SEWAGE SLUDGE APPLICATION IN SEMIARID GRASSLANDS: EFFECTS ON VEGETATION AND WATER QUALITY

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ABSTRACT

The effects of municipal sewage sludge (45 Mg ha^{-1} oven dry weight basis, surface-applied) on semiarid rangeland in central New Mexico were studied over a two-year period. Sewage sludge application increased total nitrogen (TKN) and extractable ammonium (NH_4-N), nitrate (NO_3-N), and phosphorus (P), but did not significantly increase soil organic carbon (OC) in the mineral soil underlying the applied sludge. Extractable soil copper (Cu), lead (Pb), and zinc (Zn) increased significantly following sludge application, but remained at low concentrations (<4.0 mg kg^{-1}) well below established regulatory standards. Soil cadmium (Cd) levels remained below detection limits.

Above-ground plant cover on treated plots increased following the sludge application, but only in response to rainfall. Root growth was significantly less on treated plots. Above-ground plant tissue from sludge-treated plots had significantly greater TKN and Cu.

Results from natural storms and rainfall simulation experiments showed that surface-applied sludge significantly reduces surface runoff and total sediment yield. Given the immediate reductions in runoff and total sediment yields and the favorable vegetative response to the added sludge, surface application of municipal sewage sludge to semiarid rangelands has the potential to reduce deleterious runoff and sediment from Southwestern rangelands.

Keywords: wastewater biosolids, plant response, soil organic matter, soil organic carbon, plant nutrients, heavy metals, plant litter decomposition, above- and below-ground plant response, runoff, sediment yield, erosion.

INTRODUCTION

Research Goals

Accelerated erosion from rangeland increases sediment in drainages and streams causing serious water quality problems in New Mexico and other areas of the Southwest. Municipal sewage sludge (wastewater biosolids) has proven beneficial to rangeland as a soil fertility amendment. However, sewage sludge may contain trace element contaminants that could potentially pollute surface and groundwater resources. Research is needed that determines the effects of sewage sludge on vegetation, soils, and hydrologic response and examines the potential for water contamination in semiarid grassland environments treated with sludge.

We hypothesized that increased plant growth resulting from sewage sludge application to rangeland would decrease runoff and sediment yields in treated areas without contaminating water resources by sludge-borne constituents. Our preliminary research findings suggest that heavy metal uptake by vegetation and transport of potential contaminants from sludge in treated areas via runoff is minimal (Fresquez et al. 1991, Aguilar and Loftin 1992). If sludge application on semiarid rangeland is proven environmentally sound, this practice could be conducted on entire watersheds leading to improved water quality.

Study Objectives

Municipal sewage sludge was used as a soil organic matter and nutrient amendment in an attempt to increase rangeland productivity and above-ground plant cover and to reduce soil erosion. Our objectives were to: 1) determine the effects of sewage sludge application on soils and vegetation in a grassland environment within the Sevilleta National Wildlife Refuge, central New Mexico, 2) determine how subsequent changes in vegetation following sludge application influence runoff water and sediment yields, and 3) determine the fate of potential sludge-borne contaminants.

Background Information

Approximately six million metric tons of municipal sewage sludge are produced annually in the United States (U.S. Environmental Protection Agency 1990). Disposal of this waste product has become a major problem for large metropolitan areas, particularly those along the heavily populated eastern seaboard. Large urban areas of the Southwest, including the city of Albuquerque, also produce sewage sludge from wastewater treatment plants, and disposal remains a problem. Currently, much of Albuquerque's sewage sludge is tilled into the soil on rangeland set aside specifically for sludge disposal. Safe and economical sludge disposal, not rehabilitation of the rangeland affected, is the city's primary objective. However, innovative ways of beneficially using solid-waste products, including sewage sludge, must be continually developed. The use of municipal sewage sludge for rehabilitating degraded rangeland represents an alternative to sludge disposal.

Sewage sludge can essentially act as a fertilizer treatment. Albuquerque municipal sewage sludge contains high amounts of total nitrogen (TKN), phosphorus (P), nutrient cations (K, Ca, Mg) and organic carbon (OC) (Table 1). For this reason we expect an increase in plant production and vegetative cover in degraded rangeland following sludge application.

Table 1. Mineral nutrient content of anaerobically treated municipal sewage sludge as compared with manure, straw and wood chips. [adapted from Fresquez et al. 1988]

	TKN[1]	P	K	Ca	Mg	OC	C:N	pH
	------------------ (mg kg^{-1})				------------------	(g kg^{-1})		
Sewage Sludge	10600	632	234	843	55	61	6	7.22
Manure	4190	1162	4279	21	17	290	69	8.57
Straw	1520	101	1123	46	16	218	143	7.95
Woodchips	520	2	121	21	6	170	327	5.74

[1] Total nitrogen (TKN) determined by macro-Kjeldahl procedure; P determined from 1:5 substance-water extract; Ca, Mg, and K are water-soluble indices; organic carbon (OC) determined by Walkey-Black method; pH determined from saturated paste.

Sewage sludge has been successfully used for agricultural purposes (Berglund et al. 1984, Catroux et al. 1981) and in mined land reclamation efforts (Sopper and Kerr 1979). More recently, a pioneering study in a semiarid grassland environment showed that degraded rangeland responds favorably to the application of sewage sludge as a fertilizer and organic matter amendment (Fresquez et al. 1990b). Study results showed that a one-time surface application of 22.5 to 45.0 Mg ha^{-1} (oven dry weight basis) of anaerobically digested sewage sludge did not lead to soil or plant tissue contamination

(Fresquez et al. 1990a, Fresquez et al. 1991). Information on the effects of surface-applied sludge on runoff and sediment yields was not obtained in this study.

Vegetation cover can influence the hydrology of a watershed (Kirkby 1978). Plant foliage acts as a buffer to shield the soil surface from the impact of raindrops. Precipitation intercepted by vegetation may evaporate before reaching the soil. Surface runoff is impeded by vegetation, thereby resulting in increased infiltration. Increased soil organic matter, a direct result of sludge application and an indirect result of subsequent increased vegetative growth, will also enhance infiltration and precipitation retention.

Increased water uptake and transpiration by plants can further reduce runoff and erosion by lowering soil-water content prior to high-intensity rainfall. These phenomena act to decrease water and sediment yield from semiarid rangeland in response to the short duration, high-intensity summer thunderstorms common to these environments.

Despite the potential benefits to rangeland, a primary concern limiting the use of municipal sewage sludge as a soil amendment is the potential for introducing contaminants to the food chain and surface and groundwater resources. The effects of sewage sludge on soil organic matter and trace element interactions are also major concerns. Depending upon its source, sludge contains varying concentrations of aluminum (Al) and boron (B), and the trace elements cadmium (Cd), copper (Cu), nickel (Ni), lead (Pb), and zinc (Zn) (Sommers 1977). Ryan et al. (1982) identified Cd as the sludge-borne metal with the greatest potential hazard when sludge is applied to land. Heavy-metal movement in soils is dependent upon numerous factors including soil pH, cation exchange capacity (CEC), and organic matter content. However, metal mobility is restricted in alkaline arid and semiarid soils due to the formation of insoluble complexes formed with carbonates and phosphates (Bohn et al. 1979). The potential contaminants we evaluated in this study include NO_3-N and the heavy metals Cd, Cu, Pb and Zn. These specific elements were chosen for study because of their potential accumulation and, in the case of NO_3-N, mobility in soil/vegetation systems (Adriano 1986, Holtzclaw et al. 1978, Sidle and Kardos 1977, Williams et al. 1980).

In semiarid grasslands, surface runoff during high-intensity summer storms is likely to be the primary means of contaminant transport in sludge-amended areas (Bolton et al. 1991). Loss of contaminants by volatilization may occur under certain conditions. As much as 60% of the N in surface-applied sludge can be lost within months after application (Ryan and Keeney 1975, Suhr 1982, Terry et al. 1978). These N losses are attributed chiefly to volatilization as gaseous ammonia (NH_3). Contaminants not lost from grasslands through runoff, leaching, or volatilization will remain in the soil and may be adsorbed and assimilated by vegetation.

Although ground water contamination by NO_3^- generated through sludge decomposition and subsequent nitrification is possible, the probability for contamination is very low in arid and semiarid regions because of low-leaching potential. Careful site selection should alleviate any concerns of ground water contamination by NO_3^- in sludge-treated areas in the semiarid Southwest. The majority of metal contaminants are relatively insoluble above neutral soil pH (Adriano 1986). Alkaline soils coupled with low annual rainfall should rule out leaching as a major means of metal contaminant transport. Research on soil-water flux within the Rio Puerco Watershed and in the City of

Albuquerque's Soil Amendment Facility has shown that little change in water content occurs below one meter in response to precipitation in this semiarid environment (Aguilar and Aldon 1991, Glass et al. 1991). Transport by runoff water remains a possibility and, thus, determining the potential for surface water contamination by sludge was a major objective of this research.

Previous Rangeland Sludge Application Research

USDA Forest Service scientists conducted the first in-depth study of the effects of sewage sludge application to degraded semiarid rangeland (Fresquez et al. 1990a, 1990b, and 1991). Dried, anaerobically digested sewage sludge from the City of Albuquerque was surface-applied to a degraded, semiarid grassland site within the Rio Puerco Resource Area in north-central New Mexico. The Rio Puerco basin, an extremely degraded watershed with a long history of heavy livestock grazing, is one of the most eroded and overgrazed river basins in the arid West (Sheridan 1981). Sewage sludge was applied (one-time application) at 0, 22.5, 45, and 90 Mg ha^{-1} (0, 10, 20, and 40 tons $acre^{-1}$, respectively, based on oven-dried weight basis) to each of four plots (3 X 20 m) in a completely randomized block design containing a total of 16 plots. The site was characterized as a broom snakeweed (*Gutierrezia sarothrae* (Pursh) Britt. & Rusby)/blue grama (*Bouteloua gracilis* (H.B.K.) Lag.)-galleta (*Hilaria jamesii* (Torr.) Benth.) plant community on a moderately deep, medium-textured soil (Fresquez et al. 1990b). In this study, adding sludge had a tremendous impact on soil chemistry and nutrient content on treated plots (Fresquez et al. 1990a). Total N, extractable P, water soluble K, and electrical conductivity (EC) increased linearly with increased sewage sludge application during the study's first year. However, soil organic matter in mineral soil below the sludge layer did not increase until after the fifth growing season. The delayed soil organic matter response was an indirect effect of the enhanced nutrient availability with subsequent increased below-ground plant productivity and soil microbial productivity in response to the sludge amendment.

Soil pH dropped from 7.8 to 7.5 in the 90 Mg ha^{-1} treatment during the first growing season, and to 7.4 in the second growing season, probably as a result of slightly acidic leachates from the applied sludge (Fresquez et al. 1991). Acid-producing microbial reactions in the soil (i.e., nitrification) may have also contributed to the decrease in soil pH. Although soil pH decreased over time because of the sludge amendments, only diethylenetriaminepentaacetic acid (DTPA) extractable soil Cu and Cd increased to concentrations slightly above the limits considered acceptable. Tiedemann and Lopez (1982) considered >10 to 40 mg kg^{-1} DTPA extractable Cu and >0.1 to 1.0 mg kg^{-1} DTPA extractable Cd phytotoxic and undesirable in soil. However, these levels were exceeded only in the 45 and 90 Mg ha^{-1} applications following the sludge-induced drop in soil pH. Changes in other trace elements produced by the different sludge amendments are described in Fresquez et al. (1990b and 1991). The higher trace element concentrations resulting from the sludge amendments were probably directly related to sludge decomposition rather than to the solubilization of pre-existing soil micronutrients as a result of decreased pH (Fresquez et al. 1991).

Although the sludge amendments decreased total plant density, species richness, and species diversity (index of numbers of different species in relation to the total number of plants per given area), cover and yield of blue grama significantly increased on treated plots (Fresquez et al. 1990a). Blue grama production was significantly greater for all sludge-amended plots during the first and second growing seasons, with yields ranging from 1.5 to almost 3.0 times greater in the treated plots than in the unamended (control) plots. Summer precipitation during the second growing season was exceptionally high and the highest yields of dry matter production occurred during this period. After the fifth growing season, blue grama production remained higher in the 45 and 90 Mg ha^{-1} sludge-amended plots, although the benefits of the added sludge had greatly diminished for the lowest (22.5 Mg ha^{-1}) sludge amendment.

The sludge amendments also significantly increased the nutritional value of blue grama. Tissue N, P, K, and crude protein increased with the application of sludge to recommended tissue concentrations (Fresquez et al. 1991). Furthermore, most of the trace metals, including Cu and Cd, in blue grama plant tissue did not increase significantly during the five-year study, thereby reducing concerns that these toxic elements could be transferred to grazing animals. This is a significant finding because concerns over heavy metal accumulations frequently limit sewage sludge application to land. Based on these cumulative results, Fresquez et al. (1991) concluded that a one-time sludge treatment ranging from 22.5 to 45 Mg ha^{-1} (10-20 tons $acre^{-1}$) would yield the best vegetation response without potential environmental harm.

An unexpected benefit from the sludge treatment was a decrease in broom snakeweed, a toxic, non-palatable, competitive range plant (Fresquez et al. 1990a). The number of broom snakeweed plants in the sludge-amended plots decreased progressively over a four-year period (1985-1988) following the sludge treatments. The exact mechanism(s) responsible for broom snakeweed decline remains unclear, but, it was concurrent with an increase in blue grama and other desirable forage plants. The observed decrease in broom snakeweed represents a significant finding in rangeland restoration research. Budd (1989) reported that broom snakeweed occupies over 16 million hectares in New Mexico, including over 62% of the state's grazing rangeland.

MATERIALS AND METHODS

Study Site Description

The study was conducted within the Sevilleta National Wildlife Refuge (SNWR). The wildlife refuge (Fig. 1), administered by the United States Department of Interior (USDI) Fish and Wildlife Service, provided an excellent opportunity to test for treatment effects within semiarid rangeland. The refuge has been designated as a Long-Term Ecological Research Site (LTER) and receives funding from the National Science Foundation. Climate at the SNWR is arid to semiarid with mean annual precipitation ranging from 200 to 250 mm (Moore 1991). The SNWR is fenced and has not been grazed by livestock for over 17 years. Although the refuge has had a history of heavy livestock grazing, the range condition is presently good. Vegetation on the refuge is dominated by semiarid shortgrass prairie and great basin shrubland at low elevations and pinyon/juniper woodland at higher elevations.

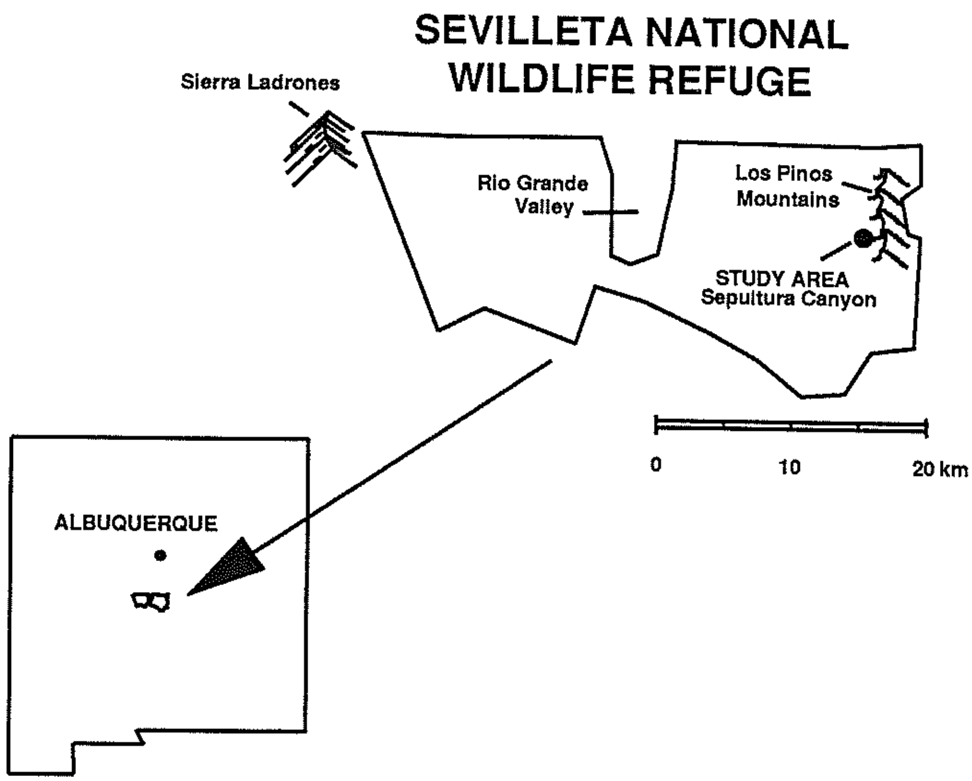

Figure 1. Detail of the Sevilleta National Wildlife Refuge, New Mexico. The study site was located near the mouth of Sepultura Canyon, on the west-facing slopes of the Los Pinos Mountains.

A blue grama/hairy grama-dominated 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, hairy grama (*Bouteloua. hirsuta* Lag.), black grama (*B. eriopoda* Torr.), ring muhly (*Muhlenbergia torrei* (Kunth) Hitchc.), three awn (*Aristida spp.*), and the shrubs broom snakeweed, yellow spiny aster (*Haplopappus spinulosus* (Pursh) DC.), narrow-leaf yucca (*Yucca glauca* Nutt.), winterfat (*Ceratoides lanata* Pursh) J.T. Howell), four-wing saltbush (*Atriplex canescens* (Torr. & Frem.) Wats.), and groundsel (*Senecio douglasii* DC. var. *longilobus* L. Bensen). Soils at the site are mapped as the Harvey-Dean Association (U.S. Dept. of Agriculture SCS 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 from 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 according to the USDA Soil Conservation Service descriptions.

Plot Establishment

We established three pairs of runoff plots, each consisting of a treated (sludge-amended) and a control (no sludge) plot, 10 m long and 3 m wide, within each of the two slope gradient classes (Fig. 2). A one meter corridor was established between the two plots in each pair. The three paired plots established at the lower component of the landscape (6-7% slope gradient) were designated as 1L-T (for treated, sludge-amended) and 1L-C (for control, no sludge), 2L-T, 2L-C, 3L-T, and 3L-C. We designated the three paired plots located in the upper portion of the study area (10-11% slope gradient) as plots 1H-T, 1H-C, 2H-T, 2H-C, 3H-T, and 3H-C. Individual runoff plot dimensions (3 X 10 m) were identical to those utilized by USDA Agricultural Research Service researchers

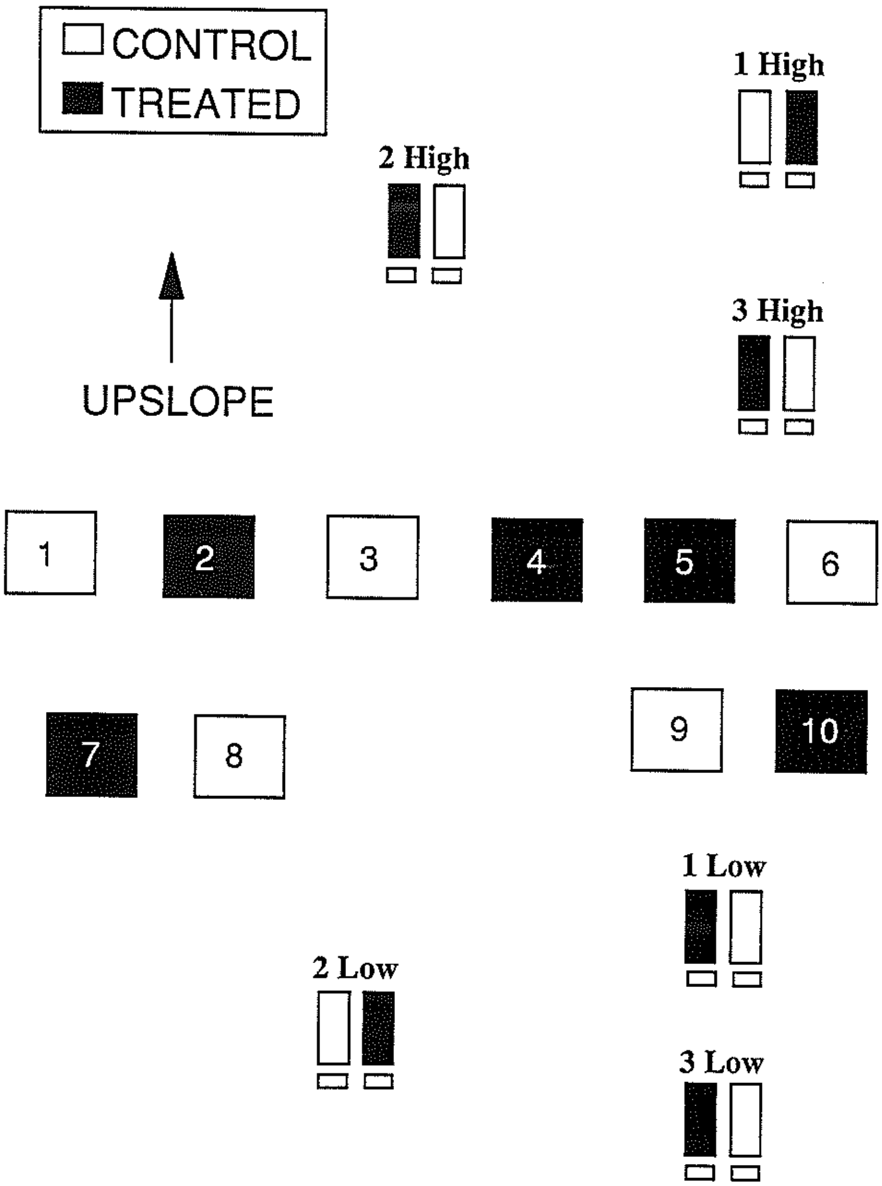

Figure 2. Experimental plots were oriented with three pairs of 3 X 10 m runoff plots in a high slope (10-11%) position, three pairs of runoff plots in a low slope (6-7%) position and five pairs of 10 X 10 m plots on an intermediate slope.

involved in the Water Erosion Prediction Project (WEPP) (U.S. Dept. of Agriculture ARS 1987). Therefore, data obtained in this study could be applied to WEPP models for larger scale predictions on runoff and sediment yield in semiarid rangeland.

Aluminum flashing (15 cm or 6 inch) bordered the runoff plots on three sides to exclude external runoff and contain surface runoff generated from the plot during rainfall events. Galvanized steel livestock tanks (1800 L) buried at the end of the plots collected the water (Fig. 3). We leveled the tanks and positioned them with a 1% drop in slope across the length of the tanks to facilitate water measurements and sampling following small runoff events. Soil was then back-filled and tamped around each tank. Gaps between plot boundaries and tank edges were lined with concrete to prevent leakage and loss of runoff water. We placed water-sealed lids (framed fiber-glass roofing panels) over the tanks to exclude direct rainfall inputs and to reduce evaporation following rainfall events.

We calibrated each runoff collection tank by recording the water level in the tank during sequential additions of water. We then developed a regression curve relating the height of water accumulation at the lowest end of the tank to the quantity of water in the tank and subsequently used this relationship as a runoff quantity prediction curve. This method enabled us to obtain an estimate of runoff yield from each plot by measuring the height of the runoff collected in the tank.

We sampled soil and vegetation within plots (five control and five sludge-amended), established on soil/vegetation assemblages similar to those on the runoff plots. Figure 2 shows the relative locations of the 10 X 10 m soil/vegetation sampling plots and the 3

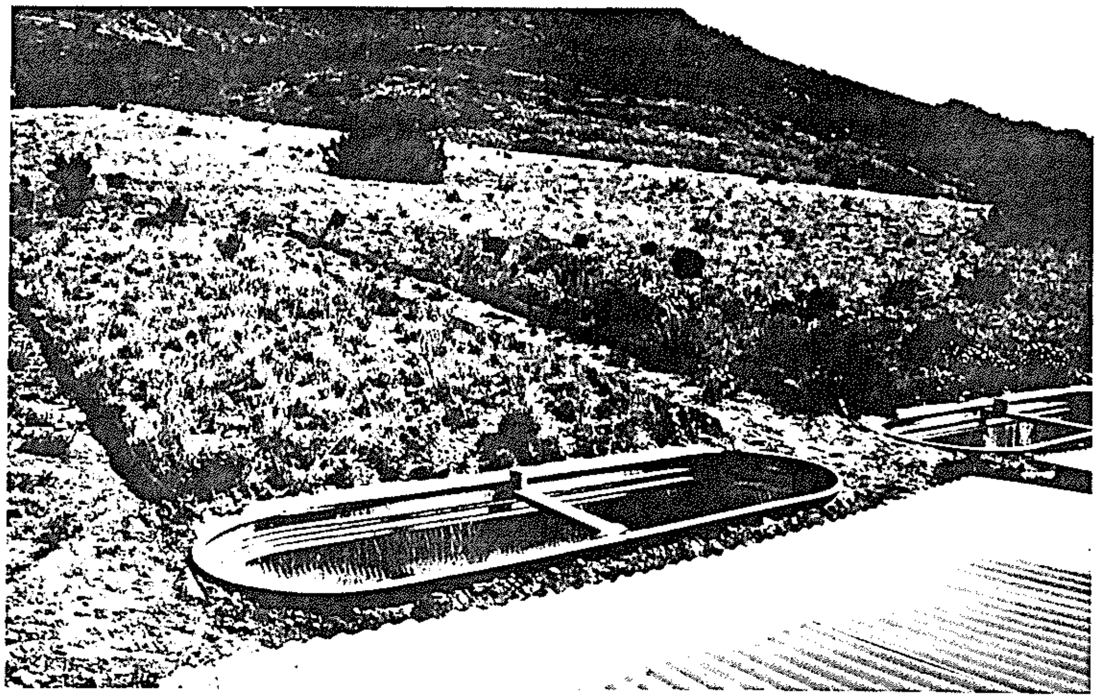

Figure 3. Runoff plots (3 X 10 m) were bordered by aluminum flashing which directed internal runoff into collection tanks at the base of the plots. This photograph was taken in May, 1992. Note the vegetative response to the sludge amendment (treated plot on right).

X 10 m runoff plots. This randomized block design was used to increase the interspersion of treatments throughout the study site (Hurlbert 1984). The larger plots were used for experiments and field activities that would disturb the soil and ground cover and might otherwise influence the surface hydrology of the runoff plots (e.g., root growth estimates and below-ground litter decomposition studies). The sludge amendment on these 10 X 10 m plots was also 45 Mg ha^{-1}.

Sludge Application

The sludge treatment, applied in early April 1991, consisted of a one-time surface application of 45 Mg ha^{-1} (oven dry weight basis) or 20 t $acre^{-1}$ of municipal sewage sludge provided by the City of Albuquerque. The City's 5-year average sludge metal concentrations are in Appendix A, Table 1. The projected metal-loading rates associated with the one-time application at 45 Mg ha^{-1} (Table 2 - Appendix A) were well below current federal limitations for sludge application to non-agricultural lands (U.S. Environmental Protection Agency 1993). Spring precipitation at the SNWR is either in the form of snow or low-intensity rainfall, and the sludge was applied at this time to minimize losses that might occur during high-intensity summer and fall convective thunderstorms. Albuquerque sewage sludge is anaerobically digested and mechanically de-watered to approximately 18% dry solids (82% water). The product is a gelatinous substance that is relatively easy to transport and handle. A front-end loader was used to apply the required quantity of sludge on the treated plots. The sludge was then spread manually within each plot using rakes and shovels to obtain a uniform application (e.g., 45 Mg ha^{-1}

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 sludge. Although sludge handling presented no known health hazard we took precautions, including the use of protective clothing, to prevent direct contact with the sludge.

Precipitation and Runoff Measurements

We measured total precipitation at the study site with two standard rain gauges (rainfall collection buckets). A self-activating recording rain gauge was also installed at the site on August 1, 1991, allowing measurements of rainfall rate (mm hr^{-1}). We collected samples of runoff water, obtained after stirring the runoff in the collection reservoirs to uniformity, in acid-washed plastic bottles following runoff-producing rainfall and immediately after each rainfall simulation experiment. The samples were then transported to the lab in an ice chest for subsequent analyses. We preserved samples tested for total phosphorus (TP) and nitrate (NO_3-N) with concentrated sulfuric acid (2.0 ml H_2SO_4 L^{-1} of water sample) and those tested for metal concentrations were preserved with concentrated nitric acid (5.0 ml HNO_3^--N L^{-1} of water sample). All chemical analyses in the runoff water samples were conducted within 28 days after collection. We collected other runoff water samples for analyses of total sediment, total suspended solids (TSS), total volatile suspended solids (TVSS), TKN, and nitrite-nitrate nitrogen (NO_2-NO_3).

Rainfall Simulation

We used a large rainfall simulator in September 1991 to allow observations of rangeland response to high-intensity rainfall under controlled conditions (Ward 1986). The simulated rainfall experiments would insure the collection of runoff yield data from our plots 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^{-1} for 30 minutes). The rainfall simulator (Fig. 4) was designed to deliver rainfall at approximately the same droplet size and velocity as would a natural storm.

