UTILIZING ANIMAL WASTE AMENDMENTS

TO IMPAIRED RANGELAND SOILS TO REDUCE RUNOFF

A Thesis

by

DIANA M. THOMAS

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

May 2011

Major Subject: Water Management and Hydrological Science

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ABSTRACT

Utilizing Animal Waste Amendments to Impaired Rangeland Soils to Reduce Runoff.

(May 2011)

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Composted biological wastes contain vital plant nutrients that assist in plant growth as well as contain organic matter that promotes good soil conditions; both aid in rangeland restoration. Most importantly, it has the potential to restore water availability through increased infiltration and reduced runoff. In this thesis, local sources of composted dairy manure are utilized for application onto the degraded Fort Hood Western Training Grounds in central Texas in hopes to restore the rangeland for continued military training. Small scale rainfall simulations are applied two and eight months post-application of seven different agronomic rates of composted waste treatment (0, 5, 10, 15, 20, 25, and 30 y^3/acre) in order to determine changes in infiltration rates.

July 2004 rainfall simulations, two months post application, indicate that composted wastes have not had sufficient time to incorporate into the soil matrix. Percent organic matter of the parent soil is the only significant variable of impact on maximum infiltration capacity. Composted waste treatments are concluded to have no effect on infiltration rates for any of the application rates in the summer rainfall simulations and are observed to exhibit very high variability in the amount of infiltration by a plot.

January 2005 rainfall simulations, eight months post waste application, are observed to continue the trend of high variability across all treatment application rates. This variability is attributed to masking any potential effects from the treatment

applications. Overall, this high natural variability disables the detection of potential effects of waste application treatments leading to the conclusion that composted waste applications do not affect infiltration on the Fort Hood Western Training Grounds. Runoff nutrient analysis observed nitrate-N to be well below Texas drinking water standards for all plots and phosphate to be above non-standardized values known to cause problematic algal growth. Natural rainfall events at intensities needed to generate runoff observed in this study are rare; therefore, nutrient pollution concern for local water bodies is low.

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INTRODUCTION

In 2001, the Texas Commission on Environmental Quality (TCEQ) issued two Total Maximum Daily Loads (TMDLs) for the North Bosque River watershed in an effort to reduce phosphorous loading by 50% within the river (Richards et al. 2008). The headwaters of the North Bosque River are located in the highest producing dairy county in Texas, leading multiple independent studies to identify confined dairy feeding operations as the main contributor to phosphorous in runoff. Additionally, it is a common practice of dairies to sell composted manure as fertilizer for local cropland, further increasing river phosphorous levels from these sources (Santhi et al. 2001; Bekele et al. 2006; Wagner 2010). In response to the TMDL, the Texas Legislature subsidized a composted dairy manure export program to remove an estimated 50% of the land-applied wastes to property outside the watershed, which has been utilized to stabilize exposed roadsides at construction sites and degraded areas of the Fort Hood Western Training Grounds (Richards et al. 2008). The Fort Hood military installation in central Texas has been experiencing degradation of their rangeland training areas due to heavy armory traffic used in training exercises, preventing recovery of the area for future use (Fort Hood Range Revegtation Pilot Project 2010). Composted dairy manure contains vital plant nutrients that assist in plant growth (Walter and Calvo 2009) as well as organic matter that promotes good soil conditions both aiding in rangeland restoration (Khaleel et al. 1981); but most importantly, it has the potential to restore water availability through increased infiltration

This thesis follows the style of *Rangeland Ecology and Management.*

and soil water storage. Water availability has been shown to be the predominant factor determining productivity on rangelands (Thurow et al. 1987), often dictating the rate of post disturbance recovery (Thurow 2000). By applying multiple rates of composted dairy manure onto a selected area within the Fort Hood Western Training Grounds, we will be able to establish the minimum rate and optimum rate of amendments needed in order to restore soil water functions and begin the feedback loop of restoration.

