	Grass	0.15	0.24	0.46	1.96	0.22
	Ryecorn	0.13	0.13	0.16	0.99	0.11
	Legume	0.20	0.23	1.24	1.95	0.28
	Capeweed	0.15	0.24	1.02	1.66	0.32
N-Viro	Grass	0.11	0.19	0.42	1.90	0.22
	Ryecorn	0.13	0.13	0.17	1.19	0.11
	Legume	0.18	0.22	1.17	2.30	0.28
	Capeweed	0.15	0.23	1.06	2.55	0.32
30 t/ha cake	Grass	0.18	0.34	0.42	2.12	0.26
	Ryecorn	0.17	0.18	0.23	1.45	0.17
	Legume	0.25	0.31	1.00	2.30	0.26
	Capeweed	0.33	0.35	1.09	2.94	0.39
60 t/ha	Grass	0.20	0.37	0.51	2.11	0.26
	Ryecorn	0.20	0.19	0.28	1.41	0.17
	Legume	0.29	0.32	1.10	2.18	0.27
	Capeweed	0.31	0.34	0.92	2.46	0.41
120 t/ha	Grass	0.25	0.42	0.58	2.09	0.31
	Ryecorn	0.22	0.23	0.34	1.46	0.20
	Legume	0.32	0.37	1.13	2.14	0.26
	Capeweed	0.35	0.40	1.08	2.56	0.41

heavier than those grazing dewatered cake plots in all measurements since lambing in June 1994. This difference reflected the better quality pasture available in control and N-Viro Soil treatments where the legume content was higher and there was less carry-over of ryecorn residue.

The effects of a better balanced diet was also reflected in wool production with ewes grazing the control and N-Viro Soil areas producing 0.5 kg more wool than dewatered cake treatments over the 12 month period (Table 7). There was also a trend toward better wool quality in the control and N-Viro Soil plots where >87% of fleeces were classed in the superior line (AAAA) whereas a high proportion of fleeces (>27%) from ewesgrazing 60 and 120 t/ha cake treatments were down graded due to tenderness or short staple length (Table 7). An assessment of the economic impact of these differences in wool yield and quality will be made when samples are analysed for fibre diameter and tenderness.

The 95% joining rate achieved from the January joining produced a 75% marking rate of lambs measured in September, 1994. At weaning in October, lambs from the control and N-Viro Soil treatments weighed 27.4 kg compared with only 22.2 kg for plots treated with dewatered cake. This superiority in lamb growth rate on the control and N-Viro Soil plots was continued in the post-weaning performance of lambs measured in November, 1994.

Differences in sward structure, pasture composition and pasture quality account for the

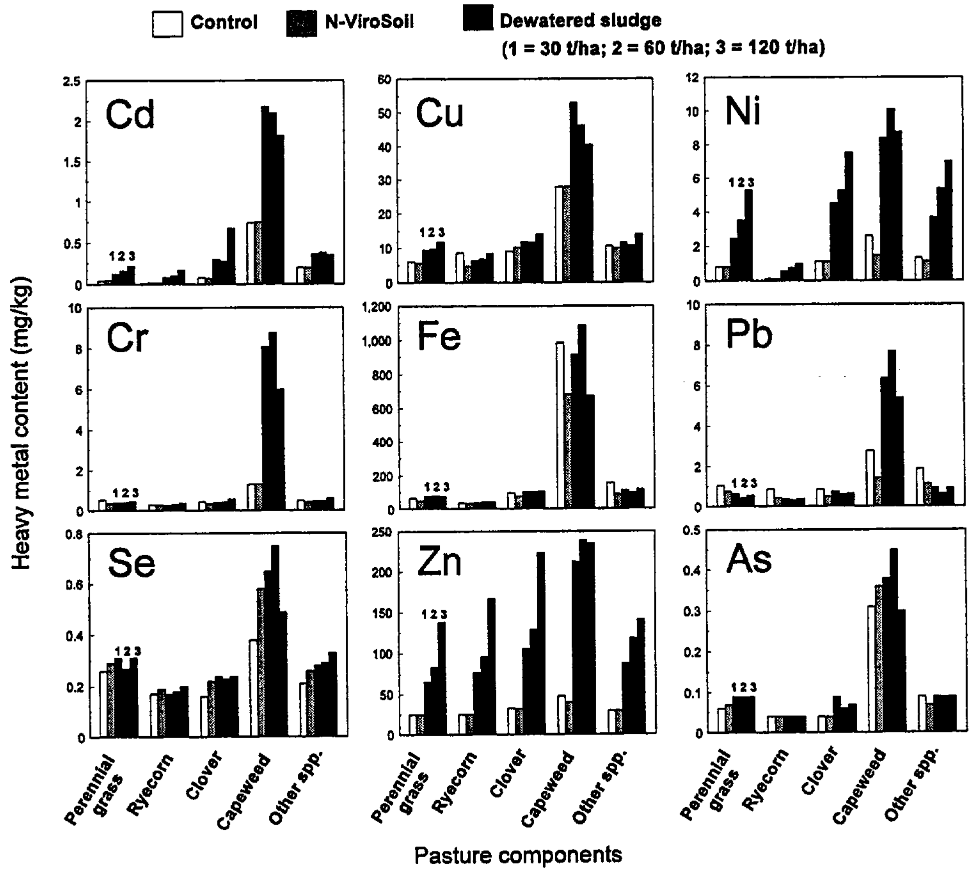

Figure 4. Effect of application of sewage products on the heavy metal content of pasture components measured in August, 1994, at Goulburn.

differences in lamb and weaner performance. The lower pasture height and higher legume content in the control and N-Viro Soil plots enabled ewes to produce more milk, and lambs and weaners to select a better quality diet than animals grazing sludge treated plots, particularly in the June to August period.

During dry periods when pasture production could not meet the requirements of the grazing livestock, supplementary feed at a maintenance level was supplied at the same rate to all treatments. Following local farmer practice, sheep nuts were fed directly onto the ground which exposed animals to possible ingestion of soil, but soil intake was not measured.

Heavy metals and organochlorines in sheep: Fifteen ewes were slaughtered prior to exposure to sewage products to establish background levels of heavy metals and organochlorines in tissues and fat. All concentrations of As, Cd, Cu, Hg, Pb, Se, Ni and Zn in liver, kidney and muscle samples were well below the maximum permissible concentracions (MPCs), and organochlorines in fat samples were below detection.

Table 7. Wool production from Merino ewes grazing pastures treated with sewage products at Goulburn.

Treatments	Wool production parameters				
	Wool cut (kg/ewe)	Fleece quality (% in each category)			Skirting ratio (Fleece:skirting)
		AAAA	FLC	AA	
Control	3.77	87[1]	10[1]	3[1]	6.1 : 1
N-Viro Soil	3.65	90	7	3	6.4 : 1
30 t/ha cake	3.22	90	3	7	6.4 : 1
60 t/ha cake	3.19	63	20	17	6.2 : 1
120 t/ha cake	3.20	73	27	0	6.0 : 1

[1]AAAA denotes superior fleeces, FLC fleeces showing tenderness and AA shorter stapled fleeces.

Analyses of milk samples collected in July, 1994, showed no consistent differences in heavy metals attributable to application of sewage waste. All organic contaminants were below detection level. Although milk samples were free from contamination, assay of tissues from lambs slaughtered at weaning (October, 1994) showed a 3 to 7 fold increase in Ni concentration in liver, kidney and muscle samples of lambs raised on dewatered cake treatments compared to the control and N-Viro Soil where the Ni level was <0.05 mg/kg. The selection of fresh, green pasture which was high in Ni (Figure 4) was the most likely source of this contamination. Analyses of tissue samples collected in December, 1994, of ewes with >12 months exposure to sewage contaminants are still to be analysed.

No change in heavy metal levels were detected by analysis of wool samples taken at shearing in October, 1994. However, there were detectable levels of total DDT and total PCBs, but since these were similar in all treatments, sheep management practices prior to exposure to sewage products is the most likely source of these organics.

Discussion and Conclusion

Although sewage waste and farm-yard manures have been used for some time to produce pastures and forage crops for consumption by livestock, the effects of contaminants on the quality of products derived from livestock grazing sewage treated land need further investigation. The importance of such research was highlighted by the large number of missing values for many of the important metal contaminants summarized by the USEPA (1989a) for risk pathways associated with the transfer of sewage pollutants from sewage to livestock and thence to humans.

There is a particular need to validate key data in Australia where environmental conditions (especially soils with unusual characteristics) and livestock production systems differ from those in Europe and North America. The very low maximum residue limits on exported animal products is another reason that such research is required in Australia.

Areas of interest for use of sewage products in pasture ecosystems in NSW include: (1) determination of appropriate one-time application rates for pastures based on nutrient loadings; and (2) the absorption in topsoil, loss through surface erosion, movement into groundwater, uptake in plants and ingestion by livestock of contaminating metals and organics when sewage waste is applied to acidic, infertile, low cation exchange soils.

This project, which is planned to continue until 2002, was established to quantify the benefits and risks involved in applying sewage waste to perennial grass pastures grown on poor, acid soil and grazed year-round by breeding Merino sheep. In general, since uptake of potentially toxic pollutants of sewage waste by livestock occurs through ingestion of plants, soil and water, the combined effects of these three pathways needs to be assessed. However, in this study the impact of uptake of pollutants via water was eliminated by providing wholesome stock water through an articulation system from an uncontaminated source upstream from the project site. Such systems are common place in commercial sheep enterprises in the Southern Tablelands region.

Transfer of biosolid pollutants from soil either directly to grazing livestock or via plants is a function of factors such as: the contaminants present in the sewage; the cumulative biosolid application rate; characteristics of the sewage product; the application method; soil pH; climate; and plant species and cultivars grown (Chaney *et al.* 1987b). The results presented, though equivocal at this point, highlight the changes in water, soil and plant components that result when sewage products are applied at high rates to soils more acidic than those used elsewhere in the world, and the risks they present to contaminate grazing sheep.

Movement of contaminating metals within the pasture ecosystem was consistent with their known bioavailability: Zn > Cu > Ni > Pb. Loadings of these metals covered a sufficient range to provide a good test of their behavior under field conditions representative of 70% of the target area for use of sewage waste. Based on the MCLs specified in the Interim Code of Practice (1994), the concentration of Zn and Cu will determine the use rate in pasture ecosystems for the majority of the sewage waste produced in NSW. However, as is the case with all contaminants, the concentration of Zn and Cu varies widely according to industrial processing on a day-to-day basis. In this study, concentrations in dewatered cake ranged from 2216 to 4538 mg/kg for Zn and 1129 to 1552 mg/kg for Cu. This means that if both metals were present at the low range, about 90 t/ha of dewatered cake could be applied to remain within the MCL, but if both were present at the high range, only about 40 t/ha could be applied to keep the MCL for Zn below 200 kg/ha.

Cadmium which is a sewage contaminant of concern in Australia because of the low maximum residue set for export products (Langlands *et al.* 1988) was present at a much lower concentration (8-15 mg/kg) in the sewage waste applied at Goulburn than the 32 mg/kg reported as the average for the Malabar STP. However, even at this higher Cd concentration, the rate of dewatered cake application would still be determined by Zn and/or Cu content. Only for sewage wastes from mining and highly industrialized areas such as Mount Isa and Port Kembla would Cd content determine application rate (de Vries 1983). The movement of metal contaminants reported here confirm previous studies (*eg.* Lund *et* *al.* 1976; Lawrie *et al.* 1991) where Zn, Cu and Cd were detected down to a depth of >30 cm in soils with pH <5.5. Soil pH, CEC and texture are the soil properties influencing the movement of heavy metals. Rapid leaching of metals in the light textured, acidic Goulburn soils highlights the importance of selecting the right soil types for sewage waste application. The range of soil types under study at Goulburn will enable the most appropriate soil type to be identified.

The movement of N and P in our study is also consistent with the physical characteristics of the Goulburn soils which results in low nutrient retention in the A horizon. The dense subsoil of low porosity (58% clay) may provide a barrier to pollution of ground water, but lateral movement may occur given favourable moisture conditions. No lateral movement has been measured to date, but this may reflect the very dry conditions experienced in 1993/4. Below average rainfall also limited the response of pasture production to the N and P inputs derived from sewage waste. High rainfall variability in the Southern Tablelands region highlights the need to continue the monitoring of this site to assess the long-term bioavailability of sewage derived metals and the impact of nutrients on the environment.

The addition of 7.5 t/ha of N-Viro Soil (alkali-stabilized sewage waste) increased soil pH sufficiently to correct acidity. However, contrary to results reported by Willett *et al.* (1984), N-Viro Soil was not superior to lime in promoting downward movement of Ca and neutralization of sub-soil acidity. Nor did sewage derived Ca produce higher Ca in plant tops relative to the lime-treated control as reported by Willett *et al.* (1984).

Due to the dry conditions encountered since July 1993, the effect of the large loadings of all macro-nutrients except K has been smaller than expected. However, application of dewatered cake significantly increased the content of all essential nurtients in sown grasses and legumes. Although the total P loading was well above the requirements of pasture species and could possibly lead to P toxicity (Kirkham 1982), the concentrations measured in tops of various pasture species were below toxic levels (Rueter and Robinson 1986).

In many studies, the benefical effects of nutrients is offset by the accumulation of toxic elements in plant tops (Sterritt and Lester 1980). However, for many pasture species there is little information on the accumulation of heavy metals and their effect on growth and reproduction. Our results showed that the sown perennial grasses accumulated Zn and Ni, and that Cd, Zn and Ni levels were elevated in clover tops. However, the concentrations did not cause any observed phytotoxicity and did not pose a risk to sheep as they were well below the MTLs. These were the same metals reported by Baxter *et al.* (1983) to accumulate in native pasture (mainly Buffalograss - *Buchloe dactyloides*) grown on a sewage disposal site in Denver, and by Lawrie *et al.* (1991) for sown pastures treated with dewatered cake in NSW.

In contrast, capeweed accumulated all metals at high rates and exceeded the MTL for Cd, Cu and Fe. Uptake rates were often four times those measured for legumes and grasses. In Western Australia, capeweed has been implicated in the Cd accumulation in soft tissues of sheep grazing pastures previously fertilized with superphosphate containing a high Cd content. Based on our preliminary findings, we recommend that management should aim to keep such broadleaf forbs at minimal levels in pastures grown on soils treated with sewage waste.

Provided phytotoxicty does not occur first, excessive animal residues from As, Hg, Fe, Ni, Co, Mo, Zn and a range of organic compounds can occur. The two routes for the transfer of these contaminants to sheep in this study were by ingestion of soil or through eating plants with elevated levels of sewage pollutants. It is not possible to differentiate the importance of these two pathways. However, Fleming (1986) calculated that about 14% of dry matter intake of sheep maintained in all year grazing systems is ingested soil. Undoubtedly, this pathway may have been an important source of contamination before full ground cover was achieve, particularly in the 120 t/ha treatment where some biosolid remained on the surface. However, since the first summer (1993/4), ground cover has exceeded 80% due to ryecorn stubble and perennial grasses, and reduced the potential for soil ingestion even when sheep were fed supplement directly on the ground.

This means that contaminants accumulated in plant tops has constituted the main hazard for sheep in this study. However, after one year of exposure, production and health of ewes and lamb has not been affected by grazing pastures treated with sewage products. The common sheep production parameters of liveweight, wool production and marking percentage suggest that the absence of legume from pastures in the dewatered cake treatment exerted a larger production penalty than contaminants derived from sewage waste. However, accumulation of Cu and Zn are likely to occur over time. For example, Ross and Short (1990) reported that it took 3 years for metal residues to reach significant levels in sheep grazing similar pastures treated with Sydney sewage waste. Continued monitoring of the current program is needed to elucidate possible risks associated with metal accumulation in the pasture ecosystem.

Conclusion

Based on our findings, we suggest that 60 t/ha may be a feasible one-time application rate of Malabar STP dewatered cake that maximizes benefits from organic matter and nutrients on soils and pasture, and minimizes environmental pollution and spoilage of livestock products. However, monitoring of this project will continue for at least another 5 years when a full economic and environmental assessment of the effects of sewage waste application on a pasture ecosystem will be possible.

Acknowledgments

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References

Addiscott, T. (1988) Farmers, fertilizers and the nitrate flood. *New Scientist*, **8:** 50-54.

Awad, A.S., Ross, A.D. and Lawrie, R.A. (1989) Guidelines for the use of sewage sludge on agricultural land. 1st Edition. NSW Agriculture & Fisheries, Sydney.

Baxter, J.C., Aguilar, M. and Brown K. (1983) Heavy metals and persistent organics at a sludge disposal site. *Journal of Environmental Quality*, **12:** 311-316.

Chaney, R.L., Bruins, R.J.F., Baker, D.E., Korcak, R.F., Smith, J.E. and Ryan, J.A. (1987a) Effects of long-term sludge application on accumulation of trace elements by crops. In: A.L.Page, T.J. Logan, and J.A. Ryan (Editors), "Land Application of Sludge", pp 53-66, Lewis Publishers, Chelsea.

Chaney, R.L., Bruins, R.J.F., Baker, D.E., Korcah, R.F., Smith, J.E. and Cole, D. (1987b) Transfer of sludge applied trace elements to the food chain. In: A.L.Page, T.J. Logan, and J.A. Ryan (Editors), "Land Application of Sludge", pp 67-99, Lewis Publishers, Chelsea.

Copeland, B. (1990) Domestic Catchment Sampling Programme Report. Trade Waste Unit, Water Board, Sydney.

Curran, P. (1991) Scope of eutrophication problems. In: "Proceedings of Chemical Summit", Sydney

Finney, I., and Rawlinson, L. (1990) Sludge and wastewater re-use. In: D. Elmes and J. Mallen-Cooper (Editors), "The Agricultural and Horticultural Use of Sewage Products", Australian Institute of Agricultural Science, Ocassional Paper No. 55.

Fleming, G.A. (1986) Soil ingestion by grazing animals: a factor in sludge-treated grassland. In: R.D.Davis, H.Hueni and P.L'Hermite (Editors), "Factors Influencing Sludge Utilisation in Europe", pp. 43-50, Elsevier, London.

Kirkham, M.B. (1982) Agricultural use of phosphorus in sewage sludge. *Advances in Agronomy*, **35:** 129-163.

Langlands, J.P., Donald, G.E., and Bowles, J.E. (1988) Cadmium concentrations in liver, kidney and muscle in Australian sheep and cattle. *Australian Journal of Experimental Agriculture*, **28:** 291-297.

Lawrie, R.A., Awad, A.S. abd Short. C. (1991) Final Report: Utilisation of Sewage Sludge on Agricultural Land. Project P86/03, NSW Agriculture, Orange.

Lindsay, D.G. (1983) Effects arising from the presence of persistent organic composunds in sludge. In: R.D.Davis, G.Hucker, and P. L'Hermite (Editors), "Environmental Effects of Organic Contaminants in Sewage Sludge", D.Reidel Publishing Co., Dordrecht, pp. 19-26.

Logan, T.J., and Page, A.L. (1989) Peer review committee analysis of the proposed Rule 40 CFR parts 257 and 503; standards for the disposal of sewage sludge. Co-operative State Research Service Technical Committee W-170, University of California, Riverside.

Lund, L.J., Page, A.L. and Nelson, C.O. (1976) Movement of heavy metals below sewage disposal ponds. *Journal of Environmental Quality*, **5:** 331-335.

Michalk, D.L., Curtis, I.H., Joshua, W.D., Bamforth, I.S., Salt, M. and Simpson, P.C. (in press) Monitoring the effects of sewage sludge on a pasture ecosystem grazed by sheep in south-eastern Australia - Preliminary results. In: "Watershed Management: Planning for the 21st Century", American Society of Civil Engineers.

Ross, A.D. and Shrot, C. (1990) "Utilisation of Sewage Sludge on Agricultural Land - Animal Trial", Final Report, Project P86/03, NSW Agriculture.

Ross, A.D., Awad, A.S., and Lawrie, R.A. (1989) Factors affecting the application of sewage sludge to agricultural land in New South Wales. Proceedings AWWH Regional Conference, Bowral, October, 1989.

Ross, A.D., Lawrie, R.A., Whatmuff, M.S., Keneally, J.P. and Awad, A.S. (1991) "Guidelines for the Use of Sewage Sludge on Agricultural Land", 3rd Edition, NSW Agriculture, Orange.

Rueter, D.J. and Robinson, J.B. (Editors) (1986) Plant Analysis: An Interpretation Manual, Inkata Press, Melbourne.

Sidle, R.C., and Kardos, L.T. (1977) Transport of heavy metals in a sludge-treated forested area. *Journal of Environmental Quality*, **5:** 97-103.

Seawright, A.A. (1989) "Chemical and Plant Poisons" Animal Health in Australian Volume 2, 2nd Edition, Department of Primary Industry and Energy, Canberra.

Seiler, H.G., Sigel, H. and Sigel, A. (1988). "Handbook on Toxicity of Inorganic Compounds", Marcel Dekker Inc., New York.

Sterritt, R.M., and Lester, J.N. (1980) The value of sewage sludge to agriculture and effects of the agricultural use of sludge contaminated with toxic elements: A review. *Science and the Total Environment*, **16:** 55-90.

Taylor, R.W., Duseja, D.R., and Thangudu, P.R. (1982) Sewage sludge effects on soil: Heavy metal accumulation and movement. *Journal of Environmental Science and Health Association*, **17:** 427-441.

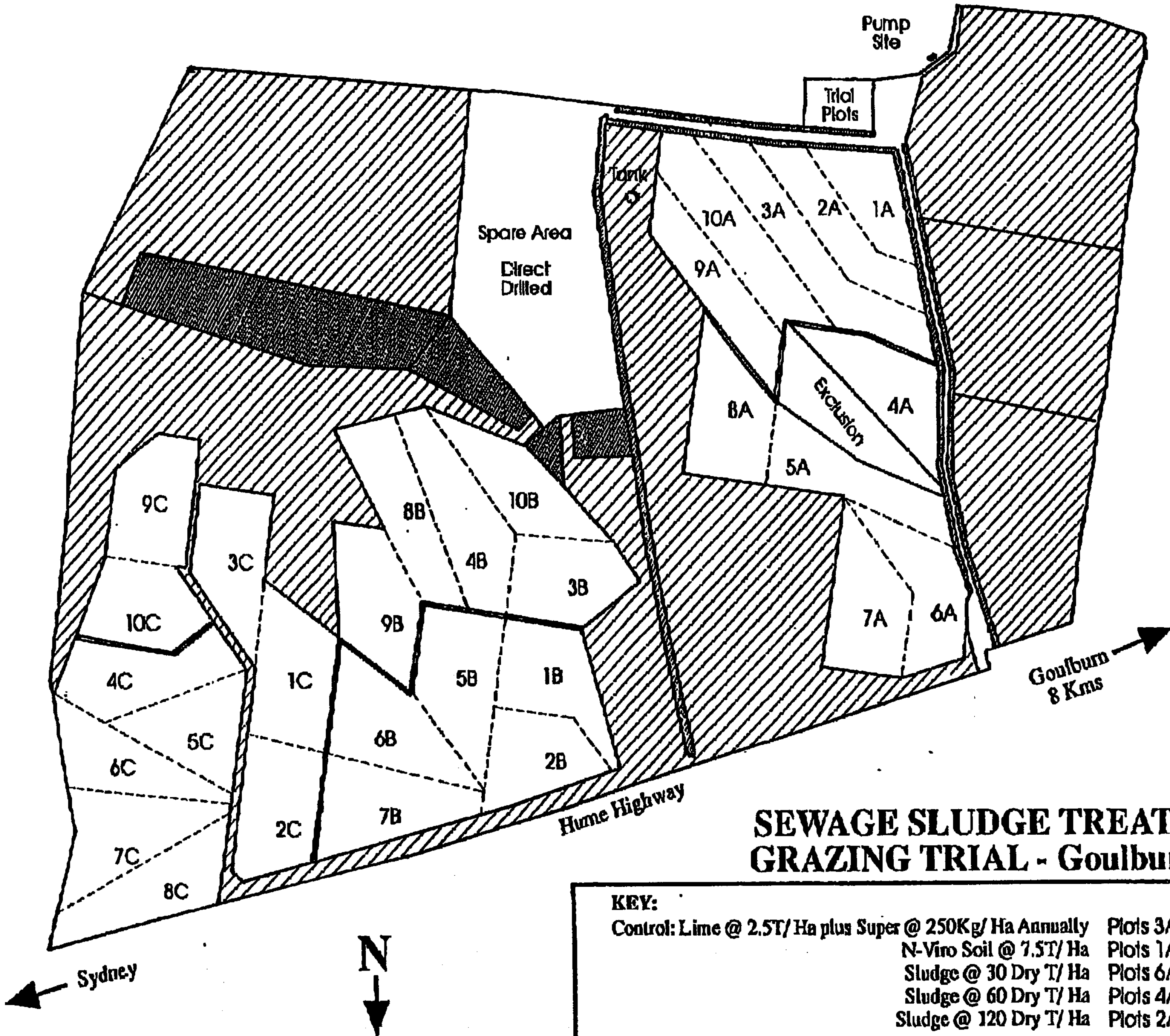
Tothill, J.C., Hargreaves, J.N.G., Jones, R.M. and McDonald, C.K. (1992) BOTANAL - A comprehensive sampling and computing procedure for estimating pasture yield and composition. 1. Field sampling. Tropical Agronomy Technical Memorandum, CSIRO Division of Tropical Crops and Pastures, No. 78, pp 24.

U.S. Environmental Protection Agency (1989a) Standards for the disposal of sewage sludge; proposed rule. 40 CFR parts 257 and 503. *Federal Register, 54:* 5746-5901.

de Vries, M.P.C. (1983) Investigations of twenty Australian sewage sludges - their evaluation by chemical analysis. *Fertilizer Research*, **4:** 75-87.

Willett, I.R., Jakobsen, P., Cunningham, R.B. and Gunthorpe, J.R. (1984) Effect of lime-treated sewage sludge on soil properties and plant growth. 1. Report for 1981 to 1982. Division of Soils, Divisional report No. 67, CSIRO, Canberra

Williams, S.H., and David, D.J. (1976) The accumulation in soil of cadmium residues from phosphate fertilizers and their effect on the cadmium content of plants. *Soil Science*, **121:** 86-93.



SEWAGE SLUDGE TREATED PASTURE GRAZING TRIAL - Goulburn, Australia

KEY:

Control: Lime @ 2.5T/ Ha plus Super @ 250Kg/ Ha Annually	Plots 3A, 7A, 2B, 4B, 8C, 10C
N-Viro Soil @ 7.5T/ Ha	Plots 1A, 8A, 1B, 9B, 1C, 6C
Sludge @ 30 Dry T/ Ha	Plots 6A, 10A, 5B, 10B, 2C, 4C
Sludge @ 60 Dry T/ Ha	Plots 4A, 9A, 7B, 8B, 5C, 9C
Sludge @ 120 Dry T/ Ha	Plots 2A, 5A, 3B, 6B, 3C, 7C

Spare Area

Trees (Direct Sown & Tube Stock)

Sawmill Waste Utilization for Reclaiming Bentonite Mine Lands[1]

G.E. Schuman and E.M. Taylor, Jr.[2]

Abstract

Large scale bentonite mining began in the Northern Great Plains in the 1930's; however, few areas were reclaimed until enactment of reclamation laws in the 1970's. These abandoned bentonite mined lands are extremely difficult to reclaim because of adverse chemical and physical properties of the spoil material, the limited inherent topsoil resource, and the arid/semiarid climate of the region. Wood residue and gypsum were found to be effective in improving spoil physical and chemical properties. Wood residue improved water infiltration and leaching of the salts from the root zone. Gypsum ameliorated the sodicity by supplying a calcium source that could replace the sodium on the exchange complex of the spoil system. These improvements in the spoil quality enabled vegetation establishment and the long-term productivity of these lands. Technology was developed that led to the reclamation of about 6000 hectares of abandoned bentonite spoils in Wyoming alone.

Introduction

Lands disturbed by bentonite mining in the Northern Great Plains are difficult to reclaim because of the adverse chemical and physical properties of the spoil material, the limited inherent topsoil, the arid/semiarid climate of the area, and to some extent, the mining methods utilized. Although large scale bentonite mining began in the region in the 1930's, few areas were reclaimed until after enactment of reclamation laws in the early 1970's. Ninety percent of the United State's supply of bentonite is mined in the Northern Great Plains states of Montana, South Dakota, and Wyoming (Ampian, 1980).

Natural revegetation and man assisted reclamation plantings of non-topsoiled, non-amended bentonite spoils have resulted in no to poor plant establishment (Dollhopf and Bauman, 1981; Sieg et al., 1983), indicating that spoil modification is essential for successful revegetation of these sodic, saline, high clay content spoils. Reclamation of abandoned bentonite spoils has met with limited success (Hemmer, et al., 1977; Bjugstad et al., 1981; Dollhopf and Bauman, 1981) unless organic amendments were utilized. Schuman and Sedbrook (1984) reported on the effectiveness of sawmill wastes (bark, chips, and sawdust) in improving the physical

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characteristics of the spoil, greatly enhancing water infiltration, and promoting vegetation establishment. Dollhopf and Bauman (1981) evaluated the use of woodchips and manure in conjunction with straw mulch. They obtained good first year plant densities with woodchips; however, the manure only resulted in about 23% of the density obtained with the woodchips. They also evaluated the effectiveness of gypsum, calcium chloride, and sulfuric acid, but these amendments resulted in poor initial seedling establishment. They concluded that the poor establishment observed was because inorganic chemical amendments did not alter the physical conditions of the spoil rapidly enough to enable successful seedling emergence, but that over time these amendments may prove beneficial. These findings were not unexpected since inorganic amendments used alone may require considerable time (perhaps years) to improve spoil physical characteristics through the replacement and leaching of sodium under natural precipitation conditions.

The research cited above demonstrates the importance of a bentonite spoil amendment program to achieve rapid and successful reclamation where adequate topsoil resources are limited. To be successful, the high clay spoil amendment program must immediately improve water infiltration for vegetation establishment and establish leaching to ensure the effectiveness of any inorganic amendment used to ameliorate sodicity problems. If both of these requirements are not met, long-term reclamation may not be achieved or may require considerable time.

The purpose of this paper is to discuss the role of organic amendments for achieving immediate improvement in water relationships of the spoil, and inorganic amendments for long-term amelioration of the sodic properties of the spoil.

Evaluation of Organic Amendments

Study Design

A field study was initiated in 1981 on about 2 ha of leveled abandoned bentonite spoils near Upton in northeastern Wyoming (Smith, 1984). These spoils were typical of the abandoned spoils associated with the Mowry shale formation (Table 1). Soils overlying bentonite deposits are formed from very fine clay and shale particles. The soils have low inherent fertility. The area is characterized by broad, flat sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) and grass covered valleys separated by low Ponderosa pine (*Pinus ponderosa*) covered ridges. The long-term average annual precipitation of this area is 370 mm, occurring mostly as spring and summer rainfall. Approximately 60% of the precipitation occurs during April through July. The average frost free period is 165 days.

The study involved the evaluation of the feasibility and effectiveness of Ponderosa pine sawmill wood residue (woodchips, bark, and sawdust) as an amendment to improve the physical characteristics of the spoil and thereby aid revegetation. The study was a split-plot design with a split-block within the wood residue treatments (Figure 1). Four wood residue rates (0, 45, 90, and 135 Mg/ha), four nitrogen fertilizer rates (0, 2.5, 5.0, and 7.5 kg N/Mg wood residue) and two seed mixtures (native and introduced species) were established. Since nitrogen application rate was based on wood residue amendment rate, nitrogen was applied at the rate of 0, 112, 224, and 336 kg N/ha on the 0 wood

Table 1. Physical and chemical characteristics of pretreatment bentonite spoil samples, Upton, WY, 1981. (Smith, 1984).

Parameter	Mean and Standard Error*
Particle-size-separates (%)	
Sand	10.8 ± 0.8
Silt	29.6 ± 0.8
Clay	59.6 ± 1.1
Saturation Percentage (%)	80.9 ± 1.7
NO_3-N (mg/kg)	7.7 ± 0.4
NH_4-N (mg/kg)	2.6 ± 0.1
Total Kjeldahl-N (mg/kg)	751.1 ± 5.8
P (mg/kg)	8.1 ± 0.3
C (mg/kg)	10.0 ± 1.0
pH	6.8 ± 0.1
Electrical Conductivity (dS/m)	13.4 ± 1.1
Water-Soluble Cations (mg/kg)	
Ca	187.9 ± 9.2
Mg	73.6 ± 4.2
Na	3613.7 ± 101.4
K	32.0 ± 0.8
Sodium-adsorption-ratio	63.1 ± 1.2

*Particle-size-separates obtained from 5 observations. All other parameters are a mean of 144 samples.

residue treatment. These nitrogen rates were equal to those applied on the 45 Mg/ha wood treatment. Phosphorus fertilizer was applied uniformly at the rate of 90 kg P/ha. The spoil material was ripped to alleviate compaction and the wood residue and fertilizer amendment applied and incorporated into the surface 30 cm of spoil using a large disk. The plots were drill seeded at the rate of 682 PLS (pure live seeds)/m2 of the grass mixtures and 32 PLS/m^2 of Gardner saltbush (*Atriplex gardneri*). Smith et al. (1985) provide complete details on the grass species and sampling methodology.

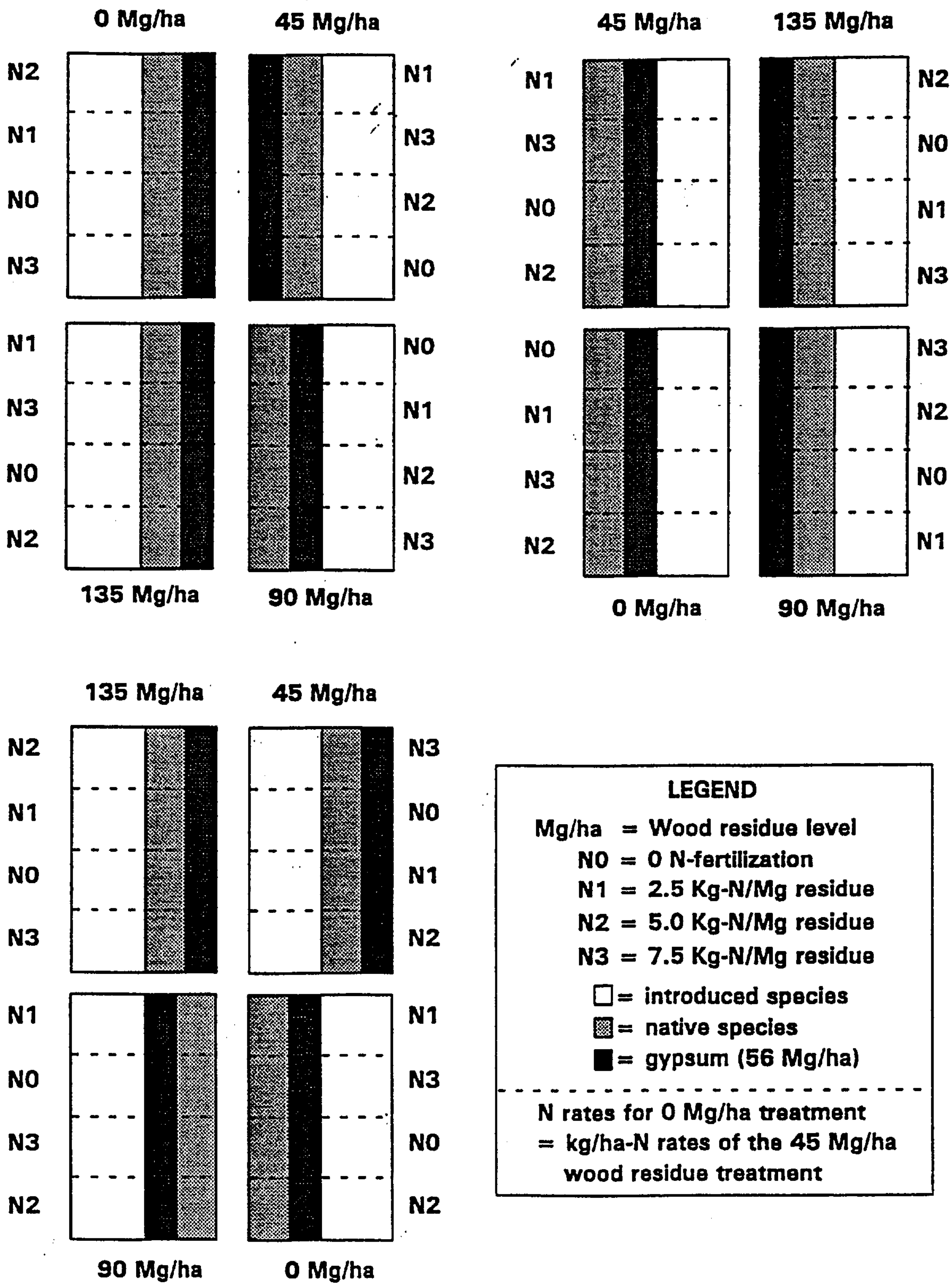

Figure 1. Field plot diagram of bentonite amendment project, Upton, WY.

Spoil Responses

Edaphic responses to wood residue amendment were exhibited in three ways: increased water infiltration and storage, decreased salinity due to leaching, and increased sodicity. The addition of 90 and 135 Mg/ha of wood residue significantly increased the spoil-water content in the surface 30 cm of spoil (Table 2).

Table 2. Spoil-water content in 0, 45, 90, and 135 Mg/ha wood residue amended spoils at three spoil depths, Upton, WY, June 1982.

Wood Residue (Mg/ha)	Spoil Depth (cm)		
	0-15	15-30	30-45
	Water Content (g/kg)		
0	153a	139a	140a
45	167a	137a	127a
90	239b	170ab	129a
135	249c	183b	140a

Values within a spoil depth with the same letter are not significantly different, $P \leq 0.05$.

The decreased effect of wood residue on spoil-water at the deeper spoil depth is related to the fact that amendment incorporation only occurred to 30 cm; therefore, percolation below that depth was limited in the early phase of the study. The increased water movement into the spoil resulted in significant leaching of soluble salts from the surface 15 cm of the spoil from 1981 to 1984 (Figure 2). In 1985, a severe drought resulted in a significant upward migration of salts in the surface 15 cm as a result of upward water movement in response to high evapotranspiration demands. Although upward salt migration occurred during the drought year, the electrical conductivity (EC) of the 0-15 cm spoil depth did not exceed the 1982 levels observed before leaching occurred.

Leaching of soluble salts is desirable and necessary to encourage soil development and long-term stability of these lands. However, the leaching process resulted in an increase in spoil sodium-absorption-ratio (SAR) in the residue amended plots over time (Figure 3). The pool of soluble sodium (91% of soluble cations) was so large that as leaching occurred, the relative proportion of sodium in the system compared to calcium and magnesium became greater, thereby increasing the SAR. This increase in SAR can have significant negative long-term effects on plant nutrition, spoil physical qualities, and subsequent maintenance of the vegetation community. Increased SAR indicates that chemical amendments are necessary in addition to the wood residue, to ensure long-term reclamation and revegetation success.

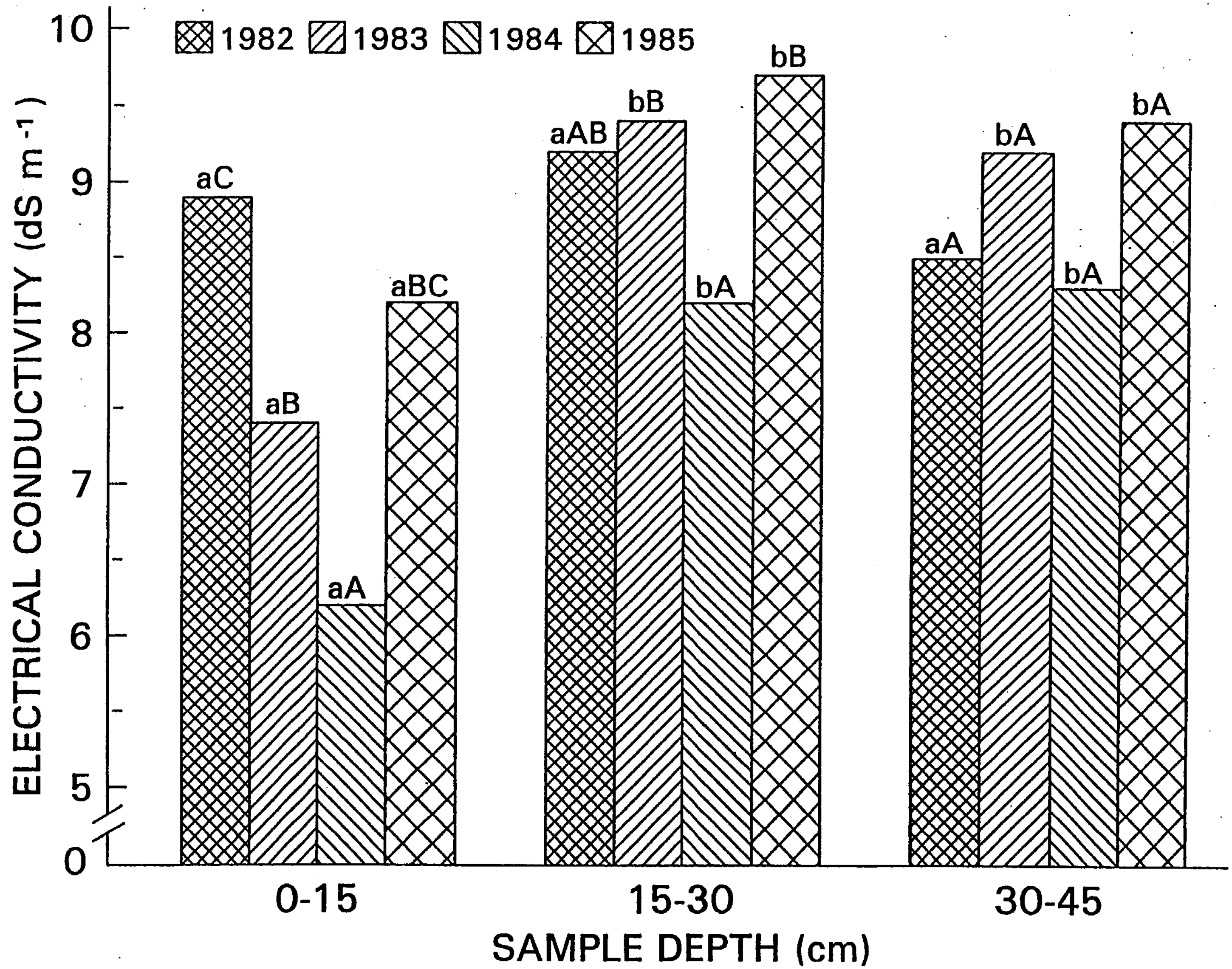

Figure 2. Mean electrical conductivity averaged across wood residue and N fertilizer treatments, 1982-1985, at three spoil depths. Means within a year among spoil depths with the same letters (lower case) or among years within a spoil depth (upper case) are not significantly different, P≤0.05. (Belden 1987).

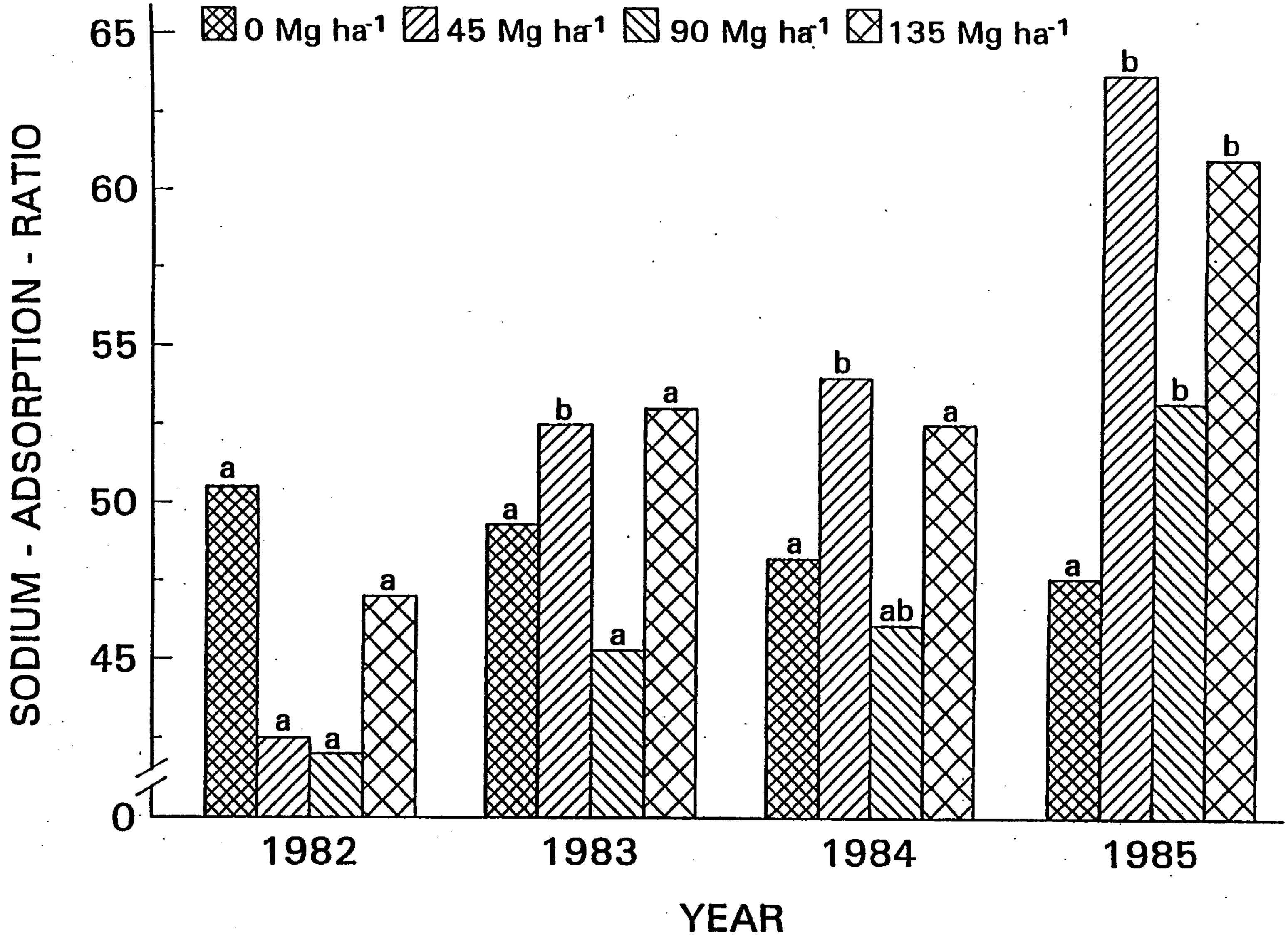

Figure 3. Mean sodium-adsorption-ratio averaged across N fertilizer treatments and spoil depths, at four wood residue levels, 1982-1985. Means among years with the same lower case letter are not significantly different, P≤0.05. (Belden 1987) success.

Plant Responses

Early seedling density exhibited little species mixture response with slightly greater seedling densities for the native mixture at the 0 and 45 Mg/ha wood residue amendment and no differences between the two species mixtures at the 90 and 135 Mg/ha treatments. Average seedling density was significantly improved by wood residue amendments because of its positive effect on spoil-water content, crusting, and bulk density. Seedling density was 14, 41, 60, and 70 plants/m^2, for the 0, 45, 90, and 135 Mg/ha wood residue treatments, respectively. Perennial grass production increased as wood residue rates increased, with maximum production occurring at the 135 Mg/ha wood residue level (Table 3). Production in 1985 was extremely

Table 3. Perennial grass production (averaged across species mixtures and nitrogen fertilizer treatments) in response to wood residue rate, Upton, WY, 1983-1986. (Means among wood residue rates within a year followed by the same letter are not significantly different, $P \leq 0.05$. (Smith, 1984; Belden, 1987; and Schuman, unpublished data).

	Wood Residue Rate (Mg/ha)			
Year	0	45	90	135
1983	59a	669b	1748c	2550d
1984	80a	361a	1220b	1956c
1985	10a	55a	148a	448a
1986	15a	116a	324a	886b

low compared to previous years because of the extreme drought that year. Precipitation from August 1984 through June 1985 (17 cm) was approximately 60% of normal for the period. However, normal precipitation occurred in late summer 1985 and throughout the 1986 growing season, resulting in the improved production observed in 1986. Visual observation of the plant community in 1986 indicated that some plant mortality had occurred as a result of the drought; therefore, predrought production levels may not be reached again for several years. Perennial grass production also responded to nitrogen fertilizer addition in 1983 and 1984 with peak biomass occurring at the 2.5 and 5.0 kg N/Mg wood residue rates. In contrast, nitrogen had no appreciable effect on production in 1985 and 1986. With the exception of tall wheatgrass (Thinopyrum ponticum), all successfully established grass species were rhizomatous (see Smith et al., 1986 for details of individual species response to the amendments). This suggests that sod-forming grasses are generally better suited than bunchgrasses for revegetation of bentonite spoils. The predominance of sod-forming grasses on clay soils in the region support this observation (Weaver and Albertson, 1956). Rhizomes have been noted to exhibit physical resistance to breakage and the capacity for regrowth and/or increased production if breakage occurs in a high clay soil (White and Lewis, 1969). Findings have demonstrated that plant species potentially useful and/or successful in revegetation of wood residue amended, abandoned bentonite spoils should have at least some of the following characteristics: sod-forming morphology, drought and salt tolerance, adaptation to clay texture, and adaptation to a shallow, poorly drained spoil/soil environment (Smith et al., 1986).

EVALUATION OF INORGANIC AMENDMENTS

An inorganic amendment study was designed to evaluate the feasibility and effectiveness of amending previously revegetated bentonite mined lands. Gypsum was surface-applied, in April 1987, at the rate of 56 Mg/ha to approximately 40% of each of the native seed mixture plots on the 1981 study (Figure 1). This level of gypsum amendment was calculated to reduce the exchangeable-sodium-percentage (ESP) of the sodic spoil to 15.

Spoil samples were collected in October 1986 and in May 1988-1990 to evaluate the effects of gypsum. Gypsum amendment significantly increased EC at all spoil depths (Figure 4). Such increases in salinity could result in reduction of germination and seedling establishment if applied during initial reclamation. Indeed, if gypsum had been applied during initial reclamation of this site, the EC would have been even higher in these spoils, since by 1987, leaching from the surface spoil depth had occurred over several years to an EC level that was considerably below the initial 1981 EC of 13.4 dS/m (Table 1). However, the observed increase would have been less had the gypsum been incorporated into 30 cm of spoil rather than surface applied. Such a large increase in salinity may have influenced seedling establishment

in Dollhopf and Bauman's (1981) inorganic amendment study since they obtained fair-to-good seedling establishment using organic amendments. Even though the EC increased in 1988, it decreased significantly over the next two years. This decrease is attributed to leaching (Schuman and Meining, 1993). Analysis of vegetation production data suggested no detrimental effects of the increase in EC on the established plant community (Schuman, et al., 1994). Gypsum amendment significantly reduced the ESP of the 60-cm spoil profile (Figure 5). These changes began to become evident within 13 months after treatment. The gypsum amendment also significantly increased the spoil-water storage in the 60-cm spoil profile (Figure 6). Therefore, inorganic amendments, such as gypsum, can significantly improve the physical and chemical properties of these spoils.

WOOD RESIDUE DECOMPOSITION

Sustained long-term success of these reclaimed lands depends on the continued improvement and development of a 'soil'. These 'new soils' must develop active microbial functions to ensure nutrient availability through sustained nutrient cycles. Evaluation of wood residue decomposition, using an ashing technique, indicates that microbial functions have begun to develop. Wood residue decomposition after 1, 2, 3, and 5 years was 10.7, 11.0, 16.5, and 26.3%, respectively (Figure 7). The single nitrogen addition in 1981 had a pronounced effect on decomposition during the 5-year period (Figure 8). Whitford et al. (1989) concluded that in semiarid rangelands where moisture availability affects nitrogen immobilization and mineralization, high C:N ratio amendments (such as wood residue) can be beneficial. They suggested that more resistant sources of organic mulches are superior to readily decomposed materials, such as crop residues because they provide a slow release of organic particles that serve as energy sources for the microflora.

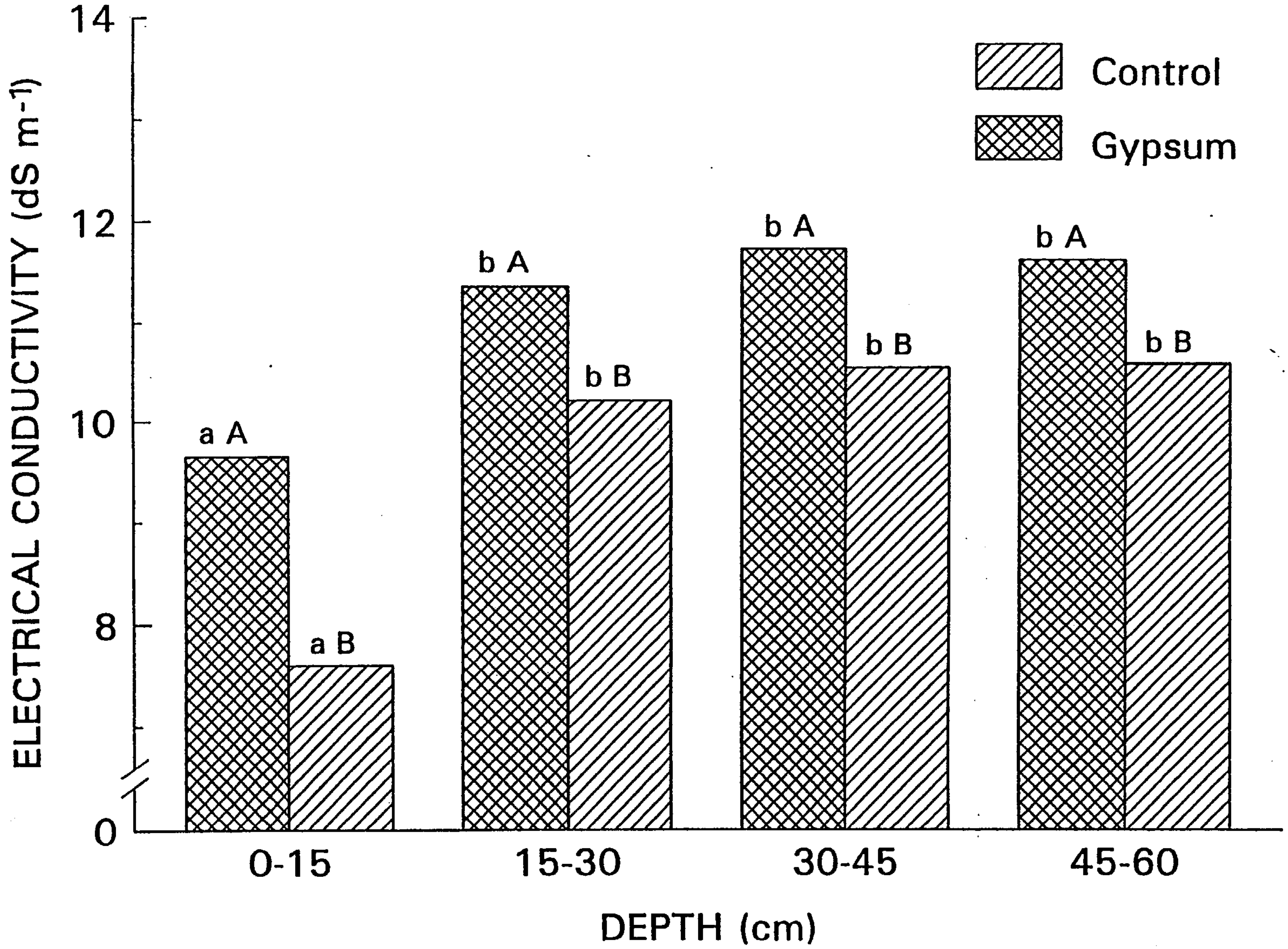

Figure 4. The effect of gypsum amendment on the electrical conductivity of wood residue amended bentonite spoil at four spoil depths, 1988-1990. Means with the same letter within a treatment (lower case) or within a depth (upper case) are not significantly different, $P \leq 0.10$. (Schuman and Meining 1993).

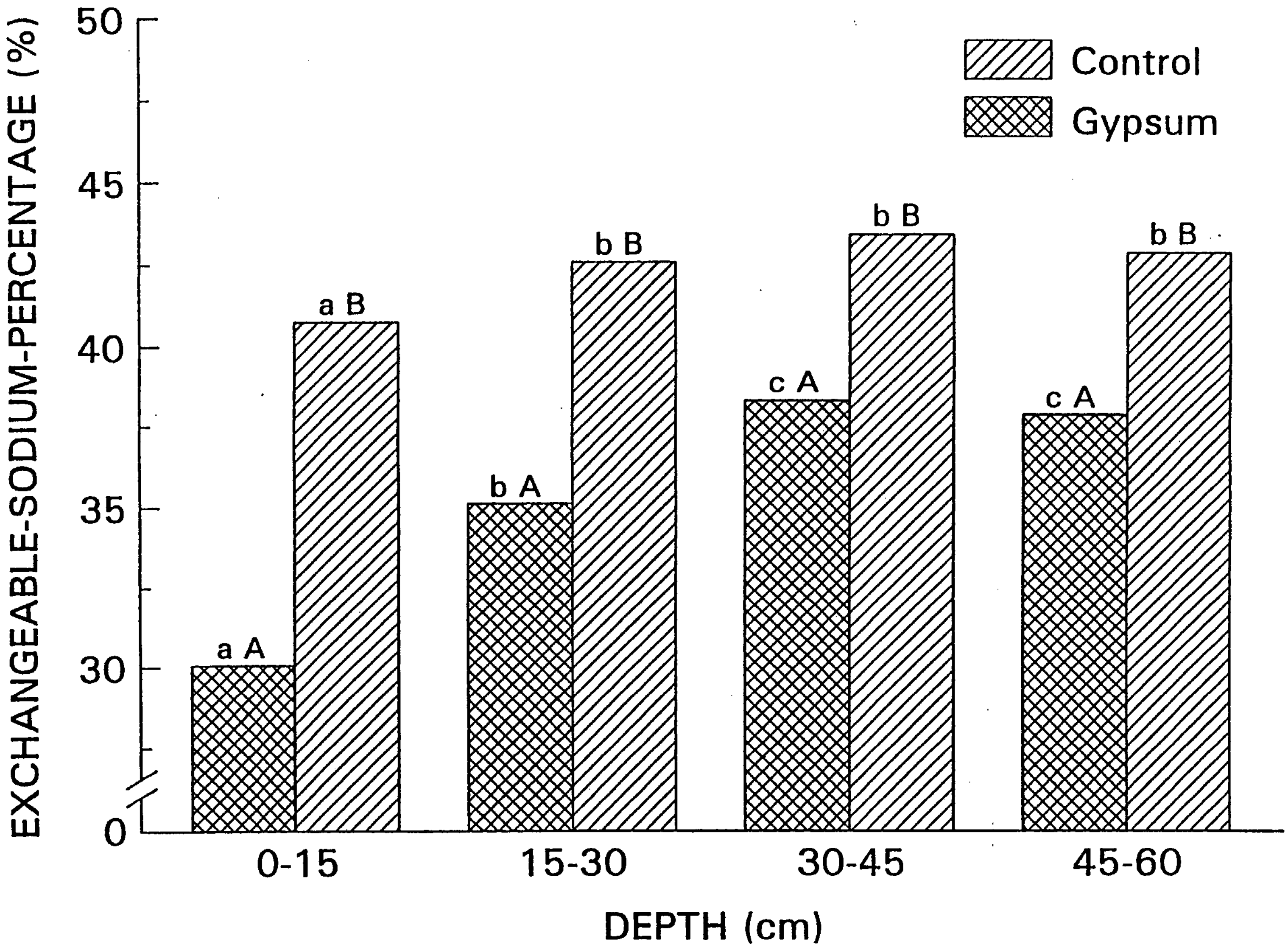

Figure 5. The effect of gypsum amendment on the exchangeable sodium-percentage of wood residue amended bentonite spoil at four depths, 1988-1990. Means with the same letter within a treatment (lower case) or within a depth (upper case) are not significantly different, P≤0.10. (Schuman and Meining 1993)

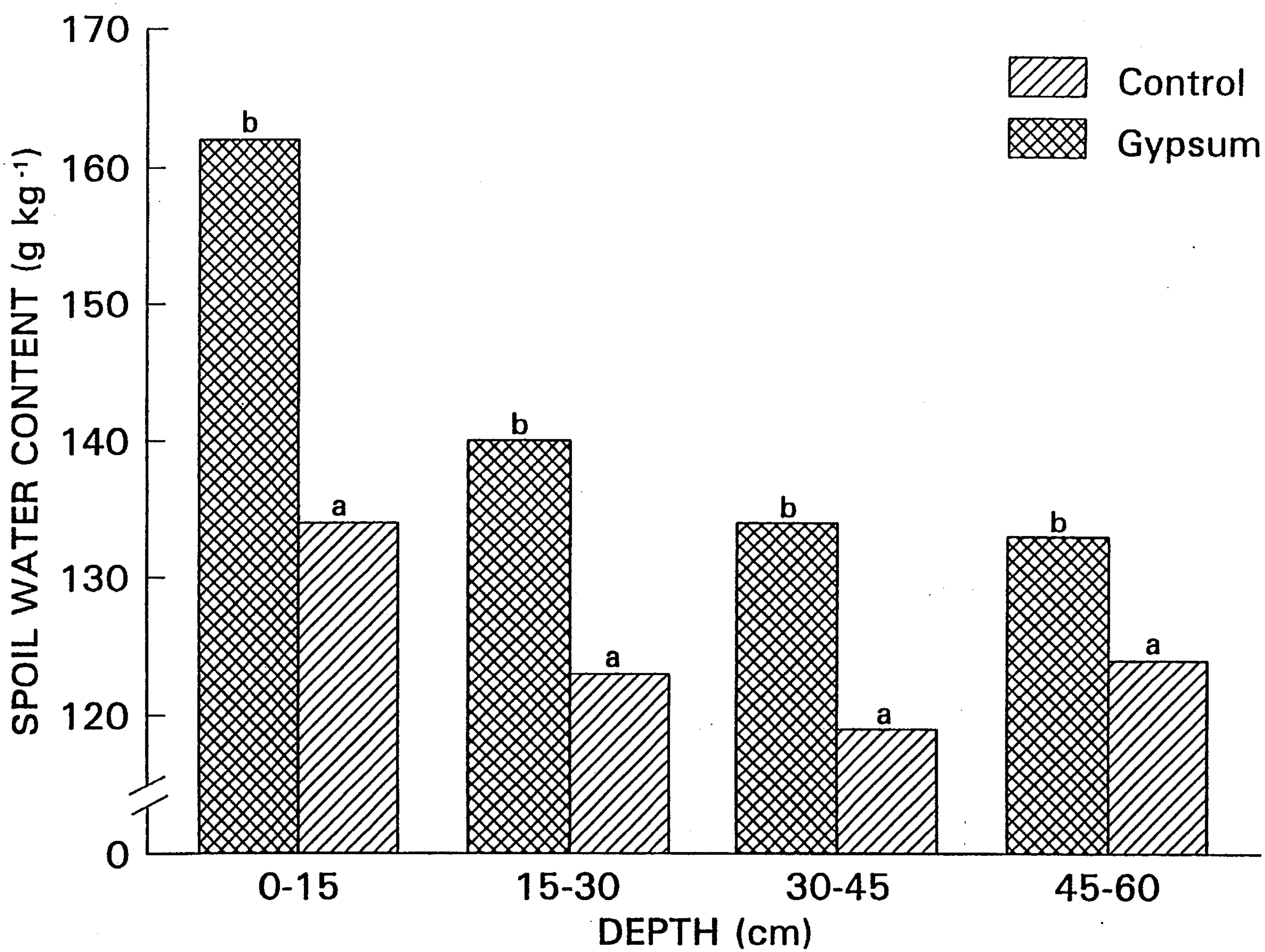

Figure 6. Response of spoil-water content of revegetated saline-sodic bentonite spoil to gypsum amendment, 1988-1990. Means within a spoil depth with the same letter are not significantly different, $P \leq 0.10$. (Schuman and Meining 1993)

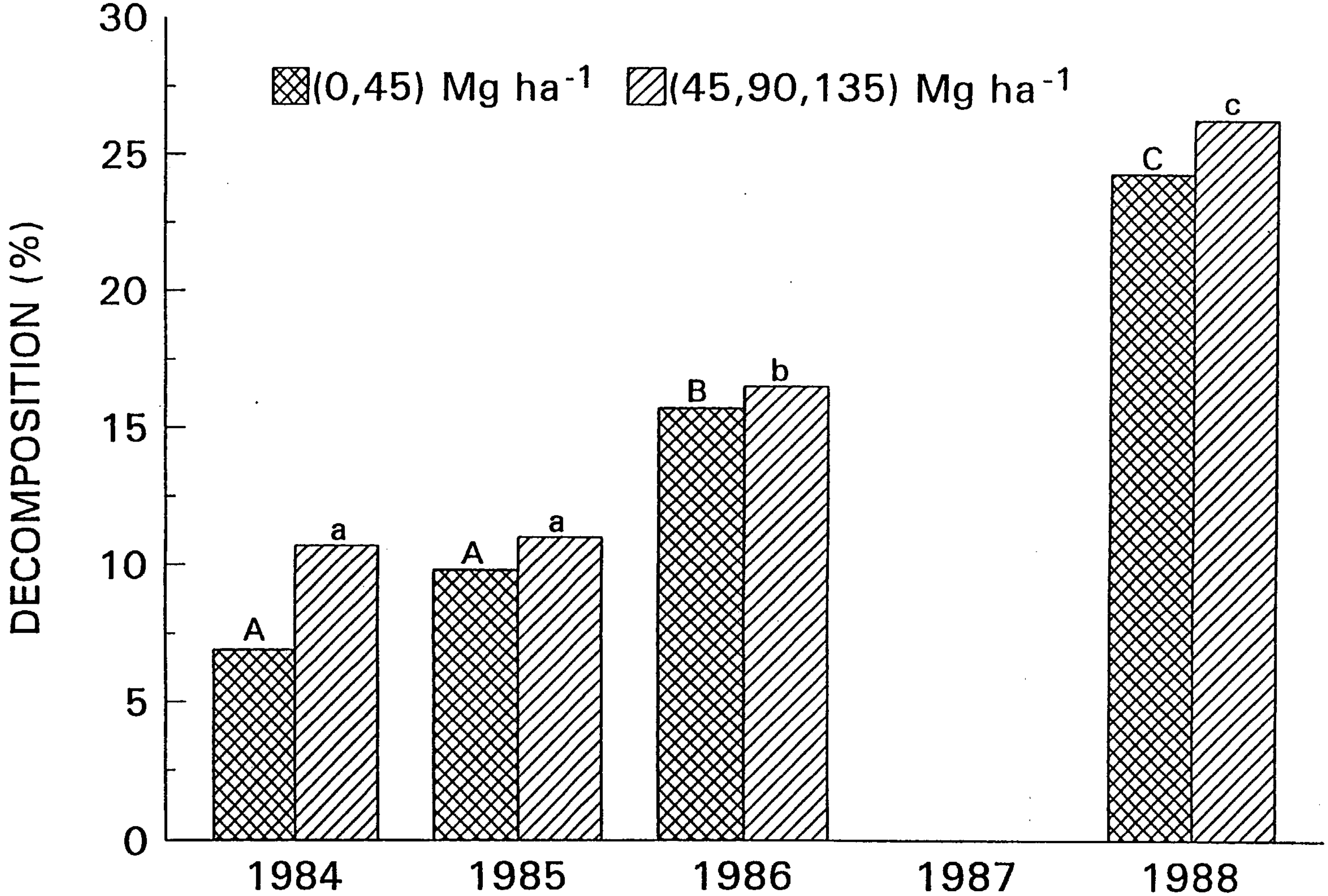

Figure 7. Decomposition of wood residues amended to bentonite spoil for a 5-year period. Data analyzed separately for 0 and 45 Mg/ha and 45, 90, and 135 Mg/ha wood residue treatments because of N application differences. Means within the 0 and 45 Mg/ha and 45, 90, and 135 Mg/ha treatments with the same upper and lower case letter, respectively, are not significantly different, P≤0.05. (Schuman and Belden 1991)

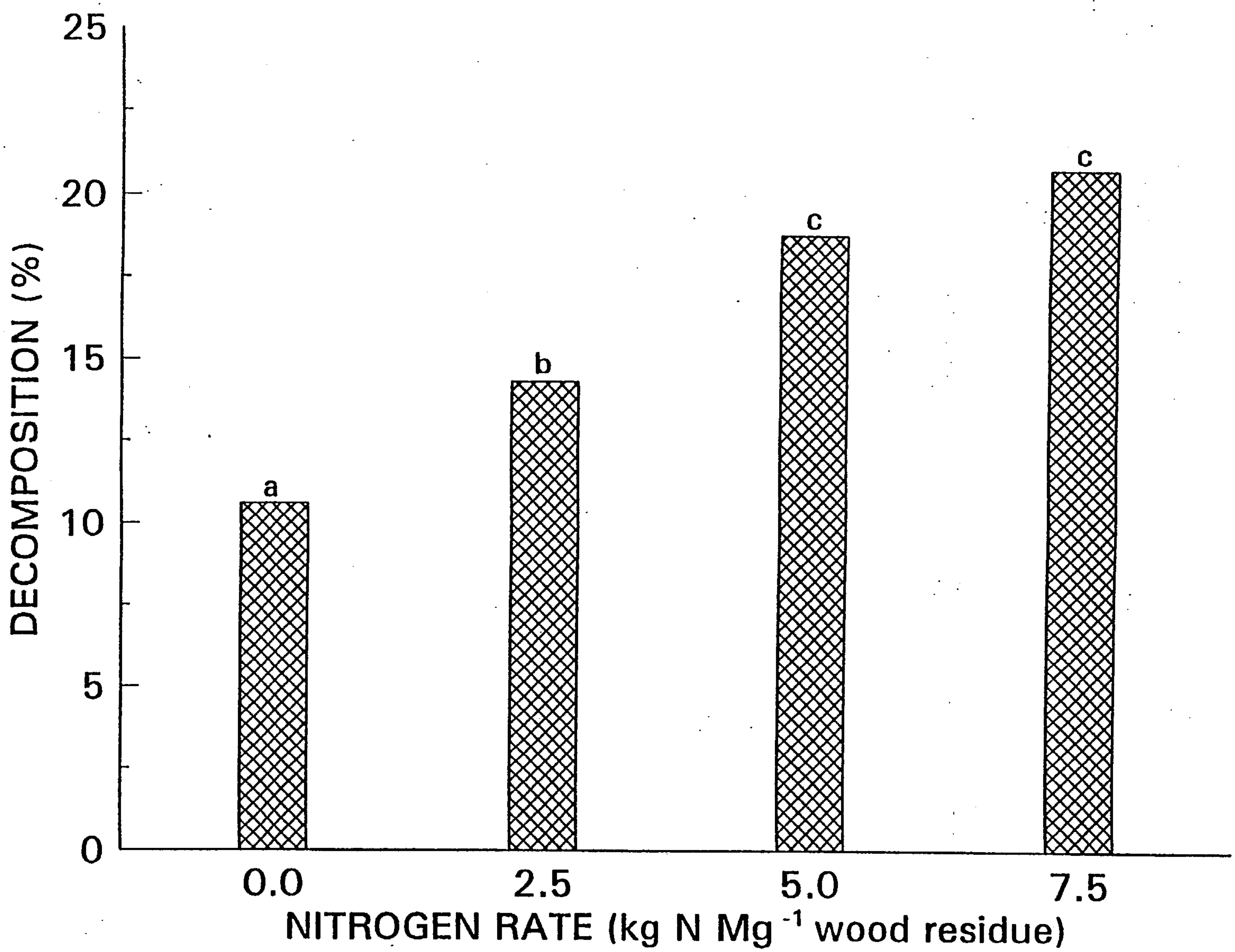

Figure 8. Decomposition of wood residue amended to bentonite spoils as a function of N-fertilizer application rate (averaged across wood residue rates and years). Means with the same letter are not significantly different, $P \leq 0.05$. (Schuman and Belden 1991)

Summary

Wood residue amendment resulted in immediate improvement of the physical characteristics of bentonite spoils, enabling improved water infiltration and storage, leaching of soluble salts, and the concurrent establishment of a desirable and productive plant community. This research also pointed out the need for inorganic amendments, such as gypsum, to be incorporated with the wood residue to ameliorate spoil sodicity. Use of both organic and inorganic amendments are necessary to enable immediate revegetation of these lands and ensure sustainability of the reclaimed ecosystem. Organic amendments utilized should not be easily decomposed since they play an important role improving the physical and biological conditions of the bentonite spoil until natural soil development occurs and nutrient cycles become established.

This technology has been readily adapted to large scale reclamation of abandoned bentonite spoils. Over 6000 hectares of abandoned spoils have been reclaimed using this technology.

Reclaiming abandoned bentonite lands in the Northern Great Plains eliminates significant environmental problems and greatly enhances aesthetic and economic value of these lands. Much of the information and technology described is directly applicable to active bentonite mine land reclamation even when topsoil salvage is practiced, especially since topsoil quantities are generally limited.

Acknowledgement

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References

Ampian, S.G. 1980. Clays. *In*: Mineral facts and problems. USDI. Bureau of Mines Bull. 671. U.S. Government Printing Office,Washington, D.C.

Belden, S.E. 1987. Edaphic responses to wood residue and nitrogen amendment of abandoned bentonite spoil. M.S. Thesis. University of Wyoming, Laramie, WY.

Bjugstad, A.J., Yamamoto, T., and Uresk, D.W. 1981. Shrub establishment on coal and bentonite clay mine spoils. p. 104-122. *In*: Shrub Establishment on Disturbed Arid and Semi-Arid Lands. Wyoming Game and Fish Commission, Cheyenne, WY.

Dollhopf, D.J. and Bauman, B.J. 1981. Bentonite mine land reclamation in the Norther Great Plains. Montana Agr. Exp. Stn. Res. Rept. 179.

Hemmer, D.S., Johnson, S., and Beck, R. 1977. Bentonite mine related reclamation problems in the Northwestern States. Montana Dept. of Lands (Old West Regional Grant No. 10570164).

Schuman, G.E. and Belden, S.E. 1991. Decomposition of wood-residue amendments in revegetated bentonite mine spoils. Soil Sci. Soc. Am. J. 55:76-80.

Schuman, G.E., DePuit, E.J., and Roadifer, K.M. 1994. Plant responses to gypsum amendment of sodic bentonite mine spoil. J. Range Manage. 47:206-209.

Schuman, G.E. and Meining, J.L. 1993. Short-term effects of surface applied gypsum on revegetated sodic bentonite spoil. Soil Sci. Soc. Am. J. 57:1083-1088.

Schuman, G.E. and Sedbrook, T.A. 1984. Sawmill wood residue for reclaiming bentonite spoils. Forest Products J. 34:65-68.

Sieg, C.H., Uresk, D.W., and Hansen, R.M. 1983. Plant-soil relationships on bentonite mine spoils and sagebrush-grassland in the northern high plains. J. Range Manage. 36:289-294.

Smith, J.A. 1984. Wood residue and fertilizer amendments for reclamation of orphan bentonite mine spoils. M.S. Thesis, University of Wyoming, Laramie.

Smith, J.A., DePuit, E.J., and Schuman, G.E. 1986. Wood residue and fertilizer amendment of bentonite mine spoils: II. Plant species responses. J. Environ. Qual. 15:427-435.

Smith, J.A., Schuman, G.E., DePuit, E.J., and Sedbrook, T.A. 1985. Wood residue and fertilizer amendment of bentonite mine spoils: I. Spoil and general vegetation responses. J. Environ. Qual. 14:575-580.

Weaver, J.E. and Albertson, F.W. 1956. Grasslands of the Great Plains. Johnson Publishing Co., Lincoln, NE.

White, E.M. and Lewis, J.K. 1969. Ecological effect of a clay soil's structure on some native grass roots. J. Range Manage. 22:401-404.

Whitford, W.G., Aldon, E.F., Freckman, D.W., Steinberger, Y., and Parker, L.W. 1989. Effects of organic amendments on soil biota on a degraded rangeland. J. Range Manage. 42:56-60.

Session II

Eleven Years of Biosolids Application to Dryland Winter Wheat†

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Abstract

We established four study sites on two farms in eastern Adams County, Colorado in 1982 to compare the effects of applying Littleton/Englewood biosolids (sewage sludge) to NH_4NO_3 on yields, protein, and elemental content of dryland hard red winter wheat (Triticum aestivum L.). Biosolids application ranged from 0 to 27 Mg ha^{-1} and N fertilizer ranged from 0 to 134 kg ha^{-1}. We focused on the 0 and 56 or 67 kg N ha^{-1} and the 6.7 and 26.8 Mg ha^{-1} treatments. Following summer dryland winter wheat management practices, we added the same N fertilizer or biosolids rate every other year six times to one set of plots and five times to the other. Three sets of plots consisted of Weld loam soils (Typic Argiustoll) and the fourth site contained a Platner loam soil (Abruptic Argiustoll).

Results averaged over years showed that neither N fertilizer nor biosolids significantly affected grain yields or grain Mo concentrations. Projected income averaged \$333 ha^{-1} for the N fertilizer treatment and \$384 ha^{-1} for the 6.7 Mg biosolids ha^{-1} treatment. The higher biosolids rate (26.8 Mg ha^{-1}) produced higher grain P levels at three sites and Cu at two sites than did N fertilizer. The Ni grain concentrations were significantly different between some treatments and locations but we did not notice a consistent trend. The most consistent biosolids effects were found for grain protein, where both biosolids rates produced higher protein than the control (0 kg N ha^{-1}) at three sites, and Zn concentrations, where the higher biosolids rates produced higher concentrations than the control and the 56 or 67 kg N ha^{-1} treatment at all sites. Biosolids appear to be a viable alternative to N fertilizer if applied at an agronomic rate (6.7 Mg ha^{-1} or less).

Introduction

The application of sewage biosolids to agricultural land is the major method of biosolids recycling or disposal in the USA (USEPA, 1983). Land application of biosolids allows cities to beneficially use this waste material by recycling plant nutrients such as N and P. As land available for landfills decreases and air quality restrictions prevent incineration, the beneficial use of biosolids will increase. We must carefully manage land application of biosolids so as to avoid excessive trace-metal uptake by plants or accumulation in soils of trace metals, contamination of groundwater with NO_3-N, and persistence of pathogenic organisms and toxic organic compounds (Logan and Chaney, 1983; Sommers and Barbarick, 1986).

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Our research objective is to compare the effect of Littleton/Englewood biosolids and N fertilizer rates on dryland winter wheat (Triticum aestivum L., 'Vona') grain yield, elemental content of the soil and wheat grain and straw, estimated income, and soil NO_3-N accumulation. Three student theses (Lerch 1987, 1991; Utschig, 1985), technical reports (Barbarick et al. 1984, 1986a, 1986b, 1987a, 1987b, 1991, 1992a, 1992b; Ippolito et al., 1992, 1993, 1994; Lerch et al. 1988, 1989, 1990a; Utschig et al. 1985), and journal articles (Barbarick and Workman, 1987; Barbarick et al., 1995; Lerch et al. 1990b, 1990c, 1992, 1993a, 1993b; Utschig et al., 1986) detail the 11 years of results from this study. Rather than present all of the data, we will focus on the effects of selected rates of N fertilizer and biosolids on selected plant parameters and on residual NO_3-N accumulation with depth.

Materials and Methods

Our long-term field study was initiated in August 1982 on two locations near Bennett, CO in Adams County. We used two sets of plots (A for those established in 1982; B for those established in 1983) on both the East and West Bennett sites, since we grew hard red winter wheat in a dryland-summer-fallow-rotation system. Table 1 indicates the rotation used for locations A and B at both sites.

Table 1. Crop rotation, biosolids, and N fertilizer application history at each location.