The rainfall simulator, consisting of 15 sprinklers mounted on 3 m tall standpipes, distributed water simultaneously to each runoff plot pair so infiltration and runoff yield could be observed and recorded on the two plots (control and treated) concurrently. At the conclusion of each simulation run, the total amount of runoff was recorded and runoff samples were collected for analysis of total sediment, TSS, TVSS, TP, NO_2-NO_3, and heavy metal contents.

We evaluated the sludge's water absorption capability during the rainfall simulation experiments by placing 30-50 g of air-dry sludge on five 30 cm^2 screen mesh platforms along the plot boundaries. The samples received the same amount of precipitation as the plots during the rainfall simulation experiment. The samples were collected immediately after the rainfall simulation experiments and placed in air-tight containers and compared to five additional sludge samples at field moisture content taken prior to the experiment.

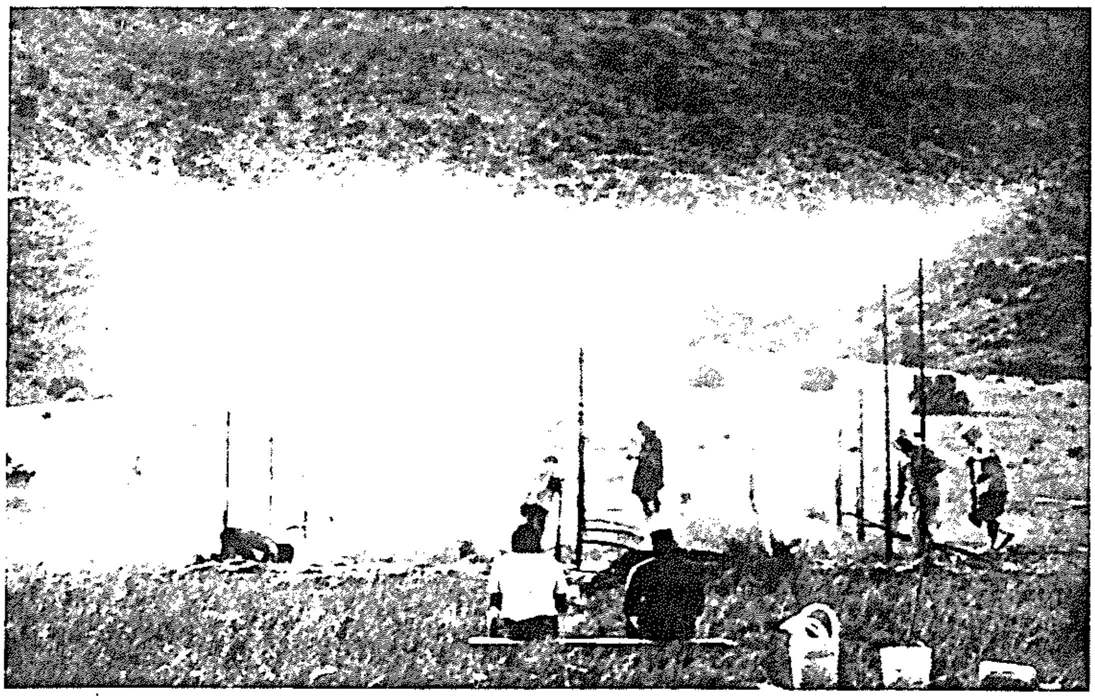

Figure 4. The rainfall simulator consisted of 15 standpipes (3 meters in height) that uniformly distributed rainfall simultaneously to a pair (control and treated) of runoff plots. During the experiments, 40 to 80 mm of precipitation were applied in a 30 minute period and differences in runoff volume and sediment yield were recorded.

Leaching Assessment

In May 1993, six soil pits were excavated to a depth of 1 m from three control and three sludge-treated plots. Soil samples were collected in 10 cm increments and analyzed for NO_3-N and TKN. A detailed description of the soil's pedogenic development was completed.

Soil and Vegetation Analyses

We collected soil samples prior to the sludge application and thereafter twice per year in all 10 X 10 m plots. The first post-application sampling period followed the winter precipitation period (April) and the second sampling period (October) followed the late summer to early fall convective thunderstorm period.

Five composite soil samples were collected from each of 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 consisted of five randomly selected subsamples. Results from other studies (Davis et al. 1988) suggested the depth resolution of this sampling procedure would allow us to adequately trace the vertical movement of specific nutrients and contaminants through the soil profile.

Range quality was based on above- and below-ground plant biomass, forage quality, and the presence/absence of undesirable plant species. Field vegetation characterization (above-ground foliar cover) followed the same general schedule as soil characterization and sampling. Vegetation samples were collected for plant tissue analysis (four composite samples per plot, five subsamples per composite) prior to sludge application in Spring 1991, and again in Fall 1992. Forage quality was determined by analyzing plant tissues of various species for contaminant levels.

Changes in above-ground plant cover were determined using a line-intercept technique developed by SNWR-LTER researchers. We sampled ten, 3 m transects within each runoff plot, for a total of 30 m per plot. A plot bridge was used when transects were recorded from the runoff plots. The plot bridge spanned the 3-meter width of the plots and allowed us to record plant intercepts without having to trample the plants and soil within the plots. We used five, 10 m transects to obtain estimates of vegetation abundance within each 10 X 10 m plot. Transect intercepts for individual plant species and other ground cover categories including plant litter, bare soil, gravel, and sludge were recorded and analyzed.

We estimated below-ground plant response to surface-applied sewage sludge using a root ingrowth technique (Steen 1984, Steen 1991). This technique was used to compare plant root response, not to estimate below-ground plant productivity, within the control and treated plots. We removed 10 soil cores (dimensions 20 X 10 cm) from within each 10 X 10 m plot. The soil from each core was sieved to remove roots and coarse fragments > 2.0 mm in diameter. A nylon mesh bag was then inserted into the hole and back-filled with the sieved soil. The samples were later collected by cutting around the bags to sever any roots that had grown into them and pulling the cores from the ground. The roots were separated from the soil with a 2.0 mm screen and the sieved soil was then placed back into the core-hole. We transported the nylon mesh bag and associated roots to the lab and air-dried them for at least 48 hours. The roots were then separated from the bag and weighed, ground, and ashed. We collected half the root ingrowth bags from the field in Fall 1991 and the second half in Fall 1992.

Treatment effects on above- and below-ground plant litter decomposition were carried out using both blue grama and four-wing saltbush tissue. The New Mexico State University/Soil Conservation Service Plant Materials Center, Los Lunas, NM, donated the blue grama tissue (leaves and stems). Saltbush leaves were collected and dried at room temperature. Five grams of plant material were placed into nylon or fiberglass mesh bags. We placed ten bags of each litter type on the ground surface and ten additional bags were buried 5 cm below-ground in each plot. Four bags from each litter type/position combination were collected immediately to correct for litter losses that may have occurred during transportation and implementation. The remaining 12 litter bags were left in the field for one growing season (spring through fall). At the end of the growing season the litter bags were collected and air-dried. We combined litter samples in the same litter type/position combination to make four samples per plot (blue grama above- and below-ground and saltbush above- and below-ground samples). The samples were ground and ashed to correct for mineral content. We based litter quality on estimates of TKN, ash content, and soluble salt content. This procedure was conducted during both the 1991 and 1992 growing seasons.

Laboratory Analyses

For the most part, NMSU's Soil, Water, and Air Testing Laboratory completed the soil, plant, and water analyses. The Forest Sciences Laboratory, USDA Forest Service, Albuquerque conducted additional analyses.

NO_3^- and NO_2^- in runoff water samples were determined by flow injection analysis according to the colorimetric Cd-reduction method. Concentrations of Cu, Cd, and Pb were determined by inductively coupled plasma emission spectroscopy (ICP) (Soltanpour et al. 1982). Total sediment (g) was estimated from 1.0 L runoff samples. The sediment in the water sample was allowed to settle out of suspension through time (generally 7-10 days). We then siphoned most of the water from the jar and oven-dried the remaining sample at 105° C and weighed the dried sediment. The TSS were determined by first filtering a sample of the runoff through a standard glass fiber filter, then evaporating the water from the filtrate at 105° C and weighing the sediment. We determined TVSS by ashing the TSS sediment in a muffle furnace at 550° C, then reweighing the sediment sample (e.g., TVSS = TSS - TSS (ashed)). U.S. Environmental Protection Agency (1983, 1986) publications describe all methods employed for sediment and chemical analyses of runoff samples.

Soil samples were analyzed for pH, EC, OC, TKN, KCl extractable NH_4-N and NO_3-N, $NaHCO_3$ extractable P and extractable heavy metals (Cd, Cu, Pb, Zn). In addition, we conducted a textural analysis of the soils during the first sampling period. Saturated pastes were used for EC and pH measurements. Organic-C contents were determined by chromic acid oxidation using the modified Walkley-Black method described by Nelson and Sommers (1975). We determined total nitrogen on Kjeldahl digests (Bremner and Mulvaney 1982). Labile, potentially plant-available, metal contents were determined by ICP from extracts obtained using the diethylenetriaminepentaacetic acid (DTPA) method developed by Lindsay and Norvell (1978). We used $NaHCO_3$ extractable P following the methodology outlined by Olsen and Sommers (1982) as an index of plant-available P in alkaline soils. The KCl extractable NH_4-N and NO_3-N were determined colorimetrically following the procedures outlined by Bremner (1965).

Plant tissue samples were oven dried at 80° C for 48 hours and ground for subsequent analysis at the NMSU Soil, Water, and Air Testing Laboratory. They were analyzed for total N using the Kjeldahl nitrogen method (Bremner and Mulvaney 1982). The DTPA extractable metal contents (Lindsay and Norvell 1978) in the plant tissues were determined by ICP analysis.

Statistical Analysis

Analysis of variance (ANOVA) techniques were used to test for statistically significant differences between treated and control plots in: 1) plant cover, root growth, and plant litter decomposition, 2) runoff and sediment yield, and 3) trace elements in water, soil, and vegetation. Repeated measures analysis of variance (Statistical Package for the Social Sciences) was used to test for significant main factors (treatment, time and depth), significant interactions, normally distributed data, and homoscedasticity (homogeneous variances). If the data were normally distributed and had homogeneous variance, we proceeded with a one-way ANOVA followed by a Tukey's multiple comparison test. If we encountered problems with non-normality or heteroscedasticity we relied on the Pillai's statistic which is "more robust" to these problems than the F statistic. We then proceeded with a one-way ANOVA and the appropriate multiple comparison test. Unless otherwise noted, we adapted a type I error rate of $\alpha = 0.05$ for all statistical analyses.

RESULTS AND DISCUSSION

SLUDGE EFFECTS ON SOILS

Soil Physical Properties

Table 2 presents the particle-size distribution and bulk densities of representative soil samples from the study area. The textural classification for all three depths was sandy loam.

Table 2. Particle-size distribution and bulk density of SNWR study site soils as a function of depth.

Depth	Sand	Silt	Clay	B.D.
(cm)	------------------	%	------------------	($g\ cm^{-3}$)
0-5	67	27	6	1.61
5-10	63	26	11	1.38
10-15	59	26	15	1.31

Table 3 compares the chemical composition of the sewage sludge with composite soil samples collected from the upper 15 cm within the study site. The C:N ratio of the soil and sludge was 9.45 and 7.67, respectively. Heavy metal concentrations in the soil are two to three orders of magnitude below the concentrations in the sludge. Table 3 also lists the estimated (calculated) levels of OC, TKN, P, Cu, Pb, and Zn that would be present in the soil if the sludge were directly incorporated into the top 15 cm of soil. We based phosphorus and heavy metal estimates on extractable forms, thus, the values reported do not represent total amounts. The potential "plant available" levels of these constituents could be greater if non-extractable forms in the sludge were transformed to extractable forms in the soil.

Table 3 also lists soil and sludge nutrient and heavy metal contents, based upon mass per unit area (kg ha^{-1}), with the sludge contribution based upon an application of 45 Mg ha^{-1}. The EPA (1993) loading limits for Cu, Pb, and Zn are listed for comparison with totals in the soil and sludge. The cumulative totals (soil + sludge) remain well below the U.S. Environmental Protection Agency pollutant loading limits.

Soil Organic Carbon

No significant differences in soil OC occurred during the course of this study between the treatments or among sampling periods (Fig. 5). Surface-applied sludge will unlikely increase the organic matter content of the underlying mineral soil as has been demonstrated in projects where sludge was directly incorporated into the soil (Khaleel et al. 1981; Mitchell et al. 1978).

Table 3. OC, TKN, extractable P, and heavy metals in SNWR study site soils and Albuquerque sludge.

	OC	TKN	P	Cu	Pb	Zn
	---------- % ----------		------------------------ mg kg^{-1} ------------------------			
Soil	0.832	0.088	3.6	2.01	0.81	0.47
Sludge	20.1	2.62	23,700	635	232	921
Combined[1]	1.25	0.14	500	15.29	5.67	19.7
	------------------------ kg ha^{-1} ------------------------					
Soil[1]	17,888	1,892	7.74	4.32	1.74	1.01
Sludge[2]	9,045	1,178	1,066	28.57	10.44	41.44
EPA Limits[3]	----	----	---	1,500	300	2,800

[1] Calculations based upon the upper 15 cm of soil.
[2] Estimates based upon an application of 45 Mg ha^{-1}.
[3] EPA Cumulative Pollutant Loading Rates.

Soil Nitrogen

Sewage sludge application significantly increased soil NH_4-N contents in treated plots at all three depths (Fig. 6). As a result of the relative immobility of NH_4-N, the greatest accumulations occurred in the 0-5 cm depth on the treated plots.

The N dynamics on the treated plots exhibited patterns similar to those reported for other disturbed systems (Kelling et al. 1977, Vitousek et al. 1982). The increase in soil NH_4-N in Fall 1991 likely resulted from free NH_4^+ present in the sludge and from rapid

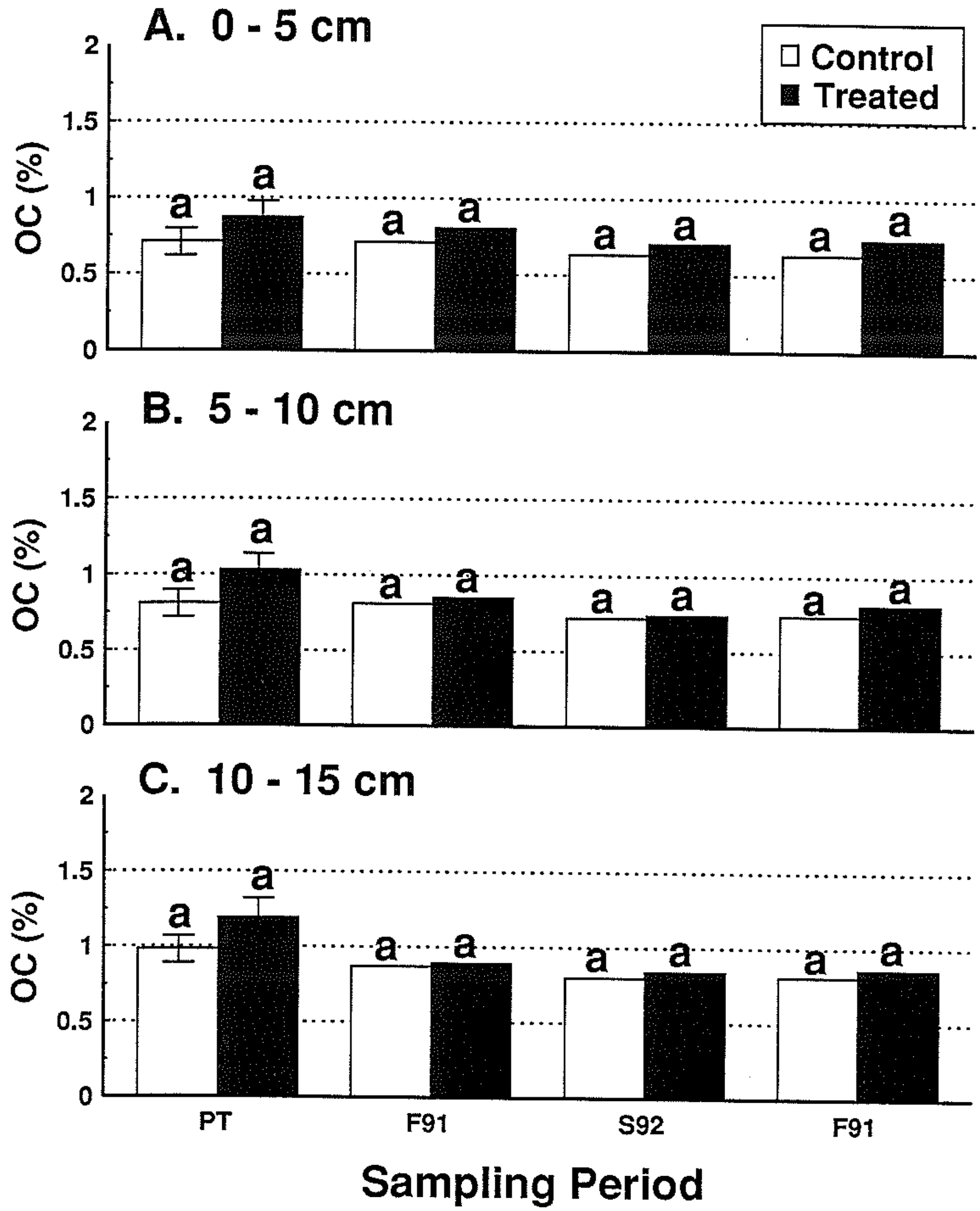

Figure 5. Soil OC (at three depths) in control and sludge-treated plots. Soils were sampled at three depths, 0-5, 5-10, and 10-15 cm in four sampling periods, pretreatment (PT), Fall 1991 (F91), Spring 1992 (S92), and Fall 1992 (F92). Values reported are means with standard errors. Bars, within a depth, with the same letter are not significantly different at the p = 0.05 level.

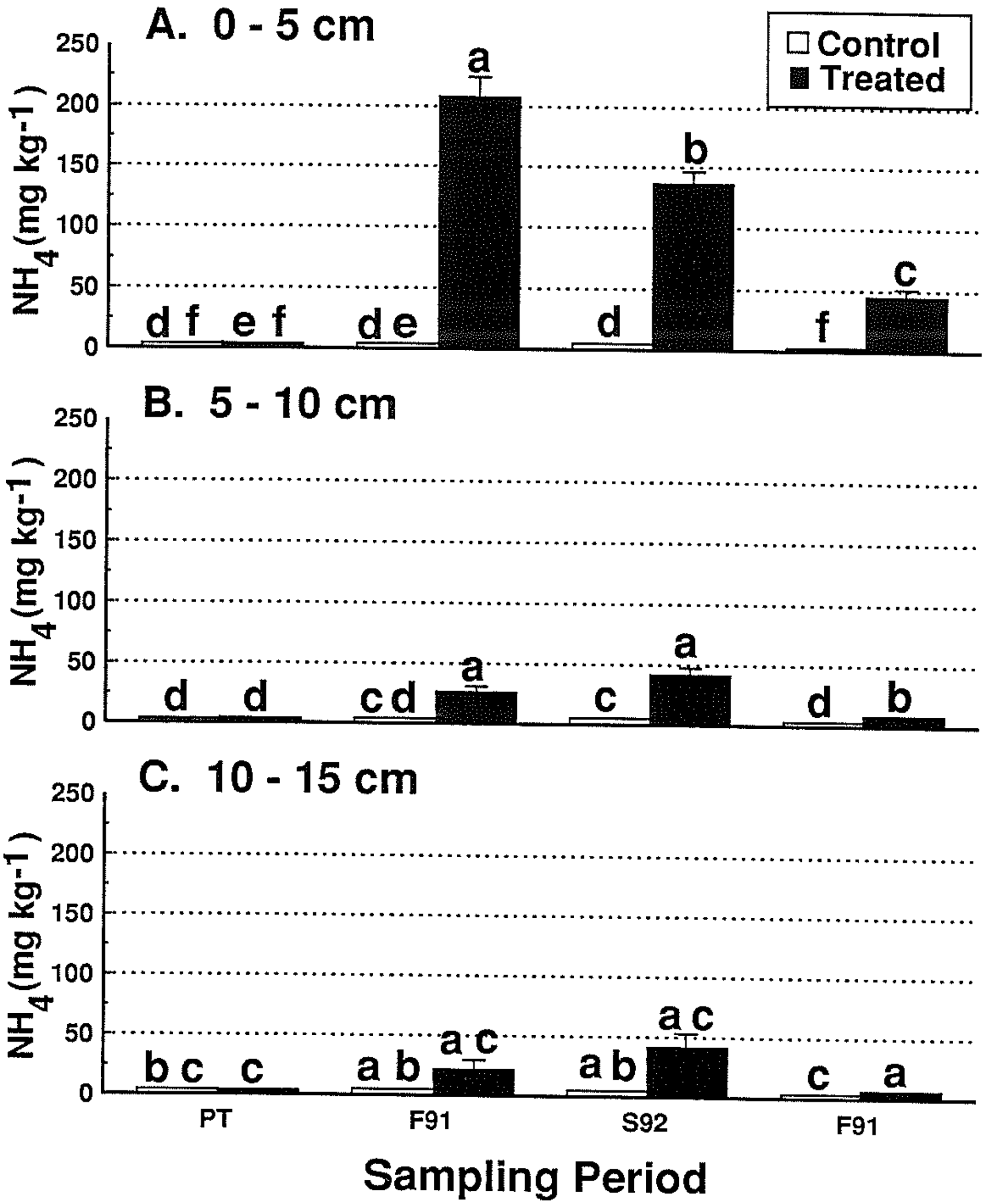

Figure 6. KCl extractable soil NH_4-N (at three depths) in control and sludge-treated plots. Soils were sampled at three depths, 0-5, 5-10, and 10-15 cm in four sampling periods, pretreatment (PT), Fall 1991 (F91), Spring 1992 (S92), and Fall 1992 (F92). Values reported are means with standard errors. Bars, within a depth, with the same letter are not significantly different at the p = 0.05 level.

mineralization of N from readily decomposable organic compounds. In our study the main pulse of NH_4^+ input had moved through the treated soils by Fall 1992 and we anticipate that levels will revert to normal (control) levels by Fall 1993. Sommers et al. (1976), in a survey of eight sludge types from Indiana, found that inorganic N content averaged 1.94%, with 90% of the inorganic N present in the form of NH_4-N. However, most of the N in sludges is in organic form and 15% to 50% of this is readily mineralized within weeks of application (Epstein et al. 1978, Kelling et al. 1977, Magdoff and Amadon 1980, Parker and Sommers 1983, Ryan et al. 1973, Serna and Pomares 1992).

Sludge treatment significantly increased soil NO_3-N levels at all depths and sampling periods (Fig. 7) Nitrate is produced through the chemoautotrophic oxidation of NH_4^+. The time required for this process to occur is reflected in the time lag observed between the major NH_4-N pulse/peak which occurred in Fall 1991, and the first major NO_3-N peak that occurred in Spring 1992. Nitrate is a negatively charged ion and is very mobile in soils. Nitrate that is not assimilated by plants or microorganisms is therefore readily leached as water infiltrates the soil. Our data clearly indicate that a pulse of sludge-derived NO_3-N has moved through the top 15 cm of treated soil. Through time, NO_3-N levels declined in the upper 0-5 cm and concurrently increased in the lower depths. Additional NO_3-N measurements we collected from soil cores taken from the treated plots revealed that, although the highest concentrations (133 mg kg^{-1}) occur between 0 - 40 cm, elevated levels (25 mg kg^{-1}) were detectable to a depth of 80 cm (Fig. 9, page xx).

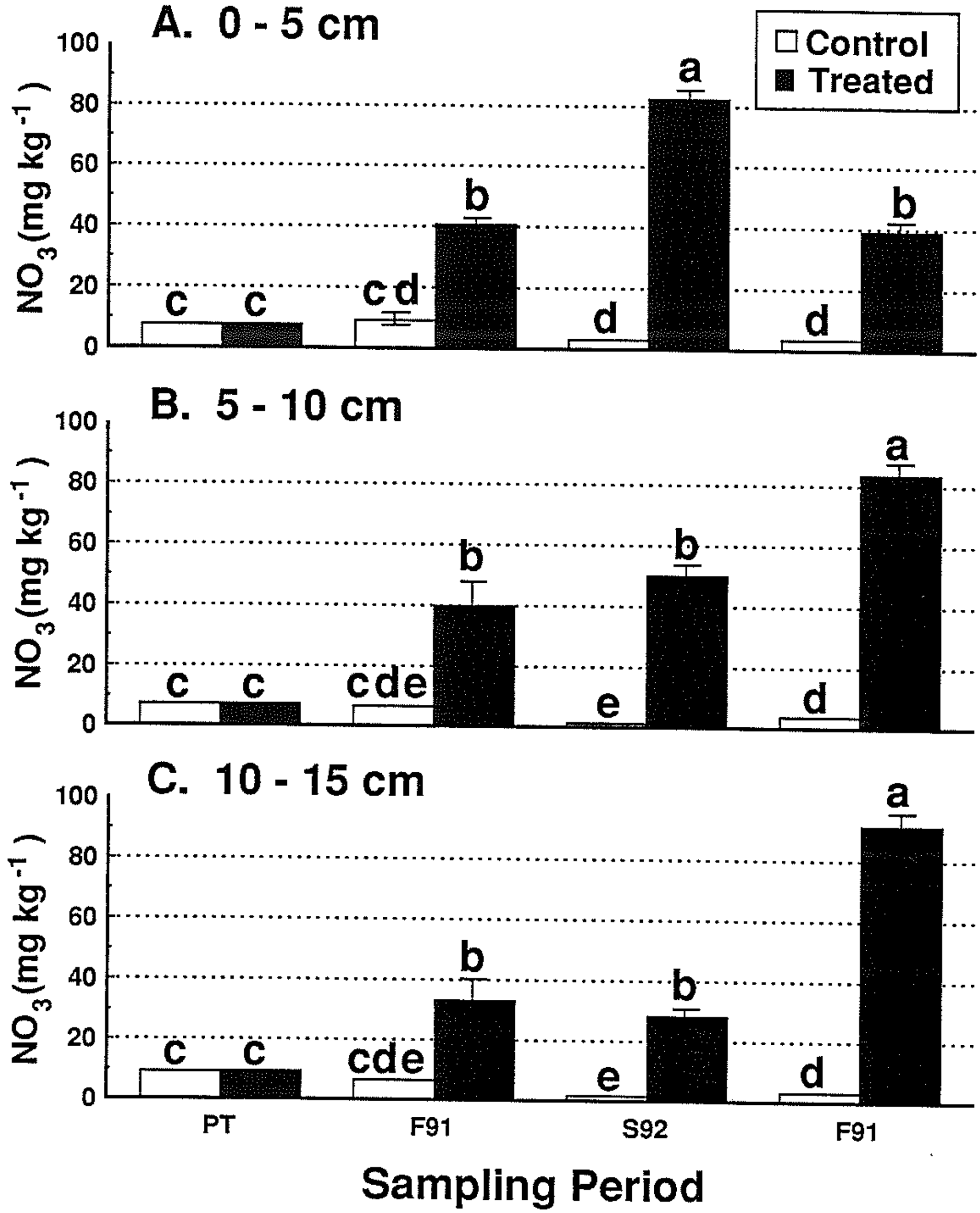

Figure 7. KCl extractable soil NO_3-N (at three depths) in control and sludge-treated plots. Soils were sampled at three depths, 0-5, 5-10, and 10-15 cm in four sampling periods, pretreatment (PT), Fall 1991 (F91), Spring 1992 (S92), and Fall 1992 (F92). Values reported are means with standard errors. Bars, within a depth, with the same letter are not significantly different at the p = 0.05 level.

We observed no significant changes in soil TKN in the control soils during the course of the study (Fig. 8). Sludge application significantly increased soil TKN only at the 0-5 cm depth, and this increase remained significant on the final sampling period.

Figures 9 and 10 show the Spring (May) 1993 distribution of soil NO_3-N and TKN, respectively, to a depth of 1 m. Nitrate levels from treated plots are significantly greater than control plots at all depths except 0-10 and 30-40 cm depths (Fig. 9). Comparisons of TKN and the NO_3-N distributions with depth indicate that the increases in treated plot NO_3-N have little or no effect on the total N content in the soils. Total N concentrations remained essentially the same at all depths in the control and treated soils despite the sludge amendment.

The movement of NO_3^- to lower soil depths may represent N losses from the soil, however, the likelihood of NO_3^- actually leaching entirely out of the soil profile and vadose zone to underlying groundwater resources is small (Wight and Black 1979; Woodmansee 1978). The presence of a caliche layer beginning at a depth of 70 cm represents the long-term average depth of downward water movement in the soil, and ground water at the site is approximately 30 m below the surface. Nitrate that accumulates within the caliche layer may serve as a N reservoir for deep rooted plants.