LITERATURE REVIEW

Biological Wastes

Agricultural livestock wastes (coincidentally, human wastes are similar) are often collected and combined together for storage as a slurry of solid and liquid wastes, which contains vital agricultural nutrients like potassium, phosphorous, and nitrogen excreted by individual animals (Hjorth et al. 2009). Despite seasonal variations in animal diet, there is very little variation in untreated slurry composition; consequently, slurries maintain many components of commercial fertilizers and can be utilized as such on croplands and rangelands (Hjorth et al. 2009). Depending on species, 55% to 95% of all nitrogen consumed by livestock is excreted in wastes as organically bound nitrogen, the primary similarity between commercial fertilizer and biological wastes (Hjorth et al. 2009). Biosolids (a term most often used to identify slurry from a wastewater treatment plant) are generally 40-70% organic matter, maintain a pH between 6 and 8, and contain heavy metals, pathogens, and recalcitrant compounds that can be detrimental to environmental health (Haynes et al. 2009).

The common heavy metals found in biosolids are divided up into four groups based on their bioavailability. Silver, chromium, tin, titanium, yttrium, and zirconium are the lowest soluble metals, have the lowest bio-uptake, and pose the lowest risk (Haynes et al. 2009). Arsenic, mercury, and lead strongly adsorb to soil but some biological uptake can occur that tends to accumulate in the non-edible regions (roots); therefore, these metals pose little risk (Haynes et al. 2009). Copper, manganese, nickel, and zinc accumulate in plants, but are phytotoxic at levels well below human toxicity and are, therefore, a low risk (Haynes et al. 2009). Cadmium, cobalt, and selenium accumulate in plants but are not phytotoxic at levels that are toxic to humans; therefore, these metals can accumulate at levels that pose serious health risks, making them the metals of highest risk (Haynes et al. 2009). Solid-liquid separation of wastes for easier handling, transport, and application results in heavy metals concentration within the solid portion and may pose an environmental threat in the wastes' designated future use (Hjorth et al. 2009). Petersen et al (2010) studied methods to reduce metal loading in animal wastes through alteration of livestock diet and found that a decrease in the amount of trace minerals, zinc and heavy metals in livestock feed had no detrimental growth effects on the animal and decreased the amount of these minerals found in manure (Petersen 2010).

Hormones and pharmaceuticals used in veterinary practices can be found in small amounts in animal wastes, very little is known about their environmental effects and degradation (Haynes et al. 2009). In 2001 in the United States, 16 million kilograms of antibiotics were used in veterinary clinics, with 70% applied in non-therapeutic uses (i.e. growth enhancers for added meat production) (Zhao et al. 2009). Most veterinary antibiotics are water soluble and are poorly absorbed in the intestines of livestock leading to detection of 30% to 90% of ingested antibiotics being excreted in feces or urine (Zhao et al. 2009). These antibiotics accumulate in water resources to levels considered toxic to ecosystems despite decomposition from exposure to natural elements (Zhao et al. 2009). Along with antibiotics, the presence of hormones in the environment can alter sensitive aquatic biology endocrine system functions at levels as low as a few nanograms per liter (Zhao et al. 2010). Hormones are naturally excreted from livestock, although many synthetics are utilized as growth promoters as well, and tend to be concentrated in manure (Zhao et al. 2010). In the year 2010, Zhao et al. (2010) estimate total excretion of hormones (natural and synthetic) in the United States to be 330 metric tons. In an effort to understand hormone persistence and origin in the environment, Zhao et al. (2010) studied water samples from three streams originating on a certified organic dairy farm (no use of hormones as growth promoters) and compared results to pure manure effluent to determine how much of the estrogen (the most common agricultural hormone naturally and synthetically) is sourced from the farm (Zhao et al. 2010). The dairy farm in the study participates in Whole Farm Planning practices such as application of manure as fertilizer onto their own property during the growing season from April to August, suggesting runoff from manure-applied lands to be yet another source of estrogen in local streams (Zhao et al. 2010). Two common forms of estrogen, E2α and E2β, were observed in stream waters consistently throughout the spring and intermittently throughout the rest of the year (Zhao et al. 2010). During times of land application of manure, estrogen concentrations in stream water increased, though levels were less than 1 nanogram per liter (Zhao et al. 2010). Summer and fall water samples rarely detected estrogen and the authors hypothesize this is due to the rapid degradation of natural estrogen due to high temperatures (Zhao et al. 2010). Conversely, conditions in winter like low temperature and low microbial activity enable estrogens to persist in the soil and be lost in runoff during spring snow melt, accounting for the high spring estrogen concentrations observed in water samples (Zhao et al. 2010). Under aerobic lab conditions, the authors were able to show 99.8% degradation of estrogen within eight months, coinciding to the average length of time manure is stored in open lagoons on the dairy farm and justifying the overall low amounts of estrogen measured in water samples (Zhao et al. 2010).