Crop Year	Location A	Location B
1982-83	Biosolids, N, Wheat	Fallow
1983-84	Fallow	Biosolids, N, Wheat
1984-85	Biosolids, N, Wheat	Fallow
1985-86	Fallow	Biosolids, N, Wheat
1986-87	Biosolids, N, Wheat	Fallow
1987-88	Fallow	Biosolids, N, Wheat
1988-89	Biosolids, N, Wheat	Fallow
1989-90	Fallow	Biosolids, N, Wheat
1990-91	Biosolids, N, Wheat	Fallow
1991-92	Fallow	Biosolids, N, Wheat
1899	Biosolids, N, Wheat	Fallow

East Bennett site A consisted of a Weld loam soil (fine, montmorillonitic, mesic Aridic Paleustoll); West Bennett site A consisted of a Platner loam soil (fine, montmorillonitic, mesicAbruptic Aridic Paleustoll). Weld loam soil constituted both experiments at site B.

Selected properties for baseline soils taken from 0-15, 15-30, and 30-60 cm are listed in Table 2.

Table 2. Selected baseline soil chemical characteristics.					
Property	Depth cm	East 1982	West 1982	East 1983	West 1982
pH	0-15	6.7	6.5	6.6	7.5
	15-30	6.9	6.6	7.4	7.3
	30-60	7.8	7.5	7.1	7.5
E.C., ds m^{-1}	0-15	0.1	0.1	0.3	0.4
	15-30	0.1	0.2	0.5	0.4
	30-60	0.2	0.3	0.5	0.4
O.M., g kg^{-1}	0-15	9	10	10	10
	15-30	6	9	9	11
	30-60	7	8	11	10
NO_3-N, mg kg^{-1}	0-15	3	2	4	4
	15-30	1	7	3	2
	30-60	1	1	4	4
AB-DTPA, mg kg^{-1}					
P	0-15	11	9	11	18
	15-30	4	4	3	17
	30-60	1	1	8	18
K	0-15	375	462	383	343
	15-30	266	438	388	316
	30-60	184	356	382	335
Zn	0-15	0.7	1.6	0.9	0.7
	15-30	0.6	0.6	0.6	0.6
	30-60	0.4	0.3	1	0.8
Fe	0-15	13	18	11	7
	15-30	11	14	8	9
	30-60	10	9	16	10
Mn	0-15	14	11	27	13
	15-30	3	4	12	10
	30-60	2	2	28	14
Cu	0-15	2.1	2.5	2.6	2.7
	15-30	2.3	2.5	3	2.6
	30-60	2.6	3	2.6	2.4

We applied biosolids and N fertilizer treatments (NH_4NO_3; 34-0-0) to plots of 4.4 by 17.1 m in 1982, 1984, 1986, 1988, 1990, and 1992 at site A. The dry biosolids application rates were 0, 7, 14, 28, and 42 Mg ha^{-1} and N fertilizer rates were 0, 28, 56, and 112 kg N ha^{-1}. In 1983, 1985, 1987, 1989, and 1991, we added the same rates of biosolids used at site A to plots of 3.3 by 17.1 m at site B; N fertilizer rates at site B were 0, 34, 67, 101, and 134 kg N ha^{-1}. We used four replications of all treatments at all experimental sites. In 1982 and 1983, we applied a more liquid biosolids (35 and 42 g solids ha^{-1}; Table 3) to our bermed plots by transfer from semi-tanker trucks through fire hose. Our farmer cooperators disked the biosolids and N treatments after the liquid biosolids had infiltrated into the soil and dried on the surface. The pre-weighed NH_4NO_3 treatments were spread uniformly by hand. In subsequent years, we weighed the drier biosolids (528 to 850 g solids kg^{-1}; Table 3), spread them evenly over the plots by a front-end loader, and then hand raked to improve the uniformity of distribution. In the years that we used dried biosolids, we roto-tilled all plots to a depth of 10 to 15 cm.

For this report we will present only the data for the control (0 kg N ha^{-1}), a typical N fertilizer rate (56 kg N ha^{-1} at site A; 67 kg N ha^{-1} at site B), our recommended rate of biosolids for dryland wheat (6.7 Mg biosolids ha^{-1} per application), and an excessive rate of biosolids (26.8 Mg biosolids ha^{-1} per application). Properties of biosolids applied each year are given in Table 3. The biosolids were classified as Grade I for 3 years and as Grade II for 8 years (Colorado Department of Health, 1985). Grade I biosolids, if dewatered and stored for one year prior to application may be applied to any land for any beneficial use. Grade II biosolids can be used for beneficial use on agricultural and disturbed lands only (Colorado Department of Health, 1993). The Mo concentration would dictate whether the biosolids in any given year are either Grade I (less than 18 mg kg^{-1}) or Grade II (between 18 and 75 mg kg^{-1}). The Grade I Mo limitation is currently on hold until more information on Mo concentration effects on plant toxicity is reported.

Table 3. Characteristics of biosolids applied to East and West Bennett sites from 1982 to 1992 (year applied).

Parameter	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	avg.±sd
Solids, g kg^{-1}	35	42	528	680	850	850	780	760	720	560	760	597±294
Total N, g kg^{-1}	128	90	29	22	10	13	9	38	30	50	46	42±37
NO_3-N, g kg^{-1}	3	2	□1	1	□1	2	□1	□1	□1	□1	□1	□1
NH_4-N, g kg^{-1}	50	57	8	□1	2	3	□1	6	6	11	12	14±20
P, g kg^{-1}	28	21	11	27	8	13	16	21	34	33	37	23±10
K, g kg^{-1}	9	7	3	3	2	3	3	3	2	4	4	4±2
Cu, mg kg^{-1}	1040	877	462	1159	359	422	917	884	865	807	863	787±259
Zn, mg kg^{-1}	1980	1300	751	2538	618	681	1054	1255	1303	1317	1303	1282±56 6
Ni, mg kg^{-1}	70	94	56	91	47	50	66	78	93	91	177	83±36
Pb, mg kg^{-1}	270	161	531	322	206	204	228	134	134	129	20	213±133
Cd, mg kg^{-1}	□30	15	8	11	10	9	12	9	9	7	7	10±2
Mo, mg kg^{-1}	59	20	12	38	7	9	20	36	32	45	22	27±16
Location where biosolids were applied	A	B	A	B	A	B	A	B	A	B	A	

Grain yields were determined by harvesting a 15.2 by 1.2 m area. We measured grain elemental concentrations of Cd, Cu, Mo, Ni, P, Pb, and Zn in concentrated HNO_3 digests (Havlin and Soltanpour, 1980) by the inductively coupled plasma-atomic emission spectrophotometer (ICP-AES). We calculated mean concentrations for each treatment and each location. This allowed us to compare results from all four locations even though the A plots received six applications while the B plots received five applications of biosolids. We compared these means statistically using oneway analyses of variance. We calculated mean separation values only for those cases where the probability level for the oneway analyses of variance were less than 0.10 (F-LSD).

For estimated income calculations, we based grain prices (including protein premiums when given) for each year on the most current quotes for that harvest year. Nitrogen fertilizer costs were based on anhydrous NH_3, since it is the most commonly used material in eastern Colorado for dryland wheat production. The Littleton/Englewood biosolids and its application are currently free to farmers within a 65-km radius of the waste treatment facility.

Immediately following each harvest, soil samples from 0-20 and 20-60 cm were taken. Proceeding the 1992 harvests, soil samples from 60-90, 90-120, and 120-180 cm also were obtained. After the 1993 harvest, soil samples from 60-100, 100-150, and 150-200 cm were acquired. We determined the NO_3-N concentration in $CuSO_4/Ag_2SO_4$ extracts using the NAS-Szechrome colorimetric procedure (Szekely, 1968). We statistically analyzed the treatment differences for the mean of each depth increment by oneway analyses of variance. We again used mean separation tests (α = 0.05 probability level) for those concentrations that were significantly different.

Results and Discussion

Grain Elemental Concentrations

Even after five or six applications of biosolids, we have not added large quantities of most of the biosolids-borne trace elements (Table 4). These additions, however, have led to significant total accumulations in soils (Ippolito et al., 1993, 1994). Despite this soil accumulation, grain concentrations of these trace elements have not approached critical health levels (Logan and Chaney, 1983; Sommers and Barbarick, 1986).

Table 4. Total amounts of selected trace elements added with the two biosolids rates after five or six applications.

	Plot A, 6.7 Mg ha^{-1}	Plot B, 6.7 Mg ha^{-1}	Plot A, 26.8 Mg ha^{-1}	Plot B, 26.8 Mg ha^{-1}
		kg ha^{-1}		
Cu	30.2	121	27.8	111
Zn	47	188	47.5	190
Ni	3.41	13.6	2.71	10.8
Pb	9.31	37.2	6.36	25.5
Cd	0.41	1.63	0.34	1.37
Mo	1.02	4.07	0.99	3.97

The N fertilizer or biosolids treatments did not significantly affect mean yields (Fig. 1). Essentially, bioisolids treatments have performed equally, in terms of grain yield, with the control (0 kg N ha^{-1}) or the typical N fertilizer rate (56 or 67 kg N ha^{-1}).

We found that both biosolids rates increased grain protein above the control (0 kg N ha^{-1}) at all locations except the East B site (Fig. 2). For the same three locations, the highest biosolids rate produced higher grain protein than the N fertilizer treatment (56 or 67 kg N ha^{-1}). We observed no differences in grain protein between the two biosolids rates.

The range in income was from a low of \$263 for the control at West Bennett B to a high of \$491 for the 6.7 Mg ha^{-1} biosolids rate at East Bennett A (Fig. 3). The overall averages were \$313, 333, 384, and 355 ha^{-1} for the control, N fertilizer treatment, 6.7 Mg biosolids ha^{-1} rate, and 26.8 Mg biosolids ha^{-1} treatment, respectively. This income range draws wheat producers' attention.

As shown in Fig. 4, the higher biosolids treatment (26.8 Mg ha^{-1}) produced higher grain P concentrations than the typical N fertilizer rate at three locations. The lower biosolids treatment (6.7 Mg ha^{-1}) created higher P levels than the N fertilizer rate at East A and West A. Average grain P concentrations ranged from 3.2 mg kg^{-1} for the N fertilizer at East A to 4.7 mg kg^{-1} for the 26.8 Mg biosolids ha^{-1} at East B and West A. We did not anticipate large differences in grain P, since the initial plant-available P levels in the surface soils ranged from moderate to high (Table 2). Barbarick et al. (1995) also showed that grain P approached a maximum plateau asymptotically (it leveled off) as biosolids-bourne P rates increased .

The higher biosolids rate produced higher grain Cu levels (Fig.5) than the control at East A and West A. As with P, grain Cu has plateaued as amount of biosolids Cu applied has increased (Barbarick et al., 1995).

Biosolids and N fertilizer application did not significantly affect grain Mo concentrations (Fig. 6). These data plus the information from Barbarick et al. (1995) indicate that the original USEPA (1993) and Colorado Department of Health and Environment (1993) limit of 18 mg Mo kg^{-1} for Grade I biosolids may be too restrictive for dryland winter wheat.

The 26.8 Mg biosolids ha^{-1} rate has produced higher grain Ni levels (Fig. 7) than the control and the N fertilizer treatment at West A. Unlike the other nutrients studied, Ni has demonstrated a linear increase in concentration as the amount of biosolids Ni applied has increased (Barbarick et al., 1995).

Both biosolids rates produced significantly higher grain Zn concentrations than the control and the N fertilizer treatments at all locations (Fig. 8). The higher biosolids rate also resulted in significantly higher Zn levels than the lower biosolids treatment at West A. Our soils were originally low in plant-available Zn (Table 2) so that the biosolids apparently supplied plant-available Zn.

Soil NO_3-N Concentrations

B Plots

Five applications of 26.8 Mg biosolids ha^{-1} resulted in significantly higher NO_3-N concentrations at 0 to 20 cm than any other treatment at East Bennett (Fig. 9). The high biosolids rate also resulted in significant accumulations of NO_3-N at 20 to 60 and 150 to 200 cm compared to the control and the N fertilizer treatment. At the 60- to 100-cm depth, NO_3-N levels associated with the high biosolids rate was significantly higher than the control. The 6.7 Mg biosolids ha^{-1} treatments contained significantly more NO_3-N than the control and 67 kg fertilizer N ha^{-1} treatment at the 0- to 20-cm depth; it was not significantly larger than the other two treatments at deeper depths.

Biosolids applied five times at 26.8 Mg ha^{-1} produced significant NO_3-N accumulations compared to all other treatments for the 0- to 100-cm depths at West Bennett (Fig. 10) and compared to the control and N fertilizer at the 150- to 200-cm depth. Five applications of 6.7 biosolids Mg ha^{-1} did not increase soil NO_3-N levels compared to the control and 67 kg N fertilizer ha^{-1} at any depth.

The residual NO_3-N to 100 cm for the 6.7 Mg biosolids ha^{-1} treatment pose a potential for NO_3-N leaching. The large amounts of N applied with the first application in 1983 produced most of the NO_3-N accumulation (Table 3)(Utschig et al., 1986). The four subsequent additions consisted of dried biosolids (>50% solids; Table 3) which resulted in lower total applied N levels. Without the first large N application, the residual carryover would more nearly match that of the 67 kg N fertilizer ha^{-1} treatment.

The high biosolids rate supplies excessive N. Thus, the potential for NO_3-N leaching is high.

A Plots

Six applications of 26.8 Mg biosolids ha^{-1} resulted in significantly larger accumulations of soil NO_3-N when compared to the other three treatments for all depths at East Bennett (Fig. 11) and all but the 150- to 200-cm depths at West Bennett (Fig. 12). Residual NO_3-N was significantly higher with the six additions of the low biosolids rate than with the control or with 56 kg N fertilizer ha^{-1} at the 20- to 150-cm depths at East Bennett but was not significantly different than the control or 56 kg N fertilizer ha^{-1} treatment at all soil depths at West Bennett. Again, we can attribute most of the NO_3-N carryover to the first large application in 1982 (Table 3; Utschig et al., 1986), and we would have a closer match to the residual NO_3-N found with 56 kg N fertilizer ha^{-1} if we had not used more liquid biosolids for the first addition.

We do not recommend the 26.8 Mg biosolids ha^{-1} treatment, since it caused elevated NO_3-N concentrations at most soil depths. Knowledge of the quantities of different N forms in biosolids and the form of the material (liquid, dewatered, dried) is required to determine optimum application rates.

Conclusions

Five or six applications of 6.7 or 26.8 Mg biosolids ha^{-1} increased wheat grain protein, P, Cu, and Zn concentrations compared to those from a control (0 kg N ha^{-1}) or N fertilizer (56 or 67 kg N ha^{-1}) treatment. We found inconsistent trends in grain levels of Ni; and, no significant effects on grain yields and grain Mo concentrations. Projected income ranged from an average of \$313 ha^{-1} for the control to \$384 ha^{-1} for the low biosolids rate.

The soil NO_3-N concentrations associated with the higher biosolids rate pose a potential for NO_3-N leaching. Nitrate-N accumulated in the lower biosolids treatments to a depth of about 100 cm. We attributed most of the higher levels of NO_3-N in the upper depths to the high application of N associated with the liquid materials that we initially applied to each set of plots (1982 and 1983). This biosolids management technique is no longer used by Littleton/Englewood so that a large application of N would not normally occur.

Nitrogen management is the largest challenge in determining safe biosolids application rates. Barbarick et al. (1995) have shown that grain trace-element concentrations tend to approach a maximum (a plateau) as biosolids addition increase. Consequently, accurately determining the agronomic rate to meet the N needs of the target crop should help prevent excessive trace element uptake by plants and NO_3-N leaching below the root zone.

Figure 1. Grain yields averaged over location, 1982-93.

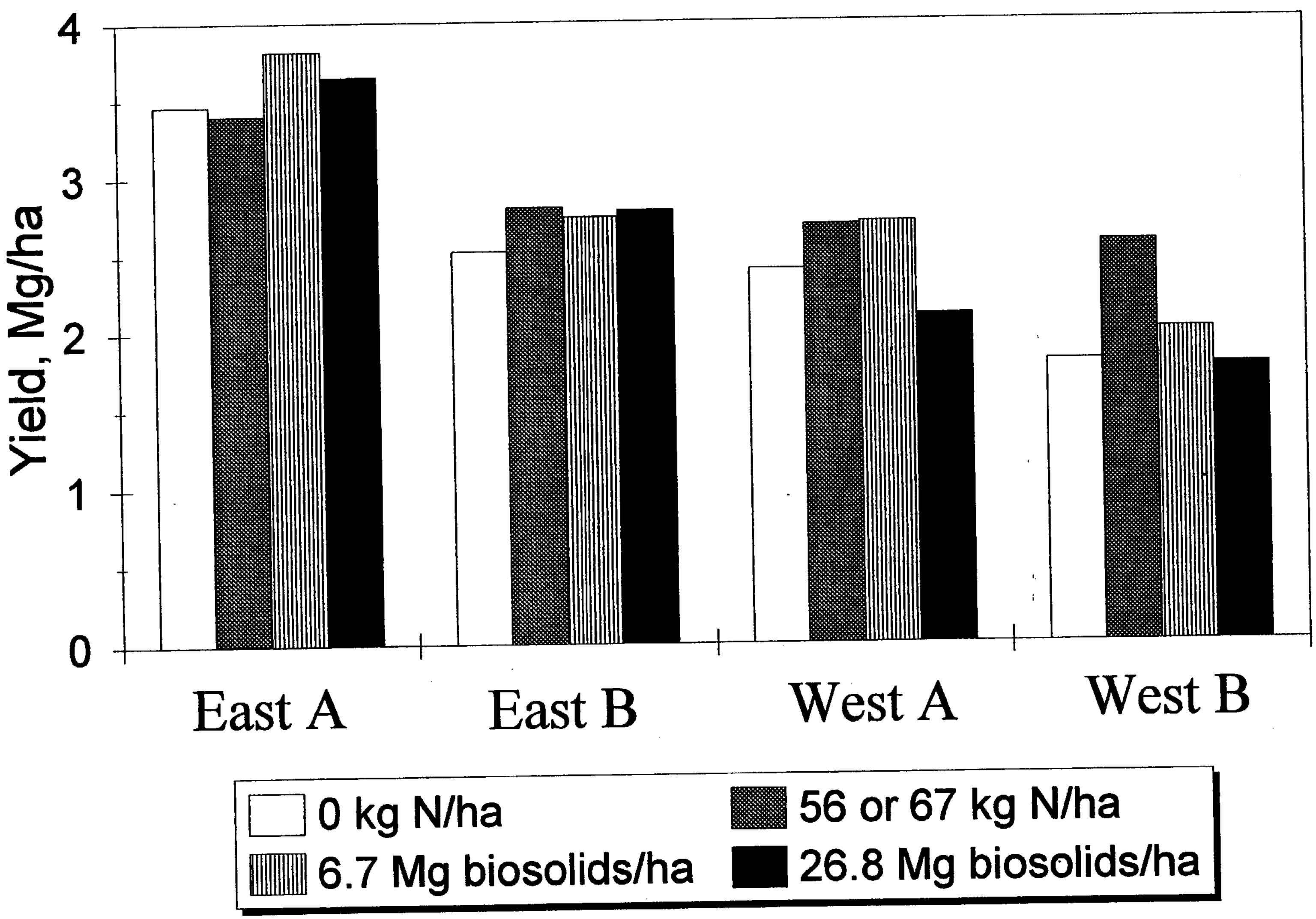


Figure 2. Grain protein concentrations averaged over location, 1982-93.

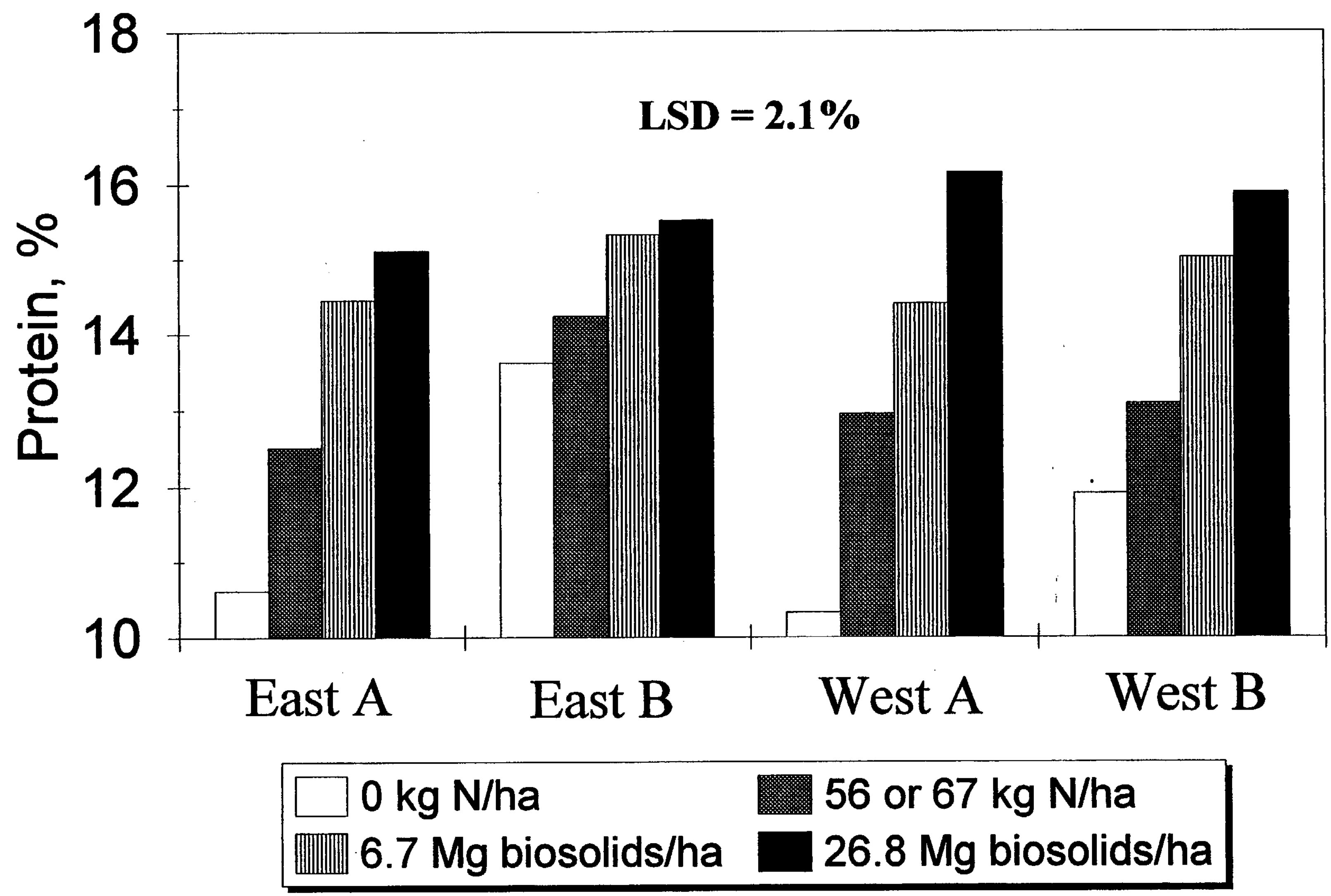


Figure 3. Projected income averaged over location, 1982-93.

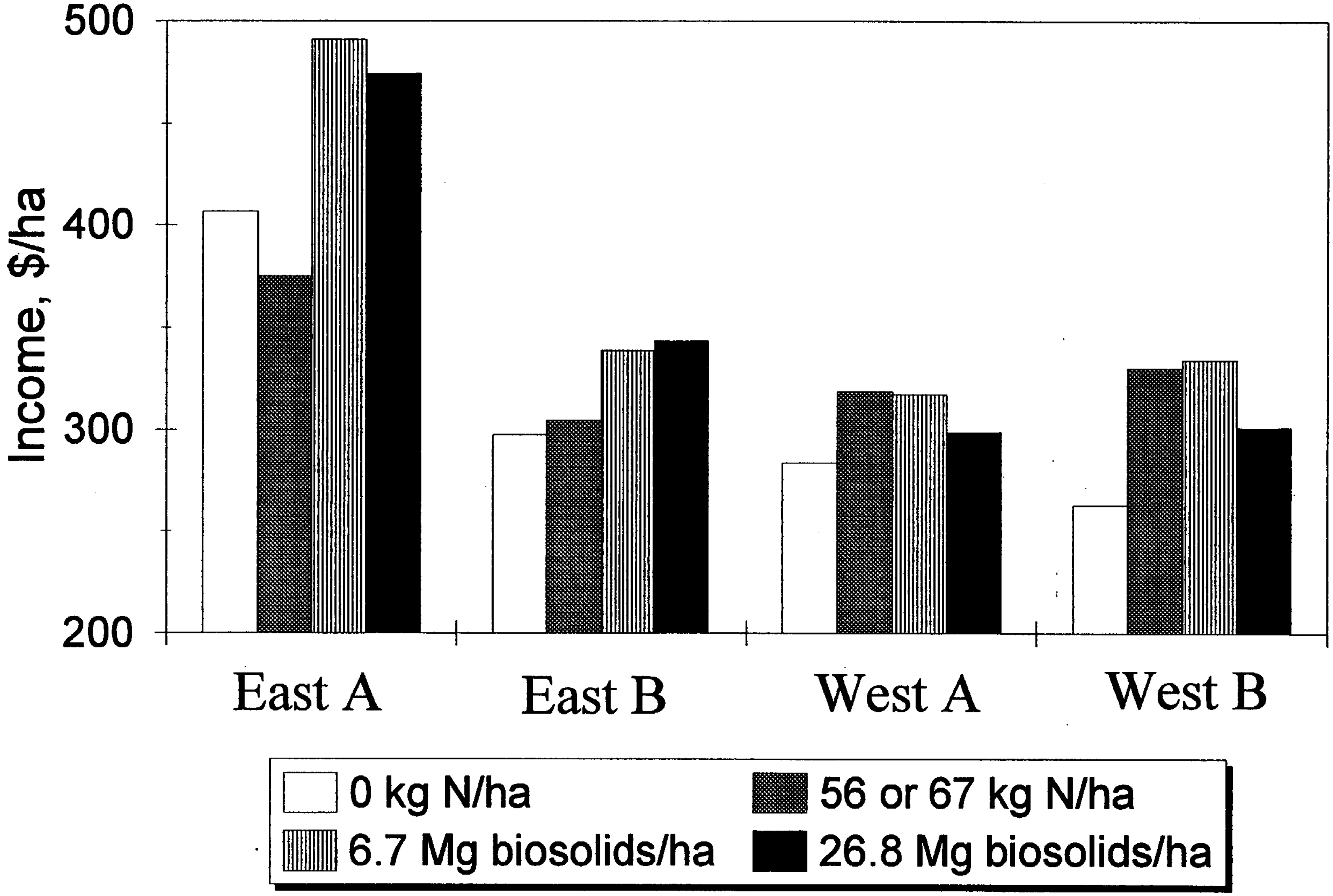


Figure 4. Grain P concentrations averaged over location, 1982-93.

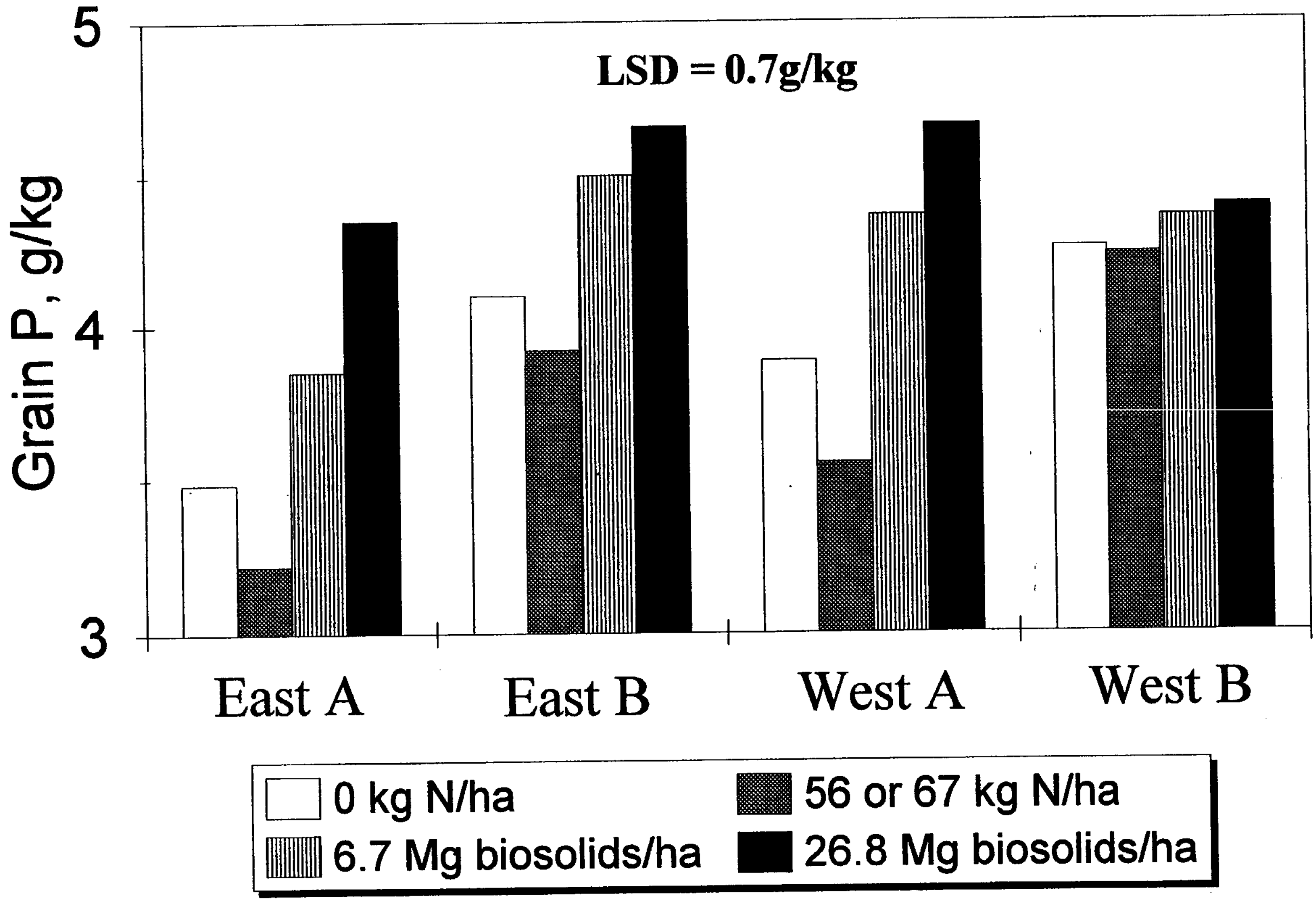


Figure 5. Grain Cu concentrations averaged over location, 1982-93.

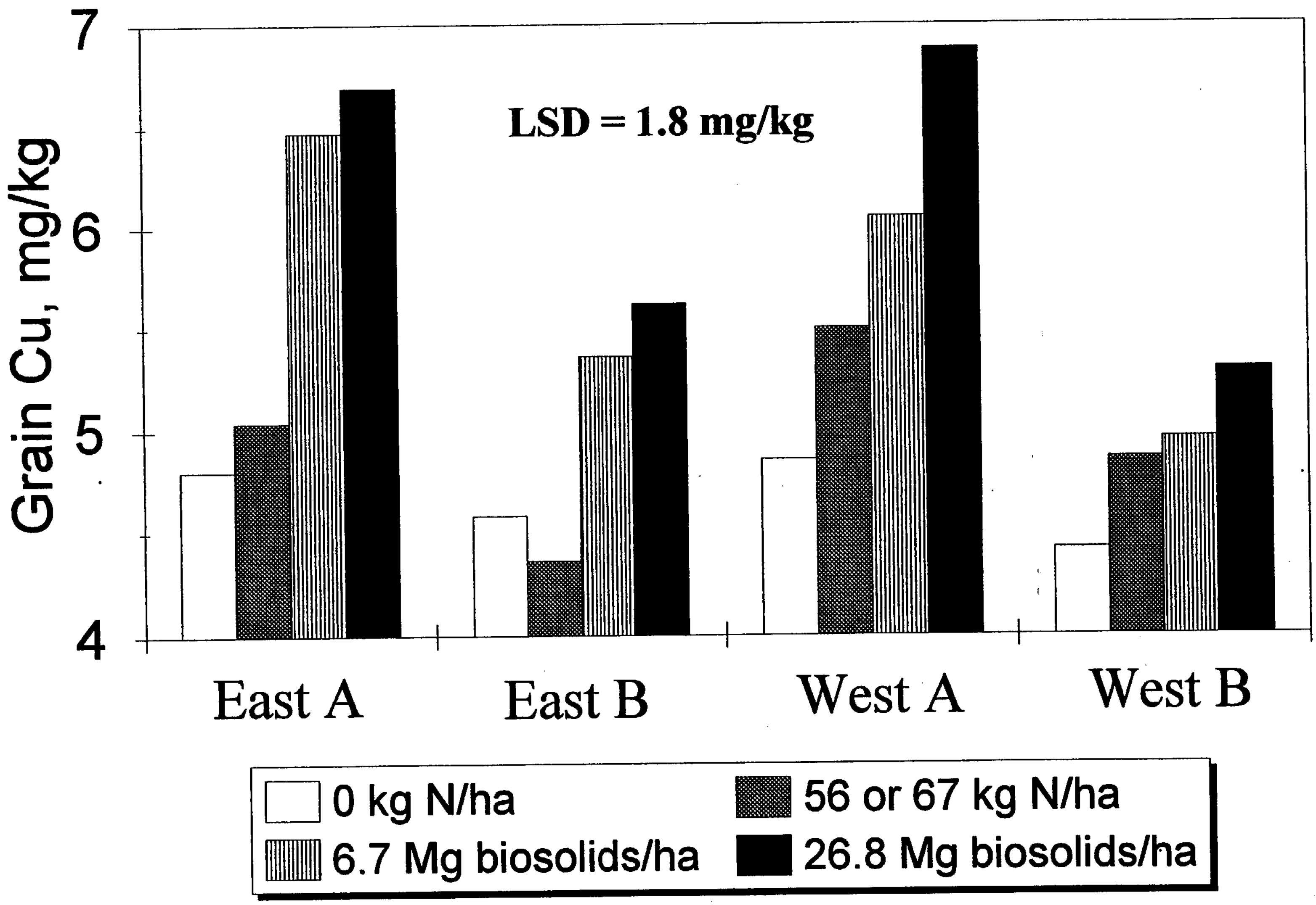


Figure 6. Grain Mo concentrations averaged over location, 1982-93.

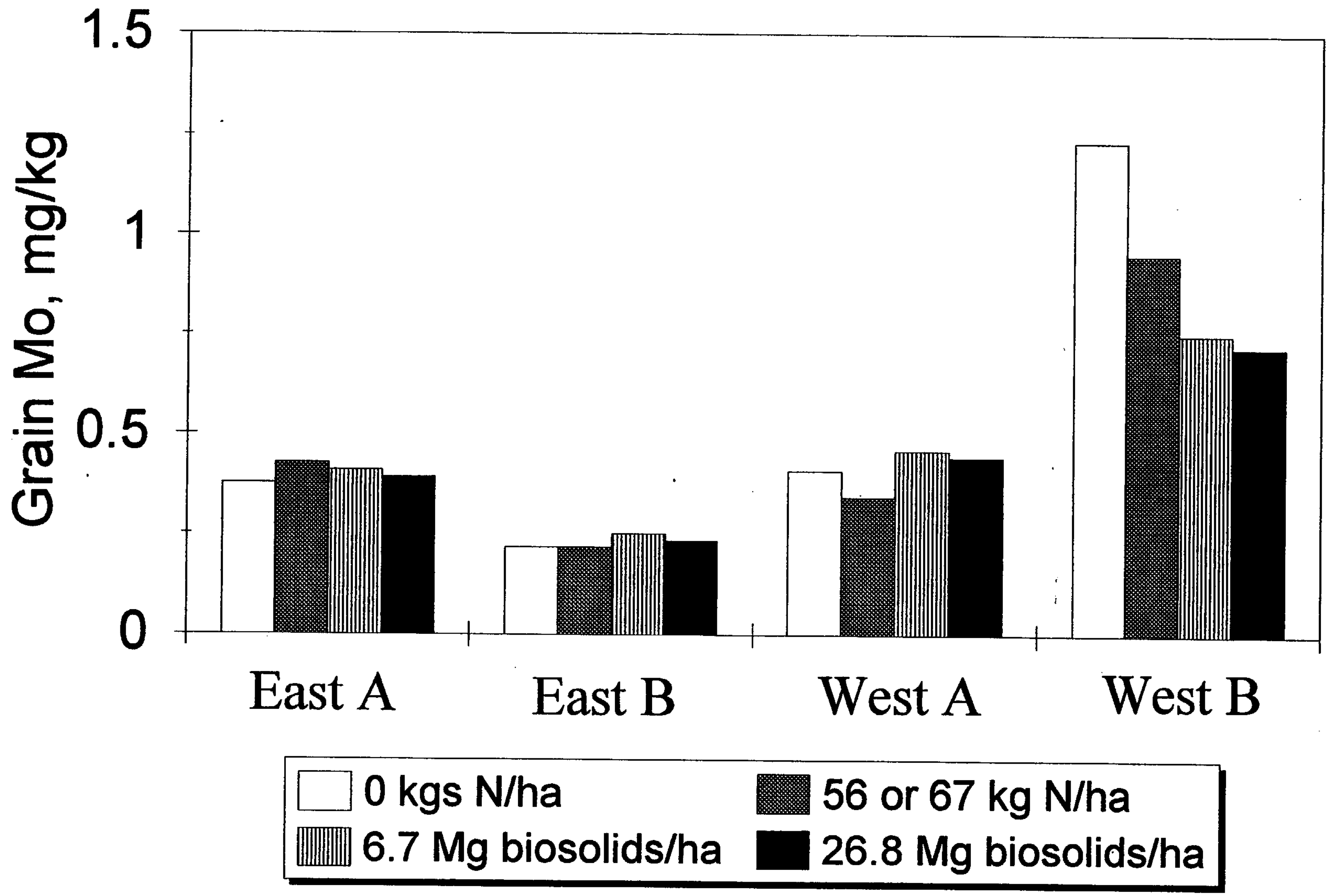


Figure 7. Grain Ni concentrations averaged over location, 1982-93.

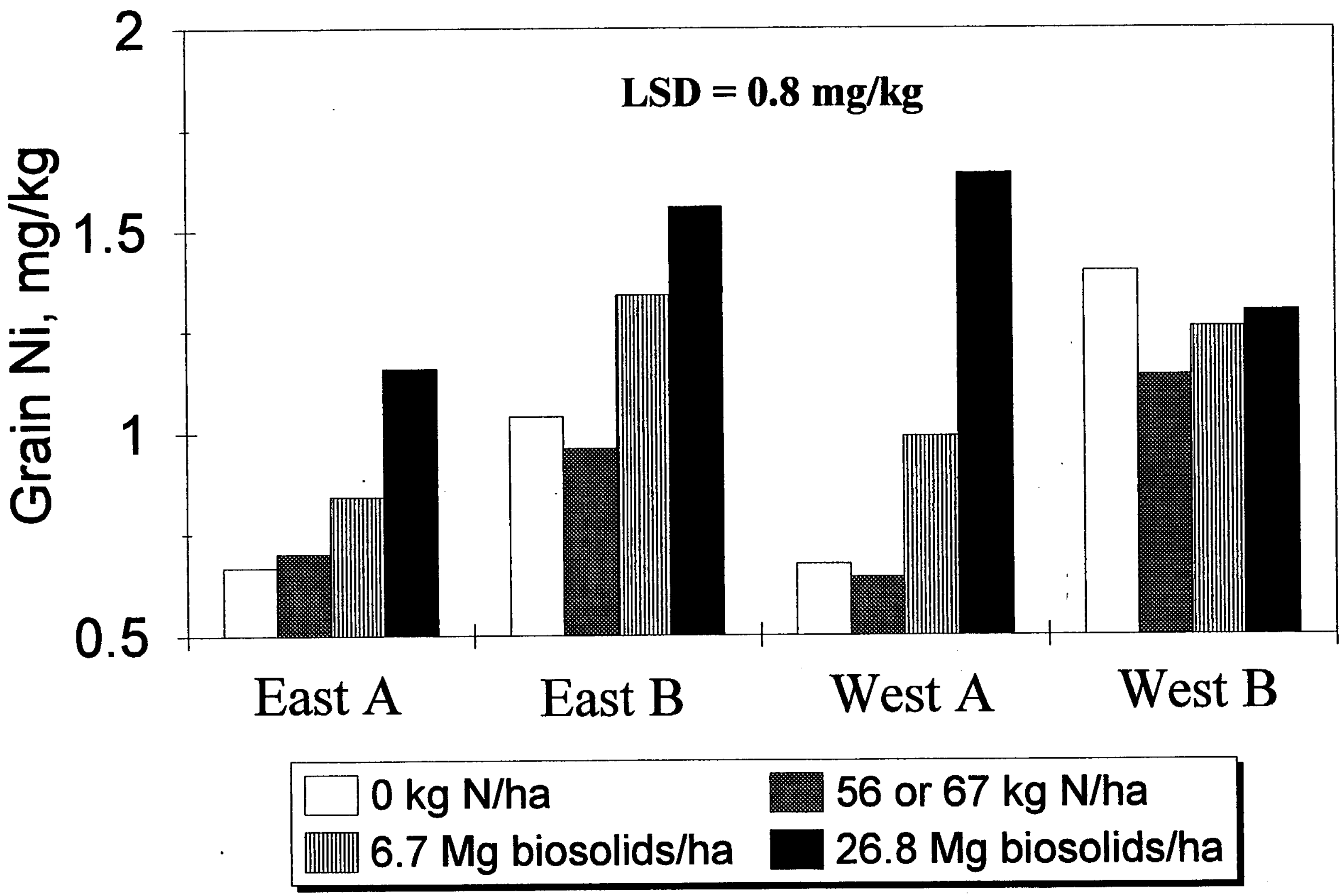


Figure 8. Grain Zn concentrations averaged over location, 1982-93.

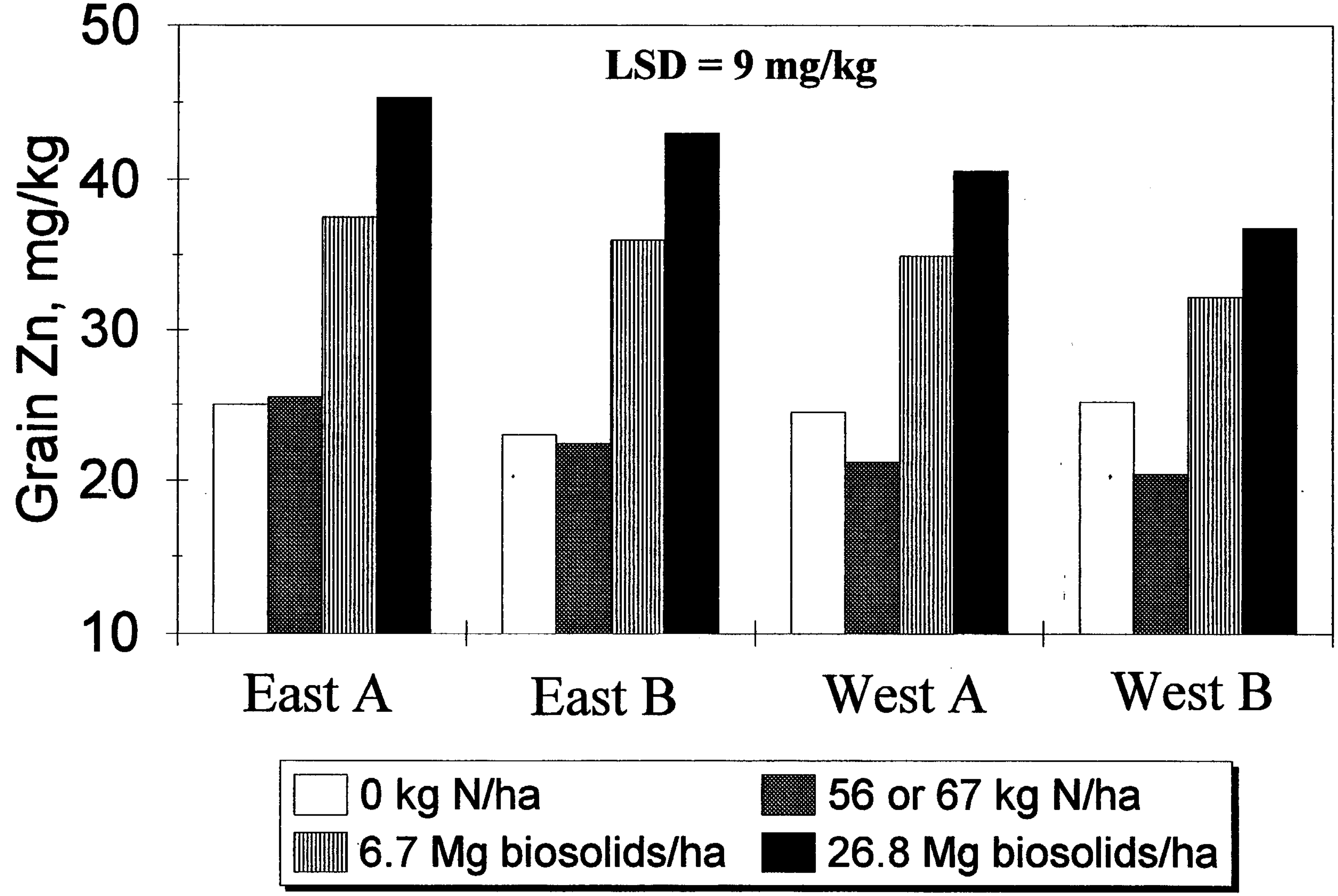


Figure 9. The effect of selected biosolids and N rates on harvest soil nitrate-N, 1991-92, East Bennett B plots.

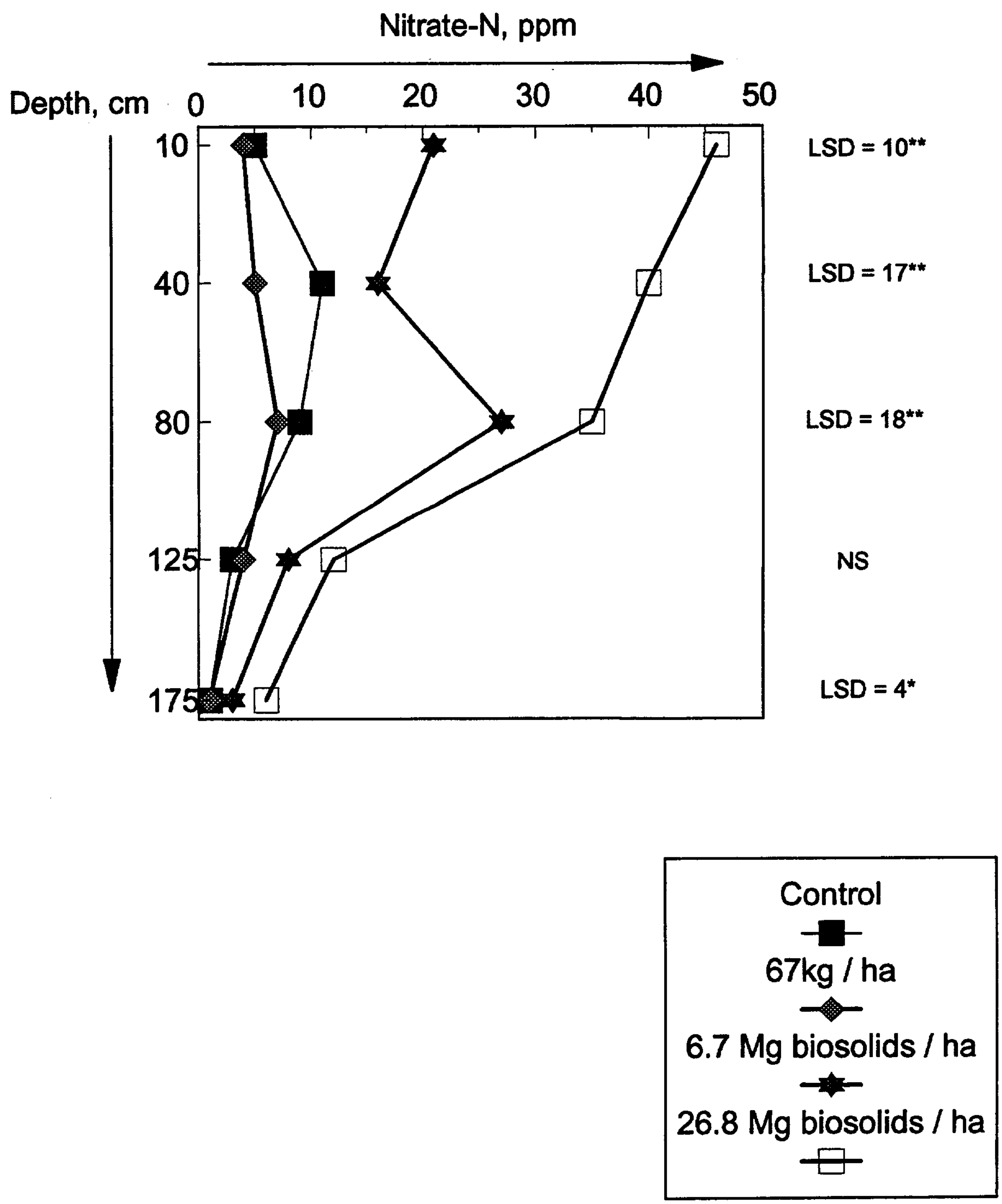


Figure 10. The effect of selected biosolids and N rates on harvest soil nitrate-N, 1991-92, West Bennett B plots.

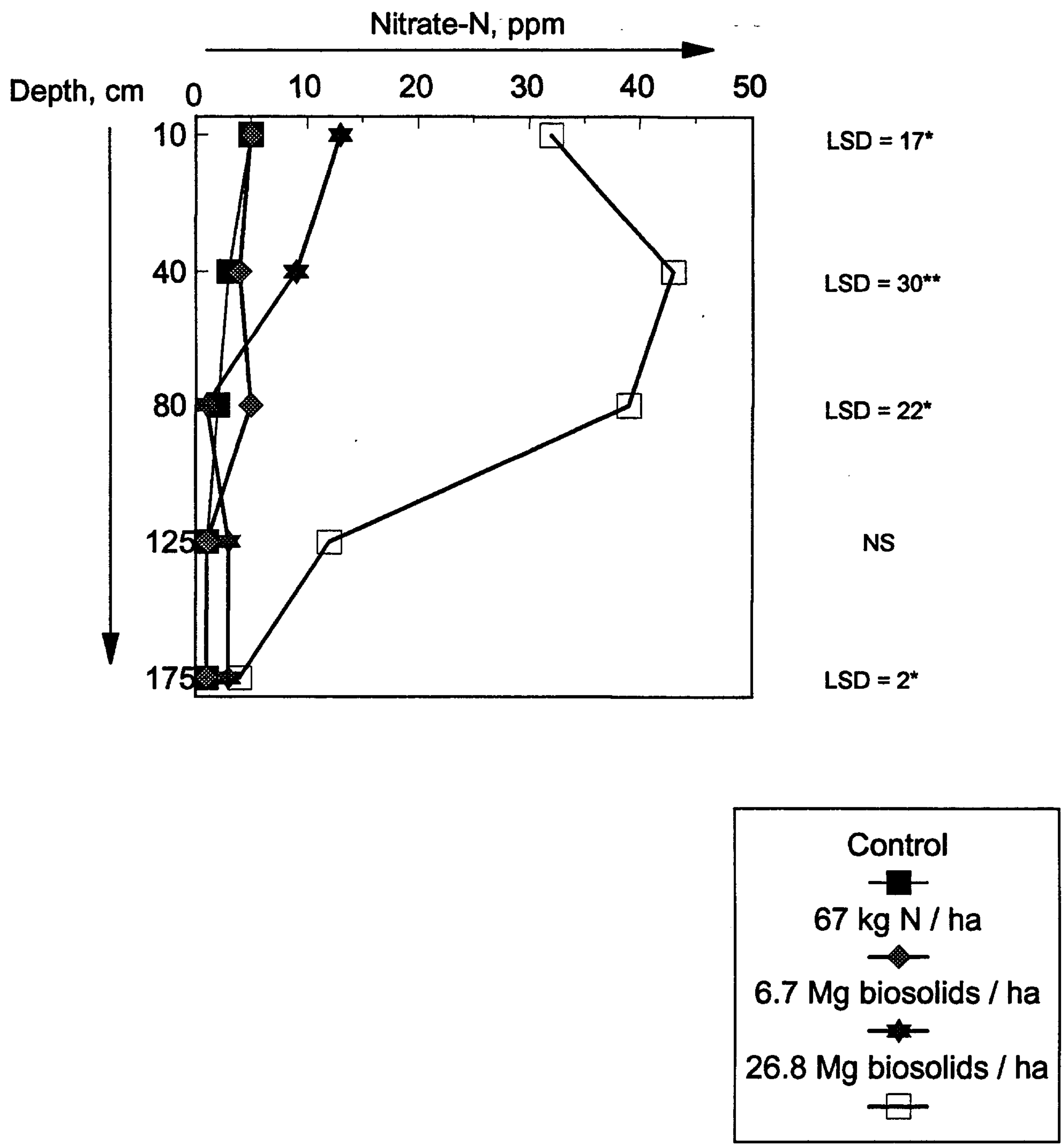


Figure 11. The effect of selected biosolids and N rates on harvest soil nitrate-N, 1992-93, East Bennett A plots.

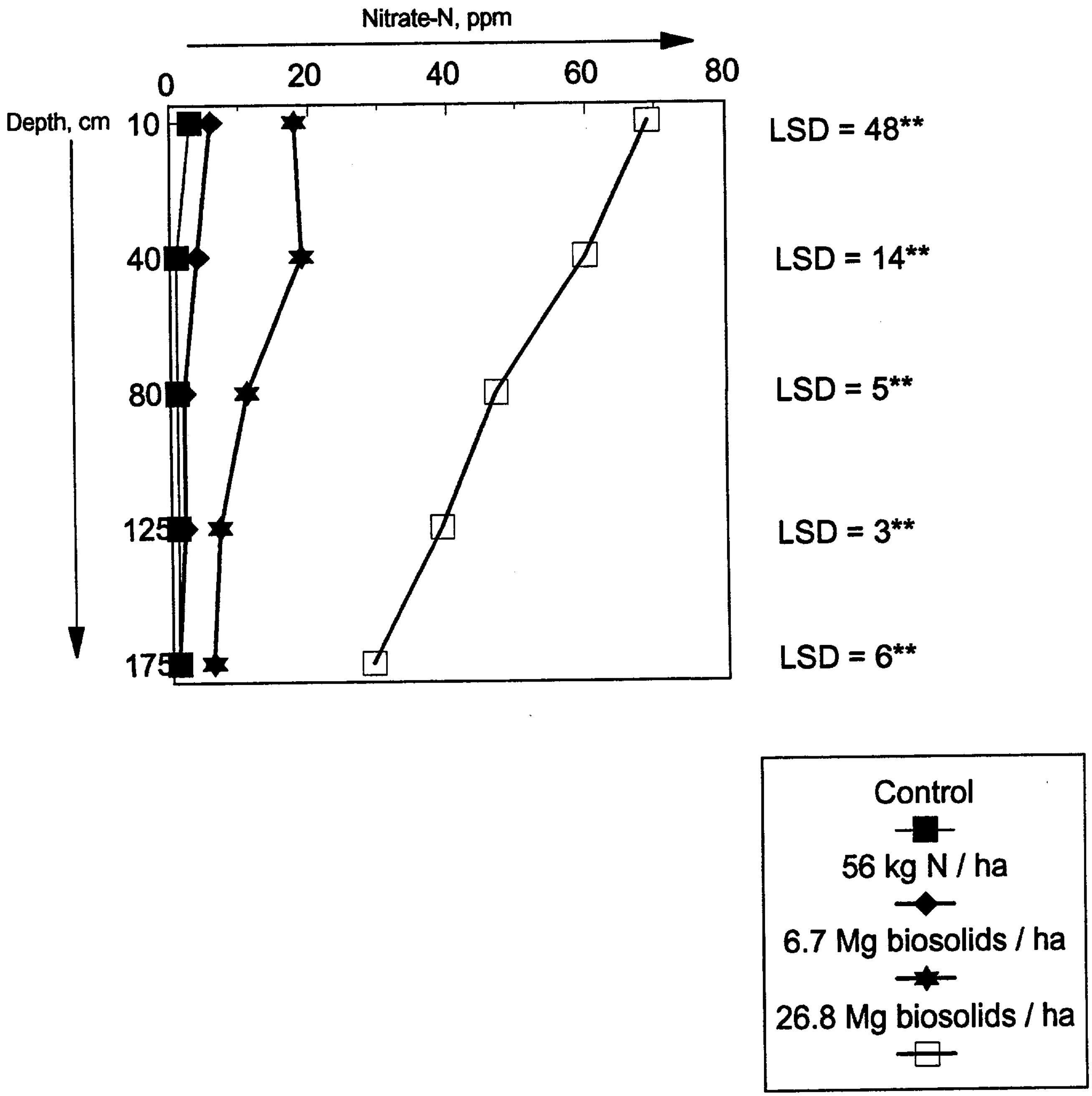
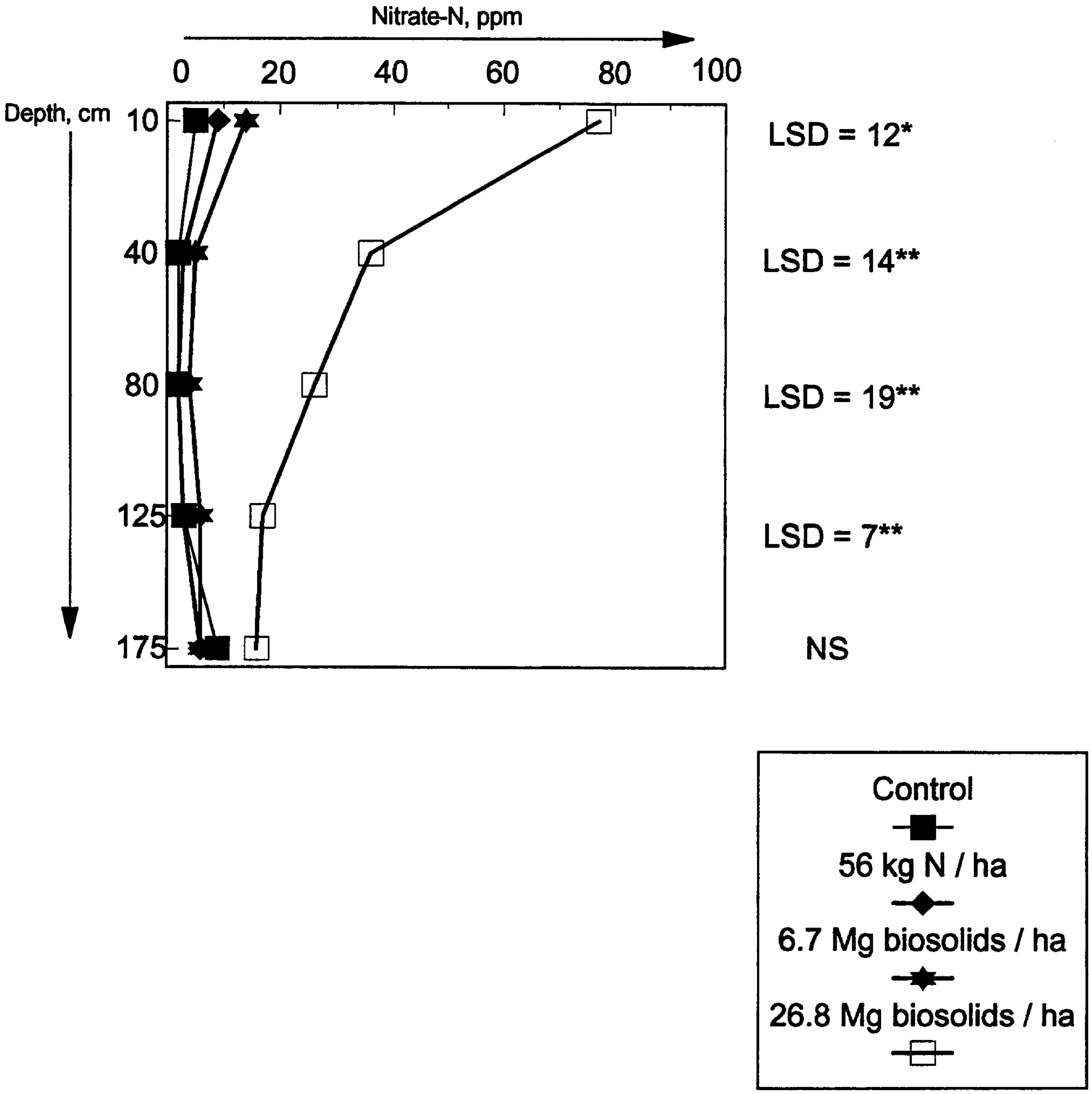


Figure 12. The effect of selected biosolids and N rates on harvest soil nitrate-N, 1992-93, West Bennett A plots.



REFERENCES

Barbarick, K. A., and S. M. Workman. 1987. NH_4HCO_3-DTPA and DTPA extractions of sludge-amended soils. J. Environ. Qual. 16:125-130.

Barbarick, K.A., J.A. Ippolito, and D.G. Westfall. 1995. Biosolids effect on P, Cu, Zn, Ni, Mo concentrations in dryland wheat. J. Environ. Qual. (In Press).

Barbarick, K.A., J. Ippolito, D.G. Westfall, R.H. Follett, and R. Jepson. 1992a. Fate of sewage sludge nitrogen applied to soils. Colorado Agricultural Experiment Station Bulletin TB92-3.

Barbarick, K. A., R. Lerch, J. M. Utschig, D. G. Westfall, R. H. Follett, T. M. McBride, and W. F. Owen. 1987. Application of municipal sewage sludge to dryland winter wheat. Waste Management Conference - Focus on the West Proceedings. Fort Collins, CO.

Barbarick, K. A., R. Lerch, D. G. Westfall, R. H. Follett, and T. M. McBride. 1987. Application of anaerobically digested sewage sludge to dryland wheat. Technical Report 87-10. Colorado Agricultural Experiment Station.

Barbarick, K. A., R. N. Lerch, D. G. Westfall, R. H. Follett, J. Ippolito, and R. Jepson. 1991. Application of anaerobically digested sewage sludge to dryland winter wheat. Technical Report TR91-5, Colorado Agricultural Experiment Station.

Barbarick, K.A., R.N. Lerch, J.M. Utschig, D.G. Westfall, R.H. Follett, J. Ippolito, R. Jepson, and T. McBride. 1992b. Eight years of sewage sludge addition to dryland winter wheat. Colo. Agric. Exp. Stn. Bulletin. TB92-1.

Barbarick, K. A., J. M. Utschig, D. G. Westfall, R. H. Follett, and T. McBride. 1984. Application of the liquid/anaerobically digested sewage sludge from Littleton/Englewood to dryland winter wheat in Bennett, Colorado, 1982-83. Progress Report No. 10, Colorado State University Agricultural Experiment Station.

Barbarick, K. A., J. M. Utschig, D. G. Westfall, and R. H. Follett. 1986a. Liquid sludge incorporation and nitrogen availability to dryland winter wheat. Technical Bulletin 86-2. Colorado Agricultural Experiment Station.

Barbarick, K. A., D. G. Westfall, R. H. Follett, R. Lerch, and T. M. McBride. 1986b. Application of anaerobically digested sewage sludge to dryland winter wheat. Technical Report 86-9. Colorado Agricultural Experiment Station.

Colorado Department of Health. 1985. Domestic Sewage Sludge Regulations. Denver, CO.

Colorado Department of Health. 1993. Biosolids Regulation 4.9.0.

Havlin, J.L., and P.N. Soltanpour. 1980. A nitric acid plant tissue digest method for use with inductively coupled plasma spectrometry. Comm. Soil Sci. and Plant Anal. 11:969-980.

Ippolito, J., K.A. Barbarick, D.G. Westfall, R.H. Follett, and R. Jepson. 1992. Application of anaerobically digested sewage sludge to dryland winter wheat. Colorado Agricultural Experiment Station Technical Report. TR92-5.

Ippolito, J., K.A. Barbarick, D.G. Westfall, R.H. Follett, and R. Jepson. 1993. Application of anaerobically digested sewage sludge to dryland winter wheat. Colorado Agricultural Experiment Station Technical Report. TR93-5.

Ippolito, J., K.A. Barbarick, D.G. Westfall, and R. Jepson. 1994. Application of anaerobically digested biosolids to dryland winter wheat. Colorado Agricultural Experiment Station Technical Report. TR94-6.

Lerch, R.N. 1987. Performance of sewage sludge versus nitrogen fertilizer applied to dryland winter wheat. M.S. Thesis. Colorado State University. Fort Collins, CO.

Table 1. Biosolids sources, processing, and selected chemical analyses.

Biosolids Source	Biosolids Digestion Process	Moisture Reduction Process	Field Locations	Biosolids Analysis			
				P mg/kg	Cd mg/kg	Cu mg/kg	Zn mg/kg
A	anaerobic	drying bed	1 and 2	22600	5.3	616	893
B	anaerobic	belt filter press	3, 4, and 5	27000	7.7	840	860
C	anaerobic	belt filter press	6	26200	14.8	405	883

Table 2 . Biosolids nitrogen analyses and suggested application rates*.

Field Location	Biosolids Analyses			Fertilizer Guide Fert. N Rqmt. kg/ha	Estimated biosolids plant-available N g/kg	Suggested biosolids appl. rate Mg/ha
	Total Solids g/kg	Total N g/kg	Ammonium-N g/kg			
1	800	40.4	10.4	73	11.2	6.5
2	900	45.0	7.2	28	11.2	2.5
3	200	51.7	14.0	67	14.5	4.6
4	200	43.0	15.4	52	13.2	3.9
5	200	43.0	11.8	63	12.1	5.2
6	200	40.3	9.7	0	11.0	0.0

See Table 3 for calculation of fertilizer N requirement for each location.

* Biosolids appl. rate calculations from WSU Cooperative Extension Bulletin EB 1432, "Sewage Sludge Guidelines for Washington, Part 3":
Estimated biosolids plant available N (g/kg dry biosolids) = 0.2 * biosolids organic N + 0.5 * biosolids ammonium-N
Suggested N application rate (Mg/ha dry biosolids) = Recommended fertilizer N rate (kg/ha)/biosolids plant-available N (g/kg)

Table 3. Field location data and preapplication soil chemical properties

Field Location	Months from application to harvest	Winter wheat variety	Average annual precipitation	Soil chemical analyses: Preapplication Soil Nitrate-N Avg.	Std. Dev.	Fertilizer Guide Fert N Rqmt.	Organic Matter	pH	Olsen P	Exch. K
			cm	kg/ha	kg/ha	kg/ha	g/kg		mg/kg	mg/kg
1	14	Sprague	25	46	8	73	22	6.6	19.7	815
2	22	Eltan	25	76	30	28				
3	15	Eltan	27	49	8	67	12	6.7	11.3	344
4	21	Eltan	27	52	3	52	13	6.4	12.0	446
5	21	Tres	29	37	12	63	13	6.5	18.0	488
6	14	Rely	36	227	54	0	28	6.3	12.1	594

Soil sampling depth: NO3-N: 90 cm; organic matter, pH, P, and K: 30 cm

N fertilizer requirement (4.0 Mg grain/ha yield goal) calculated based on Washington State
Univ. Fertilizer Guide 34, "Dryland Wheat Nitrogen Needs" using the following factors:

N mineralized from soil organic matter per 12 months (kg/ha) = average annual precipitation (cm) * 0.88

Available soil NH4-N = 35 kg N/ha (average of 3 mg/kg in 90 cm profile)

Nitrogen requirement per unit yield = 0.045 kg N/kg grain

Field Plot Techniques. Field plots were arranged in a randomized complete block design with 3 to 5 replications at all locations. Small plots (4 x 8 m) were used at locations 1 and 2. For locations 1 and 2, biosolids were weighed and applied manually, and grain was harvested from a 1.5 x 7 m swath in the center of each plot with a Hegi small plot combine. Field-scale plots were used at locations 3, 4, 5, and 6. Biosolids were applied with a John Deere 455 Hydropush manure spreader to 2.5 ha plots. Grain was harvested from one or two combine header widths in the center of each plot (about 1.2 ha).

Field Location Information (Table 3). Soils at all locations were developed under grasslands (Haploxerolls great group), with a silt loam surface horizon susceptible to wind erosion. All locations were managed using a wheat/fallow production system (one year of crop, one year of fallow in a two year crop production cycle). Biosolids were applied in the fall after crop harvest (Locations 2, 4, and 5) or during the summer fallow (Locations 1, 3, and 6).

Tillage, wheat variety, herbicide and other cultural practices were those routinely used by the cooperating grower. Wheat varieties at Locations 1, 2, 3, and 4 were common soft white wheat varieties, with moderate straw strength, selected primarily because of their resistance to snow mold. Varieties at locations 5 and 6 were soft white club wheats with greater straw strength (less susceptible to lodging). The field locations represent two counties in central Washington (Locations 1-4) and two counties in eastern Washington (locations 5 and 6). Locations 1 and 2 were in the same field (about 75 m apart). Locations 3 and 4 were located in nearby fields (about 1 km apart). Location 1 was harvested in 1992, locations 3 and 6 in 1993, and locations 2, 4, and 5 were harvested in 1994.

Grain testing. Grain yields are reported on an "as is" basis (about 10 % moisture content). Subsamples of the grain harvested from each plot were collected in the field. Grain was cleaned to remove chaff and grain test weight (bulk grain weight per unit volume) was determined with a standard USDA grain grading apparatus. Grain nitrogen was determined via a LECO combustion analyzer. Grain N uptake was calculated by multiplying the grain N content by the corresponding grain yield (minus 10 % moisture).

Soil sampling and analysis.
Preapplication soil samples (0-90 cm) were collected immediately prior to biosolids application. Surface (0-30 cm) samples were analyzed for pH, Olsen (bicarbonate) P, and exchangeable K; nitrate-N was determined for the 0-90 cm depth (Table 3).

Postharvest samples for determination of nitrate-N, ammonium-N and sulfate-S were collected during the late summer or early fall following crop harvest, prior to significant precipitation. Soil samples were collected in 30 cm depth increments to a depth of 90 cm (Locations 1, 2, and 5) or 120 cm (Locations 3, 4, and 6), using a hydraulic auger (Kauffman sampler; Albany, OR) mounted on a small tractor. Approximately 10 to 15 cores (0-30 cm depth) and 3 to 6 cores (30-60, 60-90, and 90-120 cm depths) were collected from each plot. For conversion of soil concentrations to kg per hectare, we assumed a soil bulk density of 1.3 g/cm^3 at all locations. Soil nitrate was determined via a colorimetric method or a cadmium reduction method. Soil reference samples run with each batch of unknown samples showed both nitrate analysis methods to be accurate and precise. Soil sulfate-S was determined via a turbidimetric method. Surface (0-30 cm) samples were analyzed for ammonium-N via a colorimetric method.

Postharvest soil samples (0-10 cm) were analyzed for Olsen (bicarbonate) P and DTPA Zn. Twenty to 30 cores were composited per plot.

Above-ground biomass and grain yield components were determined at locations 4 and 5 on 2 m of row per plot. At harvest, we counted the number of spikes per m of row and weighed intact plants. Grain yield was determined by running the harvest bundles through a Hegi combine.

Statistical analyses were performed using the SAS system (SAS Institute, Cary, N.C.). We did not have the same number of biosolids treatments at each location, and biosolids analyses varied between locations. To compare similar biosolids N rates across locations we grouped application rates of 200 to 400 kg N/ha and called this treatment "300 kg N/ha" and grouped higher rates of application 500 to 700 kg N/ha and called this treatment "600 kg N/ha". We used these treatment groups, 300 and 600 kg N/ha as biosolids, for orthogonal contrasts across locations. The combined data set (all locations) was analyzed as a split-plot design with locations as main plots and fertilizer treatments as subplots. Statistical analyses within a location were performed using analysis of variance procedures for a randomized complete block design.

Results and Discussion

Comparisons across locations (Table 4).
Statistical analysis using locations as main plots, and fertilizer treatments as subplots showed highly significant ($P < 0.01$) effects of fertilizer application on grain yield, test weight, grain N, grain N uptake and postharvest soil nitrate-N. The fertilizer treatment * location interaction was also highly significant ($P < 0.01$) for all parameters (Table 4), indicating that the magnitude of fertilizer treatment effects varied between locations.

The mean squares generated by the analysis of variance (Table 4) show the relative importance of sources of variation. Location had a larger effect on yield and N uptake than did fertilizer treatment, showing the unpredictable nature of yield where precipitation is the major limiting factor. Fertilizer treatments had a stronger effect than location on grain N, postharvest soil nitrate-N, and grain test weight. The consistency of these measurements across locations shows that biosolids was a dependable nutrient source. The lower test weight across locations was due to increased vegetative growth and increased plant water stress during the grain fill period.

Averaged over locations, biosolids at 300 kg N/ha and inorganic N fertilizer at 55 kg N/ha had the same yield and postharvest nitrate-N. Biosolids at 300 kg N/ha significantly ($P < 0.05$) reduced test weight and increased grain N compared to inorganic N fertilization. Increasing the biosolids application rate from 300 to 600 kg N/ha significantly reduced grain yield, test weight, and increased grain N and postharvest nitrate-N.

Table 4. Biosolids effects on grain yield, grain quality, and postharvest soil nitrate-N. Averages, significance, and analysis of variance across all locations

Fertilizer Source	Total N applied per hectare	Grain Yield	Grain Test Wt.	Grain N	Grain N Uptake	Postharvest Soil Nitrate-N
	kg/ha	Mg/ha	kg/m3	g/kg	kg/ha	kg/ha
None	0	3.56	766	15.8	52.5	38.5
Anhydrous/aqua ammonia	55	3.90	770	17.9	64.2	51.6
Biosolids	300	4.09	761	20.5	77.6	64.9
Biosolids	600	3.53	747	22.9	72.0	143.1
Source of variation	df	P>F				
Locations	4	**	**	**	**	**
Fertilizer Treatments	3	**	**	**	**	**
Fertilizer Trt. * Location	12	**	**	**	**	**
Coefficient of variation (%)		10.78	1.4	8.96	14.08	40.88
LSD (5 %)		0.25	6.70	1.08	5.85	19.63
Source of Variation	df	Analysis of Variance Mean Squares				
Blocks	4	0.5	222	4	98	1628
Locations	4	14.9	1940	59	8510	10510
Main plot error	3	0.4	153	6	108	1489
Fertilizer Treatments	3	1.4	2006	198	2387	44485
Fertilizer Trt. * Location	12	0.8	881	16	545	2954
Subplot Error	47	0.2	114	3	88	936

Note: Location 4 deleted from averages and ANOVA because it lacked a 300 kg N/ha biosolids treatment

** Significant at the 1 % probability level

Table 5. Statistical significance of grain and soil N measurements. Locations 1-6.

Contrast	Field Location					
	1	2	3	4	5	6
	Yield, Mg/ha					
Zero N vs. biosolids (300 kg N/ha)	**	NS	+	**	**	NS
Anhdrous/aqua ammonia vs. biosolids (300 kg N/ha)	NS	NS	NS	NS	NS	NS
Biosolids (300 kg N/ha) vs. biosolids (600 kg N/ha)	NS	**	**	M	NS	*
Coefficient of variation (%)	8.6	8.2	14.5	9.3	7.2	7.9
Standard error of the mean	0.23	0.37	0.73	0.30	0.25	0.27
	Grain test weight, kg/m3					
Zero N vs. biosolids (300 kg N/ha)	+	NS	*	+	**	*
Anhydrous/aqua ammonia vs. biosolids (300 kg N/ha	NS	NS	**	**	NS	NS
Biosolids (300 kg N/ha) vs. biosolids (600 kg N/ha)	NS	*	*	M	**	NS
Coefficient of variation (%)	1.4	1.4	2.3	1.2	0.4	0.6
Standard error of the mean	10.9	10.5	17.3	9.3	2.9	4.9
	Grain N, g/kg					
Zero N vs. biosolids (300 kg N/ha)	**	+	**	**	**	NS
Anhydrous/aqua ammonia vs. biosolids (300 kg N/ha	**	NS	**	**	**	NS
Biosolids (300 kg N/ha) vs. biosolids (600 kg N/ha)	**	**	NS	M	**	NS
Coefficient of variation (%)	7.2	9.5	10.9	7.1	3.9	10.2
Standard error of the mean	1.33	1.89	2.43	1.34	0.69	1.82
	Grain N Uptake, kg/ha					
Zero N vs. biosolids (300 kg N/ha)	**	NS	**	**	**	NS
Anhydrous/aqua ammonia vs. biosolids (300 kg N/ha	**	NS	**	*	**	NS
Biosolids (300 kg N/ha) vs. biosolids (600 kg N/ha)	**	+	**	M	NS	+
Coefficient of variation (%)	10.3	11.6	16.5	13.4	8.2	13.0
Standard error of the mean	4.68	9.45	16.40	7.73	4.57	7.23
	Postharvest soil nitrate-N, kg/ha					
Zero N vs. biosolids (300 kg N/ha)	NS	NS	NS	**	*	NS
Anhydrous/aqua ammonia vs. biosolids (300 kg N/ha	NS	NS	NS	**	*	NS
Biosolids (300 kg N/ha) vs. biosolids (600 kg N/ha)	**	**	*	M	**	**
Coefficient of variation (%)	24.7	30.9	71.4	11.1	38.9	23.7
Standard error of the mean	12.2	20.8	50.1	4.9	33.7	20.9

M = missing data makes this contrast impossible

Note: For Location 4 contrasts, zero N or aqua/anhydrous ammonia treatments compared with biosolids at rate of 600 kg N/ha.

+,*,** Significant at the 10, 5, and 1 % probability levels, respectively

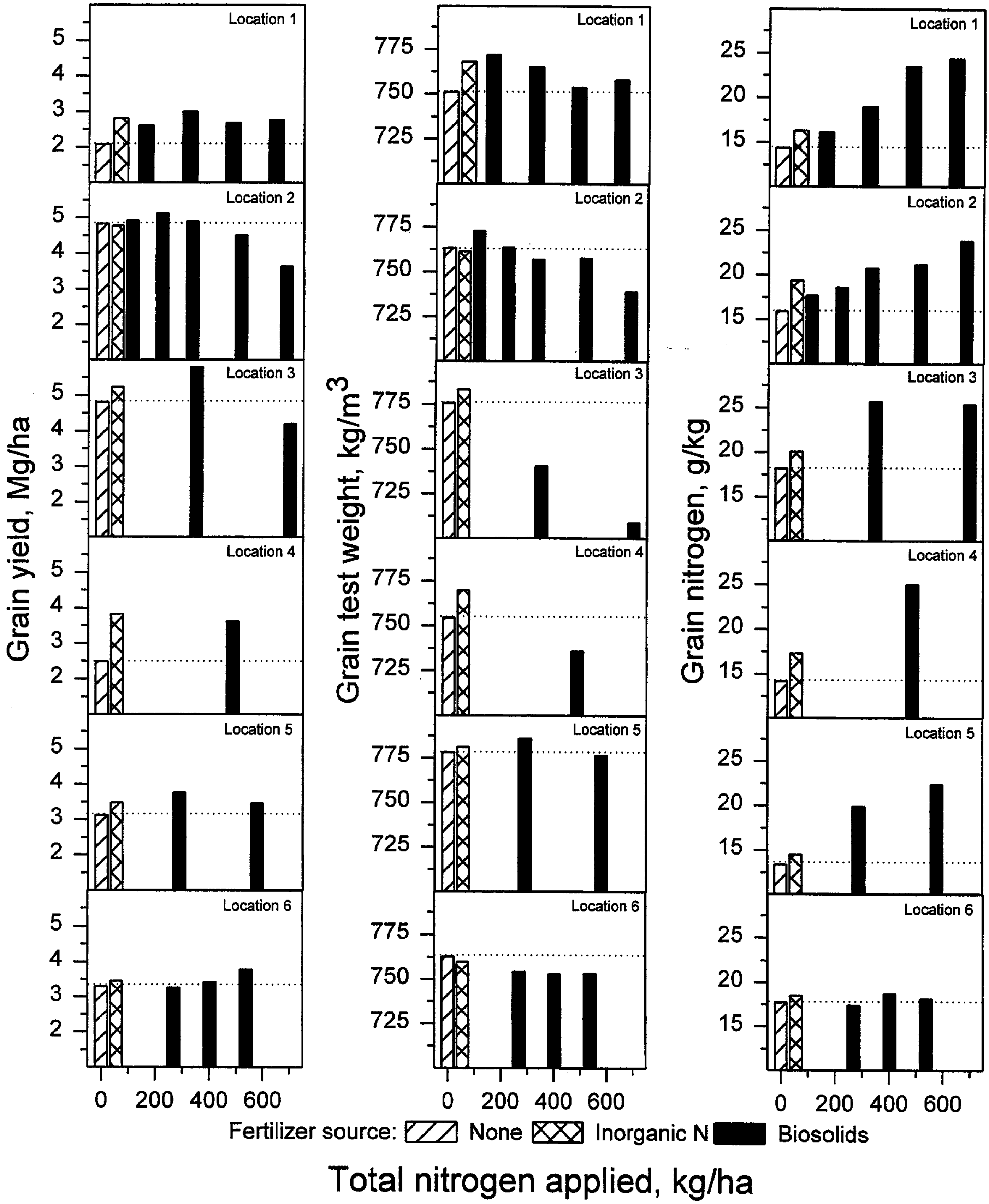


Figure 1. Biosolids and inorganic N fertilizer effects on grain yield, grain test weight and grain nitrogen. Horizontal dotted lines equal values for zero fertilizer treatment.