Soil Phosphorus

Extractable P levels in control plots remained consistent throughout the duration of the study (Fig. 11). In general, P levels were highest at the surface and decreased with depth for both control and treated plots. The high levels of P in the sludge produced

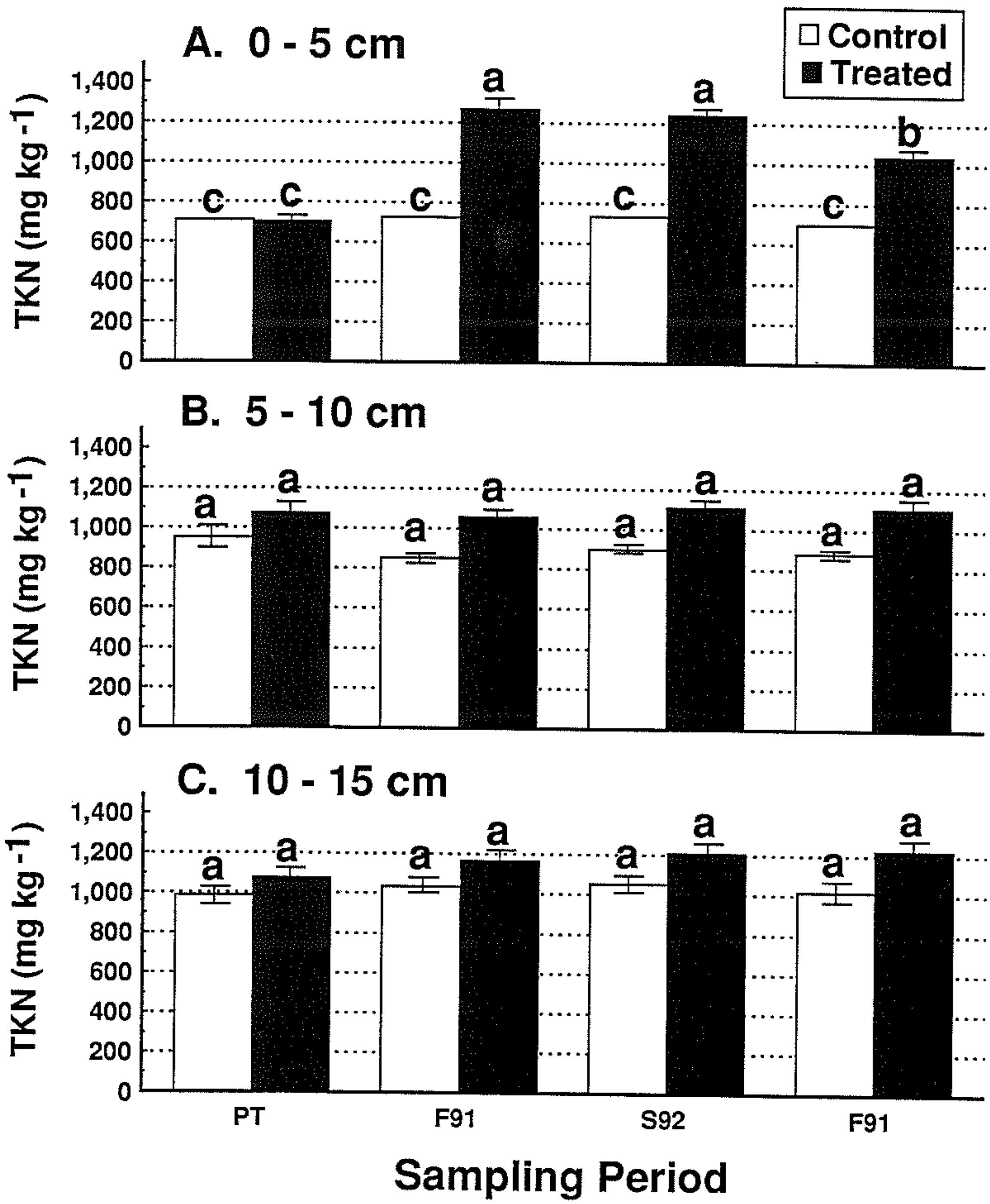

Figure 8. Total Kjeldahl nitrogen (at three depths) in soils from control and sludge-treated plots. Soils were sampled at three depths, 0-5, 5-10, and 10-15 cm in four sampling periods, pretreatment (PT), Fall 1991 (F91), Spring 1992 (S92), and Fall 1992 (F92). Values reported are means with standard errors. Bars, within a depth, with the same letter are not significantly different at the $p = 0.05$ level.

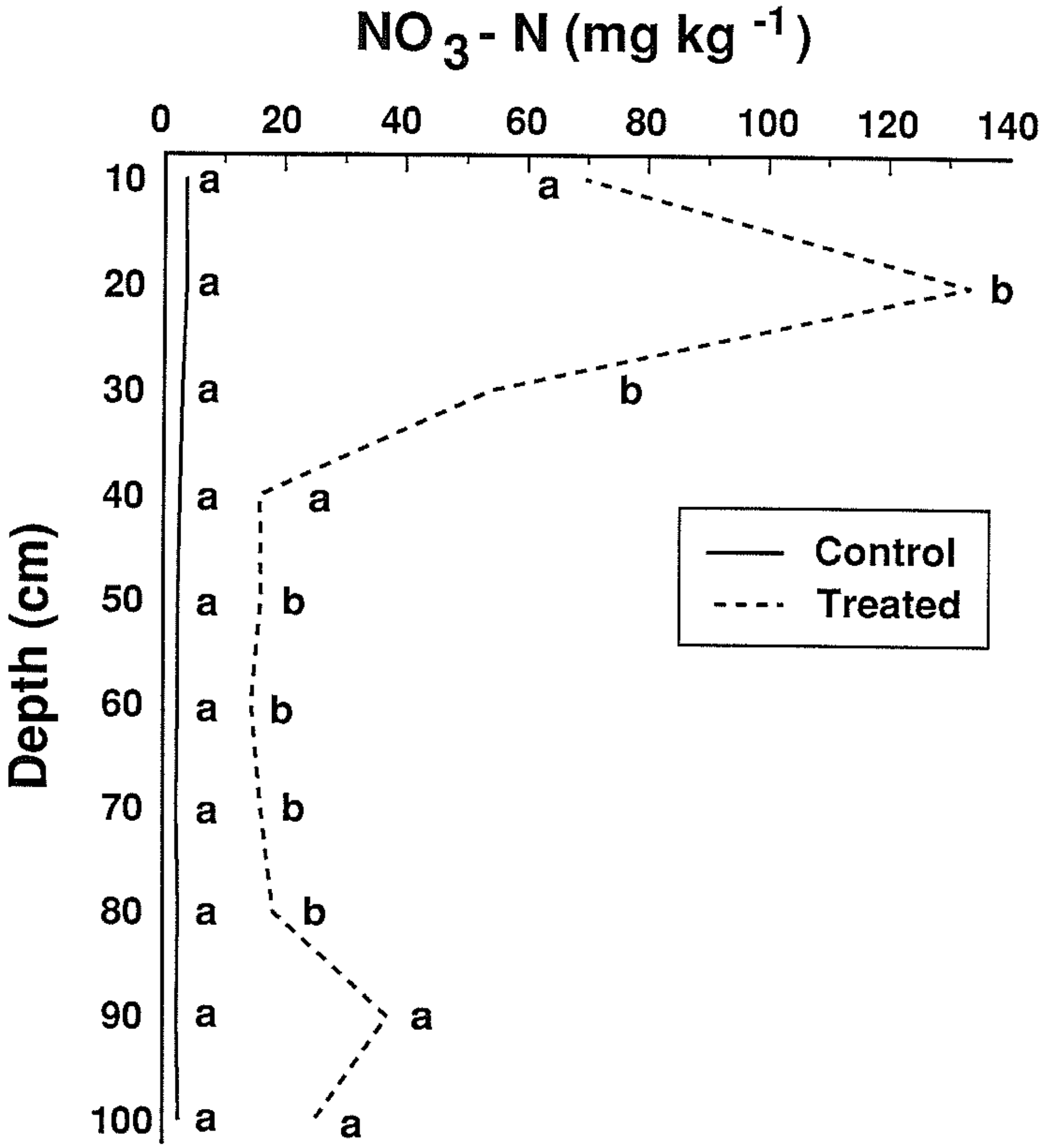

Figure 9. KCl extractable soil NO_3-N distribution to 1.0 m for control and sludge-treated plots. Points within a depth with the same letter are not significantly different at the $p = 0.05$ level.

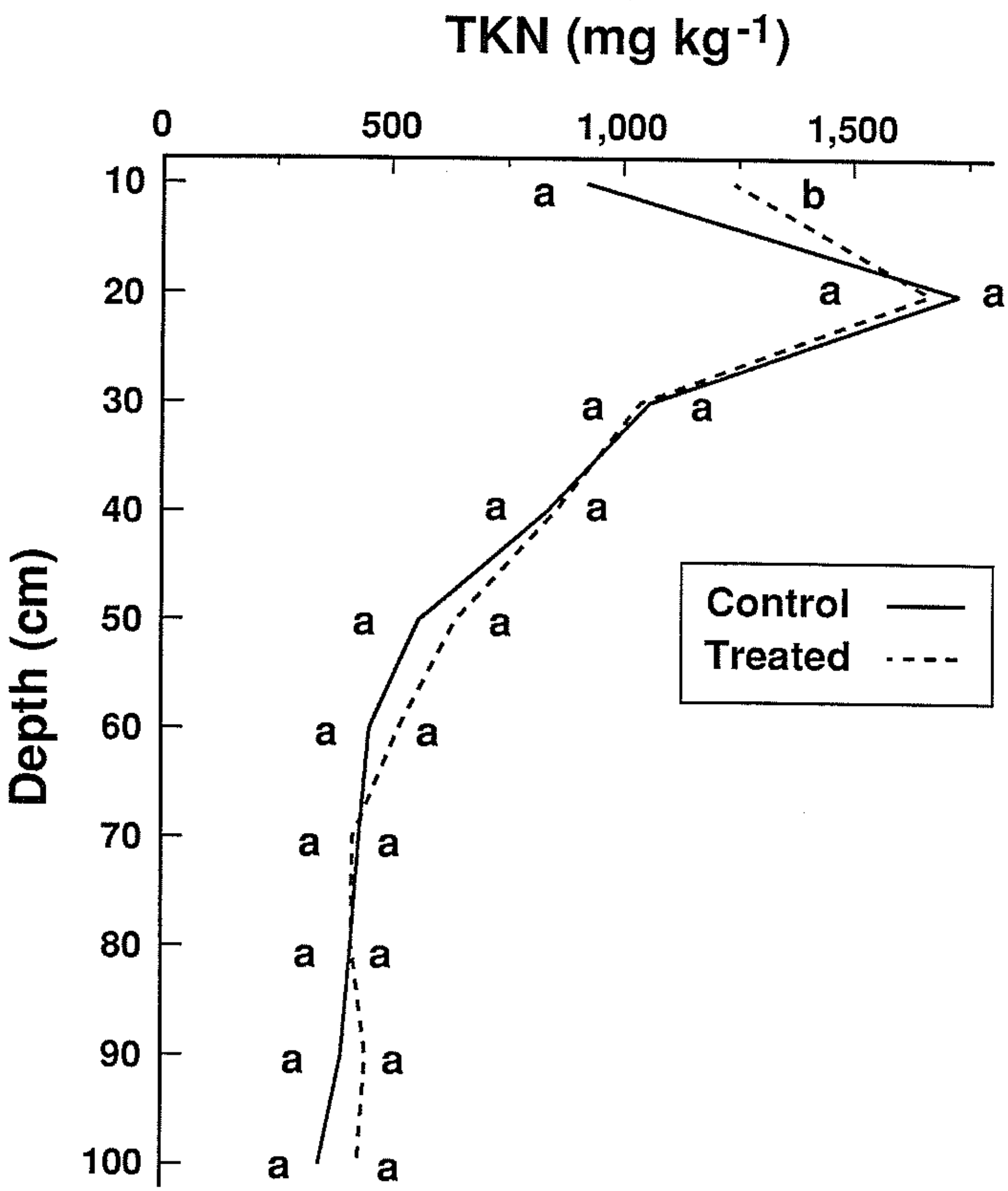

Figure 10. Total Kjeldahl nitrogen distribution to 1.0 m for soils from control and sludge-treated plots. Points within a depth with the same letter are not significantly different at the p = 0.05 level.

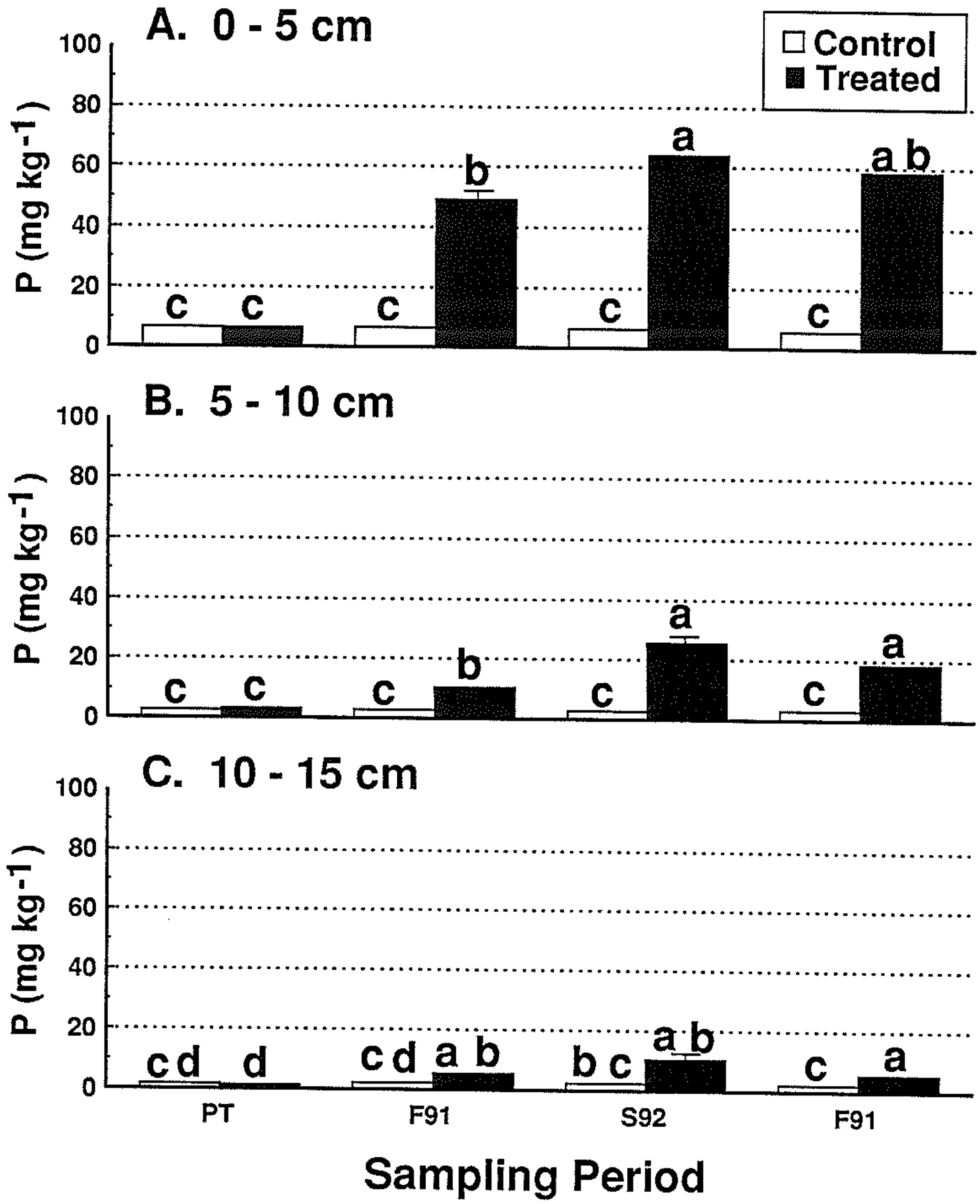

Figure 11. $NaHCO_3$ extractable soil P (at three depths) in control and sludge-treated plots. Soils were sampled at three depths, 0-5, 5-10, and 10-15 cm in four sampling periods, pretreatment (PT), Fall 1991 (F91), Spring 1992 (S92), and Fall 1992 (F92). Values reported are means with standard errors. Bars, within a depth, with the same letter are not significantly different at the $p = 0.05$ level.

significantly greater levels of extractable P in the treated soils. Treated soil P contents were significantly greater than those in control soils for all depths and sampling periods, except at the 10-15 cm depth interval in Spring 1992.

Phosphorus is relatively immobile in soils and should not leach. Kelling et al. (1977) reported significant amounts of extractable P in soils for at least two years following sludge application. Kardos and Hook (1976) reported that no more than three percent of the total P was lost from soils that received sewage effluent applications for 9 to 11 years. Precipitation with Ca, Fe, and Al cations in soils can remove labile P (Soon et al. 1978). Long-term monitoring of these treated soils will be required to determine the magnitude and persistence of sludge-induced extractable P levels.

Soil Electrical Conductivity

A potential negative side-effect of sludge application to rangeland is soluble salt accumulation. Increased soil salt concentrations increase the water stress on plants and other soil organisms. Electrical conductivity is used to indirectly estimate soluble salt levels in soils.

Sludge treatment significantly increased soil EC at all depths and sampling periods except in the 5-10 cm depth in Fall 1991 (Fig. 12). The data indicate the pulse of soluble salts has been leached out of the 0-5 cm depth and into the lower soil depths. Salts will likely continue leaching through the soil until they are either assimilated and immobilized, or are re-precipitated within or below the existing caliche layer. The negative effects, if any, of the increased soluble salt contents released from the sludge should be short-lived

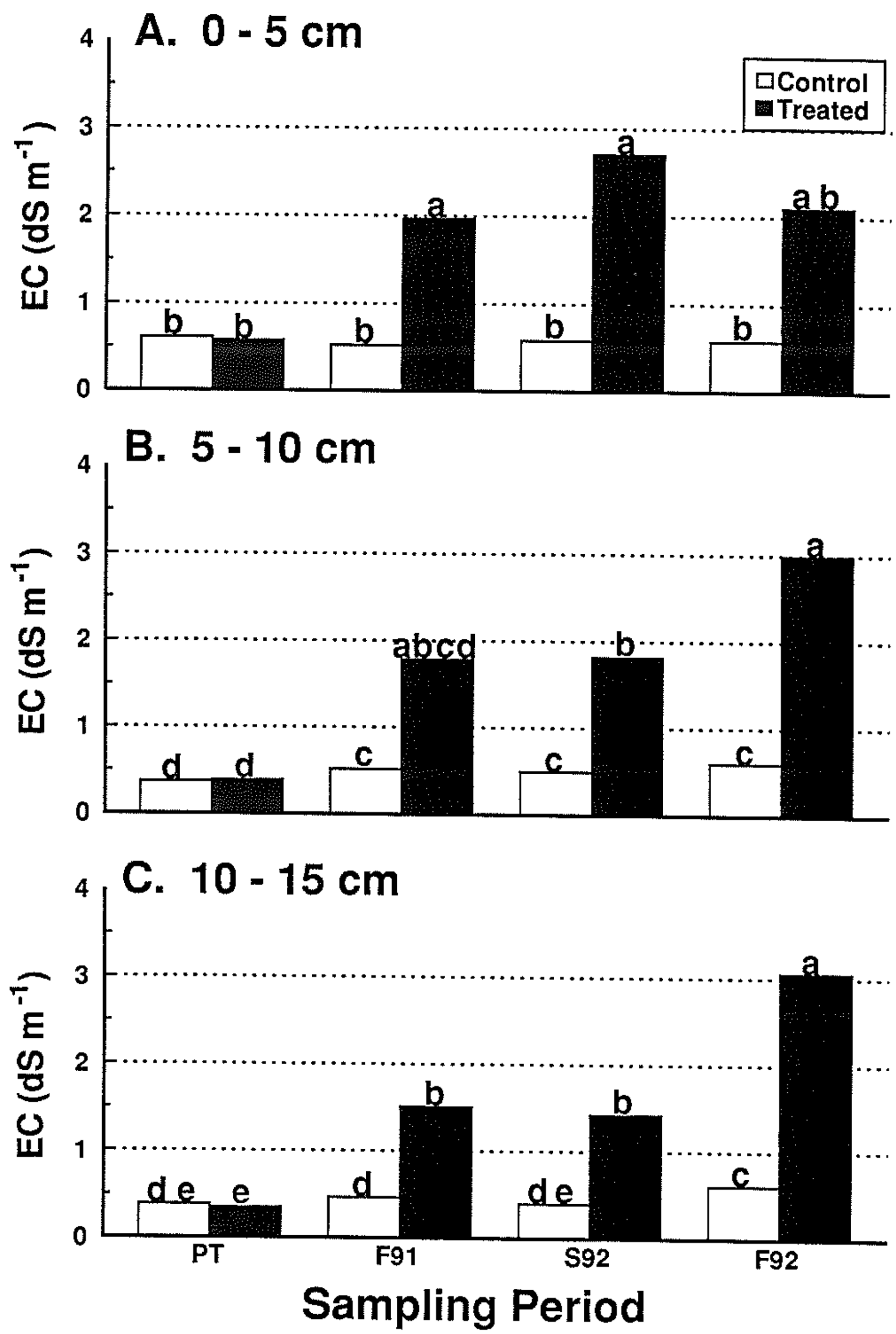

Figure 12. Soil electrical conductivity (at three depths) in control and sludge-treated plots. Soils were sampled at three depths, 0-5, 5-10, and 10-15 cm in four sampling periods, pretreatment (PT), Fall 1991 (F91), Spring 1992 (S92), and Fall 1992 (F92). Values reported are means with standard errors. Bars, within a depth, with the same letter are not significantly different at the p = 0.05 level.

since the salts appear to be leaching to lower depths below the zone of major plant root activity in these relatively coarse-textured soils.

Soil pH

Soil pH is an important factor influencing heavy metal mobility in soils. Addition of sewage sludge can decrease soil pH and can therefore increase heavy metal solubility and bioavailability in treated areas. Sludge treatment significantly decreased soil pH in the last two sampling periods (Fig. 13). Soil pH in the treated plots remained above 7.0 thus, sludge amendment should not have produced a significant increase in heavy metal solubility or availability. The decrease in soil pH may have been the result of several interacting processes, including the direct effect of sludge itself (pH = 6.5), leaching of organic acids, sludge decomposition, nitrification of NH_4^+, and carbonic acid produced through microbial activity during decomposition. These processes should decrease with time as the sludge decomposes. Soil pH should return to control levels as soil carbonates neutralize the sludge-induced acidity.

Soil Heavy Metals

One of the primary concerns of sewage sludge application to land is the accumulation of sludge-borne metals in soils and plants. Heavy metals in sludge can accumulate in soils to phytotoxic levels under certain conditions. Subsequent accumulation of heavy metals in plant tissues can result in biomagnification and potential toxicity within consumer organisms. Four heavy metals, (Cd, Cu, Pb, and Zn), were

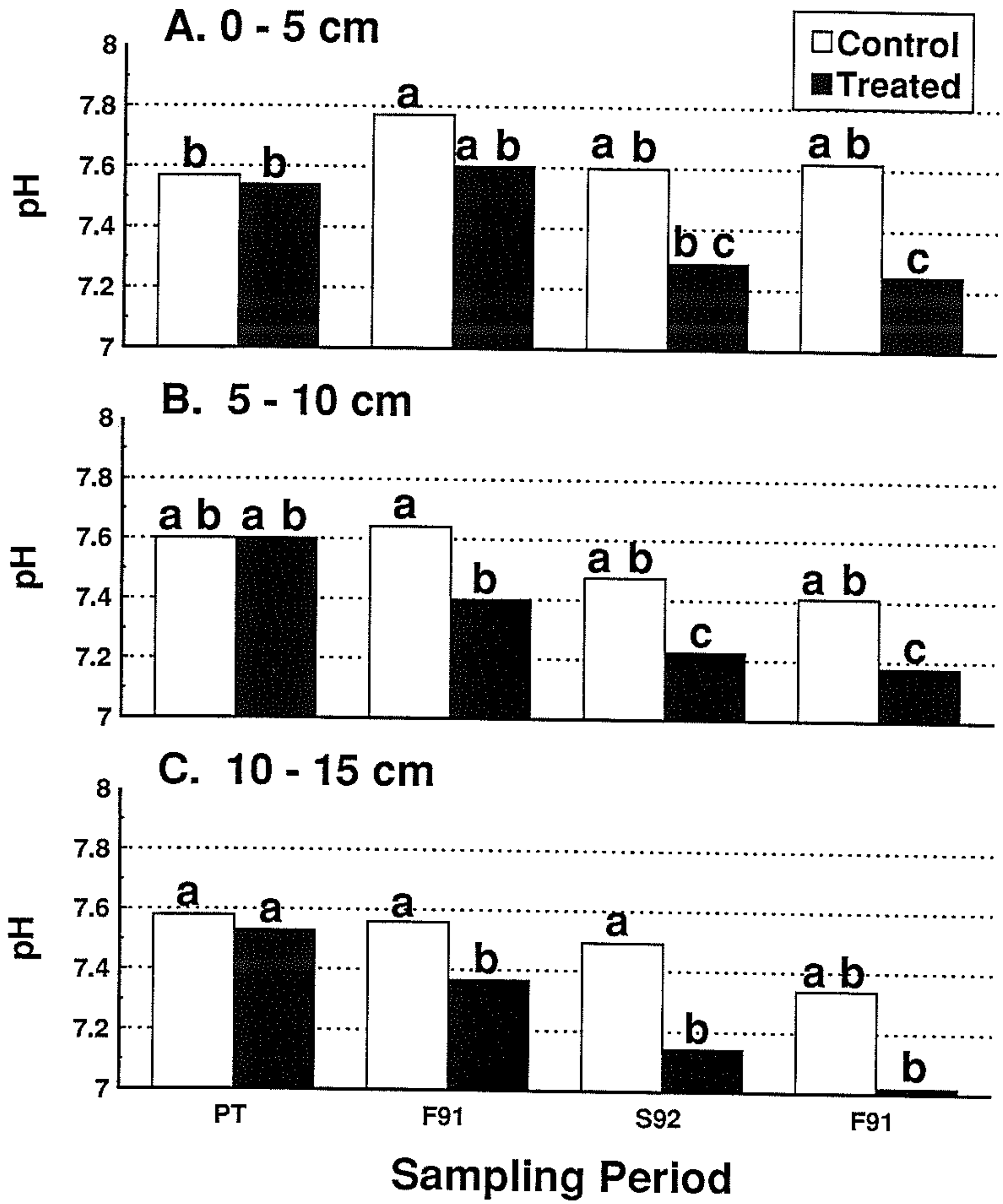

Figure 13. Soil pH (at three depths) in control and sludge-treated plots. Soils were sampled at three depths, 0-5, 5-10, and 10-15 cm in four sampling periods, pretreatment (PT), Fall 1991 (F91), Spring 1992 (S92), and Fall 1992 (F92). Values reported are means with standard errors. Bars, within a depth, with the same letter are not significantly different at the p = 0.05 level.

chosen for analysis in this study, because of their potential toxicity and availability to plants (Williams et al. 1980). Previous research on New Mexico rangeland using Albuquerque sewage sludge indicated that Cu may be the most problematic because of its relatively high concentration in sludge and potentially phytotoxic effects (Fresquez et al. 1991).

Soil Cd levels were below detection limits (<0.02 mg kg^{-1}) throughout the course of the study. When Cd leached from sludge into the soil it was probably diluted below the point of detection. Sewage sludge significantly increased Pb content only in the surface depth in Spring and Fall 1992 (Fig. 14). Lead should remain at very low levels in the surface soil because of low concentrations in Albuquerque sludge couple with inherently low solubility of Pb in soils.

Significantly higher levels of soil extractable Zn were observed in sludge-treated plots than in the control plots for all post-treatment time periods in the 0-5 cm and 5-10 cm depths and during Spring 1992 in the 10-15 cm depth (Fig. 15). The elevated surface concentrations of Zn when compared to the two lower depths in the control plots may also be the result of atmospheric inputs. We anticipated elevated levels of Zn in the treated plots because of high sludge Zn concentrations and the metal's relatively high solubility. Surface Zn concentrations on treated plots reached a maximum of 2.0 mg kg^{-1} (0.167 g/.05 m^3) in Fall 1991 and Spring 1992.