Petersen (2010) varied nutrient content of swine diets to reduce nutrient concentrations in wastes without detrimental production effects on swine growth. It was concluded that adjusting the nutrient content of diet to reflect the specific needs of a particular sex or life phase can significantly reduce the amount of nutrients excreted in manure without significantly affect the growth or marketability of livestock (Petersen 2010). Similarly, the use of more digestible forms of nutrients in feed, the incorporation of bacillus bacteria to reduce nitrogen volatilization, and the inclusion of phytase as the main form of phosphorous in feed can all reduce the total amount of nutrients excreted without affecting the growth of livestock as well (Petersen 2010). Phosphorous excretion is species dependant; cows retain most of the phosphorous ingested while pigs excrete 50-60% of total phosphorous ingested (Hjorth et al. 2009). Due to this, Petersen (2010) focused on the alteration of phosphorous in swine diet and observed that using other forms of phosphorous in swine feed can reduce the total amount of phosphorous excreted, and, thus, the amount of phosphorous entering the environment (Petersen 2010). The author noted that the more fiber contained in feed, the more waste was produced; increasing the total amount of solids contained in storage slurries will decrease the uniformity of the slurry which can affect the uniformity of nutrient application onto agricultural lands (Petersen 2010).

Biological Waste Disposal. In the United States, the main method of agricultural waste disposal is through storage in open lagoons. Disadvantages include requiring large acreage for storage, a high potential of water contamination, release of repulsive odors, and incubation of various pathogens (Vanotti et al. 2009). Waste disposal alternatives include disposal in a landfill, incineration, dumping at sea, and land application. Land application is the most economical of waste disposal methods available, the most beneficial to the surrounding environment (Haynes et al. 2009), and is one of the lowest users of non-renewable energy of all types of waste disposal mechanisms (Parker et al. 2006). Agricultural crops have been shown to similarly respond to waste applications as they respond to mineral fertilizers without risk of high uptake of heavy metals characteristic to many biosolids (Parker et al. 2006). Biosolids also contain inorganic compounds and organic matter that decay at varying rates causing them to persist in the applied soil and can be identified several years post application (Jaynes and Zartman 2005). Infascelli et al. (2009) compared animal waste production to the local need of nitrogen fertilizers in several provinces in Italy. The authors showed a wide disparity in the amount of nitrogen produced through manure and the nitrogen needed as fertilizer. In one province alone, crop nitrogen needs were only 4,000 tons/year while the production of nitrogen in manure was over 10,000 tons per year (Infascelli et al. 2009). This amount of nitrogen loading on agricultural lands and subsequent accumulation in waters due to runoff is unsustainable (Infascelli et al. 2009). The authors cited a need for more research into the costs of spreading manure onto fields and the amount of benefit in dollars provided by the manure to the crops in order to determine if this method of disposal was really feasible or not as the practice requires a lot of man hours, use of machinery, and use of non-renewable energy (example: oil to run tractors used in application) (Infascelli et al. 2009).

Overapplication of untreated wastes onto landscapes can result in increased salinization of soil, accumulations of toxic concentrations of heavy metals, decreased soil aeration, emissions of ammonia gas, use of non-renewable energy through transport and application of wastes, and the spread of livestock diseases (Hjorth et al. 2009). Hjorth et al. (2009) used separation technology to try and reduce the nutrient loads in animal wastes by separating the nutrient rich dry matter from the liquid slurry; which resulted in less cost for transport to agricultural fields through reduction of weight and the increased ability to produce energy through incineration of waste with a lower moisture content (Hjorth et al. 2009). Alternatively, composting wastes as a treatment pre-land application has been shown to reduce environmental hazards (Webber et al. 2010).