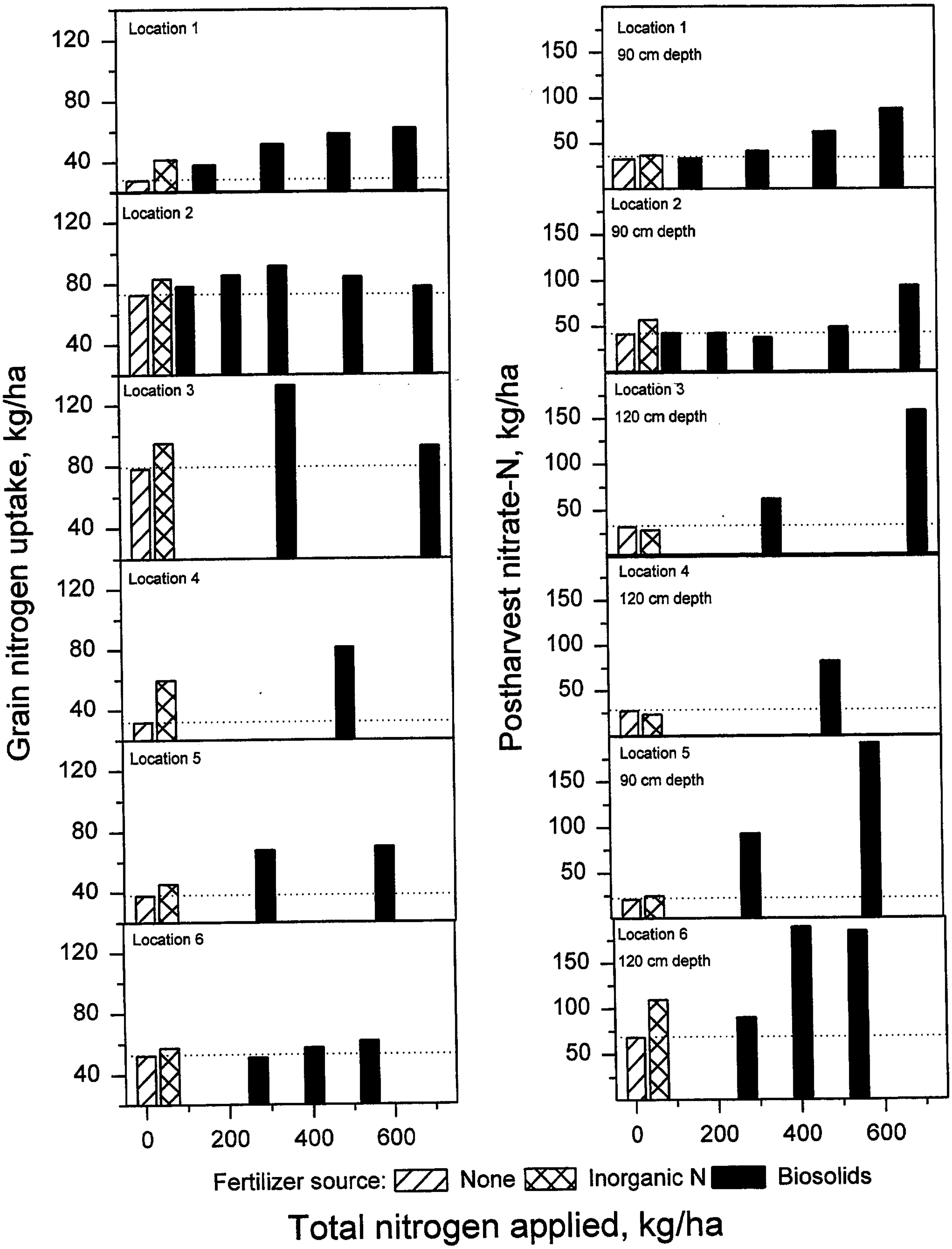


Figure 2. Biosolids and inorganic N fertilizer effects on grain nitrogen uptake and post harvest soil nitrate-N. Horizontal dotted lines equal values for zero fertilizer treatment.

Comparisons by location (Fig 1 and 2; Table 5).

Grain yield. Preapplication nitrate-N analyses (90 cm depth) showed that all sites except location 6 should respond to N fertilization (Table 3). Other nutrients tested were present at adequate levels. Soil P (0 - 30 cm) was well above (locations 1 and 5) or slightly above (locations 3, 4, and 6) the recommended minimum soil P level for maximum yield of dryland winter wheat (10 mg/kg Olsen P). Exchangeable K was well above recommended minimum (100 mg/kg). Surface soil pH values ranged from 6.3 to 6.7 (near neutral).

Fertilization significantly increased grain yield at 4 of the 6 locations (Table 5; Fig. 1). Yield responses were largely due to the N supplied by biosolids. The locations that did not respond to fertilization had high preapplication soil nitrate-N analyses (Table 3). The WSU fertilizer guide for dryland wheat correctly predicted non-responsive sites; it recommended application of reduced N rates (location 2; 28 kg N/ha) or no N (location 6). Locations that did respond to fertilization (locations 1, 3, 4, and 5) had estimated fertilizer N requirements of 52 to 73 kg N/ha for a target yield of 4 Mg/ha (60 bushels per acre).

Using the current WSU guidelines for biosolids application, suggested biosolids application rates for the responsive sites were 3.9 to 6.5 Mg/ha (Table 2). This is below the minimum application rate for most manure spreaders currently used to apply biosolids, and below the biosolids application rates used at our sites (except location 1 and 2). Actual yield responses (Fig. 1) show that low (<300 kg N/ha) rates of biosolids were needed for maximum yield at all locations except location 6. The cause of the significant yield increase at the highest rate of biosolids application at location 6 is unknown. Preapplication and postharvest soil tests at location 6 showed high nitrate levels, so the yield response was probably not due to nitrogen.

The yield data demonstrated that higher rates of N application were not needed in years with above average yield potential. Locations 1 and 2 (same field, different harvest years) both produced near-optimum yields with 300 kg N/ha biosolids applied, although maximum yields were 3.0 Mg/ha for Location 1 and 5.0 Mg/ha for Location 2. Locations 3 and 4 (adjacent fields, different harvest year) had similar increases in yield due to fertilization (approximately 1.2 Mg yield increase over the zero N treatment), although maximum yields were 5.8 Mg/ha for Location 3 and 3.8 Mg/ha for Location 4.

The largest yield reductions at high biosolids application rates occurred when wheat plants lodged (fell over). We observed some lodging at the 300 kg N/ha rate and severe lodging at the 600 kg N/ha rate at locations 2 and 3. Some lodging was also observed at Location 4 with the 600 kg N/ha rate. Locations 2, 3, and 4 had the same wheat variety, Eltan, which has only moderate straw strength. Locations 5 and 6 had the potential for lodging (high N supply and above average precipitation), but had wheat varieties with greater straw strength. Location 1 had a variety even more susceptible to lodging than Eltan, but low precipitation during the crop year limited excessive growth.

Test weight (Fig.1; Table 5). Test weight is a measure of grain plumpness. Decreases in test weight are usually associated with plant water stress during the grain fill period. Lodging and excessive vegetative growth increase plant water stress. Large reductions in grain test weight were associated with lodging at high biosolids application rates (Locations 2, 3, and 4). Where lodging did not occur (Locations 1, 5 and 6), even high rates of biosolids produced grain with acceptable test weight. Low rates of biosolids increased test weight at locations 1 and 5, probably because of improved root development and extraction of soil water.

Grain N (Fig. 1; Table 5). High grain N is undesirable in soft white winter wheat. High protein grain is less suitable for production of cakes, crackers, cookies and other confectionery items. Some export markets specify soft white wheat less with less than 18 g/kg N. Currently, there are no premiums paid to growers for low protein wheat, so high protein does not negatively affect the grower. Grain N increased with biosolids application rate at all locations except location 6. Excluding location 6, low N grain (<18 g N/kg) was produced only at biosolids application rates of less than 200 kg N/ha (Locations 1 and 2). Grain N levels continued to increase at locations 1, 2, and 5 at biosolids rates above that needed for maximum yield.

Grain N uptake (Fig. 2; Table 5). This measurement shows how much N is removed from the field at grain harvest. Because of lodging, grain N uptake was actually reduced at locations 2 and 3 as the biosolids rate increased from 300 to 600 kg N/ha. Grain N uptake with biosolids at 300 kg N/ha was higher than with inorganic N fertilization at locations 1, 3, and 5 demonstrating that biosolids supplied a greater quantity of N than the inorganic N fertilizer.

Postharvest nitrate-N (Fig. 2; Table 5) was similar for biosolids at 300 kg N/ha and inorganic N fertilizer at all locations except location 5. Postharvest nitrate-N increased significantly when the biosolids rate was increased from 300 to 600 kg N/ha at all locations. This was expected, since yield was maximized at biosolids application rates of 300 kg N/ha.

Biomass and Yield Components (Tables 6 and 7). Locations 4 and 5 showed changes in biomass production and yield components due to biosolids fertilization. The yield response to biosolids at Location 4 was largely due to increased tillering (more spikes per unit area). At Location 5, the biggest factor in the yield response was a greater number of kernels per spike. At both locations, yield increases from more spikes and/or larger spikes were offset by a lower grain kernel weight. Some of the light kernels produced with biosolids may have been blown out the back of the combine when the large plots were harvested. We got this idea by comparing the hand harvest yields with the combine yields. For the biosolids treatments, the grain yield determined by hand harvesting (Tables 5 and 6) was about 1 Mg/ha higher than for combine harvest (Fig. 1). For the zero N treatment, hand harvest and combine harvest gave similar yield estimates.

The differences in yield response pattern between Location 4 and 5 were probably strongly influenced by wheat variety. The Eltan variety used at Location 4 responded to increased biosolids application rates by producing large quantities of tillers and straw. In contrast, the Tres variety at location 5 did not increase tiller production above that produced with inorganic N fertilization, even with 900 kg N/ha applied as biosolids. Increased straw production is desirable

Table 6. Biosolids effects on biomass yields and grain yield components (Location 4).

Fertilizer Source	Total N Applied	Dry Matter Yield			Yield Components		
		Total	Grain	Straw	Kernels per spike	Spikes	Thousand Kernel Weight
	kg/ha	Mg/ha	Mg/ha	Mg/ha		per m2	g
None	0	6.9	2.4	4.5	25.0	288	33.3
Anhydrous/aqua ammonia	56	11.4	3.9	7.5	29.0	394	34.7
Biosolids	486	16.3	5.3	11.0	30.5	565	30.1
Biosolids	971	12.9	3.9	9.0	34.0	423	27.0
Contrasts:							
Zero N vs. anhydrous/aqua ammonia		NS	+	+	**	NS	NS
Zero N vs. biosolids		*	**	**	**	**	+
Anhydrous/aqua ammonia vs. biosolids		NS	NS	+	NS	NS	*
Coefficient of variation (%)		28.9	22.3	22.0	8.5	18.1	9.8
Standard error of the mean		1.12	2.65	1.76	2.52	75.65	3.07

+,*,** Significant at the 10, 5, and 1 % probability levels, respectively

Table 7. Biosolids effects on biomass yields and grain yield components (Location 5).

Fertilizer Source	Total N Applied	Dry Matter Yield			Yield Components		
		Total	Grain	Straw	Kernels per spike	Spikes	Thousand Kernel Weight
	kg/ha	Mg/ha	Mg/ha	Mg/ha		per m2	g
None	0	7.3	3.2	4.2	41.8	218	35.0
Anhydrous/aqua ammonia	56	9.1	3.9	5.2	43.6	258	34.7
Biosolids	289	10.2	4.3	6.0	53.3	268	30.2
Biosolids	578	10.0	4.4	5.6	57.1	257	30.1
Biosolids	867	10.6	4.5	6.1	54.7	288	28.8
Contrasts:							
Zero N vs. anhydrous/aqua ammonia		NS	NS	NS	NS	+	NS
Zero N vs. biosolids		**	**	**	**	**	**
Anhydrous/aqua ammonia vs. biosolids		NS	NS	NS	**	NS	**
Coefficient of variation (%)		13.6	14.8	16.0	7.4	10.4	4.2
Standard error of the mean		0.55	1.40	0.87	3.72	26.80	1.33

+,*,** Significant at the 10, 5, and 1 % probability levels, respectively

in terms of soil conservation and the opportunity for a straw harvest. It is a negative when it reduces grain yield by increasing plant water stress. Choosing the right variety is an important component of a program to beneficially use biosolids in dryland cropping systems.

Extractable soil nutrients (S, Zn, and P; Fig. 3 and Table 8).

Sulfate-S was significantly increased ($P < 0.10$) by biosolids fertilization at all locations. The increase in extractable sulfate-S with biosolids was equal to 3.1, 4.9, and 3.8 g S/kg biosolids at locations 3, 4, and 5, respectively. Based on this estimate of sulfate-S per unit of biosolids, the increase in soil sulfate-S after application of biosolids at 300 kg N/ha should be about 27 kg S/ha. Sulfur uptake by wheat grain is usually less than 10 % of the N uptake. Since grain N uptake at a high yielding dryland site is about 100 kg N/ha (Fig. 2), we would expect S removal to be less than 10 kg/ha. Therefore, the sulfate-S generated by application of biosolids at a rate of 300 kg N/ha should be enough to satisfy crop requirements for 2 or 3 cropping cycles, provided leaching losses are not significant.

Olsen (bicarbonate) P was significantly increased ($P < 0.01$) by biosolids fertilization at all locations. The depth sampled (10 cm) probably does not fully represent the depth of biosolids incorporation; additional P from biosolids application is probably present below 10 cm. Across locations, application of biosolids at a rate of 300 kg N/ha (175 kg P/ha) increased bicarbonate P from 20 mg/kg to 33 mg/kg. The additional P removed with biosolids fertilization at 300 kg N/ha was about 2 kg P/ha at locations 4 and 5 (data not shown). The P supplied by biosolids should be enough to satisfy crop requirements for many (10 + ?) crops.

Zinc (DTPA extraction) was significantly increased ($P < 0.01$) by biosolids fertilization at all locations. Even at high biosolids rates, extractable Zn levels in our study are far below levels toxic to plants. The increase in plant-available Zn probably does not represent a major soil fertility benefit. Wheat is very tolerant of low soil Zn availability, and yield response to applied Zn on wheat has not been demonstrated in our area. Application of high rates of P fertilizer sometimes result in a crop Zn deficiency. Phosphorus induced Zn deficiency appears highly unlikely with biosolids application, since both nutrients are applied simultaneously

Summary and Conclusions

Anaerobically-digested, dewatered biosolids are a dependable nutrient source for dryland cropping systems. Adequate N for maximum yield was present at all locations with a biosolids application rate of 300 kg total N/ha. A single application of biosolids supplied enough P, S, Zn for two or more crops. In addition to the benefits reported here, growers also observe significant benefits in reducing wind erosion from biosolids application. Long-term research (6 to 10 years after a single application) is needed to measure the residual effects of biosolids on site productivity.

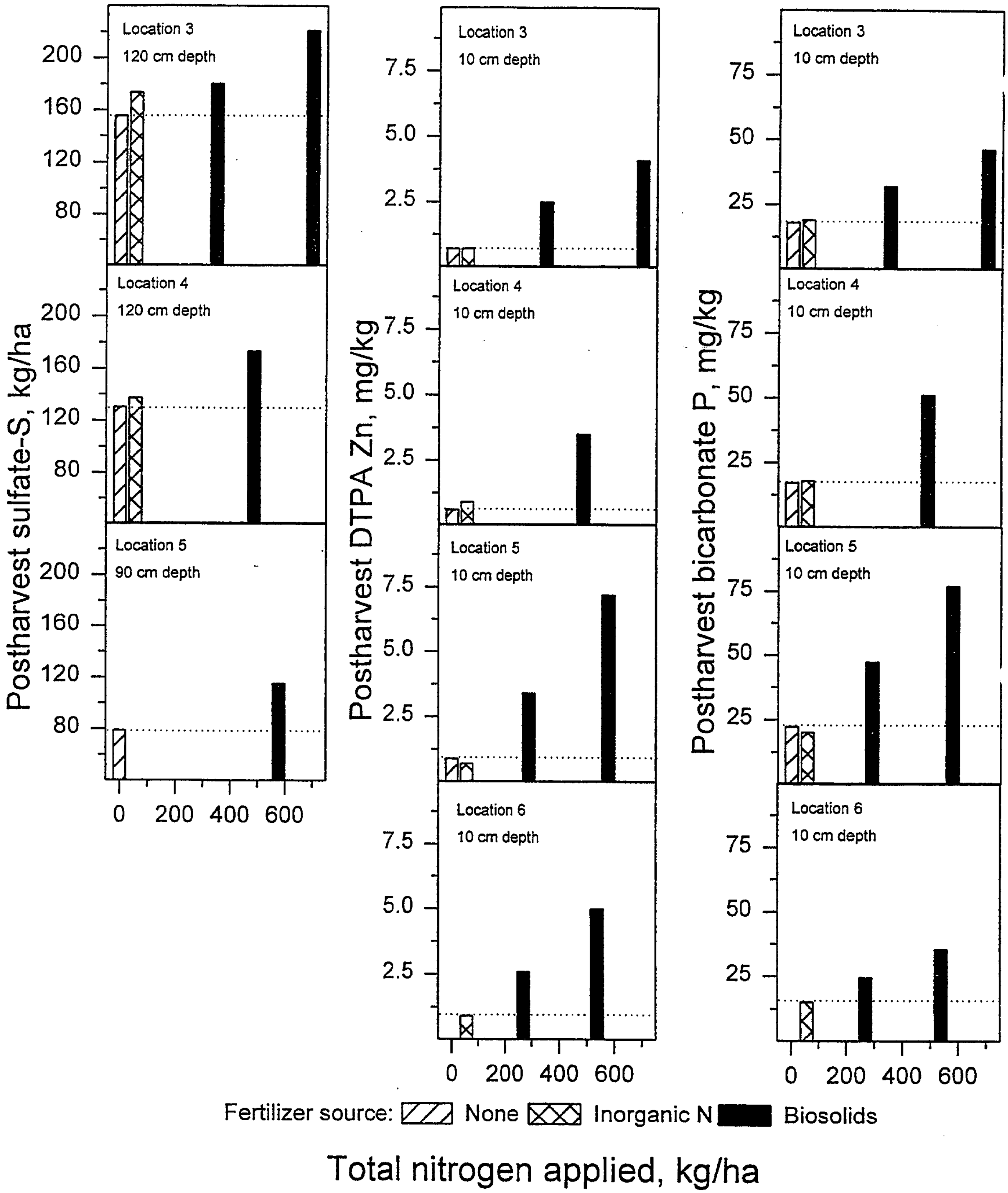


Figure 3. Biosolids and inorganic N fertilizer effects on postharvest soil sulfate-S, DTPA Zn and bicarbonate P. Horizontal dotted lines equal values for zero fertilizer treatment.

Table 8. Statistical significance of extractable soil nutrient measurements. Locations 1-6.

Contrast	Field Location			
	3	4	5	6
	Sulfate-S, kg/ha			
No biosolids vs. biosolids	+	+	*	M
Biosolids (300 kg N/ha) vs. biosolids (600 kg N/ha)	NS	M	M	M
Coefficient of variation (%)	20.4	13.7	14.5	M
Standard error of the mean	37.3	20.5	15.8	M
	Bicarbonate P, mg/kg			
No biosolids vs. biosolids	**	**	**	**
Biosolids (300 kg N/ha) vs. biosolids (600 kg N/ha)	+	M	**	*
Coefficient of variation (%)	27.2	35.0	12.3	19.6
Standard error of the mean	7.9	10.1	5.3	4.9
	DTPA Zn, mg/kg			
No biosolids vs. biosolids	**	**	**	**
Biosolids (300 kg N/ha) vs. biosolids (600 kg N/ha)	NS	M	**	**
Coefficient of variation (%)	54.3	45.4	22.6	18.8
Standard error of the mean	1.1	0.8	0.7	0.5

M = missing data makes this contrast impossible

+,*,** Significant at the 10, 5, and 1 % probability levels, respectively

A true "agronomic rate" (exactly matching N supplied with crop N needs) is probably about 5 Mg/ha biosolids (2 dry tons/acre) for our 25 to 35 cm precipitation zone. This would supply about 200 kg total N/ha. This application rate is below the spreading capability of most manure spreaders. A reasonable solution to this problem is to apply a spreadable biosolids rate and eliminate, or reduce N fertilization for succeeding crops, based on soil nitrate tests. After the first crop, we recovered most of the postharvest nitrate-N from the 0-60 cm depth (data not shown). Under our low precipitation conditions, this residual nitrate-N will be available for uptake by the second crop after biosolids application. An application rate of 7 to 10 Mg/ha (3 to 4 dry tons/acre) may provide enough N for 2 or 3 crops.

Grower confidence in biosolids fertilization is essential to continued biosolids utilization in dryland cropping systems. Varieties that do not lodge with high fertility are needed to ensure consistent yield performance with biosolids. In our study, we observed large reductions in grain yield and grain test weight associated with lodging. The high grain N produced by biosolids fertilization is a potential problem for soft white wheat production, but could be an asset for production of hard red winter or hard red spring wheat.

The greatest benefit to the grower from biosolids application comes from the first biosolids application at a site. Biosolids recycling programs designed to maximize beneficial use should concentrate on one-time applications over large acreages, rather than repeated applications on small acreages. Local biosolids permitting agencies can promote beneficial use by streamlining the permit process for dryland sites which have a very low risk of off-site pollution.

Acknowledgments

We wish to express our appreciation to the Northwest Biosolids Management Association for financial support. We thank cooperating wheat growers for their assistance with the field test plots. We thank Bob Stevens and Virginia Prest for their assistance during plot harvest. We thank Steve Fransen for assistance with statistical analysis.

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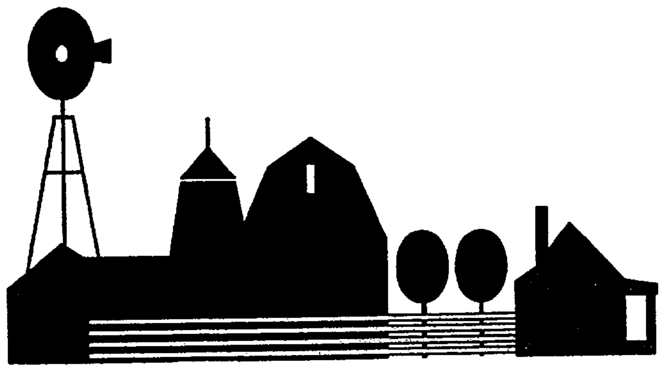
Biosolids

Biosolids

From An Agricultural Perspective



Gary Wegner
Wegner Ranch
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Reardan, WA 99029
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Gary Wegner, an Agricultural Perspective

As an American Farmer, my main concern is producing crops that grow fast and healthy, creating healthy food that tastes good and is good for you. To accomplish this, I need a healthy soil. A soil that is not deficient in nutrients. A soil that contains as much organic matter as possible. As an American Farmer, I can increase the organic matter content of my soil, by returning lots of plant residue (stems, leaves and roots) to the soil. When residue is mixed with the soil, it decays and becomes the part of the soil we call organic matter.

What determines how much residue my crops produce? The type of crop, the moisture available for the crop, and the availability of nutrients. A soil that has a shortage of one, or more nutrients, will not produce up to it's potential, even if all other factors are optimal. That is why the addition of Biosolids to my soil at Wegner Ranch has had such a dramatic affect.

- Organic matter is an important soil constituent that influences virtually all chemical, biological and physical processes that occur in the soil.
- Organic matter is the "glue" that holds soil together, so that it can resist wind and water erosion.
- Organic matter contributes to the "cation exchange capacity" of a soil, providing sites for the retention of basic cations such as potassium (K), calcium (Ca) and magnesium (Mg).
- Organic matter is an insoluble "store-house" of nutrients (food for plants), bit by bit the nutrients are taken from the store-house, and converted to soluble nutrients which are transported inside the plant to cells where they become building material for the plant.
- Mineralization, is the term used to describe this process, which is the microbial conversion of organic compounds to inorganic molecules that are available for plant uptake.

Recycling of Biosolids is the ***ultimate recycling program.***

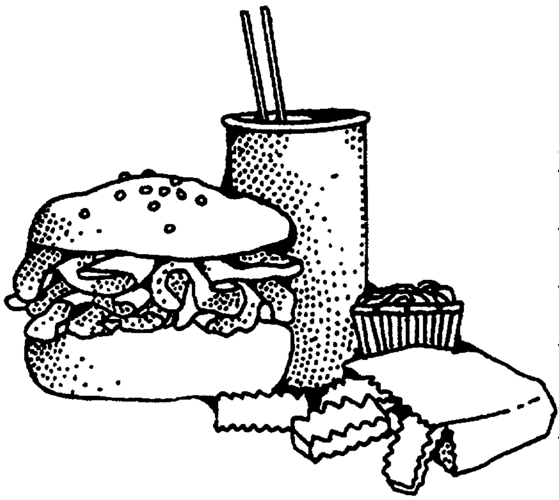
Here is how it works: Plants grow in soil, using energy from the sun (photosynthesis) and nutrients from the soil. Plants are either harvested for food, or eaten by animals. The animals that eat the plants, are themselves harvested for food. We find these nutrients (that were once in the soil) at the meat counter, in the cereal box that we buy at the supermarket, or in the steak that we enjoy at a fine restaurant. These nutrients we derive from the food we eat, serve to nourish our bodies.

Our bodies utilize the nutrients, and then shed what is normally called waste. *(Actually, "waste" is just excess nutrients, like nitrogen, and all the other nutrients that our cells cast off. ------- Cast off like spent fuel, just to be replaced with more nutrients and new sources of energy.)* These biologically processed nutrients (waste), move through the pipes of the wastewater system of a city, to a treatment facility. The facility could best be described as a "bacteria farm", where bacteria are fed, much like cattle in a feed lot. In the end, these beneficial bacteria make up what we call Biosolids. The Biosolids are hauled back to farm land, where they rejuvenate the soil, simply by returning vital nutrients. The soil is renewed, ready to grow lush crops for consumption by humans and animals, all over again!

Do you agree that Biosolids Recycling is, without a doubt, the ***ultimate recycling program?***

It is important know more about <u>Essential Nutrients:</u>

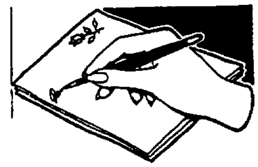
The "heavy metal group":



Cr	Chromium
Co	Cobalt
Cu	Copper
Fe	Iron
Mn	Manganese
Mo	Molybdenum
Ni	Nickel
V	Vanadium
Zn	Zinc

The "other" Essential Nutrients:

B	Boron
Ca	Calcium
C	Carbon
Cl	Chlorine
H	Hydrogen
K	Potassium
Mg	Magnesium
N	Nitrogen
Na	Sodium
O	Oxygen
P	Phosphorus
Se	Selenium
S	Sulfur



The Nutrient Quiz

1. A deficiency of ____________can affect as many as 60 enzymes and most major metabolic processes in the human body. (Human Nutrition)

2. ____________primary role is to carry oxygen and carbon dioxide within the red blood cell from one body tissue to another. (Human Nutrition)

3. Photosynthesis can produce carbohydrates from CO_2 and H_2O, but the process cannot go on to the production of proteins, nucleic acids, and so on unless ____________________is available. (Soils & Soil Fertility)

4. There's mounting evidence that wheat growers are overlooking a simple, inexpensive crop input that could return an investment two to three fold. That input is___________, a crop nutrient that may be in short supply on more than 70% of wheat fields in the High Plains. (Farm Journal 5/94)

5. In its biologically active form, _______________helps insulin metabolize fat, turn protein into muscle and convert sugar into energy.

6. _________________is a mineral that helps maintain the balance of fluid within the cells of the body. It also aids in nerve conduction and helps maintain normal blood pressure. (Encarta)

7. ____________helps maintain the balance of fluid outside the cells of the body. It also aids in nerve conduction and muscle contraction. (Encarta)

8. Sources of ________________include milk products and dark-green leafy vegetables (such as broccoli). (Encarta)

9. ____________________deficiency is extremely rare because most foods contain the mineral. (Encarta)

10. About 90 percent of __________________is stored in bone, where it can be reabsorbed by blood and tissue. (Encarta)

11. ________________is a macronutrient constituent of organic compounds and, like nitrogen and phosphorus, is an integral part of their structures. (Soils & Soil Fertility)

12. _________________is involved in chlorophyll formation and is important as a coenzyme that is needed to activate several plant enzymes. (Soils & Soil Fertility)

These steptoe barley plants are shown at 50% of their actual size.

They were 37 days old. These samples were taken 25 feet from each other.

What was the difference?

The plant on the right received the recommended agronomic rate of Biosolids.

The plant on the left, received the normal amount of nitrogen fertilizer used at Wegner Ranch.

Plants respond to the full spectrum of essential nutrients found in Biosolids.

Plant and animal growth is limited by the availability of water, sunlight and nutrients. Any nutrient can be limiting. A deficiency of one nutrient, can affect the utilization of another nutrient.

At Wegner Ranch we know that our soils were all deficient in Zinc. Agronomists tell us that Zinc affects the utilization of Phosphorus. That is just one of the reasons that we get a very significant response to Biosolids applications.

Gary Wegner, an Agricultural Perspective

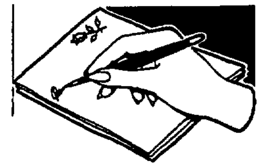
It's only human nature to resist change. As a third generation, American Farmer, I know my family has been faced with many changes. When my Grandfather sold his last horse, he had made some serious decisions. He had kept just a few of his best horses back, just in case those new tractors didn't work out. Driving a tractor was a lot different than holding leather reins for 14 hours a day. **It is human nature to resist change.** But good choices can mean a better future, or what we call progress.

I began trying to get Biosolids in 1986. I met resistance from my father (my partner), then my County Commissioners and some neighbors. With the help of a farsighted County Commissioner, we formed the Lincoln County Soil Safety Committee. Applications of Biosolids began in the Spring of 1988. My father saw the benefits and became supportive, as did many others. My cousin, an Agronomy Professor at Iowa State University, visited our farm and encouraged our use of Biosolids. We were cautious, and we were careful, but it is has been the most beneficial decision we have ever made.

We are concerned about the soil and we are concerned about the crops that we grow. That is why we use Biosolids.

We can see the soils and the crops improve every year. Extra residue, producing additional organic matter, feeding better varieties of crops, to produce better food for your table. That is just good farm management. That is the result of a good Biosolids Recycling Program.

- Gary Wegner is a graduate of Washington State University, with a B.S. Degree in Animal Nutrition.
- Wegner Ranch, 25 miles West of Spokane, Washington, and includes 1,525 acres of tillable land, devoted to wheat, barley, canola and hull-less barley production.
- Prior to returning to his family farm, Gary was a Farm Management Instructor at Centralia College, a developer of training aids for the Farm Credit Banks and a sales representative for the Ralston Purina Company.



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7. Sodium helps maintain the balance of fluid outside the cells of the body. It also aids in nerve conduction and muscle contraction. (Encarta)
8. Sources of Calcium include milk products and dark-green leafy vegetables (such as broccoli). (Encarta)
9. Phosphorus deficiency is extremely rare because most foods contain the mineral. (Encarta)
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11. Sulfur is a macronutrient constituent of organic compounds and, like nitrogen and phosphorus, is an integral part of their structures. (Soils & Soil Fertility)
12. Copper is involved in chlorophyll formation and is important as a coenzyme that is needed to activate several plant enzymes. (Soils & Soil Fertility)

N-Viro Soil as a Gypsum Replacement in Cotton Production on a Sodic Saline/Alkali Soil

B. McCullough-Sanden, H. McCutchin, R. Bailey, T. Logan, and B. Harrison

Introduction

N-Viro Soil (NVS) is produced by the N-Viro advanced alkaline stabilization process for treatment of municipal sewage sludge. The material is a dry, granular, pasteurized product with an alkaline pH (12), calcium carbonate equivalency of about 50 % (comprised mostly of $CaCO_3$ and a small amount of $Ca(OH)_2$), and NPK content of about 1, 0.5 and 0.5-1 %, respectively. It has been used as an agricultural limestone substitute, and as a soil substitute for reclamation, horticulture and landfill cover. Because of its stable structure, high organic matter content and relatively high content of soluble Ca, it appeared to be a possible substitute for gypsum in the reclamation of sodic soils.

In the U.S. there are 16 western states with alkaline/sodic soils that could potentially use significant quantities of N-Viro Soil (NVS) in agriculture. West of Nebraska, there is one N-Viro facility, and it produces only landfill cover. Mike Scharp prepared a 1993 report on NVS marketing for Kern County, California for N-Viro International Corporation. Scharp found that the data required to market NVS needed development. ScharpÕs conclusion was that NVS is not going to fit into its regularly accepted markets that have been developed in the east and midwest; for example, as a lime replacement.

California's San Joaquin Valley, a crop growing capital of the world, has saline/alkaline soil. Gypsum is applied in order to farm these areas. Soluble calcium in the gypsum replaces sodium on exchange sites in the soil and maintains water penetration. This addition of soluble calcium improves the soil physical properties of water infiltration and aggregation. In 1992, in Kern County, 144,796 tons of gypsum were purchased and used for this purpose. Within the state of California, 846,905 tons of gypsum were purchased and applied. More than twice as much gypsum was applied agriculturally in California as compared to lime. Based on this background information it was decided to do a trial to test the usefulness of NVS as an alternative to gypsum. This research forms a block in a solid foundation of knowledge of the performance of NVS on sodic/alkaline soils that could open additional markets in the Western states and other areas like Israel that have sodic/alkaline soils with water penetration problems.

Gypsum has been used for many years as a Ca^{2+} source to replace Na^+ from the soil exchange complex. This exchange generally increases infiltration rates and improves physical structure of sodic soil. The reason that gypsum ($CaSO_4.2H_2O$) is applied in comparison to agricultural limestone ($CaCO_3$) is that it contains more soluble calcium. The concentration of soluble calcium in NVS and gypsum was compared and found to be similar (around 2.4%, Table 1). Since NVS is additionally a source of low level nutrients, organic matter and has moisture holding properties, it is a product that shows promise of improved performance over that of the several hundred thousand tons of gypsum now being sold.

The project involved the cooperation of Dr. Richard Bailey, Western Region, N-Viro International Corp.; Hal McCutchan, Agricultural Project Engineer, Residuals Processing, Inc.; Blake Sanden, Farm Advisor, University of California Cooperative Extension; Professor Terry Logan, School of Natural Resources, Ohio State University; and others. Because soluble calcium levels were found to be comparable in the gypsum and NVS samples it seemed feasible to replace the function of gypsum with NVS. A field project was initiated and, with the assistance of Blake Sanden, a cooperating farm that typically applied gysum to the soil to grow cotton was located.

Study Objectives

1. To determine the value of NVS for reclamation/infiltration as a replacement for bulk gypsum in a production field crop setting on a saline/sodic/alkali soil.

2. To determine if soil concentrations of heavy metals were measurably increased above native levels.

3. To access crop benefits from the addition of nutrients and organic matter in NVS.

Study Description and Testing

Preparation and Testing of NVS

Bakersfield ash from Hondo Chemical Company was tested by The Medical College of Ohio (MCO) and found to be an adequate admixture souce to insure adequate heat generation to achieve PFRP. Admixture metal data was reviewed to be sure that the material met N-Viro/503 standards. MCO mixed the Hondo ash with biosolids from the Hyperion Treatment Plant in Los Angeles to produce NVS. A trial NVS production was processed on 1/10/94 near the study site.

The prepared NVS was sampled over time at eight intervals (Day 1 after preparation through Day 22). The samples were analyzed by the Environmental Science Lab at OSU for electrical conductivity, solids, $Ca(OH)_2$ (reported as a percentage of moist sample weight), and soluble Ca content (reported on a dry weight basis) (Table 1). A gypsum sample from American Aggregates was tested for soluble Ca and had a concentration of 2.4 %. Dry bulk density was 0.5 g/cm^3 and 1.15 g/cm^3 for the Day 22 NVS and for the Aggregates gypsum, respectively.

Southwestern Kern County is relatively close to the large urban areas of southern California. It has an extensive year round agricultural land base . The site was located on the southern boundary of the dry Buena Vista lakebed in southwestern Kern County. The soil type is Westhaven silt loam that is considered a thermic Typic Torrifluvent, that is rated Class II with irrigation and Class VII without irrigation.

Table 1. Soluble and total Ca, Mg, Na in amendments.

Amendments	Soluble			Total		
	Ca	Mg (mg/kg)	Na	Ca	Mg (mg/kg)	Na
Gypsum	20635	135	33	51470	6053	52
NVS	28706	32	52	110188	4497	168

Study design

Treatments were randomized within a block and replicated four times. Pre-application soil samples were taken to a depth of 1 foot on 2/294. All soil samples were taken with an open face Lord-type probe. Eight to ten soil plugs were taken from each plot and composited before analysis.

Three treatments were used, gypsum, at 3.5 ton/acre (representative of a standard amendment practice), NVS, at 4.5 ton/acre, and a control (no amendment added). All rates are reported on a fresh weight basis. The treatments were applied volumetrically based on bulk density values and disced into the soil on 2/11/94.

The field was preirrigated on 3/3/94 and planted on 4/4/94. From 4/22 to 5/20 the weather was much cooler than normal and an unusally high amount of rainfall occurred (1.5 in) which resulted in a poor stand (55 - 65% of desirable plant numbers). The combination of osmotic stress due to high levels of soil salts and cool temperatures enabled soil borne fungal pathogens (possibly thielaviopis, pythium, rhizoctonia) to cause the death of a large percentage of the young cotton seedlings. The effect was random and not associated with any treatment. The southwest corner of the farm (including more than half of the study) appeared to be more affected than the rest of the field. Neighboring farms had similar problems. The first irrigation occurred on 6/8/94 and pressure bomb determinations of leaf water potential were taken on 6/22 along with mid-season soil samples. Samples were composited among plots by treatment for determination of saturation extract and total metal content (Tables 2 and 3). The samples were tested for the metals listed in Title 40 CFR Part 503 (1993) and California Title 22. This compositing by treatment did not allow statistical analysis on the soil metals data. Standard salinity analyses for Western soils were completed in duplicate and analyzed statistically.

Three sets of plant mapping data were collected from all plots on 8/8/94, 8/22/94, and 9/9/94 as a comparison of plant growth. Leaf petioles were collected at the same time and analyzed for NO_3-N, PO_4-P, K, Mg, Ca, Zn, and Cu.

During the growing season a total of seven irrigations were completed; all irrigations were made by gated pipe with tailwater recycled to a following set or a lower field. The first three irrigations were applied in alternate furrows with sets of 18 to 24 hours in duration. The last four irrigations were applied to every furrow for a set time of 12 hours. On 8/13/94 the seventh and final irrigation was applied. At that time, a modified two-point furrow evaluation was performed on three of the four replicates for each treatment. This evaluation enabled the development of advance curves, infiltration functions and mean infiltration. The worst ÔsealingÕ conditions for sodic soils usually occur at the end of the season making this the optimal time to test for residual treatment benefits.

Table 2. Factors influencing reclamation/infiltration on a saline/sodic/alkali soil. Soil samples (0-12 in) taken on 2/2/94 and 6/22/94,
before and after amendment application. Mean and standard deviation of 4 replications.

Treatment	pH				Elec. cond. (dS/m)				SAR			
	2/2/94	SD	6/22/94	SD	2/2/94	SD	6/22/94	SD	2/2/94	SD	6/22/94	SD
Control	7.8	0.1	8.0	0.1	6.5	1.8	6.8	2.2	12.6	2.8	11.4	4.6
Gypsum	7.8	0.1	8.0	0.1	6.7	1.5	7.8	1.4	12.8	3.4	11.8	1.9
N-Viro	7.7	0.1	8.0	0.1	6.2	1.8	7.3	0.8	11.0	3.3	11.4	2.0

Treatment	ESP				Boron (mg/kg)				Cl (mg/kg)			
	2/2/94	SD	6/22/94	SD	2/2/94	SD	6/22/94	SD	2/2/94	SD	6/22/94	SD
Control	14.7	3.0	13.2	4.6	5.1	2.3	10.0	5.3	569	389	438	254
Gypsum	14.9	3.6	13.9	2.1	4.3	2.4	9.0	3.8	678	360	831	303
N-Viro	12.9	3.7	13.4	2.3	3.2	0.7	9.4	4.7	368	193	473	162

Table 3. Irrigation efficiencies for final irrigation on 8/13/94. All values are the mean of three replications of each treatment.

Treatment	Distribution uniformity (%)	*	Tailwater Runoff (%)	*	Irrigation Efficiency (%)	*	Max. infiltration 12 hour set (inch)	*	Mean infiltration over furrow (inch)	*
Control	0.93	a	0.47	a	0.24	a	0.55	a	0.55	a
Gypsum	0.99	a	0.68	a	0.32	a	0.78	a	0.78	a
NVS	0.99	a	0.76	a	0.53	a	1.41	b	1.34	b

Distribution uniformity = % low quarter average/field average infiltration
Tailwater runoff = % total applied water running off end of field
Irrigation efficiency = % applied water beneficially used without tailwater reuse

* = Duncan's New Multiple Range Test for statistical significance at the .05 level

Timeline Recap of 1994 Season

Date	Activity
2/2	Flag field plots and collect pre-treatment soil samples
2/11	Apply N-Viro Soil and gypsum to plots
2/14	Materials disced into soil
3/3	Preirrigate field
4/4	Plant field
4/22	Install neutron probe access tubes in every plot. Plants starting first true leaf. Subsoil saturated at 4-6Õ depth -- perched water
4/22-5/20	Weather much cooler than normal, 1.5Ó rain
5/23	Stand only 55 - 65 % of normal
6/8	First irrigation
6/22	Second set of soil samples, install tensiometers, commence pressure bomb readings
6/25	Second irrigation
7/7	First plant mapping
7/9	Third irrigation
7/20	Fourth irrigation
7/21	Apply Zephyr for spider mite control
7/28	Fifth irrigation
8/5	Sixth irrigation
8/8	Second plant mapping, pull petiole samples for tissue analysis
8/13	Seventh and final irrigation, conduct irrigation/infiltration evaluation
8/22	Pull second petiole samples
9/9	Pull third petiole samples
9/14	Apply sodium chlorate and Gramoxone for defoliation
9/16	Third and final plant mapping
9/30	Harvest of field
10/4	Harvest the four western most plots of trial. Grower failed to notify before harvest of other plots.

Final yields were disappointing due to the poor stand in the study area. Since yield data could be collected on only four plots out of the twelve, yield comparisons in relation to treatment effects could not be made.

Results and Discussion

Salinity analyses

The salinity parameters did not seem to be affected by the treatments. Neither the gypsum nor the NVS treatment caused an increase or decrease in soil chloride concentrations, electrical conductivity, pH, ESP or SAR (Table 2). The salinity threshold for yield decline is commonly cited as 7.7 dS/m, and the plot mean among treatments for the 6/22/94 sample analysis was 7.3 dS/m (Table 2). Some parameter values showed a marginal (not over 20%) increase during the interval between soil sampling events due to capillary rise of salts and the lessening of water infiltration. Boron was an exception in comparison to the other measurements and concentrations nearly doubled on all plots during the time period between soil sampling dates. The toxicity threshold for boron is considered to be between 6-15 mg/kg and the 6/22/94 sample analysis had values of around 9 mg/kg (Table 2). The high soil boron and electrical conductivity values might have caused some plant toxicity that helps to explain the cause of the poor plant stands in the field study.

Irrigation evaluations
A significantly higher infiltration rate was found in the NVS treatment for the final irrigation on 8/13/94. A power advance function was determined by measuring in-field advance for three points in three replications of each treatment. Advance in the control treatment was linear, meaning that water only filled minor cracks and then ceased to infiltrate. Furrow bottoms were quite flat by this time in the season and an average depth and width were determined to estimate water storage in the furrow. This storage volume within the furrow coupled with the power advance was applied to a border-type volume balance model to generate the infiltration function. Infiltration was computed along the furrow length according to the advance and the effective Ôon timeÕ for that segment of the furrow, thus allowing for development of distribution uniformity, tailwater runoff, and irrigation efficiencies. Mean infiltration over the entire furrow length for the NVS treatment was more than 1 to 1.5 times that of gypsum and more than twice that of the control at 1.34, 0.78, and 0.55 inches, respectively, for a 12 hour set (Table 3). No significant differences between treatments were found for distribution uniformity, tailwater runoff, and irrigation efficiencies (Table 3).

The infilration difference between treatments is highly significant and unexpected this late in the season. That the infiltration in the gypsum plot declined to the level of the control is reasonable as the dissolution of this material is fairly rapid early in the season and no longer available to displace incoming sodium. Similarly, free calcium from the NVS material should no longer be available. A plausible explanation for the increase in infiltration with the NVS treatment is that organic compounds within the NVS (from the sludge component), and structural stability of the NVS itself, are acting as long-term binders to maintain aggregate stability over the season.

Trace element concentrations
Total trace element concentrations and concentrations in saturated extracts were determined on the gypsum and NVS amendments and on the control and unamended soil (Table 4). Concentrations of all elements were low in gypsum and NVS, with only Cu, Pb and Zn in NVS at concentrations much above background soil. NVS concentrations were well within the 503 concentration limits for EQS sludge. Because of the low trace element concentrations in the amendments, and the low application rates, there were no increases in total soil concentrations above background. Only Cu and Zn concentrations in saturation extracts of NVS were elevated compared to gypsum or control soil, but application of either amendment had no effect on concentrations of trace elements in saturation extracts. Low trace element concentrations in NVS and the low rate of NVS application for sodic soil reclamation suggest that trace element accumulation above normal background levels is not anticipated.

Plant mapping and tissue analysis
The plant mapping data taken in July, August and September showed no significant differences

between treatments for plant height, vegetative and fruiting node, and plant height to node ratio (Table 5). The final mapping on 9/16/94 showed a general decrease in plant height of 2.9 to 3.7 inches when compared to the 8/8/94 data. Desiccation due to salinity stress is the most likely cause as the last irrigation was applied 8/13/94.

A series of petiole analyses taken 8/8, 8/22, and 9/9 (Table 6), corresponding to the period of maximum fiber elongation, yielded no significant difference with respect to NO_3-N, PO_4-P, K, Zn, and Cu. Tissue nitrate levels were barely in the sufficiency range for all sampling periods (Basset and MacKenzie, 1978, de Tarr, 1994). Potassium was nearly double minimum levels at 7-8 %. All others were easily within the sufficiency guidelines with the exception of Cu which fell to the minimum in the 9/9/94 sampling.

Table 4. Total and saturated extract metals (mg/kg) for gypsum, N-Viro Soil, and composite soil samples (0-12").

			Total					
			Pre-application soil 2/2/94			Post-application soil 6/22/94		
Metal	Gypsum	NVS	Control	Gypsum	NVS	Control	Gypsum	NVS
Antimony	<3.6	<27.2	<3.5	<3.5	<3.5	0.5	0.6	0.7
Arsenic	4.7	3.9	10.6	11.8	10.4	8.7	9.2	9.3
Barium	47	551	158	149	158	137	140	124
Beryllium	<0.6	<4.5	1.2	1.1	1.2	1.2	1.1	1.2
Cadmium	<0.6	<9	<0.6	0.8	0.7	0.3	0.3	0.3
Chromium	7.8	41.3	47.3	43.2	46.5	4.7	4.2	4.5
Cobalt	3.6	9.4	15.1	14.2	14.3	13.4	14.1	12.6
Copper	6.9	149.0	22.8	22.5	20.8	26.3	23.9	41.0
Lead	<3.6	51.8	28	23.4	26.2	11.4	11.4	11.2
Mercury	<0.1	0.7	<0.1	<0.1	<0.1	0.1	<0.1	<0.1
Molybdenum	<1.2	<9	<1.2	<1.2	<1.2	<1.2	<1.1	<1.1
Nickel	4.9	40.9	34.6	33	32.8	31	32.1	29.8
Selenium	<0.4	<0.54	<0.4	<0.4	<0.4	1.4	1.3	1.4
Silver	<0.6	13	<0.6	<0.6	<0.6	0.2	0.2	0.2
Thallium	44.2	<45.4	<5.9	<5.9	<5.8	0.6	<0.5	0.5
Vanadium	17.4	49.3	59.1	54.2	60.4	47.2	50.9	43.2
Zinc	23.8	334	103	97.7	96.6	93.9	99.9	88.3
			Saturated Extract					
Antimony	<0.5	<0.5	<0.5	<0.5	<0.5	<0.03	<0.03	<0.03
Arsenic	<0.03	<0.03	0.16	0.14	0.14	1.3	1.1	1.3
Barium	<2.0	<2	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0
Beryllium	<0.1	<0.1	<0.1	<0.1	<0.1	0.01	0.008	0.009
Cadmium	<0.2	<0.2	<0.2	<0.2	<0.2	0.01	0.01	0.01
Chromium	<0.2	0.4	<0.2	<0.2	<0.2	0.1	0.08	0.08
Cobalt	<0.2	<0.2	0.2	0.2	0.2	0.2	0.2	0.2
Copper	<0.1	5.7	0.2	0.2	0.2	0.17	0.15	0.15
Lead	<2.0	<2	<2.0	<2.0	<2.0	1.1	1	1.1
Mercury	<0.005	<0.01	<0.005	<0.005	<0.005	<0.01	<0.01	<0.01
Molybdenum	<0.2	<0.2	0.2	0.2	0.2	<0.2	<0.2	<0.2
Nickel	<1.0	<1	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0
Selenium	<0.03	0.04	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03
Silver	<0.1	<0.1	<0.1	<0.1	<0.1	<0.01	<0.01	<0.01
Thallium	<1.0	1.7	<1.0	<1.0	<1.0	<0.04	<0.04	<0.04
Vanadium	0.2	0.7	0.2	0.2	0.2	0.2	0.2	0.2
Zinc	0.2	2.3	0.2	1.2	0.7	0.8	0.7	0.9

Table 5. Plant mapping data taken during 1994 growing season. Each value is the mean of four treatment replications.

Measurement	Date	Control	*	NVS	*	Gypsum	*
		Treatment					
Plant height (in)	7-Jul	20.9	a	21.0	a	22.0	a
	8-Aug	27.2	a	27.8	a	28.2	a
	16-Sep	25.3	a	23.5	a	25	a
No. of veg. nodes	7-Jul	7.4	a	10.1	b	10.1	b
	8-Aug	8.0	a	7.2	a	8.0	a
	16-Sep	12.9	a	12.6	a	12.8	a
No. of fruiting nodes	7-Jul	7.2	a	5.7	a	5.2	a
	8-Aug	10.9	a	12.1	a	11.2	a
	16-Sep	7.8	a	7.8	a	8.0	a
Total nodes	7-Jul	14.6	a	15.9	b	15.2	ab
	8-Aug	18.9	a	19.2	a	19.2	a
	16-Sep	20.6	a	20.4	a	20.8	a
Height to node ratio	7-Jul	1.4	b	1.3	a	1.4	b
	8-Aug	1.4	a	1.4	a	1.5	a
	16-Sep	1.2	a	1.2	a	1.2	a

* = Duncan's New Multiple Range Test for statistical significance at the .05 level

Table 6. Cotton petiole tissue analysis for samples taken during the 1994 growing season. Each value is the mean of four treatment replications.

Measurement	Date	Control	*	NVS	*	Gypsum	*
		Treatment					
NO3-N (mg/kg)	8-Aug	2794	a	2312	a	3520	a
	22-Aug	2124	a	2835	a	2277	a
	9-Sep	868	a	1575	a	1500	a
PO4-P (mg/kg)	8-Aug	1378	a	1274	a	1299	a
	22-Aug	1436	a	1526	a	1419	a
	9-Sep	1211	a	1271	a	1014	a
K (mg/kg)	8-Aug	7	a	8	a	8	a
	22-Aug	6	a	6	a	6	a
	9-Sep	6	a	7	a	6	a
Zn (mg/kg)	8-Aug	33	a	42	a	28	a
	22-Aug	28	a	20	a	30	a
	9-Sep	21	a	21	a	19	a
Cu (mg/kg)	8-Aug	15	a	17	a	17	a
	22-Aug	8	a	7	a	7	a
	9-Sep	6	a	6	a	5	a

* = Duncan's New Multiple Range Test for statistical significance at the .05 level

Conclusions

The application of NVS at 4.5 fresh t/ac (2.47 t/ac dry weight) in comparison to the application of a locally available mixed-grade mined gypsum applied at 3.5 t/ac (2.95 t/ac dry weight) proved to be a superior amendment with respect to maintaining infiltration late in the season. Even though the flyash used to produce NVS is a highly alkaline material, buffering by the sludge component of NVS and the high background level of resident salts resulted in negligible changes in alkalinity in this field for all treatments. No significant differences or even trends were found with respect to changes in SAR, other salts, or heavy metals that were due to applied treatments. Increasing salinity, boron and chlorides during the season due to capillary rise of perched water was the dominant factor and appeared to mask any benefit that might have accrued due to improved infiltration in the NVS or gypsum treatments. Thus, no significant differences were found with respect to plant nutrition, stature, and possibly yield.

Long-term additions of NVS would build soil reservoirs of phosphorus, potassium and some micro nutrients along with free Ca^{++} and may provide superior reclamation on sodic soils not impacted by major drainage problems.

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References

Basset, D.M. and A.J. MacKenzie. 1978. Plant analysis as a guide to cotton fertilization. Soil and Plant Tissue Testing in California. Univ. of CA, Bull 1879. pp.16-17.

de Tarr, W. 1994. Personal communication. USDA/Univ of CA Cotton Field Station, Shafter, CA.

USEPA. 1993. Standards for the use and disposal of sewage sludge. Title 40 Code of Federal Regulations Part 503. pp. 703-706.

A Comparison of Production of Dryland Wheat Between Use of Anhydrous Ammonia and Biosolids In Southeastern Colorado

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Introduction

The USEPA (1989) reports that the application of biosolids to agricultural land is the most common use/disposal method in current practice. Surveys of biosolids use/disposal practices among wastewater treatment facilities in Colorado indicate a substantially higher rate of land application than is practiced nationally. An estimated eighty-five percent of the state's biosolids production is beneficially utilized as a soil conditioner/fertilizer. Agricultural land application represents an attractive use option due to the proximity of agricultural land to front range population centers. Additionally, the quality of biosolids generated in the state is typically high; instances of biosolids containing substantial concentrations of contaminants are infrequent. Agricultural use on land cultivated in dryland winter wheat is the predominant mode of beneficial use in Colorado. In recent years the state has experienced an influx of biosolids from out-of-state facilities for land application. Biosolids generated at several wastewater treatment plants serving the New York City area were transported and applied to cropland in southeast Colorado between April, 1992 and December, 1994. Within that period some 29,373 metric tons (34,402 short tons) of biosolids were applied, primarily to land cultivated in dryland winter wheat.

The potential movement of heavy metals present in land applied biosolids represents a minimal risk to public health or to the environment, particularly if the biosolids are low in metal content and the biosolids application rate is dictated by the nitrogen requirement of the crop (USEPA, 1983). Alkaline soil conditions further limit mobility of most potentially toxic trace metals found in biosolids (Logan and Chaney, 1983). Evaluation of long term biosolids application to cropland cultivated in dryland winter wheat (Lerch et al., 1990a, Barbarick et al., 1992) support these earlier findings and have demonstrated the sustainability of biosolids application. The eight year study conducted on experimental sites located near Bennett, Colorado has also illustrated the economic viability of biosolids use as an alternative to commercial fertilizer (Lerch et al., 1990b, Barbarick et al., 1992)). The importation and application of New York biosolids in Colorado provided an opportunity to validate earlier findings under production agriculture conditions.

The overall objective of the study was to compare the effect of New York biosolids and commercial anhydrous ammonia on dryland winter wheat grain production. Data was also generated to allow analysis of the uptake of trace metals present in the biosolids by the crop.

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Materials and Methods

There are thirteen wastewater treatment facilities serving New York City. The Colorado Department of Public Health and Environment, Water Quality Control Division, and the U. S. Environmental Protection Agency, Region VIII, in early 1992, authorized the application of biosolids from ten of those facilities to sites in Kiowa and Prowers Counties, Colorado. Biosolids were surface applied to the study sites during the period between October, 1992 and July, 1993 and were incorporated as the soil was worked prior to seeding. Biosolids samples were collected at the time of application. Average biosolids composition is given in Table 1.

Table 1. Characteristics of biosolids applied to dryland winter wheat fields.

Parameter	Field 1	Field 2	Field 3	Field 4	Limit[1]
N, total, %	6.22	2.35	1.95	2.08	
NH_4-N, %	0.96	1.13	0.65	0.55	
NO_3-N, %	0.003	0.002	0.003		
P, %	2.29	0.84	0.32	0.70	
K, %	0.14	0.09	0.10	0.07	
Cd, mg/kg	11.6	7.9	8.7	8.4	85
Cr, mg/kg	176.3	180.0	223.3	227.0	3000
Cu, mg/kg	1653.3	1036.8	1146.7	1150.0	4300
Fe, %	2.1	1.6	1.7		
Pb, mg/kg	262.5	345.8	375.0	304.0	840
Mo, mg/kg	6.5	12.1	17.8	11.0	75
Ni, mg/kg	28.4	71.2	93.8	74.0	420
Zn, mg/kg	2041.7	1197.5	1223.3	1120.0	7500

[1] Colorado and federal regulatory maximum concentration for land application.

The biosolids applied comply with the maximum constituent limits contained in the Colorado Biosolids Regulation (CDPHE, 1993) and in federal regulations at 40 CFR Part 503 (USEPA, 1992) for land application of biosolids. Both regulations establish a two tiered system of concentration limits. Biosolids which comply with the more stringent set of limitations (Colorado Grade I or federal Table 3 standards) have fewer restrictions placed upon their use.

The New York biosolids met the more stringent state and federal concentration limits with the exception of lead which periodically exceeded the Grade I limit of 300 mg/kg. Nonetheless, all lead analyses fell well within the regulatory maximum lead concentration for land application of 840 mg/kg.

Four fields were identified which had received biosolids during that period and which were seeded in winter wheat (*Triticum aestivum* sp.) during the fall of 1993. Winter wheat cultivation in Colorado is practiced using a dryland summer fallow system. The crop is sown in the fall months and harvested the following spring. The land then remains fallow for approximately eighteen months, allowing precipitation to replenish the soil moisture. Biosolids application may take place at any point within the fallow period.

Each field was paired with a field which had received applications of anhydrous ammonia prior to seeding with winter wheat. Soils at each site are predominantly Wiley silt loam. Each set of paired fields was selected based upon the proximity of the paired fields, wheat cultivar, and cultural practices (each set of paired plots was farmed by the same individual producer). Biosolids application rates (Table 2) were determined for each field based upon background soil nitrogen concentration and percent organic matter content (Follett, et al., 1991).

Yield data, soil, grain and straw samples were collected at harvest in June, 1994. Yield data was determined by combine harvesting of 1.2 X 15.2 meter (4 X 50 foot) plots (Table 2). Soil, grain and straw samples were collected from 7.9 X 30.5 meter (26 X 100 foot) plots located within each set of paired fields. Three individual grain, straw and soil samples were collected from each plot. All samples were analyzed by the Colorado Soil Testing Lab, Colorado State University. Soil samples were collected from the uppermost 30.5 centimeters (12 inches). Soil sampling results are reported in Table 3. Results of grain and straw analyses are given in Table 4.

Table 2. Effects of biosolids and anhydrous ammonia on yield and protein content of wheat grain.

	Biosolids applied			Anhydrous applied	
Fields	application rate[1]	yield[2]	protein content[3]	yield[2]	protein content[3]
1	5.0	34.3	18.9	33.6	18.1
2	2.2	27.0	14.4	31.7	15.0
3	3.1	42.3	15.0	41.8	14.8
4	2.4	47.3	15.6	43.5	15.3

[1] Metric tonnes/hectare
[2] Bushels/acre
[3] %

Table 3. Effects of biosolids and anhydrous ammonia on elemental composition of soil at harvest.

	Biosolids applied		Anhydrous applied	
Parameter[1]	total[2]	available[3]	total[2]	available[3]
K	7616.9	841.5	6662.3	788.1
P	663.9	6.0	672.3	4.1
Cd	1.2	0.1	1.2	0.1
Cu	19.3	4.6	17.1	3.5
Cr	11.7	0.2	10.7	0.2
Fe	22283.9	5.1	21242.3	3.9
Pb	9.2	2.2	8.8	1.7
Mo	0.5	0.1	0.5	0.1
Ni	15.2	0.8	14.3	0.7
Zn	63.5	0.8	56.9	0.4

[1] mg/kg
[2] HNO_3-$HClO_4$
[3] AB-DBTA extraction

Results and Discussion

Grain yields for the four paired fields are presented in Table 2. Although fields to which biosolids had been applied exhibited higher yields in three of the four paired sets of fields, no significant yield response was indicated. Lerch et al. (1990), and Barbarick et al. (1992) report that dryland hard red winter wheat yield does not typically vary to a significant degree between experimental plots fertilized with commercial nitrogen fertilizer and biosolids applied plots. Both commercial nitrogen fertilizer and biosolids applied at agronomic rates (6.7 metric tons/hectare or 3.0 tons/acre) will consistently result in higher yields when compared to non-fertilized control plots (Barbarick et al., 1992).

Grain protein contents for the four paired fields are also presented in Table 2. Protein content was greater in grain harvested from the biosolids treatment in three of the four paired sets of fields. Grain protein was higher for the biosolids applied fields in the same three sets of paired fields which experienced greater yields. As with the yield data, however, no significant difference was found between the biosolids treatment and the anhydrous ammonia treatment. Studies conducted at the Bennett, Colorado site (Lerch et al., 1990b, Barbarick et al., 1992) indicate a linear relation between increased wheat grain protein content and increased biosolids and commercial nitrogen application rates. Barbarick reports consistently higher protein content in wheat cultivated on biosolids applied soils over crops fertilized with commercial nitrogen fertilizers (Barbarick et al., 1992).

Review of the biosolids application data for the individual fields (Table 2) reveals that the three fields which exhibited higher yields and protein contents than the corresponding, anhydrous treated fields received higher whole biosolids applications than the paired field set where the anhydrous application resulted in a higher yield and grain protein content. The biosolids application rate was determined for each field based upon the plant available nitrogen content of the biosolids at the time of biosolids application, and the residual nitrogen (as nitrate) available in the soil and the soil organic matter content prior to biosolids application. Thus, the biosolids application rates were adjusted so that the amount of plant available nitrogen was similar for each field. Biosolids contain significant amounts of phosphorus and plant micro-nutrients: copper, iron, and zinc (Table 1). The application of these constituents at the greater biosolids application rates ranges from 109 to 227 percent of those added at the lowest biosolids application rate.

The addition of macro- or micro-nutrients other than nitrogen, either individually or in combination, may be responsible for the observed differences in yield and protein content. The total (HNO_3-$HClO_4$ extractable) and plant available (AB-DPTA extractable) elements in the soil at harvest indicates generally higher soil concentrations of both constituent forms in the biosolids applied soils when compared to those fields receiving the anhydrous ammonia treatment (Table 3). Any differences, however, are not statistically significant. Long term biosolids application, at application rates ranging from 6.7 to 40.4 metric tonnes per hectare (3.0 to 18.0 short tons/acre) per cropping cycle has been shown to result in significantly increased soil concentrations of both total and available forms of zinc and copper as well as other metals (Lerch et al., 1990a, Barbarick et al., 1992). Continued biosolids application may show significant increases in soil concentrations of micro-nutrients and other elements.

Analyses of grain (Table 4) indicated that the concentration of several micro-nutrients present in greater concentration in the biosolids (iron and zinc) were higher in the crops raised on biosolids amended soils. Several other elements were present in higher concentration in the grain cultivated on anhydrous applied sites. No statistically significant difference between elemental grain content for crops cultivated on the biosolids applied as opposed to those raised on the anhydrous applied fields was observed. Elemental analyses of wheat straw also indicated higher concentrations of iron and zinc for biosolids fertilized crops. Like the grain samples, however, no statistically significant difference was observed for those elements or for any other elements analyzed. Barbarick et al. (1993) reported significantly higher concentrations of copper and zinc in grain cultivated on biosolids applied plots when compared to control plots, particularly at higher (26.9 metric tonnes/hectare or 12 short tons/acre) biosolids application rates. Significant differences were not consistently found at lower (6.7 metric tonnes/hectare or 3.0 short tons/acre) application rates. The concentrations of all of the elements analyzed, in both grain and straw, were well below levels which could represent a potential hazard to livestock or humans. This is consistent with results reported by Barbarick et al, (1993).

Table 4.	Effects of biosolids and anhydrous ammonia on elemental content of wheat grain and straw.			
	Biosolids applied		Anhydrous applied	
Parameter[1]	wheat	straw	wheat	straw
N, %	2.9	0.3	2.5	0.3
P	2984.0	230.4	2815.0	244.2
Cd	0.5	1.8	0.5	0.6
Cr	0.6	0.5	0.7	0.5
Cu	5.9	2.3	6.1	2.1
Fe	44.7	76.7	40.6	73.4
Pb	2.5	2.5	2.7	2.5
Mo	0.6	0.5	1.1	1.1
Ni	1.8	0.6	1.5	1.0
Zn	23.6	3.8	22.4	3.7

[1] ppm except as noted

Conclusions

Land application of biosolids to dryland winter wheat fields in southeastern Colorado may result in increased grain yield and grain protein content when compared to commercially available anhydrous ammonia. These increases are possibly due to elements present in the biosolids and subsequently available in the soil following biosolids application which are crop micro-nutrients.

The addition of heavy metals in the biosolids, at application rates limited by agronomic crop requirements, did not result in significantly higher concentrations of those parameters in either grain or straw.

References

Barbarick, K. A., R N. Lerch, J. M. Utschig, D. G. Westfall, R. H. Follett, J. Ippolite, R. Jepson, and T. M McBride. 1992. Eight years of application of sewage sludge to dryland winter wheat. Technical Bulletin TB92-1, Colorado State University, Department of Agronomy, Cooperative Extension Service. Fort Collins, CO.

Colorado Department of Public Health and Environment, Water Quality Control Division. 1993. Biosolids Regulation. Denver, CO.

Follett, R. H., P. N. Soltanpour, D. G. Westfall, and J. R. Self. 1991. Guide to fertilizer recommendations in Colorado. XMC-37. Colorado State University, Department of Agronomy, Cooperative Extension Service. Fort Collins, CO.

Lerch, R.N., K. A. Barbarick, D. G. Westfall, R. H. Follett, T. M McBride, and F. W. Owen. 1990a. Sustainable rates of sewage sludge for dryland winter wheat production: I. Soil nitrogen and heavy metals. J. Prod. Agric. 3(1):60-65.

Lerch, R.N., K. A. Barbarick, D. G. Westfall, R. H. Follett, T. M McBride, and F. W. Owen. 1990b. Sustainable rates of sewage sludge for dryland winter wheat production: II. Production and income. J. Prod. Agric. 3(1):66-71.

Logan, T. J., R. L. Chaney, 1983. Utilization of municipal wastewater and sewage sludge on land-metals. p. 235-323. *In* A. L. Page (ed.) Proc. 1983 Workshop on utilization of municipal wastewater and sewage sludge on land, Denver, CO. January. University of California, Riverside.

U. S. Environmental Protection Agency. 1983. Land application of municipal sludge. EPA 625/1-83-016. Office of Research and Development, Municipal Environ. Res. Lab. Cincinnati, OH.

U. S. Environmental Protection Agency. 1989. Standards for the Disposal of Sewage Sludge; Proposed Rule. Federal Register 54(23):5746-5902.

U. S. Environmental Protection Agency. 1992. Standards for the Disposal of Sewage Sludge; Final Rules. Federal Register 58(32):9248-9415.

Session III

The Effect of Biosolids Application on Native Shortgrass Prairie Vegetation and Soils

Edward F. Redente, Rebecca L. Harris-Pierce, and Kenneth A. Barbarick

Abstract

Three biosolids studies were initiated in 1991 in Larimer County, Colorado: 1) single applications of sewage sludge of 0, 2.5, 5, 10, 21, and 30 Mg ha^{-1} (0, 1, 2, 5, 10, 15 tons $acre^{-1}$), 2) single applications of composted sewage sludge of 0, 2, 4, 14, 21, and 32 Mg ha^{-1} (0, 1, 2, 6, 10, 15, tons $acre^{-1}$), and 3) co-application of three rates of alum (5, 11, and 22 Mg ha^{-1}) (2, 5, 10 tons $acre^{-1}$) with 10.5 Mg ha^{-1} (5 tons $acre^{-1}$) of sewage sludge. The goal of this study was to evaluate the effects of biosolids application on native rangeland soils and vegetation and to determine if this is a viable recycling method.

The application of sewage sludge increased concentrations of NH_4-N and NO_3-N in both 1992 and 1993 at soil depths up to 30 cm (12 in) Electrical conductivity also increased in both years. Levels of P, Cu, Zn, Pb, Cd, Mo, and organic matter increased in the top 8 cm (3 in) of soil in 1993; probably a result of the breakdown of the sewage sludge and incorporation into the soil surface.

The application of biosolids produced a favorable response in total canopy cover, with perennial forbs and blue grama (*Bouteloua gracilis*) showing an increase in 1992. Total aboveground biomass did not exhibit a significant response, but perennial forbs and grasses, especially warm season grasses responded with increases in 1992 and 1993. Shrub biomass increased in 1992, almost entirely because of the favorable response of fringed sage (*Artemisia frigida*).

All four dominant species (*Bouteloua gracilis, Buchloe dactyloides, Agropyron smithii,* and *Artemisia frigida*) exhibited increases in tissue concentrations of N, P, and K for both years with increases in biosolids application rates. Plant tissue metal concentrations were variable and remained below phytotoxic and maximum livestock intake levels.

Introduction

The United States produces approximately six million metric tons of municipal sewage sludge annually (U.S. Environmental Protection Agency 1990). The sludge consists mainly of biodegradeable organic matter along with significant amounts of inorganic materials. Municipal sewage sludge contains all nutrient elements essential for plant growth (Melsted 1973). Measurable quantities of trace metals are also present, and are possible contaminants of the soil-plant-animal-human food chain. Recycling of sludge in an economically and environmentally acceptable manner that maximizes the beneficial aspects, and minimizes the adverse effects, presents a challenge.

Most municipal sludge is now disposed of by land application, incineration, landfilling, or lagooning. Land application, which is the controlled spreading of sewage sludge into or onto the soil surface, is becoming more popular as a sludge disposal option. Agriculture and mined-land reclamation efforts have successfully used sludge as a fertilizer and mulch (Berglund, et al. 1984; Catroux, et al. 1981; Sopper and Kerr 1979). A New Mexico study on a degraded grassland showed sewage sludge acted as a fertilizer and organic matter amendment (Fresquez, et al. 1990a). The study further showed that a one-time surface application of 22.5 to 45 Mg ha^{-1} (10 to 20 tons $acre^{-1}$) of anaerobically digested sewage sludge did not lead to contamination of soil and plant tissue (Fresquez, et al. 1990b, 1991).

In a rangeland study, Fresquez et al. (1990a,b) applied three rates (22.5, 45, and 90 Mg ha^{-1}) (10, 20, 40 tons $acre^{-1}$) of municipal sewage sludge on a degraded, semiarid grassland in New Mexico. Soil nutrients important to plant growth, including N, P, and K, increased linearly with the application of sewage sludge. With the exception of organic matter and pH, all other soil chemical properties, including electrical conductivity, were significantly higher in the 90 Mg ha^{-1} (40 tons $acre^{-1}$) sludge treatment than in the unamended control. Organic matter levels did not change with the application of sludge after one growing season. After two growing seasons, however, they reported higher organic matter levels in the low and intermediate sludge treatments as compared with the control. Soil pH decreased linearly with the application of sewage sludge. Total Cd and Pb levels in the soil did not change significantly, while several micronutrients, such as Cu, Mn, and Zn, increased as a result of sewage sludge application. Low and intermediate sludge application treatments resulted in higher plant yields than either the unamended control or the highest sludge application rate. Blue grama, in particular, responded significantly to the application of sewage sludge, with yields that nearly doubled.

Fresquez et al. (1990a,b) indicated that N, P, K, Cu, Mn, and Zn content of blue grama, galleta (*Hilaria jamesii*), and bottlebrush squirreltail (*Sitanion hystrix*) showed a linear increase with increases in sludge application rate. However, Cd and Pb did not increase with application rate because of low levels in the sludge itself. Other elements such as Al and especially Fe decreased with the application of sewage sludge, possibly because of a dilution effect associated with greater biomass production. Total plant foliar cover also increased linearly with the application of sewage sludge. The dominant species prior to soil amendments were broom snakeweed (*Gutierrezia sarothrae*), blue grama, and galleta. Blue grama density did not change with sludge amendment. Fendler threeawn (*Aristida longiseta*), ring muhly (*Muhlenbergia torreyi*), bottlebrush squirreltail, and especially broom snakeweed decreased in density with increasing sludge application rates after the first growing season. These species are generally indicators of a degraded rangeland and reduction in their densities can be considered an improvement of range condition.

The goal of our study was to evaluate the effects of sludge application on native rangeland soils and vegetation and to determine if this is a viable recycling method. Our specific objectives were to determine the effects on: 1) soil chemical properties, 2) plant species composition and production, and 3) plant tissue concentrations of nutrients and metals.

Methods

Experimental Design

The three studies we initiated in 1991 consisted of: 1) single applications of sewage sludge, 2) single applications of composted sludge, and 3) co-application of alum and sewage sludge. Each treatment plot measured 15m by 15m (50 ft by 50 ft) The single application of sewage sludge consisted of 6 treatments including 0, 2.5, 5, 10, 21, and 30 Mg ha^{-1} (0, 1, 2, 5, 10, 15 tons $acre^{-1}$). The composted sludge applications had 6 treatments including 0, 2, 4, 14, 21, and 32 Mg ha^{-1} (0, 1, 2, 6, 10, 15 tons $acre^{-1}$). The alum and sewage sludge experiment consisted of 10.5 Mg ha^{-1} (5 tons $acre^{-1}$) of sewage sludge in combination with application rates of alum of 5, 11, and 22 Mg ha^{-1} (2.5, 5, 10 tons $acre^{-1}$). All treatments for each experiment were replicated four times in a randomized complete block design.

Soils data

Baseline soils data were collected prior to the application of sludge. One composite soil sample (taken from five corings) was obtained from each treatment to a depth of 30 cm (12 in) These samples were separated into three depths: 0-8 cm (0-3 in), 8-15 cm (3-6 in), and 15-30 cm (6-12 in). Each sample was analyzed for pH, electrical conductivity (Workman et al. 1988), % organic matter, particle-size distribution, NO_3-N and NH_4-N (Keeney and Nelson 1982), and total P, K, Fe, Mn, Cu, Zn, Na, As, Cd, Cr, Pb, Hg, Mo, Ni, and Se. Sampling and analyses were repeated (except for particle-size distribution) in 1992 and 1993. Nutrient and trace element concentrations were determined by digesting in 4N HNO_3 (Bradford, et al. 1975) and using Inductively Coupled Plasma Optical Emission Spectrometer (ICP).

Vegetation data

The vegetation was sampled in August 1991, July 1992 and July 1993. Each treatment plot was sampled for cover and aboveground production by species, as well as plant-tissue nutrient and metal concentrations of the four dominant plant species. Cover measurements by species were made using the point method involving 50 individual points along a randomly placed transect and included the amount of ground covered with litter and rock. Aboveground production was measured by harvesting vegetation occurring within a rectangular 0.5 m^2 quadrat placed in a stratified-random manner.

Plant-tissue samples were taken of the four dominant plant species, blue grama, buffalograss (*Buchloe dactyloides*), western wheatgrass, and fringed sage, from each treatment in the three application studies. Aboveground plant samples were collected, washed in deionized water, oven-dried, ground, digested in concentrated nitric acid (Havlin and Soltanpour 1980), and analyzed for total P, K, Fe, Mn, Cu, Zn, Na, As, Cd, Cr, Pb, Hg, Mo, Ni, and Se using the ICP. The total N concentration was determined in H_2SO_4-H_2O_2 digests (Thomas et al. 1976), by a Lachat Flow Injector.

Statistics

Vegetation and soil data from 1992 and 1993 for the sewage sludge, compost, and alum sludge studies were analyzed using a standard randomized complete block analysis (Steel and Torrie 1980). Analysis of variance along with linear and quadratic regressions were performed using SAS programming language. Parameters were considered significant at $P \leq 0.05$ and $r^2 \geq 0.4$.

Results

Soil concentrations of NH_4-N and NO_3-N increased with biosolids (sewage and composted sludge) application rates in both 1992 and 1993 at all three soil depths (Harris-Pierce 1994). However, nitrogen concentrations were in general lower in 1993 than in 1992. Electrical conductivity (EC) also increased at all depths in both years as the application rate increased. Phosphorus, Cu, Zn, Pb, Cd, Mo, and organic matter all increased with application rate in the top 8 cm (3 in) of soil in 1993.

Total canopy cover increased in 1992 and 1993 with the application of sewage and composted sludge (Harris-Pierce 1994) (Figures 1 and 2). This increase was attributable to the addition of nutrients, particularly N. Perennial forb cover increased with application rates in 1992. Although forb cover also increased in 1993 the treatments were not significantly different from the control. Blue grama cover increased in 1992; the reponse in 1993 was generally not as large as in 1992 and was not significant.

Total biomass did not show a significant response to increases in biosolids application rates (Figures 1 and 2). There were increases recorded for specific species and lifeforms but not for overall biomass. As biosolids application rates increased there were corresponding increases in blue grama (1992 and 1993), western wheatgrass (1993), fringed sagebrush (1992), perennial forbs (1992 and 1993), and shrubs (1992).

Tissue concentrations of N, P, and K within the four dominant species (western wheatgrass, fringed sage, blue grama, and buffalograss) increased in 1992 and 1993 as a result of biosolids application (Harris-Pierce 1994). The plant uptake of other micronutrients and metals varied among species. The response was not consistent but concentrations remained below phytotoxic and maximum livestock intake levels.

The addition of alum with sewage sludge produced very little response in soils and vegetation (Harris-Pierce 1994). Alum applied in this manner did not appear to have any effect, beneficial or adverse.

Discussion

The application of sewage sludge provides all nutrients required for plant growth, especially N. Application rates may be dictated by the ability of plants to utilize N and reduce the potential for N leaching. In addition to the accumulation of N in soils and groundwater, another hazard in applying sewage sludge to land is the buildup of trace metals in the soil. High concentrations of trace metals can lead to reduced plant growth and to high metal concentrations in plant tissues that then enter the food chain.