Soil extractable Cu was significantly greater in the sludge-treated plots than in control plots for all sampling periods at the 0-5 and 5-10 cm depths (Fig. 16). High Cu concentrations in pre-treatment soils (especially at lower depths) in all plots may be the

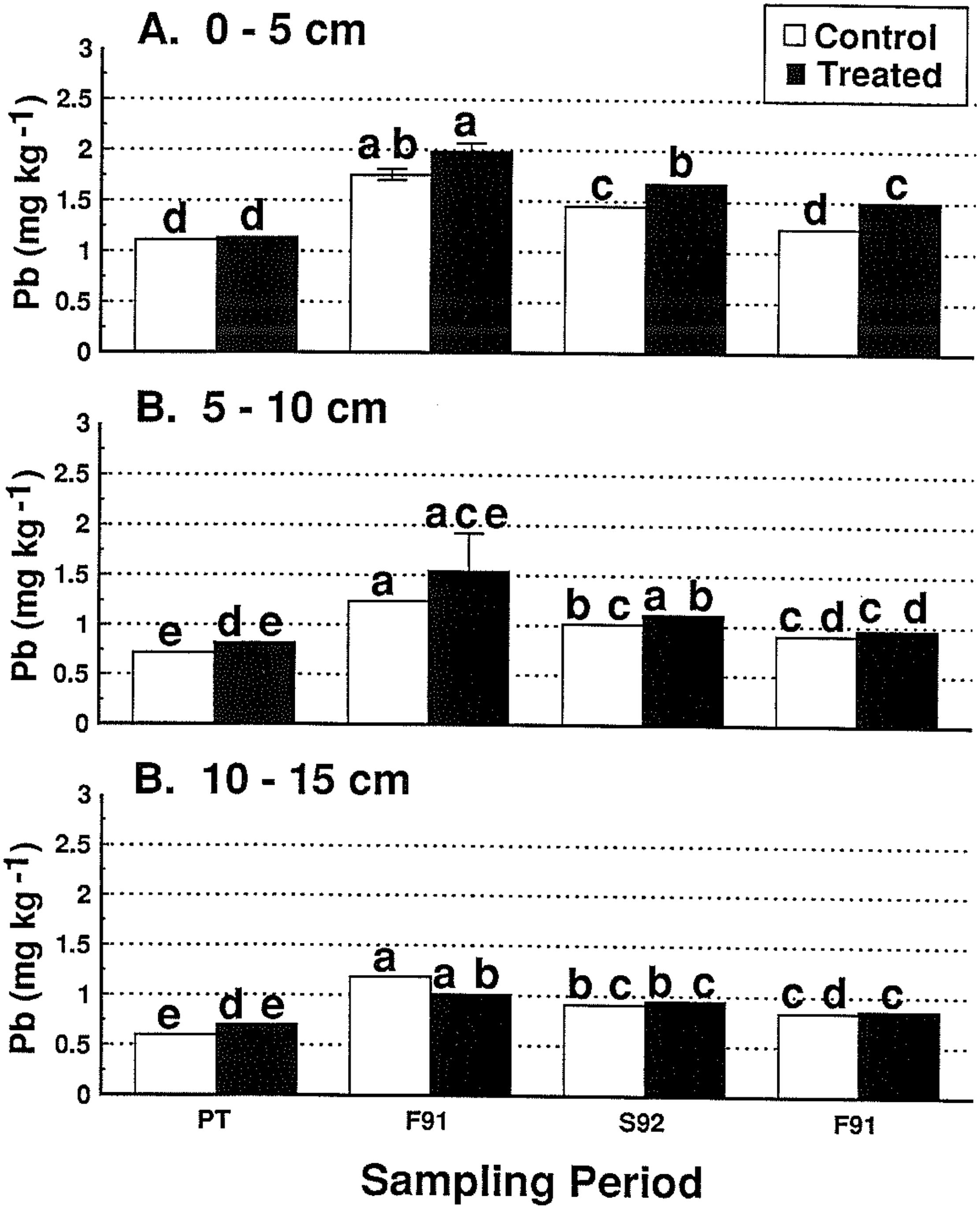

Figure 14. DTPA extractable soil Pb (at three depths) in control and sludge-treated plots. Soils were sampled at three depths, 0-5, 5-10, and 10-15 cm in four sampling periods, pretreatment (PT), Fall 1991 (F91), Spring 1992 (S92), and Fall 1992 (F92). Values reported are means with standard errors. Bars, within a depth, with the same letter are not significantly different at the p = 0.05 level.

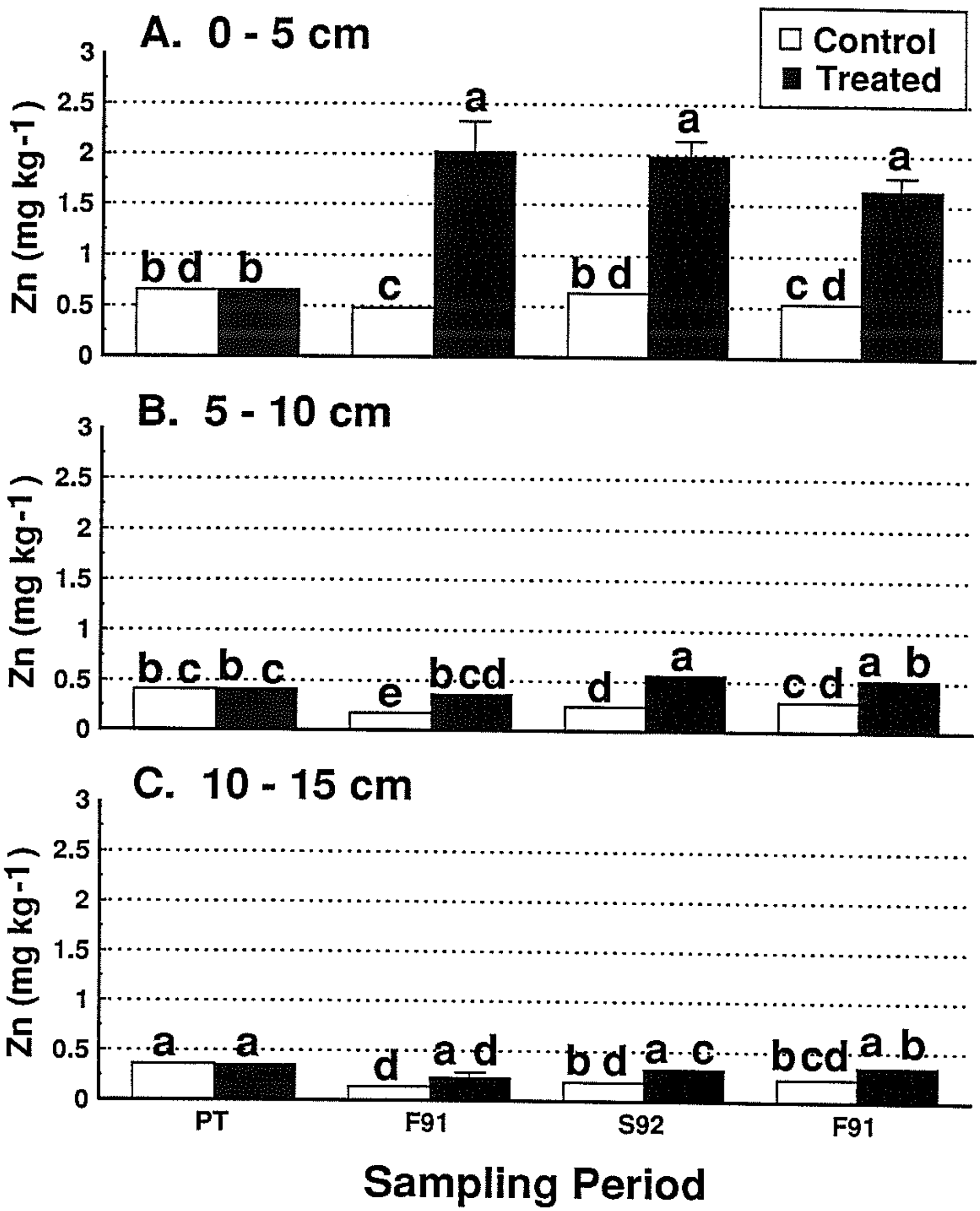

Figure 15. DTPA extractable soil Zn (at three depths) in control and sludge-treated plots. Soils were sampled at three depths, 0-5, 5-10, and 10-15 cm in four sampling periods, pretreatment (PT), Fall 1991 (F91), Spring 1992 (S92), and Fall 1992 (F92). Values reported are means with standard errors. Bars, within a depth, with the same letter are not significantly different at the p = 0.05 level.

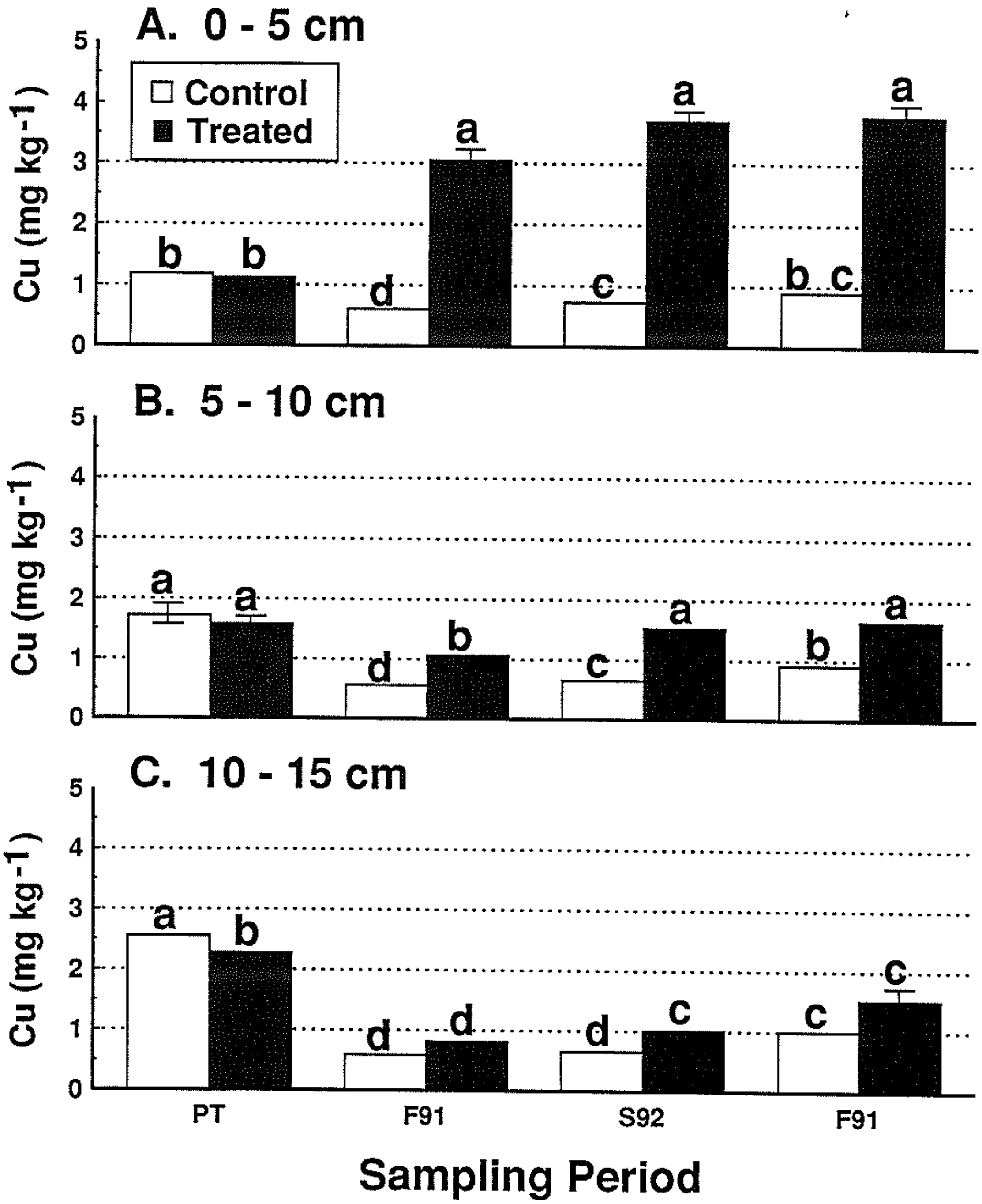

Figure 16. DTPA extractable soil Cu (at three depths) in control and sludge-treated plots. Soils were sampled at three depths, 0-5, 5-10, and 10-15 cm in four sampling periods, pretreatment (PT), Fall 1991 (F91), Spring 1992 (S92), and Fall 1992 (F92). Values reported are means with standard errors. Bars, within a depth, with the same letter are not significantly different at the $p = 0.05$ level.

result of analytical error. New Mexico State's analytical lab was experiencing trouble with the DTPA extraction of Cd at this time and there may have been a problem with the Cu extractions which the lab did not detect. The sludge-induced increases in soil extractable Cu, despite the relative insolubility of soil Cu, may be explained, in part, by the movement of sludge-borne soluble Cu-organic complexes (Baham and Sposito 1983) or the movement of Cu-carbonate precipitates (Brookins 1988).

In general, our study results compare favorably with results from other research on metal mobility in soils and sludges. Only a very small fraction of soil metals, residual or introduced, are found in the soluble or exchangeable pools. Sposito et al. (1982) determined that less than 5% of the total soil metal content would be available for plant uptake. Although significant increases in Cu, Pb, and Zn were recorded following the sludge application, the total heavy metal loading in the treated soils was well below current or established EPA limits (Table 3, page 26). Heavy metal accumulation in rangeland soils to the extent that these metals might be taken up in unacceptable quantities by plants should not occur with sewage sludge applications in the range of 45 Mg ha^{-1} as long as these inputs do not exceed the soil's fixing capability and/or soil pH is not lowered to the point where the metals will solubilize and become available for plant uptake.

SLUDGE EFFECTS ON VEGETATION

Above-ground Plant Cover

We collected above-ground plant cover data from both the 3 X 10 m runoff plots and the 10 X 10 m large plots. The two plot types were analyzed separately because the large plots had been disturbed more during sample collection and because the runoff plots received an additional 50 mm of rainfall during a rainfall simulation study. Vegetative response was similar on both plot types despite these differences.

Prior to the sludge application (PT sampling period), we observed no significant differences in above-ground plant cover between control and treated 10 X 10 m plots (Fig. 17). Sewage sludge application had no significant effect on above-ground plant cover on these plots. No significant differences were recorded between control plot cover with time on the 3 X 10 m plots (Fig. 18), however, a significant increase in above-ground plant cover on treated plots was recorded during the Spring 1992 sampling period.

The quantity and periodicity of rainfall following a sludge application will control vegetative response (Wight and Black 1979). Rainfall during the 1991 late summer through fall rainy season was over 180 mm, however, a significant increase in above-ground plant cover between treatments did not occur on either plot type. This was partially the result of the recently applied wet sludge, which had covered much of the vegetation. The vegetation had to grow out from under the sludge before it could be accounted for with the line-intercept transect technique used to measure vegetative cover. Even though vegetative growth was good during this period, the net increase in plant cover did not surpass the cover loss incurred during the sludge application.

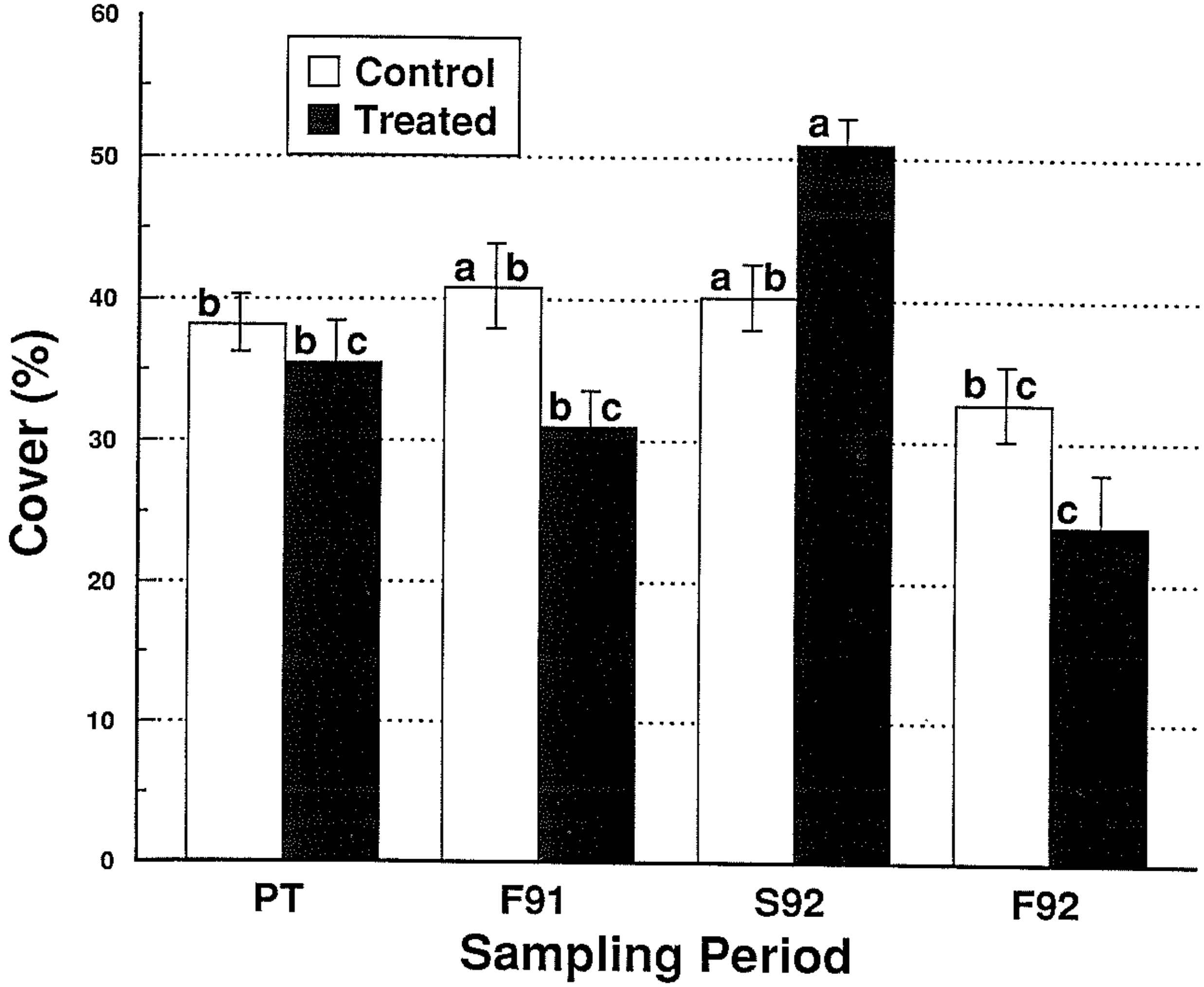

Figure 17. Total above-ground plant cover on large (10 X 10 m) control and sludge-treated plots. Data presented are for four sampling periods, pretreatment (PT), Fall 1991 (F91), Spring 1992 (S92), and Fall 1992 (F92). Values reported are means with standard errors. Bars with the same letter are not significantly different at the p = 0.05 level.

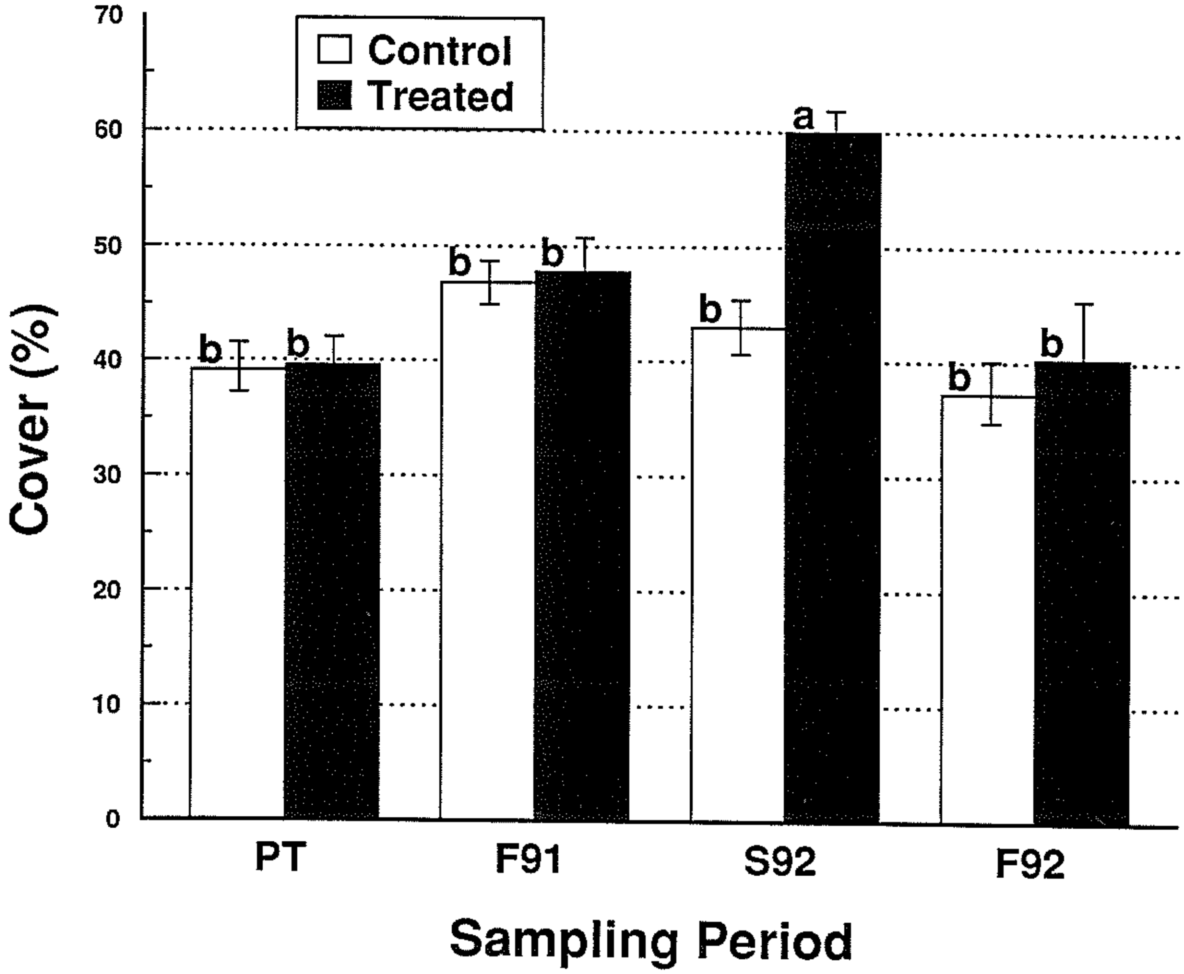

Figure 18. Total above-ground plant cover on control and sludge-treated runoff (3 X 10 m) plots. Data presented are for four sampling periods, pretreatment (PT), Fall 1991 (F91), Spring 1992 (S92), and Fall 1992 (F92). Values reported are means with standard errors. Bars with the same letter are not significantly different at the p = 0.05 level.

The treated plot above-ground plant cover response in Spring 1992 was primarily the result of the 1991-1992 winter through spring period which produced approximately 200 mm of rainfall. The following fall season (1992) was relatively dry (approximately 90 mm) and resulted in a decrease in above-ground plant cover.

Root Growth

We evaluated plant root growth in the large 10 X 10 m plots in Fall 1991 (six months after the sludge application) and again in Fall 1992 (18 months after sludge application). Root growth response was significantly less on the sludge-amended plots than on the control plots (Fig. 19).

Root growth may not have been stimulated on the treated plots by the increased nutrient availability. Chapin et al. (1987) hypothesized that some plants may not allocate C for root growth unless they are nutrient limited. One interpretation of the Chapin et al. hypothesis is that when nutrients are readily available for plant uptake, it is more efficient for the plant to allocate C resources to the above-ground supportive and photosynthetic tissues. However, this interpretation does not hold true for all circumstances. One study on N fertilization of blue grama grassland in New Mexico found no fertilizer effect on below-ground biomass (Pieper et al. 1973). Similar studies in other regions reported an increase in below-ground biomass (Black and Wight 1979; Goetz 1969; Power and Alessi 1971). Blue grama roots are very sensitive to soil moisture (Ares 1976). Therefore it is possible that high-nutrient availability, coupled with low soil water availability, may have acted to decrease below-ground standing biomass on the sludge-treated plots.

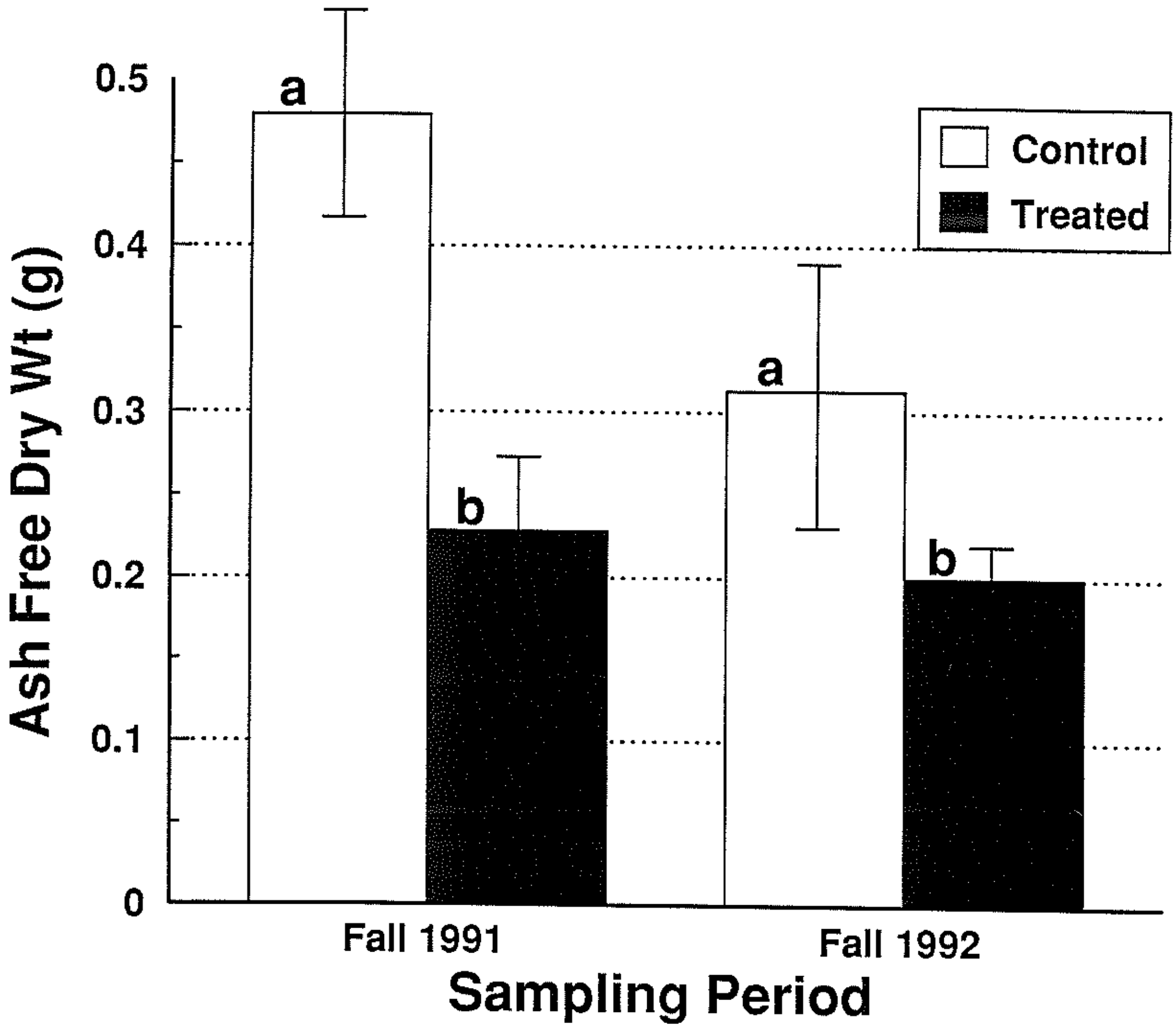

Figure 19. Root growth response on control and sludge-treated plots. Values reported are means with standard errors. Bars with the same letter are not significantly different at the $p = 0.05$ level.

Plant Tissue Analysis

Total N, Cu, and Zn levels in blue grama grass and yellow spiny-aster, after two growing seasons (Fall 1992) are reported in Table 4. Levels of Cd and Pb in plant tissues were below detection limits. Zinc levels were not significantly affected by sludge amendment. Sludge significantly increased Cu and TKN contents of both species.

Table 4. TKN, Cu, and Zn concentrations in plant tissues. Means, control versus treated, followed by the same letter are not significantly different at the P = 0.05 level.

	%		mg kg^{-1}			
	TKN	SE	Cu	SE	Zn	SE
Blue Grama Grass						
Control	1.158 a	0.027	2.313 a	0.152	13.750 a	1.447
Treated	2.165 b	0.024	4.563 b	0.729	11.656 a	1.110
Yellow Spiny-Aster						
Control	1.745 a	0.062	8.181 a	0.226	30.909 a	2.602
Treated	2.492 b	0.062	23.636 b	3.234	41.545 a	5.200

Sewage sludge application can influence the quality, as well as the quantity, of above-ground vegetation. Nitrogen assimilation and storage in plant biomass is an important component of nutrient cycling and critical to the success of a grassland restoration project. Total above-ground N should increase with a sludge-induced increase in above-ground biomass. The increase would be even greater if the biomass increase is accompanied by an increase in the N concentration of plant tissues. Assimilation and accumulation of heavy metals in plant biomass must also be monitored in sludge-amended sites because high concentrations in plant tissues can lead to potential phytotoxicity and/or transfer through the food chain, thereby resulting in bioaccumulation of metals at higher trophic levels.