Airborne endotoxins were found by Ko et al. (2010) to be associated with confined animal feeding operations, with maximum observed cultures corresponding to storage of biological wastes in traditional open lagoons. Concentrations of airborne bacterial cultures ranged in values from 10^2 to 10^5 and were correlated to the amount of endotoxins in the air

(Ko et al. 2010). Environmental conditions affected the number of endotoxins recorded in samples with temperature and relative humidity negatively correlated to endotoxin observations (Ko et al. 2010). Closed waste storage systems such as various composting technologies, separation systems, and gasifying technologies were observed to release fewer endotoxins than lagoon storage (Ko et al. 2010).

The composting processes results in a loss of organic matter from the wastes, but also kills many pathogens and provides chemical stabilization and maturation of humic substances (Haynes et al. 2009). Composting of manure has been shown to reduce environmental hazards and to convert mineral nitrogen into more stable organic forms, reducing the loading from nutrients often found in runoff from sites where wastes have been topically applied (Webber et al. 2010). Composting of wastes has also been shown to degrade antibiotics, reducing their release into water resources when applied as amendments onto land (Zhao et al. 2009). One successful composting method is windrow composting, which consists of piling manure in long narrow piles and turning the piles on a regular basis (Webber et al. 2010). To reduce leaching of nutrients during the composting process it is beneficial to place the piles on a rather impermeable surface such as concrete, fly ash, or gravel, and to incorporate vegetative filter strips nearby local water resources (Webber et al. 2010). Vegetative filter strips have been shown to significantly reduce runoff and sediments in runoff from composting facilities into local waterways in central Iowa (Webber et al. 2010).

Waste Issues in the Bosque Watershed. The Bosque Watershed covers five central Texas counties and includes the North Bosque River, the Middle Bosque River, the South Bosque River, Hog Creek, and the final destination of Lake Waco. White et al. (2010) used Soil and Water Assessment Tool (SWAT) models to simulate stream flow and nutrient loading from the entire Bosque Watershed to quantify Lake Waco algal productivity and to establish if algal levels are predominantly controlled by the nutrient concentrations from one of the contributing rivers. From the SWAT model, the authors determined that the North Bosque River is responsible for 72% of streamflow, 79% of inorganic nitrogen, and 88% of inorganic phosphorous found in Lake Waco (White et al. 2010). The model simulation leads to the conclusion that the Middle Bosque River, the South Bosque River, and Hog Creek together source a very small amount of nitrogen and phosphorous in Lake Waco (White et al. 2010). The authors state that approximately 1.0% of total phosphorous in Lake Waco is due to atmospheric deposition and that the disturbance of sediments containing adsorbed phosphorous also contributes to total phosphorous levels recorded within the lake (White et al. 2010).

Wagner et al. (2010) studied phosphorous and nitrogen inputs into Lake Waco and determined that dairy concentrated feeding operations within the North Bosque River sub-catchment were a major contributor. Active dairy farms within the entire Bosque Watershed are concentrated within the North Bosque River sub-watershed; consequently, the North Bosque River was found to have high total nitrogen, high total phosphorous, and low nitrogen to phosphorous ratios (Wagner 2010). Low nitrogen to phosphorous ratios flowing into Lake Waco cause algal blooms degrading water quality for all uses, which includes recreation and drinking water (Wagner 2010). The authors determined that dairy operations and waste application fields contribute 34% to 42% of the total phosphorous in the North Bosque River and 23% to 28% of the total phosphorous in Lake Waco (Wagner 2010). Wagner et al. (2010) calculated nutrient export coefficients for the entire sub-watershed of the North Bosque River (they divided the total amount of nutrient in kilograms sampled in the river by the total land area) and determined phosphorous contributions to be 0.14-12.5 kg/ha/yr and nitrogen contributions to be 0.3-34.1 kg/ha/yr across the entire extent of the sub-watershed (Wagner 2010). This leads the authors to conclude that dairy operations “have the greatest potential for making a difference in Lake Waco through watershed management” (Wagner 2010).