The organic components of sludge decompose to CO_2, water, and residual soil organic matter. The soluble inorganic constituents are leached away by drainage waters, while insoluble products accumulate in the soil and become part of the soil matrix. Water percolating through the soil carries soluble salts, mainly Na, K, Ca, Mg, and NO_3. The major exchangeable cations in the

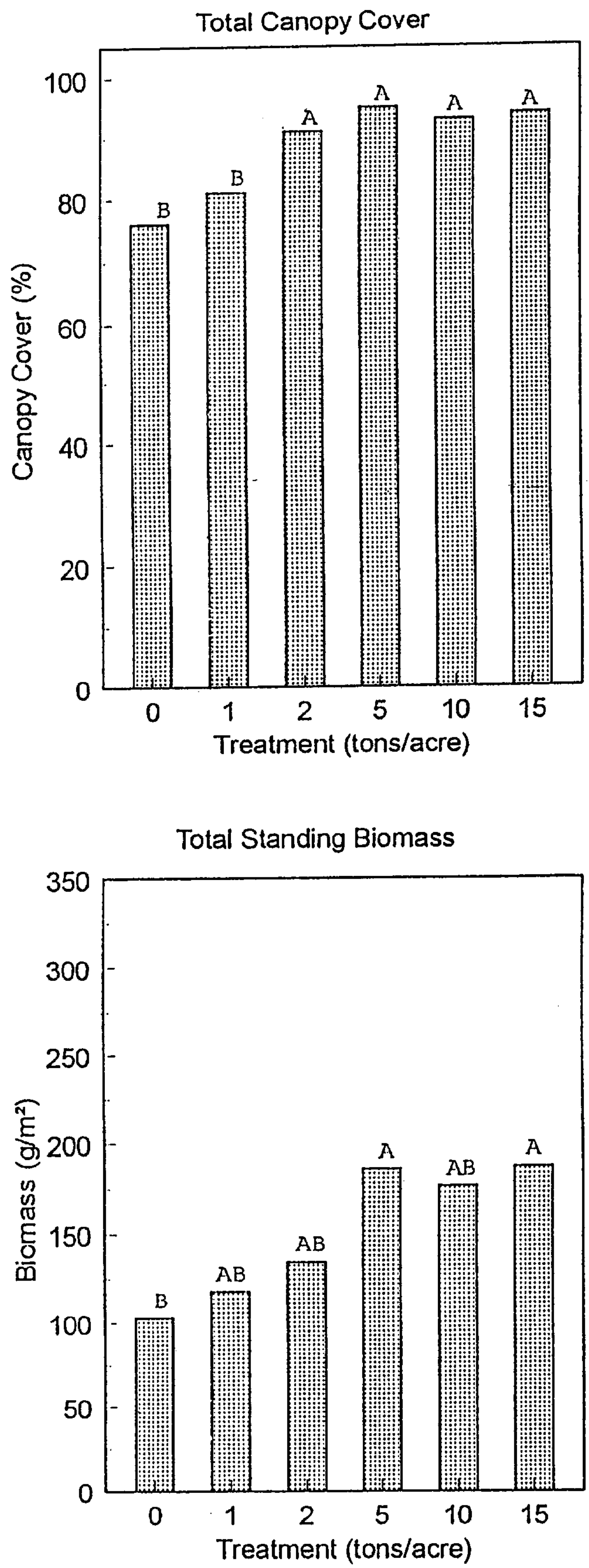
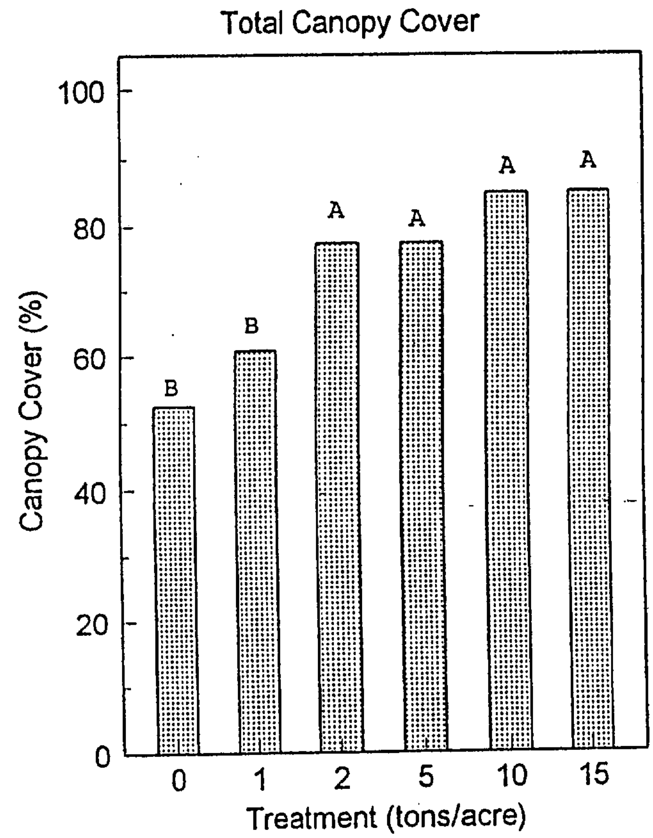


Figure 1. Total canopy cover (%) and total aboveground biomass (g m^{-1}) for sewage sludge plots for 1992. Bars with different letters indicate significant differences between treatments at $P \leq 0.05$.



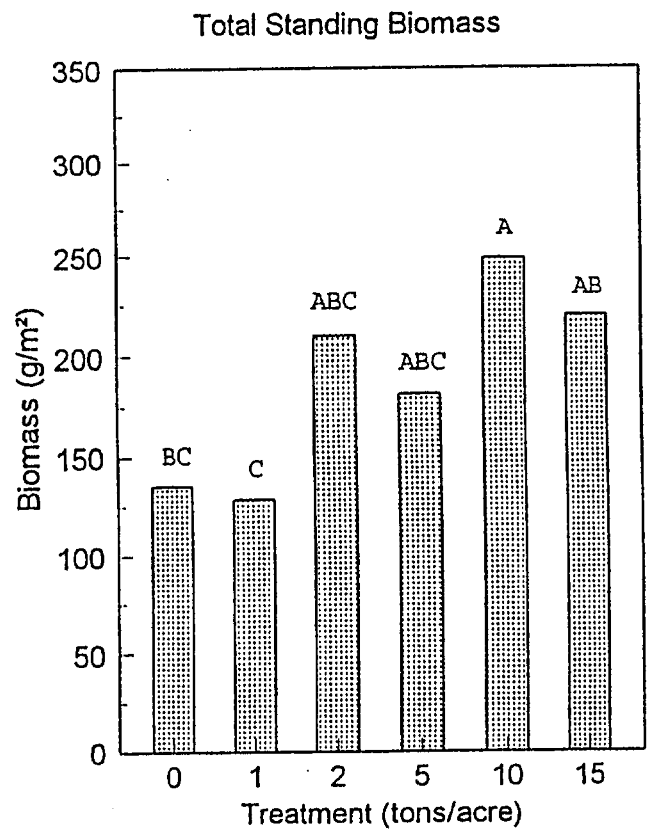


Figure 2. Total canopy cover (%) and total aboveground biomass (g m^{-1}) for sewage sludge plots for 1993. Bars with different letters indicate significant differences between treatments at $P \leq 0.05$.

soil are Ca, Mg, Na, and K; their passage through the soil is slowed by the degree to which they can be retained by exchange sites. Elements of lower solubility include Zn, Cd, Pb, Cu, Ni, Cr, Hg, Mn, Co, P, As, Se, and Mo. These elements can remain sufficiently soluble under various soil conditions (depending on pH) and can be available to plants and enter the food chain (Lindsay 1973).

Increases in soil concentration of the nitrogen species are attributable to the levels added with the biosolids. Lower levels of NH_4-N in 1993 versus 1992 are a result of nitrification to NO_3-N and subsequent leaching through the soil, as evidenced by the higher levels of NO_3-N at 15-30 cm (6-12 in) in 1993. Plant uptake and retention of N in living plants and litter in 1992 also reduced the amount of N available in 1993.

Increases in electrical conductivity probably resulted from salts added with the biosolids. Salts are added to flocculate and settle solids during wastewater treatment (Dowdy, et al. 1976). Higher EC at the lower depths is indicative of salts leaching through the soil profile. However, electrical conductivity remained at levels that were not of concern at all depths for all rates (<1 dS m^{-1}).

Phosphorus and trace metal concentrations were generally higher in 1993 than 1992 because of break down of the biosolids over time and incorporation into the soil surface. Increases in these elements in the soil can be attributed to the amounts added with the sewage sludge.

Cover in all treatments, including the control, was lower in 1993 when compared with 1992. The difference was likely due to climate; 1993 was a cooler, drier year than 1992. An increase in forb cover was an expected result with additional N, it is not clear, however if this trend will continue over time. Fresquez et al. (1990a,b) also noted an increase in foliar cover, particularly for blue grama, with the addition of sewage sludge.

An increase in forb biomass was expected with the addition of N at levels not commonly found on the shortgrass prairie. Lauenroth, et al. (1978) observed a similar increase in forbs with addition of N on a semi-arid grassland in eastern Colorado.
Blue grama and fringed sage seemed to take advantage of the additional nitrogen, although the fringed sage did not respond the second year following application, possibly a result of the climatic difference between years. Fresquez et al. (1990b) reported that blue grama nearly doubled in weight with the addition of sewage sludge. Western wheatgrass did not respond the first year (1992), but showed a marked increase the second year (1993). This may have been related to cooler temperatures and less competition from warm season species.

Increases in tissue concentrations of N, P, and K were directly attributable to the additional nutrients provided by the application of sewage sludge. Fresquez, et al. (1991) reported a similar increase in N, P, and K in blue grama tissue with increases in sewage sludge application rate. Couillard and Grenier (1989) reported a positive correlation between growth on sludge-amended soils and the N and P content of plant tissues.

Conclusions

It is apparent from our research and earlier research that the addition of sewage sludge can produce a beneficial response in a semi-arid rangeland. The addition of nutrients, provided at reasonable levels, can enhance production and improve range condition as measured by cover and production of desirable species. Most of the native species, especially blue grama, respond favorably to the nutrients found in the sewage sludge.

Sewage sludge also contains metals that can build up in the soil and be taken up by plants and enter the food chain. Low rates of sludge application produced a vegetative response and although metals showed some increase in soils and plant tissues, concentrations were well below levels of concern. Previous research has shown that plant availability of sludge-borne metals is highest during the first year sludge is applied, with declines thereafter. Declines in metal concentrations are therefore expected in this study over time.

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Literature Cited

Berglund, S., R.D. Davis, and P. L'Hermite (eds.). 1984. Utilization of Sewage Sludge on Land: Rates of Application and Long-term Effects of Metals. D. Reidel Publishing Co., Dordrecht, Holland.

Bradford, G.R., A.L. Page, L.J. Lund, and W. Olmstead. 1975. Trace element concentrations of sewage treatment plant effluents and sludges: Their interactions with soils and uptake by plants. J. Environ. Qual. 4:123-127.

Catroux, G., P. L'Hermite, and E. Seuss (eds.). 1981. The Influence of Sewage Sludge Application on Physical and Biological Properties of Soils. D. Reidel Publishing Co., Dordrecht, Holland.

Couillard, D. and Y. Grenier. 1989. Effect of applications of sewage sludge on N, P, K, Ca, Mg, and trace element contents of plant tissues. The Sci. of the Total Env. 80:113-125.

Dowdy, R.H., R.E. Larson, and E. Epstein. 1976. Sewage sludge and effluent use in agriculture. In Land Application of Waste Materials. Soil Conservation Society of America. Ankeny, Iowa. pp.138-153.

Fresquez, P.R., R.E. Francis, and G.L. Dennis. 1990a. Soil and vegetation responses to sewage sludge on a degraded semiarid broom snakeweed/blue grama plant community. J. Range Manage. 43:325-331.

Fresquez, P.R., R.E. Francis, and G.L. Dennis. 1990b. Effects of sewage sludge on soil and plant quality in a degraded semiarid grassland. J. Environ. Qual. 19:324-329.

Fresquez, P.R., R. Aguilar, R.E. Francis, and E.F. Aldon. 1991. Heavy metal uptake by Blue grama growing in a degraded semiarid soil amended with sewage sludge. J. Water, Air, & Soil Poll. 57-58:903-912.

Harris-Pierce, R.L. 1994. The Effect Of Sewage Sludge Application on Native Rangeland Soils and Vegetation. Masters thesis Colorado State University. pp.

Havlin, J.L., and P.N. Soltanpour. 1980. A nitric acid plant digest method for use with inductively coupled plasma spectrometry. Comm. Soil Sci. and Plant Anal. 11:969-980.

Keeney, D. R., and D. W. Nelson. 1982. Nitrogen-Inorganic forms. pp 648-649 *In* A. L. Page (ed.) Methods of Soil Analysis, Part 2, 2nd edition. Chemical and Microbiological Properties. American Soc. Agon. Madison, WI.

Lauenroth, W.K., J.L. Dodd, and P.L. Sims. 1978. The effects of water- and nitrogen-induced stresses on plant community structure in a semi-arid grassland. Oecologia 36:211-222.

Lindsay, W.L. 1973. Inorganic reactions of sewage wastes with soils. In Recycling Municipal Sludges and Effluents on Land. National Association of State Universities and Land-Grant Colleges. Washington, D.C. pp 91-96.

Melsted, S.W. 1973. Soil-plant relationships (some practical considerations in waste management). In Recycling Municipal Sludges and Effluents on Land. National Association of State Universities and Land-Grant Colleges. Washington, D.C. pp 121-128.

Sopper, W.E. and S.N. Kerr (eds.). 1979. Utilization of Municipal Sewage Effluent and Sludge on Forest and Disturbed Land. The Pennsylvania State University Press, University Park, Pennsylvania.

Steel, R.G.D., and J.H. Torrie. 1980. Principles and Procedures of Statistics: A Biometrical Approach. McGraw-Hill Publishing Co., New York.

Thomas, R.L., R.W. Sherd, and J.R. Mayer. 1976. Comparison of conventional and automated procedures for nitrogen, phosphorus, and potassium analysis of plant materials using a single digestion. Agron. J. 59:240-243.

United States Environmental Protection Agency. 1990. 40 CFR - Part 503, National sewage sludge survey: Availability of information and data and anticipated impacts on proposed regulations. Federal Register, Vol. 55, No. 218, pp. 47210- 47283.

Workman, S.M., P.N. Soltanpour, and R.H. Follett. 1988. Soil testing methods used at Colorado State University for the evaluation of fertility, salinity, and trace element toxicity. Agricultural Experiment Station Technical Bulletin LTB88-2. Colorado State University, Fort Collins, Colorado.

Biosolids Fertilization of a Colorado Shrubland

Bonnie L. Pierce, Edward F. Redente, and Kenneth A. Barbarick

Abstract

A study of the effects of municipal biosolids application on a semiarid sagebrush ecosystem was initiated in July 1991 at Wolcott, Colorado to evaluate the interactions of elemental content of the biosolids, the soil environment, and the plant community. Under proper management, millions of hectares of rangeland in the U.S. may benefit from the nutrient and soil building properties provided by biosolids. This investigation offered one of the first opportunities to study ecosystem responses of a semiarid shrubland in the western U.S. to biosolids fertilization.

Semiarid shrubland attributes which may present unique challenges to land managers considering biosolids fertilization include extensive areas of bare ground, naturally occurring high concentrations of trace elements in soils and plants, steep slopes, immature soils, presence of deep-rooted accumulator plant species, and spatial and temporal variability in precipitation.

The results of this study indicate that 2 years following a single surface treatment of biosolids, N and P uptake was enhanced in 3 dominant perennial grasses. Molybdenum and Se concentrations were significantly decreased with increasing biosolids application rates in 2 perennial grasses and 1 perennial forb, respectively. Additionally, soil pH, EC, NO_3-N, NH_4-N, and trace element concentrations were unaffected by treatment at rates below 20 Mg biosolids/ha and no significant changes in plant production, cover, seral stage, or species composition were identified at any treatment rate 2 years following land application of biosolids.

Keywords: biosolids, shrubland, sagebrush, semiarid, organic fertilizers, nitrate, trace metals

Introduction

The Wolcott, Colorado biosolids study was initiated in July 1991 in a cooperative research effort between the U.S. Environmental Protection Agency (USEPA), the Colorado Department of Public Health and Environment (CDPH&E), Colorado State University (CSU), the Bureau of Land Management (BLM), and the Upper Eagle Valley Consolidated Sanitation District. The study was conducted on BLM land, located in a cold desert sagebrush community at an elevation of 2,225 meters. The results from this study were used to evaluate the interactions among the nutrient content of the biosolids, the soil environment, and the plant community for the development of appropriate management practices for the beneficial use of biosolids on semiarid rangelands.

The primary objective of the study was to determine if the application of biosolids had measurable effects of plant cover and production, plant uptake of metals and nutrients, and the chemical properties of soils. Specific objectives were to:

1: Determine if plant species composition changed with increasing biosolids application rates;
2: Ascertain whether total plant foliar cover and herbaceous production increased with increasing biosolids applications rates;
3: Evaluate total plant tissue concentrations of nutrients and metals with increasing biosolids application rates, and
4: Determine if soil chemical properties and total concentrations of nutrients and metals changed with increasing biosolids application rates.

Shrubland Nutrient Requirements

Little information is known about rangeland nutrient requirements because of the wide variety of rangelands and associated plant species, and because fertilization of rangelands is not a common practice. Determining nutrient requirements for semiarid shrublands is further complicated by the spatial variability of many soils and the variability in timing and microscale distribution of precipitation at some sites (Rickard et. al., 1988). Most research accomplished to date has focused on the nutrient requirements of specific crops or rangeland species.

Nitrogen is commonly recognized as the most important growth limiting nutrient in most terrestrial ecosystems (Mengel and Kirkby, 1987). Numerous researchers have reported that during short periods when soil moisture is adequate, nutrient supply can be limiting to arid and semiarid ecosystems (James and Jurinak, 1978; Breman and deWit, 1983; and Sturges, 1986). Burke (1989) in a field experiment on a high elevation sagebrush ecosystem in Wyoming, with loamy sands and sandy soils, reported that soil N pools varied significantly across the landscape, with larger N pools associated with higher vegetation cover and downslope and leeward topographic positions.

The seral position of a plant species may also determine its nutrient requirements. Redente et. al (1992), under controlled greenhouse conditions, investigated the nutrient response patterns of 5 semiarid plant species common to sagebrush ecosystems. The investigators reported that early seral species produced more biomass but had lower tissue nutrient concentrations of N and P, especially at high nutrient levels, while late seral species, in response to low availabilities of N and P, sequestered nutrients within a smaller amount of biomass. At low nutrient availabilities, the late seral species demonstrated a competitive advantage in obtaining N and P.

Mengel and Kirkby (1987) provide an example of the amounts of N and P a cool season, perennial grass typically removes from the soil during one growing season. They reported, as a general guideline, a nutrient uptake of 66 kg N/ha and 13 kg P/ha to produce a yield of 6,000 kg/ha of common Timothy grass (*Phleum pratense)* in one growing season (their calculations are based on the data of Eakin, 1972). Stubbendieck et. al. (1992) list this species as a non-drought tolerant grass which responds to N fertilization.

A number of factors control nutrient cycling in the soil-plant system. Nutrient input and output processes have been characterized for many ecosystems, but specific nutrient requirements for semiarid rangeland systems have not been defined. Nutrient loss by leaching is lower in semiarid ecosystems due to lower annual precipitation, however, plants need adequate soil moisture to obtain the nutrients from the soil. The lack of continuous vegetation cover in semiarid regions suggests a lower supply of soil moisture and nutrients. The addition of organic fertilizers may increase the supply of nutrients to some rangeland species and may improve soil structure to aid in water and nutrient retention, however, increased nutrient losses may also occur from leaching, volatilization, and erosion. Because water may be the most limiting growth factor in semiarid rangelands, simply increasing nutrient supply may be without effect for improving conditions for plant growth. Alleviating this problem by increasing soil-water retention capacity through improved soil structure may result in yield increases for forage grasses on semiarid rangelands.

Biosolids as Fertilizers

The use of municipal biosolids as a N-source is widely recognized for its short-term fertilization effects (Hall, 1984). With appropriate application rates, plants such as perennial grasses, may benefit from the long-term slow release of organic-N from biosolids (Hall, 1984). Mengel and Kirkby (1987) report that the main advantage of slow-release organic-N fertilizers is that a single application can often be used to meet plant needs.

Most biosolids land application systems have been designed for use with croplands, forests, drastically disturbed lands (Sopper et al, 1977), and most recently semiarid grasslands (Fresquez et al, 1989, 1990a, 1991, and Harris-Pierce, 1994).

Numerous differences exist between semiarid grasslands and shrublands which could affect ecosystem responses to land application of biosolids. Besides obvious differences in plant community composition, semiarid shrublands usually do not have continuous vegetation cover and have considerable bare ground exposed (Blaisdell et. al., 1982). Sturges (1975) concluded from his research at the Stratton Sagebrush Hydrology Study Area in Wyoming that increased sediment movement across soil surfaces can be expected in semiarid shrublands in years with high snowmelt rates or with intense summer storms that produce overland flow, no matter how well the watershed is vegetated. This may have important implications for the potential contamination of surface waters in regions of greater slope.

Current USEPA regulations for biosolids application for beneficial use (Numerical Criteria for Part 503 Rule) lists pollutant loading limits for 11 trace elements which may be contained in biosolids and maximum allowable cumulative loading rates. Although natural soil levels of these metals were considered in establishment of the numerical criteria (USEPA, 1992), certain land applications may require evaluation of abnormally high levels of these metals in rangeland soils and the potential for some plant species to accumulate large quantities of metal (e.g., Se) (Beath et. al., 1939a, 1939b, and 1940) from deeper soil horizons. Upon decomposition of plant species which accumulate heavy metals, there is potential for increases in plant-available metals in surface soils which could affect other plant species, e.g., forage plants.

Methods and Materials

Study Site Description

The study site is located 2 km north of the town of Wolcott, Colorado on BLM land. The research site closely matches the surrounding land in terms of physical soil properties, slope, elevation, climatic conditions, and plant communities. The Eagle river, a tributary of the Colorado river, flows from east to west approximately 1.8 km south of the study site, and a sanitary area landfill is located directly north of the study site. The average elevation for this region is 2,225 meters.

Biosolids Characteristics

Biosolid samples from the Upper Eagle Valley wastewater facility were taken from July to September 1991 prior to their surface application at Wolcott, CO. Analysis of its elemental content and chemical properties are given in Table 1, and indicates that it meets current USEPA regulations for application for beneficial use (Numerical Criteria for Part 503 Rule). The biosolids qualify as "high-quality sludge" (meets the "Alternate Pollutant Limit") indicating that as long as agronomic rates are applied, there is no time limitation for addition of these biosolids.

Soils and Climate

The soils are primarily of the Tanna-Pinelli complex, 12 to 25% slopes, consisting of fine, montmorillonitic Aridic Argiborolls and fine, montmorillonitic Borollic Haplargids (SCS, 1992). Soils of the southeastern portion of the study site include numerous small areas of the Moyerson-Rock Outcrop complex, which are shallow Ustic Torriorthents (immature soils derived from shale) (SCS, 1992).

The Wolcott study site soils are extremely heterogeneous. Effective rooting depths range from 25 cm (Moyerson Complex) to 152 cm or deeper (Pinneli soils) (SCS, 1992), and the available water-holding capacity ranges from very low (Moyerson Complex) to high (Pinneli soils). Permeability is slow on Tanna soils, and all soils have a moderate to high hazard of water erosion with rapid runoff during summer precipitation (SCS, 1992). Extensive areas of bare ground are found at the study site ranging from 37% bare ground in 1991 to 26% in 1993 on control plots (Pierce, 1994).

The climate in this region is characterized by an average annual precipitation of 330 to 381 mm, an average annual temperature of 4.4 to 6.7^0 C, and an average frost-free period of 80 to 90 days (SCS, 1992). The average seasonal snowfall is 127 to 152 cm with areas of snowpack accumulating during the winter, depending on topographic position (SCS, 1992).

The wettest year during the study was 1993, although drought conditions existed throughout the investigation. Climatic data from the weather station at Eagle FAA airport near Wolcott, CO shows an annual total precipitation of 256 mm in 1991, 223 mm in 1992 and 295 mm in 1993. Additionally, in 1993, the study site received 87 mm more total precipitation from January through July, than the prior year.

Table 1. Biosolid chemical properties and nutrient concentrations from the Upper Eagle Valley Sanitation District. Baseline data were collected July-September 1991.		
	Units	1991
Chemical Property		
pH	none	7.7
EC	ds/m	13.0
Total solids	%	28.0
Volatile solids	%	71.8
Nutrient		
Total N	%	1.70
Organic N	%	1.24
NO_3-N	mg/kg	0.159
NH_4-N	mg/kg	168
K	mg/kg	0.44
P	mg/kg	1.98
Zn	mg/kg	1210
Fe	mg/kg	1.24
Cu	mg/kg	567
Ni	mg/kg	14.7
Mo	mg/kg	2.92
Al	mg/kg	1.60
Cd	mg/kg	4.78
Cr	mg/kg	22.1
Pb	mg/kg	47.4
As	mg/kg	3.48
Hg	mg/kg	2.71
Se	mg/kg	4.66
Ag	mg/kg	37.9

Chemical analyses of baseline soils shows low soil fertility with an estimated 7.6 kg available N/ha and 5.5 kg available P/ha in the upper soil profile (0-60 cm) in 1991 (Pierce, 1994). Available soil N includes NO_3-N and NH_4-N , and available soil P is PO_4^{3-} measured in baseline soils and using an assumed soil bulk density of 1.33 g/cm^3. Table 2 indicates that 3 trace elements are found at naturally high levels in baseline soils.

Plant Community

In 1991, perennial grasses, mid-seral shrubs, and big sagebrush (*Artemisia tridentata)* were the dominant species at the study site. On a similar sagebrush ecosystem in Colorado, McLendon and Redente (1990) characterized successional patterns over a 12- year period showing that this combination of plant species indicates an ecosystem at some point between a mid to late seral stage. The six dominant species are western wheatgrass, bluebunch *wheatgrass (Agropyron spicatum*), Indian ricegrass *(Oryzopsis hymenoides*), pulse milkvetch (*Astragalus tenellus*), big sagebrush (*Artemisia tridentata*), and Douglas rabbitbr*ush (Chrysothamnus viscidiflorus*).

Experimental Design
The experimental design for this study consisted of a randomized, complete block design with nine treatments. Each treatment was a single surface application of biosolids from the Upper Eagle Valley Consolidated Sanitation District at the following rates: 5, 10, 15, 20, 25, 30, 35, and 40 Mg/ha. The dimensions of each treatment measured approximately 31 by 107 meters and were replicated four times.

Sampling Procedures
We collected baseline soils data in July 1991, prior to biosolids application, to a depth of 180 cm with a deep core-

Table 2. Comparison of soil nutrient and metal concentrations at different soil depths in 1991 (controls) to normal and critical soil levels (potential for phytotoxicity or zootoxicity) (based on Mengel and Kirkby, 1987 and USEPA, 1992).

	Elements		
	Mo mg/kg	Cd mg/kg	Zn mg/kg
Normal Soil Levels	0.6-3.5	unknown	50
Critical Soil Levels (potential for phyto-toxicity or zoo- toxicity)	100	3	300
USEPA Background Soil Levels	2.00	0.20	54.0
Study Site Soils			
0-30 cm depth	10.9	*6.16	126
30-60 cm depth	11.7	*8.37	128
60-90 cm depth	12.5	*5.71	115
90-120 cm depth	10.7	*5.31	101
120-180 cm depth	11.1	*4.78	95.8
150-180 cm depth	7.95	*3.34	61

* Exceeds critical soil level

sampling instrument. Samples were taken at depths of 0-30 cm, 30-60 cm, 60-90 cm, 90-120 cm, 120-150 cm, and 150-180 cm. Soils were analyzed for pH, EC, NO_3-N, NH_4-N levels, and total P, K, Cu, Fe, Mn, Zn, Cd, Pb, Ni, Al, Ti, Mo, Cr, Sr, B, Ba, Ca, Na, and Mg (Pierce, 1994). We completed follow-up monitoring, one and two years after treatment application, in July 1992 and August 1993. Analysis of the 1992 soils data included pH, EC, and NO_3-N and NH_4-N levels to a depth of 30 cm. The samples were divided into three depths: 0-10 cm, 10-20 cm, and 20-30 cm. Analysis of the second year (1993) soil samples were the same as the baseline samples at
the following depths: 0-10 cm, 10-20 cm, 20-30 cm, 30-60 cm, and 60-90 cm.

Biosolid samples from the wastewater facility were taken prior to land application at Wolcott, CO. We analyzed the samples for total P, K, Cu, Fe, Mn, Zn, Cd, Pb, Ni, Al, Ti, Mo, Cr, Sr, B, Ba, Ca, Na, and Mg, as well as pH, EC, % total solids, % volatile solids, and NO_3-N and NH_4-N levels (Pierce, 1994). We collected surface biosolid samples in July 1992 from the 40 Mg/ha treatments and analyzed for total % N, NH_4-N, NO_3-N, P, K, Cu, Fe, Mn, Zn, Cd, Pb, Ni, Al, Ti, Mo, Cr, Sr, B, Ba, Ca, Na, and Mg. We again collected surface biosolids samples in August 1993 from the 40 Mg/ha treatments and analyzed for total P, K, Cu, Fe, Mn, Zn, Cd, Pb, Ni, Al, Ti, Mo, Cr, Sr, B, Ba, Ca, Na, and Mg.

We collected baseline vegetation data on the test plots prior to treatment application. Each treatment was sampled for canopy cover by species, and each control plot was sampled for herbaceous production and plant tissue nutrient and metal concentrations.

We measured aboveground herbaceous production by harvesting vegetation by species in rectangular 0.5 m^2 quadrats placed in a stratified-random manner in each treatment. We clipped all biomass either at ground level or at the point of entry into the quadrat volume. Baseline data included harvesting 15 quadrats per control treatment across all four blocks. We sampled aboveground production for all treatments in July 1992 and 1993. In all 3
years, shrub height and canopy width were measured for shrubs situated within the 0.5 m^2 quadrats.

We completed baseline cover measurements using the individual point method along randomly placed transects within each treatment. These measurements included the amount of ground covered by rock, litter, and live vegetation. We took baseline measurements in all treatments and included the placement of two 100 point transects per treatment. Measurements for July 1992 and 1993 were the same, except that we sampled four transects per treatment.

Plant tissue samples were taken from each control plot prior to biosolids application and from all treatments in July 1992 and 1993 to determine nutrient and metal concentrations. Aboveground plant samples from each of six dominant species were collected, washed in distilled water, oven-dried at 60° C, ground, digested in concentrated nitric acid, and analyzed using the ICP for total N, P, K, Ca, Mg, Na, Cu, Fe, Mn, Zn, Cd, Pb, Ni, Al, Ti, Cr, Sr, B, and Ba.

Statistical Analysis
Statistical testing of the data included linear and quadratic regression analyses and analyses of variance. We used the F test to determine significance at the 0.05 and 0.01 level of probability. We utilized linear and quadratic regression analyses and analyses of variance of 1992 through 1993 data to determine if any climatic or treatment effects existed for the 2 years following treatment application. Significant results for regression analyses were based on a least squares fit of all sample data to account for variation within and between replications. We considered results as significant for regressions with R^2 values ≥ 0.35 We used the LSD test to determine means separation between treatments for 1992-1993 analyses of variance.

Results and Discussion

The application of municipal biosolids had measurable effects on the chemical properties of plant tissues and soils at Wolcott, CO two years following land application, but showed no effect on plant species composition, and total plant foliar cover and herbaceous production, (Pierce, 1994). In addition, systematically varying treatment rates enabled us to determine inhibitory biosolids rates (above 20 Mg/ha) and determine the rates at which plant nutrient needs may have been exceeded (above 10 Mg/ha).

Nutrient Removal by Plants
The addition of low to moderate levels of N and P from a slow release organic fertilizer, such as municipal biosolids, to a mid seral sagebrush ecosystem might be predicted to result in a favorable environment for uptake of these nutrients by the mid seral species. Data indicates that the 3 dominant perennial grass species, western wheatgrass, bluebunch wheatgrass, and Indian ricegrass at Wolcott, CO did increase their plant tissue concentrations of N (Figure 1), and western wheatgrass and Indian ricegrass increased their uptake of P (Figure 2), but did not produce more biomass with treatment (Pierce, 1994). The nutrient response pattern of late seral species, typical in sagebrush ecosystems, as reported by Redente et. al. (1992), indicates a competitive advantage over early seral species in obtaining N and P at low nutrient availabilities. Late seral species show higher concentrations of N and P in smaller amounts of biomass at lower nutrient availabilities in contrast to the response of early seral species which produce more biomass but have lower tissue nutrient concentrations.

The addition of 10 Mg/ha of Upper Eagle Valley municipal biosolids to native soils provides a total of 66 kg N/ha for plant use (this value includes the available soil NO_3-N and NH_4-N). This nutrient level appears to provide a favorable environment for uptake of N and P by the 3 dominant perennial grass species in the first two years following biosolids application. The increasing quadratic response of N plant tissue levels to increasing biosolids rates for the 2 wheatgrasses (Figure 1) appears to indicate a diminished treatment response at levels above 10 Mg/ha.

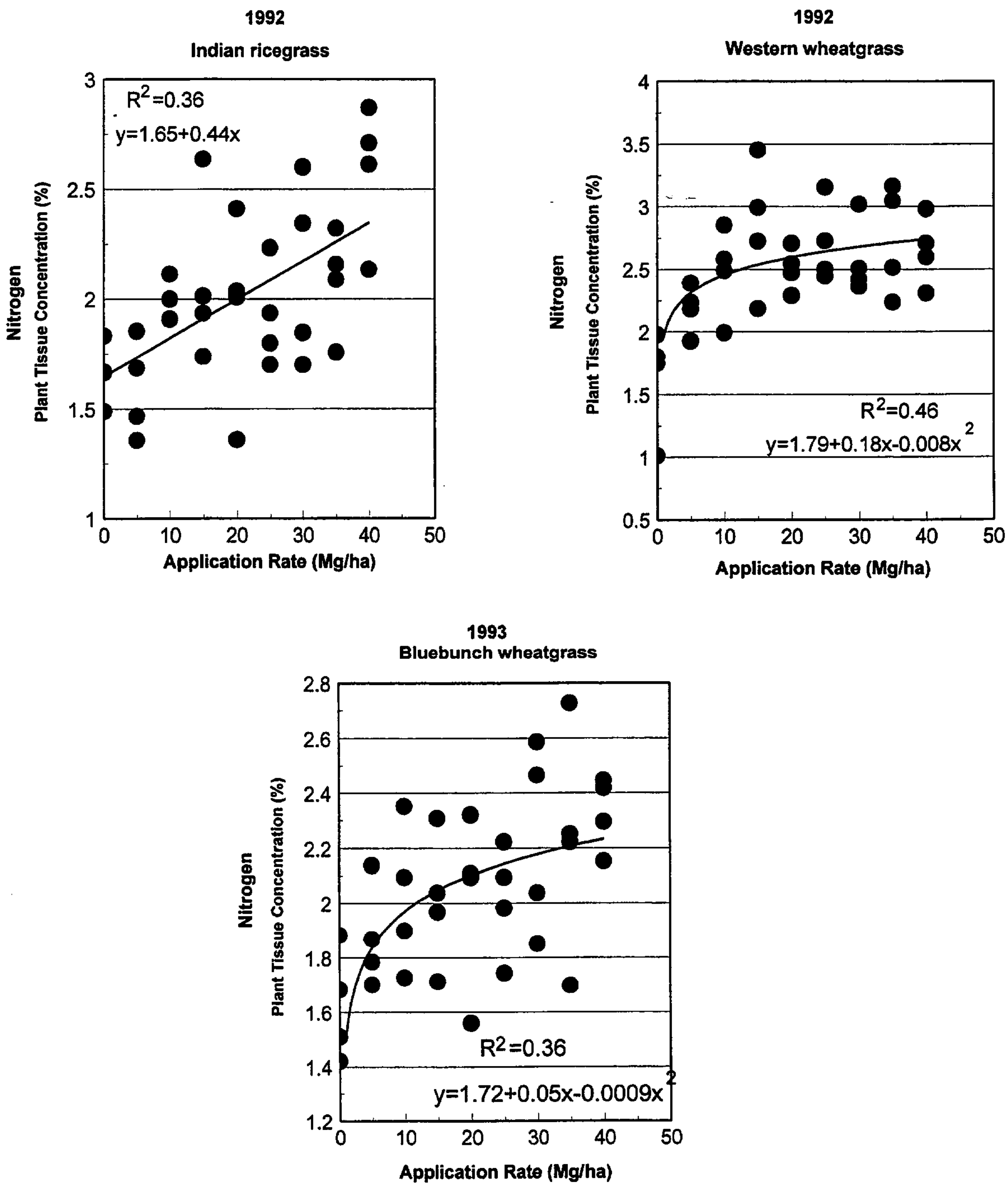


Figure 1. Plant tissue concentrations of N in 3 dominant, perennial grasses, one and two years following treatment application. Data were collected in July 1992 and 1993.

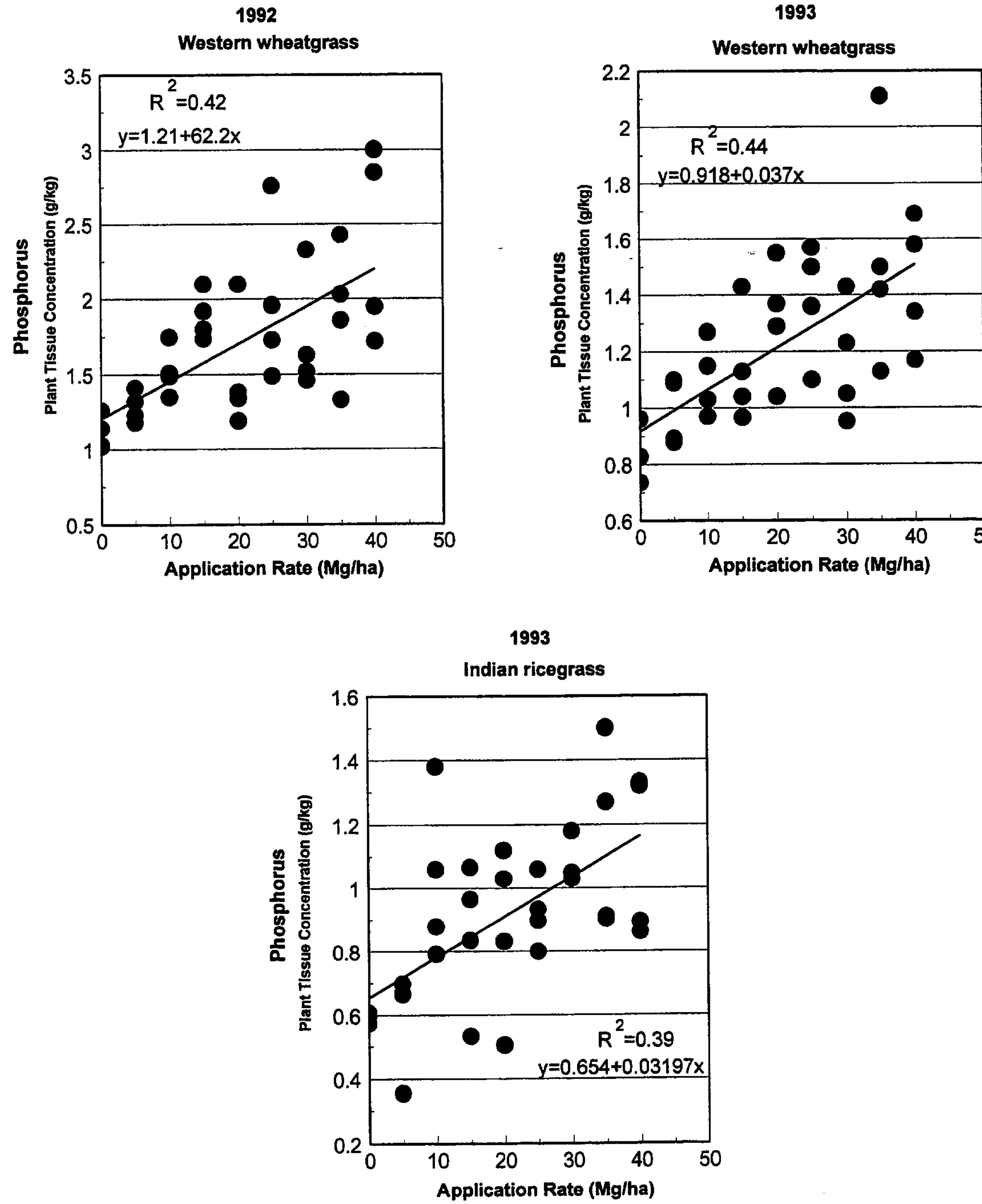


Figure 2. Plant tissue concentrations of P in 2 dominant, perennial grasses, one and two years following treatment application. Data were collected in July 1992 and 1993.

Fate of Elements with More Toxic Effects

Analyses of plant tissue taken from control plots from 1991 to 1993 indicates high levels of Mo and Cd in 4 of the 6 dominant species at Wolcott, CO (Table 3). Three of the dominant species, pulse milkvetch, big sagebrush, and Douglas rabbitbrush are unpalatable to most animals (Stubbendieck, 1992), but may result in increased metal concentrations in upper soil horizons, as the element is brought up from deeper soils and deposited on surface soils through the processes of litter fall and decomposition. In this manner Mo may become more available to other plants, e.g., forage plants.

Trace element concentrations in forage plants which can cause zootoxicity varies with plant species, animal species, time and length of grazing, and percent of animal diet (Church, 1988). The critical levels cited in Table 3 have been determined based on observations of adverse health effects in cattle or sheep grazing areas with grasses exceeding these critical levels. Length of grazing season and percent of diet were not specified in the data of Church (1988).

Table 3. Comparison of tissue nutrient and metal concentrations for four dominant species from 1991-1993 (controls) to normal, phytotoxic, and zootoxic levels (based on Logan and Chaney, 1983 and Mengel and Kirkby, 1987).

Element	
	Mo mg/kg
Normal levels	0.1-0.3
Phytotoxic levels for agronomic crops	100
Critical levels (may cause zootoxicity if included in diet of animals)	5
Study Site Plants	
Pulse milkvetch - 1991	*9.00
Pulse milkvetch - 1992	*5.92
Pulse milkvetch - 1993	*9.25
Indian ricegrass - 1993	*7.26
	Cd mg/kg
Normal levels	unknown
Phytotoxic levels for agronomic crops	5-10
Critical levels (may cause zootoxicity if included in diet of animals)	0.5-1
Study Site Plants	
Big sagebrush - 1992	*0.73
Big sagebrush - 1993	*0.61
Douglas rabbitbrush - 1992	*0.52

* Exceeds critical levels

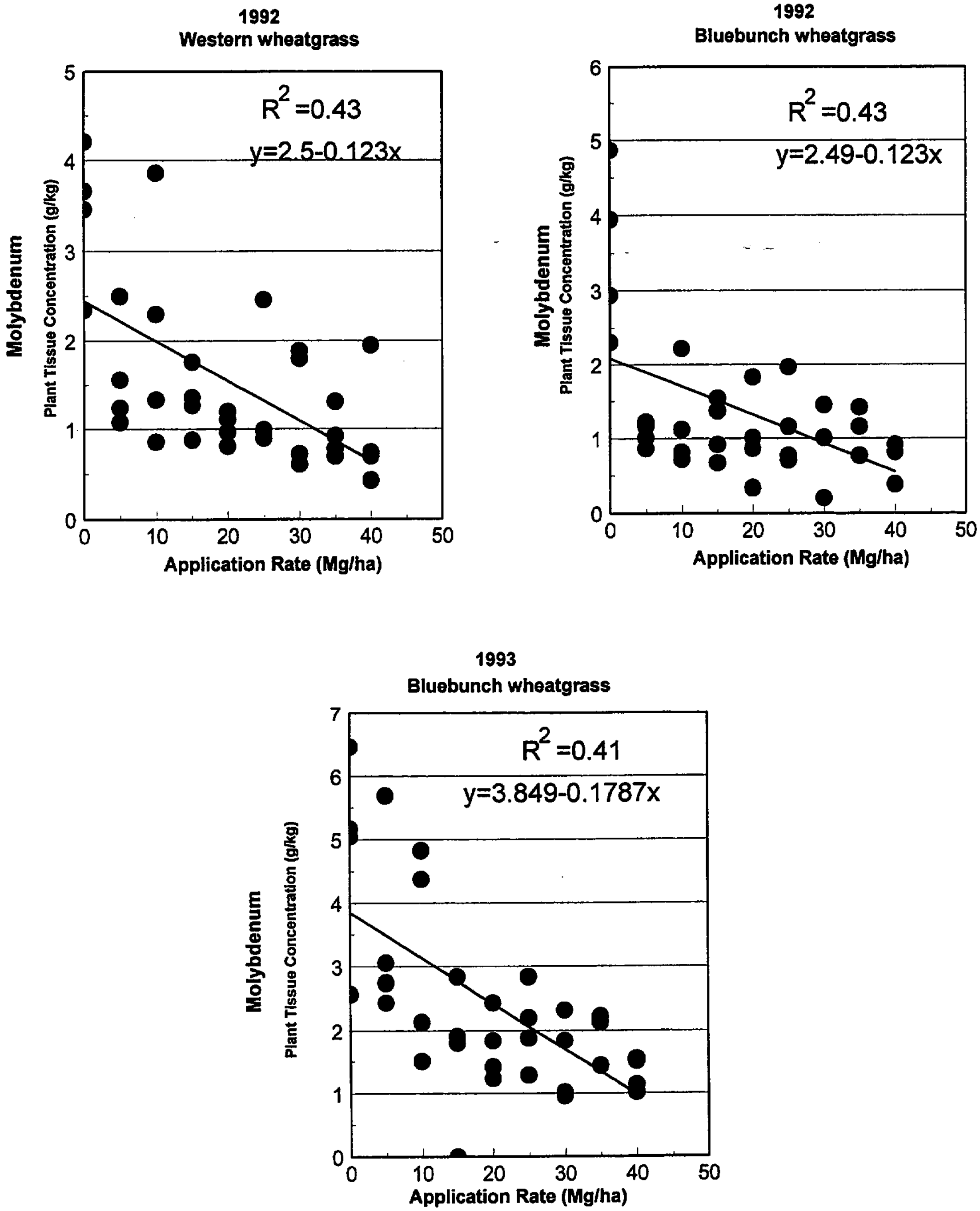
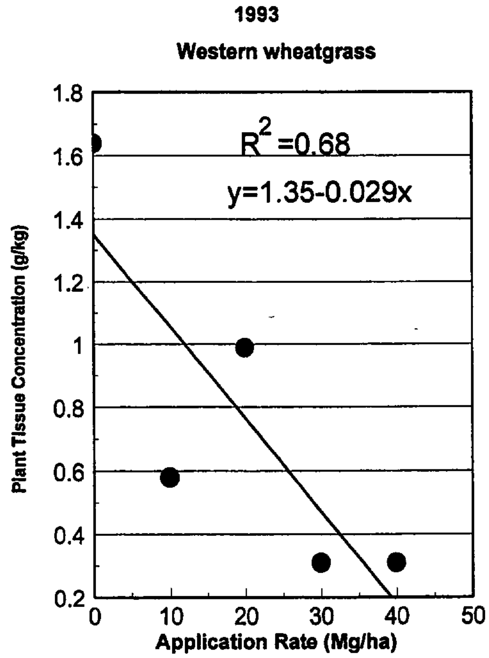


Figure 3. Plant tissue concentrations of Mo in 2 dominant, perennial grasses, one and two years following treatment application. Data were collected in July 1992 and 1993.



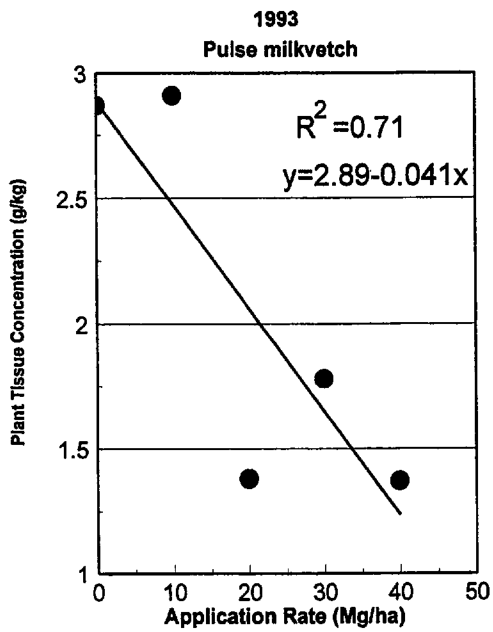


Figure 4. Plant tissue concentrations of Se in a dominant, perennial grasses andperennial forb, one and two years following treatment application. Data were collected in July 1992 and 1993.

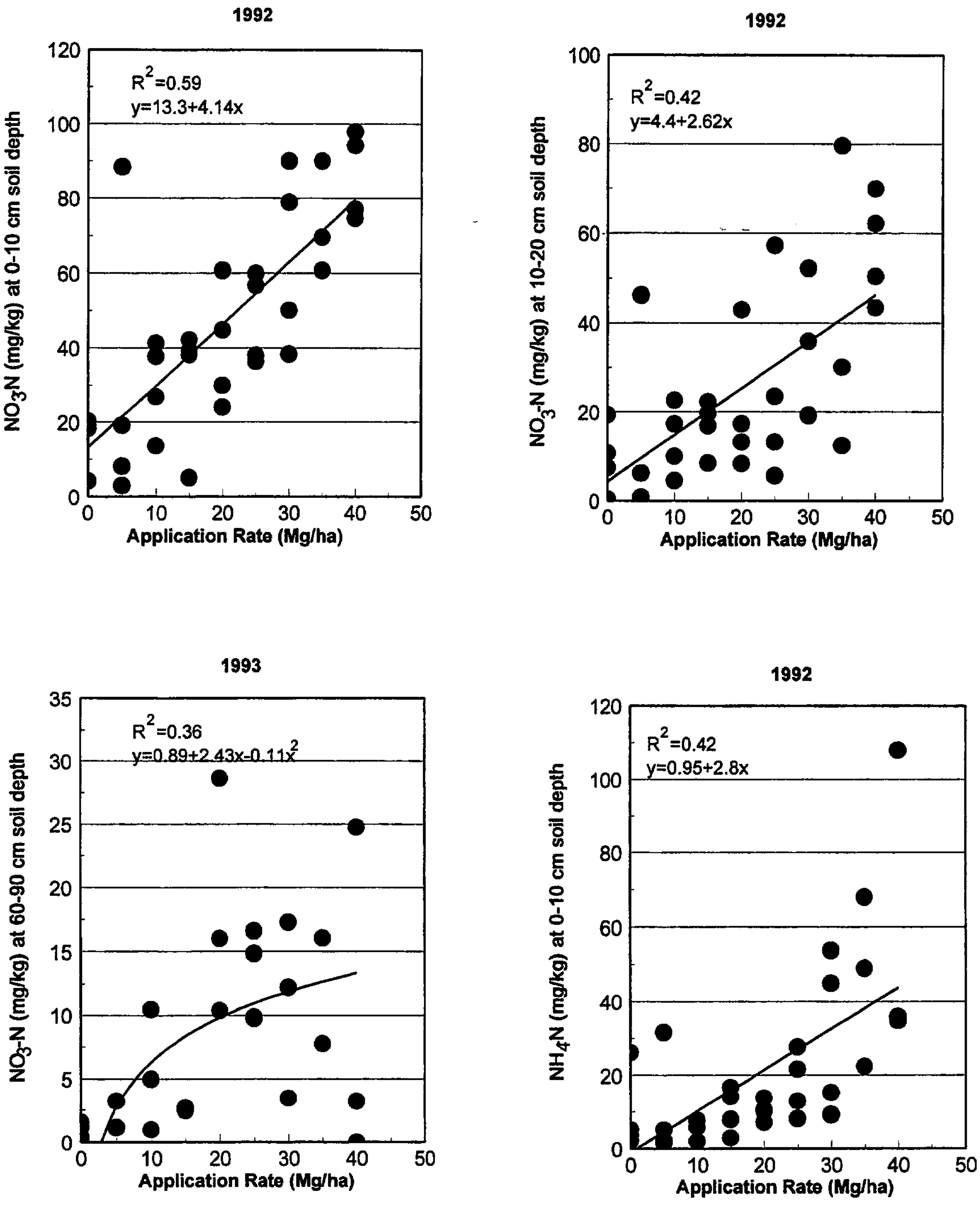


Figure 5. Soil NO_3-N and NH_4-N at various soil depths one and two years following treatment application. Data were collected in July 1992 and 1993.

Kabata-Pendias et. al. (1984) report that the species of higher plants that show a tolerance to high levels of trace elements belong most commonly to the plant families *Cruciferae, Gramineae, Luguminasae*, and *Chenopodiaceae*. Seventeen species from these families are found at the study site (Pierce, 1994).

The application of biosolids appears to reduce potentially toxic levels of Mo in western wheatgrass and bluebunch wheatgrass to safer levels in these dominant perennial grasses (Figure 3). Plant tissue concentrations of Se in western wheatgrass and pulse milkvetch indicates decreases with an increase in treatment rate at the $p \leq 0.10$ level of significance in 1993 (Figure 4).

The decrease in plant tissue concentrations of Mo in plant tissues of the dominant grasses at Wolcott, CO with low biosolids application rates (e.g., 10 Mg/ha) is the most striking result of this study. These perennial grasses provide valuable forage for domestic livestock and wildlife (Stubbendieck, 1992), and the improvement in plant tissue quality is a favorable result. Molybdenum is known to cause toxicity when consumed at high levels in the diet of ruminants (Mengel and Kirkby, 1987), and naturally high levels of this element were measured at the study site in the dominant grasses. Biosolids treatment may reduce Mo plant tissue concentrations in dominant species at Wolcott,CO to safer levels for animal consumption.

Biosolids Effects on Soils

The effect of biosolids application on the study site soils are primarily on surface soils one year following treatment. Application rates above 20 Mg/ha significantly increase soil NO_3-N at the 0-10 cm soil depth in 1992 (Figure 5) to levels of potential concern for contamination of surface waters if significant runoff occurs with intense summer storms or spring snow melt. Soil NH_4-N levels also increased significantly one year following treatment, with increasing application rates, at the 0-10 cm soil depth (Figure 5).

Two years following treatment, soil NO_3-N levels were significantly increased at the 60-90 cm soil depth (Figure 5) and reached levels of potential concern for leaching into and contaminating groundwaters only at the 20 Mg/ha treatment rate. The 1993 results, however, indicate very little overall movement of NO_3-N into soils to 90 cm at any other application rate (Pierce, 1994).

Analyses of soil pH and EC, and soil nutrient and metal concentrations indicates these properties are not significantly affected by biosolids treatment.

Biosolids Application and the Environment

The naturally high levels of Zn found in the soils, and the level of Zn in the Upper Eagle Valley biosolids, in combination, become the limiting factor for land application of biosolids at Wolcott, CO. Applied at 10 Mg/ha, and accounting for natural soil Zn concentrations at the study site, 193 surface applications can be added before USEPA cumulative loading limits are reached (Pierce, 1994).

The two soil elements of potential concern at the study site are Mo and Cd, because they exist at naturally high levels in the soil profile (Pierce, 1994). The USEPA Part 503 Rule does not restrict application of Upper Eagle Valley biosolids at Wolcott, CO (USEPA, 1992), but suggests these soil levels be monitored to avoid exceeding maximum allowable pollutant concentrations.

The USEPA used the median background concentrations of metals in agricultural soils to represent the background soil levels of inorganic biosolids pollutants in their exposure assessment for determination of numerical criteria (Part 503 Rule) for land application of biosolids (USEPA, 1992). The exposure assessment cautions that when background soil levels are significant compared to maximum allowable pollutant concentrations, the allowable pollutant-loading from biosolids should be reduced (USEPA, 1992). Comparison of soil concentrations of Mo and Cd from baseline soils (1991) with USEPA pollutant ceilings indicates that, although these levels are naturally high, they do not represent a significant problem for biosolids application.

Cadmium can be transported readily from soils via plant roots to upper plant parts and recycled in the upper soil mantle by plant decomposition. Cadmium is a cumulative metal that can be stored in the kidneys, liver, and spleen of animals and humans (Mengel and Kirkby, 1987). Molybedenum concentration above 5 mg/kg in dry matter of forage can be toxic to animals. Ruminants, in particular, are susceptible to high Mo levels in plant tissue (Mengel and Kirkby, 1987). Excess Mo in the diet appears to interfere with normal Cu absorption and utilization because Mo is transformed in the rumen to thiomolybdate which binds Cu and prevents Cu absorption from the intestines, thereby causing the disease molybdenosis (Schrader and Thomas, 1981). Based on naturally high soil Cd and Mo levels at Wolcott, CO, these metals should be monitored on a long-term basis (i.e., measured 10 years following treatment), if land application of biosolids continues at this location, to ensure biosolids treatment do not significantly alter soil Mo or Cd levels and pose a threat to animal or human health.

Conclusions

Application of municipal biosolids at Wolcott, CO after 2 growing seasons has provided an enriched level of essential nutrients for plant use, and has lowered potentially toxic levels of Mo plant tissue of 2 of the 6 dominant species at the study site at the 10 Mg/ha application rate. This low treatment level does not contaminate soils from 0 to a 90 cm depth with excessive levels of nutrients, metals, or soluble salts.

Little change was found in total or species-level plant cover and aboveground biomass in both years following treatment with biosolids. No significant shift in seral stage of this semiarid sagebrush ecosystem was detected with treatment.

We found significant increases in the essential nutrients N and P in 3 of the dominant forage grasses at the study site in 1992 and 1993. We expectd this increase in plant tissue nutrient concentration as a result of addition of nutrients from the biosolids.

Most soil chemical properties were not significantly altered at treatment rates below 20 Mg/ha, at all soil depths to 90 cm, 2 years following biosolids application. Higher treatment rates pose a potential hazard for contamination of surface waters by NO_3-N in the first year following land application in semiarid ecosystems subject to significant runoff with intense summer storms or spring snowmelt. No sizeable movement of NO_3-N, NH_4-N, soluble salts, or metal pollutants was detected to a 90-cm soil depth 2 years following treatment, and no change in soil pH was identified.

The naturally high concentrations of Cd and Mo in plant tissue and soils at Wolcott, CO, although not limiting to land application of biosolids, should be monitored on a long-term basis to ensure biosolids treatment do not significantly alter these levels to pose a threat to animal or human health.

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References

Beath, O. A., C. S. Gilbert, and H. F. Eppson. 1939a. The use of indicator plants in locating seleniferous areas in the western U.S. I. General. Amer. J. of Botany. 26:257-269.

Beath, O. A., C. S. Gilbert, and H. F. Eppson. 1939b. The use of indicator plants in locating seleniferous areas in the western U.S. II. Correlation Studies by States. Amer. J. of Botany. 26:296-315.

Beath, O. A., C. S. Gilbert, and H. F. Eppson. 1940. The use of indicator plants in locating seleniferous areas in the western U.S. III. Further Studies. Amer. J. of Botany. 27:564-573.

Blaisdell, J. P., R. B. Murray, and E. McArthur. 1982. Managing intermountain rangelands-sagebrush-grass ranges. Intermountain Forest and Range Experiment Station. General Tech. Report INT-134.

Breman, H., and C. T. deWitt. 1983. Rangeland productivity and exploitation in the Sahel. Science 221:1341-1347.

Burke, I. C. 1989. Control of nitrogen mineralization in a sagebrush steppe landscape. Ecology 70(4):1115-1126.

Church, D. C. (ed.) 1988. The Ruminant Animal: Digestive Physiology and Nutrition. Prentice Hall, Englewood Cliffs, N.J. p.355-395.

Eakin, J. H. 1972. Food and fertilizers. p. 1-21. In The Fertilizer Handbook, The Fertilizer Inst., Washington.

Fresquez, P. R., and G. L. Dennis. 1989. Composition of fungal groups associated with sewage sludge amended grassland soils. Arid Soil Research and Rehab. 4:19-32.

Fresquez, P. R., R. E. Francis, and G. L. Dennis. 1990a. Sewage sludge effects on soil and plant quality in a degraded, semiarid grassland. J. Environ. Qual. 19:324-329.

Fresquez, P. R., R. E. Francis, and G. L. Dennis. 1990b. Soil and vegetation responses to sewage sludge on a degraded semiarid broom snakeweed/blue grama plant community. J. Range Mgmt. 43(4):325-331.

Fresquez, P. R., R. Aguilar, R. E. Francis, and E. F. Aldon. 1991. Heavy metal uptake by blue grama growing in a degraded semiarid soil amended with sewage sludge. Kluwer Academic Publishers, Netherlands. 900-912.

Hall J. E. 1984. The cumulative and residual effects of sewage sludge nitrogen on crop growth. p. 73-83. In J. H. Williams, et. al. (ed). Long-Term Effects of Sewage Sludge and Farm Slurries Applications. 1984. Elsevier Applied Science Publishers. London and N.Y..

Harris-Pierce, R. L. 1994. The effect of sewage sludge application on native rangeland soils and vegetation. Masters Thesis. Colorado State Univ.

James, D. W., and J. J. Jurinak. 1978. Nitrogen fertilization of dominant plants in The northeastern great basin deserts. p.219-231. In N. E. West et. al. (ed). Nitrogen in Desert Ecosystems. U.S. Internat. Program Series No. 9. Dowden, Hutchinson, and Ross, Stroudsburg, Pennsylvania, USA.

Kabata-Pendias, A. and H. Pendias. 1984. Trace Elements in Soils and Plants. CRC Press, Inc., Boca Raton, FL. p.109-116 and p. 199-206.

Logan, T. J., and R. L. Chaney. 1983. Utilization of municipal waste water and sludge on land-metals. p. 235-326. In A. L. Page et. al. (ed). Utilization of Municipal Waste Water and Sludge on Land. Univ. of California, Riverside.

McLendon T. and E. F. Redente. 1990. Succession patterns following soil disturbance in a sagebrush steppe community. Oecologia. 85:293-300.

Mengel, K., and E. A. Kirkby. 1987. Principles of Plant Nutrition. 4th Ed. Internat. Potash Institude, Switzerland.

Pierce, B. L. 1994. The effect of biosolids application on a semiarid rangeland site in Colorado. Masters Thesis. Colorado State Univ.

Rickard, W. H., L. E. Rogers, B. E. Vaughan, and S. F. Liebetrau 1988. Shrub-Steppe: Balance and Change in a Semi-arid Terestrial Ecosystem. Battelle Memorial Institute, Pacific Northwest Laboratories, Richland, WA.

Redente, E. F., J. E. Friedlander, and T. McLendon. 1992. Response of early and late semiarid seral species to nitrogen and phosphorus gradients. Plant and Soil. 140:127-135.

Schrader, L. E., and R. J. Thomas. 1981. Nitrate uptake, reduction, and transport in the whole plant. p. 49-93. In J. D. Bewley (ed.). Nitrogen and Carbon Metabolism. Vol 3. Martinus Nijhoff and Dr. W. Junk Publishers. The Hague, Boston, London.

Soil Conservation Service, U. S. Department of Agric. 1992. Soil Survey of Aspen-Gypsum Area, Colorado, Parts of Eagle, Garfield, and Pitkin Counties.

Sopper, W. E., and S. N. Kerr. 1977. Utilization of municipal sewage effluent and sludge on forest and disturbed land. Proceedings of a symposium conducted by the School of Forest Resources and the Institute for Research on Land and Water Resources. Pennsylvania State Univer.

Stubbendieck, J., S. L. Hatch, and C. H. Butterfield. 1992. North American Range Plants, 4th Ed., Univer. of Nebraska Press. Lincoln and London.

Sturges, D. L. 1975. Hydrologic relations on undisturbed and converted big sagebrush lands: The status of our knowledge. Rocky Mountain Forest and Range Exper. Station. USDA Forest Service Research Paper RM-140.

Sturges, D. L. 1986. Responses of vegetation and ground cover to spraying a high elevation, big sagebrush watershed with 2,4-D. J. of Range Mgmt. 39:141-146.

U. S. Environmental Protection Agency. 1992. Technical support document for land application of sewage sludge. Vol. I and II. EPA 822/R-93-001a. Office of Water. Eastern Research Group. Lexington, MA.

Ammonia Volatilization from Biosolids Applied to Semiarid Rangeland

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Abstract

Proper quantities of applied biosolids are typically based upon nitrogen available to plants. Ammonia volatilization from anaerobically-digested, dewatered biosolids was measured following biosolid application at rates of 0.0, 6.7, and 17.9 dry Mg ha^{-1}. Volatilization studies using two similar soils with different gravel contents were conducted under three temperature regimes. Volatilization followed a diurnal fluctuating pattern and was significantly reduced by day three for all trials. Cumulative volatilization losses, after 210 h, ranged from 11.5 kg ha^{-1} in the cool-season trial for the 6.7 dry Mg ha^{-1} application rate to 35.5 kg ha^{-1} in the hot-season trial for the 17.9 dry Mg ha^{-1} rate. The influence of soil gravel was not significant ($P > 0.01$). A prediction equation is being developed to model volatilization losses as a function of site-and season-specific potential evapotranspiration, biosolid content application rate, and time after application.

Keywords Biosolids, ammonia volatilization, semiarid, collector systems, potential evapotranspiration.

Introduction

Methods to determine the maximum rate of biosolid application for beneficial use have changed over the last decade in an effort to minimize negative environmental impacts associated with land application of biosolids. Generally, current rate limits are based on the balance between crop N needs and by the plant-available N (PAN) content of the applied biosolids. PAN, not total N, is used as the limiting factor because much of the N in fertilizers, including biosolids, is lost from the soil-plant system and is not available for plant uptake.

Many studies, including laboratory and field work with various organic fertilizers (urea, biosolids, and slurry), have indicated that gaseous loss of ammonia is one of the major pathways of N loss from fertilizer applications (Beauchamp *et al.*, 1978; Adamsen and Sabey, 1987; Fine *et al.*, 1990). For example, Beauchamp *et al.* (1978) reported that 56-60% of the ammoniacal N (21-22% of the total N applied) in anaerobically-digested liquid biosolids was volatilized within 5-7 d after application. Other studies have attempted to quantify gaseous loss of ammonia from organic fertilizer application, including biosolids (Beauchamp *et al.*, 1982). Much of the early work involved laboratory experiments and extrapolation to field conditions, but recent ammonia volatilization work has focused on field quantification (Freney *et al.*, 1983). Research focused on initial volatilization, which occurs by NH_3 diffusion from the biosolid-water solution and along with the drying of the sludge, parameters that control diffusion and drying are of interest. Many studies, including work by Beauchamp *et al.* (1980), and Beauchamp *et al.* (1982) have attempted to determine relationships between diffusion/drying parameters (including air temperature and ambient partial pressures) and volatilization. For instance, Beauchamp *et al.* (1978) concluded that air temperature is especially important in the first two to three days after biosolid application with other environmental and soil parameters later increasing in importance. With diffusion/drying parameters in mind and from observations in this research, drying potential played a major role in volatilization, especially during the first few days following biosolid application. This study focused on field quantification of volatilization under semiarid/arid rangeland conditions.

The objectives of this study were 1) to quantify volatilization of ammonia from anaerobically-digested, dewatered biosolids; and 2) to predict volatilization losses from biosolids applied to semiarid rangeland under various seasonal conditions.

Materials and Methods

Ammonia volatilization from applied biosolids was measured with semi-open, dynamic NH_3 collectors (0.575 m x 0.405 m x 0.3 m). The collectors (Fig. 1 a,b) are "semi-open" because they permit gaseous exchange with the atmosphere via diffusion and "dynamic" because they sweep gases from the soil surface (Marshall and Debell, 1980). The collectors were placed in the field to allow the diurnal cycle and its associated phenomena, including fluctuation of solar radiation, air temperature, and relative humidity, to be maintained or at the least approximated.

Each collector was equipped with a small fan with a flow rate of approximately 340 L min^{-1}. The fan provided an air exchange rate in the collector chamber of approximately six volumes min^{-1}. This air exchange rate was sufficient to keep chamber temperature near ambient levels. With this air flow rate, possible reduction of the diffusion gradient caused by the build up of gaseous ammonia in the collection was assumed to be minimal.

Each unit also was equipped with a bubbler collection system with 0.1 N H_2SO_4 absorbing solution to trap NH_3 and a small diaphragm pump with a 1.6 L min^{-1} flow rate. Operation of each ammonia collection system included 56 min h^{-1} with the fan on and the bubbler off (Fig. 1a) and four min h^{-1} with the bubbler on and the fan off (Fig. 1b). The concentration of ammonia in the absorbing solution after each time period was determined as noted in Methods of Air Sampling and Analysis (Lodge, 1989).

The efficiency of ammonia collection systems was evaluated with three 20 cm constant-rate NH_3 emission tubes. By comparing the total NH_3 emitted per time period with the total NH_3 collected per period during the four min h^{-1} collection periods, the efficiency of the collection systems was calculated. A flow rate correction based on bubbler flow rates compared to the flow rate used in the efficiency test was also utilized.

Properties of surface soil samples from two soil series, one gravelly and one nongravelly, were evaluated (Table 1). Chilicotal very gravelly loam (loamy, skeletal, mixed thermic Ustollic Calciorthid) and Berino loam taxajunct (fine loamy, mixed, thermic, Ustollic Haplargid) were selected because of their similar chemistry; both soils have similar soil textures and parent

Table 1. Initial (prestudy) average sludge and selected soil surface properties		
Sludge Property	**Mean Value**	
pH	6.6	
NH_3-N (g kg^{-1})	17.0	
TKN (g kg^{-1})	52.1	
Water Content (%)	74.0	
Soil Property	**Chilicotal Soil**	**Berino Soil**
CEC (cmole kg^{-1} soil)†	20.4	22.5
$CaCO_3$ Content (g kg^{-1})	7.78	4.50
Sand (g kg^{-1})	595	670
Silt (g kg^{-1})	260	183
Clay (g kg^{-1})	145	147
pH	7.80	7.74
Organic C (mg kg^{-1})	3500	3600
TKN (mg kg^{-1})	625	496
Extractable NH_4^+ (mg kg^{-1})	1.91	1.73
NO_3^--N (mg kg^{-1})	19.1	20.3

†These values seem unreasonably high in relation to the low clay content of these soils.

materials. Soils with similar chemistry were selected to minimize the differences of soil chemistry influence on ammonia volatilization.

A 4.5 kg sample of Berino loam was placed in the base of nine ammonia collectors. A 4.0 kg sample of Chilicotal very gravelly loam was placed in the base of the nine collectors with 0.50 kg of the natural surface gravel placed on the soil's surface. Large clumps of plant matter were removed prior to sludge application. To reestablish a soil crust, each plot received 0.25 L of tap water applied as simulated rainfall distributed evenly over the soil surface. The plots were left overnight to dry.

Immediately prior to biosolid application, samples were collected for determination of selected parameters. Biololids were applied midday at 0.0, 6.7, and 17.9 dry Mg ha^{-1}. These rates were chosen to correspond to a control group (0.0 Mg ha^{-1}), the current level of biosolid application at a semiarid/arid range site near Sierra Blanca, Texas (6.7 Mg ha^{-1}), and a possible future increased rate (17.9 Mg Ha^{-1}).

Immediately after biosolid application, an ammonia collector system was placed over each base. The collectors were set to bubble for the first four-minute ammonia collection period after 55 min and again every hour for 210 h. The first time period was a 6 h period corresponding to the afternoon hours. Starting with time period number two, the time periods were 12 h and corresponded to daytime and nighttime periods. The purpose of time periods was to differentiate daytime ammonia volatilization losses from nighttime losses.

A completely randomized design was used to assign treatments, each with three replicates, to plots within the experimental area. A repeated measures technique was used to test the influence of time (corresponding to the time periods mentioned above) on ammonia volatilization. Volatilization tests were performed under hot (20 to 36°C), cool (-4 to 14°C), and intermediate (6 to 23°C) temperature regimes during 1993 and 1994 to evaluate ammonia volatilization from biosolids under various seasonal conditions.

Results and Discussion

Several relationships were apparent when the volatilization studies were compared. No statistical comparison can be made between all three studies, as night volatilization losses were not measured in January; therefore, these relationships are based on observation. Ammonia volatilization from applied biosolids followed a diurnal fluctuating pattern for both the hot and intermediate temperature regime trials (nighttime temperature extremes precluded measurements in the cool-season trial) and was significantly reduced by day three for all of the trials (Fig. 2, 3, 4). Hargrove (1988) cited many studies indicating diurnal fluctuation; therefore, this result was anticipated. Cumulative volatilization data are presented in Fig. 5, 6, and 7. At the 6.7 dry Mg ha^{-1} application rate, cumulative (210 h) ammonia volatilization losses ranged from 11.5 kg ha^{-1} in the cool-season trial (Fig. 7) to 18.3 kg ha^{-1} in the hot-season trial (Fig. 5). These represent a range of 9.5 to 16.6% of applied NH_3-N. At the 17.9 dry Mg ha^{-1} application rate, cumulative (210 h) volatilization losses ranged from 26.2 kg ha^{-1} in the cool-season trial to 35.5 kg ha^{-1} in the hot season trial. These represent a range of 8.3 to 12.1% of applied NH_3-N.

Statistical analysis of the data showed that the influence of biosolid application rate and time after application were significant effects ($P < 0.01$) during the first 210 h after application, as initial volatilization is controlled by environmental factors. Potential evapotranspiration (Table 2) seemed to play a large role in volatilization, especially over the first few days following sludge application.

Table 2. Potential evapotranspiration (PET) in mm d^{-1}, calculated using MacPet® (Lascano and Salisbury, 1993).

Hot Season		Cool Season		Intermediate	
Day	PET	Day	PET	Day	PET
8/7/93	5.2	1/7/94	2.6	3/12/94	2.0
8/8/93	7.7	1/8/94	4.3	3/13/94	4.5
8/9/93	9.8	1/9/94	5.0	3/14/94	7.5
8/10/93	8.0	1/10/94	4.1	3/15/94	6.9
8/11/93	8.2	1/11/94	3.0	3/16/94	7.7
8/12/93	8.7	1/12/94	3.3	3/17/94	14.2
8/13/93	6.6	1/13/94	3.8	3/18/94	9.8
8/14/93	8.3	1/14/94	3.9	3/19/94	10.7
8/15/93	7.0	1/15/94	1.9	3/20/94	13.3

This PET/volatilization relationship was clearly evident in the intermediate temperature trial (Fig. 8). On day one, when the PET (Table 2) was extremely low (2.0 mm) because of relatively low temperature and high humidity, little ammonia loss occurred when compared to day one of the other studies. Then, on day two as PET increased to 4.5 mm, a large increase in volatilization occurred.

In the hot-season trial with relatively high PET values, the largest first day losses occurred, but volatilization rates were extremely low by day three. In comparison, in the cool-season trial with much lower PET values, the day one losses were much smaller, but substantial volatilization occurred through day three. These observations indicate that slow drying allows significant volatilization to occur for a longer period than does rapid drying.

Comparison of cumulative losses between the studies is difficult for several reasons. First, nighttime losses were not measured in the January study. Also, the relatively low PET value of day one in the March study led to an unusual volatilization pattern compared to the other studies. Generally, cumulative volatilization losses were highest for the August study and lowest for the January study (even with estimated nighttime losses included in the January totals). The same pattern exists for cumulative PET values over the first three days following application with the highest totals for August and the lowest total for January. Only the first three days were considered because 77%-94% of total volatilization losses occurred over the first three days in each study.

The short time of soil/sludge contact might also explain why the influence of soil type (gravelly versus nongravelly) was not significant. In the initial period following sludge application, environmental parameters have an overriding influence on volatilization. In time, as the soil/sludge interaction becomes more intimate, soil/sludge interaction, as well as environmental conditions, would be expected to make a significant impact on volatilization. The magnitude of soil/sludge interaction would be different for gravel-covered soils and gravel-free soil surfaces, possibly resulting in significant volatilization differences.

Because of the importance of PET, a model of ammonia volatilization as a function of PET would be useful. In this model, an estimate of the length of phase one and phase two drying would be necessary. The length of phase one is especially important because of the large amount of NH_3 volatilized during that time. It was assumed that phase one drying lasted for 12 h; however, for low (<2.3 mm d^{-1}) PET on the day of application, a 36 h duration of phase one was assumed to be more accurate.

Model Development

A model is being developed based on the assumption that volatilization occurs by NH^3 diffusion from the biosolid-water solution and along with evaporation of the biosolid water. Specifically, volatilization is controlled by diffusion and drying parameters which are modeled using potential evapotranspiration (PET). In order to use this model to predict cumulative volatilization losses as a function of time after application, the following inputs are needed: biosolid NH^3-N content, biosolid water content, daily PET values, biosolid application rate, and maximum temperature on day of biosolid application.

The first step in model development involves deriving a drying potential approximation equation given daily PET values. Drying of biosolids and volatilization occurs in two phases similar to the drying phases of soil described in Ritchie (1972). Therefore, the influence of drying potential on volatilization was separated into two phases. In phase one, cumulative drying potential from application time to 12-36h. The evaporative surface area of biosolid also decreases during phase one drying. For phase two drying, the cumulative drying potential varies as the square root of time Ritchie (1972). The total amount of NH_3-N volatilized at time since biosolids application is determined by summing phase one and phase two drying.

In developing this predictive model, a proportionality coefficient for the cumulative loss will first be determined by comparing measured and predicted values with the least-squares fit method for individual data sets (treatment means averaged across soil type for each trial). The proportionality constant will need to consider the maximum temperature on day of application and rate of applied biosolids.

Summary

A predictive model being developed in this study allows users to input typical (site-and season-specific) PET data, biosolid application rate, NH_3-N content of sludge, H_2O content of sludge, and the maximum temperature on day of application (T_{max}) and estimate loss of PAN due to ammonia volatilization. This model should assist regulators and operators of semiarid/arid rangeland application projects in achieving the proper balance between plant N needs and N application rates of surface-applied biosolids. An environmentally sound management plan would attempt to achieve this N balance, and, therefore, eliminate negative environmental impacts of excess N loading.

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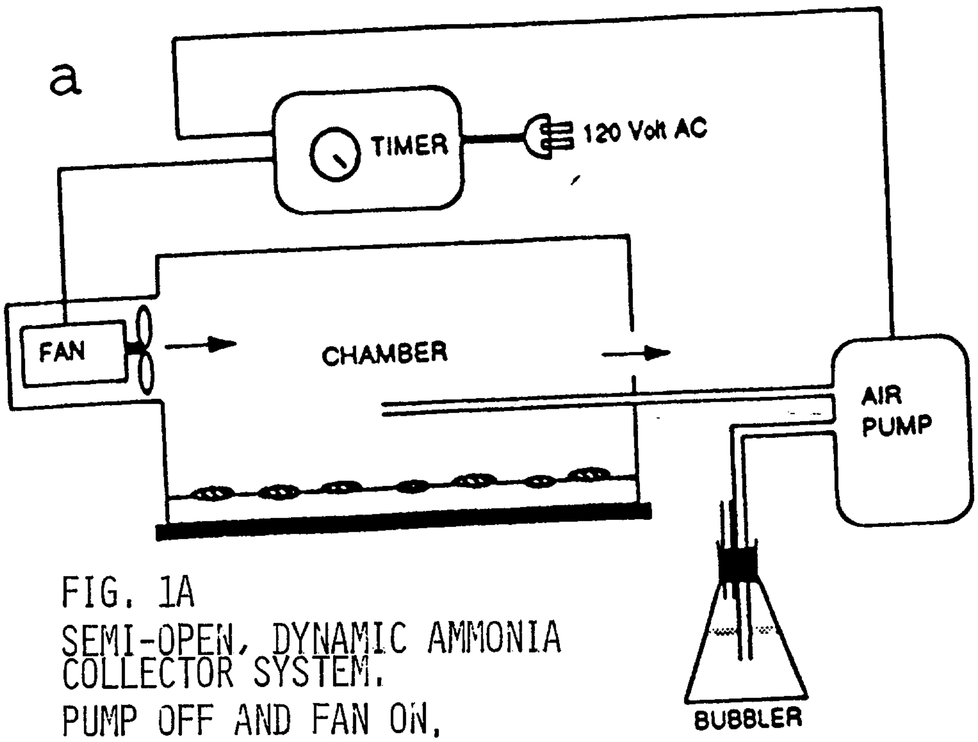


FIG. 1A
SEMI-OPEN, DYNAMIC AMMONIA COLLECTOR SYSTEM.
PUMP OFF AND FAN ON.