SLUDGE EFFECTS ON PLANT TISSUE DECOMPOSITION

Accumulation of soil organic matter and organically bound plant nutrients represents an important step in the recovery of degraded lands. Decomposition of organic matter and subsequent mineralization of associated plant nutrients is necessary for the long-term growth, productivity, and stability of a restored ecosystem. The sludge treatment effect on the surface (above-ground) and buried (below-ground) decomposition of two plant litter types was evaluated within the control and treated plots. Sludge amendment significantly increased decomposition of blue grama litter in 1992 but not in 1991 (Fig. 20). We recorded significantly greater plant litter decomposition below-ground compared to above-ground, regardless of treatment.

Decomposition of saltbush litter (Fig. 21) was not affected by sludge amendment. However, decomposition was significantly greater below-ground than at the ground surface during both growing seasons.

The most intriguing questions arising from these decomposition studies is why blue grama litter experienced significant weight loss on sludge-treated plots in 1992, but not in 1991, and why the same phenomenon was not observed for the four-wing saltbush litter. As previously stated, climatic conditions were noticeably different between the two

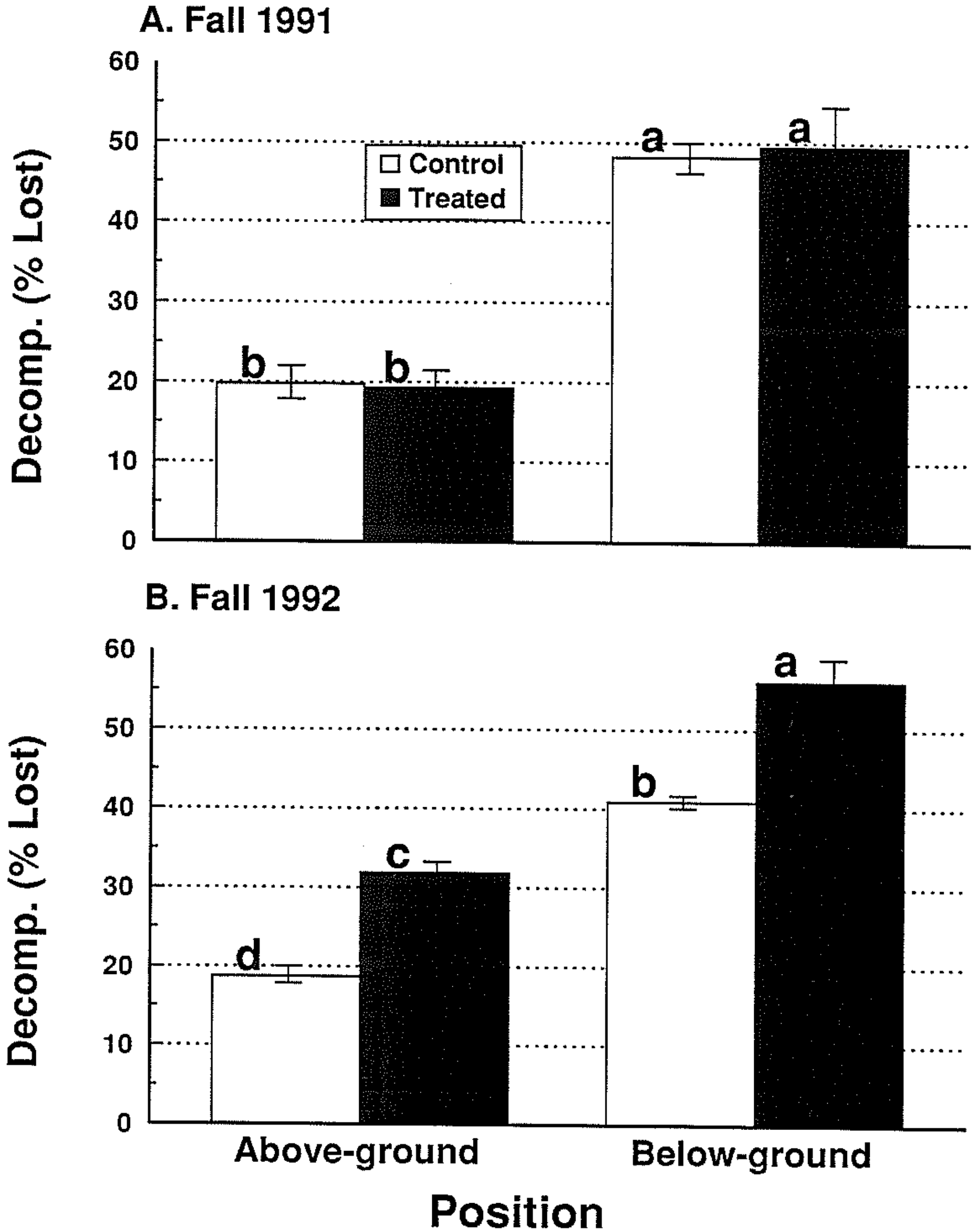

Figure 20. Blue grama litter decomposition on control and sludge-treated plots. Data presented are the results of two separate decomposition experiments conducted during the 1991 and 1992 growing seasons (spring - fall). Values reported are means with standard errors. Bars with the same letter, within a season, are not significantly different at the $p = 0.05$ level.

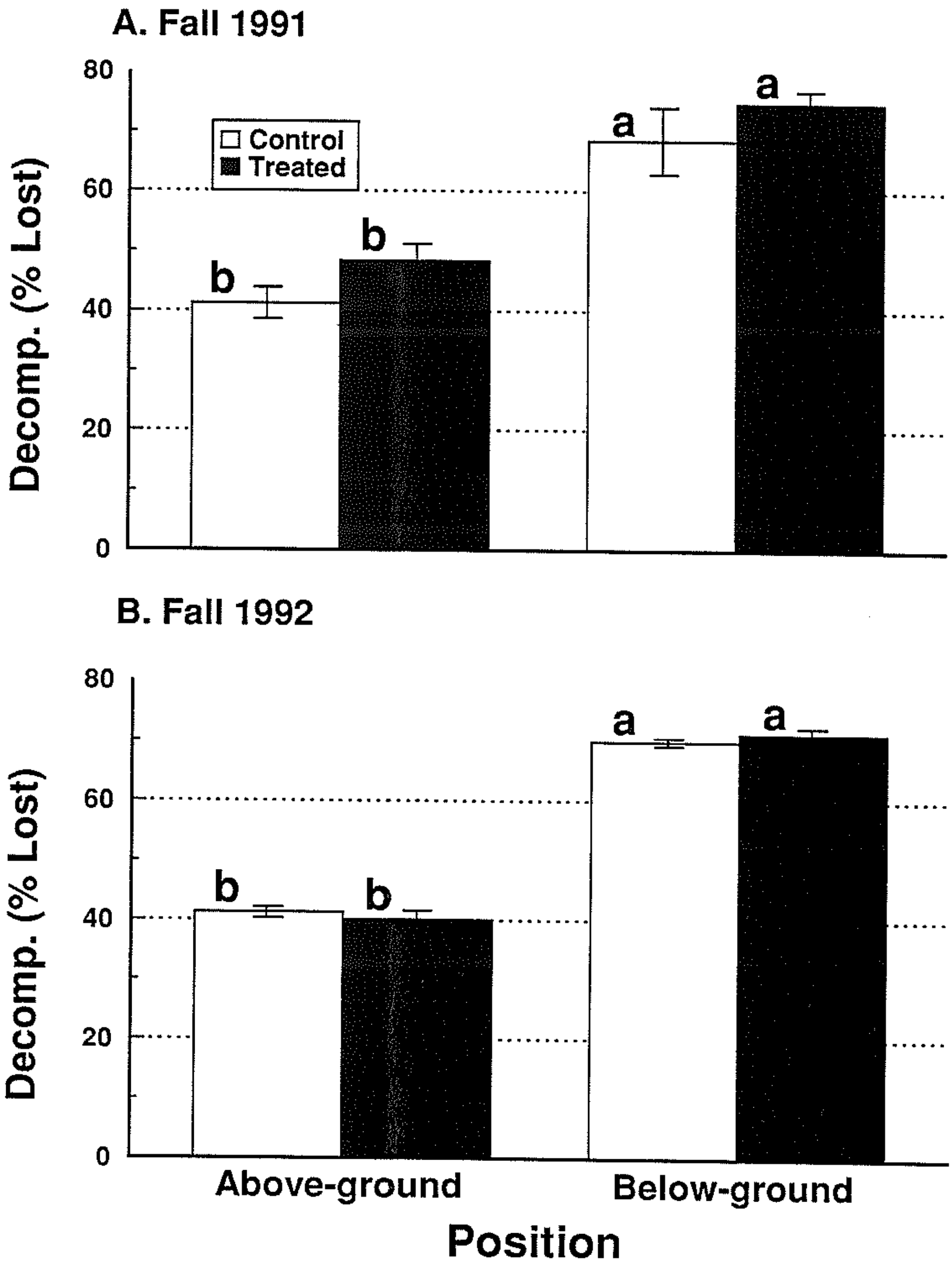

Figure 21. Four-wing saltbush litter decomposition on control and sludge-treated plots. Data presented are the results of two separate decomposition experiments conducted during the 1991 and 1992 growing seasons (spring - fall). Values reported are means with standard errors. Bars with the same letter, within a season, are not significantly different at the p = 0.05 level.

growing seasons. The first growing season following the sludge application was relatively moist (180 mm) while the second growing season was comparably dry (75 - 80 mm). Additionally, soil NH_4-N was relatively high, and NO_3-N and P were relatively low, at the end of the 1991 season in comparison to the levels at the end of the 1992 season. Interactions among the aforementioned factors, coupled with the initial differences in plant litter characteristics, make interpretation of the decomposition study data difficult. Although initially there were no significant differences in OC content (ash free dry weight/2) between the two litter types, saltbush litter had a significantly lower C:N ratio and a higher percentage of water soluble constituents (Table 5). Apparent differences in weight loss between the two tissue types may have resulted from losses in soluble salts in saltbush litter, rather than actual differences in decomposition between the two plant litter types. Because similar decomposition losses were not observed with both litter types, we conclude that complex interactions among soil and plant tissue variables, and not climate, were chiefly responsible for the differences in litter decomposition.

Table 5. Litter quality analyses for plant tissue used in decomposition studies. Means followed by the same letter, within the same row (same variable), are not significantly different at the P = 0.05 level.

Variable	Blue Grama	Saltbush
OC(%)	45.0 a	41.0 a
TKN (%)	0.27 b	1.33 a
C:N Ratio	167 a	31 b
Water Soluble Constituents (%)	3.9 b	22.7 a

Organic matter mineralization is a microbially mediated process and differences in mineralization rates are often the result of differences in plant tissue quality. The C:N ratio for blue grama and saltbush litter were 167 and 31, respectively. Decomposition of the blue grama litter is more likely to be N-limited because of its larger C:N ratio. The sludge treatment significantly increased plant-available (extractable) N. Consequently, the increase in soil N availability should have had a greater impact on the decomposition of blue grama litter than on the decomposition of saltbush litter (Bosatta and Berendse 1984; Parnas 1975). Differences in blue grama decomposition between 1991 and 1992 may also have been due somewhat to differences in litter quality, however, this hypothesis could not be tested because litter quality was not measured for the plant material used in the 1991 decomposition experiments.

While study results have not clearly demonstrated sludge-treatment effects on plant litter decomposition, other studies have documented an increase in microbial activity following sludge application to soils (Dennis and Fresquez 1989, Fresquez and Dennis 1990, Mitchell et al. 1978, Pagliai et al. 1981, Seaker and Sopper 1988, Stevenson et al. 1984). The apparent discrepancy between our findings and those of other researchers may be the result of several factors. Other studies did not differentiate between microbial activity associated with the decomposition of sludge organic matter and the decomposition of plant litter, and a large percentage (50 - 85%) of sludge organic matter is readily decomposed (Hartenstein 1981). Sludge-borne carbon sources rather than residual soil C pools may have increased microbial activity. It is also possible that the soil N limitations on microbial activity prior to sludge application in the other studies were greater than the soil limitations in this study and this stimulated a greater response from the added sludge.

SURFACE HYDROLOGY

Runoff Produced by Natural Storms

During the course of our three-year study, natural storms that produced runoff occurred only in 1991 and 1993. Analyses of the 1991 storms are included in Figure 22 and results from two 1993 storms producing runoff are in Figure 23. The term "runoff yield" used in the figures refers to the ratio of millimeters (mm) of runoff measured in the collection tanks at the base of the plots to the total mm of rainfall occurring during the related storms.

Runoff was recorded from four natural storm events of high intensity during July and August 1991 (Fig. 22). Sludge-treated plots had significantly less runoff than the control plots. An analysis of variance comparing runoff yields among plot pairs using slope gradient as the variable found no significant differences ($p \geq 0.05$) based on slope class, except for July 25, 1991. Runoff yields were greatest during high-intensity storm events. Runoff yields from the control plots increased progressively with increased precipitation and storm duration. Storm duration measurements for July 1991 were not available because we had not installed self-activating recording rain gauges prior to the storm event. Control plot runoff values ranged from 5% to 27% of the total precipitation input. We attributed differences in runoff yield among control plots within the two slope gradients to differences in resistance to surface-water flow brought about by microtopographical variation and differences in plant cover. Apparently, surface roughness, which greatly reduced surface water flow and, in turn, enhanced infiltration, was the major factor responsible for the reduced runoff in the sludge-amended plots. Two natural storms in 1993 again demonstrated the sludge's effectiveness at reducing runoff (Fig. 23). The August 28 storm was particularly intense, with 62.2 mm of rainfall occurring in one hour. Large quantities of surface runoff generated throughout the study site resulted in external runoff entering the plots and collection tanks on plot pairs 1L, 3L, and 3H, thus invalidating the treatment comparisons for these plot pairs. However, three runoff plot pairs (2L, 1H, and 2H) maintained their integrity during this intense storm and clearly demonstrated the sludge's effectiveness in reducing surface runoff.

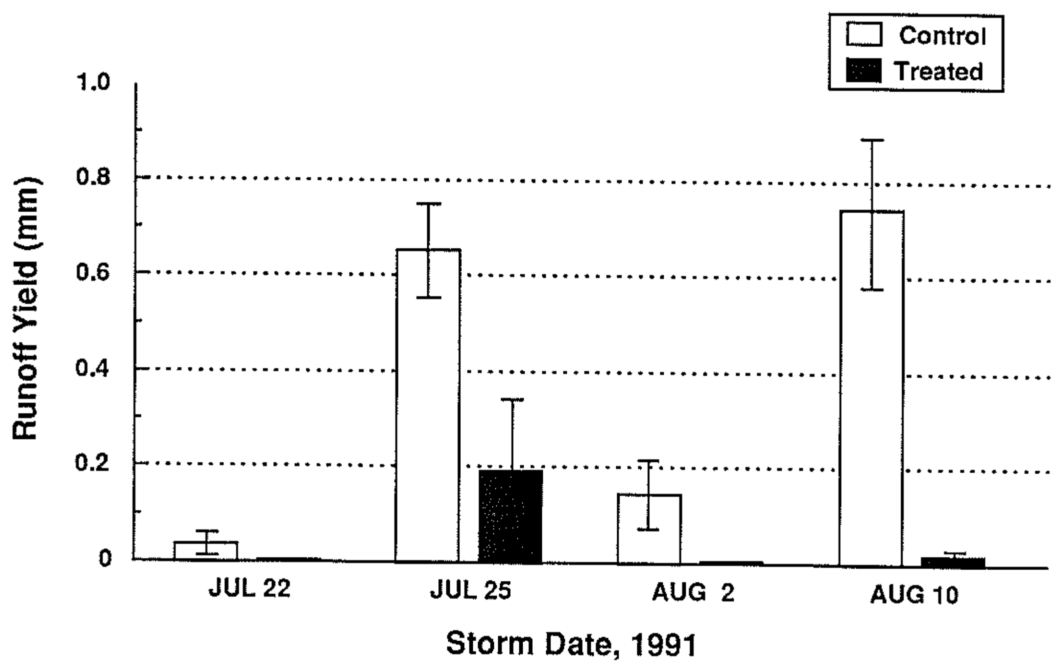

Figure 22. Mean runoff yield from all treated (n=6) versus all control plots (n=6) during four natural storms in 1991. (Note: differences in mean runoff between control and treated plots were significant at $p < 0.05$ for all storms except the July 22 event, wherein p=0.09).

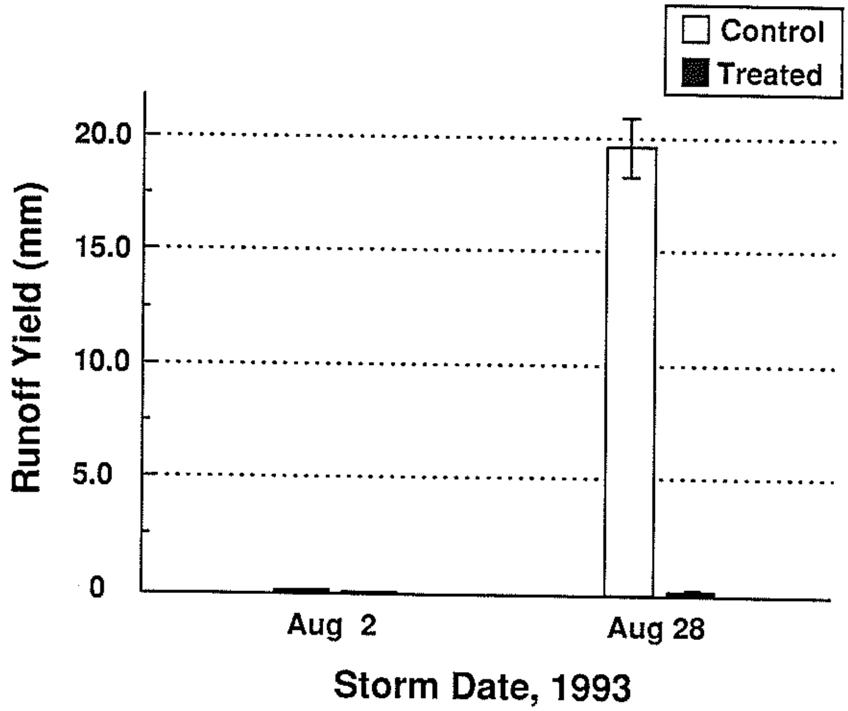

Figure 23. Mean runoff yield from treated versus control plots during two natural storms in 1993. Differences between treated and control plots were significant ($p = 0.05$) for all plot pairs.

Runoff Produced by Rainfall Simulation

Runoff yields produced during rainfall simulation experiments conducted during the month of September in each of the three years of our study are listed in Appendix B - Tables 1, 2, and 3 and also shown graphically in Figures 24 through 26. Expression of runoff yield as runoff per mm of precipitation standardized the runoff for comparison among plots because precipitation input among and between the plot pairs varied somewhat due to wind gusting.

In 1991, we subjected each of the six plot pairs to one rainfall simulation experiment. Table 1 (Appendix B) contains the plot characteristics and rainfall-runoff information from these simulations. As evidenced by the infiltration and runoff/rainfall ratios (runoff yield) for the experiments (Fig. 24), very little runoff was generated from the sludge-amended plots (treated plots - labeled "T") in comparison with the natural plots (control plots - labeled "C").

The soil infiltration rate at each plot during the six rainfall experiments was estimated by finding the average runoff rate from the slope of the runoff hydrograph for each plot and comparing that value to the average rainfall rate (see Appendix C). Hydrographs were generated only for those plots that produced runoff.

Additional indications of the absence of runoff from the sludge-amended plots were the rate at which water first appeared on the surface of the plot (i.e., the measured time to ponding indicating that the rainfall rate and infiltration rate were equal on at least part of the plot) and the time needed to drain the plot after the rainfall had ceased during the experiment (labeled as time to runoff stop, Table 1 - Appendix B). In general, ponding occurred about three times faster on the control plots than on the treated plots.

During the 1991 rainfall simulation experiments, we determined that the small amount of runoff generated from treated plots was produced only from the narrow concrete lip at the bottom edge of the plots. Given that runoff from the treated plots ceased immediately after rainfall had ceased during the simulation run, we concluded that "true" runoff within the plots did not occur. Following these first-year experiments, we modified our tank lids to extend slightly further onto the lower plot boundaries to exclude direct precipitation onto the narrow concrete lips and, thus, eliminate runoff originating from these areas.

Differences in runoff cessation times were recorded between the upper control plots (10-11% slope gradient) and the lower control plots (6% slope gradient), with runoff ceasing approximately three times faster on the upper plots. We attributed these differences to two factors: 1) steeper slope gradients on the upper control plots producing more rapid surface movement of runoff, and 2) greater infiltration in the upper plots because of a greater abundance of coarse fragments on the ground surface, thus resulting in lower quantities of runoff and ultimately faster cessation times.

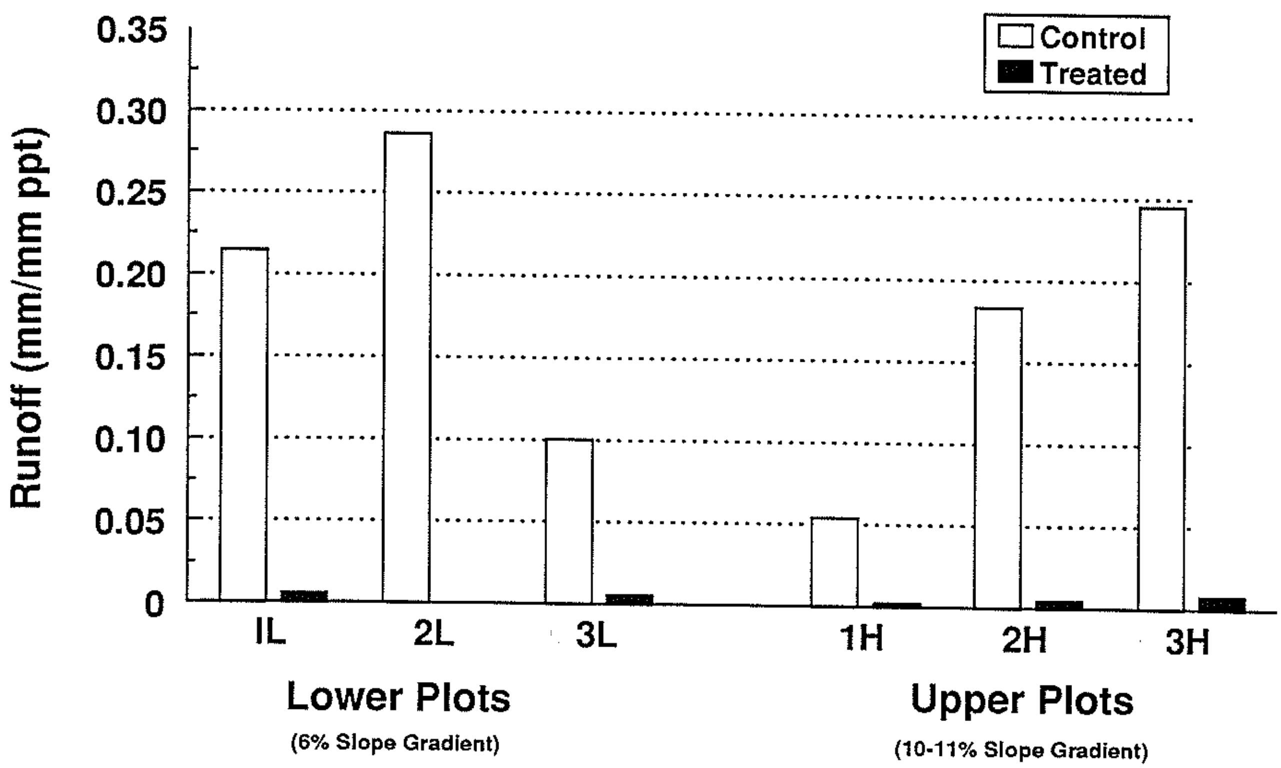

Figure 24. Differences in runoff yield between control and treated plots during 1991 rainfall simulation experiments. All experiments were conducted with antecedent soil moisture conditions at field moisture content.

In 1992, similar rainfall simulation experiments were conducted on each plot pair. However, very little runoff was generated from any of the runoff plots, including the controls, because of the extremely dry conditions that year created by the abnormally low summer precipitation. We conducted a second rainfall experiment (very wet run) to simulate the hydrological conditions that would result from two consecutive storms separated by only a short time interval. The "very wet" runs were conducted only one hour after the initial experiments with the prospect that higher antecedent soil and sludge moisture conditions would produce a higher probability for runoff. The "very wet" runs, denoted by a 'VW', were performed on plot pairs 2-Low and 1-High (Fig. 25 and Table 2 - Appendix B).

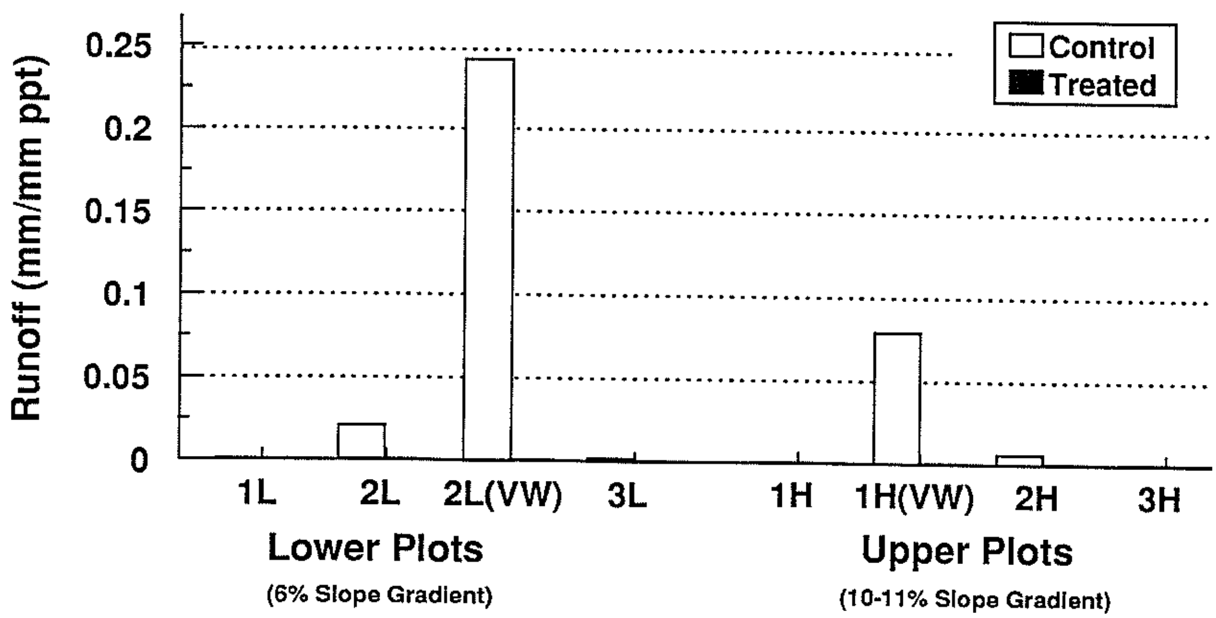

Figure 25. Differences in runoff yield between control and treated plots during 1992 rainfall simulation experiments. Plot pairs 2L and 1H were subjected to a second rainfall experiment (VW = very wet run) one hour following the initial rainfall experiment to increase the probability for runoff.

No runoff was generated from the treated plots during the "very wet" runs and we concluded the sludge effectively prevented runoff, even under very wet pre-existing conditions. The time to ponding, when it did occur on the treated plots, occurred six to 20 times slower than on the control plots (Table 2 - Appendix B). Appendix C contains the runoff hydrographs used in calculating soil infiltration rates for these experiments.

In 1993, we again performed the usual "dry" simulation runs on all six plot pairs. In addition, we conducted "very wet" runs on plot pairs 2L and 1H, and a "wet" run (20 hours following the dry run) on plot pairs 2H (Fig. 26 and Table 3 - Appendix B). Treated plots were more responsive to the rainfall events in 1993 than in the two previous years, although runoff yields from these plots were still limited in comparison to runoff yields from the control plots. We noted that rills were beginning to form in the treated plots, especially in their lower sections, whereas these were not present in the previous two years. We also observed small patches of bare ground in the treated plots, again in the lower plot areas, which were not evident in 1991 and 1992. We believe these rills and bare ground areas resulted from erosion and sludge movement during the intense September 28, 1993 storm that produced 6.20 cm (2.44 in.) of rainfall in one hour. However, we cannot discount natural decay of the sludge over the three-year study period as a contributing factor.