Nemec et al. (2010) used Bacterial Source Profiling to identify fecal coliform sources contributing to the North Bosque River, which has been on Texas Impaired Waters list since 1992 (Nemec and Massengale 2010). Bacterial Source Tracking uses a library of known bacterial sources and allows comparison with unknown bacteria samples from the environment to allow determination of the unknown source (Nemec and Massengale 2010). Bacterial Source Tracking in this study had a high rate of source identification success, leaving only 11% of samples unidentified (Nemec and Massengale 2010). The authors found that cattle accounted for 43% of fecal pollution in the North Bosque Watershed, while 27% of fecal pollution came from sewage from municipal waste treatment facilities (Nemec and Massengale 2010). Fecal coliform colonies in water samples were highest in the head waters of the North Bosque River and observed colonies decreased as samples were taken farther downstream, ending with the lowest observed colonies at the mouth of Lake Waco (no samples were taken within the actual lake) (Nemec and Massengale 2010). Even though 43% of fecal coliform sources were from cattle, only 2% of watershed property at the time of this study were used as concentrated feeding operations or waste application fields (Nemec and Massengale 2010). The authors recommend that Best Management Practices (BMPs) to reduce fecal contributions from cattle operations will have the greatest effect on fecal contamination in the North Bosque River watershed (Nemec and Massengale 2010).

Excessive algal growth in the North Bosque River lead the TCEQ to implement a Total Maximum Daily Load (TMDL) in February 2001, identifying waste application fields as a major contributor as wastes from dairy cow operations locally are added to crop or pasture land as fertilizer (Bekele et al. 2006). The North Bosque River headwaters are within Erath County, which is also the top dairy producing county in Texas (Bekele et al. 2006). There are approximately 100 operational dairies with 40,500 cows (this fluctuates yearly with market feed and dairy prices), of which their waste is applied to 95 square kilometers of land (Santhi et al. 2001). This predominantly rural area has no significant point source of pollution, leaving reduction of non-point source pollution the goal of the TMDL; so, a voluntary composting program was started in 2000 to export dairy manure outside the watershed and, thus, reduce nutrient loads to the North Bosque River (Bekele et al. 2006). Within the watershed, about 50% of the dairies participated in the composting program totaling an export of 500,000 metric tons of waste in four years (Bekele et al. 2006). The authors state that measured reductions in in-stream phosphorous has a nonlinear response to land improvements and often leads to a lag response in water quality (Bekele et al. 2006). Phosphorous can continue to be released from re-suspension of sediments that contain adsorbed phosphorous and the high variability of short-term water data can mask small improvements in water quality (Bekele et al. 2006). After only four years, the composting program has reduced soluble reactive phosphorous concentrations by 19% to 23% in the areas that had the most participation in the haul-off program (Bekele et al. 2006).

Santhi et al. (2001) used a SWAT model to assess the dairy BMPs under consideration for the TMDL applied to the North Bosque River watershed (Santhi et al. 2001). Land within the watershed is predominantly agricultural range and pasture with a little cropland in the south, and there are eight wastewater treatment plants (Santhi et al. 2001). Local in-stream water monitoring stations have lead to the conclusion that dairy waste application fields are a major non-point source pollution source within the watershed. Past studies collaborate this conclusion by establishing correlations between acreage under waste application and phosphorous concentrations measured in local streams (Santhi et al. 2001). Other identified phosphorous sources within the watershed are urban runoff and municipal wastewater treatment plant effluent (Santhi et al. 2001). Through SWAT modeling, the authors show that the North Bosque River benefits more from proposed dairy BMPs rather than tighter restrictions on wastewater treatment plant effluent; therefore, dairies are indentified as the primary target of the TMDL (Santhi et al. 2001). The three BMPs identified are haul off or export of waste outside the watershed, calculating crop phosphorous requirements for rates of waste application instead of nitrogen crop requirements, and phosphorous diet restrictions in animal feed (Santhi et al. 2001).