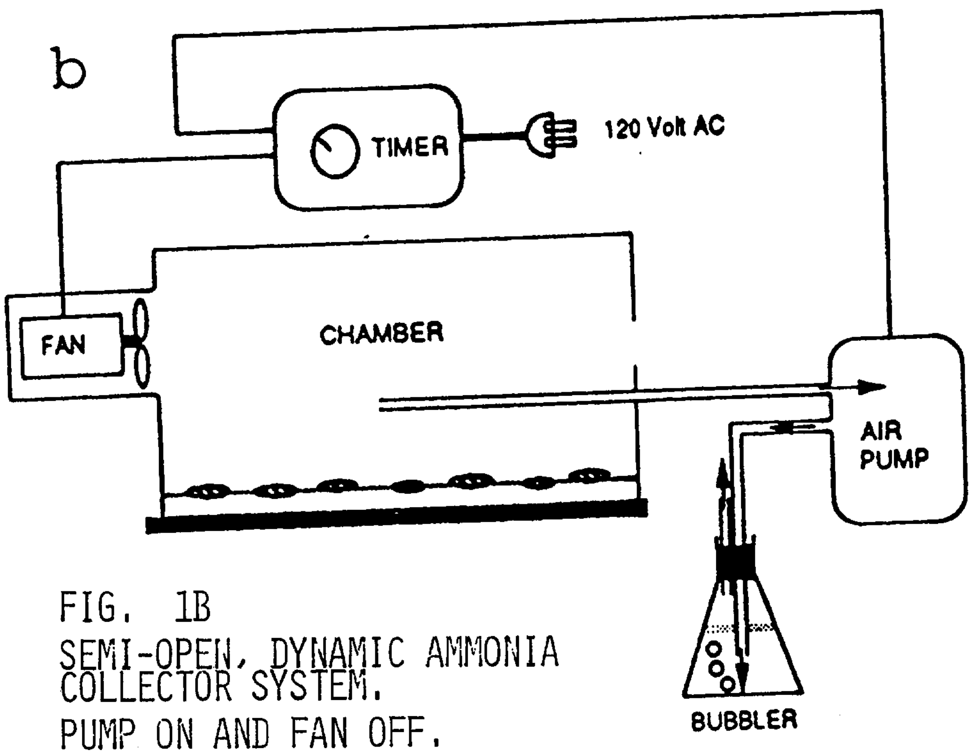
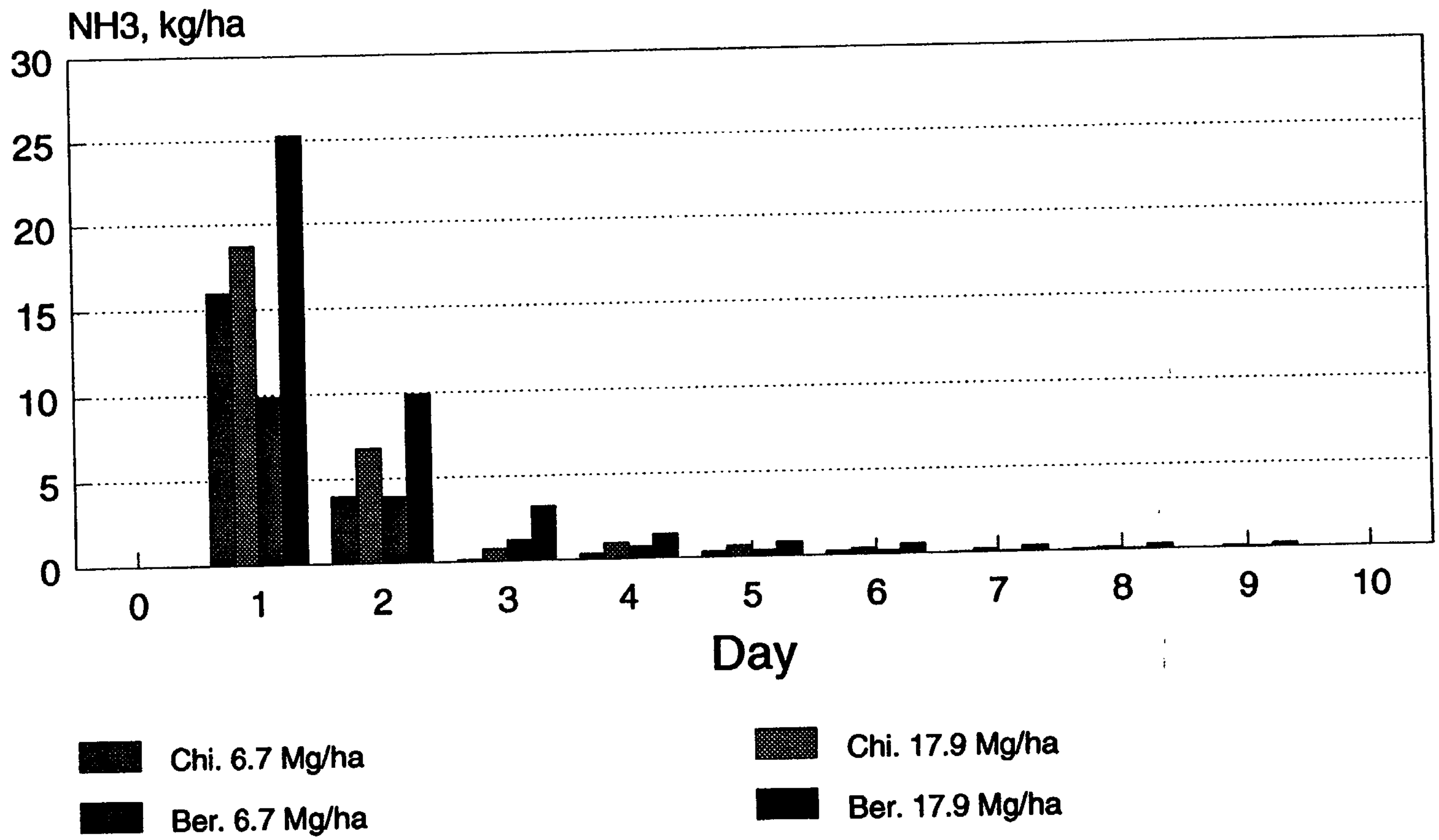


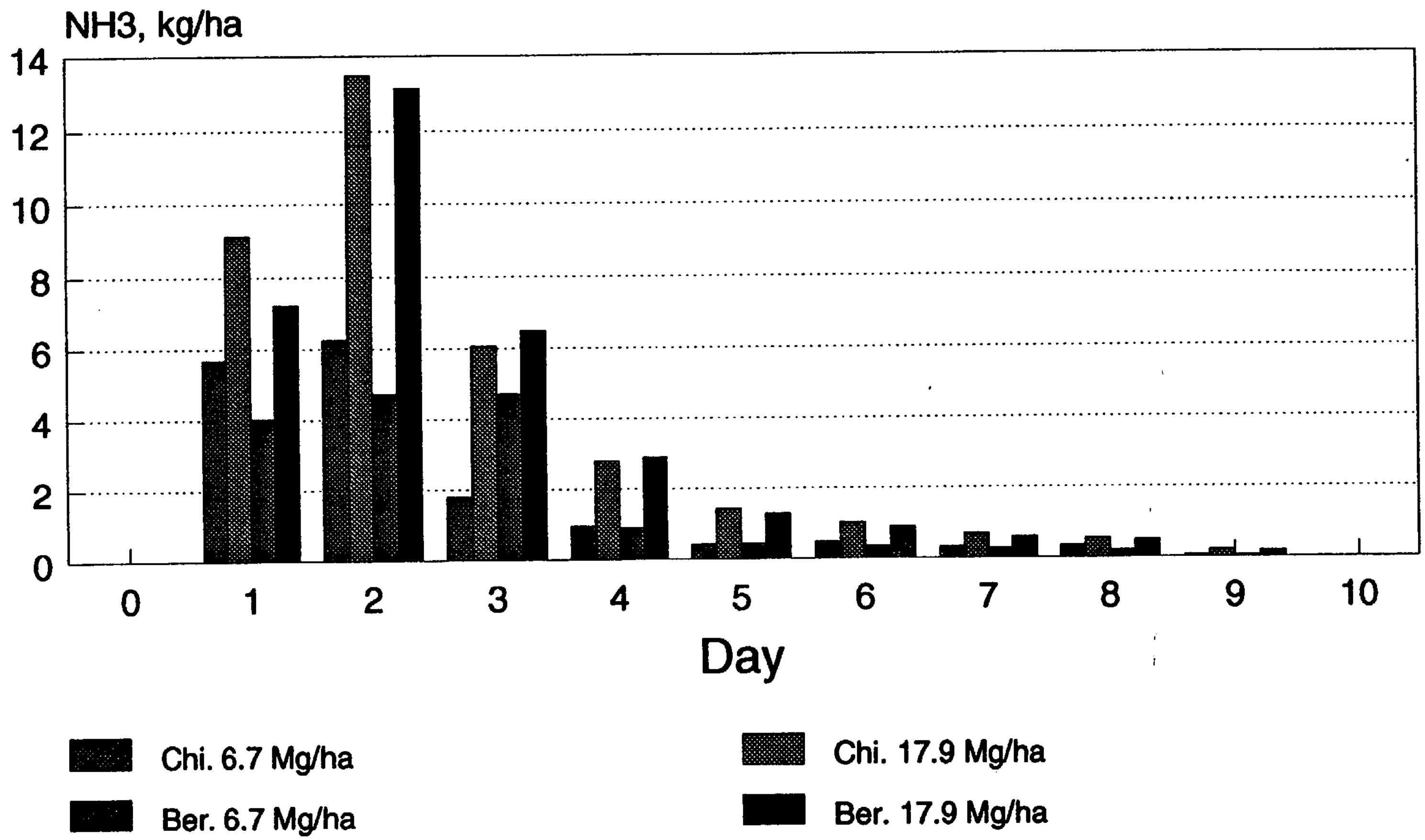
FIG. 1B
SEMI-OPEN, DYNAMIC AMMONIA COLLECTOR SYSTEM.
PUMP ON AND FAN OFF.

Fig. 2. Daily ammonia loss (August 1993).



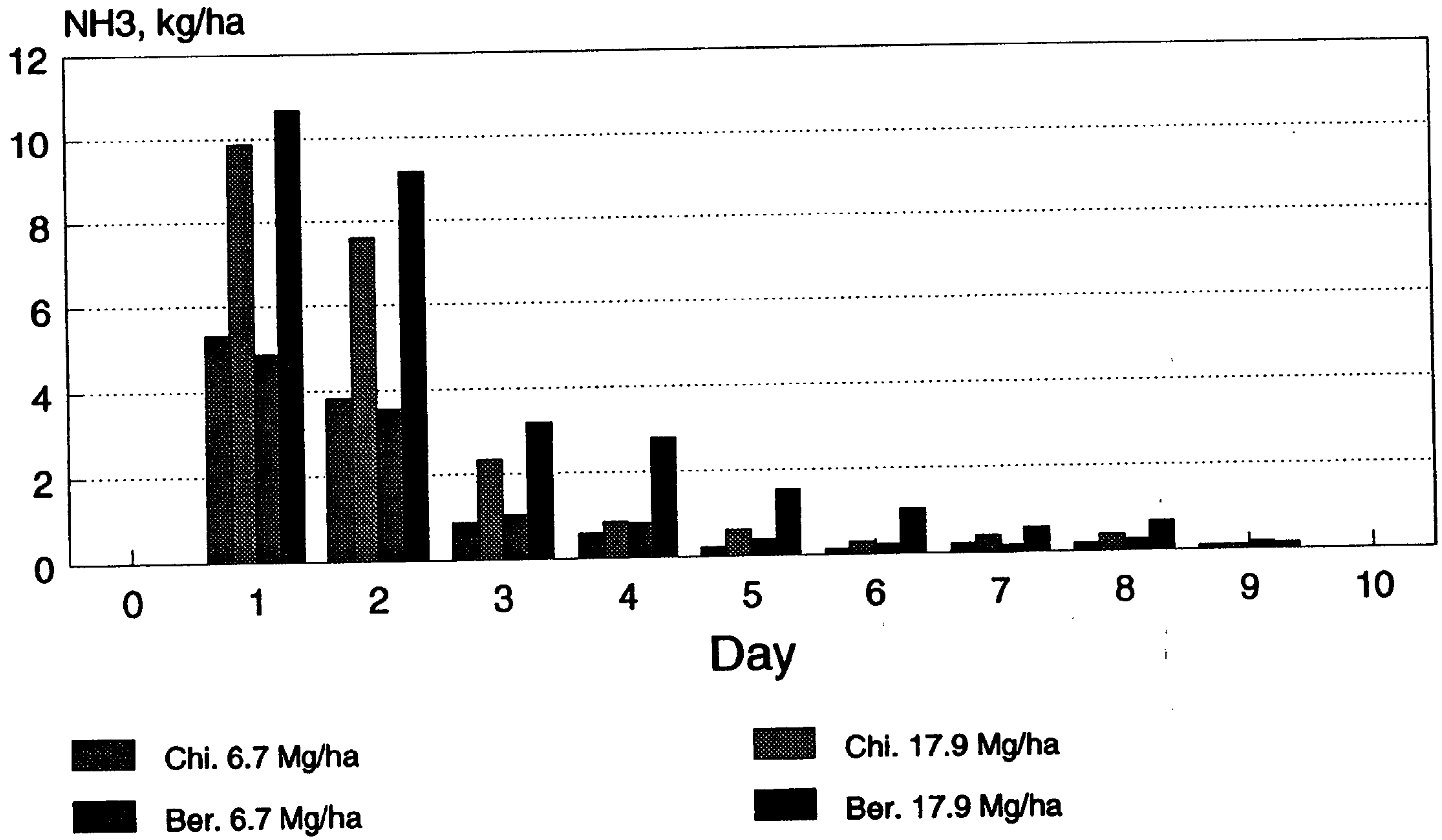
Chi. is Chilicotal; Ber. is Berino

Fig. 3. Daily ammonia loss (March 1994).



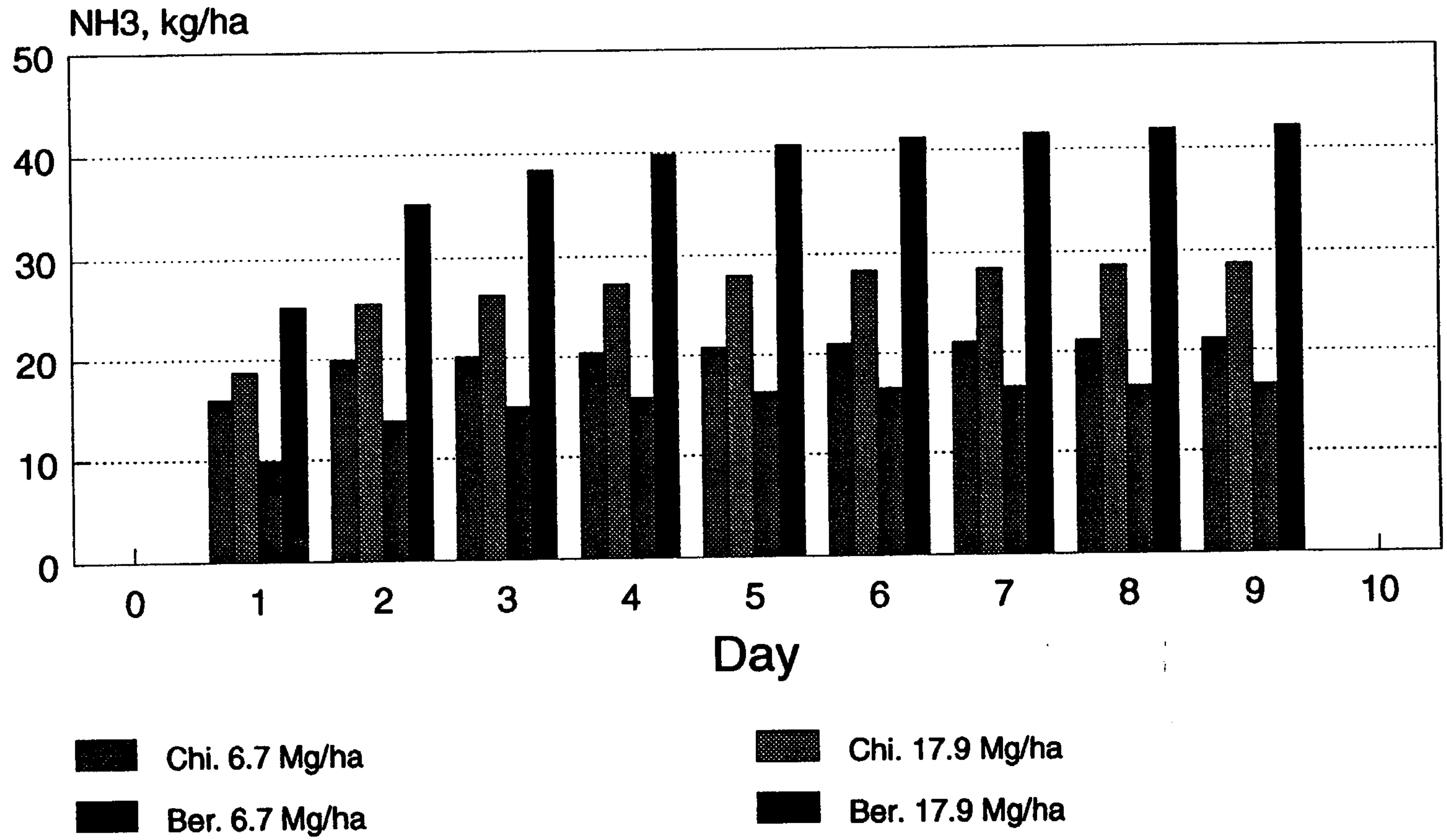
Chi. is Chilicotal; Ber. is Berino

Fig. 4. Daily ammonia loss (January 1994).



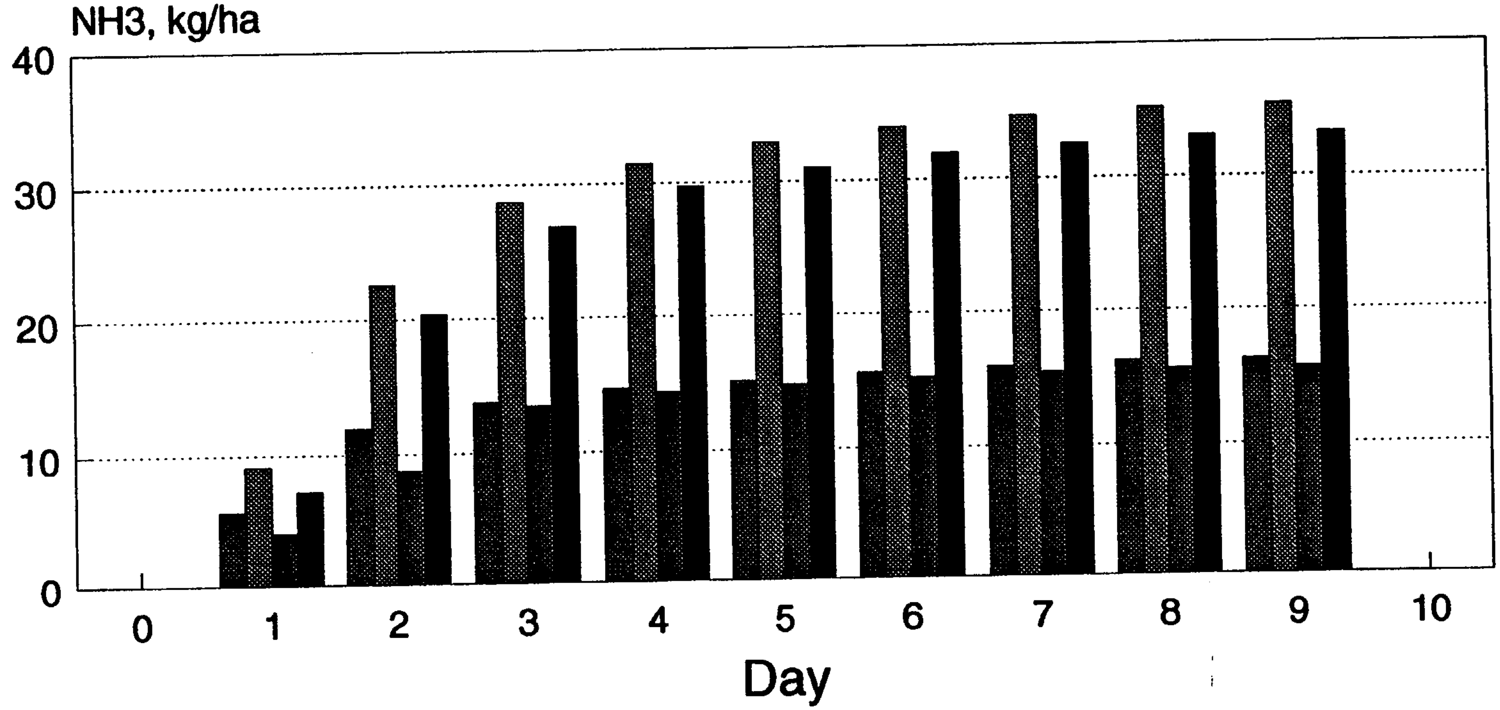
Chi. is Chilicotal; Ber. is Berino

Fig. 5. Cumulative ammonia loss (August 1993).



Chi. is Chilicotal; Ber. is Berino

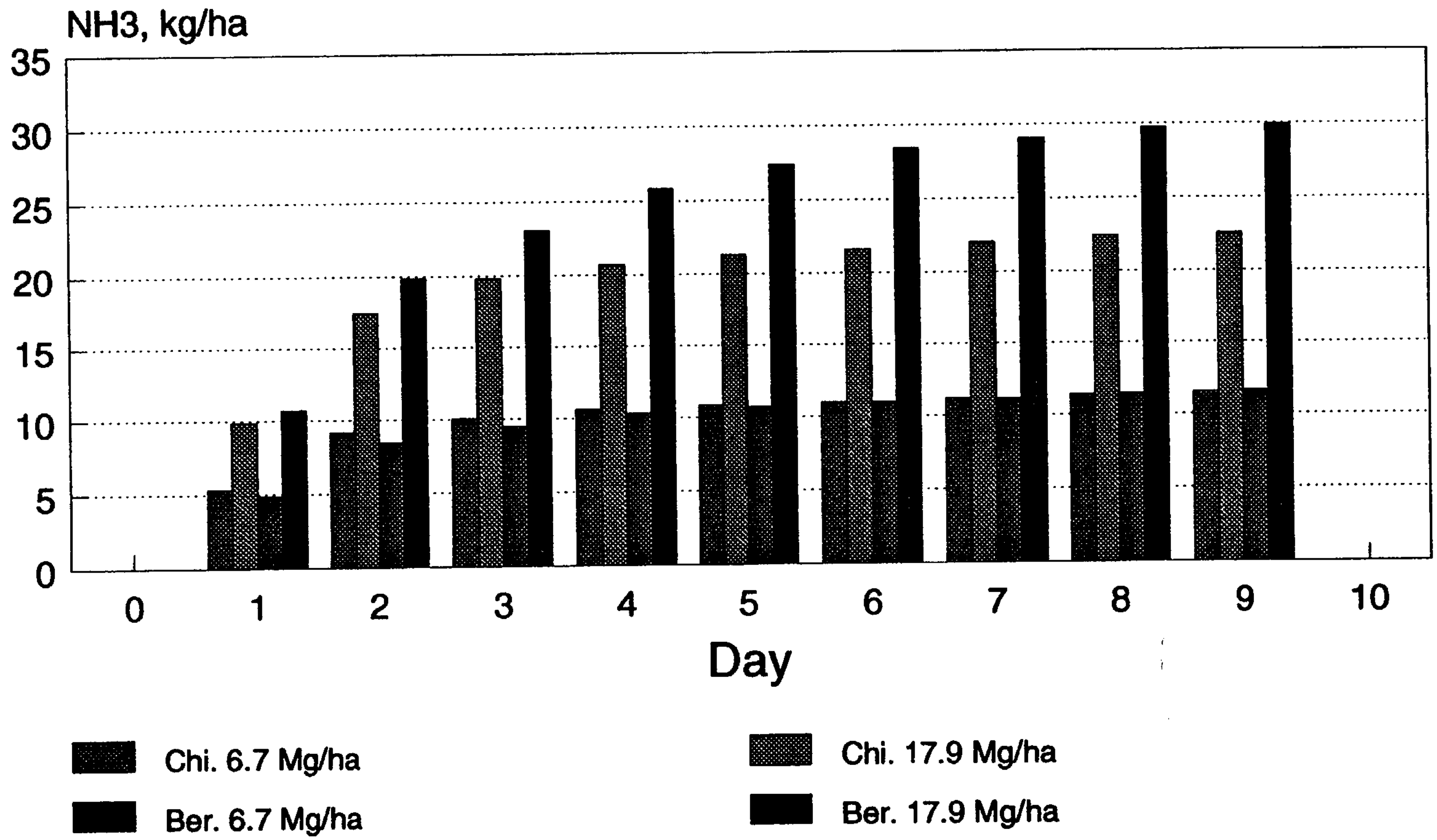
Fig. 6. Cumulative ammonia loss (March 1994).



Chi. 6.7 Mg/ha
Ber. 6.7 Mg/ha
Chi. 17.9 Mg/ha
Ber. 17.9 Mg/ha

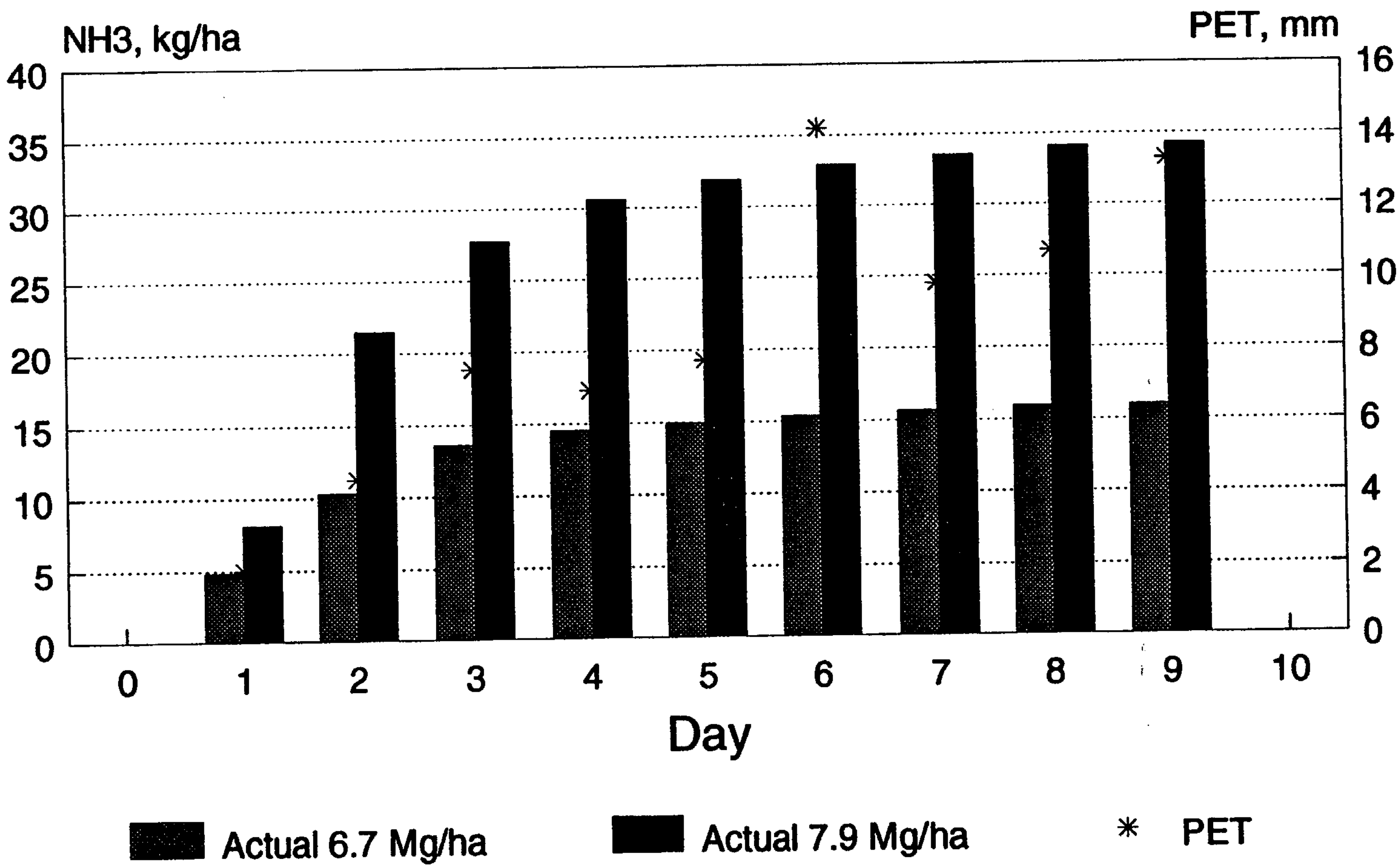
Chi. is Chilicotal; Ber. is Berino

Fig. 7. Cumulative ammonia loss (January 1994).



Chi. is Chilicotal; Ber. is Berino

Fig. 8. Cumulative vs. PET (March 1994).



Chi. is Chilicotal; Ber. is Berino

References

Adamson, F.J., and B.R. Sabey. (1987) Ammonia volatilization from liquid digested sewage sludge as affected by placement in soil. *Soil Sci. Soc. Am. J.* 51, 1080-1082.

Beauchamp, E.G., *et al.* (1978) Ammonia volatilization from sewage sludge applied in the field. *J. Environ. Qual.* 7, 141-146.

Beauchamp, E.G., *et al.* (1982) Ammonia volatilization from liquid dairy cattle manure in the field. *Can. J. Soil Sci.* 62, 11-19.

English, C.J., Jr., *et al.* (1980) Volatile ammonia losses from surface-applied sludge. *J. Water Poll. Control Fed.* 52, 2340-2350.

Fine, P., *et al.* (1989) Incubation studies of the fate of organic nitrogen in soils amended with activated sludge. *Soil Sci. Soc. Am. J.* 53, 444-450.

Freney, J.R., *et al.* (1983) Volatilization of ammonia. p. 1-32. In Gaseous Loss of Nitrogen from Plant-Soil Systems. J.R. Freney and J.R. Simpson, (Eds), Martinus Nijhoff/Dr. W. Junk Pub., The Hague, Netherlands. 367 pp.

Frost, J.P., *et al.* (1990) Effect of separation and acidification of cattle slurry on ammonia volatilization and on the efficiency of slurry nitrogen for herbage production. *J. Agri. Sci.* 115: 49-56.

Hargrove, W.L. (1988) Evaluation of ammonia volatilization in the field. *J. Prod. Agri.* 1: 104-111.

Lascano, R., and D. Salisbury. (1993) MacPet(R). Tex. A&M Misc. Pub. 3181.

Lodge, J.P., (Ed). 1989. In Methods of Air Sampling and Analysis. 3rd Edition. Lewis Pub., Chelsea, MI. 763 pp.

Marshall, U.G., and D.S. Debell. (1980) Comparison of four methods of measuring volatilization losses of nitrogen following urea fertilization of forest soils. *Can. J. Soil Sci.* 60: 549-563.

Ritchie, J.T. (1972) Model for predicting evaporation from a row crop with incomplete cover. *Water Resou. Res.* 8: 1204-1213.

Environmental Effects on the Degradation of Biosolids by Soil Microbes

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David B. Wester

Methodology Development and Initial Results

In the past, research on the microbial degradation of biosolids resulting from the land application has been centered in humid regions such as the Eastern United States. Comparatively little is known about the microbe/biosolid interactions which occur in arid areas, such as desert grasslands, which are characterized by low rainfall and above freezing temperatures. Land application on arid and semi-arid rangelands has many distinct advantages, including forage improvement and decreased environmental risk (Sosebee 1994, Wester 1994, Aguilar and Loftin 1994). As application sites in the American Southwest increase in size and number, a knowledge of the microbial degradation rates of biosolids in these environments is essential.

Prior studies of soil microbial populations have utilized methods, including light microscopy, the "Most Probable Number" method, and enumeration of plated cultures to determine changes in the soil microbial populations (Weaver, et al. 1994). These methods require the destruction of the soil sample in order to determine population size. Multiple samples must be taken in order to determine population changes over time and the inherent variation in populations from sample to sample may lead to misleading results in these studies. This paper introduces a new method which utilizes a single soil sample from which repeated measurements are obtained. Soil microbial populations remain viable throughout the sampling period. In addition, this method is integrated into environmental chambers, thus allowing for the regulation of physical conditions. The goal in developing this technique was to provide an *in-situ* method to study the long-term degradation of biosolids by microbes in an arid soil. However, the method may also be applied to other soil microbial population studies.

The development of this "live" method arose from an interest in determining the rates of the microbial degradation of biosolids at a land application site near Sierra Blanca, Texas. This site is located in the Trans Pecos region of the Chihuahuan Desert. The site is classified as a desert grassland with *Hilaria mutica, Bouteloua gracilus and Sporobolus airoides* as the dominant grasses (Wester 1994). Biosolids are being commercially applied to improve the condition of the rangeland for future grazing. Since little information on degradation rates in such an environment is available, research in this area may provide guidelines for future application practices. In addition, this research can provide insight into the ecology of microbial populations in desert soils.

This study utilizes CO_2 evolution to measure changes in microbial population size. Measuring CO_2 evolution and relating it to changes in population size has the advantage of quick analysis of samples through infrared spectroscopy. In contract to other methods, the analysis of daily samples can be completed in minutes instead of hours or days. In the past, microbial biomass has been measured by the "fumigation" method (Weaver et al. 1994). Since this method is time consuming and destroys individual soil samples, a new method was needed. The interest of our research centered around differences in microbial population size over time instead of total biomass at a single point; thus, fumigation and sample destruction was unneccessary.

Given the environmental conditions in the Trans Pecos, the question of the effects of moisture and temperature on microbial degradation of biosolids is of interest. The "live" method discusses here integrates CO_2 measurement into environmental chambers. The chambers used in the study were built and adapted specifically for this technique and are somewhat simplistic when compared to those available on the market. However, they do provide a basis from which the technique can be modified in the future.

Materials and Methods

This ongoing research currently includes three different methods: traditional microbial population plating and enumeration, a benchtop $\mathbf{CO_2}$ evolution study, and the newly developed "live" method for investigating the effects of temperature and moisture on microbial populations.

Microbial Population Enumeration

Initial testing of the soils at the Sierra Blanca site involved the standard method of microbial enumeration from a soil sample (Weaver, Angle and Bottomley 1994). Stellar clay loam (Fine, montmorillonitic, thermic Ustollic Haplargid) samples were collected in July of 1994 from existing research plots maintained by Texas Tech University. The plots sampled represented five different application rates and three times of application. Application rates sampled were 0, 7, 18, 34, and 90 Mg/ha. Applications were made in July 1993, January 1994, and April 1994. Soil samples were stored on ice and transported to the laboratory within 24 hours. In the laboratory, soil samples were divided into five 1.0g subsamples. Each subsample was stirred into 99mL of FTA Hemagglutination Buffer and placed on a wrist action shaker for 20 minutes. Serial dilutions of this solution were carried out until concentrations of 1g soil per 10,100, and 1000 liters were achieved. Each of these dilutions was then plated onto Triptic Soy Agar plates. After 24 and 48 hours of incubation at 30°C, plates were enumerated under 10x magnification. Only those plates with numbers of colonies ranging from 30 to 300 were counted.

Benchtop CO_2 Evolution Study

This study evaluated the feasibility of taking multiple CO_2 samples from growth chambers and to test the effect of autoclaving on soil and biosolid microbial populations and biosolid degradation. Stellar clay loam was collected from the application site. In order to gain homogeneity among the samples, soil was collected from locations that lacked vegetation and other types of surface cover. The top 5cm of the soil were sampled. These samples were stored on ice and returned to the laboratory, where they were homogenized by mixing and screening through a 4.76mm sieve (ASTM #4). Half the soil was autoclaved at 121°C and 19 psi for one hour. Autoclaved and nonautoclaved soil was divided into 1000g subsamples, each of which was placed into 2.0 Liter Erlenmeyer flasks. Biosolids were sampled immediately after removal from their shipment containers at the site and transported to the lab on ice. Biosolids were then autoclaved in the same manner as the soil, with half being autoclaved and half remaining as sampled. Biosolids were applied to the soil at rates of 0, 7, and 90 Mg/ha to achieve the following combinations of soil and biosolids: Soil: Biosolids; Autoclaved Soil: Biosolids; Soil: Autoclaved Biosolids; Autoclaved Soil: Autoclaved Biosolids.

Individual Erlenmeyers were fitted with two hole stoppers through which 3.2mm I.D. tubing had been threaded (Illustration 1). Tube #1 extended from 3cm beyond the top of each stopper through the stopper to 2.5cm beneath the surface of the soil. Tube #2 extended from 3cm below the bottom of each stopper through the stopper to an external length of 15cm. The external end of Tubing #2 was connected to a bacterial filter. Thirty centimeters of 3.2mm tubing connected the filters to diaphragm pumps (Penn-Plax Silencer #XP-SLUL). Pumps were set to deliver a flow rate of 0.5L/hr.

After setup, the experiment ran for 20 days. Air samples were taken periodically from the outlet of Tube #1 of each flask with a 10mL syringe. These samples were immediately injected into an infrared CO_2 analyzer (Beckman Model 365 equipped with a Hewlett Packard 3390A Integrator). The analyzer was calibrated with a 515ppm CO_2 standard. Integrated sample values were weighed against a calibration curve to obtain CO_2 concentrations for each sample. On day fourteen of the experiment, 60mL distilled H20 was added to each Erlenmeyer as a simulation of rainfall. The entire experiment was conducted at room temperature.

"Live" Method of CO_2 Evolution Analysis

Two experiments have been conducted utilizing this technique. The live method measures CO_2 from "growth chambers" set up similarly to the Erlenmeyer flasks in the benchtop study. However, each growth chamber is constructed from round, clear polycarbonate containers 18 cm in circumference and 19 cm in depth and fitted with airtight lids. Two glass tubes (3.2mm ID) have been installed in the side of each chamber: Tube A is 2.54cm long and centered in the wall of the chamber 3cm from the top while Tube B is 10cm in length and installed 2.5 cm from the bottom of the chamber so that the interior end of the tube is centered in the chamber. This end of Tube B is then covered with a nylon fabric sleeve (690 X 690 mesh/2.54cm), which prevents soil from clogging the tube. The exterior ends of both Tube A and B were connected to 30cm lengths of plastic tubing (3.2mm ID). Tube A serves as an air inlet to the growth chamber while Tube B acts as an air outlet. Copper-constantan thermocouple wire was installed in each chamber through a hold 5cm from the bottom of the container. The interior length of the wire is 8.9 cm. The holes made in the sides of the chambers to house the tubing and the thermocouple wire were sealed with silicone caulk on the exterior of the container (Illustration 2).

For each experiment, 24 growth chambers were constructed. Soil was sampled and homogenized in the same manner as in the Benchtop Study. To each chamber, 1850g of soil was added. This volume of soil brought the soil depth in each growth chamber to 5cm. At this point, the thermocouple wire is centered on the soil surface. Twelve of these chambers were "rained" on by adding 10% distilled water by volume (18mL) evenly across the soil surface. The surface was then covered with nylon mesh fabric (690 X 690 mesh/2.54cm). This fabric allows water, air, and microbes to pass through it while separating the soil from the biosolids. Thus, after the completion of each experiment, soil and biosolids can be sampled separately without cross contamination. Biosolids were applied on the top of the nylon fabric at a rate of 100g (dry mass) per chamber. This quantity of biosolids is sufficient to cover the entire surface of the soil. After application, lids were placed on the containers and secured with vinyl tape.

These growth chambers were housed in larger "environmental chambers" constructed from .3255 m^3 dormitory refrigerators. Four growth chambers were placed in each refrigerator. Six environmental chambers were adapted so that two operated at each of three temperatures-- 5, 22, and 35°C. In the coldest treatment, temperature was regulated with the thermostat equipped on each refrigerator. Since 22°C is equivalent to ambient laboratory temperatures, these refrigerators were not connected to power for the course of the study. In the warmest treatment, temperatures were regulated with household heating pads (Sunbeam King Size). Each heating pad was wrapped in 3 mil plastic and stapled to a wooden block which was then screwed to the door of the refrigerator so that the pad hung lengthwise (Illustration 3). Heating pad electrical cords exited the refrigerators via holes drilled in the doors. Temperatures were maintained with the heating pad's thermostat. Shelves in the refrigerators were adjusted so that an equal amount of air space surrounded each of the four growth chambers. Thermocouple wires from the growth chambers were fed through holes in the side of each refrigerator and connected to a recorder.

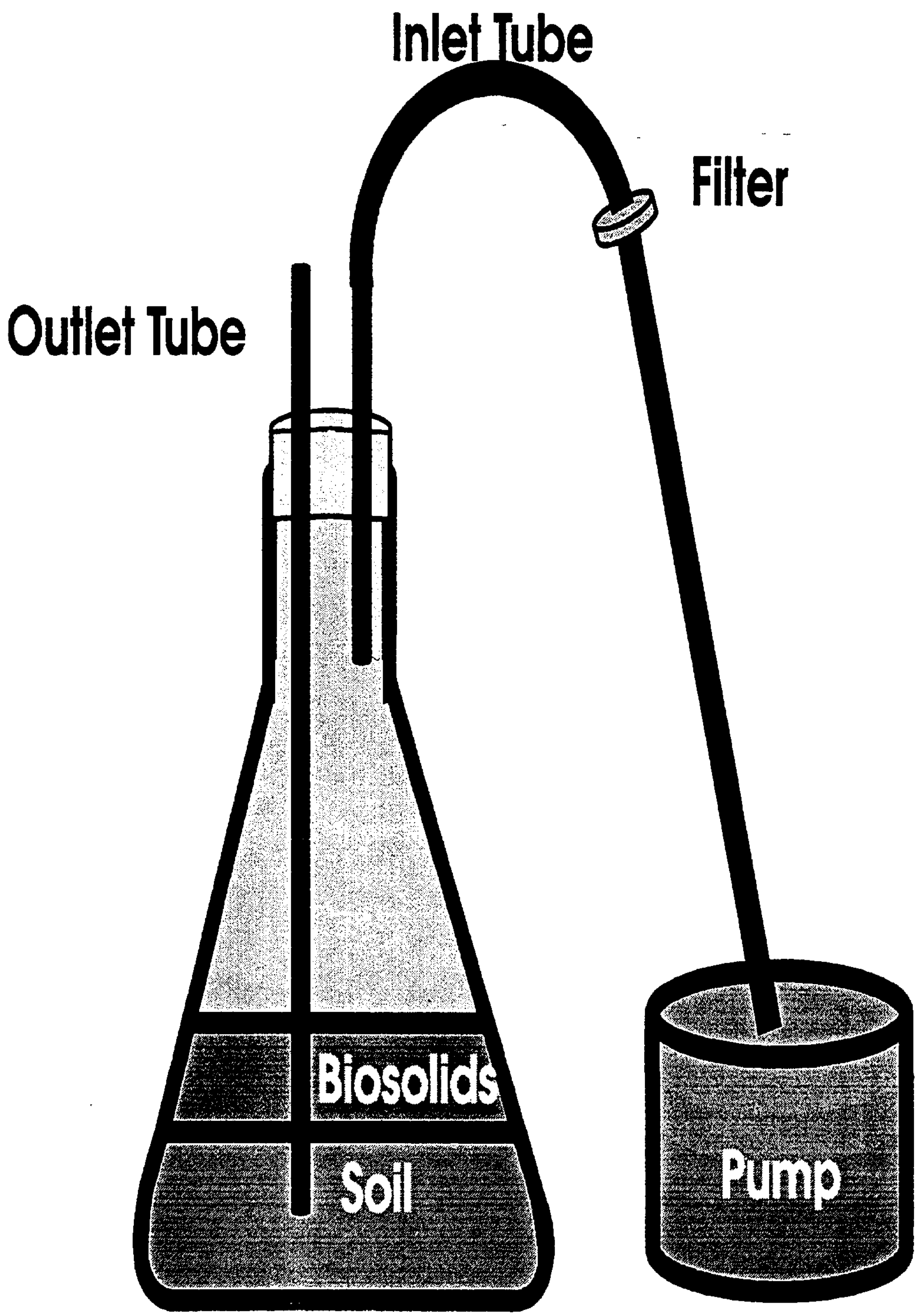
On the "inlet" side of each refrigerator, two #10 two-hole stoppers were placed in 3cm diameter holes. Glass tubing (3.2mm ID) was installed through each set of stoppers. The exterior end of this glass tubing was attached with plastic tubing (3.2mm ID) to a diaphragm pump on top of the refrigerator. Air filters (Gelman 4310) were installed in-line between the pumps and the refrigerators. The interior end of the glass tubing was connected with plastic tubing to a Tee-valve mounted on the wall of the refrigerator. Each Tee-valve splits off to two 45cm lengths of plastic tubing. This tubing is connected to the "inlet" tubing of each of the four growth chambers within the refrigerator. On the "sampling" side of each refrigerator, tubing from each container is connected directly to glass tubing running through stoppers to the outside of the refrigerators. Thus, individual samples can be taken from each growth chamber without opening the refrigerator.

One issue which required addressing was that of daily fluctuations in atmospheric CO_2 within the laboratory itself. In order to remove this artifact, we set up a benchtop control with each experiment. The basic construction of this control was the same as that of the growth chambers, but, instead of soil and biosolids, the containers were filled with autoclaved sand. Thus, CO_2, as measured from this control, would reflect ambient CO_2. Ambient CO_2 levels were then used as a baseline for growth chamber measurements.

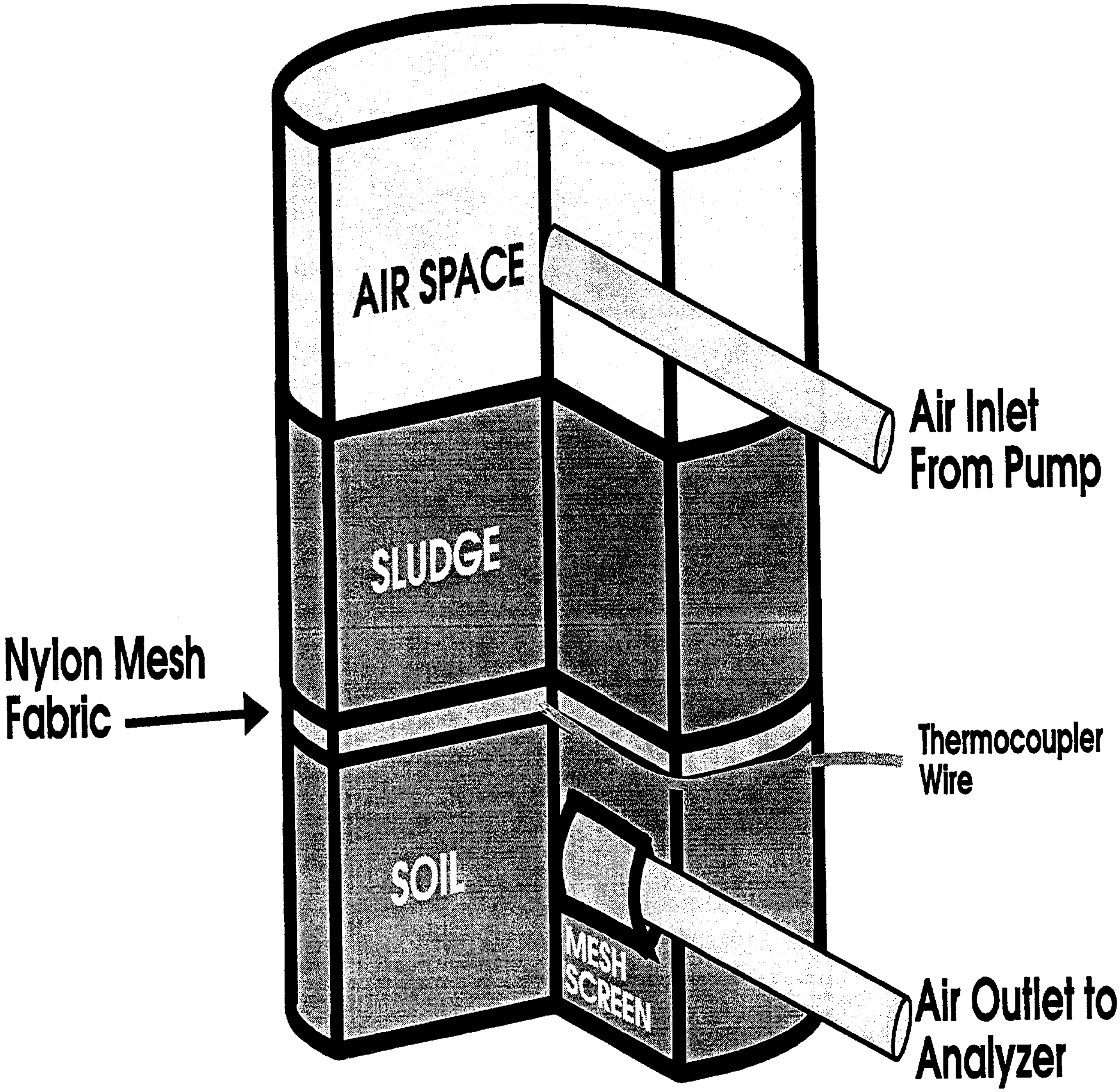
During the course of each experiment, samples were taken on a daily basis. Samples were drawn with a syringe directly from the sampling ports on each refrigerator and analyzed with infrared spectroscopy. In Experiment One, 3mL samples were taken, while 10mL samples were taken in Experiment Two. These sample sizes were based upon the expected CO_2 evolution compared with the standard calibration gas available at the time each experiment began. Experiment One was calibrated with a 515 ppm standard and Experiment Two was calibrated with a 1585 ppm standard. Both experiments lasted for one month.

ILLUSTRATION 1

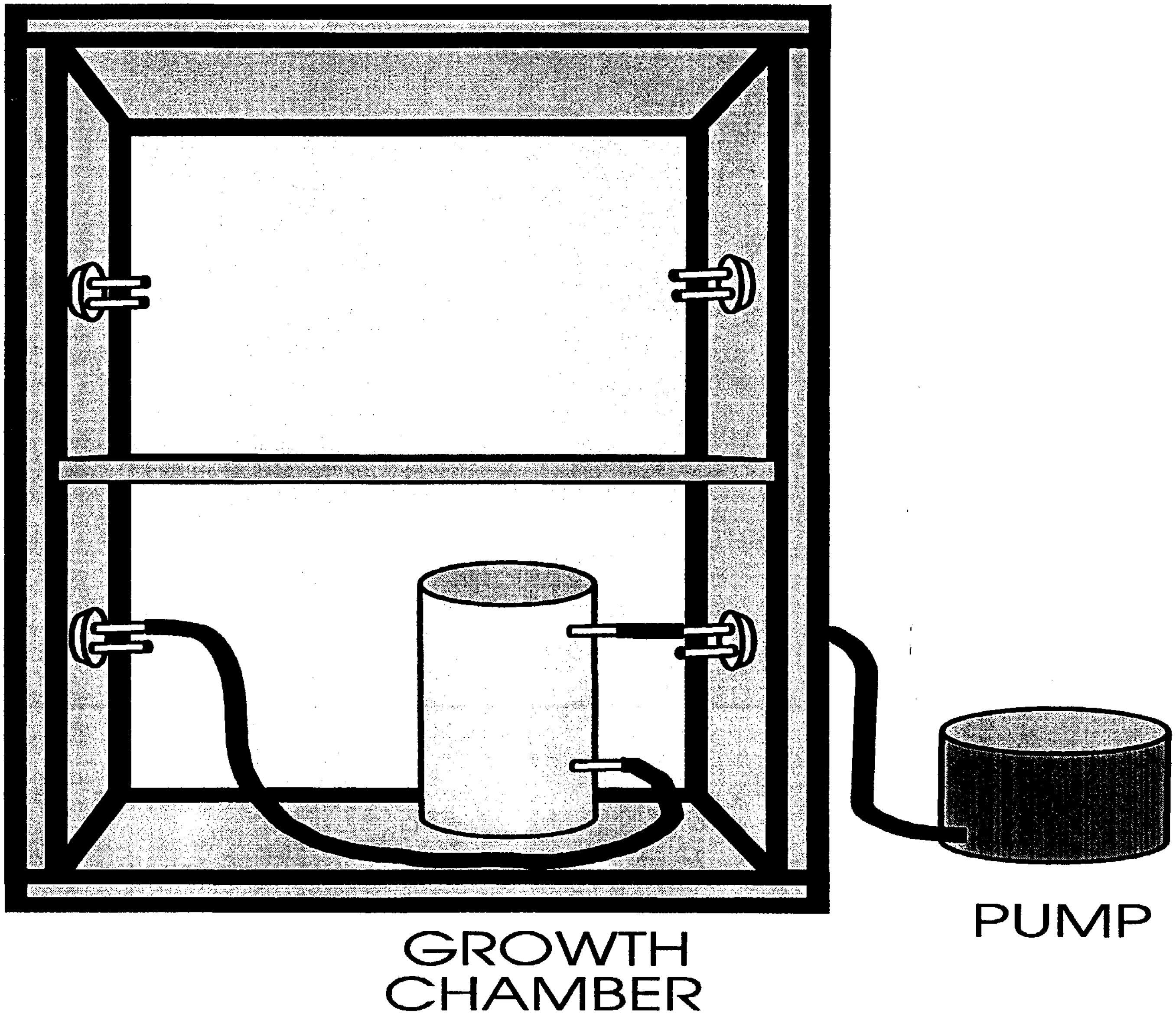
Benchtop CO^2 Evolution



GROWTH CHAMBER



ENVIRONMENTAL CHAMBER SETUP



In Experiment One, soil moisture levels in the 5°C treatments increased. Most likely, this was caused by the dewpoint temperature in the laboratory falling to below 5°C. Therefore, the second experiment included measurements of relative humidity and temperature, from which the daily dewpoint was calculated. During Experiment Two, the ambient dewpoint dropped below 5°C for only one day; thus, no significant wetting of samples occurred.

Results and Discussion

This is an ongoing study and all data have not been fully analyzed. Future results may shed light on current findings. However, our data, as measured to date, does raise some interesting questions. In the enumeration study, results indicate that microbial population size is a function of application rate and time since application of biosolids (Chart 1). Population sizes generally increased with increasing biosolid application rates while they decreased with time since application. These results prompted investigation into the actual rate of microbial degradation of biosolids. The methodology of addressing this problem is difficult. As previously mentioned, traditional methods are both destructive of individual soil samples and time consuming. Although variations between individual soil samples can be lessened with replication, the time involved to undertake such a study may be prohibitive. Thus, our goals in this study became two-fold: 1.) to determine the rate of microbial degradation of biosolids under different environmental conditions and 2.) to develop a new methodology that was nondestructive and less time consuming. Obviously, the second goal must be met before the first one can be sought out. To date, we have been involved primarily in meeting that second goal.

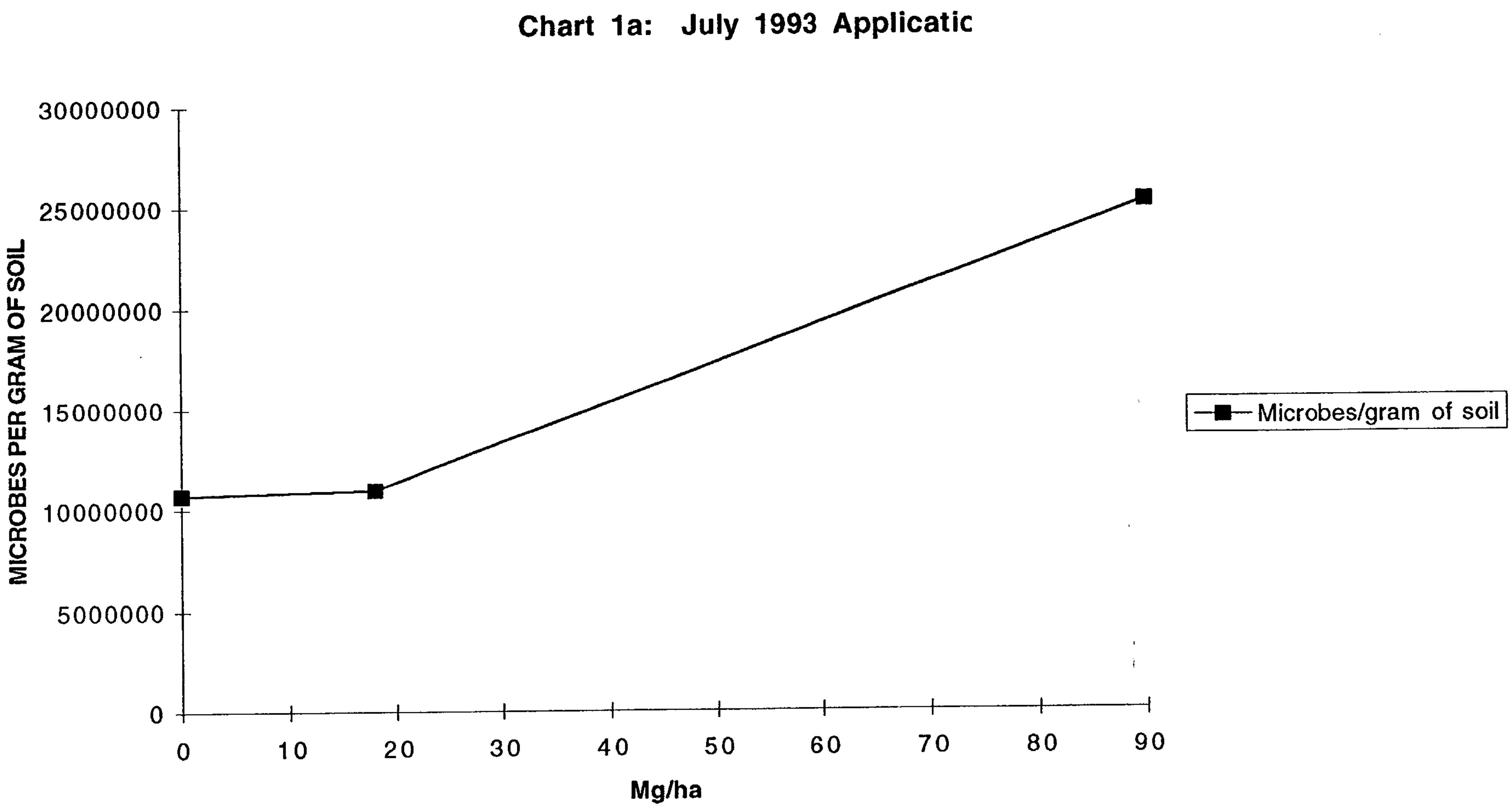
The benchtop CO_2 evolution study brought out several interesting factors. In the beginning, we had hoped to include an experimental control of autoclaved soil and autoclaved biosolids. By doing so, we hoped to block out CO_2 evolution by chemical factors. However, autoclaved samples evolved CO_2 in manners similar to non-autoclaved samples. In some cases, autoclaved biosolids appeared to be more readily degraded than untreated biosolids (Chart 2). In addition, autoclaving appeared to alter the physical and chemical composition of the soil and the biosolids. Thus, we determined that such a control would be useless in our study.

However, this study did show some promising results. As rate increased, CO_2 evolution also increased (Chart 2). This result was supported by the findings of our enumeration study. In addition, several of the replicates began to grow large fungal colonies on the sludge. This fungi began to grow at the soil-biosolid interface and spread over the entire surface of the biosolid layer. As the fungi spread, CO_2 evolution increased. At a later stage, when the fungal colonies appeared to die off, CO_2 evolution decreased (Chart 3). Thus, our system for measuring CO_2 evolution from the same soil samples over time appears to work well.

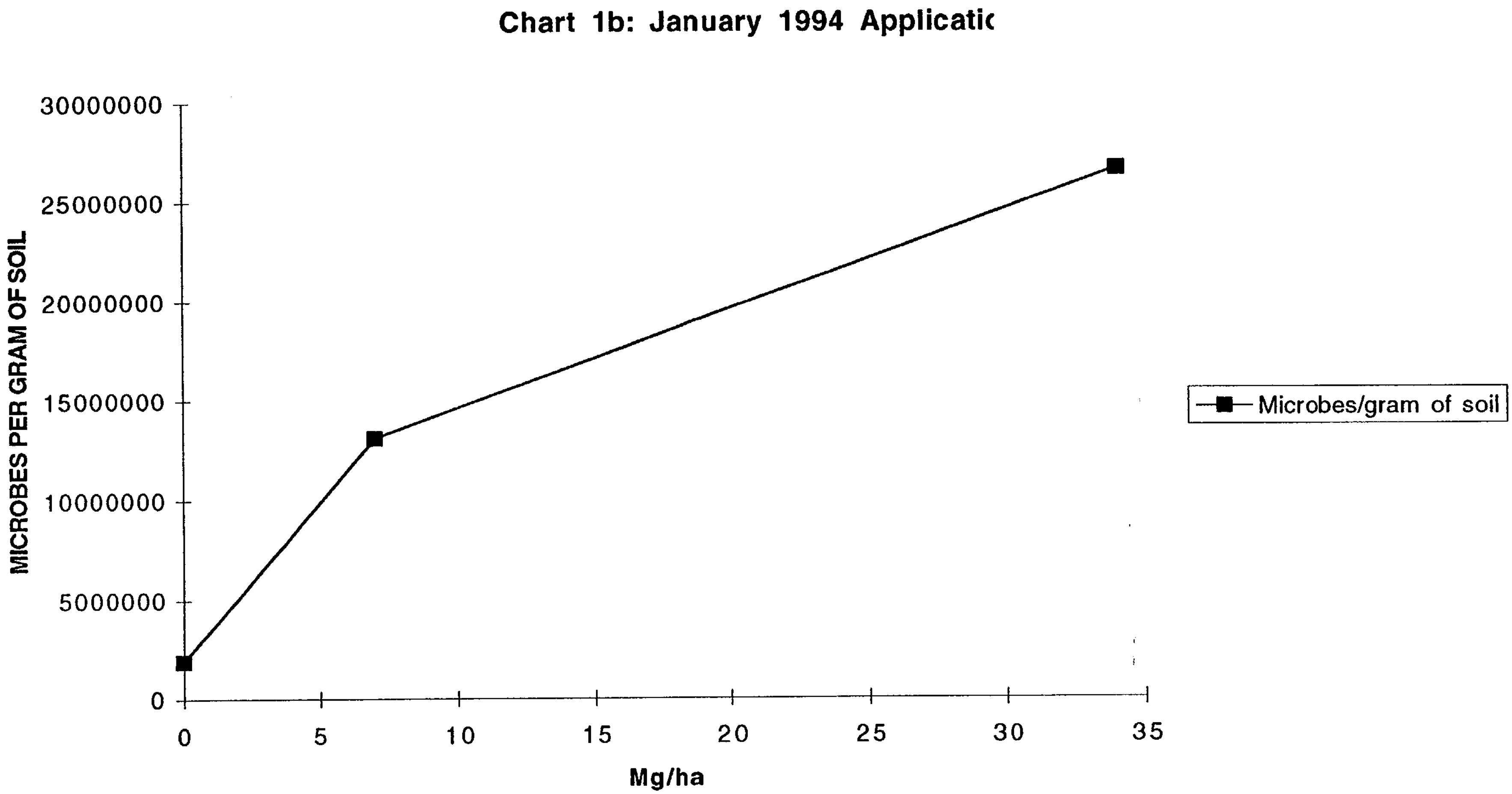
Some general inferences can be drawn from the results of both "live" method experiments. First of all, increasing CO_2 sample size in Experiment Two reduced fluctuations within treatments from those seen in Experiment One (Charts 4 and 5). In addition, the general trend of the microbial decomposition of biosolids when plotted over time occurs on a decreasing log scale.

Also, cumulative CO_2 evolution results for the experiments differ. In Experiment One, CO_2 evolution increased with increasing temperature and moisture (Chart 6). In Experiment Two, this trend is not followed (Chart 7). The disagreement between the two data sets may be caused by interference in Experiment One as a result of the smaller CO_2 sample size, or by seasonal effects on individual soil and biosolid samples. Experiment One samples were taken in September 1994 and Experiment Two samples were taken in January 1995. Future experiments should clear up the CO_2 sample size question. Currently, new studies are being planned to assess seasonal effects on experimental results and the impact of fluctuating soil moisture levels. At this stage, comments on actual degradation rates in arid environments are preliminary. Currently, we are pleased with the initial performance of our "live" method.

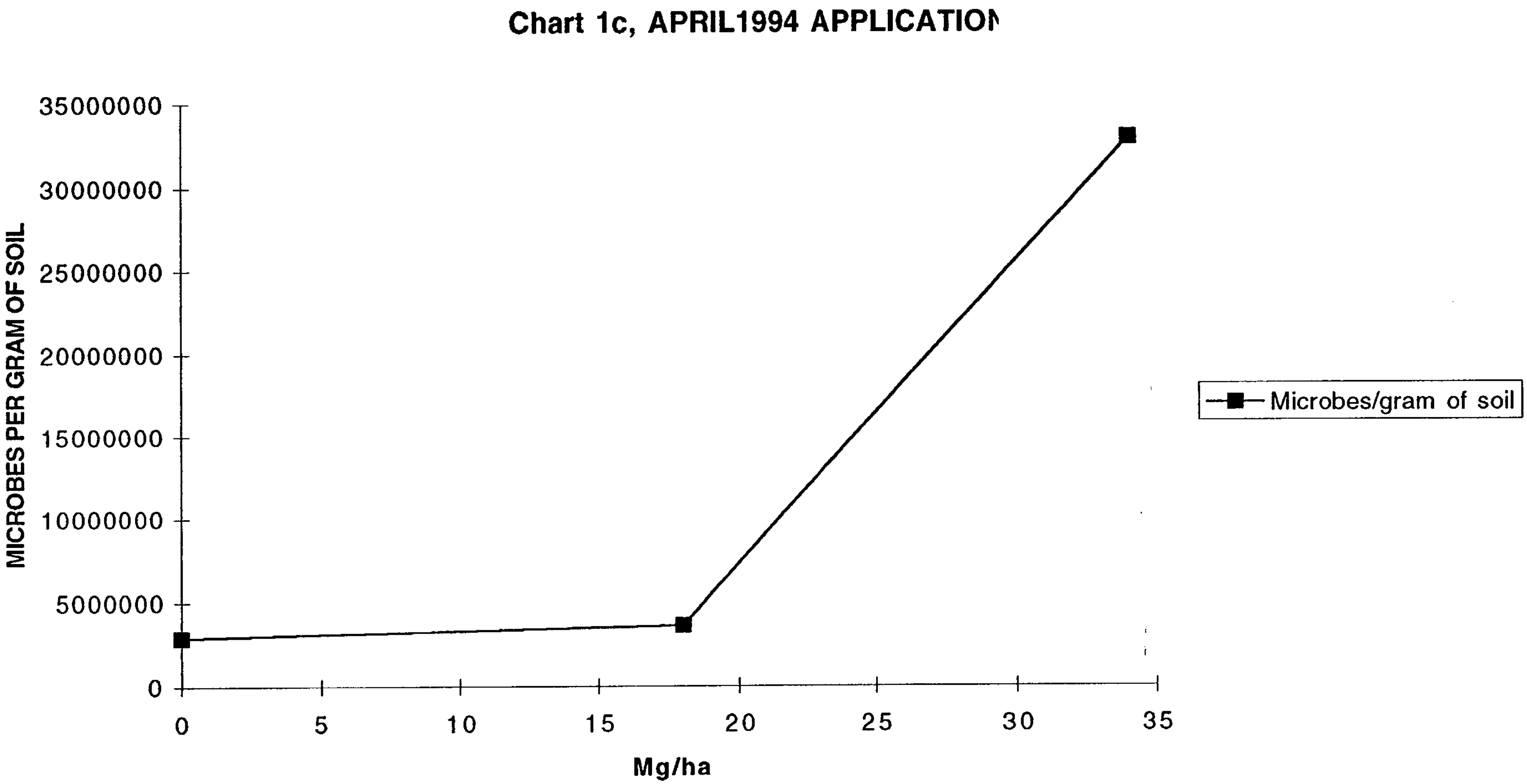
Compared data Chart 1



Compared data Chart 8

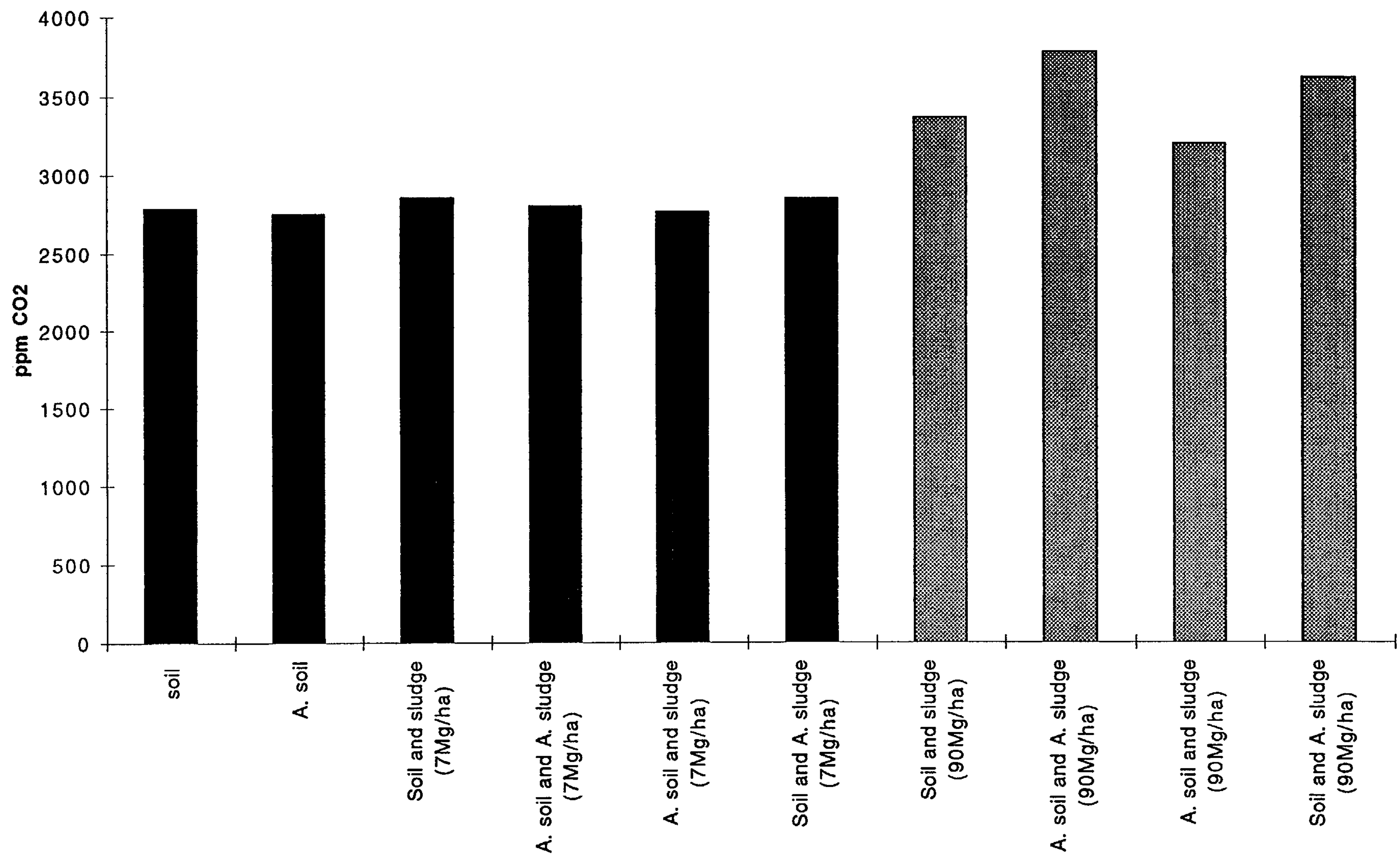


Compared data chart 3



Autoclaving Experiment 2 Chart 5

Chart 2



C-1

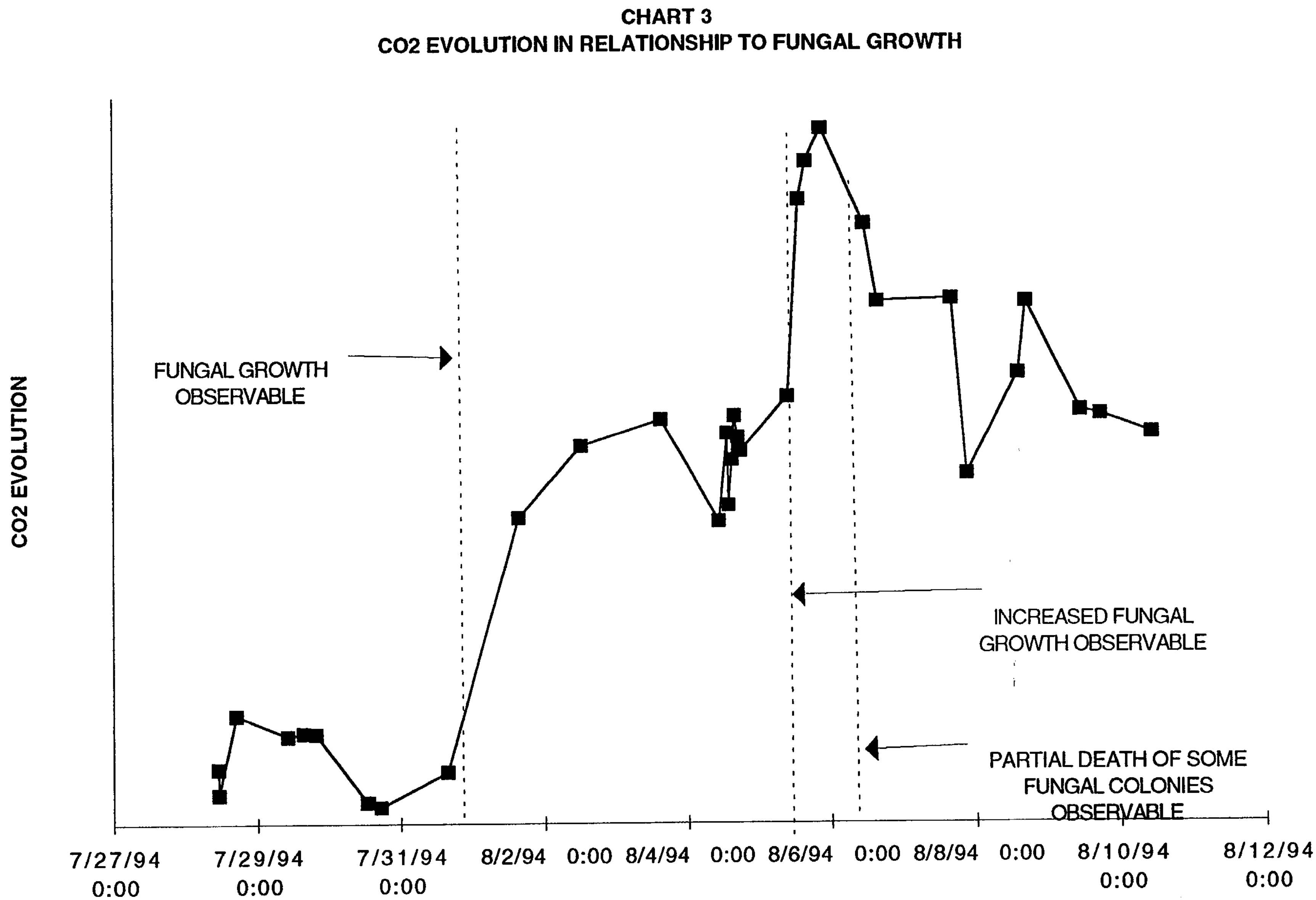
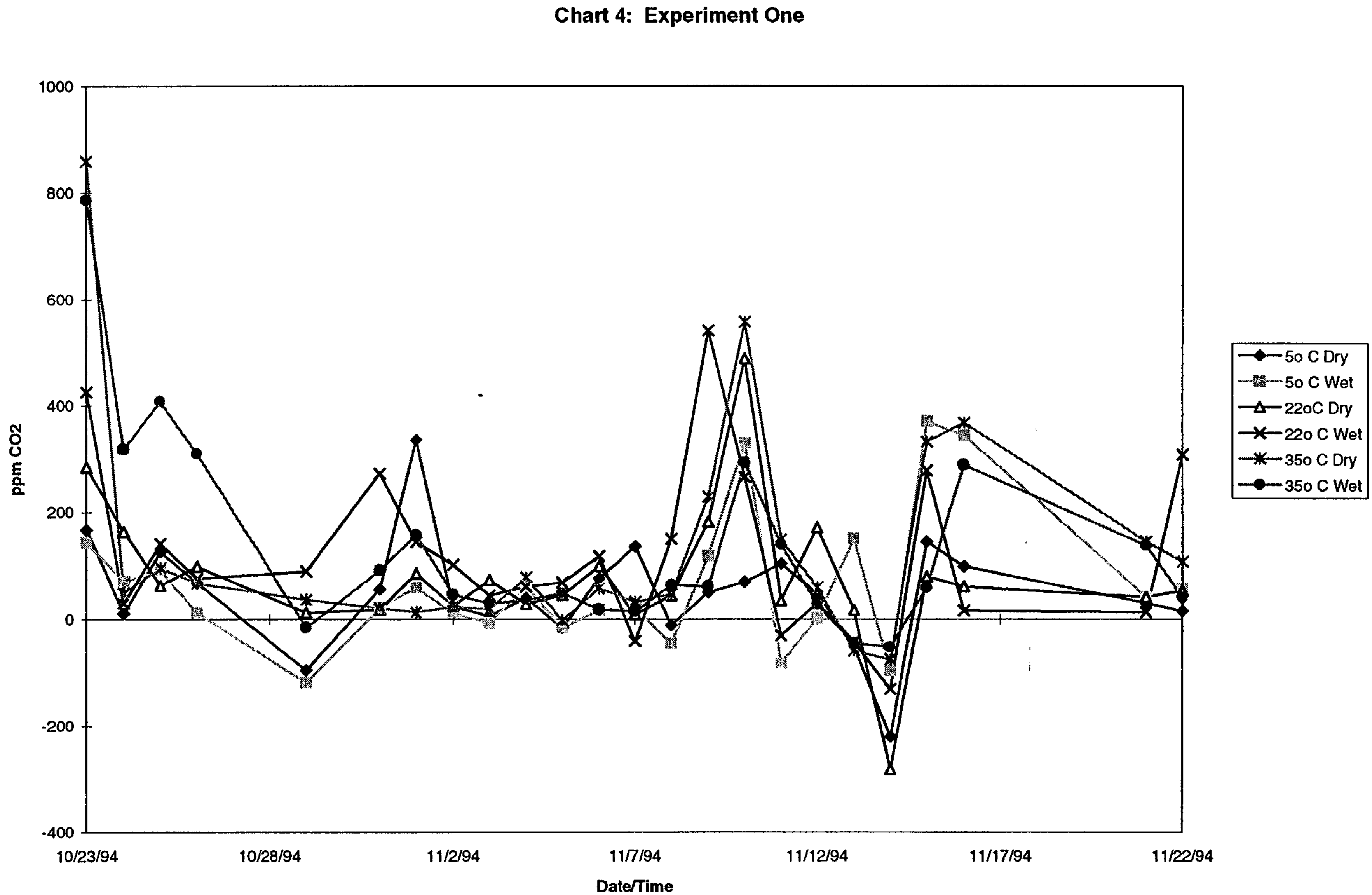
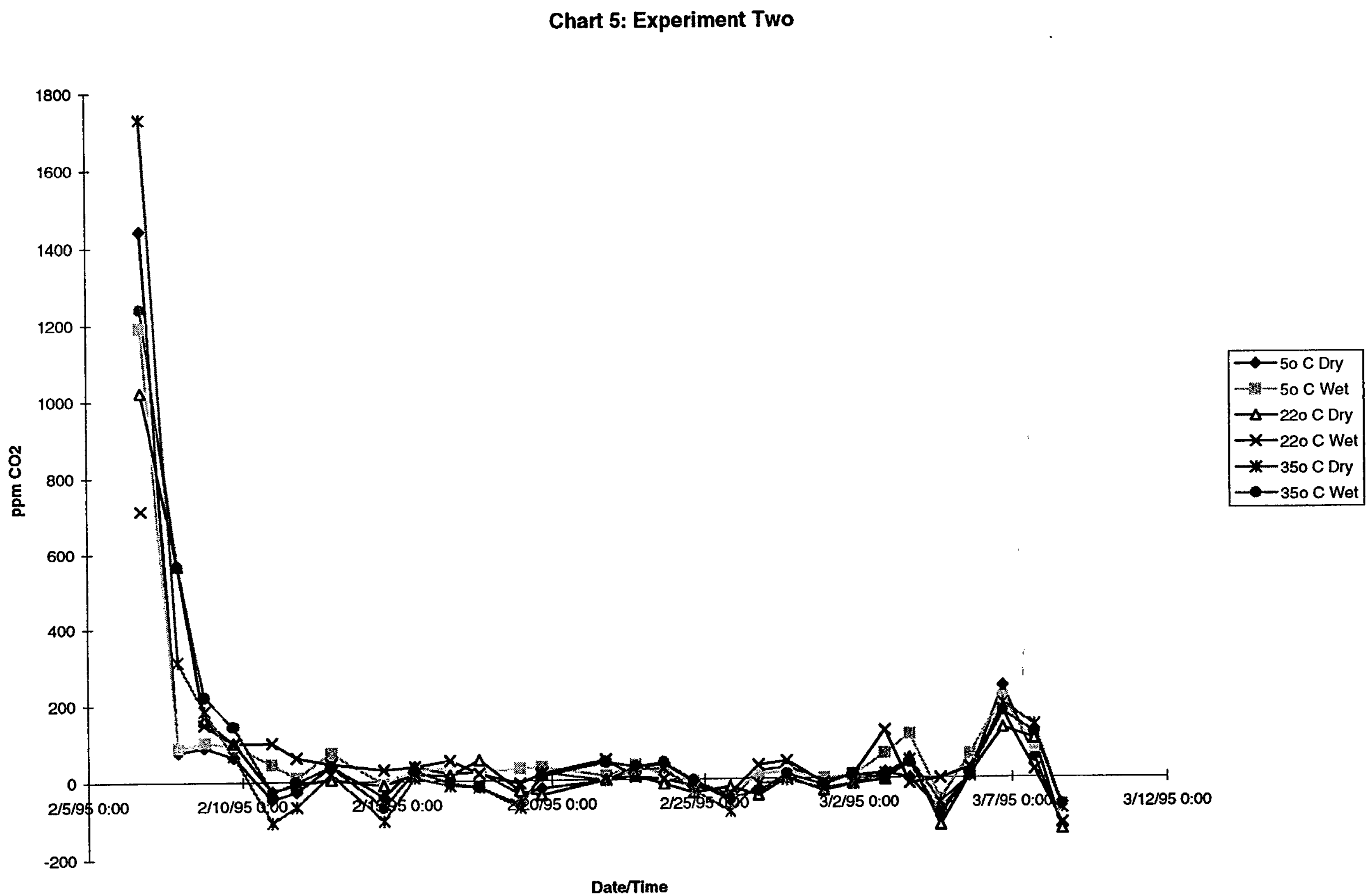


CHART 3
CO2 EVOLUTION IN RELATIONSHIP TO FUNGAL GROWTH
CO2 EVOLUTION
FUNGAL GROWTH OBSERVABLE
INCREASED FUNGAL GROWTH OBSERVABLE
PARTIAL DEATH OF SOME FUNGAL COLONIES OBSERVABLE
7/27/94 0:00
7/29/94 0:00
7/31/94 0:00
8/2/94 0:00
8/4/94 0:00
8/6/94 0:00
8/8/94 0:00
8/10/94 0:00
8/12/94 0:00

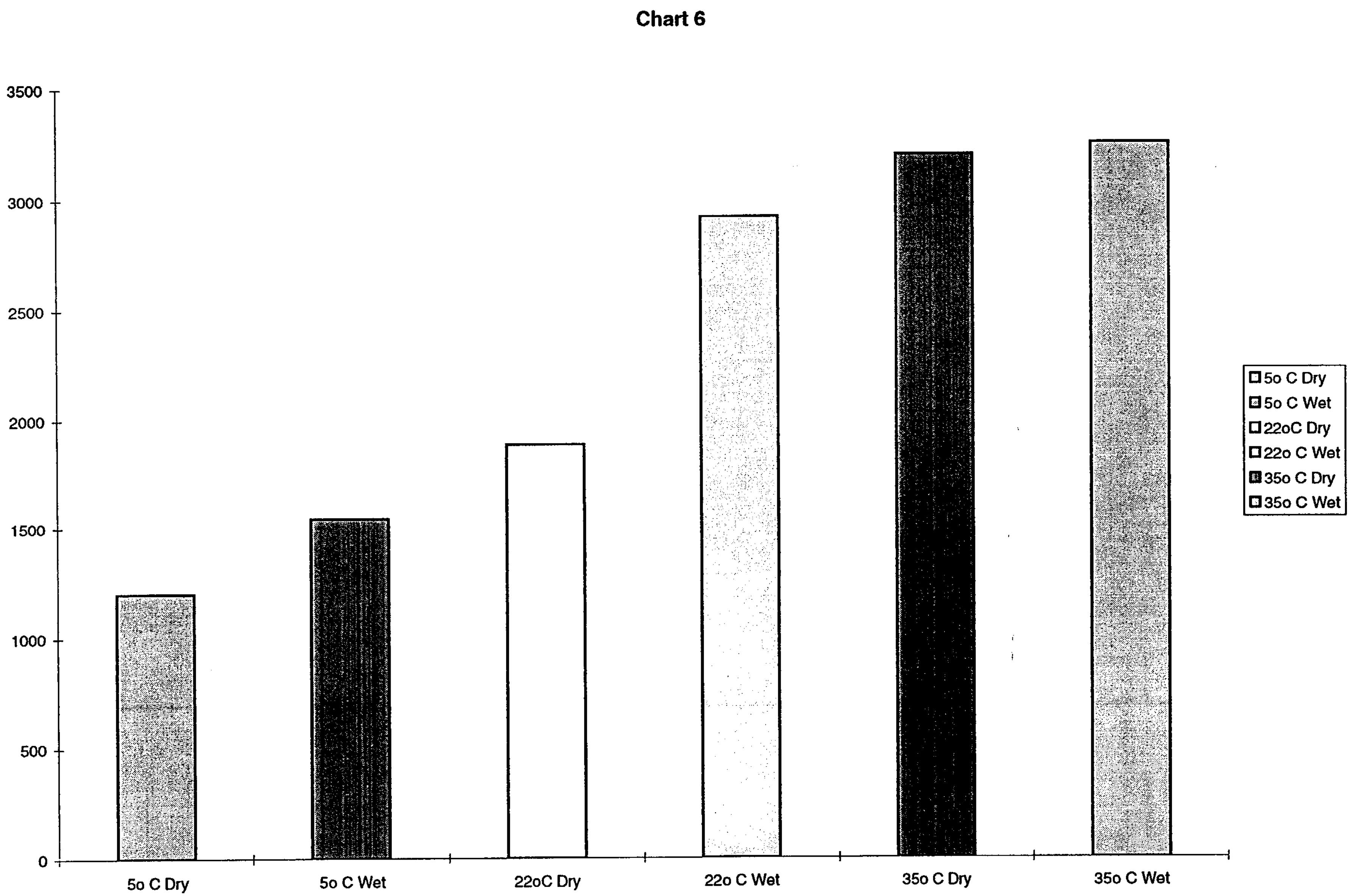
Sheet 1 Chart 15



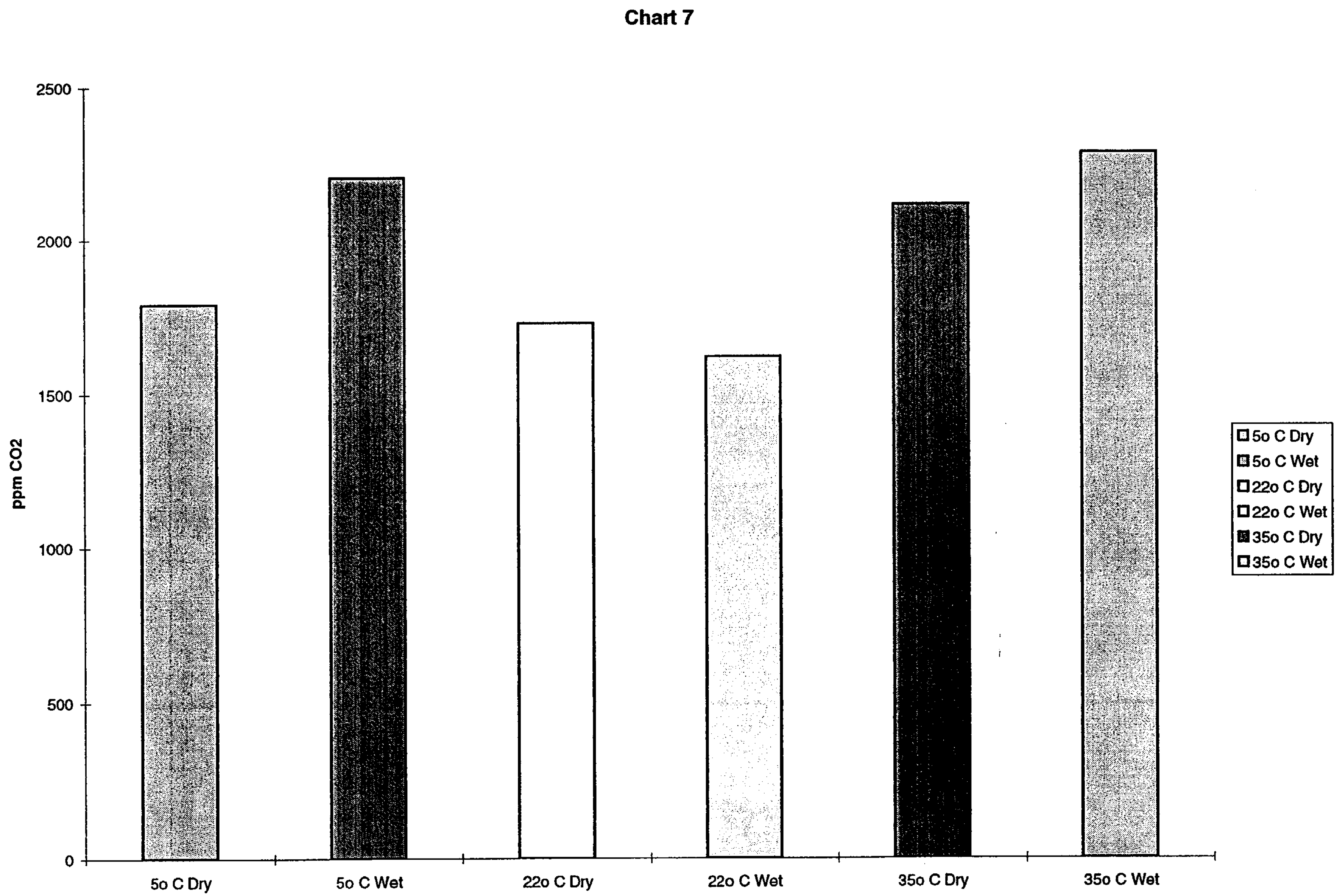
Sheet 1 Chart 17



Sheet 1 Chart 19



Sheet 1 Chart 18



Session IV

Rangeland Restoration with Surface-applied Biosolids: Effects on Soils, Vegetation, and Surface Hydrology at the Sevilleta National Wildlife Refuge, Central New Mexico

Loftin, S.R. and R. Aguilar

Abstract

A biosolids research project was initiated by the Rocky Mountain Forest and Range Experiment Station in December 1990 on the Sevilleta National Wildlife Refuge, central New Mexico. Based on previous study results from the Rio Puerco Watershed, Albuquerque biosolids (17% solids) were surface-applied at a rate of 45 Mg/ha. Vegetation was monitored for changes in aboveground cover, and total nitrogen and heavy metal content (cadmium, copper, lead, and zinc) of aboveground tissues. Soils were monitored for changes in nutrients (organic carbon, nitrogen, and phosphorus) and heavy metals. Runoff plots were established to study the effects of biosolids on surface runoff quantity. Biosolids increased aboveground plant cover (when water was available), nitrogen and copper content of aboveground tissues, and soil nutrient and heavy metal content. Surface-applied biosolids significantly reduced surface runoff from simulated rainfall. The reduction in runoff from biosolids-treated plots was still significant three years after biosolids application.