Infiltration rates during the 1993 rainfall simulation experiments, time to ponding, and time to runoff cessation measurements were consistent with the two previous years. Time to ponding occurred four to 13 times faster on the control plots than on the treated plots. In most cases, runoff ceased immediately after rainfall stopped on the treated plots. A greater time period for runoff cessation in the lower control plots compared to the upper control plots was again observed. Hydrograph curves and infiltration rate estimates for the plots during the 1993 rainfall experiments are in Appendix C.

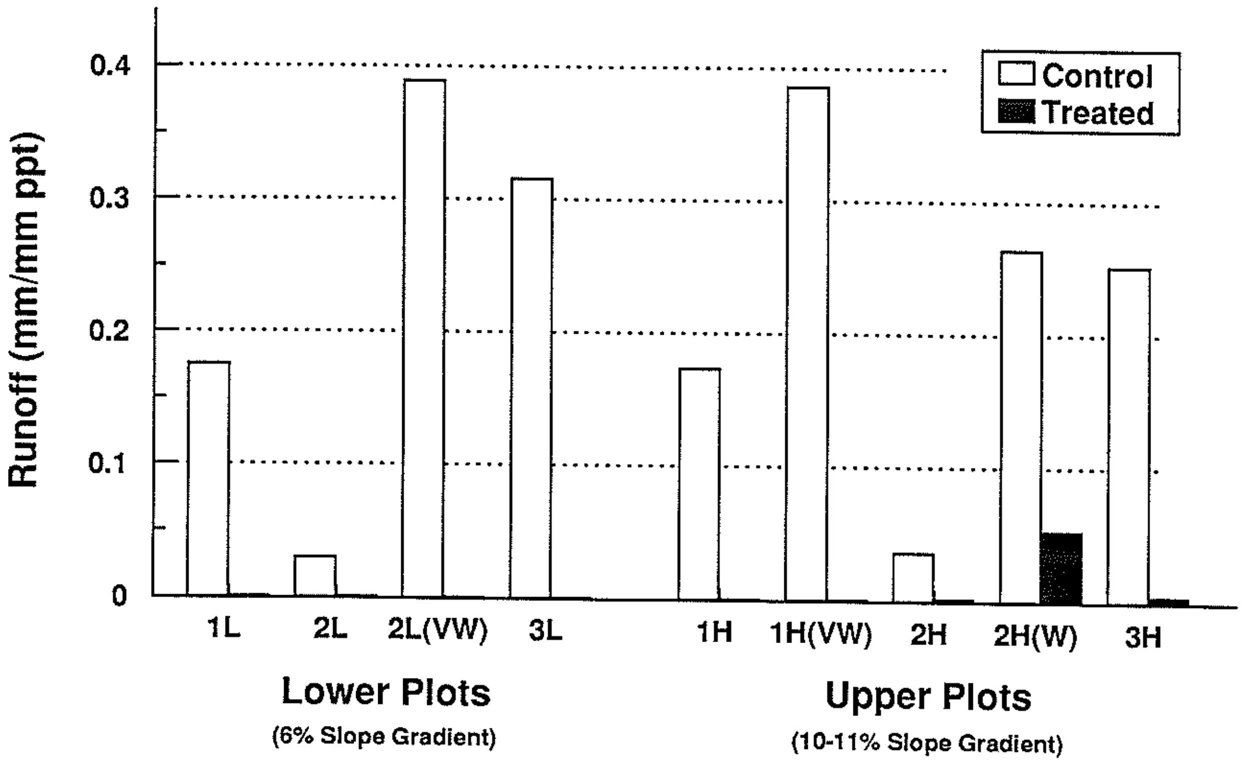

Figure 26. Differences in runoff yield between control and treated plots during 1993 rainfall simulation experiments. Plot pairs 2L and 1H were subjected to a second rainfall experiment (VW = very wet run) one hour following the initial experiment to increase the probability for runoff. Plot pair 2H was subjected to a second rainfall input (W = wet run) 20 hours following the initial "dry run" experiment because high wind gusting occurred during the initial experiment.

Runoff Quality

Total sediment yields from the runoff plots during the 1991, 1992, and 1993 rainfall simulation experiments are listed in Appendix B - Tables 4, 5 and 6. The 1991 rainfall experiments did not generate sufficient runoff from the sludge-amended plots to allow for representative total sediment samples (Table 4 - Appendix B). In those cases where the quantity of runoff accumulated in the collection tanks was low, representative samples could not be obtained without introducing positive error in our total sediment estimates. Virtually all of the runoff sample had to be collected, thereby preferentially collecting a greater proportion of the most coarse sediment particles when compared to the samples collected from tanks with large runoff volumes. For these plots we did, however, collect suitable samples for total suspended solids (TSS) and total volatile suspended solids (TVSS) as these samples required only 250 ml of runoff water each. Runoff water from treated plots generally contained greater concentrations (mg L^{-1}) of TSS and TVSS than runoff collected from the control plots (Tables 4, 5, & 6 - Appendix B). However, when considering the overwhelmingly greater runoff volumes generated from the control plots, the total amounts (g/m^2) of TSS and TVSS generated from the treated plots were much lower.

Similar patterns were observed for N and P in the runoff. Concentrations (mg L^{-1}) of TKN, NO_2-NO_3, and TP were almost always greater in the runoff water from the treated plots (Tables 7, 8, & 9 - Appendix B). However, when considering the much greater quantities of runoff generated from the control plots, the actual N and P yields that could potentially contribute to surface water degradation are appreciably higher for the control plots than for the treated plots.

Another important finding in our runoff analyses was that only a relatively small portion of TKN in the runoff water was in the form of inorganic N (NO_2^- and NO_3^-) (Tables 7 & 8 - Appendix B). This suggests that most, or in some cases all, of the TKN in the runoff was organic N, likely derived from plant litter on the control plots and litter plus sludge on the treated plots. Water-quality analyses for the 1993 rainfall simulation samples are in Table 9, Appendix B. These data show similar patterns of inorganic N to TKN ratios.

We also analyzed runoff samples from the 1991 natural storms and rainfall simulation experiments for concentrations of Cu, Cd, and Pb (Table 10 - Appendix B). Because no runoff was generated from treated plots in the 1992 rainfall experiments and only from one treated plot in the 1993 experiments we are not able to make comparisons between control and treated plots. New Mexico standards allow ≤0.01 mg L^{-1} Cd and ≤1.0 mg L^{-1} Cu in ground water (New Mexico Water Quality Control Commission 1991a). Livestock and wildlife watering standards are ≤0.05 mg L^{-1} Cd and ≤0.5 mg L^{-1} Cu (New Mexico Water Quality Control Commission 1991b). Levels of Cd and Cu in collected runoff were well below these standards, with the exception of the slightly elevated Cd concentration in runoff collected from Plot 1H-C after the July 25, 1991 storm (Table 10 - Appendix B). However, no significant treatment differences were found between mean Cd and Cu concentrations (treated versus control plots), indicating that Cd and Cu contamination in the runoff water from the added sludge did not occur. We did measure Pb concentrations exceeding current New Mexico standards for ground water (0.05 mg L^{-1}) and for livestock and wildlife watering (0.1 mg L^{-1}), but could not attribute such Pb levels to the sludge (Table 10 - Appendix B) because no significant differences were found between the Pb concentrations in runoff collected from the sludge-amended plots and the runoff from the control plots for any of the storms, including the rainfall simulation.

The elevated Pb levels in the runoff water may have resulted from Pb in the surface soil within the plots or from Pb solubilization from the galvanized steel runoff collection tanks.

CONCLUSIONS

Surface application of sewage sludge significantly increased extractable NH_4-N, NO_3-N, and P, but did not significantly increase soil OC. Soil extractable Cu, Pb, and Zn increased significantly following sludge application, but all remained at low concentrations (< 4.0 mg kg^{-1}). Soil Cd levels remained below detection. These results support previous findings that a one-time surface application of municipal sewage sludge at 45 Mg ha^{-1} will not produce heavy-metal contents in the soil that exceed limits established by regulatory agencies or the concentrations that have been shown to induce phytotoxicities (Fresquez et al. 1991).

Surface application of municipal sewage sludge significantly increased above-ground plant cover only on runoff plots in spring 1992. Analysis of above-ground plant tissues revealed significantly higher concentrations of N and Cu in the vegetation on the sludge-treated plots. Tissue concentrations of Cu in the forb yellow spiny-aster were four to five times that of blue grama grass, while N concentrations were similar. No significant differences in Zn concentration were observed between grass and forb tissues from control and treated plots. The recorded increase in nutrient availability may be responsible for significantly less root growth in sludge-amended plots. If root growth does not increase following a future decline in nutrient availability, it may be necessary to adjust sludge applications to decrease nutrient loading.

Decomposition of blue grama tissues was significantly greater on sludge-treated plots, but only during the 1992 field season. The differences in decomposition between the two tissue types (species) were attributed to interactions between tissue quality (C, N, soluble salt content, and other properties we did not measure such as lignin content), and environmental conditions (temperature, soil water, and nutrient availability).

The surface application of sludge increased rainfall infiltration, reducing the rangeland's potential for runoff and water erosion. Although runoff and infiltration rates on the control plots were comparable to those reported for other rangeland sites, both natural storms and rainfall simulation showed that runoff was virtually eliminated from sludge-amended plots. Subsequent increases in vegetation cover due to the sludge's fertilizer effect are anticipated as in the previous USDA Forest Service sludge application to rangeland study (Fresquez et al. 1990a). These anticipated increases in vegetation cover should further improve the surface hydrology of treated rangeland.

Potential degradation of surface water by sludge-borne contaminants, including heavy metals, in Albuquerque sewage sludge does not appear likely with a one-time application of 45 Mg ha^{-1} (20 tons/acre). Runoff water from treated plots generally contained greater concentrations of total sediment, total suspended solids (TSS) and total volatile suspended solids (TVSS) than runoff collected from the control plots. Yet, when considering the overwhelmingly greater runoff volumes generated from the control plots, the total quantities of total sediment, TSS, and TVSS generated from the treated plots were much lower. Similar patterns were observed for N and P in the runoff.

Cadmium and Cu collected in runoff were well below established New Mexico standards and no significant treatment differences were found between mean concentrations (treated versus control plots), indicating that Cd and Cu contamination to the runoff water from the added sludge did not occur. Pb concentrations exceeding current New Mexico standards for ground water and for livestock and wildlife watering were observed. However, we did not attribute the elevated Pb to the sludge treatment because we found no significant differences between the Pb concentrations in runoff from the control and treated plots.

Although our conclusions are based on only two growing seasons, we believe that surface application of sewage sludge may prove to be a useful tool in the restoration and management of semiarid rangelands. Furthermore, the reductions in total runoff and sediment yields indicate that sludge amendments to degraded rangeland can lead to improved surface-water quality. Sludge applications on semiarid rangeland have the potential for being environmentally and economically beneficial if these applications are based on sound guidelines developed through continuing research.

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APPENDIX A

Metal Content in Albuquerque Sludge (1987 - 1991)
(Table 1)

&

Limits versus Projected Metal Application Rates
(Table 2)

Table 1. Metal contents in Albuquerque sludge and federal (EPA and USDA) proposed limits for pollutant concentrations[1] in municipal sewage sludge.

Pollutant	EPA 1993[2]	USDA 1992[3]	EPA 1993[4]	Albuquerque Mean 1987-1991
Arsenic	41	60	75	17
Cadmium	39	25	85	15
Chromium	1200	>3000	3000	71
Copper	1500	1500	4300	704
Lead	300	300	840	271
Mercury	17	15	57	3
Molybdenum[5]	13	35	75	<23
Nickel	420	500	420	37
Selenium	36	32	100	2
Zinc	2800	2700	7500	1104

[1]All units are mg kg^{-1} dry weight.

[2]USEPA pollutant concentration limits for non-agricultural lawn or home garden application. 40 CFR Parts 257 and 503: Standards for the Disposal of Sewage Sludge; Final Rule, 1993.

[3]USDA Cooperative State Research Service Technical Committee W-170 recommended no-observed-effect levels (NOAEL). Chaney, R.L. and J.A. Ryan, "Regulating Residuals Management Practices, "Water Environment & Technology, 4(4):36-41, April 1992.

[4]USEPA maximum allowable limits of pollutants in sewage sludge. 40 CFR Parts 257 and 503: Standards for the Disposal of Sewage Sludge; Final Rule, 1993.

[5]Minimum detectable limit for molybdenum varied due to analytical method modifications and water content in sludge. Molybdenum was not detected in sludge between 1988 and 1990.

Table 2. U.S. Environmental Protection Agency limits on annual and cumulative land-application metal loading rates. Also listed are projected (Proj) metal application rates at the SNWR sludge-amendment study site. Projections are based upon a one-time surface application rate of 45 Mg ha^{-1} (20 tons/acre, dry weight equivalent).

	Soil (kg ha^{-1} yr^{-1})	Soil (kg ha^{-1})	Sludge (kg ha^{-1})
Metal	Limit[1]	Limit[2]	Projected
Arsenic	2.0	41	0.8
Cadmium	1.9	39	0.7
Chromium	150.0	3000	3.1
Copper	75.0	1500	31.7
Lead	15.0	300	12.2
Mercury	0.9	17	0.1
Molybdenum	0.9	18	<1.0
Nickel	21.0	420	1.6
Selenium	5.0	100	0.1
Zinc	140.0	2800	49.7

[1]USEPA maximum annual metal loading rates. Standards for the Disposal of Sewage Sludge, Final Rule, 1993; 40 CFR Part 503, 13(b)(4), Table 4.

[2]USEPA maximum cumulative metal loading rates. Standards for the Disposal of Sewage Sludge, Final Rule, 1993; 40 CFR Part 503, 13(b)(2), Table 2.

APPENDIX B

Plot Characteristics & Rainfall/Runoff Relationships During
Rainfall Simulation Experiments (1991 - 1993)
(Tables 1-3)

Sediment Concentrations & Total Sediment Yield From Runoff Plots
During Rainfall Simulation Experiments (1991 - 1993)
(Tables 4-6)

Nitrogen & Phosphorus Concentrations in Runoff Water During Rainfall Simulation
Experiments (1991 - 1993)
(Tables 7-9)

Heavy Metal Concentrations in Runoff Collected Following Natural Storms & Rainfall
Simulation Experiments (1991)
(Table 10)

Table 1. Plot characteristics[1] and rainfall/runoff relationships during rainfall simulation experiments[2] at the Sevilleta National Wildlife Refuge (SNWR), September 16-30, 1991.

I.D.	Rainfall (mm)	Runoff (mm)	Slope (%)	AMC[3] (%)	Infiltration (mm/hr)	Runoff Rainfall	Time to Ponding (min:sec)	Time to Runoff Stop (min:sec)
1L-C	43.0	9.2	6.4	2.8	59.8	0.214	1:34	4:46
1L-T	48.6	0.2	6.6	6.4	95.8	0.004	4:32	0:00
2L-C	48.8	13.9	5.9	2.8	62.6	0.285	0:46	5:40
2L-T	50.2	0.0	6.0	6.4	99.8	0.000	2:19	0:00
3L-C	42.2	4.0	5.9	1.8	78.4	0.095	1:26	--
3L-T	32.9	0.2	6.1	3.8	64.8	0.006	4:37	0:00
1H-C	13.2	0.7	10.3	1.8	--	0.053	2:03	2:38
1H-T	18.1	0.04	10.1	1.5	--	0.002	5:39	0:00
2H-C	35.0	6.5	9.9	1.5	55.9	0.186	2:09	1:59
2H-T	39.4	0.2	10.8	2.0	78.2	0.005	5:11	0:00
3H-C	53.1	13.4	10.2	2.8	74.5	0.252	0:59	1:40
3H-T	49.9	0.4	10.0	6.4	--	0.008	6:28	0:00

[1]All plots had an area of 30 m^2.
[2]All simulations lasted 30 minutes except 1H-T and 1H-C which ran for 13 minutes.
[3]Antecedent moisture content in upper 10 cm of soil.

Table 2. Plot characteristics[1] and rainfall/runoff relationships for rainfall simulation experiment[2] at the SNWR, September 21-24, 1992.

I.D.	Rainfall (mm)	Runoff (mm)	Slope (%)	AMC[3] (%)	Infiltration (mm/hr)	Runoff/ Rainfall	Time to Ponding (min:sec)	Time to Runoff Stop (min:sec)
1L-C	30.2	0.0	6.4	3.5	>rainfall	0.000	1:20	---*
1L-T	46.1	0.0	6.6	3.8	>rainfall	0.000	29:40	---
2L-C	47.5	1.1	5.9	4.0	83.0	0.020	0:52	0:41
2L-T	36.1	0.0	6.0	4.0	>rainfall	0.000	---	---
2L-C VW[4]	48.0[5]	11.6	5.9	13.3	---	0.242	---	---
2L-T VW	36.0	0.0	6.0	13.7	>rainfall	0.000	---	---
3L-C	36.8	0.1	5.9	2.3	73.3	0.002	0:55	0:00
3L-T	50.0	0.0	6.1	2.4	>rainfall	0.000	---	---
1H-C	26.3	0.0	10.3	5.3	>rainfall	0.000	2:00	---
1H-T	42.5	0.0	10.1	6.0	>rainfall	0.000	12:32	---
1H-C VW	46.6	3.7	10.3	10.2	76.4	0.079	1:14	1:52
1H-T VW	40.2	0.0	10.1	8.5	>rainfall	0.000	16:12	---
2H-C	47.6	0.3	9.9	2.6	94.8	0.006	2:05	1:39
2H-T	44.5	0.0	10.8	4.6	>rainfall	0.000	---	---
3H-C	26.9	0.0	10.2	9.4	>rainfall	0.000	3:09	---
3H-T	44.8	0.0	10.1	5.1	>rainfall	0.000	---	---

[1]All plots had an area of 30 m^2.
[2]All simulations lasted 30 minutes.
[3]Antecedent moisture content in upper 10 cm of soil.
[4]VW- "very wet" run, simulation run conducted within 1 hr following "dry run".
[5]Estimated
*Indicates either no ponding or no runoff occurred.

Table 3. Plot characteristics[1] and rainfall/runoff relationships for rainfall simulation experiments[2] at the SNWR study area, September 20-28, 1993.

I.D.	Rainfall (mm)	Runoff (mm)	Slope (%)	AMC[3] (%)	Infiltration (mm/hr)	Runoff/ Rainfall	Time to Ponding (min:sec)	Time to Runoff Stop (min:sec)
1L-C	39.7	7.0	6.4	1.6	54.76	0.176	2:00	5:00
1L-T	43.6	0.0	6.6	1.6	--	0.000	26:00	--*
2L-C	49.5	1.5	5.9	2.1	62.60	0.030	1:15	3:15
2L-T	45.3	0.0	6.0	2.1	--	0.000	16:00	--
2L-C VW[4]	45.4	17.7	5.9	9.8	51.40	0.390	1:00	4:00
2L-T VW	31.8	0.0	6.0	9.8	--	0.000	15:00	--
3L-C	48.1	15.2	5.9	1.8	56.84	0.316	1:30	6:30
3L-T	47.9	0.0	6.1	1.8	--	0.000	23:15	--
1H-C	37.9	6.6	10.3	2.6	56.60	0.174	3:10	0:46
1H-T	23.0	0.0	10.1	2.6	--	0.000	15:00	0:00
1H-C VW	49.3	19.1	10.3	13.1	57.60	0.387	1:14	1:20
1H-T VW	26.5	0.0	10.1	13.1	--	0.000	9:30	0:00
2H-C	27.7	1.0	9.9	1.6	53.20	0.036	3:30	2:11
2H-T	48.9	0.1	10.8	1.6	97.32	0.002	--	0:35
2H-C W[5]	35.1	9.3	9.9	7.9	53.80	0.265	1:00	2:30
2H-T W	45.7	2.4	10.8	7.9	81.70	0.053	6:00	3:00
3H-C	41.5	10.5	10.2	5.1	62.27	0.253	2:30	2:50
3H-T	45.6	0.2	10.1	5.1	91.00	0.004	4:30	0:20

[1]All plots had an area of 30 m^2.
[2]All simulations lasted 30 minutes.
[3]Antecedent moisture content in upper 10 cm of soil.
[4]VW - "very wet" run, simulation run conducted within one hour following previous "dry run".
[5]W - "wet" run, simulation run conducted within 24 hours following previous "dry" run.

Table 4. Sediment concentration (g L^{-1}), total sediment yield (g/30m^2), and suspended solids in runoff during rainfall simulation experiments at the Sevilleta study site, September 16-30 1991.

Plot I.D.	Precip.[1] (mm)	Runoff[2] (L)	Total Sediment (g L^{-1})		TSS[3] (mg L^{-1})		TVSS[4] (mg L^{-1})	
			(g L^{-1})	(g/30m^2)	(mg L^{-1})	(g/30m^2)	(mg L^{-1})	(g 30/m^2)
1L-C	43.0	286.6	0.66	189	249	71	69	20
1L-T	48.6	7.7	--*	--*	385	3	<1	<1
2L-C	48.8	427.9	0.54	233	182	78	<1	<1
2L-T	50.2	1.4	--	--	499	<1	<1	<1
3L-C	42.2	125.2	0.73	92	192	24	27	3
3L-T	32.9	11.5	--	--	352	4	132	2
1H-C	13.2	20.0	1.60	32	634	13	22	<1
1H-T	18.1	1.3	--	--	141	<1	18	<1
2H-C	35.0	196.3	0.60	120	414	81	<1	<1
2H-T	39.4	6.4	--	--	603	4	138	1
3H-C	53.1	410.2	0.74	305	339	139	<1	<1
3H-T	49.9	13.1	--	--	34	<1	<1	<1

[1]All rainfall experiments were conducted for 30 minutes, the with exception of Plots 1H-C & 1H-T where rain was applied for only 13 minutes.
[2]Runoff (liters) collected following the rainfall experiment.
[3]TSS = Total suspended solids.
[4]TVSS = Total volatile suspended solids.
*Samples were not collected due to limited runoff quantities.

Table 5. Sediment concentration (g L^{-1}), total sediment yield (g/30m^2), and suspended solids in runoff during rainfall simulation experiments at the Sevilleta study site, September 21-24 1992.

Plot I.D.	Precip.[1] (mm)	Runoff (L)	Total Sediment (g L^{-1})	Total Sediment (g/30m^2)	TSS[2] (mg L^{-1})	TSS[2] (g/30m^2)	TVSS[3] (mg L^{-1})	TVSS[3] (g/30 m^2)
1L-C	30.2	0.5	1.14	<1	--*	--*	--*	--*
1L-T	46.1	--	--	--	--	--	--	--
2L-C	47.5	33.4	0.56	19	413	14	63	2
2L-T	36.1	--	--	--	--	--	--	--
2L-C (VW)[4]	48.0	348.9	0.24	84	218	76	37	13
2L-T (VW)	36.0	--	--	--	--	--	--	--
3L-C	36.8	--	--	--	--	--	--	--
3L-T	50.0	--	--	--	--	--	--	--
1H-C	26.3	--	--	--	--	--	--	--
1H-T	42.5	--	--	--	--	--	--	--
1H-C (VW)	46.6	111.8	0.60	67	342	38	61	7
1H-T (VW)	40.2	--	--	--	--	--	--	--
2H-C	47.6	9.5	0.90	9	679	6	150	1
2H-T	44.5	--	--	--	--	--	--	--
3H-C	26.9	--	--	--	--	--	--	--
3H-T	44.8	--	--	--	--	--	--	--

[1] Rainfall experiments on all plots were conducted for 30 minutes.
[2] Total dissolved solids.
[3] Total volatile dissolved solids.
[4] VW - "very wet", simulation run conducted within one hour following "dry" run.
* No samples taken because of limited runoff quantity.

Table 6. Sediment concentration (g L^{-1}), total sediment yield (g/$30m^2$), and suspended solids in runoff during rainfall simulation experiments at the Sevilleta study site, September 20-28, 1993.

Plot I.D.	Precip.[1] (mm)	Runoff (L)	-- Total Sediment -- (g L^{-1})	(g/$30m^2$)	------ TSS ------- (mg L^{-1})	(g/$30m^2$)	------- TVSS ------ (mg L^{-1})	(g/30 m^2)
1L-C	39.2	208.5	0.57	119	517	108	76	16
1L-T	43.6	--	--*	--*	--	--	--	--
2L-C	49.5	444.5	0.37	164	321	143	44	20
2L-T	45.3	--	--	--	--	--	--	--
2L-C (VW)[2]	45.4	530.5	0.18	95	261	138	37	20
2L-T (VW)	31.8	--	--	--	--	--	--	--
3L-C	48.1	456.0	0.59	269	595	271	0.8	0.4
3L-T	47.9	--	--	--	--	--	--	--
1H-C	37.9	196.5	0.50	98	481	95	72	14
1H-T	23.0	--	--	--	--	--	--	--
1H-C (VW)	49.3	574.2	0.44	253	542	311	56	32
1H-T (VW)	26.5	--	--	--	--	--	--	--
2H-C	27.7	30.1	0.87	26	552	17	86	3
2H-T	48.9	3.7	5.76	21	1302	5	155	0.6
2H-C (W)[3]	35.1	274.2	0.47	129	62	17	0	0
2H-T (W)	45.7	72.2	2.60	188	620	45	26	2
3H-C	41.5	313.5	2.03	638	1341	420	18	6
3H-T	45.6	4.6	3.97	18	1224	6	66	0.3

[1] Rainfall experiments on all plots were conducted for 30 minutes.
[2] VW - "very wet", simulation run conducted within one hour following previous "dry run".
[3] W - "wet", simulation run conducted within 24 hours following previous "dry" run.
* No samples taken, no runoff generated on plot.

Table 7. Nitrogen and phosphorus concentrations in runoff water collected at the SNWR study site during rainfall simulation experiments, September 16-30 1991.

Plot I.D.	Precip. (mm)	Runoff (L)	TKN[1] ($mg\ L^{-1}$)	TKN[1] ($mg/30m^2$)	NO_2-NO_3[2] ($mg\ L^{-1}$)	NO_2-NO_3[2] ($mg/30m^2$)	TP[3] ($mg\ L^{-1}$)	TP[3] ($mg/30m^2$)
1L-C	43.0	286.6	1.6	459	0.00*	0*	0.19	54
1L-T	48.6	7.7	4.4	34	0.10	<1	0.58	4
2L-C	48.8	427.9	1.2	513	0.02	9	0.20	86
2L-T	50.2	1.4	6.1	9	0.01	<1	0.39	<1
3L-C	42.2	125.2	1.5	188	0.01	1	0.09	11
3L-T	32.9	11.5	5.3	61	0.00	0	0.84	10
1H-C	13.2	20.0	4.5	90	0.06	1	0.58	12
1H-T	18.1	1.3	4.6	6	0.57	<1	0.30	<1
2H-C	35.0	196.3	1.9	373	0.00	0	0.21	41
2H-T	39.4	6.4	11.4	73	1.19	8	1.15	7
3H-C	53.1	410.2	1.4	574	0.00	0	0.05	21
3H-T	49.9	13.1	1.4	18	0.00	0	0.09	1

[1]Total Kjeldahl nitrogen
[2]Nitrite + nitrate nitrogen
[3]Total phosphorus
*Zero concentrations indicate that the concentration in the rainwater exceeded that of the runoff.

Table 8. Nitrogen and phosphorus concentrations in runoff collected at the SNWR study site during rainfall simulation experiments, September 21-24 1992.

Plot I.D.	Precip. (mm)	Runoff (L)	TKN[1] (mg L^{-1})	TKN[1] (mg/30m^2)	NO_2-NO_3[2] (mg L^{-1})	NO_2-NO_3[2] (mg/30m^2)	TP[3] (mg L^{-1})	TP[3] (mg/30m^2)
1L-C	30.2	0.5	--	--	--	--	--	--
1L-T	46.1	--	--	--	--	--	--	--
2L-C	47.5	33.4	6.1	204	0.00*	0*	1.07	36
2L-T	36.1	--	--	--	--	--	--	--
2L-C (VW)[4]	48.0	348.9	0.7	244	0.06	21	0.50	174
2L-T (VW)	36.0	--	--	--	--	--	--	--
3L-C	36.8	--	--	--	--	--	--	--
3L-T	50.0	--	--	--	--	--	--	--
1H-C	26.3	--	--	--	--	--	--	--
1H-T	42.5	--	--	--	--	--	--	--
1H-C (VW)	46.6	111.8	1.3	145	0.08	9	0.85	95
1H-T (VW)	40.2	--	--	--	--	--	--	--
2H-C	47.6	9.5	15.5	147	0.00	0	3.13	30
2H-T	44.5	--	--	--	--	--	--	--
3H-C	26.9	--	--	--	--	--	--	--
3H-T	44.8	--	--	--	--	--	--	--

[1] Total Kjeldahl nitrogen
[2] Nitrite + nitrate nitrogen
[3] Total phosphorus
[4] VW - "very wet" run, simulation made within 1 hr following previous "dry" run.
* Zero concentrations indicate that the concentration in the rain water exceeded that of the runoff.