Rangeland Desertification

Water availability is the predominate factor affecting rangeland productivity (Thurow et al. 1987). Ecosystems limited by water availability have slow recovery rates and are, therefore, difficult to restore; consequently, with global climate change these dry regions are predicted to become drier, leading to higher erodability of soils and higher erosion rates (Bautista et al. 2010). Restoration of degraded rangelands requires a long-term perspective as well as an adaptive management strategy to address dynamic ecosystem reactions to initial management strategies (Bautista et al. 2010). Removal of a degrading stressor to a rangeland landscape will not result in automatic ecosystem restoration; managed inputs need to be incorporated in order for restoration processes to initiate (Thurow 2000). The rate of the recovery on rangelands is determined by the rate of improvement of soil characteristics and associated water and nutrient cycles (Thurow 2000). Rangelands in the United States have been exploited over a long history of use, which has resulted in alteration of critical water and nutrient cycles at small and large scales (Wilcox 2010). Once a community's critical point is reached, desertification is inevitable (Thurow 1991).

Water Budget Inputs. The basic rangeland hydrologic cycle inputs are precipitation, interception, surface detention, and infiltration (Thurow 1991). Infiltration rates vary by species of plant cover, amount of litter accumulated by plant species, and by season due to variations in growth dynamics (Thurow 1991). Underneath tree-dominated vegetative communities, there can be a large build up of slowly decomposing litter which intercepts some of the precipitation enabling a smaller percentage of that which reaches the surface to infiltrate to the soil below. Litter layers can even completely prevent light precipitation events from ever reaching the soil underneath (Thurow and Hester 1997). Compared to grass-dominated vegetative communities that do not have the extensive accumulated litter layer, up to 80-90% of all precipitation will reach the soil (Thurow and Hester 1997). Shifts in vegetative communities can alter the percent of the total water budget interception occupies on a landscape as interception is the capture and re-direction of precipitation from the leaves, to flow down the trunk, and condense deposition around the base of the tree (Thurow and Hester 1997). Not only do shifts from tree species to shrub or grass species alter interception rates, but shifts from one tree species to another with higher leaf surface area (i.e. junipers) can capture more precipitation and alter water budgets (Thurow and Hester 1997). The average Texas rangeland (dominated by oak mottes) in peak conditions will have no interill erosion and no surface runoff, where approximately 7% of precipitation is intercepted, and approximately 81% of precipitation is infiltrated (Thurow 1991). Oak mottes contribute an additional litter layer interception component to water budgets that captures approximately 12% of precipitation (Thurow 1991). Rangelands in prime condition dominated by bunchgrass growth forms will intercept less than 1% of rainfall, infiltrate approximately 75% of rainfall, and lose about 24% of rainfall to runoff; comparatively, sodgrass growth form dominated rangelands will intercept less than 1% of rainfall, infiltrate approximately 54% of rainfall, and lose around

45% of rainfall to runoff (Thurow 1991). Bare ground does not intercept rainfall, can infiltrate up to 25% of rainfall, and will lose up to 75% of rainfall to runoff (Thurow 1991).

The maximum infiltration rate of the soil is determined by porosity, and begins with rapid adsorption that slows down as more pore spaces become filled with water until the soil is fully saturated (Thurow and Hester 1997). If precipitation continues to fall after the infiltration rate is maximized and the soil is fully saturated, then runoff will begin. Rough surface terrain (i.e. surface depressions) further acts as precipitation detention ponds, enabling water to infiltrate into the soil, or retards the onset of runoff by holding precipitation on-site after the maximum infiltration rate is met (Thurow and Hester 1997). On rangelands under livestock production, soil surface detention depends on the grazing intensity and stocking rate; moderate stocking rates tend to increase rough surface terrain while heavy stocking rates increase soil compaction, smoothing out surface storage depressions (Thurow 1991). Livestock grazing intensity and stocking rate can also affect infiltration rates based on vegetative impacts through grazing and soil compaction through trampling (Thurow 1991).

Runoff and Erosion. The basic rangeland hydrologic cycle outputs include runoff, evaporation, transpiration, and deep drainage (Thurow 1991). The type of vegetation indirectly influences runoff on rangelands as it controls the amount of interception and accumulation of litter, as stated above. Falling raindrops contain energy that is released on impact with the soil surface and can dislodge soil particles; these loose soil particles can be acquired and transported off site by runoff. Interception dissipates the energy of raindrops and reduces its erosive force on the soil surface, in fact, energy of rainfall impact is the most important mode of soil erosion (Thurow and Hester 1997). Exposed soil surfaces are also susceptible to crusting by detached soil particles that fill in soil pore spaces on site, sealing them off to infiltration and contributing to runoff (Thurow and

Charles A. Taylor 1999). In heavy clay soils, surface crusting can reduce infiltration by up to 90% (Thurow and Charles A. Taylor 1999). In extreme cases of severely degraded soils, a decrease in vegetative cover not only leads to more runoff and erosion but will change the surface reflectivity of an area and can inhibit cloud formation decreasing rainfall patterns regionally (Thurow 1991; Thurow and Charles A. Taylor 1999).