Keywords: rangeland restoration, biosolids, sewage sludge, vegetation, soil, runoff, heavy metals.

Introduction

Improper land use practices, including over-grazing, can lead to a severe reduction in plant cover and soil productivity. This process, known as desertification, is especially common in arid and semiarid regions with sparse vegetative cover. In the United States over 225 million acres have experienced severe or very severe desertification and overgrazing is the single most important causal factor (Sheridan 1981). Southwestern rangelands have experienced heavy livestock grazing over the past century resulting in a substantial reduction in total plant cover and density (Dortignac and Hickey 1963). A loss of vegetative cover can increase the erosion potential of the soil and soil erosion can accelerate the process of desertification by removing the topsoil rich in plant nutrients and organic matter. If the cycle of degradation is not disrupted, a grassland can degrade to a desert-like system which might be virtually impossible to restore (Naveh 1988; Klein 1989).

Soil organic matter plays a key role in inhibiting the process of desertification (El-Tayeb and Skujins 1989; Parr et al. 1989) . Organic matter improves infiltration of precipitation, soil water holding capacity, and nutrient availability, all of which are important to plant recovery. Any successful attempt at grassland restoration will need to increase retention of precipitation and control runoff and erosion, increase plant growth, and re-establish a stable pool of soil organic matter.

Biosolids are an excellent choice for a soil amendment, it is readily available, contains many plant nutrients, and has excellent soil conditioning capabilities when it is incorporated into the soil (Alloway and Jackson 1991; Glaub and Golueke 1989; Parr et al. 1989). A surface application of biosolids to semiarid grasslands has been shown to increase above-ground plant cover (assuming these systems are water and/or nutrient limited) (Fresquez et al. 1990) and a subsequent decrease in surface runoff and soil erosion (Aguilar and Loftin 1992). Surface applications of biosolids make no direct input to soil organic matter (carbon), however, increases in soil organic matter could occur as an indirect result of increased plant nutrient availability and subsequent belowground productivity. The specific objectives of this research were to: 1) determine the effects of municipal biosolids application on soils and vegetation in a semiarid grassland environment, 2) determine how biosolids application influences runoff and surface water quality, and 3) assess the fate of potential biosolids-borne contaminants in the soils and vegetation.

Methods

Study Site Description

The study was conducted within the Sevilleta National Wildlife Refuge (SNWR). The wildlife refuge is administered by the United States Department of Interior, Fish and Wildlife Service and has been designated as a Long Term Ecological Research Site (LTER) under the administration of the Department of Biology, University of New Mexico. Climate at the SNWR is arid to semiarid with mean annual precipitation ranging from 200 to 250 mm.

Although the SNWR has had a history of heavy livestock grazing, no grazing has occurred on the refuge for approximately 20 years. A desert grassland community was selected for study on moderately sloping and strongly sloping components of a stable alluvial fan. The typical vegetation assemblage at the study site includes the grasses blue grama (*Bouteloua gracilis*), hairy grama (*Bouteloua hirsuta*), black grama (*Bouteloua eriopoda*), ring muhly (*Muhlenbergia torrei*), three awn (*Aristida spp.*), and the shrubs broom snakeweed (*Gutierresia sarothrae*), yellow spiny aster (*Haplopappus spinulosus*), narrow-leaf yucca (*Yucca glauca*), winterfat (*Ceratoides lanata*), four-wing saltbush (*Atriplex canescens*), and groundsel (*Senecio douglasii* var. *longilobus*). Soils at the site are mapped as the Harvey-Dean Association (USDA Soil Conservation Service 1988). The Harvey soil is classified as fine-loamy, mixed, mesic Ustollic Calciorthid and the Dean soil is classified as fine-loamy, carbonatic, mesic Ustollic Calciorthid. The soils are formed in alluvium derived dominantly from sandstone, limestone, and eolian material and are deep and well drained. Permeability ranges from moderate for the Harvey soil to moderately slow for the Dean soil. Runoff is medium and the hazard of water erosion is moderate for both soils.

Experimental Design

Soil and vegetation were sampled within ten (10 X 10 m) plots (five control and five biosolids-treated). A randomized block design was used to increase the interspersion of treatments throughout the study site (Hurlbert 1984). The 10 X 10 m plots were used for experiments and field activities that required disturbance of the soil and ground cover and might otherwise influence the surface hydrology of the runoff plots.

Three pairs of runoff plots (3 X 10 m), each consisting of a treated and a control plot, were established within each of two slope gradient classes (6% and 10-11%). The runoff plots were bordered by 15 cm (6-inch) aluminum flashing to exclude external runoff and contain surface runoff generated from the plot during rainfall events. The water was collected in 1800-liter galvanized steel livestock tanks buried at the base of the plots. Lids (framed fiberglass roofing panels) were placed over the tanks to exclude direct rainfall inputs and to reduce evaporation following rainfall events.

Biosolids Treatment

The biosolids treatment, applied in early April 1991, consisted of a one-time surface application equivalent to 45 Mg/ha (20 tons/acre on an oven-dry weight basis) of municipal biosolids provided by the City of Albuquerque. Levels of Cu, Pb, and Zn were below USEPA limits (Table 1) and the projected metal-loading rate associated with the one-time application at 45 Mg/ha was well below current federal limitations for biosolids application to non-agricultural lands (U.S. Environmental Protection Agency 1992). The biosolids were applied in the spring to minimize losses that might otherwise occur during high intensity rainfall from summer and fall convective thunderstorms, given that spring precipitation at the SNWR is either in the form of low intensity rainfall or snow.

Albuquerque biosolids are anaerobically digested and mechanically de-watered to approximately 17% dry solids (82.5% water). The product is a gelatinous substance that is relatively easy to transport and handle. Biosolids were applied to the treated plots with a front-end loader and then spread with rakes and shovels to obtain a uniform application (the 45 Mg/ha surface application resulted in a uniform "wet sludge" layer approximately 2.5 cm thick). Extra caution in minimizing disturbance to the existing native vegetation was exercised while spreading the biosolids on the plots.

Rainfall Simulation

A rainfall simulator was used in September 1991, 1992, and 1993 to evaluate rangeland response to high-intensity rainfall under controlled conditions. The rainfall simulator was designed to deliver rainfall at approximately the same droplet size and velocity as would a natural storm (Ward 1986). The simulated rainfall experiments insured estimates of runoff yield in the absence of natural storms capable of producing runoff. Each event was approximately equivalent to a high intensity summer thunderstorm common in the region (40-80 mm/hr for 30 minutes).

The rainfall simulator (15 sprinklers mounted on 3-meter tall standpipes) distributed water simultaneously to a runoff plot pair so that infiltration and runoff yield could be observed and recorded while rainfall variability was minimized. At the conclusion of each simulation run the total amount of runoff in the runoff collection tanks was measured and recorded.

Table 1. Organic carbon, plant nutrient, and heavy metal analysis of Albuquerque municipal biosolids, loading rates from the Sevilleta biosolids application project, and U.S. Environmental Protection Agency (USEPA) loading limits.

Albuquerque Biosolids

OC	TKN	P	Cu	Pb	Zn
(%)			(mg/kg)		
20.1	2.62	2.37	635	232	921
USEPA Limits (mg/kg)[1]			4,300	840	7,500

Sevilleta Biosolids Loading Rates

OC	TKN	P	Cu	Pb	Zn
(kg/ha)			(kg/ha)		
9,045	1,175	1,066	28.57	10.44	41.44
USEPA Limits (kg/ha)[2]			1,500	300	2,800

[1]USEPA ceiling concentrations for municipal biosolids (USEPA, 1993).
[2]USEPA cumulative pollutant loading rates for land application of municipal biosolids (USEPA, 1993)

Soil and Vegetation Analyses

Soil samples were collected once before the biosolids treatment (Fall/Winter 1990) and three times following the treatment (Fall 1991, Spring 1992, Fall 1992) in all 10 X 10 m plots. The fall sampling periods followed the late summer to early fall convective thunderstorm period and the spring sampling period followed the winter/spring precipitation period.

Five composite soil samples were collected from each of the three sampling depths, (0-5, 5-10, 10-15 cm) within each 10 X 10 m plot during the scheduled sampling periods. Each composite sample was composed of five randomly selected subsamples. Results from other studies (e.g., Davis et al., 1988) suggested that the depth resolution of this sampling procedure would allow an accurate trace of the vertical movement of specific nutrients and contaminants through the soil profile.

Soil samples were analyzed for pH, electrical conductivity (EC), organic carbon (OC), total nitrogen (TKN), KCl extractable NH_4-N and NO_3-N, $NaHCO_3$ extractable phosphorus, and DTPA (Diethylenetriaminepentaacetic acid) extractable heavy metals (Cd, Cu, Pb, Zn).

Characterization of vegetation (aboveground foliar cover) followed the same general schedule as soil characterization and sampling. Changes in aboveground plant cover were determined using a line-intercept technique developed by SNWR-LTER researchers. Transect intercepts were recorded for individual plant species and other ground cover categories including plant litter, bare soil, gravel, and biosolids.

Blue grama grass (four composite samples per large plot, and two composite samples per runoff plot) and yellow spiny-aster (one composite sample per large and runoff plot) aboveground tissue samples were collected in Fall 1992 and analyzed for total nitrogen and total heavy metal content.

A more detailed description of the methods and procedures is published in Aguilar et al. (1994).

Statistical Analysis

Data sets that included repeated measures (time or time and depth) were analyzed with SPSS (Statistical Package for the Social Sciences) repeated measures multivariate analysis of variance to test for significant interactions among main factors (treatment, time, and depth). Data sets that included both time and depth as repeated measures were analyzed with consecutive repeated measures MANOVA tests (depth then time) using adjusted alpha values (α=0.025) to test for significant interactions among main factors (treatment, time, and depth). Unless otherwise noted, an α (type I error rate) of 0.05 was adopted for all of the statistical analyses.

Results and Discussion

Soil Response

An increase in plant-available nutrients is an important step towards increasing aboveground plant cover and stabilizing soils. Biosolids are an excellent source of nitrogen and phosphorus (Table 1), two plant nutrients that are often limiting in terrestrial ecosystems.

Mineral nitrogen increased significantly in biosolids-treated soils (Table 2). Most of the mineral nitrogen present in anaerobically digested biosolids is in the form of ammonium (NH_4). Therefore, it is not surprising that the peak NH_4-N response occurred in the first sampling period following biosolids application (Fall 1991). Ammonium is relatively immobile in soils and, consequently, the largest response was in the surface (0-5 cm) depth. In aerated soils NH_4 is rapidly converted to NO_3 through the process of nitrification which explains why soil NO_3-N levels peaked after soil NH_4-N. Nitrate is mobile in soil and significant increases in NO_3-N were detected in the 5-10 and 10-15 cm depths. Sodium bicarbonate-extractable phosphorus, an index of plant-available phosphorus, increased significantly following the biosolids application.

Table 2. Plant nutrient and organic carbon content of control and biosolids-treated soils. Means within a depth with the same letter are not significantly different (P>0.05).

Depth	Treatment	PT	F91	S92	F92
		Sampling Period			
Ammonium Nitrogen (mg/kg)					
0-5 cm	Control	3.90df	3.98de	5.07d	3.00f
	Biosolids	3.55ef	207a	136b	44.9c
5-10 cm	Control	3.41d	4.30cd	5.50c	3.70d
	Biosolids	3.61d	26.1a	41.7a	8.56b
10-15 cm	Control	4.04bc	5.01ab	5.01ab	3.66c
	Biosolids	3.64c	21.4ac	41.5ac	6.22a
Nitate Nitrogen (mg/kg)					
0-5 cm	Control	7.62c	9.16cd	3.16d	3.64d
	Biosolids	7.52c	40.5b	82.3a	39.1b
5-10 cm	Control	7.20c	6.68cde	1.53e	3.84d
	Biosolids	7.22c	39.6b	49.9b	83.6a
10-15 cm	Control	9.17c	6.55cde	1.70e	3.23d
	Biosolids	9.28c	33.2b	28.3b	91.5a
Extractable Phosphorus (mg/kg)					
0-5 cm	Control	6.51c	6.54c	6.35c	5.55c
	Biosolids	6.41c	49.0b	64.0a	58.6ab
5-10 cm	Control	2.62c	2.93c	2.81c	3.12c
	Biosolids	3.13c	10.5b	25.7a	18.6a
10-15 cm	Control	1.68cd	2.27cd	2.40bc	2.28c
	Biosolids	1.28d	5.39ab	10.4ab	5.53a
Total Kjeldahl Nitrogen (%)					
0-5 cm	Control	709c	726c	733c	697c
	Biosolids	700c	1267a	1240a	1038b
5-10 cm	Control	950a	852a	899a	883a
	Biosolids	840a	1057a	1111a	1105a
10-15 cm	Control	988a	1035a	1053a	1017a
	Biosolids	1075a	1165a	1207a	1225a
Organic Carbon (%)					
0-5 cm	Control	0.71a	0.71a	0.62a	0.63a
	Biosolids	0.87a	0.80a	0.70a	0.73a
5-10 cm	Control	0.83a	0.81a	0.72a	0.74a
	Biosolids	1.03a	0.85a	0.74a	0.81a
10-15 cm	Control	0.98a	0.87a	0.80a	0.81a
	Biosolids	1.19a	0.89a	0.84a	0.86a

Table 3. Heavy metal and pH analyses of control and biosolids-treated soils. Means within a depth with the same letter are not significantly different (P>0.05).

Depth	Treatment	Sampling Period PT	F91	S92	F92
pH (-log[H^+]) (averaged across depths)					
0-15 cm	Control	7.58a	7.66a	7.52a	7.46ab
	Biosolids	7.56a	7.46ab	7.22bc	7.14c
DTPA Extractable Cu (mg/kg)					
0-5 cm	Control	1.18b	0.61d	0.73c	0.90bc
	Biosolids	1.12b	3.05a	3.71a	3.79a
5-10 cm	Control	1.72a	0.56d	0.65c	0.91b
	Biosolids	1.58a	1.06b	1.52a	1.64a
10-15 cm	Control	2.56a	0.59d	0.66d	1.00c
	Biosolids	2.28b	0.81d	1.01c	1.52c
DTPA Extractable Pb (mg/kg)					
0-5 cm	Control	1.11d	1.75ab	1.45c	1.23d
	Biosolids	1.13d	1.98a	1.67b	1.50c
5-10 cm	Control	0.72e	1.24a	1.02bc	0.91cd
	Biosolids	0.82de	1.54ace	1.11ab	0.97cd
10-15 cm	Control	0.60e	1.18a	0.91bc	0.84cd
	Biosolids	0.70de	1.01ab	0.95bc	0.86c
DTPA Extractable Zn (mg/kg)					
0-5 cm	Control	0.65bd	0.48c	0.63bd	0.53cd
	Biosolids	0.65b	2.02a	1.98a	1.64a
5-10 cm	Control	0.41bc	0.17e	0.24d	0.30cd
	Biosolids	0.41bc	0.36bcd	0.56a	0.52ab
10-15 cm	Control	0.36a	0.14d	0.19bd	0.23bcd
	Biosolids	0.35a	0.23ad	0.32ac	0.35ab

Phosphorus is also relatively immobile in soils and the greatest increases occurred in the 0-5 cm depth.

The majority of soil TKN is organic nitrogen and an increase in organic nitrogen and/or organic carbon represents an increase in the potential biological productivity. Organic nitrogen and carbon can be used by soil microorganisms as a nutrient and energy source which, in turn, increases soil fertility and plant and ecosystem productivity. Total soil nitrogen increased only in the 0-5 cm depth and significant increases in soil OC were not observed at any depth (Table 2). Some of the TKN response in the surface was undoubtedly due to the increase in inorganic nitrogen, however, soil microbial populations respond rapidly to increased nutrient availability and immobilize inorganic nitrogen into organic forms. This microbial response should also produce an increase in soil OC, however, the increase, if it occurred, may not have been significant relative to the amount of residual OC present in the soil.

Soil pH is an important controlling factor for nutrient and heavy metal solubility and availability to plants and microorganisms. Fortunately, most aridland soils are slightly to strongly alkaline, conditions which decrease heavy metal solubility. In addition, Chang et al. (1992) report that biosolids application to soils with a pH $\geq$ 5.5 have not produced heavy metal (Cu, Zn, Ni, Cr) toxicity problems. Consequently, the significant increases in soil acidity observed in the last two sampling periods (Table 3) should not present a problem.

Soil Cd levels were consistently below detectable limits (0.02 mg/kg) for the duration of the project (Table 3). Soil Pb increased significantly only in the 0-5 cm depth, while Cu and Zn both increased significantly at all depths. Although significant increases were observed, levels of DTPA extractable metals were very low (<4.0 mg/kg) and did not result in any observable detrimental effects on vegetation.

Vegetation Response

Surface application of municipal biosolids increased aboveground plant cover but the response was very dependent upon climatic conditions following application (Loftin and Aguilar 1994). Precipitation in central New Mexico occurs primarily in the late summer/fall and winter/spring. The 1991 summer/fall season produced over 180 mm rainfall while the winter/spring and 1992 summer/fall periods produced 200 and 90 mm, respectively. The lack of plant cover response following the 1991 summer/fall period (Table 4) is partly due to the fact that the biosolids application covered the plants and they had to grow out from under the biosolids in order to be measured by the line-intercept method. A significant increase in plant cover on treated plots occurred in Spring 1992 but only on the 3X10 m plots. The following summer/fall period was relatively dry and no significant differences in plant cover were observed on either the 10 X 10 or 3 X 10 m plots.

Without adequate water, no amount of fertilizer or mulch will produce a vegetation response. Surface-applied biosolids might actually inhibit uptake of water under some circumstances. Many aridland plants have surface roots which can uptake water after a very light rainfall. Biosolids, or any other mulching material, would absorb the water from a light shower and the

Table 4. Total aboveground plant cover (%) in 10X10 and 3X10 m plots. Means within a plot size with the same letter are not significantly different (P>0.05).

		Sampling Period			
Plot Size	Treatment	PT	F91	S92	F92
10 X 10 m	Control	38.2b	40.8ab	40.2ab	32.7bc
	Treated	35.5bc	31.0bc	51.0a	24.1c
3 X 10 m	Control	39.2b	46.9b	43.1b	37.4b
	Treated	39.6b	47.8b	60.1a	40.4b

water would evaporate before reaching the soil. Of course when rainfall is sufficient to penetrate the biosolids layer and wet the underlying soil, the biosolids act as a mulch which inhibits evaporation and increases soil water availability. The magnitude, rate, and direction of vegetation response to biosolids application is dependent upon the climatic conditions following application. Thus it is difficult to predict the immediate response of vegetation to biosolids application at any one site. However, visual observations of the experimental plots in subsequent years indicate that aboveground cover has been increasing.

Changes in plant tissue nitrogen provide an index of the nitrogen captured by the system during the relatively brief period of elevated nitrogen availability following the biosolids amendment. Changes in heavy metal content of plant tissues provide an indication of heavy metal uptake and potential transport to higher trophic levels. Blue grama grass and yellow spiny-aster were chosen for tissue analysis because they are the most abundant grass and forb, respectively, on the site. Both species showed significantly greater aboveground tissue TKN and Cu on treated plots, while Zn levels were not significantly different (Table 5). Plant tissue Cd and Pb were below detection limits (1.0 and 10.0 mg/kg, respectively) on both treated and control plots and are not included in Table 5.

Hydrological Response

Results from 1993 rainfall simulations support previously published results from 1991 and 1992 (Aguilar and Loftin 1992; Aguilar et al. 1994). Surface runoff from biosolids-treated plots was significantly less than runoff from controls for all simulation experiments (Table 6). Due to dry soil conditions it was necessary to conduct "wet" and "very wet" simulations in 1992 and 1993 in order to generate runoff from either control or treated plots. Wet and very wet simulations were conducted within 24 and 1 hr, respectively, of the initial simulation. Although not statistically testable, the results from the wet and very wet runs indicate that surface-applied biosolids reduced surface runoff even under conditions of high (near saturation) soil moisture content. During simulations, water on the control plots quickly puddled (standing water) on

Table 5. TKN, Cu, and Zn concentrations in aboveground tissues from blue grama grass and yellow spiny-aster. Means, within a species and analysis, followed by the same letter are not significantly different (P>0.05).

Treatment	TKN (%)	Cu (mg/kg)	Zn (mg/kg)
Blue Grama Grass			
Control	1.18b	1.95b	12.40a
Treated	2.19a	3.75a	9.95a
Yellow Spiny-Aster			
Control	1.75b	8.18b	30.91a
Treated	2.49a	23.63a	41.54a

bare soil and then flowed downslope from one bare spot to the next. Surface-applied biosolids filled bare spaces and initially absorbed precipitation until saturated. Even when treated areas puddled after the biosolids became saturated, the water rarely flowed because of the increased surface resistance. Water was held in place until it infiltrated the soil.

Conclusions

Surface application of biosolids to a degraded semiarid grassland increased plant-available nitrogen (NH4-H and NO3-N) and phosphorus, and increased the total nitrogen content of aboveground plant tissues. Soil and plant tissue heavy metal levels were also increased. Soil heavy metal contents did not reach levels which are likely to be phytotoxic, and no phytotoxic effects were observed. Unfortunately, information is currently not available to evaluate the potential long-term effects of increased Cu contents in aboveground plant tissues.

Total soil nitrogen and total plant nitrogen increased significantly following biosolids application. Increases in total soil and plant nitrogen represent positive responses by the biological components of this semiarid grassland ecosystem. The period of increased biologically-available mineral nitrogen following biosolids application is relatively brief. Nitrogen assimilated into organic biomass by plants and soil microorganisms can be retained within the system and used to support future productivity.

Soil organic carbon is an important factor in controlling nutrient and water retention and availability. Furthermore, soil organic carbon improves soil physical structure and decreases soil erosion potential. Unfortunately, no increase in soil organic carbon was detected after two growing seasons. In an earlier Forest Service biosolids project, no significant differences in soil organic matter were observed eight years after application (Loftin et al. 1995). Plant root production is the major soil organic matter source and there is evidence to support the assumption that plants will not allocate carbon to roots when nutrients are readily available (Chapin et al. 1987; Loftin and Aguilar 1994). We hypothesize that any increase in soil organic carbon, if it does occur, will be a long-term response that will occur only after plants are nutrient limited. It may take at least ten years for nutrient-limiting conditions to return following a biosolids treatment of this type.

The increase in plant nutrient availability resulted in an increase in plant cover, but only when water was available. This is an important point that needs to be stressed. Aridlands are, by definition, water-limited and no restoration program will succeed without adequate water for plant growth. It is generally accepted that a surface mulch of almost any organic material can increase the time and amount of soil water retention and our runoff data showed that surface-applied biosolids did increase the retention of precipitation following relatively heavy rainfall (40-80 mm/hr for 30 minutes). However, a surface layer of biosolids (or any other organic material) will intercept some precipitation and release it to the atmosphere via precipitation before it reaches the soil. This phenomenon could limit the effectiveness of light rainfall events.

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Table 6. Rainfall and runoff relationships during rainfall simulation experiments at the Sevilleta National Wildlife Refuge[1]. Values are means of three plots from the low (L) or high (H) slope class unless otherwise noted as "wet" (W), or "very wet" (VW) which are values from individual rainfall simulations. Runoff/rainfall means, within a date and slope class, followed by the same letter were not significantly different (P>0.05).

Date	Plot	Slope (%)	Rainfall (mm)	Runoff (mm)	Runoff/Rainfall
September 1991					
	Control (L)	6.1	44.7	9.0	0198a
	Treated (L)	6.2	43.9	0.1	0.003b
	Control (H)	10.1	33.8	6.9	0.164a
	Treated (H)	10.3	35.8	0.2	0.0056b
September 1992					
	Control (L)	6.1	38.2	0.4	0.0079a
	Treated (L)	6.2	44.1	0.0	0.000b
	Con (L) **VW**[2]	5.9	48.0	11.6	0.242
	Trt (L) **VW**	6.0	36.0	0.0	0.000
	Control (H)	10.1	33.6	0.1	0.0029a
	Treated (H)	10.3	43.9	0.0	0.000b
	Con (H) **VW**	10.3	46.6	3.7	0.079
	Trt (H) **VW**	10.1	40.2	0.0	0.000
September 1993					
	Control (L)	6.1	45.8	7.9	0.174a
	Treated (L)	6.2	45.6	0.0	0.000b
	Con (L) **VW**	5.9	45.4	17.7	0.390
	Trt (L) **VW**	6.0	31.8	0.0	0.000
	Control (H)	10.1	35.7	6.0	0.154a
	Treated (H)	10.3	39.2	0.1	0.002b
	Con (H) **W**[3]	9.9	35.1	9.3	0.265
	Trt (H) **W**	10.8	45.7	2.4	0.053
	Con (H) **VW**	10.3	49.3	19.1	0.0387
	Trt (H) **VW**	10.1	26.5	0.0	0.000

[1] All plots were 30 m^2 and rainfall simulation experiments lasted 30 minutes.
[2] VW - "very wet" - second simulation conducted within 1 hr following initial simulation.
[3] W - "wet" - second simulation conducted within 24 hr following initial simulation.

Literature Cited

Aguilar, R. and S.R. Loftin. 1992. Sewage sludge application in semiarid grasslands: effects on runoff and surface water quality. pp 211-220. *In*: C.T. Ortega-Klett (ed.) *Proceedings of the 36th Annual New Mexico Water Conference*. New Mexico Water Res. Research Inst. Nov. 7-8, 1991, Las Cruces, NM.

Aguilar, R., S.R. Loftin, and P.R. Fresquez. 1994. Rangeland restoration with treated municipal sewage sludge. *In*: Clapp, C.E., W.E. Larson, and R.H. Dowdy. *Sewage Sludge: Land Utilization and the Environment*. ASA-CSSA-SSSA, Madison, WI.

Alloway, B.J. and A.P. Jackson. 1991. The behavior of heavy metals in sewage sludge-amended soils. Science of the Total Environ. 100:151-176.

Chang, A.C., T.C. Granato, and A.L. Page. 1992. A methodology for establishing phytotoxicity criteria for chromium, copper, nickel, and zinc in agricultural land application of municipal sewage sludges. J. of Environ. Qual. 21(4):521-536.

Chapin, F.S.III., A.J. Bloom, C.B. Field, and R.H. Waring. 1987. Plant responses to multiple environmental factors. Bioscience 37(1):49-57.

Davis, R.D., C.H. Carlton-Smith, J.H. Stark, and J.A. Campbell. 1988. Distribution of metals in grassland soils following surface applications of sewage sludge. Environ. Poll. 49:99-115.

Dortignac, E.J., and W.C. Hickey, Jr. 1963. Surface runoff and erosion as affected by soil ripping. Presented at the Second Federal Interagency Conference on Sedimentation held at Jackson, MS on January 1963; U.S. Dept. of Agric., Forest Service, Washington, DC 13 p.

El-Tayeb, O.M. and J. Skujins. 1989. Introduction: potential of biological processes in desertification control. Arid Soil Res. and Rehab. 3:91-98.

Fresquez, P.R., R.E. Francis and G.L. Dennis. 1990. Soil and vegetation responses to sewage sludge on a degraded semiarid broom snakeweed/blue grama plant community. J. of Range Manage. 43:325-331.

Fresquez, P.R., R. Aguilar, R.E. Francis and E.F. Aldon. 1991. Heavy metal uptake by blue grama growing in a degraded semiarid soil amended with sewage sludge. J. of Water, Air and Soil Poll. 57-58:903-912.

Glaub, J.C. and C.G. Golueke. 1989. Municipal organic wastes and composts for arid areas. Arid Soil Res. and Rehab. 3:171-184.

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecol. Mono. 54(2):187-211.

Klein, D.A. 1989. Cellulose functions in arid soil development. Arid Soil Res. and Rehab. 3:185-198.

Loftin, S.R. and R. Aguilar. 1994. Semiarid rangeland response to municipal sewage sludge: plant growth and litter decomposition. pp 221-229. *In*: Clapp, C.E., W.E. Larson, and R.H. Dowdy (eds.). *Sewage Sludge: Land Utilization and the Environment.* ASA-CSSA-SSSA, Madison, WI.

Loftin, S.R., R. Aguilar, and P.R. Fresquez. 1995. Rangeland restoration with surface-applied biosolids: effects on soils and vegetation of the Rio Puerco Watershed, Northcentral NM. (this issue).

Naveh, Z. 1988. Multifactorial reconstruction of semiarid Mediterranean landscapes for multipurpose land uses. *In:* Allen, E.A. (ed.). *The reconstruction of disturbed arid lands: an ecological approach.* AAAS Selected Symposium 109.

Parr, J.F., R.I. Papendick, S.B. Hornick, D. Colacicco. 1989. Use of organic amendments for increasing the productivity of arid lands. Arid Soil Res. and Rehab. 3:149-170.

Sheridan, D. 1981. Desertification of the United States. Council on Environmental Quality. U.S. Government Publishing Office. Washington D.C.

U.S. Department of Agriculture, Soil Conservation Service. 1988. *Soil Survey of Socorro County Area, New Mexico.* U.S. Government Printing Press, Washington D.C.

U.S. Environmental Protection Agency. 1993. 40 CFR - Parts 257 and 503, Standards for the disposal of sewage sludge; final rule.

Ward, T.J. 1986. A study of runoff and erosion processes using large and small area rainfall simulators. New Mexico Water Resources Research Institute (WRRI) Report No. 215. New Mexico State University, Las Cruces, New Mexico.

Utilization of Municipal, Industrial and Animal Wastes on Semiarid Rangelands

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In recent years, cities and industries have begun to consider rangelands as potential sites for disposal of municipal, industrial and animal wastes. Numerous studies have demonstrated that municipal sewage sludge and feedlot animal wastes can be successfully used as fertilizer and mulch on croplands, but few studies have applied these wastes to rangelands. Therefore little information is available concerning the effects of waste applications to rangelands where incorporation of the waste material into the soil is not feasible, and where the vegetation is comprised of perennial grasses rather than annual crops. We initiated a study in April, 1993, to determine the effects of industrial, municipal and animal waste applications on soil, plant and water quality properties of semiarid rangelands. The study is being conducted at the Central Plains Experimental Range (CPER) near Nunn, CO, on short grass range, and at the High Plains Grasslands Research Station (HPGRS) near Cheyenne, WY, on mixed grass range. At both sites a series of thirty 9-by-9-m plots for vegetation/soil evaluations and twenty 3-by-9-m plots for hydrologic investigations were established. Treatments consisted of surface applications (23 Mg/ha) of (1) fresh feedlot cattle waste, (2) composted feedlot cattle waste, (3) phosphogypsum (a by-product of the phosphorus fertilizer industry and 92% $CaSO_4$) (4) composted sewage sludge, and (5) control (no treatment). Soil samples were collected in 1993 prior to waste application for evaluating baseline soil properties, and in 1994 to assess the effects of waste application on soil properties. Vegetation samples for evaluating forage production/quality and species composition were collected at peak production in 1993 and 1994. Runoff water quality and quantity were evaluated with a rotating boom rainfall simulator in May and August, 1993.

The composted sewage sludge and composted manure contained high levels of N and P, which resulted in increased aboveground plant production and increased concentrations of N and P in plant tissue in the 1993 growing season. In the spring of 1994, elevated levels of NO_3-N and bicarbonate-extracted P were found in the soil profiles of these two waste treatments, but due to severe drought throughout the 1994 growing season, total plant production did not respond to the increased nutrient levels. During simulated rainfall events, elevated levels of N and P were found in the runoff water from the composted manure and composted sewage sludge treatments. The fresh manure and phosphogypsum amendments did not improve total plant production, although plant N concentrations were increased in some plant species by the fresh manure amendment, and both the fresh manure and phosphogypsum amendments increased the P concentration of warm season grasses. Elevated levels of NO_3-N and bicarbonate-extracted P were not found in the soil profiles except at the soil surface. Runoff during simulated rainfall events on the fresh manure and phosphogypsum amendments were not elevated in N, but were elevated in P.

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Evaluating Infiltration-Related Water Quality Impacts Associated with Land Application of Biosolids in Arid and Semi-arid Settings

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Abstract

Land application of municipal wastewater biosolids has many benefits in arid and semi-arid settings, including increases in plant productivity, decreases in sediment runoff and increases in infiltration rates. Increased infiltration of biosolids-derived leachate could have adverse water quality effects on the receiving waters. These may include ground water degradation due to deep percolation of leachate and surface water contamination associated with return flows of the leachate to wetlands, streams and lakes. The heavy reliance on water supplies in arid and semi-arid settings warrants an investigation of the quantity and quality of liquid infiltrating beneath land application sites. Results from land application sites in Colorado and New Mexico where infiltration monitoring has been conducted are presented. Monitoring methods used to evaluate infiltration-related water quality impacts associated with land application of biosolids in semi-arid settings are discussed. Infiltration studies from these and other sites in arid and semi-arid environments indicate that infiltration from normal rainfall events is usually restricted to the upper two meters, although extreme precipitation events can cause much deeper infiltration.

Introduction

Land application is a viable option for disposing of municipal wastewater biosolids. This disposal method involves the surface spreading of anaerobically digested and partially dewatered biosolids on designated lands. Land application has been used at many sites in the arid and semi-arid setting in the western U.S., both as a soil amendment to increase plant productivity and to decrease sediment yield in surface streams (Gallier et al, 1993; Aguilar and Loftin, 1992; Mostaghimi et al, 1992; Aguilar and Aldon, 1991a; Lerch et al, 1990; Sabey et al, 1990; King et al, 1988). Land application of biosolids is likely to increase in the future to take advantage of these and other benefits.

Biosolids application is not without its risks to the surrounding environment, however. If undertaken incorrectly, biosolids application can lead to the accumulation of heavy metals in soil (Chang and Page, 1993; Liang and Corey, 1993; Fresquez et al, 1990; Welch and Lund, 1989; Davis et al, 1988; Williams et al, 1987; Tackett et al, 1986; Sommers and Nelson, 1981) and to degradation of surface water (Coupe and Macy, 1993) and ground water quality (Berg et al, 1987; Goda et al, 1986; O'Brien and Mitsch, 1980; Baxter et al, 1977; Trout et al, 1975). Water is almost always in short supply in arid and semi-arid regions. Surface streams often are dry except during the period of snowmelt runoff in the spring or during brief periods after summer thunderstorms, so the often more reliable ground water supplies are an even more precious commodity in these regions. Due to the scarcity of water in arid and semi-arid settings it is essential that land application be undertaken in a manner that minimizes adverse water quality impacts.

A number of studies have shown that land application of biosolids decreases surface runoff. At the Sevilleta National Wildlife Refuge (NWR) in central New Mexico, Aguilar et al (1994) and Aguilar and Loftin (1992) recorded essentially no runoff leaving biosolids test plots in response to natural and simulated rainfall events over a three year period; runoff from adjacent control plots averaged about 20 percent of rainfall. At the Meadow Springs Ranch site in northern Colorado, runoff from control plots was 3.4 to 37 times that of biosolids-treated plots (RBD and CDM, 1993). Kelling et al (1977) noted a reduction in sediment load and runoff from biosolids-applied test plots, which they attributed to increased infiltration. Even in more humid settings, Mostaghimi et al (1992) found in their simulated rainfall experiments on test plots in Virginia that infiltration from biosolids-applied plots was almost twice that of the control plots.

These studies indicate that surface runoff decreases and infiltration increases in areas where biosolids have been applied. With increasing infiltration rates there is also a greater potential for water quality to be adversely affected by leachate generated from the biosolids. An objective of the current study therefore is to evaluate infiltration and infiltration-related water quality impacts associated with biosolids application in arid and semi-arid environments. Because land application sites in these environments are often in relatively remote locations where power sources are unavailable and access is difficult, an additional objective of this study is to compare the effectiveness of different water quality monitoring methods used in arid to semi-arid settings.

Processes Leading to Water Quality Degradation

In order for the threat of infiltration-related water quality impacts near land application sites to be minimized there must be an understanding of the biosolids contaminant source, contaminant migration pathways and contaminant transport processes from a hydrologic perspective. Understanding these aspects with regard to the characteristics of a given field site can lead to environmentally safer and more effective land application operations at that site.

Contaminant Sources

Contaminant sources associated with biosolids application can include infiltration of compounds directly from the biosolids liquid fraction, infiltration of biosolids-derived leachate, and mobilization of compounds in soil due to reduced soil pH. Direct application of liquid biosolids will lead to infiltration of the soluble components, and may cause deep enough migration within the soil profile to cause ground water contamination (Berg et al, 1987; Baxter et al, 1977; Trout et al, 1975). Dewatering of biosolids to approximately 15-20 percent solids can virtually eliminate this problem (Aguilar et al, 1994; RBD and CDM, 1993). Biosolids application has been shown to decrease the pH of shallow soils (Fresquez et al, 1991; Fresquez et al, 1990; Williams et al, 1987; Goda et al, 1986). Since heavy metals become more mobile at lower pH conditions (Tackett et al, 1986), it is possible that metals either naturally present in soils, or enriched from biosolids application (such as Cd, Cu, Fe, Pb, Ni and Zn) could become mobilized and migrate into deeper portions of the soils profile (Sommers and Nelson, 1981). In many soils in the western U.S., however, soils are generally alkaline so the ultimate depth of migration of metals is likely to be limited to less than 10 cm (Welch and Lund, 1989; Davis et al, 1988; Williams et al, 1987). For partially dewatered biosolids, infiltration of biosolids-derived leachate is thus the most likely source of contaminants that can migrate to and potentially cause water quality degradation. Leachate would form as precipitation interacts with the biosolids, forming a solute which contains soluble compounds from the biosolids. Whether it is surface water or ground water that is ultimately affected, and the extent of the impacts, depend in part on the migration pathways and transport processes.

Migration Pathways

Contaminant migration pathways are the routes that soluble compounds may take after infiltrating below where biosolids have been applied. Leachate will tend to infiltrate vertically downward below the surface under the force of gravity unless impeded by a low-permeability barrier or until it reaches the water table. In the upper soil horizon, infiltration may be via macropore flow or via flow through the porous soil medium. Macropore flow occurs when water moves through distinct soil regions (macropores) much more rapidly than through the surrounding soil matrix (Seyfried, 1991); this can include flow through shrinkage cracks in the soil, along plant roots, insect burrow holes or the like. Macropore flow allows dissolved compounds to migrate into the soil relatively unaffected by the attenuating processes that act to decrease compound concentrations. Macropore flow was observed to become unimportant below about 30 cm below land surface at a test site in southern Idaho (Seyfried, 1991). Flow through the porous soil matrix will therefore be the dominant process in deeper soil zones even where macropore flow exists.

One migration pathway is for leachate to infiltrate vertically downward until it encounters a low-permeability barrier, such as a clay, within the unsaturated zone soil profile. The soil moisture content will increase above this low permeability barrier as infiltration continues. If sufficient liquid exists, the leachate will eventually migrate laterally and down-dip along the top of the barrier. Depending on the local topography and location of the land application area, this perched water could migrate to and discharge at the land surface as a seep or directly to a wetland or surface water body, degrading surface water quality in these bodies. A second migration pathway exists when low permeability soil layers are not present, or are not laterally continuous. In these cases, and if sufficient leachate infiltrates, then a direct migration pathway to ground water exists. Ground water quality will then be affected by the compounds present in the infiltrating leachate. Local ground water-surface water flow dynamics may result in a third migration pathway, this time to surface water. Ground water naturally flowing to and discharging at nearby streams, lakes or wetlands as return flow could also adversely affect the receiving surface water bodies if the return flow water has become contaminated by biosolids-derived leachate. The depth to ground water, presence of low permeability perching layers with the unsaturated zone and infiltration of sufficient volumes of leachate are factors that determine which, if any, of these migration pathways will occur.

Transport Processes

Each of the migration pathways discussed above requires that infiltration be greater than the transport processes that inhibit fluid migration and decrease leachate concentrations. In arid and semi-arid environments the attenuating processes for both fluid and mass transport are significant. Infiltration of fluid below the plant evaporative root zone depth will occur only when the magnitude and duration of precipitation exceeds the effects of evapotranspiration, runoff, soil moisture retention and moisture absorption by the applied biosolids. The effects of these processes on decreasing infiltration in arid environments are discussed below.

Several investigators have quantified evapotranspiration rates in arid and semi-arid regions. A characteristic of rangelands (which are predominant in arid and semi-arid zones) are the high evapotranspiration and low water yields (Branson et al, 1972). In south Texas rangelands, Weltz and Blackburn (1993) calculated evapotranspiration rates that ranged from 72 percent of precipitation from bare sandy loam soil to 99.7 percent from grass and shrub locations. Carlson et al (1990) found that evapotranspiration accounted for over 95 percent of the water balance in grass and mesquite shrub sites in the rolling plains rangeland of Texas. Duell (1990) studied the evapotranspiration requirements of a series of native grasses in the Owens Valley of southeast California. He calculated evapotranspiration rates ranging from approximately 4 to 12 times the annual precipitation for alkali and saltgrass/rush meadow sites. Parton et al (1981) and Lauenroth and Sims (1976) found that the water demands by plants at a shortgrass prairie site in northeast Colorado were sufficiently in excess of available water that evapotranspiration consumed all precipitation during the growing season . Although it is often thought that infiltration below the plant root zone can be considered negligible during the growing season because of the high evapotranspiration rates for most arid grassland and rangeland environments (McCully and Haas, 1970), Stephens et al (1986) found during a 19 month study that deep percolation was roughly 20 percent of mean annual natural precipitation at one site near Socorro in central New Mexico.

The potential for infiltration can be reduced further due to water absorption by the applied biosolids and due to surface runoff of precipitation. Aguilar & Loftin (1992) determined that partially dewatered biosolids (17 percent solids) absorbed an estimated 83-105 percent of the precipitation that fell during a simulated rainfall experiment in central New Mexico. Epstein (1975) found that biosolids retained approximately 10 times more water than the soil control in one study. Runoff will decrease the amount of water that can potentially infiltrate. Several studies have shown that biosolids application greatly decreases runoff (Aguilar et al, 1994; RBD and CDM, 1993; Mostaghimi et al, 1992). Decreases in runoff due to land application is thought to be enhanced by increases in surface roughness associated with the applied biosolids (Aguilar et al, 1994). Surface slope can also directly affect runoff, although the amount of clay in the soil may have an equally large effect on runoff (RBD and CDM, 1993).

Migration of biosolids-derived leachate below the evaporative root zone depth is strongly influenced by the antecedent (existing) soil moisture content (AMC) and the soil moisture retention capacity. Deeper migration occurs more readily when the soil is more moist because of the associated increase in the unsaturated hydraulic conductivity at these conditions, which allows for greater fluid migration. Nash et al (1991) found that wetting of dry arid rangeland soil is a slow process and is largely a function of the soil AMC. Tromble et al (1974) found that infiltration rates were higher in areas covered with litter than in other areas due to higher AMC values beneath litter. Loaque (1992) felt soil AMC was the most critical parameter in optimizing a rainfall-runoff model of a rangeland catchment in Oklahoma. At biosolids land application sites Aguilar et al (1994) and RBD and CDM (1994) observed higher AMC values in soil beneath treated plots compared to adjacent control plots. Bruggeman and Mostaghimi (1993) found that peak loading of nutrient compounds appeared during a rainstorm, with wet initial soil conditions. The moisture retention capacity of a soil indicates how much moisture it can hold under negative pressure against the force of gravity. The field capacity of a soil generally represents its maximum soil moisture retention capacity. In general, finer grained soils have higher retention capacities. A larger moisture retention capacity means that a soil will contain more moisture at a given soil suction, thus will have a higher hydraulic conductivity which allows greater infiltration rates. Several investigators, as summarized in Khaleel et al (1981), have reported increased moisture retention capacities, at both field capacity and wilting point moisture contents, with increasing biosolids application rates. The increases were attributed to increased soil organic carbon. Thus biosolids application has been shown to increase soil antecedent moisture contents and moisture retention capacity, and thereby increase the potential for infiltration of fluid beneath land application areas.

Several studies have been undertaken to evaluate the depth to which infiltration occurs in arid and semi-arid environments. Aguilar and Aldon (1991b) set up test plots on native grassland soils in the Rio Puerco Watershed Management area in northern New Mexico to measure natural infiltration and runoff over a three year period. Using neutron probes, gravimetric analyses of soil samples and onsite raingages, they found that soil moisture fluctuated greatly in the upper 30 cm in response to precipitation events. More gradual seasonal changes in soil moisture were observed in the 50 to 100 cm depth, with increases occurring through snowmelt-based recharge in the spring or correlated to successive thunderstorms in the summer. Little change in soil moisture was observed below 100 cm depth and stayed near the estimated field capacity of the soil (0.03 MPa; 1/3 bar). Analyses of soil chemistry showed accumulations of secondary calcium carbonate at the 48-84 cm depth and gypsum deposits at the 160 to 170 cm depth. The accumulation of these soluble compounds within the soil profile was attributed to the prevailing depth of infiltration under normal precipitation events and to the maximum depth reached by water following extreme storms, respectively (Aguilar and Aldon, 1991b). Aguilar et al (1994) noted a caliche layer at a depth of 70 cm below the surface, which they felt represented the long-term average depth of downward water movement in the soil. Otoma and Kuboi (1988) simulated chloride movement beneath soil treated with biosolids and planted with ryegrass. They found that chloride, also a soluble compound, would generally accumulate in the upper meter of soil. Seyfried (1991) conducted simulated rainfall experiments on test plots of native soils in southern Idaho. He found that moisture penetrated below 42 cm during an initial 'low intensity' rainfall event (10.4 cm/hr) but that subsequent tests allowed a bromide tracer to migrate to only about 21 cm. Seyfried concluded that montmorillonite clays that were present in the test plots swelled during the initial test and inhibited moisture migration in subsequent tests.

Natural recharge was also studied at a native grass test site on the high plains near Golden, Colorado. Kiusalaas and Poeter (1992) found that moisture profiles were static below 90 cm in the late summer through winter, but that heavy precipitation in March 1992 allowed moisture to migrate to slightly over 2 m. A 10 year study conducted by Nixon et al (1972) in sandy soils and native vegetation in coastal California showed that infiltrating recharge would penetrate below the root zone and contribute to deeper recharge only once every seven years, on average, although infiltration penetrated to more than 580 cm following heavy rainfall one year. Wengel and Griffin (1979) found that leaching and accumulation of nitrate to depths of 244 cm occurred at high biosolids application rates on agricultural land in the east coast. These studies show that the transport processes discussed above limit fluid migration in arid and semi-arid environments to the upper meter of soil under normal precipitation inputs but that deeper infiltration is possible under extreme precipitation events.

A final factor affecting the potential for water quality degradation is the effectiveness of reactions that degrade or attenuate the biosolids-related leachate constituents. These include geochemical oxidation-reduction reactions, that will tend to cause metals compounds to precipitate in the soil (Liang and Corey, 1993), and aerobic/anaerobic biodegradation reactions, which will consume some of the nutrient compounds (O'Brien and Mitsch, 1980). Several investigators have found that the depth of metals migration below land application sites in semi-arid regions is limited due to these reactions (Aguilar et al 1994; RBD and CDM, 1994; Welch and Lund, 1989; Williams et al, 1987). Chang and Page (1990) found that the majority of biosolids constituents were assimilated into soils through biogeochemical reactions. The amount of compound attenuation that will occur will increase with a greater depth to water below land surface and slower infiltration rate, since these will allow a greater time period for the attenuating processes to operate.

Water Quality Monitoring in Arid Settings

The preceding discussion suggests that during extreme precipitation events or during the dormant season for plants, both the unsaturated and saturated zones in arid and semi-arid environments are potentially at risk from water quality degradation due to infiltration of biosolids-derived leachate. Monitoring of both the unsaturated and saturated zones may be appropriate based on the biosolids application rates, biosolids water content, natural precipitation, permeability of site soils, and depth to water at a given land application site. Monitoring would serve the purpose of an early warning of contamination, so that biosolids application practices could be modified without posing further risk to the nearby water resources. The following is a brief discussion of existing monitoring methods and devices for both zones.

Unsaturated Zone

Unsaturated zone monitoring devices exist for both water quantity and water quality. Commonly used moisture monitoring devices include soil moisture blocks, tensiometers and neutron probes. Moisture blocks consist of small diameter (5 cm) gypsum blocks with wire leads extending from the blocks that are installed into previously drilled holes. The relative moisture content in soil is indicated by increases in electrical conductance read on a meter. Moisture blocks can record soil conditions drier than the wilting point, but must be calibrated to site soils based on moisture characteristic curves developed from laboratory soils tests. Temperature probes co-located with the moisture blocks can help calibrate the readings to actual water contents. Moisture blocks are inexpensive but are less accurate than other devices and have a finite site life as the gypsum block slowly decays.

Tensiometers consist of an airtight water-filled tube with a porous ceramic cup attached to the bottom, and a type of vacuum gage. The soil monitoring depth corresponds to the installation depth of the ceramic cups. Tensiometers indicate soil moisture content by directly reading soil suction, which is the negative pressure exerted on residual moisture by soil capillary tension. Tensiometers record soil tension in surrounding soils by losing water out of the ceramic cup until an equilibrium vacuum pressure is reached inside and outside the tensiometer tube. Depending on the depth of installation and type of tensiometer, soil suction readings only down to about -70 centibars can be attained. Soils must also be tested in the lab to determine the relationship between soil suction and soil moisture content. Tensiometers are moderate in cost and provide accurate readings of soil tension. If installed in very dry soils, they require frequent monitoring and maintenance to stay operational.

Neutron probes record soil moisture by recording responses to radioactive source emitted from the probe. Emitted neutrons are slowed (thermalized) through interactions with hydrogen atoms in soil, so greater soil moisture content can be inversely correlated to neutron counts. The small diameter (5-7 cm) neutron probes are lowered into access tubes previously installed into the soil and measure soil moisture at any depths within the access tube. Like the other devices, the relationship between soil moisture and neutron readings must be determined for each soil type present at a site. Neutron probes are very accurate but are relatively costly. They offer an advantage of being able to monitor soil moisture over the range of depths of the access tube, allowing a more continuous soil moisture profile to be determined. Storage, use and handling of the radioactive neutron emitter may pose unacceptable logistical difficulties.

Monitoring water quality in the unsaturated zone can be accomplished with lysimeters and by direct soil sampling. Both methods are used to obtain samples, which are subsequently analyzed for the compounds of interest. Lysimeters consist of hollow tubes with ceramic porous cups at their base and are installed in predrilled holes. If sufficient soil moisture exists, water can be induced through the ceramic cups and into the lysimeter tubes by creating a negative pressure inside the tube with an external pump. Water contents must be near or above the field capacity of the soil in order for soil samples to be obtained readily. Water collected in the lysimeter tubes can then be withdrawn and submitted for laboratory analyses. Sampling soils directly can also provide information on water quality in the unsaturated zone. To evaluate infiltration-related impacts, soil samples must be collected and analyzed prior to land applying biosolids and then a comparison is made with later soil samples. Natural variation in soil composition may render the comparison of results inconclusive for metals and common ions, but should be effective for evaluating the depth and magnitude of nitrogen compounds.

Saturated Zone

Monitoring the saturated zone for infiltration-related impacts involves the installation and sampling of ground water monitoring wells. These can be small diameter (5 cm) wells constructed with PVC casing and slotted screen. Drilling methods such as hollow stem augering are preferred, since they do not introduce fluids into the borehole which could affect the chemistry of the water samples which are collected later. Hollow stem auger drilling also allows undisturbed soil samples to be collected, which is useful for characterizing soil chemistry is deeper horizons. Recharge of infiltration from land application sites can be evaluated by measuring water levels in monitoring wells. A history of previous water level measurements are needed in order to identify and factor out daily and seasonal trends in water levels. Water quality evaluations are undertaken by sampling the monitoring wells for the parameters of concern. At a minimum, the more mobile constituents in biosolids such as nitrate and other nutrient compounds should be tested for in the water samples (Berg et al, 1987; Sommers and Nelson, 1981). Several wells should be installed at the land application site to compare upgradient from downgradient water quality. In addition, at least one set of ground water samples should be collected prior to biosolids land application and analyzed to establish background conditions. Quarterly to semi-annual sampling should be a sufficient sampling frequency after land application has occurred. More frequent sampling may be warranted if the depth to water is small (Liu, 1982) or if there are other activities nearby that may adversely affect ground water quality and possibly implicate the land application activities.

Field Study Methodology

Two biosolids land application field studies located in arid to semi-arid rangeland settings have been instrumented to evaluate infiltration-related water quality degradation. These studies are part of ongoing research which has been published elsewhere (Aguilar et al, 1994; RBD and CDM, 1994) but salient aspects relating to the infiltration studies are summarized in the following section.

Meadow Springs Ranch Infiltration Studies

The Meadow Springs Ranch (hereafter referred to as the 'MSR' site) field studies are based on a series of bordered test plots located in the high plains grassland in northern Colorado. The MSR test plots consist of 60 15x15 meter "vegetative response" test plots and 24 3x10 meter "runoff response" test plots. Twenty-four of the vegetative response test plots were treated in 1991 with single applications of biosolids ranging from 0 to 34 Mg/ha; an equal number received single applications of composted biosolids at the same rates, and the remaining plots received biosolids and water treatment plant alum residuals in differing rates. The MSR runoff response test plots were evenly divided between higher (14-16 percent) and lower (7-8 percent) slopes. The runoff response plots consisted of four replications of single biosolids applications (in 1992) of 0, 17 or 34 Mg/ha at each slope type (RBD and CDM, 1994). Soils beneath both test plot sites consist of the Altvan Series of loamy soil.

Infiltration from natural precipitation events was evaluated at the vegetative response plots from autumn 1991 through autumn 1993. Monitoring consisted of a set of soil moisture blocks, tensiometers and lysimeter installed at four sites. Each set of devices was installed within the highest application rate of the biosolids and compost test plots, at an alum/biosolids test plots with an intermediate application rate, and at a nearby background site. Moisture blocks were installed to depths of 15, 30 and 45 cm, with paired tensiometers at the 30 and 45 cm depths. A suction lysimeter was also installed at each site at 45 cm. Precipitation events were recorded by an onsite tipping bucket raingage and continuous strip chart recorder. Evaporation was evaluated with an onsite Class A evaporation pan. Infiltration monitoring at the runoff response test plots consisted of moisture blocks and tensiometers installed in one control plot and one biosolids plot each at the lower and higher slopes. Moisture blocks were installed in two sets on a given plot, a depths of 15, 30, 45 and 60 cm. A single tensiometer was also installed at 30 cm at each plot. Measurements of soil moisture, precipitation and evaporation were made almost daily throughout the growing seasons in 1992 and 1993 at both sites, and weekly or as site access allowed during the winter months (RBD and CDM, 1994). Soil samples were also collected at 0-7.5, 7.5-15 and 15-30 cm depth intervals in July 1992 and July 1993 for analyses of metals, nitrate and other nutrient compounds.

In addition to monitoring natural precipitation at the MSR site, a simulated rainfall response test was conducted. This occurred in August 1992 at the runoff response test plots. Simulated rainfall was applied using a spray up - fall down system consisting of 200 cm tall sprinklers that distributed precipitation evenly across each plot. Rainfall duration was for 30 minutes at each plot, at an intensity ranging from 5.1 to 11.3 cm/hr. This is similar to the maximum intensity 30-minute storm events recorded in the region (RBD and CDM, 1994). Rainfall was recorded by a series of 12 rain gages in each plot. Runoff was determined from water levels in a calibrated holding tank located at the bottom of each bordered test plot. Infiltration was measured at hourly intervals during the day of the simulated rainfall event, and daily for several months thereafter.

A final monitoring aspect at the MSR site consisted of ground water monitoring sampling. Ten wells have been installed at the MSR site and are sampled quarterly for common ions and nutrient parameters. Water levels are gaged routinely and maps of the water table surface have been prepared to evaluate upgradient/downgradient flow directions and rates (RBD and CDM, 1994).

Sevilleta National Wildlife Refuge

The Sevilleta National Wildlife Refuge site (hereafter referred to as the 'SNWR' site) is on rangeland in central in New Mexico . The test plots at the SNWR site consist of three pairs of 3x10 m bordered plots on 6-7 percent slopes and 3 paired plots on 10-11 percent slopes. One of each pair of plots received a one-time application of biosolids in 1991 at 45 Mg/ha (20 dry tons/acre), with the other plot serving as a control (Aguilar et al, 1994). Soils beneath the site are classified as the Harvey-Dean Association and consist of fine loamy soils. The emphasis in this study was to evaluate vegetative response and runoff, so monitoring equipment used in the infiltration evaluations consisted of a calibrated runoff tank at the base of each test plot and an onsite self-activating recording rain gage. In addition, soil samples were collected in the upper 15 cm in 5 cm interval and analyzed for antecedent moisture content, nutrient compounds and metals prior to conducting the simulated rainfall experiments each year. Samples were also collected annually in 10 cm intervals down to 100 cm and analyzed for nitrate and TKN nitrogen (Aguilar et al, 1994).

Natural and simulated rainfall events were monitored in 1991 - 1993. Simulated rainfall events were conducted in a manner as described above for the MSR simulated rainfall tests. Rainfall rates ranged from 4.6 to 10.6 cm/hr for the 30 minute tests (Aguilar et al, 1994). Although unsaturated zone monitoring was not undertaken directly, relationships of runoff to rainfall on control and biosolids application test plots provide insight into the magnitude of infiltration that is possible at this site.

Results and Discussion

Results from monitoring infiltration associated with native precipitation at the land application study sites were obtained largely from the soil moisture blocks and tensiometers installed at the MSR site. The moisture block data showed responses from infiltration at all depths monitored. The 15 cm monitoring depth showed large and rapid responses to rainfall events at all test plot sites, and then equally rapid drying out of the soil. More seasonal patterns were observed at the 30 cm (12 inch) and 60 cm (24 inch) monitoring depths. Plots of soil moisture at these depths for each biosolids application type at the MSR vegetative response test plots are shown in Figures 1 and 2 for 1992 and 1993 data, respectively (RBD and CDM, 1994). Also shown on the plots are precipitation values recorded onsite. All soil moisture data shown on these figures were smoothed using an eleven-point moving average. Soil moisture values in the range of 75 to 85 percent probably equate to field capacity conditions, meaning that downward fluid flow occurs at those moisture contents.

Even with the large variation in soil moisture shown on figures 1 and 2, several trends are evident. Soils beneath the biosolids plots generally showed the highest soil moisture contents at both depths. Moisture contents were relatively high beneath the 60 cm depth through June of each year. This is probably due to wet spring conditions and lack of evapotranspiration. From July through November the soil becomes extremely dry at all depths except in response to large storm events (figures 1a, 2a). Increases in soil moisture content at the 30 cm depth lags behind precipitation events by approximately 3 to 7 days. This suggests moisture infiltration rates on the order of 10^{-5} cm/sec for the MSR soils under partially saturated conditions.

The tensiometers showed the same general trends as the soil moisture blocks. The rapid soil responses in the upper 15 cm due to precipitation and evapotranspiration effects were confirmed, as were the generally higher moisture contents beneath the biosolids plots. Conditions were generally dry enough at the MSR site during the monitoring period that on numerous occasions all of the water was withdrawn from the tensiometers by the high soil suction. This required repriming the tensiometers and loss of data until the tensiometers again came into equilibrium with the surrounding soil. As implied by the 60 cm soil moisture block data (Figures 1b, 2b), soil moisture was generally not high enough during the monitoring period to allow fluid samples to be collected by the suction lysimeters.

The simulated rainfall events conducted during the third week of August 1992 at the MSR site created distinct moisture infiltration fronts migrating into the soil columns. Plots of moisture content from the soil moisture blocks at the 7-8 and 14-16 percent slopes are shown in figures 3 and 4, respectively (RBD and CDM, 1994). The initially high moisture content readings shown on all plots represents the effects of moisture block installation in early August. The 15 cm (6 inch) moisture blocks show immediate responses to the simulated rainfall event in all plot types and slopes, followed by moisture contents approaching field capacity at the 30 cm depths. Three of the test plots also showed soil moisture contents approximately equal to field capacity at a depth of 45 cm (figures 3a, 3b and 4a), while none of the soils at the 60 cm depth reached field capacity. This indicates that most of the infiltrating water was attenuated in the upper 45 cm through water storage.

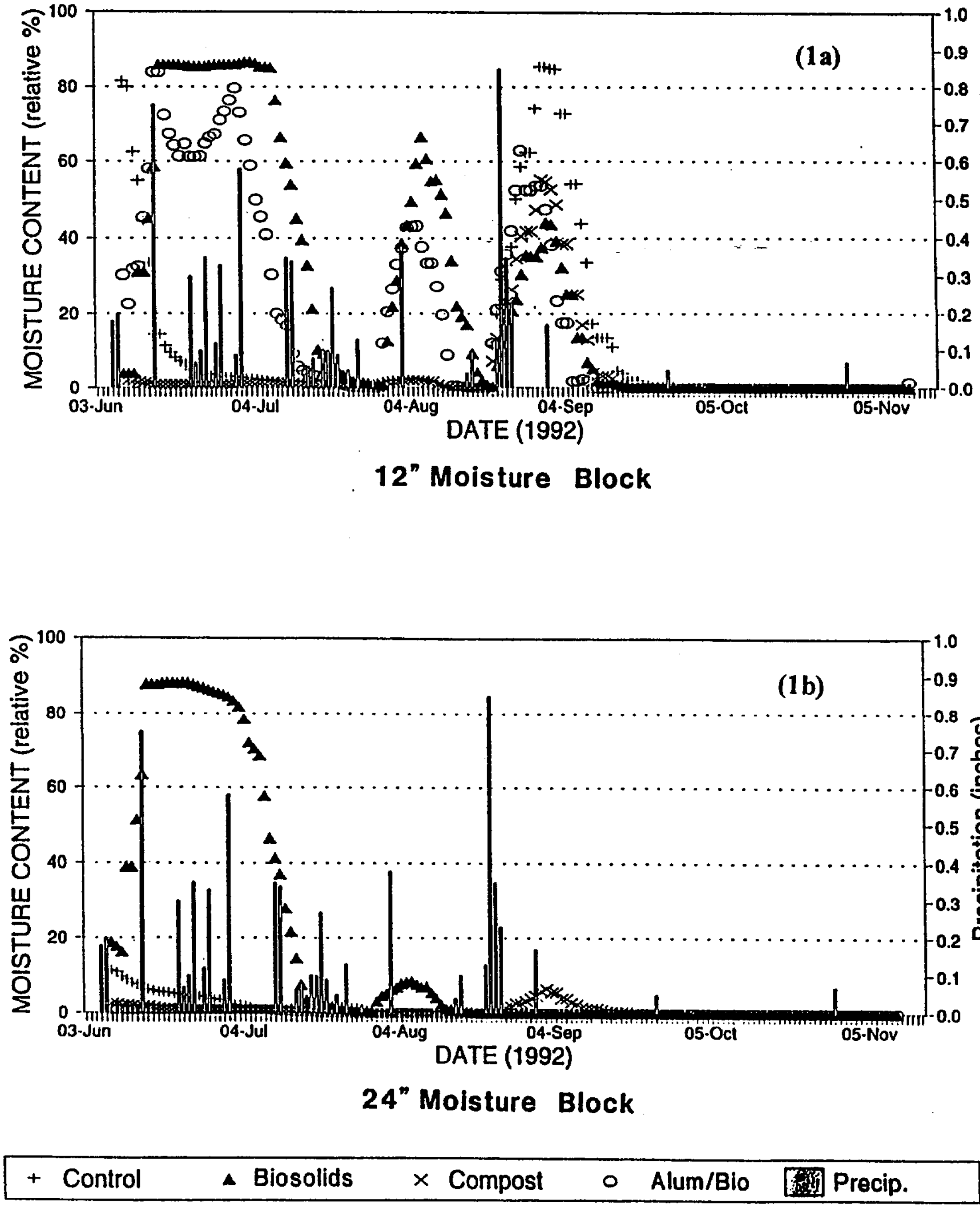

Figure 1. Meadow Springs Ranch averaged soil moisture block data for 1992. (A) 30 cm monitoring depth; (B) 60 cm monitoring depth.

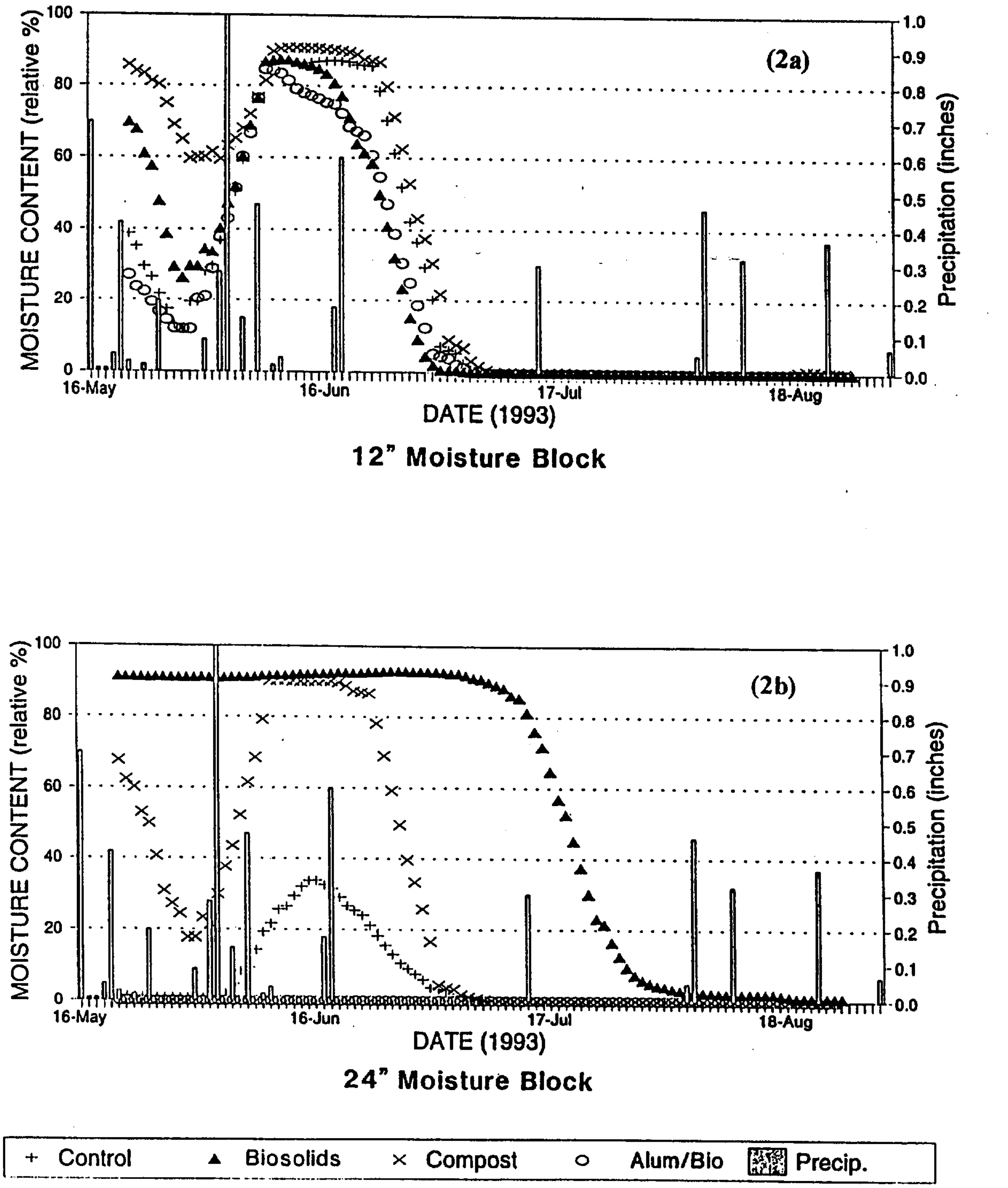

Figure 2. Meadow Springs Ranch averaged soil moisture block data for 1993. (A) 30 cm monitoring depth; (B) 60 cm monitoring depth.

The moisture fronts show a lag time as moisture infiltrates downward to each deeper moisture block. As shown on figures 3 and 4, the 30 cm moisture blocks at all four test plots showed lag times of about 3-5 days before reaching maximum moisture content. The 45 cm (18 inch) blocks show lags in moisture increase of 5-10 days from the start of the rainfall events, with the larger lag times at the biosolids-treated plots. The difference in moisture front migration rates between the control and treated plots is greatest at the 60 cm depth. Approximately 10 days were required after the simulated rainfall event for the moisture fronts to reach their maximum at the 60 cm depth beneath both control plots, while approximately 20 days were required beneath the treated plots. These travel times correspond to an average infiltration rates of approximately 7×10^{-5} cm/sec at the control plots and 3.5×10^{-5} cm/sec at the biosolids treated plots. Variation in moisture block responses at individual test plots of a given biosolids application rate is likely due to different soil properties and instrument responses to soil moisture. Lag times between precipitation events and soil moisture at depth were also observed by Nash et al (1991). The slightly lower moisture contents seen in the moisture contents of the biosolids treated plots is likely due to water being absorbed by the biosolids at the surface and less water being available for infiltration.