Table 9. Nitrogen and phosphorus concentrations in runoff collected at the SNWR study site during rainfall simulation experiments, September 20-28 1993.

Plot I.D.	Precip. (mm)	Runoff (L)	TKN[1]		NO_2-NO_3[2]		TP[3]	
			($mg\ L^{-1}$)	($mg/30m^2$)	($mg\ L^{-1}$)	($mg/30m^2$)	($mg\ L^{-1}$)	($mg/30m^2$)
1L-C	39.7	208.4	2.9	604	0.09	19	0.52	108
1L-T	43.6	--	--	--	--	--	--	--
2L-C	49.5	44.5	0.0*	--	0.04	44	0.29	13
2L-T	45.3	--	--	--	--	--	--	--
2L-C (VW)[4]	45.4	530.5	0.0	--	0.01	5	0.25	133
2L-T (VW)	31.8	--	--	--	--	--	--	--
3L-C	48.1	456.0	--	--	--	--	--	--
3L-T	47.9	--	--	--	--	--	--	--
1H-C	37.9	196.6	7.4	1454	0.10	20	0.80	157
1H-T	23.0	--	--	--	--	--	--	--
1H-C (VW)	49.3	574.2	5.8	3300	0.06	34	0.62	356
1H-T (VW)	26.5	--	--	--	--	--	--	--
2H-C	27.7	30.1	2.0	60	0.08	2	0.60	18
2H-T	48.9	3.7	11.3	42	4.36	16	1.84	7
3H-C	41.5	313.5	0.7	219	0.00	--	0.99	310
3H-T	45.6	4.6	107.7	49	2.89	13	8.33	38

1 Total Kjeldahl nitrogen
2 Nitrite + nitrate nitrogen
3 Total phosphorus
4 VW - "very wet" run, simulation made within 1 hr following previous "dry" run.
* Zero concentrations indicate that the concentration in the rain water exceeded that of the runoff.

Table 10. Concentrations (mg L^{-1}) of Cd, Cu, and Pb in runoff collected from sludge amended (T) and untreated (C) plots at the Sevilleta Wildlife Refuge, 1991. Mean Pb concentrations between treated (n = 6) and control (n = 6) plots were not significantly different at $P < 0.10$ for any of the storm events, including rainfall simulation. [adapted from Aguilar and Loftin 1992]

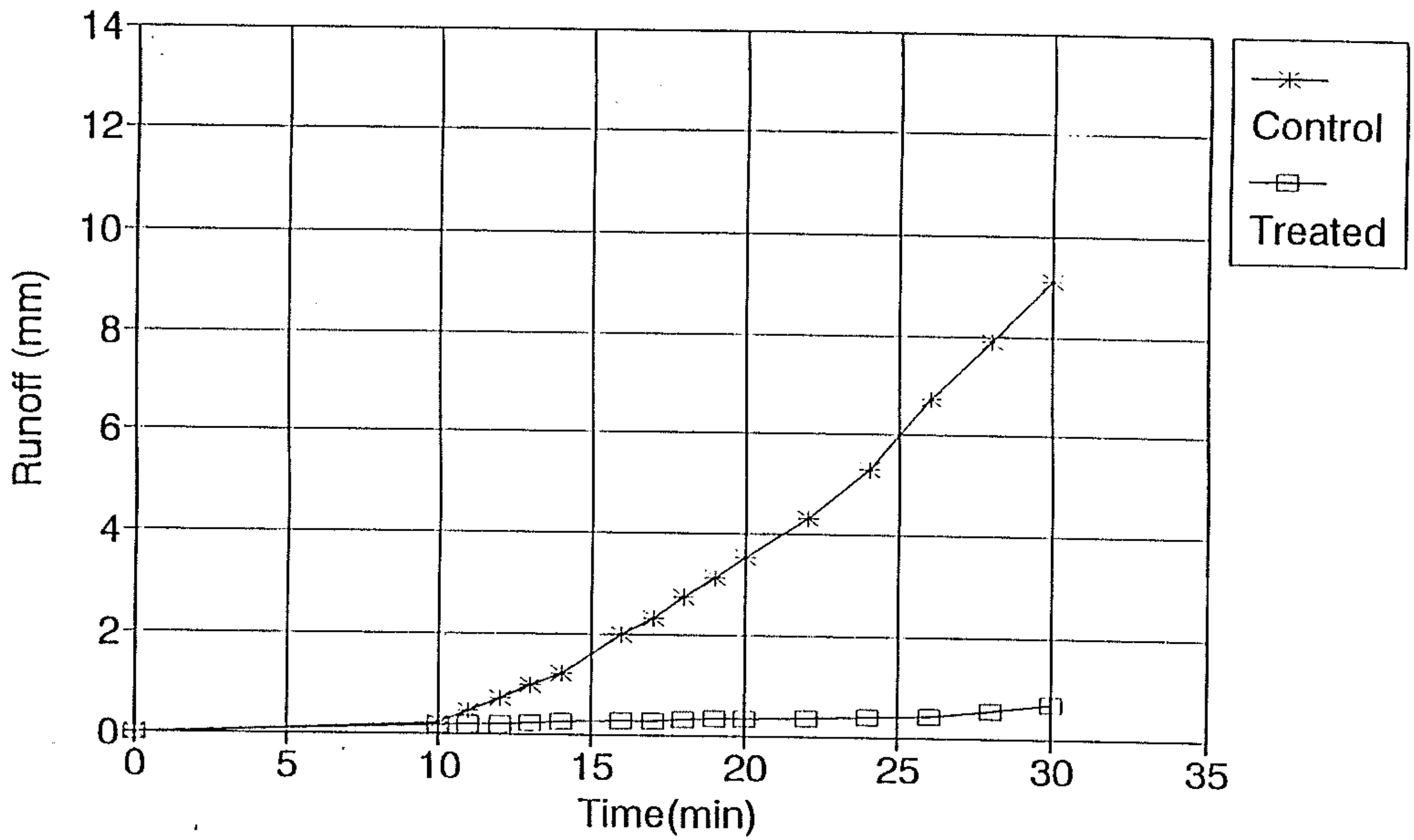
Plot	Cd	Cu	Pb	Cd	Cu	Pb
	July 25 Storm			August 2 Storm		
1L-C	<0.005	<0.02	0.31	<0.005	<0.02	0.11
1L-T	<0.005	<0.02	0.36	<0.005	<0.02	0.25
2L-C	0.005	<0.02	0.21	<0.005	<0.02	<0.05
2L-T	<0.005	<0.02	0.26	<0.005	<0.02	<0.05
3L-C	<0.005	<0.02	0.19	<0.005	<0.02	<0.05
3L-T	0.008	<0.02	0.48	<0.005	<0.02	<0.05
1H-C	0.016	0.02	0.47	<0.005	<0.02	0.09
1H-T	<0.005	<0.02	0.23	<0.005	<0.02	0.24
2H-C	<0.005	<0.02	0.66	<0.005	<0.02	0.10
2H-T	<0.005	<0.02	0.82	<0.005	<0.02	<0.05
3H-C	<0.005	<0.02	0.34	<0.005	<0.02	0.07
3H-T	<0.005	<0.02	0.37	<0.005	<0.02	0.25
	Aug. 10 Storm			Sept. rainfall simulation		
1L-C	<0.005	<0.02	0.13	<0.005	<0.02	<0.05
1L-T	<0.005	<0.02	0.20	<0.005	<0.02	0.13
2L-C	<0.005	<0.02	<0.21	<0.005	<0.02	0.40
2L-T	<0.005	0.52	2.52	<0.005	<0.02	0.11
3L-C	<0.005	<0.02	0.16	<0.005	<0.02	0.11
3L-T	<0.005	<0.02	0.05	<0.005	<0.02	0.17
1H-C	<0.005	<0.02	0.10	<0.005	<0.02	0.13
1H-T	<0.005	<0.02	0.39	<0.005	<0.02	0.13
2H-C	<0.005	<0.02	0.16	<0.005	<0.02	0.12
2H-T	<0.005	<0.02	0.14	<0.005	<0.02	0.21
3H-C	<0.005	<0.02	0.16	<0.005	<0.02	<0.05
3H-T	<0.005	<0.05	0.46	<0.005	<0.02	<0.05

APPENDIX C

Runoff Hydrographs Generated from Rainfall Simulation Experiments
(1991 - 1993)