Antecedent moisture content and runoff/rainfall data were computed for the simulated experiments at both the MSR and the SNWR sites. Assuming evapotranspiration was minimal over the 30-minute duration of the rainfall tests, average infiltration rates were calculated based on the difference between precipitation and runoff volumes and rates. A summary of these data are shown in Table 1. Soil moisture and infiltration rates are both much lower at the MSR site than the SNWR site as shown in the table. Infiltration and runoff/rainfall rates are comparable for a given slope type and biosolids application rate, suggesting that slope does not have a strong influence on these parameters. The data in Table 1 also indicates that runoff rates are much larger and infiltration rates are much smaller on control plots compared to biosolids plots. Infiltration rates shown on Table 1 for the MSR site are approximately two orders of magnitude larger than rates calculated based on the time required for the soil moisture fronts to reach a depth of 60 cm. The higher infiltration rates shown on Table 1 are due to the higher moisture content of the surface soils, which allowed more water to infiltrate into the upper soil zone. Table 1 infiltration rates also account for other water losses, such as evapotranspiration and water absorption by the biosolids, and so may be biased slightly high. Natural rainfall events that produced runoff at the SNWR were recorded only in 1991 and 1993 (Aguilar et al, 1994). Runoff and infiltration relations appear consistent with those of the simulated experiments, although the magnitudes of the natural infiltration events were small enough that definitive conclusions could not be made regarding infiltration.

Soil sample results collected from the MSR site showed increases in ammonia, nitrate, electrical conductivity, P, Cu, An, Mo and Hg for all biosolids application rates in the 0-7.5 cm depth interval compared to samples from control plots, while ammonia and nitrate concentrations also increased in soil at the 7.5-15 and 15-30 cm depth intervals (RBD and CDM, 1994). Soil sample results from the SNWR site showed similar patterns, where Cu, Pb and Zn showed increases in shallow soils down to about 10 cm compared to control plots. Nitrate, ammonia and P showed

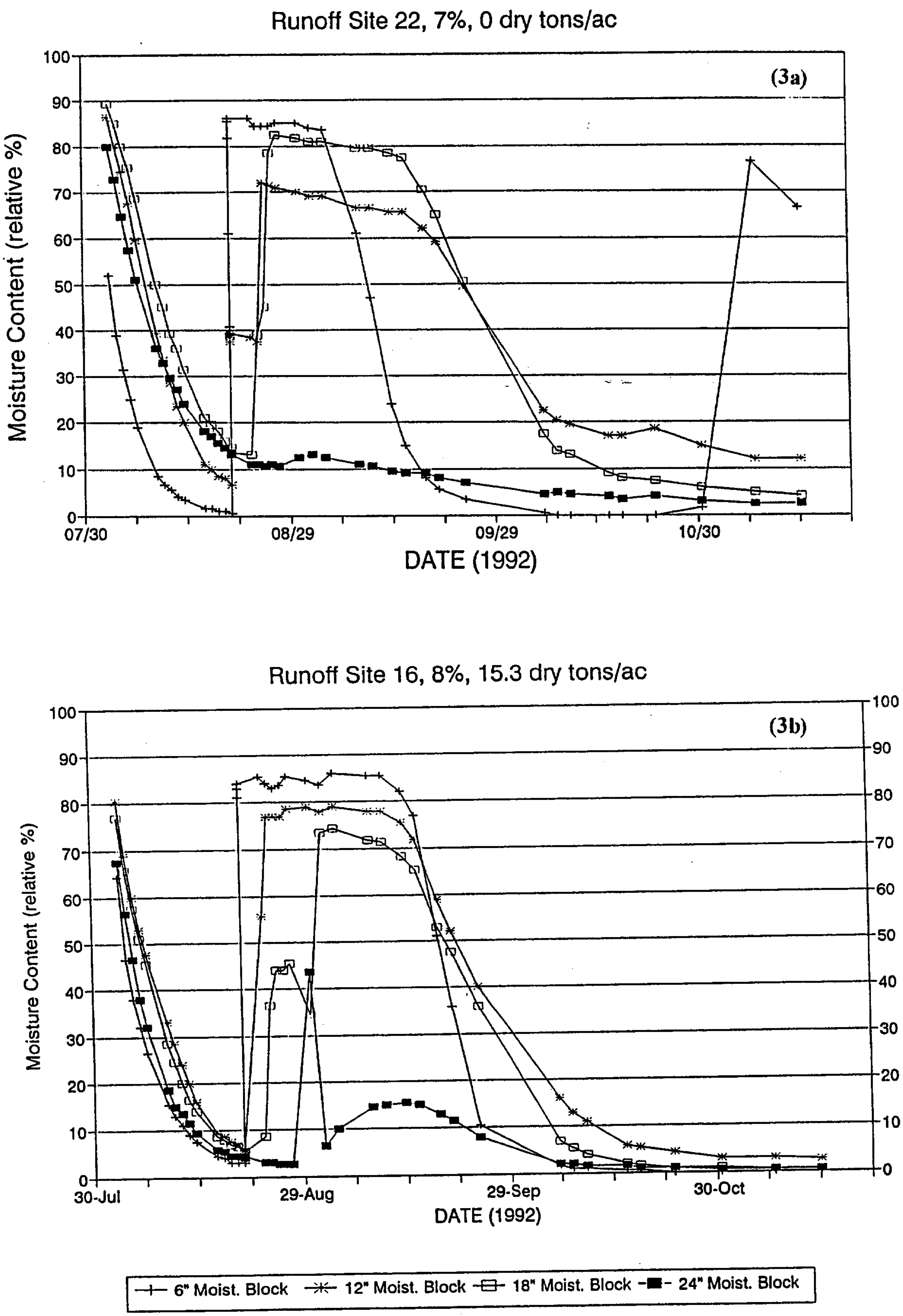

Figure 3. Meadow Springs Ranch averaged soil moisture block data for 1992 simulated rainfall experiment, on 7-8 percent slopes. (A) 0 Mg/ha biosolids; (B) 34 Mg/ha biosolids.

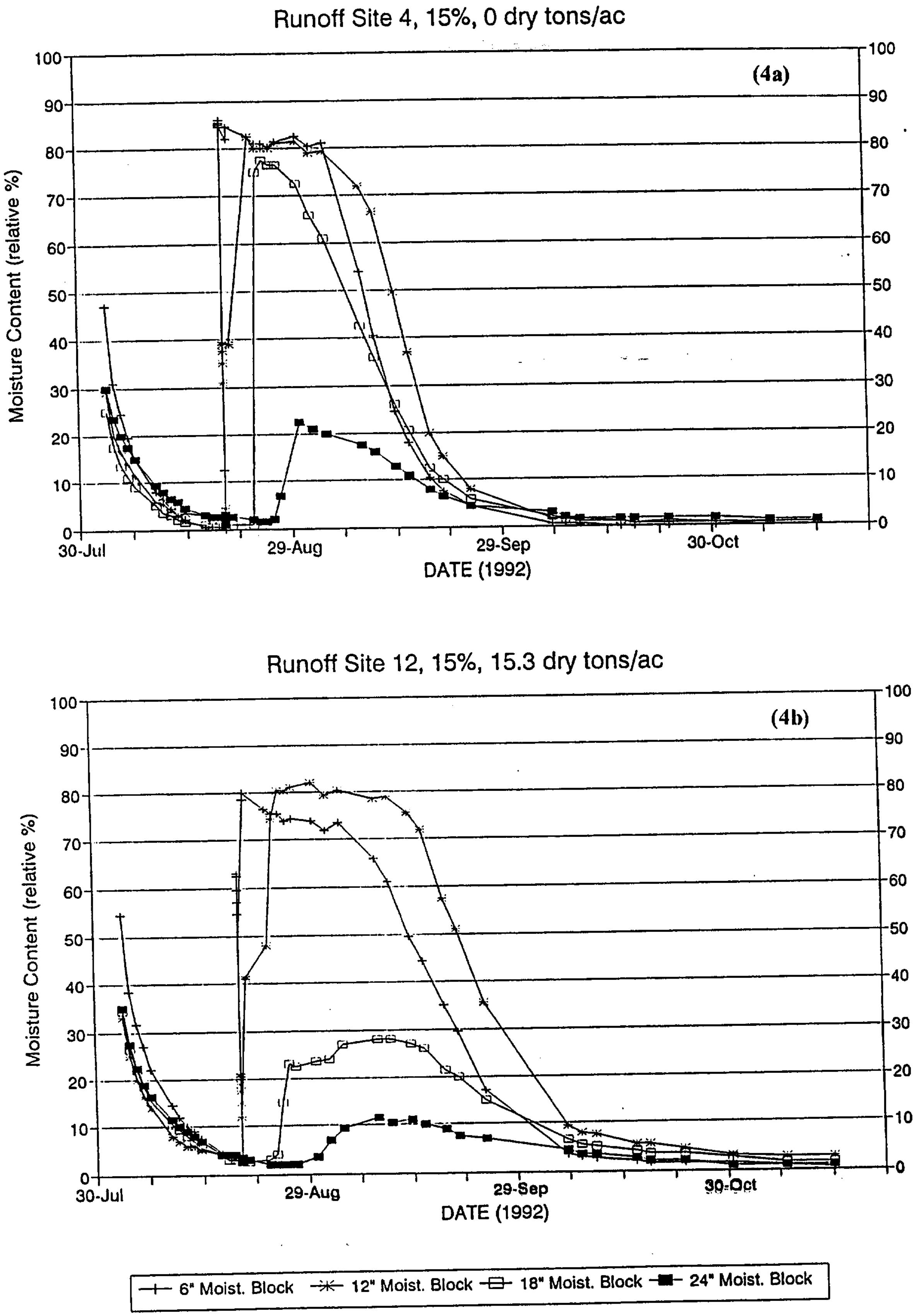

Figure 4. Meadow Springs Ranch averaged soil moisture block data for 1992 simulated rainfall experiment, on 14-16 percent slopes. (A) 0 Mg/ha biosolids; (B) 34 Mg/ha biosolids.

Table 1. Summary of Infiltration Results Simulated Rainfall Experiments on Biosolids-Ammended Test Plots

Site	Application Rate (Mg/h	Antecedent Moisture Content (%)	
		7 - 8% slope	14 - 16% slope
MSR	0	0.03 ---	0.09 (0.03)
	17	0.03 ---	0.08 (0.04)
	34	0.03 ---	0.05 (0.02)
		6 - 7%	10 - 11%
SNWR	0	2.52 (0.78)	3.63 (2.43)
	45	3.59 (1.73)	4.80 (1.84)
		Infiltration Rate (cm/hr)	
		7 - 8%	14 - 16%
MSR	0	3.8 (0.9)	2.8 (1.0)
	17	4.6 (0.9)	4.6 (0.9)
	34	4.8 (0.7)	5.4 (0.4)
		6 - 7%	10 - 11%
SNWR	0	7.4 (1.2)	6.4 (1.3)
	45	9.0 (1.2)	8.4 (1.5)
		Runoff/Rainfall	
		7 - 8%	14 - 16%
MSR	0	0.451 (0.147)	0.492 (0.053)
	17	0.302 (0.121)	0.316 (0.183)
	34	0.245 (0.130)	0.357 (0.103)
		6 - 7%	10 - 11%
SNWR	0	0.127 (0.117)	0.107 (0.102)
	45	0.001 (0.003)	0.002 (0.003)

MSR= Meadow Springs Ranch, CO; SNWR= Sevilleta Nat. Wildlife Refuge, NM.
First value is average for plots; number in ( ) is standard deviation.
Sevilleta dataset has 6 plots/slope/application rate.
Each Sevilletta plot measured 3 times (9/91, 9/92, 9/93).
Meadow Springs dataset has 4 plots/slope/application rate.

increases at the SNWR biosolids application plots compared to the control plots, and elevated nitrate concentrations were detected down to 80 cm (Aguilar et al, 1994).

Ground water monitoring performed at the MSR has shown that common ions, metals, nutrient compounds and synthetic organic compounds are within natural background levels at all wells or are below analytical detection limits. Ongoing quarterly monitoring at the MSR site will continue as field studies proceed.

Conclusions

A series of studies have shown the potential for biosolids-related leachate to contaminate underlying soils and ground water. Most infiltration-related studies conducted in arid and semi-arid environments have shown that infiltration is limited to the upper meter or less under normal precipitation conditions, but deeper infiltration is possible under extreme storm events. The data collected from the MSR site where biosolids have been applied indicate that neither natural precipitation nor leachate will infiltrate below about a meter in depth.

In a simulated rainfall experiment at the MSR site representing a large storm event, soils reached field capacity down to 45 cm, but only moderate increases in soil moisture were evidenced at 60 cm. Based on results obtained from natural precipitation events at the MSR site in 1992 and 1993, soil moisture infiltrates below biosolids land application areas to depths greater than 60 cm only in spring and early summer, or for brief periods following intense summer thunderstorms. Spring and early summer appear to be the times when deep infiltration is most likely because evapotranspiration effects are minimal and soils frequently have a relatively high moisture content due to snowmelt and spring rains. These finding are consistent with infiltration studies conducted in arid and semi-arid regions on undisturbed control plots in NM, ID, CA, TX, OK and UT.

Biosolids application was shown to cause increased soil moisture contents in all depths during the monitoring periods at both the MSR and SNWR sites. Higher antecedent moisture contents (AMCs) observed in site soils beneath biosolids test plots correlated closely with higher infiltration rates in test plots at the SNWR site. This trend could not be corroborated at the MSR site due to very low AMC values in soils there. The higher AMC contents beneath biosolids-treated plots will increase the rate of infiltration and thus the potential for deeper migration of infiltrating fluid. Land surface slope appears to make little difference in infiltration rate, whereas biosolids application rate shows a strong direct correlation to infiltration rate and soil moisture content.

Soil moisture blocks and neutron probes appear to be appropriate soil moisture monitoring devices in remote arid and semi-arid environments. Laboratory testing is needed to correlate readings from either device to actual soil moisture contents. Soil tensiometers required constant maintenance and still became dewatered frequently, necessitating repriming and a new equilibration period with surrounding soil moisture. For these reasons, use of tensiometers is not recommended unless daily monitoring and maintenance can be undertaken throughout the growing season. Due to the very dry soil conditions encountered, the suction lysimeters were not effective in collecting samples of soil moisture from the unsaturated zone. These devices may be useful in settings where relatively high moisture contents (those approaching field capacity) for a given soil are anticipated. Ground water monitoring will provide the best absolute means to evaluate changes in ground water quality associated with land application of biosolids, but may be necessary only when the unsaturated zone monitoring indicates that deep infiltration occurs at a site. Borehole drilling is recommended in order to determine the depth to ground water and evaluate the presence of low-permeability layers within the unsaturated zone.

The moisture contents observed at the MSR site indicate that the potential for lateral migration leachate to nearby seeps and the vertical migration of leachate to the water table are both very low. Ground water sampling results do not indicate impacts to the shallow aquifer system from biosolids application. Therefore, the risk to water quality degradation associated with infiltration of leachate beneath biosolids application areas at the MSR site appears minimal. The larger AMC values and higher infiltration rates present at the SNWR test plots suggest a greater potential for infiltration. The accumulation of calcium carbonate at 40-80 cm and gypsum at 160-170 cm, if indicative of the average and maximum depths of infiltration, supports the greater depth of infiltration compared to the MSR site.

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References

Aguilar, R., and Aldon, E.F. (1991a) Runoff and sediment rates on San Mateo and Querencia soils, Rio Puerco Watershed Management Area, NM. USDA Forest Service Rocky Mountain Forest Experiment Station Research Note RM-506. Albuquerque, NM.

Aguilar, R., and Aldon, E.F. (1991b) Seasonal water flux and potential for leaching in a semiarid rangeland soil. In: *Proceedings of the Fifth National Outdoor Action Conference on Aquifer Restoration, Ground water Monitoring and Geophysical Methods.* May 13-16, 1991, Las Vegas NV. National Water Well Assoc. pp 669-683.

Aguilar, R., and Loftin, S.R. (1992) Sewage sludge application in semiarid grasslands: effects on runoff and surface water quality. In: *Proceedings of the 36th Annual New Mexico Water Conference.* New Mexico Water Resources Research Institute, Technical Report No 265, NM State Univ., Nov. 7-8, 191, Las Cruces, NM.

Aguilar, R., Loftin, S.R., Ward, T.J., Stevens, K.A., and Gosz, J.R., (1994) Sewage sludge application in semiarid grasslands: effects on vegetation and water quality. New Mexico Water Resources Research Institute, Technical Report No 285, NM State Univ., Las Cruces, NM.

Baxter, J.C., Martin, W.J., Sabey, B.R., Hart, W.E., and Cohen, D.B., (1977) Comprehensive summary of sludge disposal recycling history. EPA Report EPA-600/2-77-054.

Berg, R.C., Morse, W.J., and Johnson, T.M., (1987) Hydrogeologic evaluation of the effects of surface application of sewage sludge to agricultural land near Rockton, Illinois. *Env. Geol. Notes 119.*

Branson, F.A., Gifford, G.F., and Owen, J.R., (1972) Rangeland Hydrology. Society for Range Management, Denver Colorado. Range Science Series No. 1.

Bruggeman, A.C., and Mostaghimi, S. (1993) Sludge application effects on runoff, infiltration and water quality. *Water Res. Bull.,* 29:15-26.

Carlson, D.H., Thurow, T.L., Knight, R.W., and Heitschmidt, R.K., (1990) Effect of honey mesquite on the water balance of Texas rolling plains rangeland. *J. Rangeland Mgmnt.,* 43:491-496.

Chang, A.C., and Page, A.L., (1990) Long-term environmental effects associated with land application of municipal sludges: a review. In: *Solid/Liquid Separation: Waste Management and Productivity Enhancement.* Battelle Press, Columbus, OH, pp 439-454.

Coupe, R.H., and Macy, J.A., (1993) Surface water and streambed sediment quality of streams draining surface mined land reclaimed with sewage sludge, Fulton County, Illinois. U.S. Geol. Survey Water Resources Invest. Report 93-4056.

Davis, R.D., Carlton-Smith, C.H., Stark, J.H., and Campbell, J.A., (1988) Distribution of metals in grassland soils following surface applications of sewage sludge. *Environ. Poll.,* 49:99-115.

Duell, L.F.W., (1990) Estimates of evapotranspiration in alkaline scrub and meadow communities of Owens Valley, California, using the Bowen-ratio, Eddy-correlation, and Penman-combination methods. U.S. Geol. Survey Water Supply Paper 2370-E.

Epstein, E., Effect of sewage sludge on some soil physical properties. *J. Env. Qual.,* 4:139-142.

Fresquez, P.R., Aguilar, R., Francis, R.E., and Aldon, E.F., (1991) Heavy metal uptake by blue grama growing in a degraded semiarid soil amended with sewage sludge. *Water, Air and Soil Poll.,* 57/58:903-912.

Fresquez, P.R., Francis, R.E., and Dennis, G.L., (1990) Sewage sludge effects on soil and plant quality in a degraded, semiarid grassland. *J. Env. Qual.,* 19:324-329.

Gallier, W.T., Brobst, R., Aguilar, R., Barbarick, K., and Hegeman, P. (1993) Rx for Rangelands. *Water Env. & Tech.,* 5:56-60.

Goda, T., Kuboi, T., and Fujii, K., (1986) Environmental impacts of sewage sludge applied to cropland. *Internat. J. Env. Studies,* 27:239-253.

Kelling, K.A., Peterson, A.E., and Walsh, L.M., (1977) Effect of wastewater sludge on soil moisture relationships and surface. *J. Wat. Poll. Cont. Fed.,* 49:1698-1703.

Khaleel, R., Reddy, K.R., and Overcash, M.R. (1981) Changes in soil physical properties due to organic waste applications: a review. *J. Environ. Qual.,* 10:133-141.

King, C.H., Houck, C.P., and Deadmond, L.L., (1988) Land application of Kodak Colorado Division wastewater treatment plant sludge. In: *Analytical Techniques and Residuals Management in Water Pollution Control Specialty Conference*, April 19-20, 1988, Alexandria, VA. Water Pollution Control Federation, pp 241-261.

Kiusalaas, N.J. and Poeter, E.P. (1992) Estimation of groundwater recharge using neutron probe moisture readings near Golden, Colorado. Colorado Water Resources Research Institute, Completion Report No. 165, Colo. State Univ., Fort Collins, CO.

Lauenroth, W.K., and Sims, P.L. (1976) Evapotranspiration from a shortgrass prairie subjected to water and nitrogen treatments. *Water Res. Res.* 12:437-442.

Lerch, R.N., Barbarick, K.A., Westfall, D.G., Follett, R.H., and McBride, T.N., (1990) Sustainable rates of sewage sludge for dryland winter wheat production: II. production and income. *J. Production Agric.,* 3:66-71.

Liang, Y., and Corey, R.B., (1993) Redistribution of sludge-borne cadmium, copper, and zinc in a cultivated plot. *J. Env. Qual.,* 22:1-8.

Liu, D., (1982) The effect of sewage sludge land disposal on the microbiological quality of groundwater. *Water Res.,* 16:957-961.

Loaque, K. (1992) Soil water content at R-5: Part 2, impact of antecedent conditions on rainfall-runoff simulations. *J. Hydrol.,* 139:253-261.

McCully, W.G., and Haas, R.H., (1970) An evapotranspiration model for great plains grasslands. In: *Evapotranspiration in the Great Plains.* March 23-25, 1970, Bushlands, Texas. Great Plains Agricultural Council Research Committee Publication No. 50, pp 55-65.

Mostaghimi, S., Younos, T.M., and Tim, U.S. (1992) Effects of sludge and chemical fertilizer application on runoff water quality. *Water Res. Bull.,* 28:545-552.

Nash, M.S., Wierenga, P.J., and Gutjahr, A., (1991) Time series analysis of soil moisture and rainfall along a line transect in arid rangeland. *Soil Sci.,* 152:189-198.

Nixon, P.R., Lawless, G.P., and McCormick, R.L., (1972) Soil hydrology in a semiarid watershed. *Trans., Am. Soc. Agric. Eng.,* 15:985-991.

O'Brien, P., and Mitsch, W.J., (1980) Root zone nitrogen simulation model for land application of sewage sludges. *Ecolog. Modelling,* 8:233-257.

Otoma, S., and Kuboi, T., (1988) Prediction of time variation in water and chloride profiles in a soil subject to cropping and annual application of sewage sludge. *J. Hydrol.,* 99:1-17.

Parton, W.J., Lauenroth, W.K., and Smith, F.M., (1981) Water loss from a shortgrass steppe. *Agric. Meteorol.,* 24:97-109. Elsevier Scientific Pub. Co., The Netherlands.

RBD Engineers Inc, and Camp Dresser & McKee Inc, (1994) Wastewater plan for biosolids and water treatment plant residuals, City of Fort Collins, CO.

Sabey, B.R., Pendleton, R.L., and Webb, B.L., (1990) Effect of Municipal Sewage Sludge application on growth of two reclamation shrub species in copper mine spoils. *J. Env. Qual.,* 19:580-586.

Seyfried, M.S. (1991) Infiltration patterns from simulated rainfall on a semiarid rangeland soil. *Soil Sci. Soc. Am. J.,* 55:1726-1734.

Stephens, D.B., Knowlton, R.G., McCord, J., and Cox, W., (1986) Field study of natural ground water recharge in a semi-arid lowland. New Mexico Water Resources Research Institute, Technical Report No 177, NM State Univ., Las Cruces, NM.

Sommers, L.E., and Nelson, D.W., (1981) Monitoring the response of soils and crops to sludge applications. In: *Sludge and Its Ultimate Disposal.* Ann Arbor Science, Ann Arbor, MI, pp 217-239.

Tackett, S.L., Winters, E.R., and Puz, M.J., (1986) Leaching of heavy metals from composted sewage sludge as a function of pH. *Canadian J. Soil Sci.,* 66:763-765.

Tromble, J.R., Renard, K.G. and Thatcher, A.P., (1974) Infiltration for three rangeland soil-vegetation complexes. *J. Range Mgmnt.,* 27:318-321.

Trout, J.T., Smith, J.L., McWhorter, D.B., (1975) Environmental effects of land application of digested municipal sewage sludge. National Science Foundation, Washington, DC. Applied Science and Research Applications.

Welch, J.E., and Lund, L.J., (1989) Zinc movement in sewage sludge treated soils as influenced by soil properties, irrigation water quality and soil moisture level. *Soil Sci.,* 147:208-214.

Weltz, M.A., and Blackburn, W.H., (1993) Modeling water balance with the ERHYM model on south Texas rangelands. *Water Res. Bull.,* 29:461-474.

Wengel, R.W. and Griffin, G.F., (1979) Potential ground water pollution from sewage sludge application on agricultural land. Connecticut Univ., Storrs Inst. of Water Resources Research Project Technical Completion Report, OWRT A-074-Conn(1).

Williams, D.E., Vlamis, J., Pukite, A.H., and Corey, J.E., (1987) Metal movement in sludge-amended soils: a nine year study. *Soil Sci.,* 143:124-131.

Municipal Biosolids Application Effect on Infiltration and Erosion on Desert Grasslands

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Abstract

Land application of biosolids on arid and semi-arid rangeland has led to growing interest in determining the hydrologic consequences of this practice. The objectives of this work are: 1) to evaluate the effects of varied rates of biosolids on infiltration, erosion, and soil water quality; and 2) to examine some of the mechanisms, whereby biosolids application affect infiltration. A portable single-nozzle rainfall simulator was used to simulate rainfall on 0.5 m^2 plots to determine infiltration and erosion. In the infiltration and erosion experiment, effects of vegetation cover (bare or vegetated), season of simulation (winter or summer), soil type (Chilicotal or Stellar), and rate of biosolids applied (0, 7, 34, or 90 dry Mg ha^{-1}) were studied. Microlysimeters were used in the percolation and water quality study for two soils (Armesa and Stellar) with similar rates of applied biosolids plus 18 dry Mg ha^{-1}. The quantity of water to wet the profile and the quality of the leachate were measured. Application of biosolids, especially at the 34 and 90 Mg ha^{-1} rates, increased infiltration rate and cumulative infiltration while decreasing erosion. The clay layer in the Stellar soil limited water percolation. Increases in NO_3^{1-}-N, SO_4^{2-}, and PO_4^{3-} were observed with an increase in biosolids application rate. Leachate pH tended to decrease with increasing biosolids rate. Topical application of biosolids has important hydrologic consequences in desert grasslands that depend upon kind of soil being treated and the rate of biosolids applied. Risk of groundwater contamination from soil leachate in the study area is minimal.

Keywords: biosolids, infiltration, erosion, ground water, hydrology

With the recent increase in land application of biosolids has come demand for a better understanding of its ecological implications. The application of biosolids can supply large quantities of nitrogen and phosphorus to the soil and agricultural crops (Soon et al. 1978). Use as a fertilizer is not a new practice. People have used this resource and even its untreated counterpart (raw sewage) for centuries as a soil amendment on cropland.

Fresquez et al. (1990) have demonstrated some of the benefits of biosolids application on semi-arid grasslands in New Mexico. Hydrologic consequences of biosolids application in semi-arid environments have also been examined (Aguilar and Loftin 1992, and Harris-Pierce et al. 1995). These studies found a decrease in runoff and erosion with biosolids application while maintaining surface water quality. The low precipitation in arid and semi-arid regions would seem to be favorable for maintaining surface and ground water quality.
An understanding of the effects of biosolids application on hydrology is necessary in properly managing the water resource and in explaining ecosystem responses to this treatment. The

objectives of this work are: 1) to evaluate the effects of varied rates of biosolids on infiltration, erosion, and soil water quality; and 2) to examine some of the mechanisms, whereby biosolids application affect infiltration.

Materials and Methods

Study Site

This research is being conducted in far west Texas near the town of Sierra Blanca. This is a northern Chihuahuan Desert grassland. The climate of the region is characterized by limited precipitation and high afternoon air temperatures in the summer. The annual precipitation is about 31 cm with about 65 % falling in the months of July, August, and September. The shallowest known ground water at the study site is greater than 150 meters deep and has not been a source for domestic use.

The physiography of the study area is typical of the northern Chihuahuan desert. The study area is located on a large gently sloping to nearly level bolson that is bounded by igneous mountains on the west and limestone escarpments on the north and east.

The soils at the study area are diverse. Although all the soils being studied are classified as aridisols; texture, structure, and expression of pedogenic features are widely varied. There is not a Natural Resource Conservation Service (NRCS; previously the Soil Conservation Service) soil survey available at this time for either Hudspeth or adjacent Culberson counties. The published survey for adjacent El Paso county is in geology that is strongly influenced by the Rio Grande river unlike the locally derived alluvium of the study area. Soil series have not been established by the NRCS for the study area and thus the soil series names cited in this work are tentative, providing only a field name. The three soils studied have been described and classified as: Armesa fine sandy loam, 0 to 1 percent slope (Fine-loamy, carbonatic, thermic Ustic Haplocalcids); Chilicotal gravelly sandy loam 1 to 3 percent slope (Loamy-skeletal, mixed, thermic Ustic Haplocalcids); and Stellar taxajunct sandy clay loam, 1 to 3 percent slope (Fine, mixed, thermic Ustertic Haplargids).

The Armesa soil has an 11 cm thick fine sandy loam surface horizon that overlies a 33 cm thick sandy clay loam cambic horizon. The calcic horizon at 54 cm below the surface has about 40 % by volume calcium carbonate segregations. The similar Chilicotal soil has a 12 cm thick gravelly sandy loam surface horizon that overlies a 30 to 50 cm thick gravelly sandy loam cambic horizon. The calcic horizon is generally at a depth greater than 50 cm. The Chilicotal soil has a significant cover of pebble sized rock fragments that armor the soil. The Stellar soil has a 10 cm thick non-calcarious sandy clay loam surface horizon that overlies a clay argillic horizon. The argillic horizon has well-developed prismatic structure that parts to subangular blocks in the upper 20 cm but becomes extremely dense below a depth of 30 cm. A calcic horizon is also present in the Stellar soil at depths between 50 and 100 cm.

Vegetation varies with soil and each soil studied constitutes a unique range site. A Loamy range site variant is associated with the Stellar soil. The vegetation on the Loamy site is dominated by tobosagrass (*Hilaria mutica*) with some mesquite (*Prosopis glandulosa*) and lotebush (*Ziziphus obtusifolia*). A Gravelly range site is associated with the Chilicotal soil. The Gravelly site has a sparse herbaceous cover of primarily fluffgrass (*Dasyochloa pulchella*) and the half-shrub broom snakeweed (*Gutierrezia sarothrae*). The dominant shrub is creosotebush (*Larrea tridentata*). A Sandy Loam range site is associated with the Armesa soil. The Sandy Loam range site has abundant herbaceous cover of black grama (*Bouteloua eriopoda*) and sand dropseed (*Sporobolus cryptandrus*) with shrubs of longleaf mormon tea (*Ephedra trifurca*) and Yucca (*Yucca elata*).

Rainfall Simulator

A portable single-nozzle rainfall simulator similar to the one used by N.Z. Elkins (1983), Wilcox et al. (1986), and K.E. Spaeth (1990) was used to simulate the rainfall for the infiltration and erosion experiment. The nozzle was placed 2 m above the surface of the plot. A nozzle pressure of 20.7 kPa (3 psi) was maintained during simulation to produce a mean rainfall intensity on the 0.5 m^2 plots of 16 cm hr^{-1} with a median drop size between 1.2 and 2 mm.

A drop with a diameter of 1.5 mm falling in still air from a height of 2 m has an impact velocity of 4.50 m s^{-1} or 82 % of terminal velocity (G.D. Bubenzer, 1979). Since the water at the nozzle orifice in this simulator is under pressure it is reasonable to assume that the impact velocity would be greater than that reported for free falling drops.

Infiltration Measurement

Infiltration was measured on the Gravelly site during the summer of 1994. Rate of application comprised treatments. The rates of biosolids tested were 0, 7, 34, and 90 dry Mg ha^{-1}. Each treatment was replicated 5 times. On the Gravelly site all treatments were on bare ground because of the paucity of vegetation.

Infiltration on the Loamy site was measured during both the winter and summer of 1994. Treatments on the Loamy site consisted of: 1) bare and vegetated cover conditions; 2) the biosolids rates of 0, 7, 34, and 90 dry Mg ha^{-1} topically-applied (Initial biosolids quality is presented in table 1); and 3) winter and summer seasons of simulated rainfall. Each treatment was replicated 5 times.

A square 0.5 m^2 steel frame was positioned on each plot and driven into the ground, bounding the plot edges on three sides. The fourth side of the frame, set on the lowest edge of the plot, was flush with the soil surface and was fitted with a runoff collection pan. The runoff collection pan was sheltered from the rain, but allowed runoff water to flow from the plot to a low corner in the pan. The rainfall simulator was positioned on each plot and a thirty-minute rainfall was simulated. The water used in the simulation was tap water that ultimately came from the town of Van Horn, Texas. The runoff water that collected in the lower corner of the collection pan was transferred through a nylon hose and suction pump to a graduated cylinder. The volume was then recorded for each period. Runoff was recorded every 2.5 min for the first 10 min and every 5 min for the remaining 20 min. The difference in runoff and applied rainfall for each period is the infiltration for that period.

Erosion Measurement
Erosion measurements were made on the same plots as the infiltration measurements while the infiltration measurements were being made.

After the volume of runoff had been determined for a period the runoff water was poured into a clean bucket. At the end of each of 3, 10 min periods the runoff water that had been collected during the period was thoroughly mixed and a 1 liter subsample was collected. The volume of the subsample was measured and then the sample was filtered to remove the sediment. The sediment was oven-dried at 105^0F for 24 hr and weighed to determine the sediment concentration for that period. The concentration multiplied by the volume of runoff yielded the quantity of erosion.

Microlysimeters
Soil water quality data were collected from microlysimeters. Two contrasting soils were used in this experiment, Armesa and Stellar. Soil water quality was measured on two occasions for each soil. Each soil received treatments composed of five biosolids application rates. The five biosolids application rates were 0, 7, 18, 34, and 90 dry Mg ha^{-1}. This study was a completely randomized design and each of the 5 rate treatments were replicated 5 times.

Lysimeters were constructed from polyvinyl chloride pipe (PVC) with an internal diameter of 25.4 cm. Each PVC pipe was cut into 90 cm sections, and one end was beveled or tapered so that it could be more easily driven into the soil. Holes, were drilled on either side of the pipe near the flat cut end so that U-hooks could be used to aid in moving the excavated lysimeter. The lysimeters, once they had a core soil sample, weighed approximately 23 kg for the Stellar soil and 68 kg for the Armesa soil.

The soils were prepared in the field by thoroughly wetting them to reduce resistance for inserting the lysimeter into the soil. A backhoe was used to apply force to the lysimeters, fitted with a plywood cover, which drove them into the ground. Lysimeters were inserted in plant interspaces. In the Stellar soil, tubes would only penetrate approximately 30 cm, to the top of a dense clay layer. Standing water, impounded in the lysimeter tube for approximately 2 weeks, failed to penetrate this layer. In the Armesa soil we had no difficulty inserting the tubes a full 75 cm. The lysimeters were allowed to dry for a few days and then excavated and transported to the laboratory where the leaching was to take place. Lysimeters were placed on plexiglass squares, with 13, 0.64 cm perforations that were used to create a base plate for the lysimeter. The lysimeter-plexiglass contact was caulked with silicon around the base to prevent drainage water from escaping collection. The plexiglass fitted lysimeters sat on racks that allowed easy access to the bottom of the lysimeter where a collection bucket was placed. Collection buckets were constructed of plastic and would hold approximately 4000 ml of leachate. The lysimeters were then prepared for the experiment by taping over the holes in the base plate and adding 1500 ml and 3500 ml of water to the Stellar and Armesa soils, respectively. The lysimeters were allowed to stabilize for 24 hr before the tape was removed. This volume of water was added to bring the soil to near field capacity.

After the lysimeters had been prepared, the biosolids were weighed and placed on the soil surface inside each lysimeter according to a randomization scheme. Quality for biosolids applied to each soil type is given in table 2 and table 3 for both Armesa and Stellar soils, respectively.

Leaching Process and Chemical Analysis

Tap water, from the same source as used in the infiltration and erosion study, was used in all of the wetting operations. Water was applied to the lysimeters with a sprinkler device. Using a 1000 ml plastic graduated cylinder, water was measured at 500 or 1000 ml increments and emptied into the sprinkler. Water was then applied to the lysimeters at such a rate and at intervals so water would not pond in the tube. Sufficient water was applied to exceed the field capacity of the soil and to allow collection of 1.2 liters of leachate for analysis.

From the leachate, a sample was poured into a 125 ml nalgen bottle and preserved with 6 drops of concentrated nitric acid, while the remainder was poured into a 1 liter nalgen bottle. Both bottles were then labeled to identify the sample. Five samples of tap water were also collected to quantify initial water quality.

The 30 samples in the 1 liter nalgen bottles were analyzed on site with a Hach kit for ortho-phosphates, nitrate-nitrogen, pH, sulfates, calcium hardness, total hardness, and chlorides. Sample water tested with the Hach kit was kept relatively cool to preserve the sample and analyzed within 24 hr. The 125 ml samples were sent to a commercial laboratory to be analyzed for 24 elements including: silver, aluminum, arsenic, barium, beryllium, calcium, cadmium, cobalt, copper, chromium, iron, potassium, magnesium, manganese, molybdenum, sodium, nickel, phosphate, lead, antimony, strontium, titanium, vanadium, and zinc.

The procedures of lysimeter leaching, leachate collection, and chemical analysis were done initially on freshly applied biosolids and once again 60 days later for the Armesa soil and 110 days later for the Stellar soil.

Results and Discussion

Infiltration and Erosion

Factors that were most important in affecting infiltration and erosion were soil type (site), biosolids application rate, and vegetation cover condition (not necessarily in that order). The season when rainfall was simulated was not an important factor. Due to differences in the factors tested on each soil type, a subset of the data that provides balanced combinations of factors were used to explore each effect.

A subset of these data that had balanced combinations of Stellar and Chilicotal soils was selected. Only plots with a bare cover condition and that were measured in the summer of 1994 met the balanced criteria. From this data set soil-rate interactions may be explored.

Infiltration flux curves for both soil types without the addition of biosolids (Fig. 1) show very similar curves. Infiltration flux falls sharply to a stage III infiltration of about 3.1 cm hr^{-1} and 1.9 cm hr^{-1} for Chilicotal and Stellar soils, respectively, within about 10 min from the start of simulation. Addition of 7 Mg ha^{-1} of biosolids did not yield a stage III infiltration flux that was significantly different ($P > 0.05$) from the unamended treatment for either soil (data not shown). Stage III infiltration fluxes for the 7 Mg ha^{-1} rate were 3.6 cm hr^{-1} and 2.5 cm hr^{-1} for Chilicotal and Stellar soils, respectively. Thirty-four Mg ha^{-1} of biosolids (Fig. 2) significantly elevated the stage III infiltration flux above both control and 7 Mg ha^{-1} treatments. Stage III infiltration flux for both soils increased by about 75 % above controls to 5.4 cm hr^{-1} and 3.3 cm hr^{-1} for Chilicotal and Stellar, respectively. Stage I infiltration occurred for a short period and the period of stage II infiltration was extended to about 15 min with 34 dry Mg ha^{-1} of biosolids. Stage III infiltration flux was also significantly increased in both soils with 90 dry Mg ha^{-1} (Fig. 3). With 90 Mg ha^{-1} of biosolids stage I infiltration was almost 5 min and the stage II infiltration was over 15 min. Stellar soil yielded an increase in stage III infiltration flux over the control (1.9 cm hr^{-1} to 5.3 cm hr^{-1}). Chilicotal soil had an even greater increase over the control (3.1 cm hr^{-1} to 9.2 cm hr^{-1}).

Integrating infiltration flux curves over the 30-minute simulation period yielded terminal cumulative infiltration (TCI). Figure 4 illustrates the effect of rate of biosolids on TCI for both soils. The TCI for both soils was increased with application of biosolids. The 34 and 90 Mg ha^{-1} treatments had significantly greater TCI compared to the control and 7 Mg ha^{-1} rates. The 90 Mg ha^{-1} rate on the Chilicotal soil produced an increase of 3.4 cm over the control in the 30 min simulated rainfall. Terminal cumulative infiltration for Chilicotal with 90 Mg ha^{-1} of biosolids was two and a half times greater than the unamended Chilicotal. Recall, that with the 90 Mg ha^{-1} rate, stage I infiltration occurred. Therefore rainfall intensity limited infiltration flux for a period of almost 5 min, and TCI may have been greater had the simulated rainfall been delivered at a greater intensity.

The increase in infiltration flux achieved with the addition of biosolids and subsequent reduction in runoff have acted in part to reduce erosion on experimental plots. The nearly level slope and the armoring effect inherent with lag gravel on the surface reduce the potential for soil erosion by water at this scale. In spite of low antecedent erodability on each soil, a reduction in erosion has been observed where biosolids had been applied, more so on the Stellar soil than on the Chilicotal (Fig. 5).

These data show how different soils responded to the application of varying rates of biosolids. A significant question is, what mechanisms are involved in this response? Where have 3.4 cm of water gone in the 90 dry Mg ha^{-1} biosolids amended Chilicotal treatment? At least some part of this water must be absorbed in the biosolids, but measurements of absorbed water in the biosolids following simulated rainfall is surpassingly low at just 0.4 cm. Once intermolecular water has been removed from the biosolids it is rather difficult to rehydrate this compartment. The absorbed water accounts for just over 10 % of the deficit. Other mechanisms are being explored to explain the remainder; at this time, however, the response is not clearly understood.

It is widely accepted that infiltration is increased and erosion decreased by increasing grass cover. A subset of data was used to explore the effect of vegetation cover and varied rates of biosolids on infiltration and erosion. This subset has balanced combinations of vegetated and bare cover treatments. Only the Stellar soil had both cover conditions.

Comparing vegetation cover condition without the application of biosolids (Fig. 6), infiltration flux responses were quite different. Vegetated treatments had a short period of stage I infiltration. Stage II infiltration in the vegetated treatment was much longer, and terminated at a higher infiltration flux than the bare treatment. The vegetated treatments without biosolids had an infiltration flux curve similar to that found for bare plots amended with 90 Mg ha^{-1} of biosolids. The similarities were limited to the general shape of the curves (i.e. period of each infiltration stage). The infiltration flux for bare treatments with 90 Mg ha^{-1} terminated at 4.7 cm hr^{-1} which was 3.2 cm hr^{-1} less than vegetated treatments without biosolids. In vegetated treatments, the increase in terminal infiltration flux that resulted from the application of 90 dry Mg ha^{-1} of biosolids was not as dramatic as was found in bare treatments (1.9 cm hr^{-1} vs. 3.2 cm hr^{-1}).

Adding 34 and 90 Mg ha^{-1} of biosolids to bare and vegetated treatments yielded significant increases in TCI over the control and 7 Mg ha^{-1} rates (Fig. 7). In bare treatments, 90 Mg ha^{-1} of biosolids yielded a significant increase in TCI over the 34 Mg ha^{-1} rate. The increase in TCI gained over controls with the addition of 90 Mg ha^{-1} of biosolids was lower for vegetated treatments than for bare treatments (1 cm vs. 2.3 cm).

Erosion on bare treatments was greater than vegetated treatments at all biosolids application rates (Fig. 8). In both ground cover treatments, soil erosion decreased as biosolids rate increased. The biosolids or vegetation cover intercept falling raindrops and absorb its energy so that detachment of soil particles is reduced. Additionally, the reduction in runoff acts to reduce erosion.

Soil Water Quality

The day after the tubes were brought to near field capacity, biosolids were added and the Stellar soil was leached for the first time. After applying an average of 2833 ml of water all but one lysimeter had produced approximately 1200 ml of leachate. Water applied was equivalent to a 5.6 cm rainfall event with total infiltration. No relationship was found between column depth and volume of water necessary to produce the leachate. However, with the fresh biosolids there was a significant relationship (R^2=0.3973; P=0.0010) between rate of applied biosolids and volume of water needed to produce the leachate. When the biosolids were fresh they had the ability to absorb a great deal of water. The 90 dry Mg ha^{-1} rate required about 1100 ml more water to produce leachate than the control treatment.

Of all 31 constituents analyzed, 17 tests had significant differences among rates of biosolids applied (Table 4). The 17 are Cl, total hardness, Ca hardness, Ca, Mg, Na, Sr, NO_3^{1-}-N, Ba, Mn, K, Cu, Zn, SO_4^{2-}, pH, PO_4^{3-}, and Al. Many of these followed the same general trend where the concentration of the constituent increased with increasing biosolids rate. The means for pH, Mn, Cu, Zn, Al, PO_4^{3-}, and NO_3^{1-}-N did not express a consistent trend. The soil water pH ranged from 7.8 to 8.7 for all treatments, which is near the mean pH for the tap water of 8.1. Only Pb and As exceeded EPA guidelines for drinking water at the 90 Mg ha^{-1} rate; and Pb was also slightly above the criteria for the 34 Mg ha^{-1} rate. Tested anions were all within guidelines at all rates.

Approximately 110 days after the first leaching, the Stellar soil was again leached. The biosolids had in the meantime dried considerably (much of the intermolecular water was removed).

Three-hundred ml of water was added between leaching dates to partially offset evaporation losses.

After adding an average of 3360 ml (equivalent to total infiltration of 6.6 cm of rain) of water, approximately 1200 ml of leachate was collected. There was no significant relationship between either column depth or biosolids rate and volume of water necessary to produce leachate.

For this second leaching only 5 constituents had values that were significantly different among rates (Table 5). Of these, 4 showed clear trends. Two of the anions, NO_3^{1-}-N and SO_4^{2-} generally increased linearly with increased biosolids rate. Another anion, PO_4^{3-}, as well as soil water pH had quadratic components. The pH decreased sharply with the addition of small rates of biosolids without much further reduction at the higher rates. The PO_4^{3-} increased sharply with small additions of biosolids without much further increase at the higher biosolids application rates. The range for pH of the soil water in the second leaching was slightly lower than that of the first leaching. Tested anions were all within drinking water guidelines. Lead was slightly higher than drinking water guidelines for the 90 Mg ha^{-1} rate. Also, As was slightly above guidelines for the 34 Mg ha^{-1} rate and Cr was slightly above at the 18 Mg ha^{-1} rate; interestingly, both As and Cr were above guidelines on the control or 0 Mg ha^{-1} treatment as well.

There are a number of properties of the Stellar soil which help to explain these results. The Stellar soil has strongly developed structural units which provide preferential flow paths creating little interaction with soil colloids in ped interiors. Flow into the ped interior is rather slow in comparison to that along ped faces. The column length of the Stellar soil was relatively short which decreased the solutions residence time. The non-careous surface is less effective in buffering the pH and binding metals. In the second leaching, the quality of the water may have been better as a result of the biosolids having dried out. The water in biosolids may be partitioned into three theoretical compartments; free water (largely removed in the dewatering process), capillary water (held in tension by adhesive forces between water and biosolids particles), and intermolecular water (held in the interior of biosolids organic particles). The hysteretic intermolecular water once exhausted remained dry and water added in the leaching process passed through the bulk fabric quite rapidly, resulting in poor biosolids to solution exchange.

The first leaching of the Armesa soil was done after the lysimeters were prepared by bringing them near field capacity and adding the biosolids. An average of 5720 ml of water (equivalent to total infiltration of 11.2 cm of rainfall) was required to produce 1200 ml of leachate. There was a significant relationship between column depth and volume of water used to produce leachate. Rate of biosolids was not related to the volume of water need to produce leachate.

Of the 31 constituents tested only two elements Zn and Ba had significant differences among rates of biosolids applied. Both elements generally increased with increased biosolids application rate. The range in pH for soil water from this leaching was 8.1 to 8.8, slightly higher than for the Stellar soil. All metal and anions were within drinking water standards.

Approximately 60 days later, the Armesa soil was leached a second time. An average of 5340 ml of water was required to produce about 1200 ml of leachate. This amount of water is equal to total infiltration of a 10.4 cm rain. Column depth was not related to the volume of water needed to produce leachate. However, with this soil in the second leaching there was a significant relationship (R^2=0.5416; P=0.0001) between rate of biosolids applied and volume of water needed to produce leachate. The highest rate of biosolids required the least amount of water to produce leachate. The relationship between biosolids rate and volume of water need to produce leachate found here indicates that biosolids act as a mulch, reducing evaporation loss from the lysimeters. This is opposite from the fresh biosolids on the Stellar soil where the high biosolids rates actually absorbed more water. Of the 31 constituents tested, only one, Mg, had significant differences among rates of application. The Mg concentration in the soil water increased with increased biosolids rate. The level of Mg in soil water was 26 mg l^{-1} for the control and 39.4 mg l^{-1} for the 90 Mg ha^{-1} rate. The pH values for the soil water ranged from 8.5 to 9.0. All metals tested, but one exception (18 Mg ha^{-1} rate for Pb) were well within drinking water guidelines. All 4 rates of biosolids as well as the control had high nitrate concentrations. Sulfate, although high in comparison with the other leachings was within drinking water standards. Chloride concentrations were also within drinking water standards.

The Armesa soil had a much thicker column, and the soil structure was weaker than that of the Stellar soil. Also, the Armesa soil was calcareous to the surface. These soil properties enhance the Armesa soil's ability to remove dissolved constituents from the soil solution. The Armesa soil exhibited less preferential flow, which allowed more intimate contact with soil colloids. The Armesa soil water had a longer residence time in the soil.

Conclusion

The application of biosolids, especially at 34 and 90 dry Mg ha^{-1} rates, increased infiltration flux and cumulative infiltration while decreasing erosion. The degree to which biosolids affect infiltration and erosion depend on the site conditions (i.e. soil type and vegetation cover). Infiltration increased more per Mg ha^{-1} of biosolids added on the Chilicotal soil than on the Stellar soil, regardless of vegetation cover. In contrast, infiltration increased the least per Mg ha^{-1} of biosolids added to the vegetated Stellar soil. Although the rate response attributed to biosolids was less on the vegetated Stellar soil, both infiltration flux and cumulative infiltration for the untreated vegetated Stellar soil were greater than treated bare treatment on either the Stellar or Chilicotal soil, regardless of biosolids rate. Erosion was reduced more per Mg ha^{-1} of biosolids added on the bare Stellar soil than on either the vegetated Stellar or bare Chilicotal soil.

Soil water quality for both Armesa and Stellar soils were within EPA drinking water standards with 7 Mg ha^{-1} of biosolids. The Armesa soil had fewer constituents with significant differences in leachate quality among the rates of applied biosolids. The Armesa soil, because it has a longer flow path, less prominent structural units, and is calcareous to the surface, does a more thorough job of removing dissolved constituents than the Stellar soil. Under normally expected precipitation patterns, leachate would not be expected to reach the ground water table on the study site, which is in excess of 150 meters. Total infiltration of 8.2 cm of precipitation over 24 hours (slightly more than the 10 yr return period, 24 hr storm) was required to move water through just 75 cm of the coarser Armesa soil.

These results clearly show that topical application of biosolids has important hydrologic consequences in desert grasslands that depend on kind of soil being treated and rate of biosolids applied.

Acknowledgment

We would like to thank MERCO, A Joint Venture for allowing us to conduct this research on their property and for providing much of the funding which supported this research effort.

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Literature cited

Aguilar, R., and S.R. Loftin. 1992. Sewage sludge application in semiarid grasslands: Effects on runoff and surface water quality. p. 101-111. *In* Proc. of the 36th Annual New Mexico Water Conf., Las Cruces, NM. 7-8 Nov. 1991. WRRI Rep. 265. New Mexico Water Resource Res. Inst., Las Cruces.

Bubenzer, G.D. 1979. Inventory of rainfall simulators. Proc. Rainfall Simulator Workshop. Tucson, Ariz. p. 120-130.

Elkins, N.Z. 1983. Potential mediation by desert subterranean termites in infiltration, runoff and erosion soil loss on a desert watershed. Ph. D. diss. New Mexico State Univ., Las Cruces.

Fresquez, P.R., R.E. Francis, and G. L. Dennis. 1990. Effects of sewage sludge on soil and plant quality in a degraded semiarid broom snakeweed / blue grama plant community. J. Environ. Qual. 19:324-329.

Harris-Pierce, R.L., E.F. Redente, and K.A. Barbarick. 1995. Sewage sludge application effects on runoff water quality in a semiarid grassland. J. Environ. Qual. 24:112-115.

Soon, Y. K., T.E. Bates, and J. R. Moyer. 1980. Land application of chemically treated sewage sludge: III. Effects on soil and plant heavy metal content. J. Environ. Qual. 3:497-504.

Spaeth, K.E. 1990. Hydrologic and ecological assessment of a discrete range site on the southern High Plains. Ph. D. diss. Texas Tech Univ., Lubbock.

Wilcox, B.P., M.K. Wood, T.T. Tromble, and T.J. Ward. 1986. A hand-protable single nozzle rainfall simulator designed for use on steep slopes. J. Range Manage. 39:375-377.

Tables and Figures

Table 2.

Constituent	unit	Mean
Total Kjeldahl Nitrogen	g kg^{-1}	34.7
Iron	g kg^{-1}	20.5
Calcium	g kg^{-1}	19.3
Phosphorus	g kg^{-1}	15.8
Aluminum	g kg^{-1}	9.00
Magnesium	g kg^{-1}	6.64
Manganese	g kg^{-1}	1.25
Potassium	g kg^{-1}	1.02
Zinc	mg kg^{-1}	887
Copper	mg kg^{-1}	872
Lead	mg kg^{-1}	236
Nickel	mg kg^{-1}	46.8
Boron	mg kg^{-1}	40.4
Cadmium	mg kg^{-1}	9.04

Table 1.

Constituent	unit	Mean
Total Kjeldahl Nitrogen	g kg^{-1}	40.4
Iron	g kg^{-1}	25.8
Phosphorus	g kg^{-1}	21.0
Calcium	g kg^{-1}	20.6
Aluminum	g kg^{-1}	9.76
Magnesium	g kg^{-1}	6.57
Potassium	g kg^{-1}	3.08
Zinc	g kg^{-1}	1.11
Copper	g kg^{-1}	1.01
Manganese	mg kg^{-1}	605
Lead	mg kg^{-1}	249
Boron	mg kg^{-1}	43.0
Nickel	mg kg^{-1}	36.0
Cadmium	mg kg^{-1}	23.8

Table 3.

Constituent	unit	Mean
Total Kjeldahl Nitrogen	g kg^{-1}	37.4
Iron	g kg^{-1}	28.54
Calcium	g kg^{-1}	23.2
Phosphorus	g kg^{-1}	20.1
Magnesium	g kg^{-1}	9.02
Aluminum	g kg^{-1}	7.84
Manganese	g kg^{-1}	1.25
Zinc	g kg^{-1}	1.19
Potassium	g kg^{-1}	1.02
Copper	mg kg^{-1}	430
Boron	mg kg^{-1}	37.2
Nickel	mg kg^{-1}	12.1
Lead	mg kg^{-1}	3.86
Cadmium	mg kg^{-1}	3.39

Table 4.

Constituent	unit	0 Mg ha^{-1}	7 Mg ha^{-1}	18 Mg ha^{-1}	34 Mg ha^{-1}	90 Mg ha^{-1}
Cl	mg l^{-1}	27.9 d	34.6 cd	40.3 bc	44.6 b	64.1 a
T-Hardness*	mg l^{-1}	92.0 c	95.5 c	93.9 c	125.6 b	213.5 a
Ca-Hardness	mg l^{-1}	66.2 c	55.5 c	64.4 c	91.6 b	157.6 a
NO_3^{1-}-N	mg l^{-1}	1.94 a	1.00 b	0.52 bc	0.38 c	0.56 b
SO_4^{2-}	mg l^{-1}	66.0 c	73.8 b	76.0 b	88.0 b	123.0 a
pH	---	7.99 c	8.29 ab	8.47 a	8.19 bc	8.20 bc
PO_4^{3-}	mg l^{-1}	0.77 c	1.30 ab	1.35 a	1.40 a	0.84 bc
Ca	mg l^{-1}	27.8 c	26.5 c	27.2 c	38.0 b	59.4 a
Mg	mg l^{-1}	8.38 c	8.40 c	9.22 c	11.40 b	19.70 a
Na	mg l^{-1}	155 c	175 b	200 b	186 b	272 a
Sr	mg l^{-1}	0.30 c	0.31 c	0.32 c	0.44 b	0.78 a
Ba	mg l^{-1}	0.098 b	0.095 b	0.094 b	0.100 b	0.200 a
Mn	mg l^{-1}	0.314 a	0.172 a	0.200 a	0.192 a	0.050 b
K	mg l^{-1}	5 c	5 c	7 bc	11 ab	15 a
Cu	mg l^{-1}	0.009 c	0.022 ab	0.032 ab	0.040 a	0.018 bc
Zn	mg l^{-1}	0.058 c	0.143 a	0.118 ab	0.132 ab	0.078 bc
Al	mg l^{-1}	4.00 a	2.25 abc	3.40 ab	1.80 bc	0.92 c

* T-Hardness is Total-Hardness

Table 5.

Constituent	unit	0 Mg ha^{-1}	7 Mg ha^{-1}	18 Mg ha^{-1}	34 Mg ha^{-1}	90 Mg ha^{-1}
SO_4^{2-}	mg l^{-1}	95.0 d	120.4 c	125.4 bc	137.2 ab	148.6 a
NO_3^{1-}-N	mg l^{-1}	1.42 b	4.39 b	3.96 b	3.63 b	19.17 a
pH	---	8.23 a	8.04 b	7.97 b	7.95 b	7.97 b
PO_4^{3-}	mg l^{-1}	0.304 c	0.880 b	1.074 b	1.284 ab	1.580 a
V	mg l^{-1}	0.056 a	0.042 b	0.052 ab	0.040 b	0.040 b

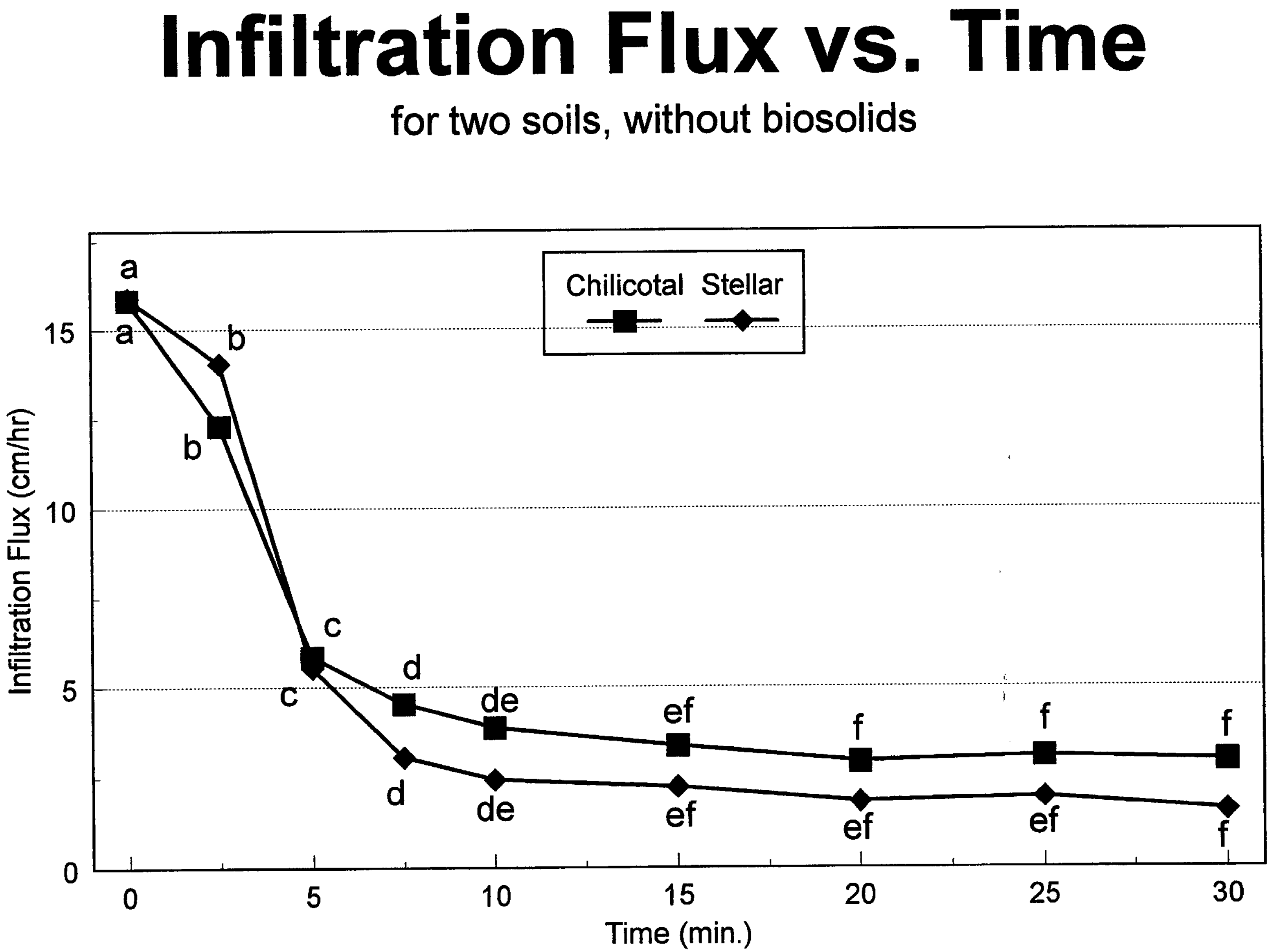

Infiltration Flux vs. Time
for two soils, without biosolids
Chilicotal
Stellar
Infiltration Flux (cm/hr)
Time (min.)
0
5
10
15
20
25
30
0
5
10
15
a
a
b
b
c
c
d
d
de
de
ef
ef
f
ef
f
ef
f
f

FIG. 1

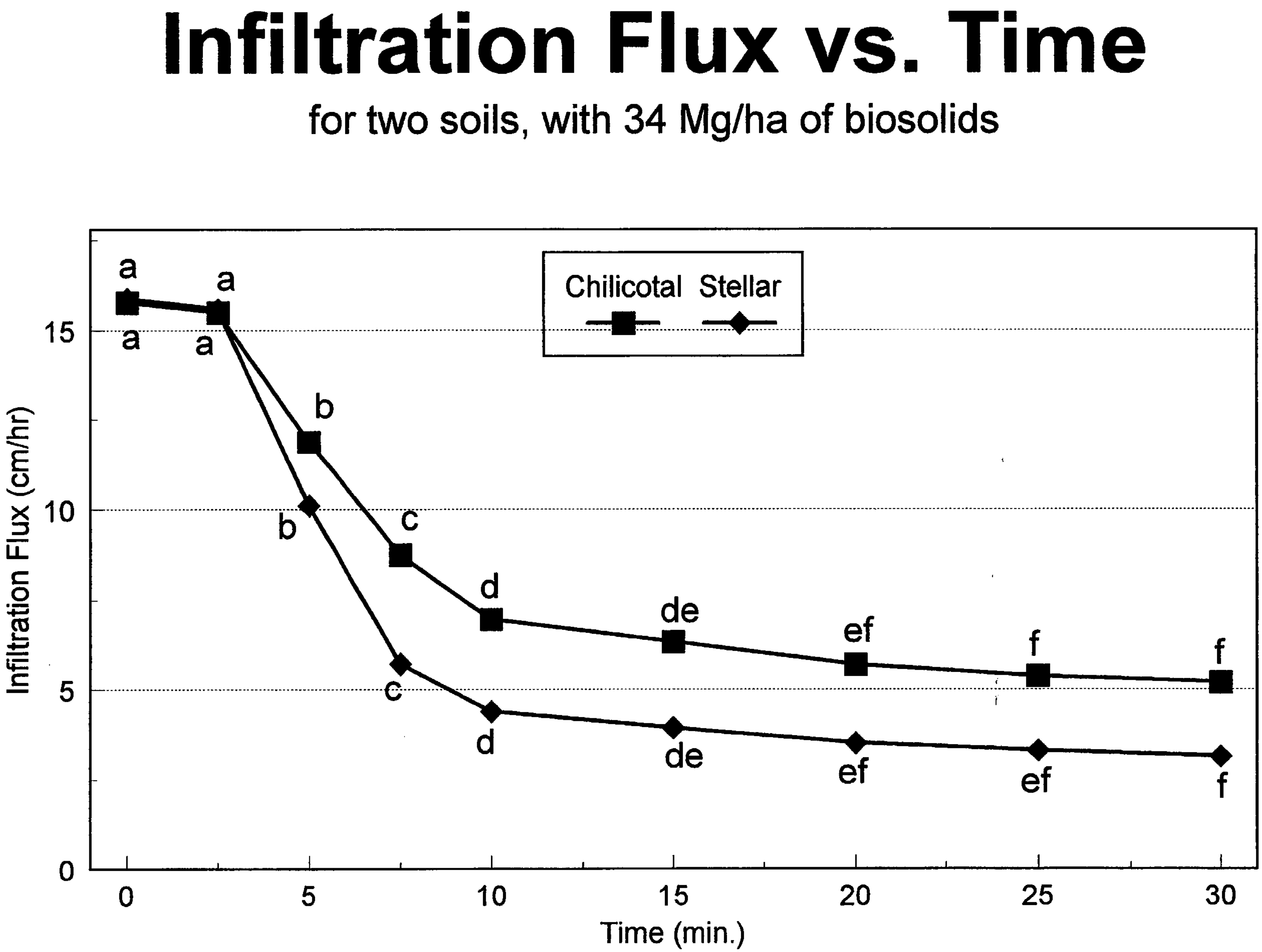

Infiltration Flux vs. Time
for two soils, with 34 Mg/ha of biosolids
Chilicotal
Stellar
Infiltration Flux (cm/hr)
Time (min.)
0
5
10
15
0
5
10
15
20
25
30
a
a
a
a
b
b
c
c
d
d
de
de
ef
ef
f
ef
f
f

FIG. 2

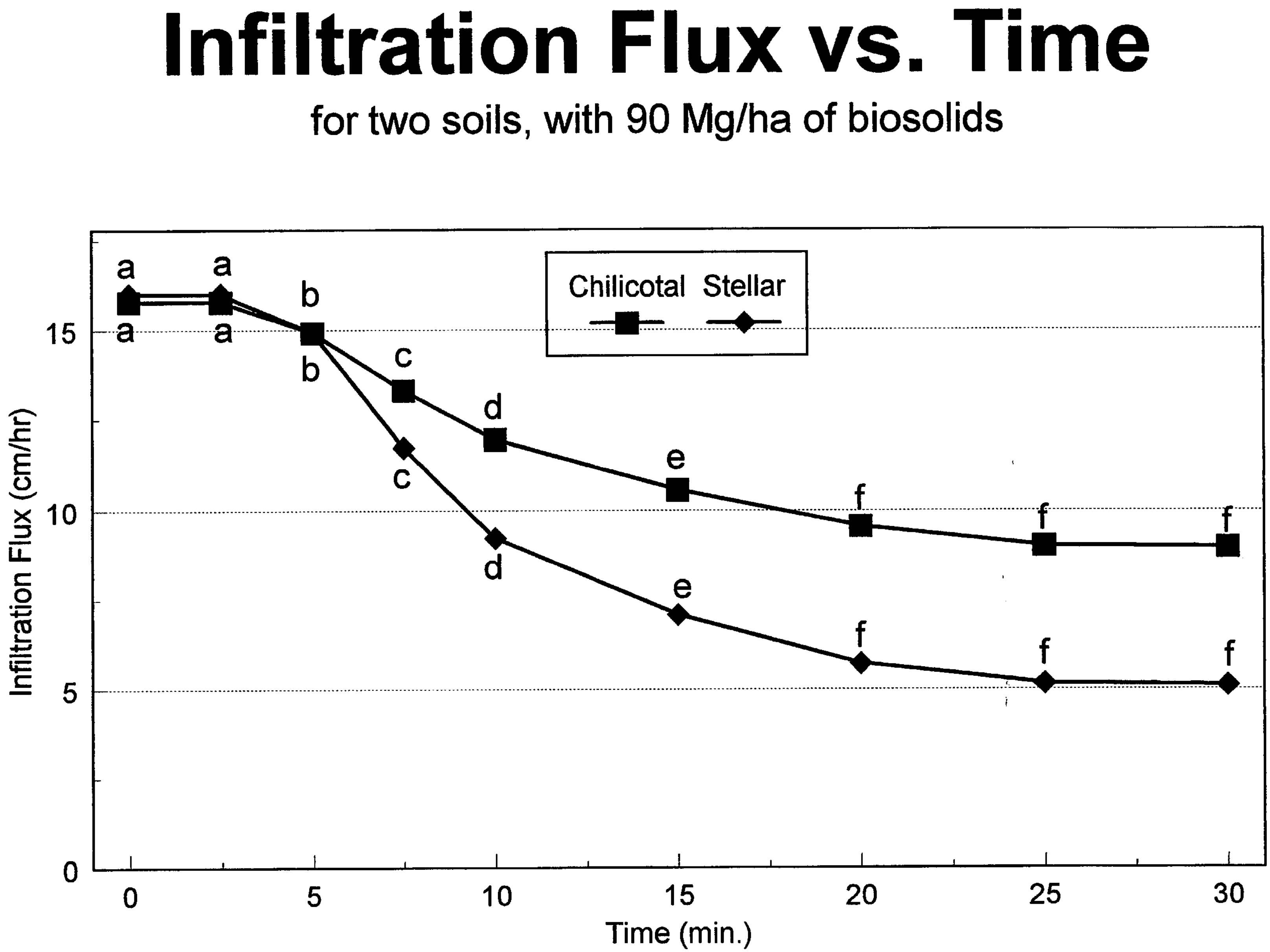

Infiltration Flux vs. Time
for two soils, with 90 Mg/ha of biosolids
Chilicotal
Stellar
Infiltration Flux (cm/hr)
Time (min.)
0
5
10
15
0
5
10
15
20
25
30
a
a
a
a
b
b
c
c
d
d
e
e
f
f
f
f
f
f

FIG. 3

Terminal Cumulative Infiltration vs. Rate

for two soils

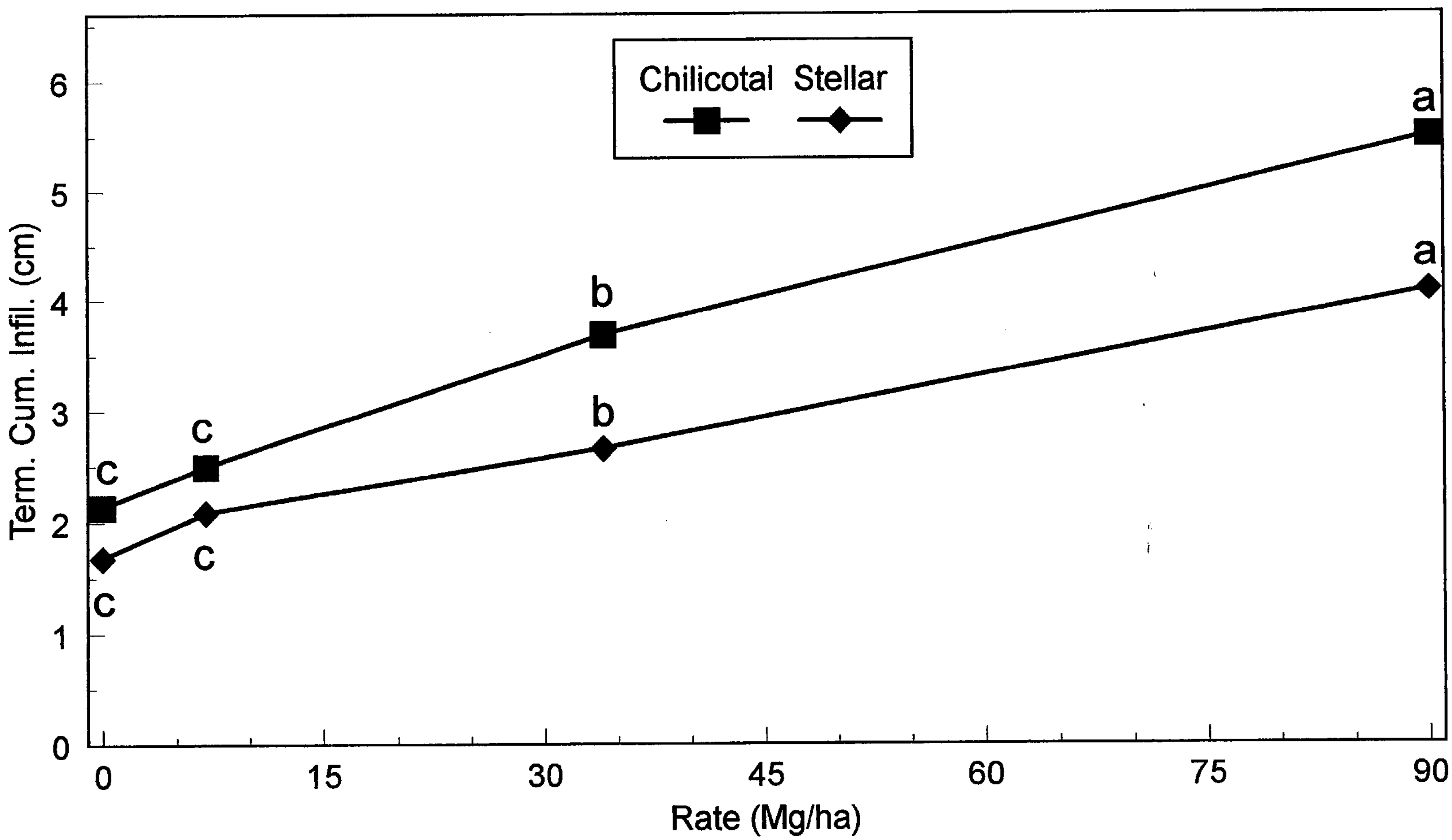


FIG. 4

Terminal Cumulative Erosion vs. Rate

for two soils

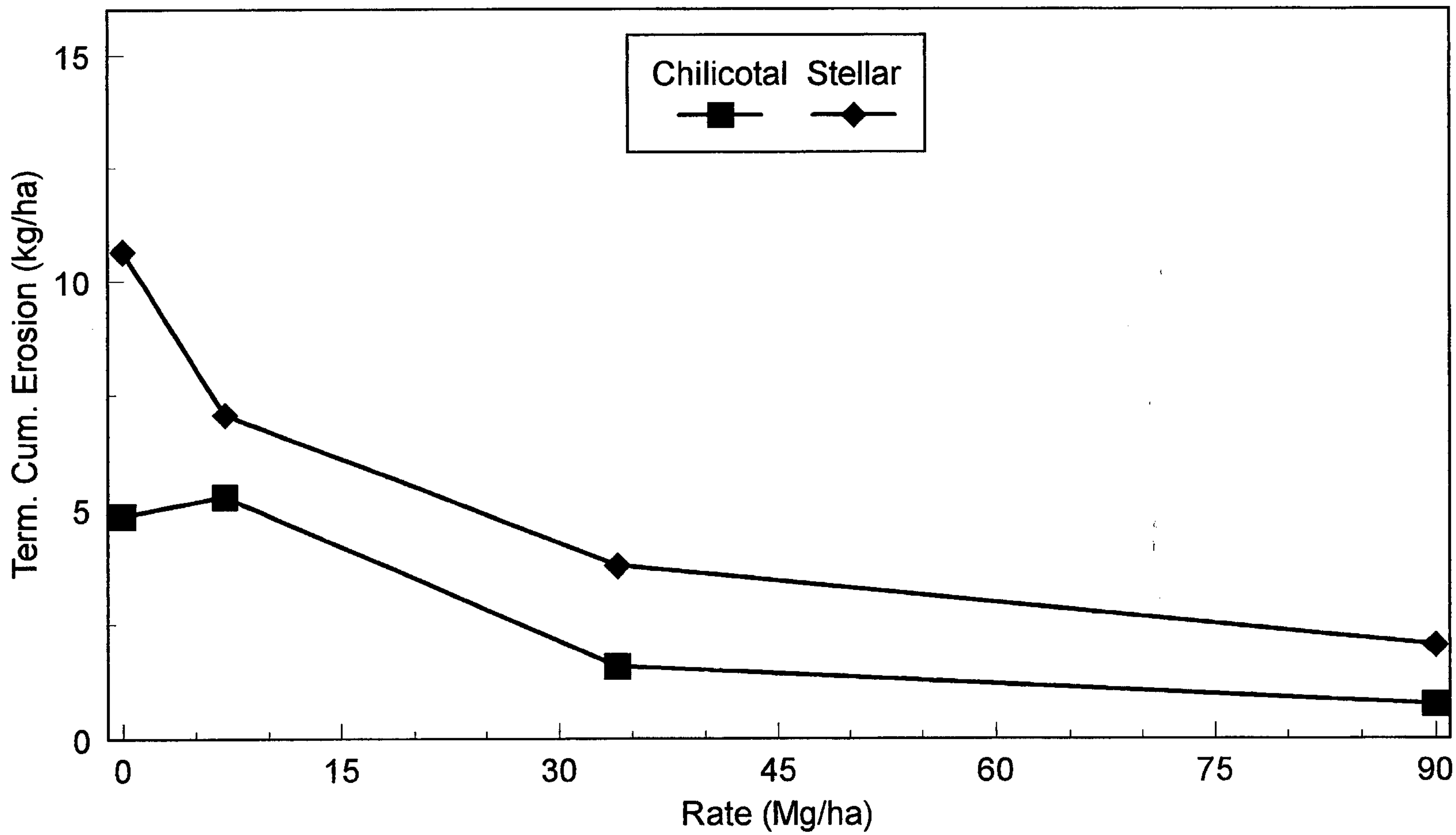


FIG. 5

Infiltration Flux vs. Time

for two cover conditions, without biosolids

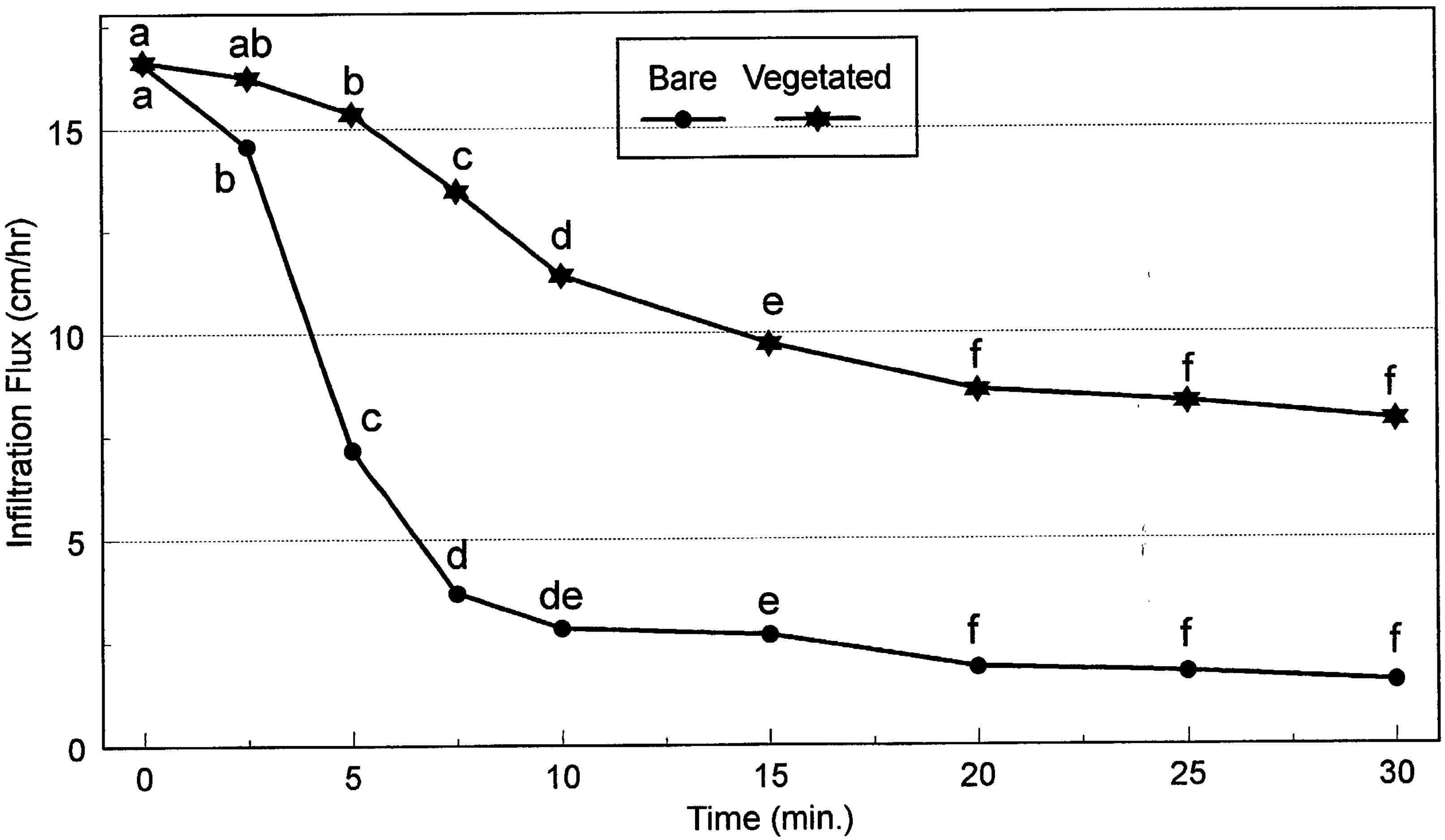


FIG. 6

Fig.7

Terminal Cumulative Infiltration vs. Rate

for two cover conditions

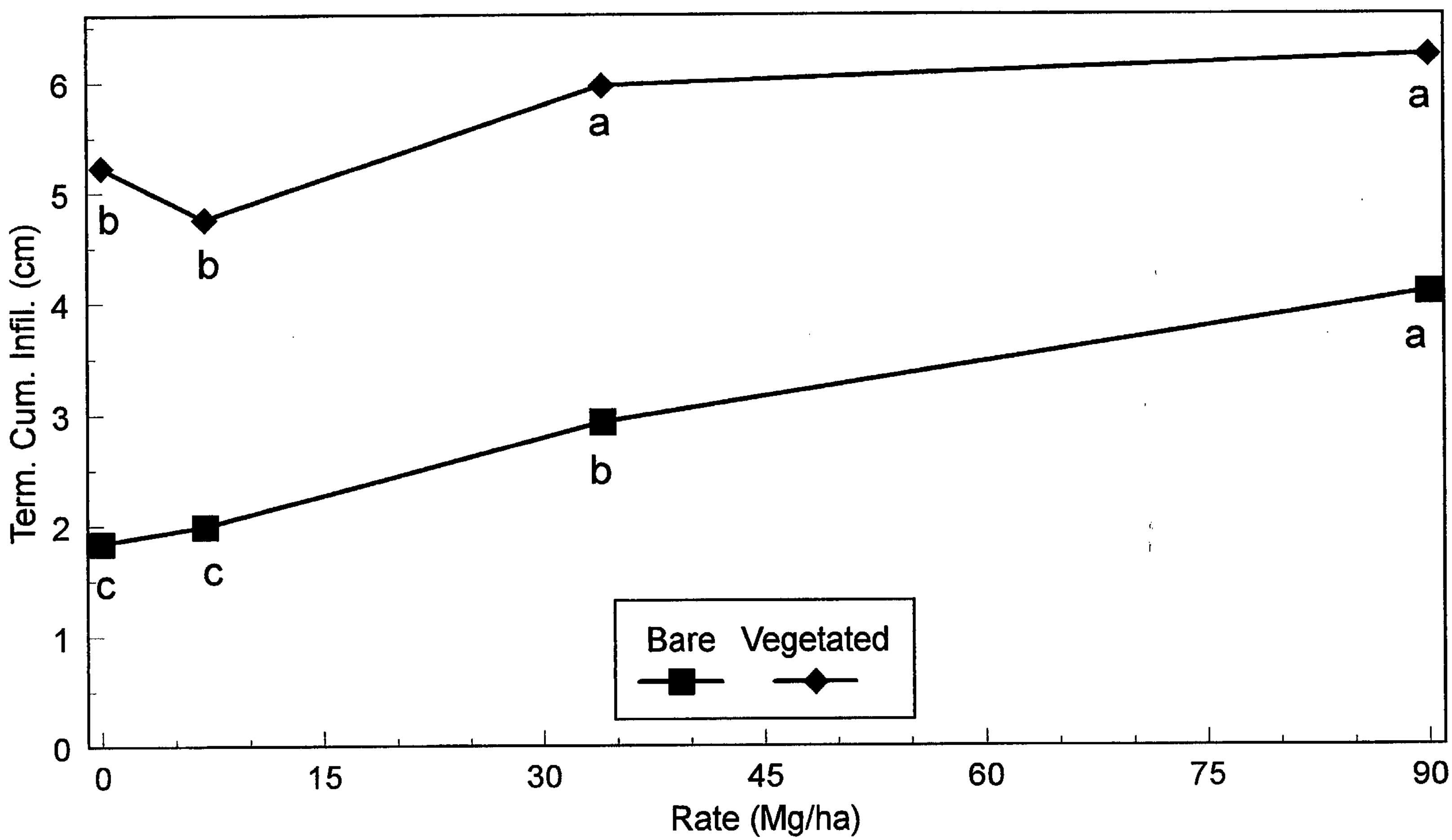
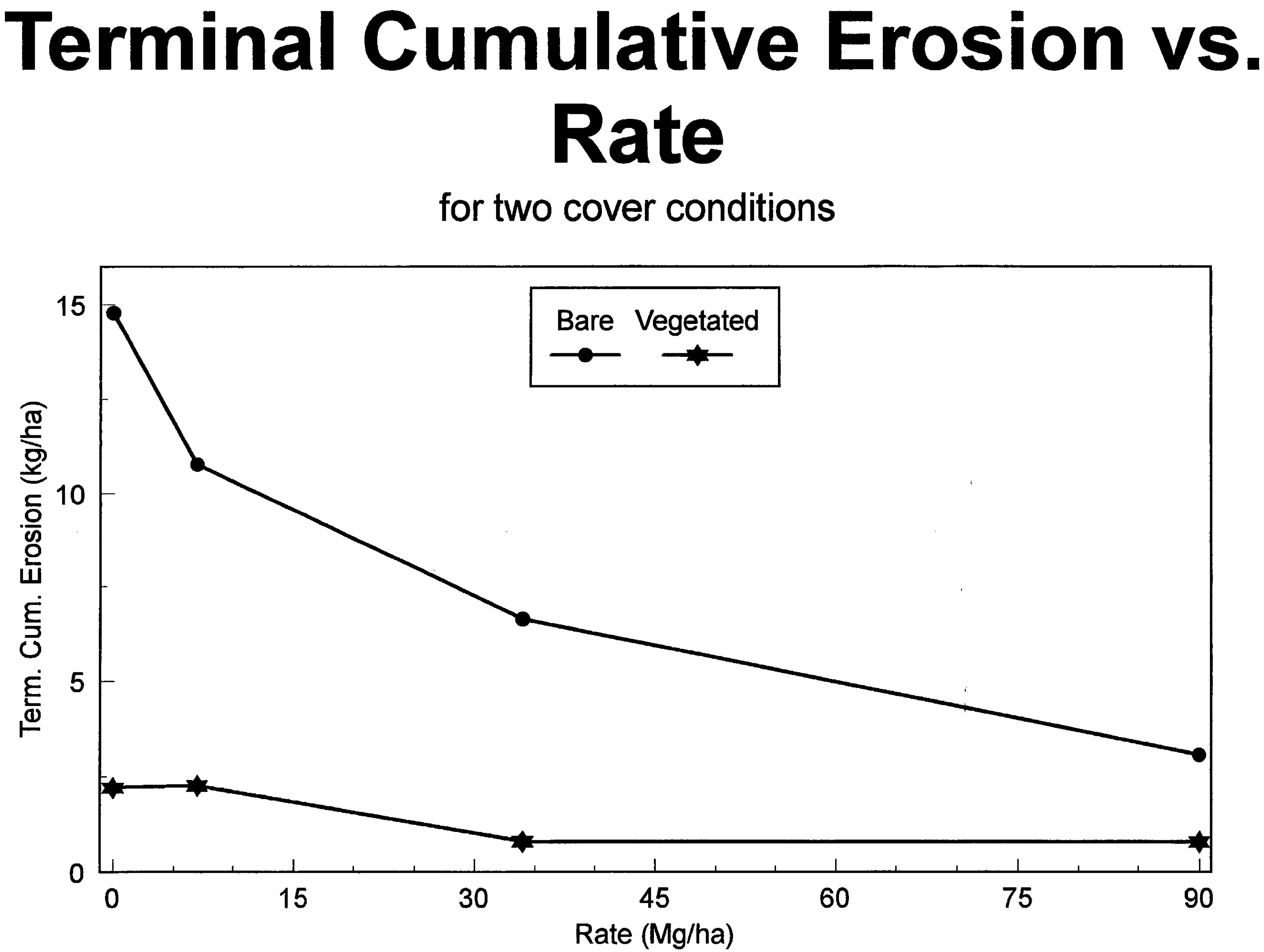


FIG. 7

Fig.8



Terminal Cumulative Erosion vs. Rate
for two cover conditions
Bare
Vegetated
Term. Cum. Erosion (kg/ha)
15
10
5
0
0
15
30
45
60
75
90
Rate (Mg/ha)

FIG. 8

Session V

Monitoring Airborne Microbial Pathogens and Indicators

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Abstract

Land disposal of sewage sludges is a common practice throughout the United States and in several regions of the world. In areas where sludge application is close to human habitation, there is potential for exposure to airborne microbial pathogens. A four month study was conducted in Texas to determine the impacts of the sludge application on the microbial air quality at different locations around the application site. None of the sites showed the presence of specific microbial pathogens though, indicator organisms such as Clostridium spp., and hydrogen sulfide producing organisms were detected at one location (hopper loading) where there was significant physical agitation of the sludge material. Preliminary DNA fingerprinting analysis however, does not indicate that the airborne clostridia originated from the sludge material. The overall results indicate that the current rangeland sludge application program does not adversely impact the microbial air quality at the rangeland-population interface.

Introduction

Municipal sewage sludges or, what are currently referred to as "biosolids" are routinely utilized on agricultural lands in various parts of the world. Presently, in the United States, as much as 33% of the domestic sewage sludge production is reportedly being applied on agricultural lands. With the ban on sludge dumping in the oceans and the increasing restrictions on land fills, disposal onto land surfaces (for beneficial reuse) becomes almost the only alternative, and is expected to increase in the coming years. As part of an interim disposal practice, New York City currently transports dewatered anaerobically digested municipal sewage sludge to a Far West Texas location where it is mechanically applied as a semi solid "cake" form on arid rangelands at an annual rate of 3 dry tons per acre.

One of the hazards associated with sewage sludges are pathogenic microorganisms. Though there have been many studies on the survival of pathogenic microbial populations in agricultural lands and aquatic systems exposed to sewage sludges, there is little information on the possible airborne transmission of microbial pathogens during land application of sludge material. Most of the currently available airborne microbial data are based on aerosol studies conducted near waste water treatment plants and at effluent spray irrigation sites. To estimate the potential public health risks associated with land application of sewage sludges, it is critical that information be available on the extent of airborne microbial pathogens.The sludge application program in Far West Texas is intended to restore depleted rangelands for cattle grazing while at the same time, serving as a means of safe disposal. Since this rangeland is located in an arid wind swept region, the potential for airborne microbial pathogens during sludge application could be considered high. Primarily, there are concerns of airborne transmission of microbial pathogens to the neighboring town of Sierra Blanca (pop: 800) located 4 miles to the closest sludge application site.

A four month study was conducted during August-November, 1994 to determine the levels of airborne microbial pathogens and indicator species during sludge application. The primary objectives of the study were to characterize the airborne microbial populations in the vicinity of sludge application, and also determine if sludge derived microbial pathogens were detectable at the rangeland-population interface.

MATERIALS AND METHODS

Site Description: The Sierra Blanca Ranch in Far West Texas is in the Chihuahuan Desert and is characterized by limited precipitation, high intensity thunderstorms, high evaporation rates, high summer air temperatures, high wind velocities and low relative humidities. The ranch occupies approximately 180,000 acres of which only 18,000 acres is currently being used for sludge application. The town of Sierra Blanca is 4 miles to the closest sludge application site. Wind velocities on the ranch rarely exceed 20 mph (average around 5 mph) though occasional gusts of upto 40 mph are reported during the Spring season. The primary direction from which the wind blows is bimodal, blowing from southerly and northwest directions. Sludge application takes place primarily on gravelly and variant loamy sites which have tobosagrass, desert holly and mesquite as some of the primary vegetation cover.

Sampling Locations: Five sampling locations were chosen viz., the "Background" (representing sites upwind of the sludge application areas), the "Interface" (representing the interface between the sludge application area and the population center at Sierra Blanca), the "Old Application" locations (representing areas where sludge had been applied 12 months previously), the "Sludge Application" areas (representing locations directly within areas under current sludge application) and the "Hopper Loading" sites where mechanized sludge applicators (hoppers) are loaded with sludge using front end loaders. The "Hopper Loading" sites served as a location where maximal physical disturbance of sludge material and soil occurred and therefore had the maximum potential for airborne microorganisms.

Sampling :The AGI-30 all glass impinger (Ace Glass, Vine, N.J.) was utilized for the sampling and the air samples were concentrated in 20 ml of 0.1% peptone. Peptone was used to aid in resuscitating potentially injured organisms. Impingement rather than impaction was employed for air sampling, because it provided protection against microbial injury during sampling and subsequent transportation to the laboratory. A total of 15 independent sampling trips were made during the course of the study. A minimum of two independent samplings were conducted at each location and five replicate air samples were collected (at each sampling) using sterile impingers. Each impinger was operated for 20 min at a flow rate of 12L/min using at Dwyer VFB series (Dwyer Instruments, MI) flow meter. The samplers were always positioned downwind, five feet above the ground which correspond to the average breathing height of an individual.

Microbiological Analyses: The 20 ml samples were initially concentrated in the laboratory to 5 ml using Centriprep-50 concentrators (Amicon, Beverly, MA) by a two-step centrifugation procedure (1000 *g* for 5 min and 1000 *g* for 1 min). Aliquots of the concentrated sample were analyzed for the following specific microbial pathogens and indicator populations:

- Aerobic heterotrophs
- Fecal coliforms, Fecal streptococci and *Salmonella* spp.
- Hydrogen sulfide producers: [Hydrogen sulfide producing bacteria (*Salmonella*, *Citrobacter*, *Clostridium*, *Proteus*, *Edwardsiella* and some *Klebsiella* species) have been shown to be associated with the presence of fecal material]
- Pathogenic *Clostridium* spp.: Since clostridial spores are present in sewage sludges in numbers several orders of magnitude greater than those in soils, and serve as a relatively better indicator for fecal contamination than coliforms, pathogenic *Clostridium* spp. were also enumerated.
- Coliphages: The soft agar overlay and the colorimetric methods were employed to detect the presence of male specific (F^+) phages and the somatic phages.

Ribotyping of *Clostridium* isolates: To determine whether the clostridial isolates (obtained from the air samples) were genetically related to those from the sewage sludge material, selected clostridial isolates were ribotyped using PCR primers specific to the 16S-23S interspacer region of eubacteria. For ribotyping, the primer pair (*TTGTACACACCGCCCGTC* and *CCTTTCCCTCACGGTACTG*) were employed in PCR amplifications using the following reaction conditions: 95°C-1 min; 55°C-1min; 70°C-3 min. The reaction products were separated on a 2% submarine NuSieve agarose gel.

RESULTS AND DISCUSSION

Aerobic heterotrophic populations: Airborne heterotrophic bacterial populations ranging from a minimum of 64 CFU/m^3 at the "Background" location to a maximum of 3,071,429 CFU/m^3 (at the "Hopper Loading" site) of air were recovered from the various locations (Table 1). There was however a significant variability in numbers of airborne bacterial populations even at a single site on different days. The "Hopper Loading" site exhibited the highest number, with populations averaging around 300,000 CFU/m^3, while the "Background" site had the lowest with a mean of 90,000 CFU/m^3 (Fig 1,Fig.2 & Fig. 3). The increased bacterial levels at the "Hopper Loading" site could be attributed to the result of the physical agitation of the sludge material during sludge loading operations. There was however, no direct correlation between windspeed and bacterial population densities even at this site (r^2=0.05) where there were significantly greater numbers of airborne bacteria than the other sites. Since the sampling (at the "Hopper Loading" site) was performed approximately 15 to 30 m downwind of the sludge loading operations, these values could represent the maximum airborne bacterial loading potential during physical agitation of sludge material (under similar weather conditions) during these operations.

Pathogen/Indicator Microbial Species: None of the sites except the "Hopper Loading" site showed the presence of pathogen indicator bacterial species. The detection limits (as per the sampling and detection protocols employed in the study of this study were 10 MPN/m^3 and 50 CFU/m^3). All the aerosol samples collected during the four sampling dates at the "Hopper Loading" site were negative for *Salmonella* spp., fecal coliforms and fecal streptococci. However, hydrogen sulfide producing bacteria were detectable using the PathoScreen medium during three out of four sampling periods. *Clostridium* spp., (averaging 508 CFU/m^3) were also detectable at the sludge loading site on two sampling occasions (Fig.4).

These airborne heterotrophic bacterial levels are significantly greater than those reported by Brenner et al (1988) who obtained bacterial levels ranging from 86 to 7143 CFU/m^3 at a wastewater spray irrigation facility in Michigan using the XM2 Biological Sampler/Collector. This is rather surprising given that bacterial cells tend to survive longer under relatively humid conditions than dry conditions. One possible reason for the elevated numbers obtained during this study could be the location of the samplers closer to the source and a better isolation medium (R2A) for stressed organisms than the Standards Methods Agar which was employed in the previous study. It is also interesting to note that even though sewage/pathogen indicators such as *Clostridium* spp., and hydrogen sulfide producers were detectable at the Hopper Loading site none of the samples showed the presence of fecal coliforms or fecal streptococci.

Coliphages: None of the samples collected during the study were positive for coliphages (F^+ or somatic) either by the colorimetric assay or the plaque assay.

***Clostridium* Ribotyping**: A total of 13 different *Clostridium* isolates were obtained during the course of this study, some directly from sludge material (at the Hopper Loading site) and four isolates from the air samples collected on 9/15/94 and 10/6/94. No attempt was made to biochemically characterize these isolates. Rather, they were ribotyped to determine whether they were genetically related to those isolates obtained from the sludge samples. Fig.5 is the ribotype pattern obtained of the clostridial isolates obtained on the various days. As can be seen from the ribotype pattern, the isolates from the air samples (Lanes G,H,I,J) show a different pattern from those from the sludge material (Lanes B,C, D, E and F) indicating that they are genetically distinct, suggesting that they do not have a common source or progenitor. It is however possible that we may have missed genetic similarities between the sluge isolates and the air sample isolates since a significantly large number of isolates were not isolated, and consequently not screened.

The data indicates that the current rangeland application program does not significantly influence the microbial air quality in terms of pathogenic or indicator microbial species at the rangeland-population interface at Sierra Blanca. Studies are currently in progress to determine the airborne microbial populations under significantly stronger wind events.

Sampling Location	Sampling Date	Average Wind Speed	Range (CFU/m^3)
Background	10/12/94	2.96	64 - 17,143
	10/19/94	0.48	107 - 500,000
Sierra Blanca Interface	8/12/94	2.56	386 - 1032
	8/30/94	2.12	5714 - 528,571
	10/27/94	2.58	148 - 23,928
Sludge Application	8/23/94	2.3	2354 - 91,429
	9/12/94	2.82	6071 - 1,214,286
	10/4/94	4.42	10,714 - 78,571
	11/7/94	5.14	17,500 - 30,714
Hopper Loading	9/6/94	3.8	4357 - 207,143
	9/15/94	4.82	17,500 - 3,071,42
	10/6/94	4.11	110,714 - 278,571
	11/9/94	3.21	7143 - 196,428
Old Application Site	9/8/94	2.56	750 - 196,429
	9/30/94	2.89	500 - 34,286

Table 1: Heterotrophic bacterial populations at various locations

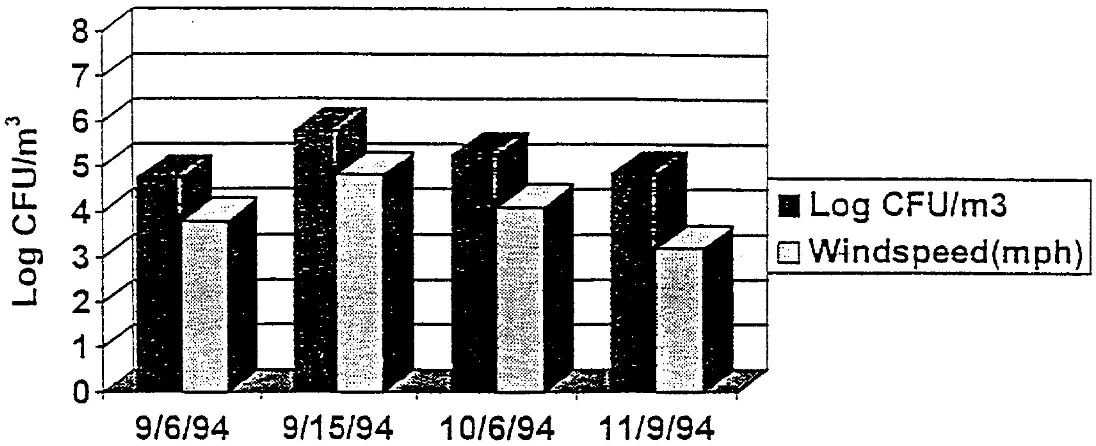


Fig. 1. Heterotrophic bacterial populations at "Hopper Loading"

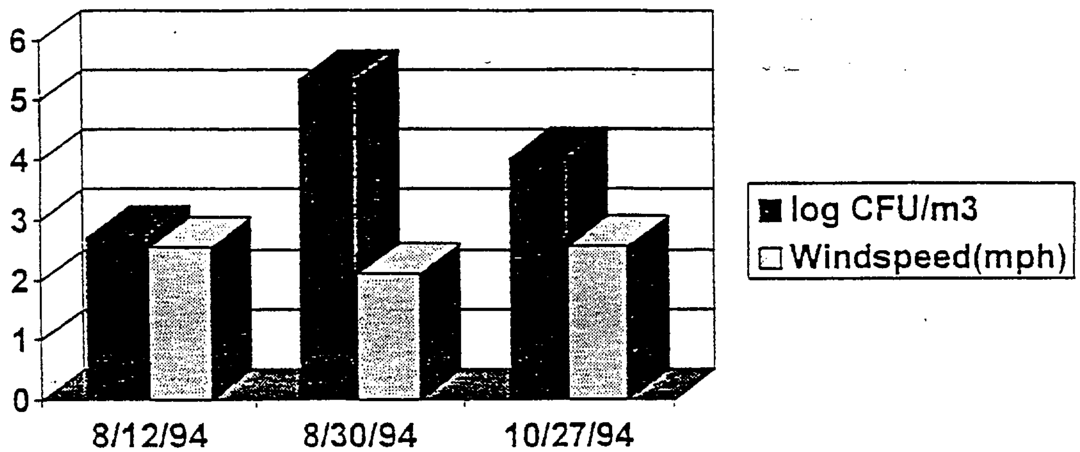


Fig. 2. Heterotrophic bacterialpopulations at rangeland-population interface.

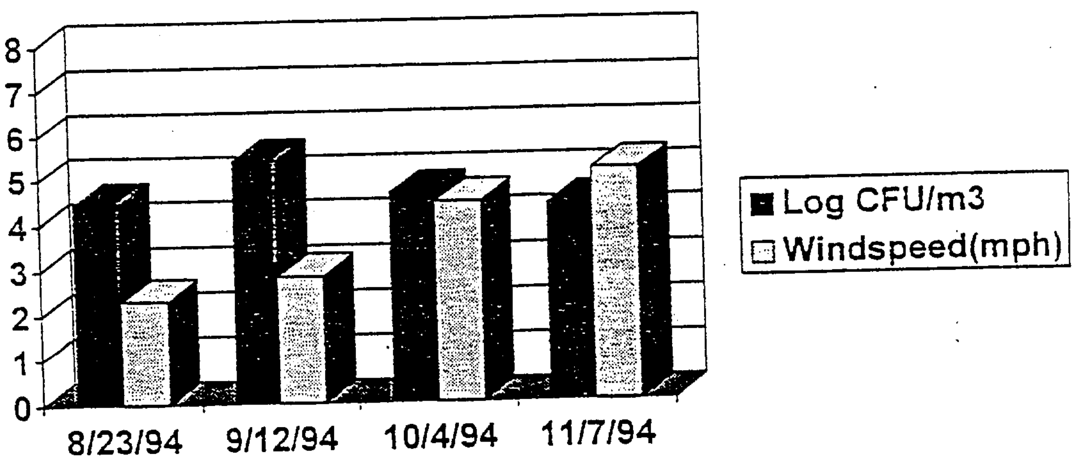


Fig. 3. Heterotrophic bacterial populations during sludge application.

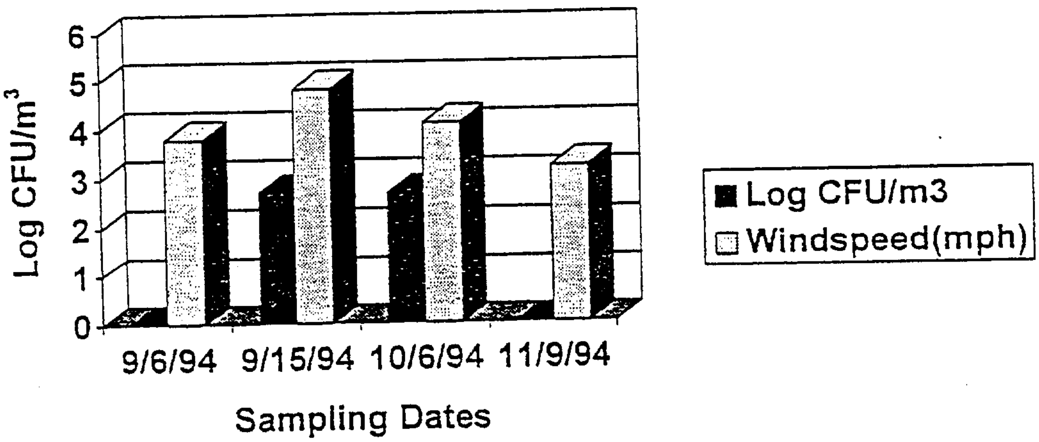


Fig. 4. Airborne Clostridium sp. at "Hopper Loading"

Ribotyping of clostridial isolates using PCR

A: : 100 bp ladder

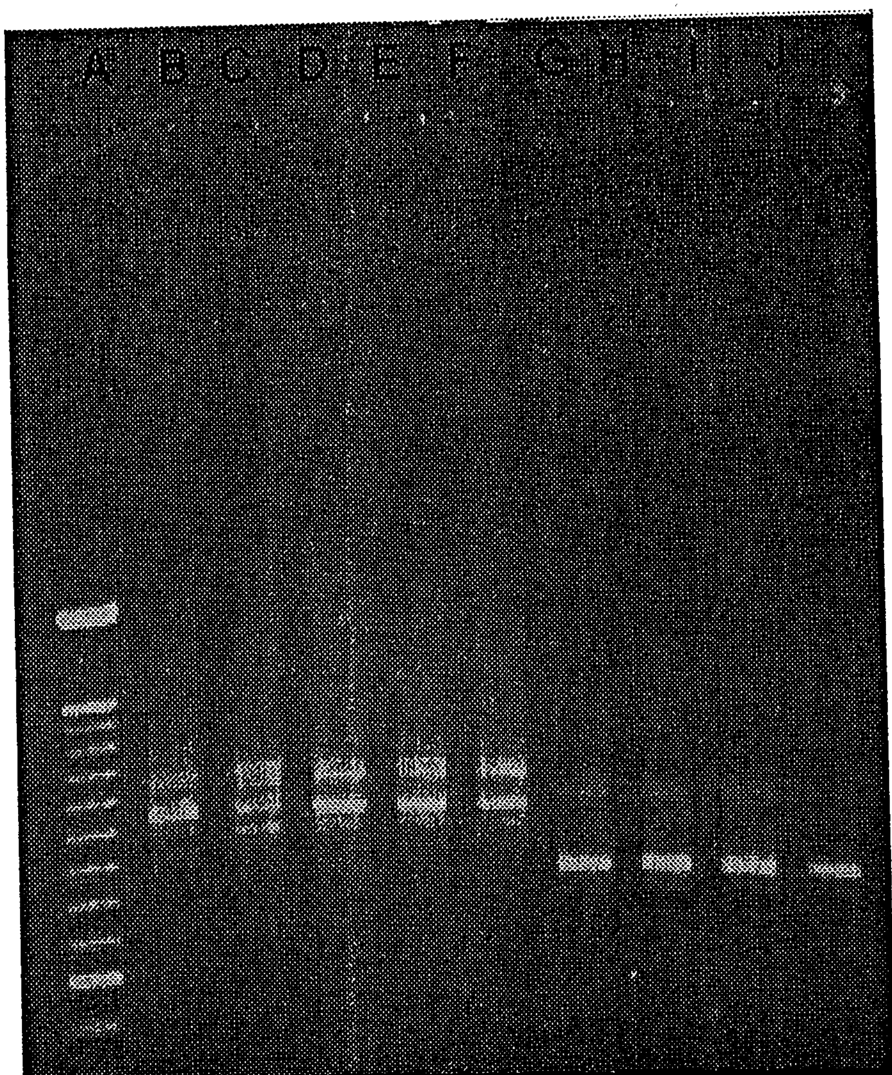
B : SLUDGE (9/15/94)

C : SLUDGE (9/15/94)

D : SLUDGE (9/15/94)

E : SLUDGE (9/15/94)

F : SLUDGE (9/15/94)



G : AIR (9/15/94)

H : AIR (9/15/940

I : AIR (10/6/94)

J : AIR (10/6/94)

References

1. American Public Health Association. 1992. Standard methods for the examination of water and waste water. 18th Ed.

2. ASTM standards on materials and environmental microbiology. 2nd. Ed. American Society for Testing and Materials. Philadelphia, PA.

3. Brenner, K.P., P. Scarpino, and C. S. Clark. 1988. Animal viruses, coliphages and bacteria in aerosols and wastewater at a spray irrigation site. Appl. Environ. Microbiol. 54: 409-415.

4. Hill, R.T., I.T. Knight, M.S. Anikis, and R.R. Colwell. 1993. Benthic distribution of sewage sludge indicated by Clostridium perfringens at a deep-ocean dump site. Appl. Environ. Microbiol. 59: 47-51.

5. Scanlan, J.W., P.J. Psaris, R.D. Kuchewither, M. Nelson, M. Metcalf, R.S. Reinero, T.G. Akero, B.P. Flynn, P.L. Schafer,and J.M. Kelly. 1989. Review of EPA sewage sludge technical regulations. J. Water Poll. Cont. Fed. 61: 1206-1213.

6. Sosebee, R.E. 1994. General climate and weather patterns of the Chihuahuan Desert. Annual Report on the basic and applied research on the beneficial reuse of biosolids on the Sierra Blanca Ranch.

Acknowledgements

The technical assistance of Kenneth Widmer, Jacob Pena and Scot Dowd are gratefully acknowledged. I thank Dr. Bob Carlile for his helpful discussions. This work was supported by a research grant from Merco Joint Venture and funds from the Texas Agricultural Experiment Station of the Texas A&M University System.

Air Quality/Wind Erosion Dynamics of the Sierra Blanca Biosolid Application Area.

B.L. Harris

Abstract

The purpose of this study primarily was to determine the extent to which applied biosolids are moved offsite by wind erosion processes. Secondarily, the study was directed to evaluate the dynamics of wind-moved biosolids at selected sites within the application area. We anticipate this study to continue at least on more year. Triplicate samplers were set at a 1 meter height at 15 locations to collect particulate mater from the air. All samples were processed at the Frank Hernandez Environmental Laboratory at the Texas A&M University Agricultural Research and Extension Center at El Paso, Texas. Very little movement of biosolids was observed. Particulate matter collection was greater near marshalling areas, but little of this material moved offsite. Sites downwind from the biosolids application site had significantly less airborne particulates than upwind sites.

Introduction

Application of biosolids is conducted in the Southwestern U.S., in part, because of climatic and environmental factors that lessen the potential of movement of the material from the application areas, especially by water. However, wind movement has not been previously evaluated. Monitoring wind movement of applied biosolids on the Sierra Blanca ranch can increase the understanding of the industry, regulatory agencies, and the general public about potentials for wind erosion of the biosolid materials.

Methodology

Ten sites were chosen and three replicates of wind erosion samplers (Fryrear, 1986)were placed at 1-m height above the ground surface at each of those sites. Additionally, five sites, each with two replicates, were chosen to place other samplers, each having samplers at 25-, 50-, and 100-cm (1-m) heights; however, this progress report only includes data from the 1-m height samplers. Descriptions of the sites can be found in Table 6. Wind direction was primarily west-southwest; therefore, samplers TAM-2, -3, and -4 were considered upwind collectors of dust generated in the natural environment. TAM-1 was near the town of Sierra Blanca. All other samplers were strategically placed in and around application and marshaling areas. (see map at the end of this report for sampler locations). Particulate samples were collected on 4/6/94, 5/13/94, 8/1/94, 9/6/94, and 10/1/94, based on data compiled regarding windspeed during the collection periods.

Those samples were shipped to the Frank Hernandez Environmental Laboratory at the Texas A&M University Agricultural Research and Extension Center at El Paso, Texas for analyses.

Heavy metal (Ni, V, Cd, Mn, Cr, Cu, and Co) content was determined using acid digestion ($HNO_3/HClO_4$) techniques and IPC determinations. Zn and Pb were not determined because of potential contamination from the galvanized sampler construction materials. There is potential for other metals levels to be affected as well. Additionally, total sample weight, organic matter and total organic carbon were determined (organic matter content was determined visually on samples collected on 4/6/94; organic matter for other samples were analytically determined). Data were analyzed statistically to determine if any significant movement of biosolid materials occurred during the sampling periods. O.M. and T.O.C. samples were pulverized and determined by NCHS-O analyzer.

Results and Discussion

Significantly more particulate matter was collected from the marshaling area from 3/1/94 to 4/6/94, compared to other collection sites (Table 1). This marshaling area was in use during the collection period. Considerable vehicular traffic occurred during this use period and generated local dust. Sample weights of the other collection sites were very minute, ranging from 22 to 62 mg (Table 1). Samples collected from the marshaling area were also significantly higher in V and Mn, compared to collection sites downwind from the application areas (TAM sites 8 and 10; Table 1).

The same trend was evident with regard to sample weight when samples were collected from 4/6/94 to 5/13/94, with the marshaling area having significantly more particulate matter (Table 2). Organic matter, Cd, Mn, Cr, and Co contents did not differ with respect to collection sites during this period. An upwind site revealed a higher V content (TAM site 2), while a higher Ni content was evident in a downwind site (TAM site 10) (Table 2).

Marshaling areas continued to show more particulate matter movement compared to all other sites during the collection period of 7/4/94 to 8/1/94 (Table 3). Significantly more Ni was detected in TTU-1 through TTU-5 samplers during this period; however, significantly lower levels of Ni were detected in TAM-9 and TAM-10 samplers located very near the TTU-1 through TTU-5 samplers, indicating the high Ni content might be associated with component materials of those particular samplers. Mn, Cr, and Co contents did not differ significantly with regard to collection sites during this period (Table 3).

Similar trends with regard to marshaling areas and total sample weight were revealed during the collection period of 8/1/94 to 9/6/94; however, this sampling also revealed that more particulate matter is detected while the marshaling areas are in use, compared to when those areas are not in use (TAM sites 70 and 71; Table 4). Higher amounts of organic matter were found in the marshaling area samples, also.

Samples collected from 9/6/94 to 10/1/94 revealed similar results regarding total sample weights, organic matter content, and Ni content (Table 5).

Overall, particulate matter movement in marshaling areas was 86.2% greater than upwind sites, and 88.7% higher than downwind sites, suggesting that most movement occurs near marshaling areas, as expected. Little, if any, of this material is moving offsite of the area. Additionally, 17.6 % less particulate matter moves in downwind sites than in upwind sites, indicating that wind erosion, and thus biosolid offsite movement, is reduced as the wind moves across the area of biosolid material application. Organic matter (associated with biosolid materials and/or plant materials) in particulate matter was 21.6 % reduced in marshaling areas, compared to upwind sites. Similarly, downwind sites had 54.6 % less organic matter in particulate matter, compared to upwind locations.

Conclusions

1) The overall amounts of both mineral and organic materials being moved onto and off the application areas is almost too small to measure; special micro-sample techniques were required for analyses.

2) Applying biosolids reduces wind erosion by roughening the surface, thereby increasing the boundary layer of air just above the ground surface.

3) Applying biosolids reduces wind erosion by supplying nutrients that promote growth of plant species, which serve as windbreaks. Increasing plant growth in the application areas helps filter particulate matter out of the air as it traverses the site.

4) While intense vehicular traffic in and around marshaling areas does increase dust in their vicinity, those activities do not increase dust movement off the application site.

5) Based on these studies to date, there should be little concern about potentials for the applied biosolids to be moved offsite by wind.

Acknowledgments

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References

D.W. Fryrear (1986) A Field Dust Sampler. Journal of Soil and Water Conservation, 41, 117.

Table 1. Total weights, organic matter, and metal contents of articulate samples collected from 3/1/94 to 4/6/94.

Site	Total Weight (g)	Organic Matter (%)	Ni (mg/kg)	V (mg/kg)	Cd (mg/kg)	Mn (mg/kg)	Cr (mg/kg)	Cu (mg/kg)	Co (mg/kg)
TAM-1	0.040 b	7.5	8.19 c	16.45 b	0.00	215.0 bc	17.75 b	83.75 ab	4.84 ab
TAM-2	0.022 b	2.5	10.83 a-c	31.80 b	0.00	208.0 bc	38.15 ab	66.15 ab	0.00 b
TAM-3	0.062 b	10.0	9.85 bc	24.15 b	0.68	218.5 bc	27.95 ab	41.05 b	1.87 ab
TAM-4	0.038 b	2.5	10.11 bc	23.10 b	2.31	209.5 bc	43.30 ab	121.50 ab	1.88 ab
TAM-5	0.025 b	2.5	18.75 a	26.75 b	2.79	299.0 b	51.00 a	219.50 a	0.00 b
TAM-6	0.070 b	0.0	14.95 a-c	21.75 b	4.32	241.0 bc	24.05 b	145.50 ab	4.83 ab
TAM-70†	2.840 a	0.5	17.60 ab	54.80 a	2.27	405.5 a	36.00 ab	92.50 ab	6.90 a
TAM-8	0.035 b	0.5	9.32 bc	22.30 b	2.59	183.0 c	28.10 ab	31.40 b	2.59 ab
TAM-9	0.049 b	12.5	12.30 a-c	23.80 b	1.73	193.0 c	23.20 b	36.50 b	4.18 ab
TAM-10	0.049 b	0.0	13.16 a-c	24.05 b	2.83	209.5 bc	42.15 ab	112.85 ab	2.40 ab

† This site was marshaling area sampled during use.

Table 2. Total weights, organic matter, and metal contents of articulate samples collected from 4/6/94 to 5/13/94.

Site	Total Weight (g)	Organic Matter (%)	Ni (mg/kg)	V (mg/kg)	Cd (mg/kg)	Mn (mg/kg)	Cr (mg/kg)	Cu (mg/kg)	Co (mg/kg)
TAM-1	0.038 bc	3.33	12.73 b	19.07 bc	2.51	193.00	17.93	36.50 b	4.60
TAM-2	0.020 cd	0.00	14.95 b	39.10 a	1.19	249.00	23.15	124.50 ab	3.78
TAM-3	0.043 b	2.00	11.79 b	23.40 b	2.43	238.00	19.93	47.70 ab	1.67
TAM-4	0.017 cd	12.50	10.80 b	12.35 bc	2.37	245.00	36.25	156.75 a	4.32
TAM-5	0.017 cd	5.00	8.89 b	16.55 bc	4.24	264.00	27.60	69.15 ab	0.00
TAM-6	0.027 b-d	10.33	6.87 b	14.29 bc	0.95	224.67	24.13	61.40 ab	4.92
TAM-71†	0.190 a	0.50	10.10 b	25.70 ab	1.35	214.00	17.47	70.35 ab	3.26
TAM-8	0.028 b-d	0.00	6.92 b	19.43 bc	1.16	212.00	24.63	28.87 b	2.63
TAM-9	0.019 cd	3.33	19.36 b	21.53 b	1.90	230.00	27.47	51.13 ab	1.77
TAM-10	0.013 d	3.33	45.53 a	4.23 c	2.84	247.00	39.67	74.60 ab	0.00

† This site was a marshaling area sampled during a post-use period.

Table 3. Total weights, organic matter, and metal contents of articulate samples collected from 7/4/94 to 8/1/94.

Site	Total Weight (g)	Organic Matter (%)	Ni (mg/kg)	V (mg/kg)	Cd (mg/kg)	Mn (mg/kg)	Cr (mg/kg)	Cu (mg/kg)	Co (mg/kg)	OC (%)
TAM-1	0.040 cd	3.06 a-c	24.10 c	29.33 a	0.86 a-c	202.67	16.60	40.73 c	5.26	1.77 a-c
TAM-2	0.122 b-d	2.68 a-d	16.37 c	17.93 b-d	1.12 ab	258.00	19.20	37.17 c	5.66	1.56 a-d
TAM-3	0.162 b	2.20 b-e	15.43 c	17.63 b-e	0.63 bc	255.67	14.07	41.23 c	5.22	1.28 b-e
TAM-4	0.049 cd	2.81 a-d	17.93 c	21.57 bc	0.12 c	170.67	14.47	50.03 c	2.26	1.63 a-d
TAM-5	0.062 cd	3.39 a	15.10 c	18.50 b-d	0.63 bc	196.30	11.06	30.10 c	4.82	1.97 a
TAM-6	0.063 cd	3.32 ab	22.07 c	22.20 b	1.29 ab	187.67	26.57	91.20 bc	5.68	1.92 ab
TAM-8	0.105 b-d	1.92 c-e	13.13 c	12.80 de	0.75 bc	218.33	13.45	47.30 c	4.86	1.11 c-e
TAM-9	0.107 b-d	2.93 a-d	17.33 c	18.40 b-d	0.58 bc	234.00	17.50	45.57 c	5.89	1.70 a-d
TAM-10	0.032 d	3.43 a	22.33 c	18.87 b-d	0.60 bc	211.33	22.53	38.67 c	10.14	1.99 a
TTU-1	0.137 bc	1.83 de	260.00 b	18.95 b-d	1.23 ab	252.00	11.95	161.50 b	6.04	1.06 de
TTU-2	0.111 b-d	1.84 de	271.00 b	15.10 c-e	1.52 a	233.00	14.50	253.00 a	5.66	1.07 de
TTU-3	0.048 cd	2.99 a-d	617.00 a	10.98 e	0.14 c	250.50	14.95	275.50 a	6.44	1.73 a-d
TTU-4	0.339 a	2.86 a-d	219.50 b	12.95 de	1.22 ab	195.50	13.60	129.50 b	4.58	1.66 a-d
TTU-5	0.167 b	1.12 e	183.50 bc	15.55 b-e	1.19 ab	244.00	10.50	105.85 bc	4.88	0.65 e

Table 4. Total weights, organic matter, and metal contents of articulate samples collected from 8/1/94 to 9/6/94.

Site	Total Weight (g)	Organic Matter (%)	Ni (mg/kg)	V (mg/kg)	Cd (mg/kg)	Mn (mg/kg)	Cr (mg/kg)	Cu (mg/kg)	Co (mg/kg)	OC (%)
TAM-1	0.026 ef	0.00 d	25.13 cd	13.60 b	4.82	240.67 a	20.10 a	51.70 e	5.88 a	0.00 d
TAM-2	0.083 b-e	1.39 bc	12.83 cd	12.37 b	0.51	199.67 b-e	18.10 a-c	31.47 e	5.53 a	0.81 bc
TAM-3	0.067 c-f	2.35 a	13.13 cd	11.87 b	3.13	228.00 a-e	15.17 a-e	38.83 e	3.09 ab	1.36 a
TAM-4	0.027 d-f	0.00 d	13.23 cd	10.31 b	1.66	148.33 f-h	11.80 c-e	36.47 e	0.00 b	0.00 d
TAM-5	0.017 f	0.00 d	16.97 cd	2.35 b	1.02	154.67 f-h	13.43 a-e	29.70 e	0.00 b	0.00 d
TAM-6	0.043 c-f	0.69 cd	13.50 cd	4.88 b	0.48	139.33 gh	19.60 ab	54.37 e	2.54 ab	0.40 cd
TAM-70†	0.347 a	2.09 ab	10.90 d	11.40 b	1.05	234.33 ab	12.83 b-e	36.77 e	4.25 ab	1.21 ab
TAM-71‡	0.137 b	2.23 ab	13.16 cd	11.10 b	1.50	187.00 d-f	15.70 a-e	44.40 e	4.40 ab	1.29 ab
TAM-8	0.033 d-f	0.00 d	12.77 cd	14.67 b	1.29	193.33 b-e	14.83 a-e	48.33 e	5.53 a	0.00 d
TAM-9	0.037 c-f	0.46 cd	14.51 cd	10.04 b	0.00	152.33 f-h	14.17 a-e	32.37 e	4.24 ab	0.27 cd
TAM-10	0.024 ef	0.00 d	14.23 cd	15.63 b	0.00	177.67 d-g	16.27 a-d	29.27 e	5.77 a	0.00 d
TTU-1	0.033 d-f	0.00 d	151.00 b-d	13.05 b	1.08	185.50 d-f	9.86 de	85.80 de	1.89 ab	0.00 d
TTU-2	0.044 c-f	0.70 cd	193.50 bc	8.66 b	2.92	125.50 h	9.41 de	181.00 b	0.00 b	0.41 cd
TTU-3	0.017 f	0.00 d	566.50 a	41.30 a	0.00	215.50 a-d	15.40 a-e	241.50 a	0.00 b	0.00 d
TTU-4	0.086 b-d	0.00 d	312.50 b	14.45 b	0.56	168.50 e-g	12.75 b-e	158.50 bc	4.64 a	0.00 d
TTU-5	0.095 bc	0.80 cd	229.50 b	12.20 b	0.83	212.50 a-d	9.21 e	111.50 cd	4.27 ab	0.47 cd

† This site was a marshaling area in use from 9/6/94 to 9/13/94.

‡ This site was a marshaling area; sample collected pre-use from 8/1/94 to 9/6/94.

Table 5. Total weights, organic matter, and metal contents of articulate samples collected from 9/6/94 to 10/1/94.

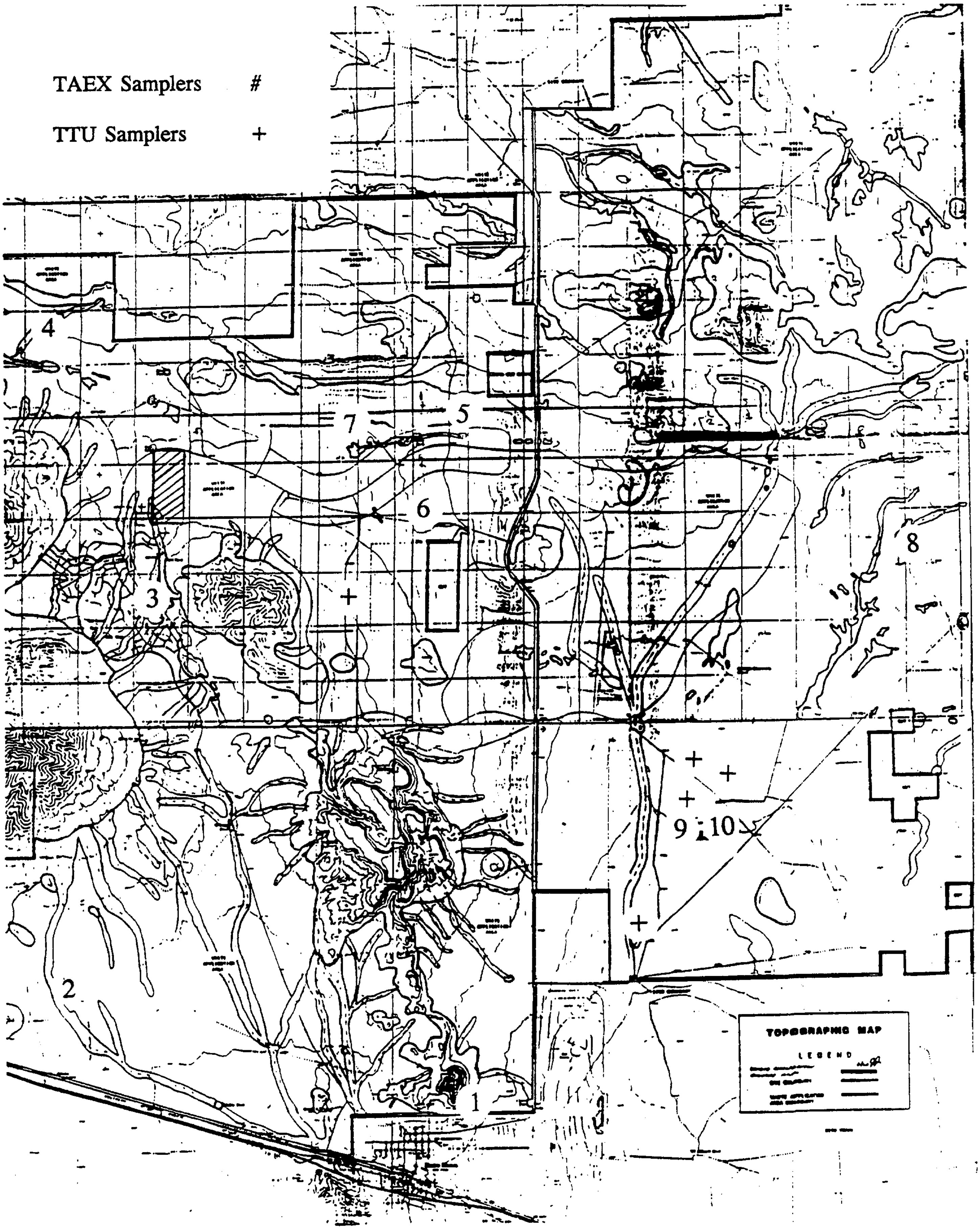
Site	Total Weight (g)	Organic Matter (%)	Ni (mg/kg)	V (mg/kg)	Cd (mg/kg)	Mn (mg/kg)	Cr (mg/kg)	Cu (mg/kg)	Co (mg/kg)	OC (%)
TAM-1	0.022 d-f	0.00 c	26.27 d	22.13 a	0.98	246.33 a-c	25.30 a	57.17 c	9.92 a	0.00 c
TAM-2	0.031 c-f	0.00 c	20.63 d	18.63 ab	0.00	186.00 b-e	15.37 b-e	34.47 c	5.00 ab	0.00 c
TAM-3	0.067 b	3.35 a	11.00 d	15.13 a-c	0.00	211.33 b-e	10.34 de	23.30 c	3.86 bc	1.94 a
TAM-4	0.020 d-f	0.00 c	18.87 d	16.73 a-c	0.94	168.67 de	16.67 b-d	57.63 c	2.42 bc	0.00 c
TAM-5	0.013 f	0.00 c	23.57 d	18.43 ab	0.00	182.00 c-e	17.00 b-d	38.57 c	0.00 c	0.00 c
TAM-6	0.021 d-f	0.00 c	14.80 d	17.33 ab	0.00	151.67 e	14.97 b-e	52.27 c	2.30 bc	0.00 c
TAM-70†	0.119 a	3.04 a	15.80 d	18.20 ab	1.03	280.33 a	18.87 b	61.23 c	5.50 ab	1.76 a
TAM-8	0.041 cd	1.28 bc	16.50 d	13.83 bc	1.46	164.33 de	13.30 b-e	42.47 c	5.21 ab	0.74 bc
TAM-9	0.028 c-f	0.00 c	13.70 d	9.91 cd	0.00	146.00 e	12.40 b-e	29.50 c	5.59 ab	0.00 c
TAM-10	0.018 ef	0.00 c	18.50 d	11.66 b-d	1.80	211.67 b-e	17.90 bc	41.13 c	4.98 ab	0.00 c
TTU-1	0.039 c-e	1.19 bc	156.50 b-d	16.80 a-c	1.66	251.00 ab	12.75 b-e	121.00 b	5.13 ab	0.69 bc
TTU-2	0.034 c-f	0.00 c	212.00 b	12.40 b-d	0.77	170.50 de	11.05 c-e	151.00 b	5.32 ab	0.00 c
TTU-3	0.016 f	0.00 c	442.50 a	6.95 d	0.00	186.00 b-e	15.65 b-e	228.50 a	3.23 bc	0.00 c
TTU-4	0.044 c	2.30 ab	203.00 bc	18.65 ab	0.00	157.00 e	12.95 b-e	123.00 b	6.06 ab	1.33 ab
TTU-5	0.114 a	0.82 bc	75.70 cd	17.60 ab	0.48	228.50 a-d	9.12 e	59.45 c	5.15 ab	0.48 bc

† This site was a marshaling area; sample collected post-use from 9/13/94 to 10/1/94.

Table 6. Site descriptions.

Agency	Sampler #	Site	Description
TAM†	1	1	Off application area; upwind; near town.
TAM	2	2	Off application area; upwind.
TAM	3	3	Off application area; upwind.
TAM	4	4	Off application area; upwind.
TAM	5	5	On application area.
TAM	6	6	On application area.
TAM	7	70	Marshaling area; mobil; in use or post-use.
TAM	7	71	Marshaling area; mobil; pre-use.
TAM	8	8	Off application area; downwind.
TAM	9	9	On application area.
TAM	10	10	Off application area; downwind; surrounded by application areas on N, W, and E sides.
TTU	1	1	Off application area; upwind.
TTU	2	1	
TTU	3	2	On application area.
TTU	4	2	
TTU	5	3	Off application area; downwind.
TTU	6	3	
TTU	7	4	Marshaling area; stationary.
TTU	8	4	
TTU	9	5	On application area; near major road.
TTU	10	5	Off application area; downwind; near major road.

† TAM samplers were replicated three times on 10 sites; TTU samplers were replicated twice on five sites.


TAEX Samplers #
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LEGEND

Biosolids: Beneficial Uses For Semi-Arid Rangeland Restoration

S. Straka

This paper presents an overview of studies related to the beneficial use of biosolids on semi-arid rangelands. This review starts with a definition of biosolids, then discusses semi-arid rangeland deterioration. The beneficial effects of biosolids applied to such rangelands are then discussed along with potential uses for biosolids as a land management tool.

What Are Biosolids?

Biosolids are defined by the Colorado Department of Public Health and Environment as: *The accumulated residual product resulting from domestic wastewater treatment works. Biosolids does not include grit or screening from a wastewater treatment works, grease, commercial or industrial sludges, or domestic and industrial septage. Biosolids do not include animal manures, untreated septage, municipal solid wastes, hazardous wastes, industrial sludges such as those generated from oil and gas refineries.* The intent of this definition is to differentiate between untreated sludge that is randomly disposed, and biosolids which are a highly treated organic amendment with land application procedures regulated by federal state and local guidelines to promote beneficial uses. This differentiation is not completely recognized, and public perception of biosolids is often of "sludge" that cities are trying to "dump" in rural areas. However, this "dumping", or land application of biosolids, can be a very effective method to restore rangeland quality in semi-arid areas by increasing soil nutrients, organic matter, electroconductivity (EC) and moisture. The restored rangelands retain more moisture, lose less soil to erosion, and yield better quality and quantity.

Biosolids are not untreated raw sewage materials; they are the treated residuals from domestic wastewater treatment processes. Biosolids have undergone screening (to remove large inorganic materials) and grit removal (to removal small inorganic material). The remaining highly organic material is then be processed in a manner certified by the U.S. Environmental Protection Agency (EPA) to destroy pathogens. Pathogen destruction may involve anaerobic digestion, aerobic digestion, heating, drying, or pH adjustment.

Biosolids that have not undergone pathogen destruction cannot be land applied

The specific biochemical properties of biosolids depend on the type of waste stream they are derived from, and the type of treatment they are produced from. Generally, biosolids are high in organic matter and nutrients, with some trace concentrations of metals and plant micronutrients. The chemical properties of biosolids from three different domestic wastewater treatment systems in Colorado are shown in Table 1. Biosolids contain a form of nitrogen that is more readily available to rangeland plants than nitrogen from commercial fertilizers (Barbarick and Westfall, 1994).

Deterioration of Semi-Arid Rangelands

Rangelands are defined as "uncultivated land capable of providing habitat for domestic and wild animals" (Holecheck et. al., 1989). Rangelands are not capable of providing a resource for cultivation due to limiting soil and moisture conditions. Extensive areas of semi-arid rangeland in the Western United States are characterized by deteriorated vegetation and soil conditions. This deteriorated state results from historical misuse and is continued by a vicious cycle of water losses. Unregulated grazing in the late nineteenth and early twentieth centuries caused severe and long-lasting damage to rangeland watersheds throughout the West (Chaney et. al., 1993). This overgrazing reduced the rangeland vegetative cover. As a result, runoff volumes increased and soil quality deteriorated, leading to further vegetative losses.

Table 1: Chemical Properties (dry weight basis) of three biosolids from Colorado.[1]

Parameter, units	Littleton/ Englewood[2]	Fort Collins[3]	Metro Denver[4]
EC[5], ds/m	11.6	5	12.7
Organic N, %	2.88	4.22	6.31
NH4-N, %	0.47	0.40	1.35
NO3-N, %	0.01	0.01	0.01
Phosphorus (P), %	2.52	1.6	2.32
Potassium (K), %	0.283	0.194	0.200
Arsenic (As), %	4	3	3
Cadmium (Cd), mg/kg	6	5	10
Chromium (Cr), mg/kg	98	40	80
Copper (Cu), mg/kg	558	553	500
Mercury (Hg), mg/kg	0.8	6.2	3.0
Lead (Pb), mg/kg	45	117	138
Molybdenum (Mo),	26	16	31
Nickel (Ni), mg/kg	85	19	41
Selenium (Se), mg/kg	13	14	4
Zinc (Zn), mg/kg	942	776	915

1 From: Barbarick, K., and Westfall, D., 1994 "Service In Action". Colorado State University Cooperative Extension bulletin No. 0.547.
2 Applied to experimental plots near Bennett, CO in August, 1993.
3 Applied to experimental plots on the Meadow Springs Ranch near Fort Collins, CO in August 1991.
4 Metrogro™ cake chemical analysis, 1993.
5 EC is a measure of the soluble salt concentration.

Soil water availability and fertility in semi-arid rangelands directly affects forage production. Deficiencies in soil water reduce plant vigor, and result in plant mortality and changes in rangeland species composition. Leaf conductance and transpiration studies in north-western Colorado showed that moisture rapidly decreases through the growing season or with rangeland disturbances, while plant photosynthetic rates decreased proportionally to losses in soil moisture (Bohnam et. al., 1990). Studies conducted to determine crop coefficients for rangelands showed that under rangeland conditions, water is often limiting and soil water evaporation cannot be ignored (Wight and Hanson, 1990). Next to soil water, soil fertility is the most limiting factor to forage production in rangelands with less than 500 mm annual precipitation (Holecheck et. al., 1989).

In an effort to halt semi-arid rangeland degradation, research has been conducted for the past 40 years to find effective land amendments to restore rangeland productivity. Studies in Northeastern Colorado in the early 1950's compared the effectiveness of manure to commercial fertilizer for rangeland rehabilitation. The manure was found to be the most effective treatment, increasing herbage yields from 15 to 50 percent. Yields from plots treated with commercial fertilizers seldom exceeded those from the untreated native range (Klipple and Retzer, 1959). Another study conducted in 1951 in Saskatchewan concluded that the benefit of heavy applications of manure are both immediate and lasting, while the effect of commercial fertilizers may not be worth the cost (Lodge, 1959). At a visit to this research site twelve years after the manure application, the beneficial effects were still evident.

Even though the beneficial effects of manure application are recognized, most ranchers do not have the mechanical resources for land application, or the money to afford the substantial cost of widespread fertilizer application. However, biosolids now provide a low cost alternative for rangeland rehabilitation. Biosolids are readily available, with many communities looking for safe, economically feasible, beneficial use opportunities (Aguilar et. al., 1992). Since the land application of biosolids is a benefit to the wastewater treatment manager, and the rancher, capitol costs for land application equipment may be shared or be considered as part of the municipality's budget for wastewater treatment.

The Effect of Dry-Land Biosolids Application

Small scale field studies have been conducted at several semi-arid rangeland sites to assess the benefits and environmental impacts of biosolids application. The environmental impacts were monitored to combat the perceived threat of introducing contaminants into the environment; the thought that currently limits public acceptance of biosolids. The results of some of these studies are summarized below.

Increased Soil Fertility
Researchers from the Rocky Mountain Forest and Range Experiment Station conducted a series of investigations at biosolids test plots in the Rio Puerco Watershed in northwest New Mexico to determine the effect of biosolids on the rangeland quality. These studies focused primarily on soil characteristics, microbial communities, and vegetation responses to the biosolids application. The researchers applied anaerobically digested biosolids at rates of 0, 22.5, 45, and 90 MG/ha to four sets of test plots, for a total of 16 plots. Samples were then taken from these prior to biosolids application, and annually (after the growing season) for the next five years.

Results from the Rio Puerco studies showed that soil EC, nutrient, micronutrient, and metals concentrations increased linearly with the biosolids application rates for the first two years of application to all the test plots. However, after two years the EC levels in the heaviest biosolids application decreased. The soil pH decreased through the sampling period, most significantly in the 90 MG/ha test plot. This was attributed to the fact that the biosolids pH was lower than that of the rangeland. Application of dried, anaerobically digested biosolids at 90 Mg/ha resulted in an increase in concentrations of Copper, Manganese and Cadmium to just above acceptable standards after four growing seasons, which the researchers attributed partially to the pH reductions.

The researchers concluded that a single biosolids application has the potential to serve as a low-grade, slow release fertilizer for this soil for several years (Dennis and Fresquez, 1989). However, they recommended that the biosolids be applied at rates of 22-45 MG/ha to prevent the potential for metals toxicity.

Increased Soil Microbial Communities

Soil fertility is also related to soil biota. Microbes and fungi are the major agents in the decomposition of organic matter and soil stabilization, and serve to increase the availability of plant nutrients, and fix nitrogen from the atmosphere. In the Rio Puerco test plots, soil bacterial, fungal, and ammonium oxidizer populations increased linearly with increasing biosolids application rate (Dennis and Fresquez, 1989). Improvement in soil fertility and organic matter as a result of biosolids application was also reflected in fungal, diversity, and in the composition of the fungal community (Fresquez and Dennis, 1989). Fungal diversity decreased but overall populations increased after biosolids application. This trend in fungal population is due to increased nutrient availability.

A previous study conducted in north-central New Mexico, Whitford et. al., (1989) also investigated the addition of organic materials on soil biota activity and the resulting effects on soil organic matter and nutrients. The application of dried municipal biosolids showed no significant effect on biomass and activity of soil microflora. However, the application of 1 MG/ha of dried municipal biosolids was lower than used in the previous studies. Also, the second year after biosolids application was much drier than the first growing season, which may have affected microbial population dynamics. The researchers also proposed that the lack of variation of nematode populations after the first year may be due to increased populations of nematode grazers.

Increased Quality and Quantity of Vegetation

The Rio Puerco biosolids application sites were also monitored by Fresquez et. al. (1990a, 1990b, 1991), to determine the effect of biosolids amendments on forage productivity, and metal uptake by the primarily blue grama/broom snakeweed plant population. Plant tissue analyses conducted one, two and five growing seasons after biosolids application showed that plant biomass, crude protein, nitrogen, phosphorus and potassium concentrations increased with biosolids applications for the first two growing seasons. However, after five growing seasons, there was no significant difference in plant quality between the amended and unamended sites. The plant tissue metals concentrations were not significantly different from those in the unamended sites, proving that the decreased soil pH did not increase metals content in plant tissue as a result of solubilization, even at the highest loading rate. Some plant metals actually decreased in concentration with application of the biosolids. The researchers proposed that this was due to the diluting effect of increased plant growth (Fresquez et. al., 1990b)

Another biosolids application study was conducted at the Meadow Springs Ranch, owned by the City of Fort Collins, in northcentral Colorado. Test plots on the Ranch were amended with dewatered and composted biosolids, applied at rates of 0, 2.2, 4.5, 11.0, 22.0 and 34.0 MG/ha. Water treatment plant residuals were also applied at rates of 5.5, 11.0 and 22.0 MG/ha. Vegetation sampled from these plots showed increased nitrogen, potassium and phosphorus concentrations in the plant tissues. Metals concentrations in plant tissues remained within the normal ranges for plant tissue. The biomass and canopy cover of the two dominant plant species, blue grama and fringed sage, showed increases in biomass as application rates increased. The water treatment plant residuals, when mixed with wastewater biosolids, had the same rangeland effect as the unblended wastewater biosolids (Gallier, et. al., 1993).

Reduced Runoff

Researchers at the Rocky Mountain Range and Forest Experiment Station conducted a further series of runoff biosolids studies at the Sevilleta National Wildlife Refuge in New Mexico. The study objectives were to determine the influencing factors for runoff from semi-arid rangeland; and to investigate runoff characteristics from areas amended with biosolids. Results from comparisons of runoff and erosion from two different soils at varying slopes showed that slope and soil textural differences influence the rate of water infiltration and resulting runoff and erosion in semi-arid rangeland (Aguilar and Aldon, 1991a). Runoff from four high intensity summer storms, and from higher intensity simulated rainfalls, was collected from plots applied with 45 MG/ha anaerobically digested biosolids. The runoff test plots were located on level areas and also on steeper slopes. The runoff volumes and constituents were compared with those from adjacent unamended plots. Comparison of the volumes indicated that runoff from the biosolids sites was significantly reduced. This was attributed to increased ground surface roughness and absorption of water by the biosolids (Aguilar and Loftin, 1991).

Analysis of the runoff for potential contamination showed that runoff from test plots carried metals in concentrations well within New Mexico groundwater and livestock water quality standards. There was no significant difference between these concentrations and those from the unamended plots. Additionally, there was no significant difference between nitrate concentrations from the test plots and the control plots after both natural and simulated rainfalls. The concentrations of nitrates in the runoff from the test plots were well below the acceptable New Mexico State standards of 10 mg/l nitrate for groundwater or surface water supplies.

The study concluded that contamination of surface water by constituents in biosolids do not appear to be a limitation for biosolids application as a fertilizer and mulch amendment in a semi-arid rangeland (Aguilar and Loftin, 1991).

Runoff response test plots were also established at the Fort Collins Meadow Springs Ranch. Anaerobically digested wastewater treatment biosolids, water treatment residuals, and a blend of the two solids were applied on short-grass prairies at rates of 0, 2.2, 4.5, 11.0, 22.0 and 34.0 dry MG/ha on land surface slopes ranging from 0 to 15 percent. (McCurry, Janonis and Gallier, 1993). Runoff analysis from these plots showed increasing sediment, metals and nutrients in the runoff from the test sites proportional to biosolids application rates. However, runoff metals concentrations did not exceed the concentrations listed by the EPA for federal drinking water standards. Aluminum and iron were the only exceptions to this trend; concentrations of these metals decreased as application rates increased (Gallier et. al., 1993). These results indicate that although runoff is decreased by biosolids application, the remaining runoff contains more concentrated solids, nutrients and metals than unamended areas.

Increased Infiltration
Infiltration rates on semi-arid rangelands are influenced by soil compaction, antecedent soil moisture, and vegetative cover. Researchers in Somalia investigating the effect of grazing on infiltration rates did not find a correlation, but did see increased infiltration in areas with higher cover and biomass accumulation (Takar, et. al, 1990). Studies on the effect of shrub management on water yield from semi-arid rangeland showed that plant cover limited cracking of the soil which reduced the depth from which evaporation occurs (Carlson et. al., 1990), therefore conserving soil moisture.

In order to determine if the water retention provided by land applied biosolids increases infiltration, monitoring of unsaturated zone leachate from biosolids application sites was conducted on the runoff test plots at the Fort Collins Meadow Springs Ranch. A set of unsaturated zone monitoring devices were installed within the test plots receiving the highest biosolids application rates and also in a control site. Results showed an increasing steady state infiltration rate with increasing application rates. Nitrogen in the soil increased with application rates, but this response was only apparent to a soil depth of 30.5 cm (Gallier et. al., 1993).

The results showing limited nitrogen leaching corresponded to a study conducted by the Rocky Mountain Range and Forest Experiment Station to determine the potential for groundwater contamination in semi-arid rangeland. Subsurface infiltration dynamics were investigated by Aguilar and Aldon (1991b) in the Rio Puerco watershed. Study results showed that leaching in semiarid environments due to saturated flow will seldom occur below 1.5 m in Querencia soils or similar soil types. The study also showed that precipitation and evaporation dictated the water content in the upper 30 cm of soil.

Biosolids Management

Research results prove that land amendment of biosolids at light to moderate loadings (0-45 MG/ha) is beneficial to semi-arid rangelands, if properly managed. Biosolids application reduces runoff, increases infiltration, and results in increase biomass growth and quality. However, heavy loadings of biosolids may result in the accumulation of metals at unacceptable concentrations, which may in turn lead to increased metal concentrations in vegetative tissues and surface water. Heavy loadings of biosolids may also result in the contamination of groundwater by nitrates if the application rate exceeds the nitrogen demand of the vegetation. Also, since runoff from biosolids amended areas is more concentrated, biosolids should not be applied in riparian zones.

The EPA issued the 40 CFR Part 503 Biosolids Rule in February 1993 to promote biosolids reuse and to ensure proper management of biosolids loading rates and locations. All biosolids land application programs must meet these regulations, in addition to any existing state and local regulations. In order to protect the environment and public health, these regulations require the application of biosolids at an agronomic rate determined by the nitrogen demand of the cover crop or vegetation. The regulations determine the total amount of biosolids that can be applied to a site (site life) based on the trace metals concentration. These loading restrictions prevent potential conditions for groundwater contamination or metals toxicity from occurring at the application site. Additionally, the regulations prevent the land application of biosolids on steep slopes, high infiltration areas, riparian zones, and other locations where concentrated runoff could be a concern (EPA 1993).

Proper management of biosolids may include objectives to improve the quality of biosolids so that they may be a safe resource at any loading rate. One author suggests the use of recombinant DNA techniques to develop a more degradable biosolids to minimize disposal problems; or to qualify municipal waste as a renewable resource and separating industrial waste streams which contribute to metals concerns (Boyle, 1990). However these options may not be socially, economically or politically feasible.

Biosolids management may also include developing a model of watershed-scale effects of biosolids application. Runoff models used to predict the effect of management practice on reducing runoff from rangeland watersheds (Osborn and Simanton, 1990) could be implemented to determine biosolids impacts on watershed hydrology.

Biosolids Use for Land Management Purposes

Properly applied biosolids are a renewable resource with many potential beneficial uses for semi-arid rangelands, including improving soil quality (as mentioned previously), cultivating or selecting for preferred plant species, manipulating animal grazing patterns, and increasing stocking rates.

Plant Manipulation

Biosolids may be used to inhibit unwanted plant species. An interesting result of the biosolids applications conducted in the Rio Puerco watershed was a dramatic decrease in the broom snakeweed population. After 4 years, the broom snakeweed plants in all of the biosolids application plots were dead while those in the unamended sites had regrowth. The researchers proposed the predominance of blue grama may be due to increased nitrogen availability in the root zone which can be utilized by the shallow rooted grasses. They concluded that the decrease of broom snakeweed and increase of blue grama on biosolids amended sites shows the potential for biosolids to be used as a slow release organic fertilizer for land management purposes (Fresquez et. al., 1990a). The blue grama dominance may also be due to increased soil moisture (Samuel and Hart, 1992) that results from biosolids dry-land application. The vegetative response is correlated by plant species at a Bureau of Land Management (BLM) site near Wolcott, Colorado, where land applied biosolids resulted in increases in three grass species (western wheatgrass, alkali bluegrass, and big sagebrush) and decrease in one forb species (nailwort). The vegetative responses were directly proportional to biosolids application rates (Gallier et. al., 1993).

Reseeding

Biosolids may also be used to aid in reseeding disturbed areas with grasses that are otherwise difficult to cultivate. When manure was applied at a rate of 12.8 KG/ha and seeded with crested wheatgrass, both the crested wheatgrass and the native grass population (including needle-and thread, blue grama and western wheatgrass) increased while the mossy groundcover of selaginella decreased (Heady, 1952). Reexamination of the test plots 12 years after manure application showed that taller plant growth was still evident in the manured areas.

Reduce Wind Erosion

Biosolids applied to eroded areas will reduce soil wind erosion. Dust is created by both wind and soil conditions (Pennisi, 1992). Larger moist biosolids particles are less likely to be suspended in the air than smaller eroded soil particles, and serve to "weigh down" the underlying soil. The increased vegetative cover that results from the biosolids application will further protect the soil from wind erosion. Biosolids application for the reduction of wind erosion has proven successful on a private ranch in eastern Oregon (Gallier et. al., 1993).

Influence Grazing Patterns

In addition to providing a tool for land management, biosolids application could also provide a means for animal management. In a study conducted in 1976 on big game rangeland in Utah, elk showed a grazing preference for fertilized areas instead of native grassland (Bayoumi and Smith, 1976). The U.S. Air Force Academy, near Colorado Springs, Colorado, improved vegetative cover with the application of biosolids at rates less that 1 MG/ha, which resulted in improved wildlife habitat. (Gallier et. al., 1993)

Increase Stocking Rates

The rangeland improvements resulting from biosolids application is beneficial to animal husbandry. Stocking rates at the private Oregon ranch where biosolids were applied to reduce wind erosion increased from one cow per 6 ha to one cow per 2.1 ha. Biosolids application to a ranch near Gunnison, Colorado has allowed grazing frequency to increase from once to two times per year (Gallier et. al., 1993).

The value of dry land application of biosolids obviously extends beyond the municipalities and engineers seeking a solution for residuals disposal. Rather than being simply a beneficial disposal solution, biosolids application on semi-arid rangeland is a benefit which extends to the rancher seeking to improve land value and animal production, to environmentalists concerned with wildlife habit and rangeland ecosystem destruction, and essentially to all semi-arid rangeland managers and users.

References

Aguilar, R., and Aldon, E. 1991a. Runoff and Sediment Rates on San Mateo and Querencia Soils, Rio Puerco Watershed Management Area, NM. USDA Forest Service Rocky Mountain Forest and Range Experiment Station, Research Note RM-506.

Aguilar, R., and Aldon, E. 1991b. Seasonal Water Flux and Potential for Leaching in a Semiarid Rangeland Soil. Proceedings if the Fifth National Outdoor Action Conference on Aquifer Restoration, Ground Water Monitoring, and Geophysical Methods. May 13-16, 1991. Ground Water Management No. 5:669-683.

Aguilar, R., Gosz, J., and Ward, T. 1992. Sewage Sludge Application in Semiarid Grasslands: Effects on Vegetation and Water Quality. *New Mexico Water Resources Research Institute Annual Report of Attainment, Project Number 1423645.*

Aguilar, R., and Loftin, S. 1991. Sewage Sludge Application in Semiarid Grasslands: Effects on Runoff and Surface Water Quality. New Mexico Water Resources Research Institute Report. 265: 101-111.

Barbarick, K. and Westfall, D. 1994. "Service in Action". Colorado State University Cooperative Extension, Bulletin No. 0.547.

Bayoumi, M., and Smith, D. 1976. Response of Big Game Winter Range Vegetation to Fertilization. *J. Range Manage.* 29(1):44-47.

Bonham, C., Mack, S., and Trlica, M. 1990. Leaf Conductance and Transpiration of Winterfat Associated with 2 Species of Wheatgrass on Disturbed Sites. *J. Range Manage.* 43(6):501-505.

Boyle, Michael. 1990. Biodegradation of Land Applied Sludge. J. Environ. Qual. 19:640-644.

Carlson, D., Thurow, T., Knight, R., and Heitschmidt, R. 1990. Effect of Honey Mesquite on the Water Balance of Texas Rolling Plains Rangeland. *J. Range Manage.* 43(6):491-496.

Chaney, E., Elmore, W., and Platts, W. 1993. Managing Change-Livestock Grazing on Western Riparian Areas. USEPA-Northwest Information Center Inc., Eagle, Idaho.

Chaney, E., Elmore, W., and Platts, W. 1990. Livestock Grazing on Western Riparian Areas. USEPA.

Colorado Department of Public Health and Environment. 1994. "Biosolids Regulations" 4.9.0 (5CCR 1002-19). Colorado Water Quality Control Comission. Denver, Colorado.

Dennis, G., and Fresquez, P. 1989. The Soil Microbial Community in a Sewage-Sludge-Amended Semi-arid Grassland. Biol. Fertil. Soils. 7:310-317.

Environmental Protection Agency. 1993. "Standards for the Use or Disposal of Sewage Sludge; Final Rule". 40 CFR Part 503. 58 Federal Register 9248.

Fresquez, P., and Dennis, G. 1989. Composition of Fungal Groups Associated with Sewage Sludge Amended Grassland Soils. Arid Soil Research and Rehabilitation. 4:19-32.

Fresquez, P., Aguilar, R., Francis, R., and Aldon, E. 1991. Heavy Metal Uptake by Blue Grama Growing in a Degraded Semiarid Soil Amended with Sewage Sludge. *Water, Air and Soil Pollution.* 57-58:903-912.

Fresquez, P., Francis, R., and Dennis, G. 1990a. Soil and Vegetation Responses to Sewage Sludge on a Degraded Semiarid Broom Snakeweed/Blue Grama Plant Community. *J. Range Manage.* 43(4):325-331.

Fresqeuz, P., Francis, R., and Dennis, G. 1990b. Sewage Sludge Effects on Soil and Plant Quality in a Degraded, Semiarid Grassland. *J. Environ. Qual.* 19:324-329.

Gallier, T., Brobst, R., Aguilar, R., Barbarick, K., Hegeman, P., Janonis, B., Salahub, D., and Wilson, S. 1993. *Water, Environment and Technology.* 5(10): .

Heady, H. 1952. Reseeding, Fertilizing and Renovating in an Ungrazed Mixed Prairie. *J. Range. Manage.* 5:144-149.

Holechek, J., Peiper, R., and Herbel, C. 1989. Range Management. Prentice-Hall. New Jersey.

Klipple, G., and Retzer, J. 1959. Response of Native Vegetation on the Central Great Plains to Applications of Corral Manure and Commercial Fertilizer. *J. Range Manage.* 12:239-243.

Lodge, R. 1959. Fertilization of Native Range in the Northern Great Plains. *J. Range Manage.* 12:277-279.

McCurry, G. , Janonis, B., Gallier, T. 1993. Methods to Evaluate Impacts of Sludge Application. in *Proceedings from the 1993 Joint CSCE-ASCE National Conference on Environmental Engineering.* 1:413-420.

Mullahey, J., Waller, S., and Moser, L. 1990. Defoliation Effects on Production and Morphological Development of Little Bluestem. *J. Range Manage.* 43(6):497-??.

Osborn, H. and Simanton, J. 1990. Hydrologic Modeling of a Treated Rangeland Watershed. J. Range Manage. 43(6):474-481.

Pennisi, E. 1992. Dancing Dust. *Science News*. 142:218-220.

Samuel, M. and Hart, R. 1992. Survival and Growth if Blue Grama Seedlings in Competition with Western Wheatgrass. *J. Range Manage.* 45(5):444-448.

Takar, A., Dobrowolski, J., and Thurow, T. 1990. Influence of Grazing, Vegetation Life-Form, and Soil Type on Infiltration Rates and Interill Erosion on a Somalian Rangeland. *J. Range Manage*. 43(6):486-490.

Whitford, W., Aldon, E., Freckman, D., Steinberger, Y., and Parker, L. 1989. Effect of Organic Amendments on Soil Biota on a Degraded Rangeland. *J. Range Manage*. 42(1):56-60.

Wight, J. and Hanson, C. 1990. Crop Coefficients for Rangeland. *J. Range Manage.* 43(6):482-485