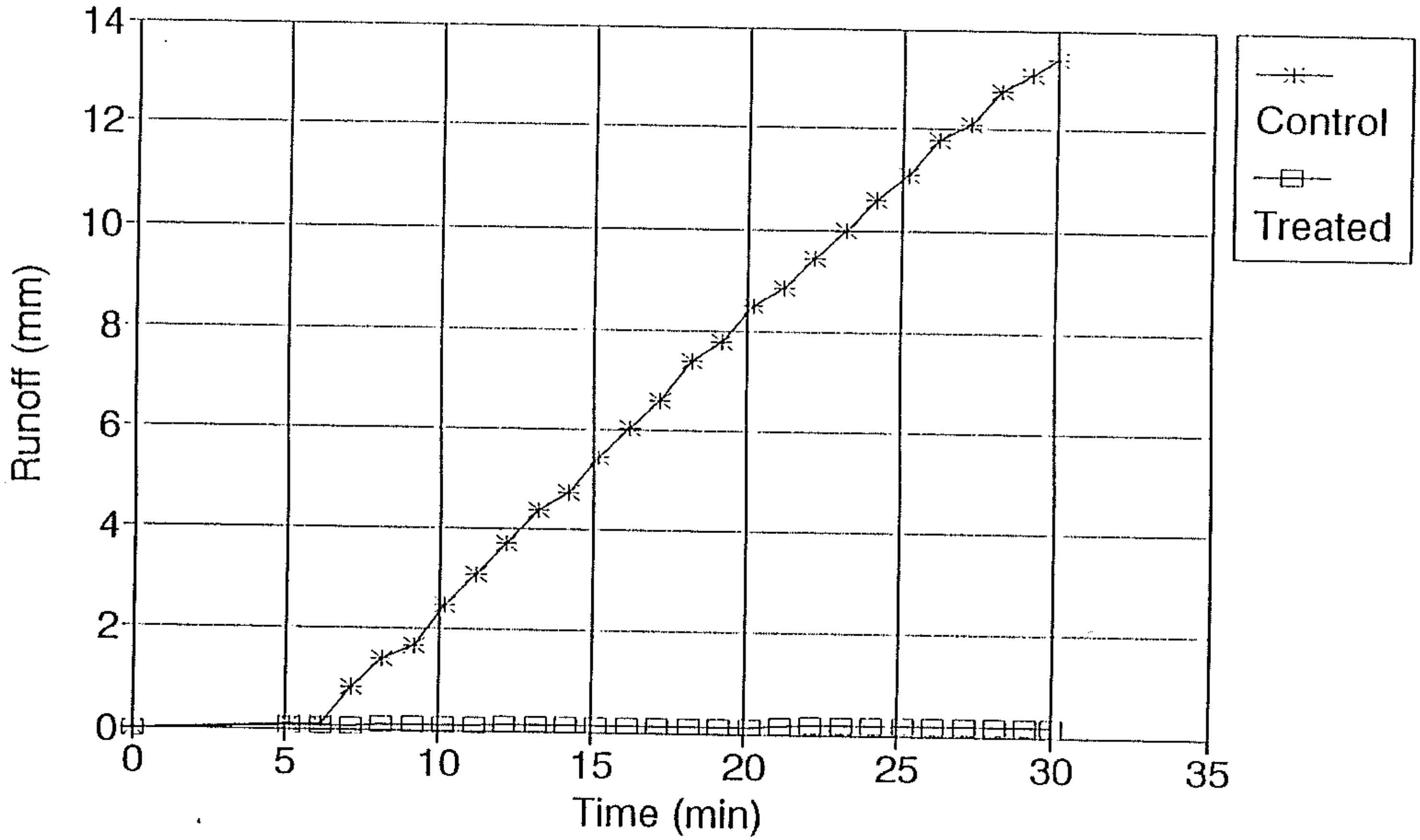
Sevilleta 1991

Plot 1L-C & Plot 1L-T



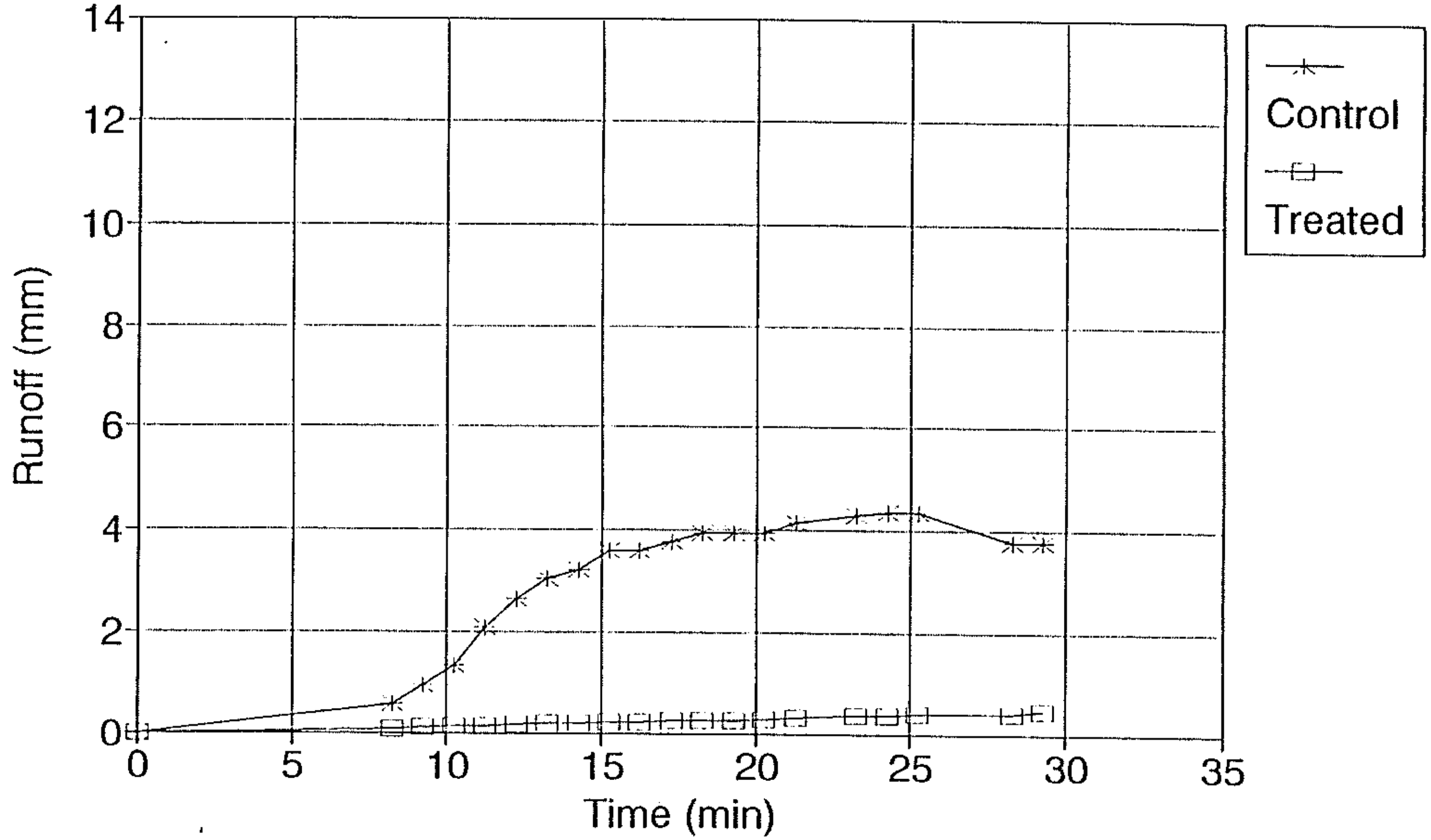
Sevilleta 1991

Plot 2L-C & Plot 2L-T



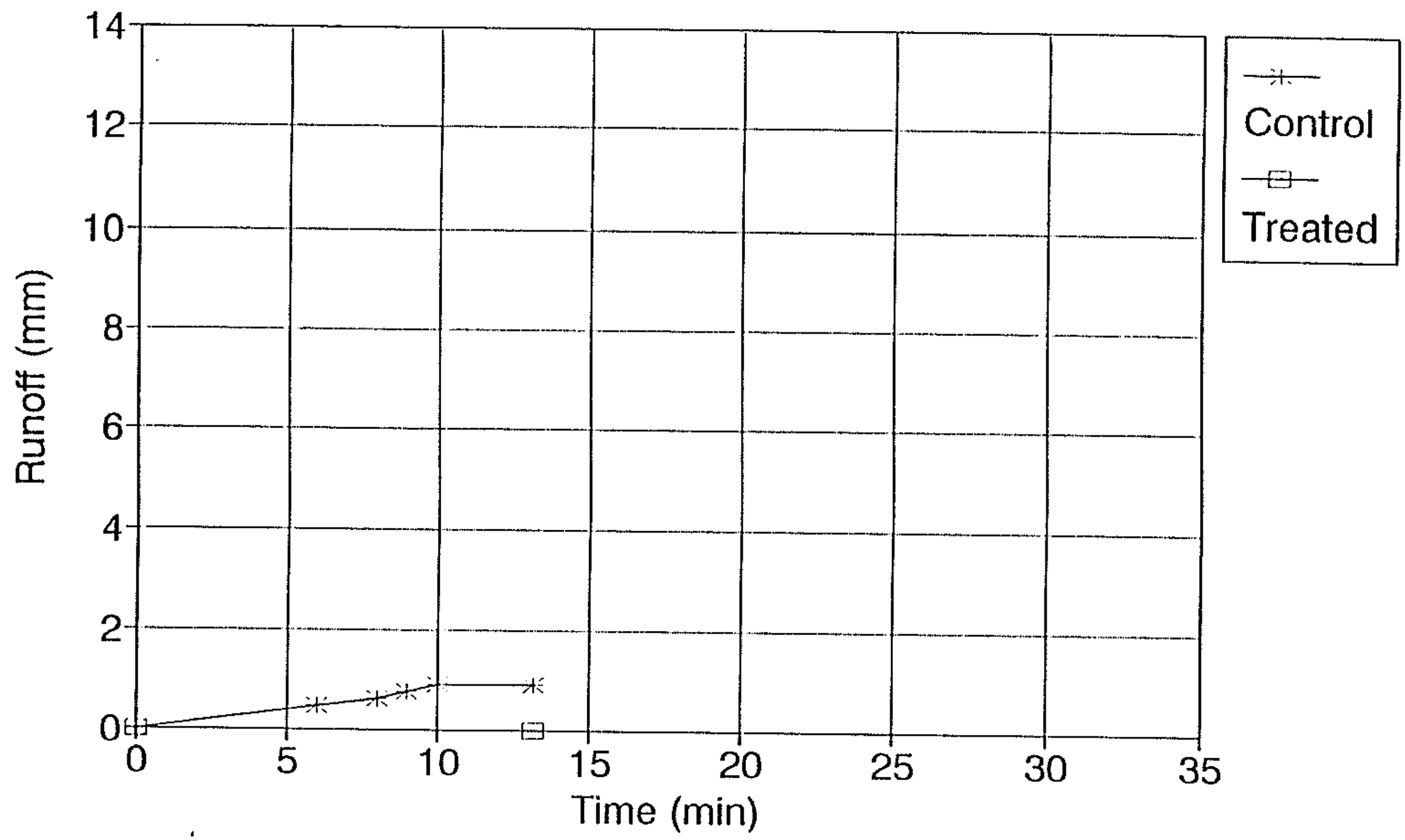
Sevilleta 1991

Plot 3L-C & Plot 3L-T



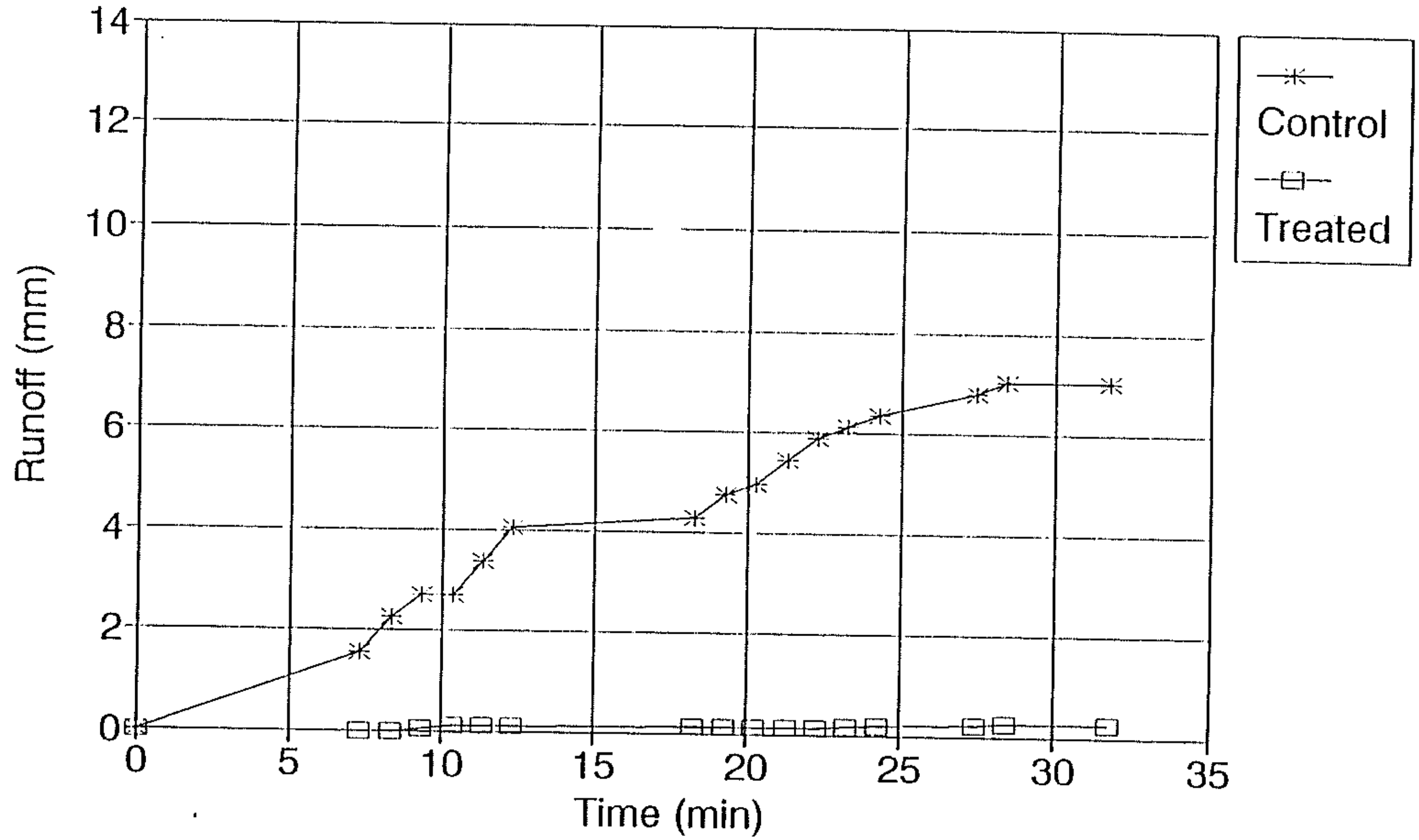
Sevilleta 1991

Plot 1H-C & Plot 1H-T

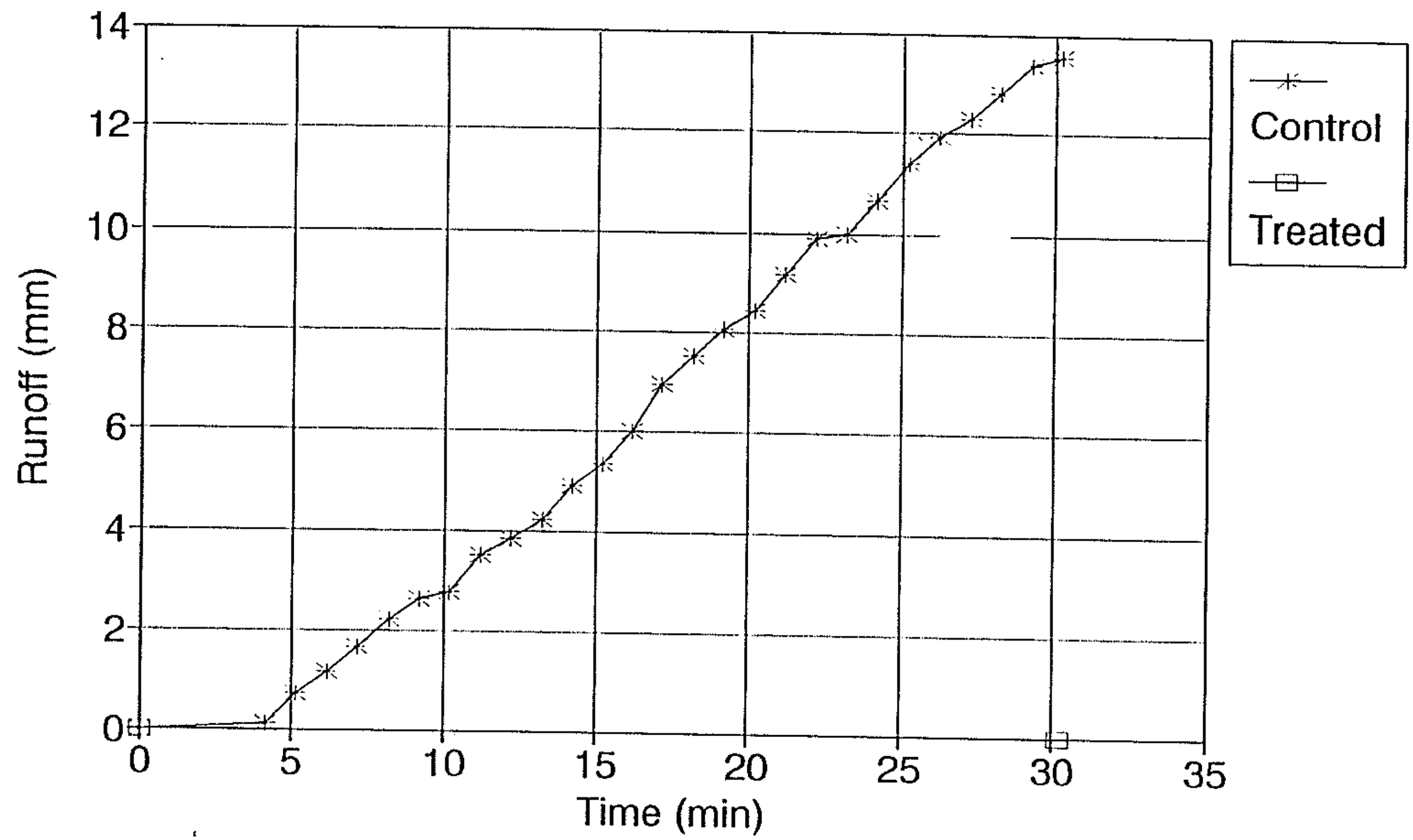


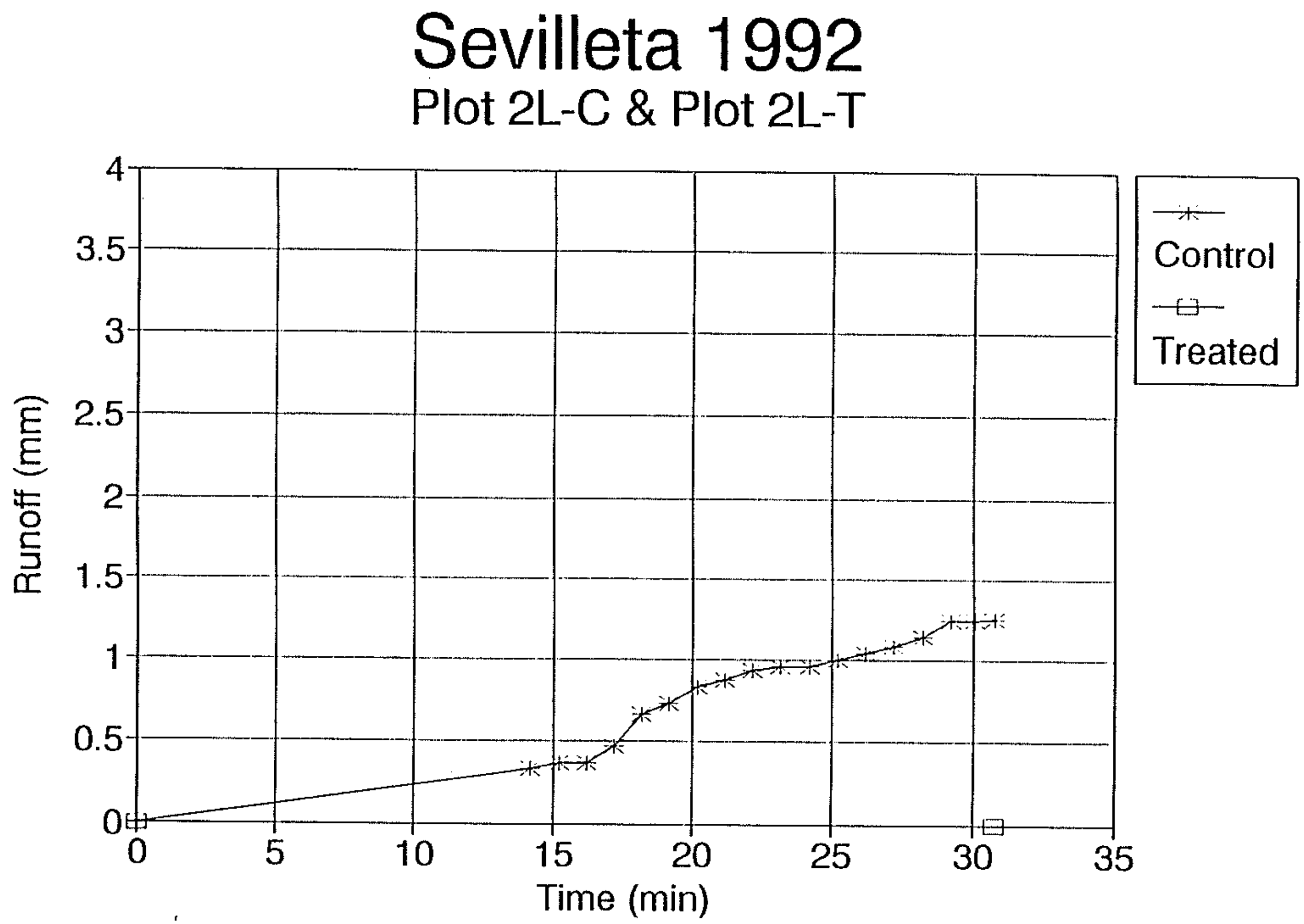
Sevilleta 1991

Plot 2H-C & Plot 2H-T



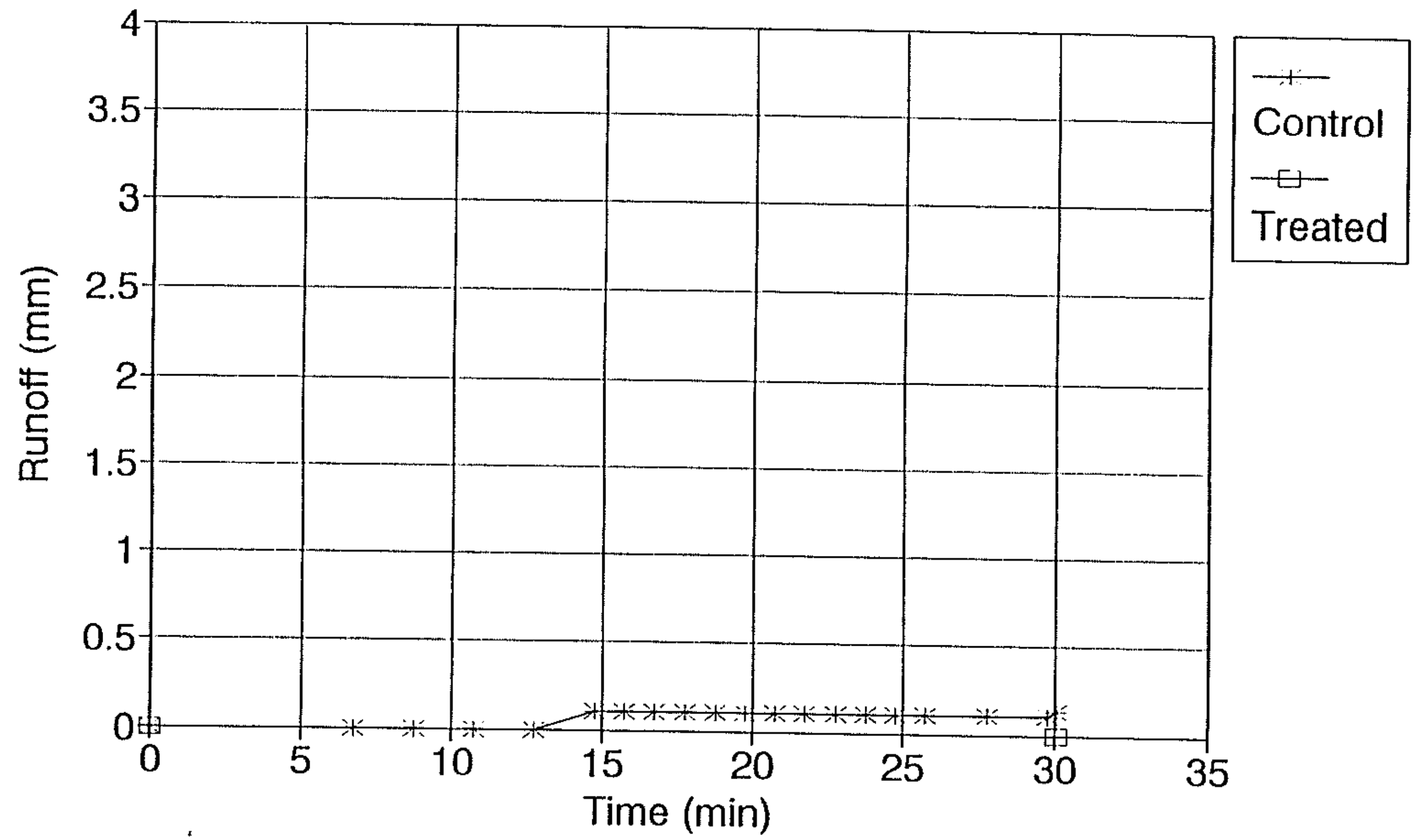
Sevilleta 1991
Plot 3H-C & Plot 3H-T




Sevilleta 1992
Plot 2L-C & Plot 2L-T
Runoff (mm)
4
3.5
3
2.5
2
1.5
1
0.5
0
0
5
10
15
20
25
30
35
Time (min)
Control
Treated

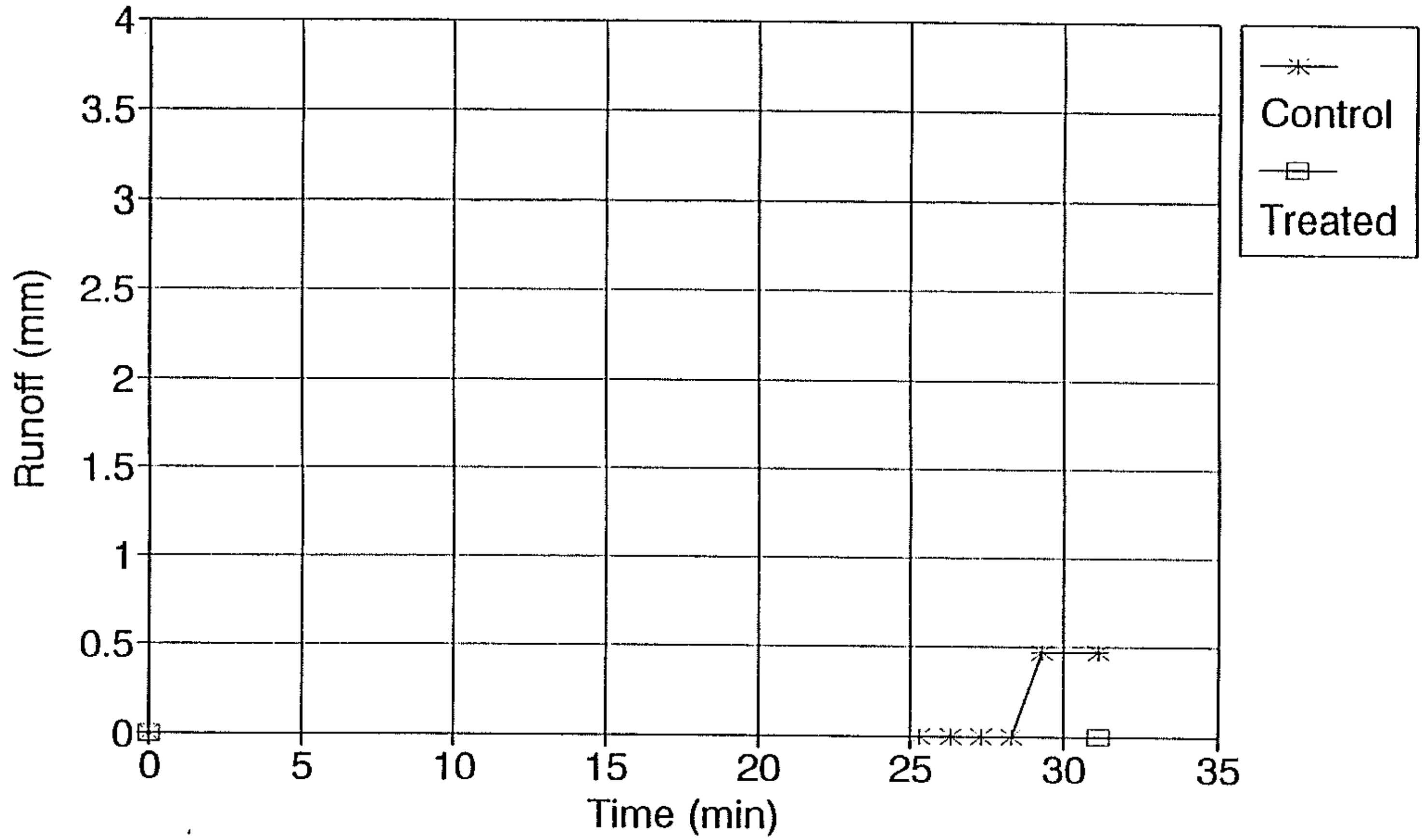
Sevilleta 1992

Plot 3L-C & Plot 3L-T



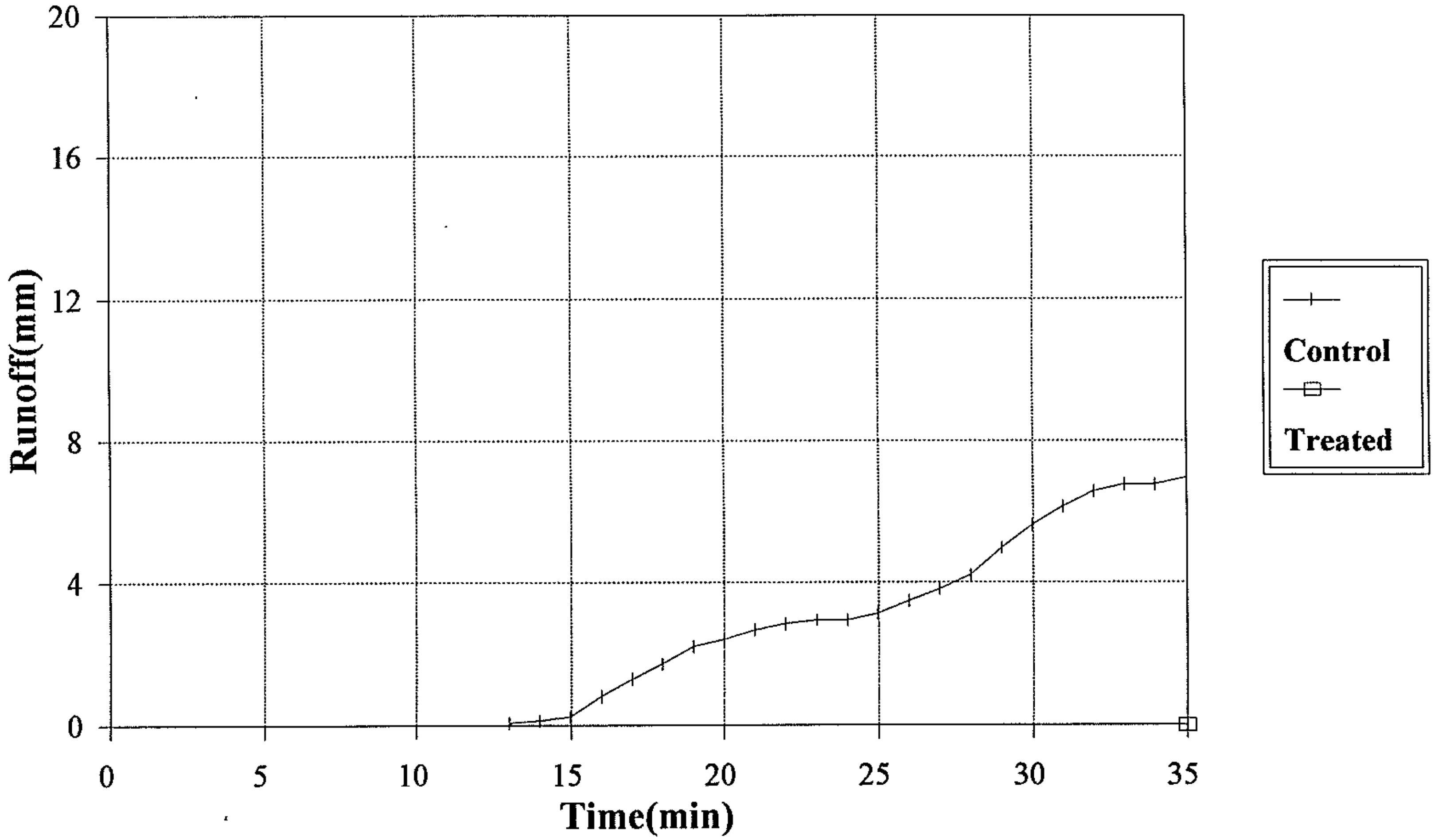
Sevilleta 1992

Plot 2H-C & Plot 2H-T



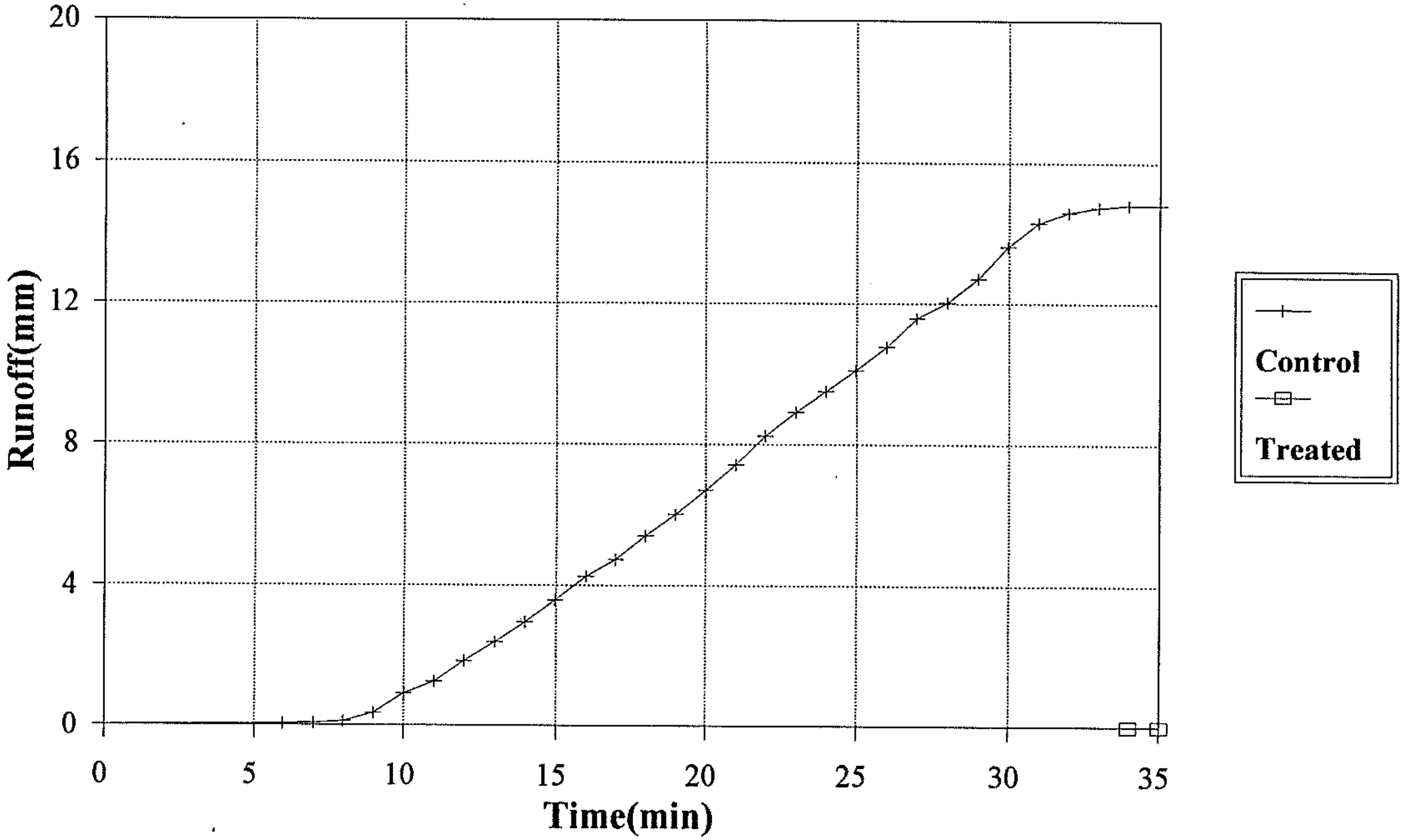
Sevilleta 1993

Plot 1L-C & Plot 1L-T



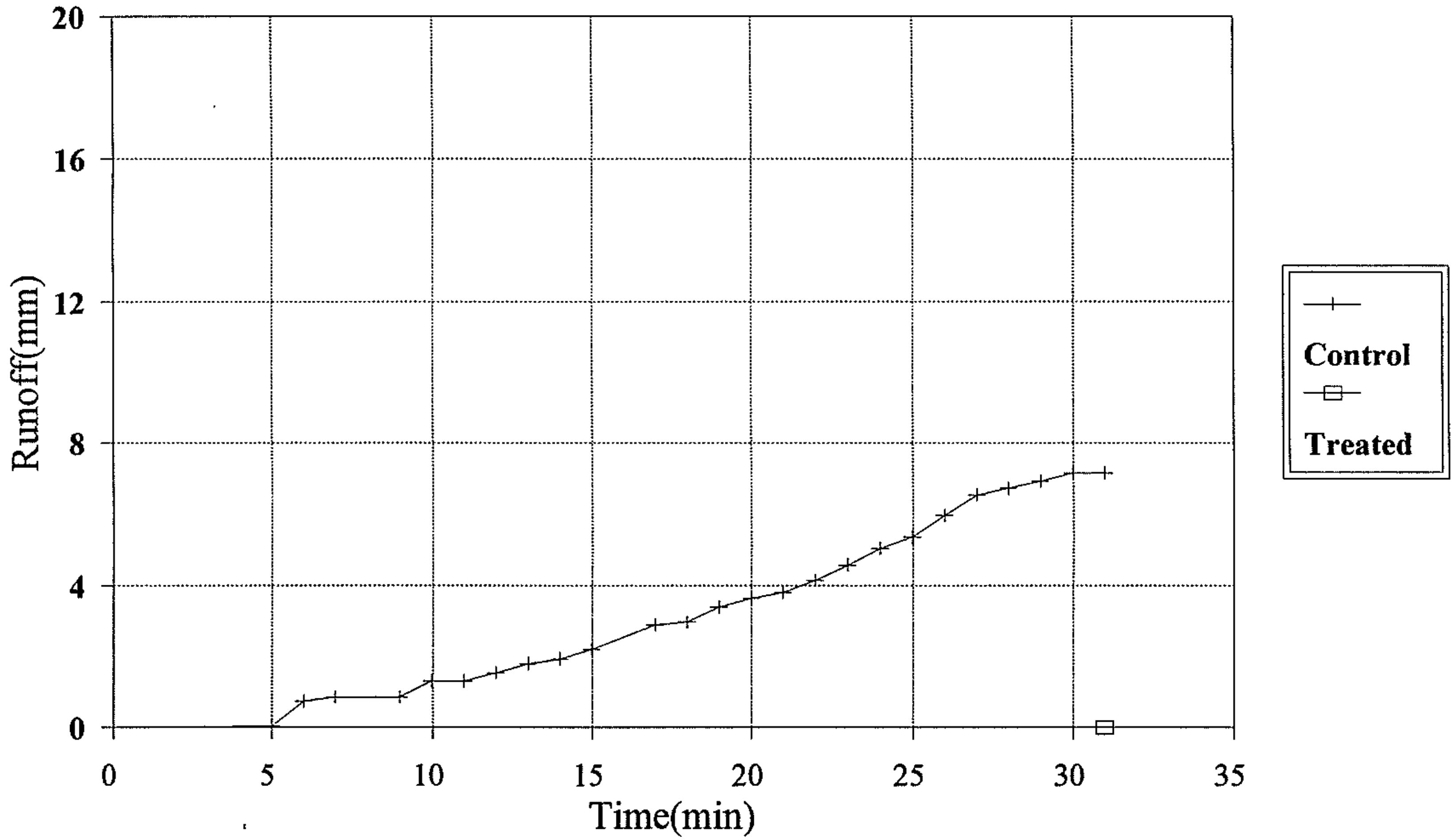
Sevilleta 1993

Plot 3L-C & Plot 3L-T



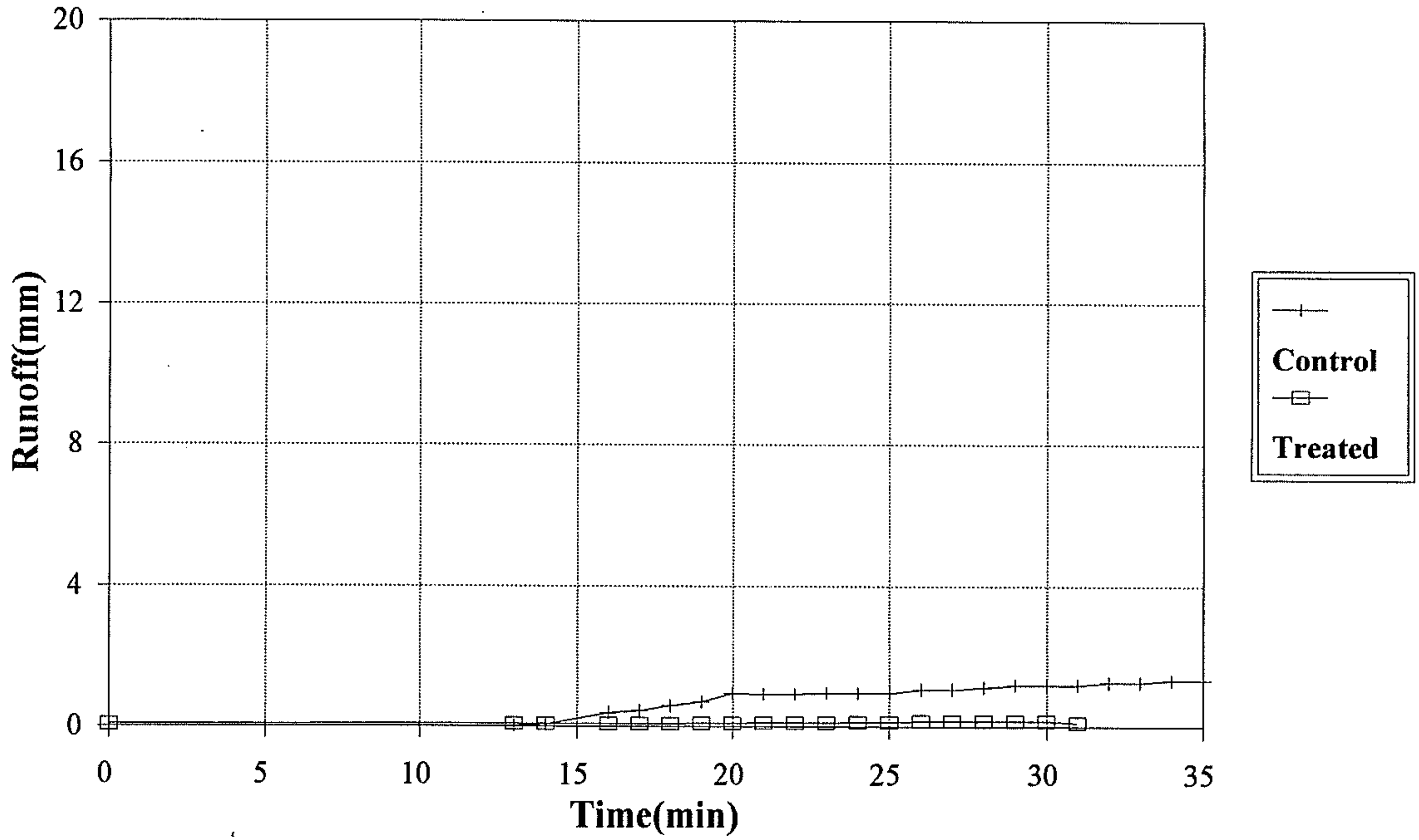
Sevilleta 1993

Plot 1H-C & Plot 1H-T



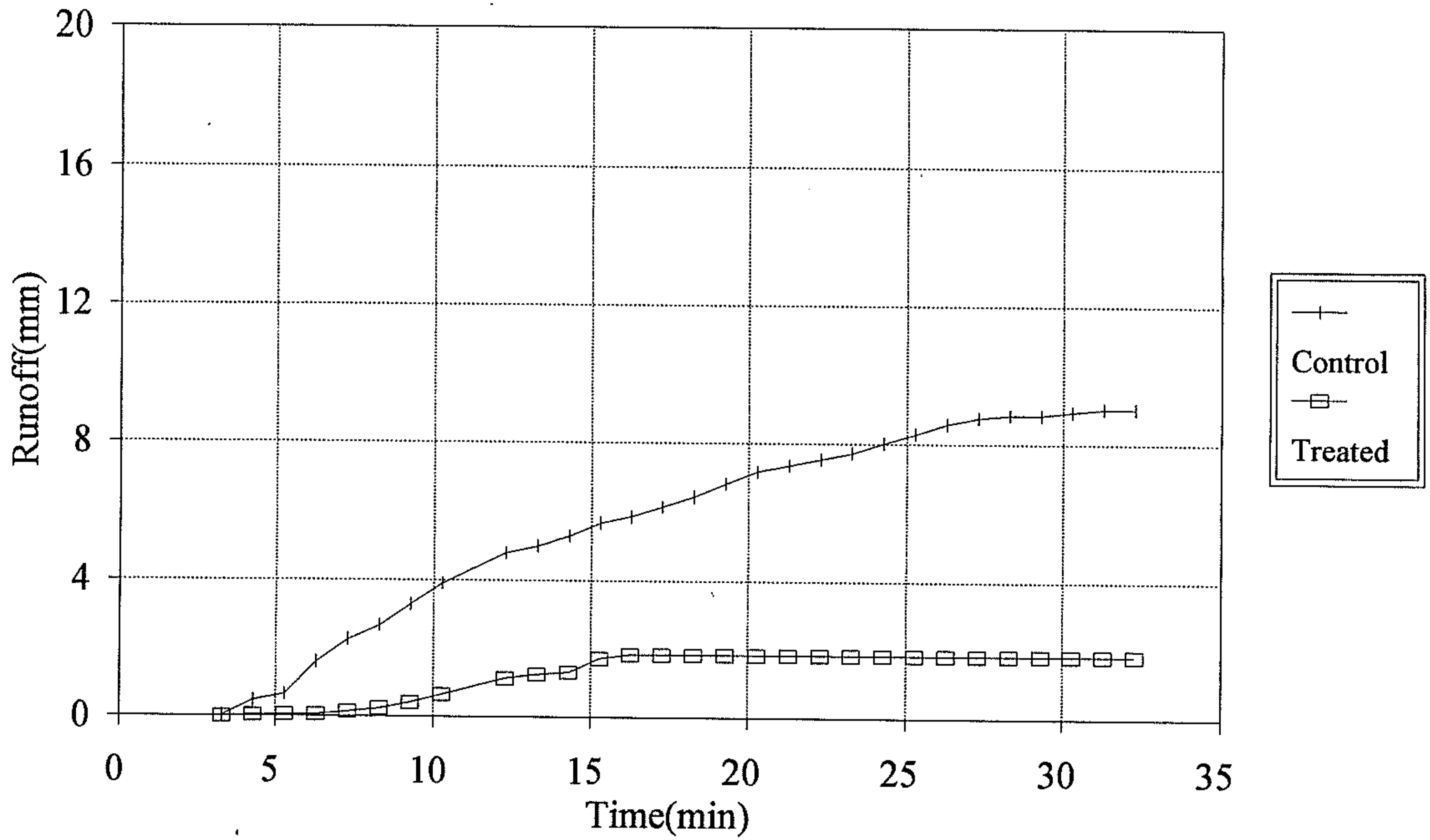
Sevilleta 1993

Plot 2H-C & Plot 2H-T



Sevilleta 1993

Plot 2H-C wet & Plot 2H-T wet



Sevilleta 1993

Plot 3H-C & Plot 3H-T