Erosion is a function of soil protective attributes such as total cover (vegetative and litter), plant biomass, and density of vegetation (Thurow and Charles A. Taylor 1999). Soil compaction and vegetation community destruction through excessive use by livestock or vehicle traffic will increase erosional forces, as demonstrated by Thurow et al. (1993) on the Fort Hood Military Base near Killeen, Texas. The authors found a general trend of increased erosion as the use of vehicle traffic increased on rangelands, with higher erosion rates observed when activity occurred on wet soil versus dry soil (Thurow et al. 1993). Collapsed pore spaces due to compaction slowed down water infiltration and decreased aeration of the soil, thus inhibiting plant root growth, nutrient uptake, and seedling emergence (Thurow et al. 1993). Reduction of vegetative communities increased the susceptibility to erosion during the next rain event, and impaired future use and productivity of the site (Thurow et al. 1993). The relationship of erosion from military vehicle use impacting vegetative communities further amplifying erosion creates a feedback loop that continues to degrade the rangeland condition through loss of soil and nutrients that cannot be regained naturally (Thurow 2000). The same is true for overstocking of livestock on rangelands, where reduction of vegetative communities increases bare ground that is susceptible to soil and nutrient loss from the destructive energy of rain drops that leads to greater grazing pressure on the vegetation that is left perpetuating the same cycle of degradation (Thurow 2000). To combat erosion and subsequent soil crusting, the only proven strategy is to accumulate cover, litter or plant biomass (Thurow and Charles A.

Taylor 1999). The more susceptible a site is to erosion processes, the higher the cost of rehabilitation (Thurow et al. 1993).

In an attempt at restoration to combat erosion and remove suspended soil sediments in runoff, Landry et al. (1998) compared the filtering ability of vegetative communities planted perpendicular to the flow of runoff on central Texas rangelands. The authors found that a mixed community of dense native grasses was best at reducing runoff volume and pollutant loads, including fecal coliform bacteria (Landry et al. 1998). For central Texas rangelands in this study, native grasses utilized were bluegrass and fescue (Landry et al. 1998). The authors observed that, as the vegetation collected sediments are heated and dried from exposure to sunlight, the fecal coliform populations decreased (Landry et al. 1998), in much the same processes as composting does to biological wastes.

Soil Characteristics. Rangelands are water limited, so maintaining high infiltration rates on rangelands is the key to keeping water onsite and preventing runoff (Thurow 2000). Soil structure determines infiltration rate; attributes such as stable aggregation, number of plant roots, number of fungal hyphae, diversity of microbial communities, and presence of organic matter are all integral to maintaining good soil structure (Thurow 2000). Soil aggregates are susceptible to dispersion from the impact of raindrops making plant foliar cover or litter cover important for maintenance of good soil aggregation (Thurow 2000). A litter layer on the surface of soil not only acts as a raindrop energy dispersant, but also contributes to soil organic matter, maintains even temperatures and moisture levels in the soil, and promotes microbial activity (Thurow 2000). Soil depth is an additional component of infiltration as it can limit total soil water storage, which is a critical component for plant production in shallow soils (Thurow 2000; Wilcox et al. 1988). Soil water storage capacity is a factor of soil adsorption and is affected by soil structure, organic matter content, and soil texture much like infiltration (Thurow 2000). Livestock production on rangelands has a trampling effect on soil, increasing compaction and bulk density from the breaking of soil aggregates by hoof force; trampling can also destroy cryptogamic crusts (cover by moss, algae, and lichens), which aid in slowing runoff and reducing evaporative loss of water from soil when these communities are present (Thurow 1991).

Nutrient distributions within the soil varies by vegetative community on rangelands; grassland soil tends to have a homogenous distribution of nutrients, while shrub-invaded grasslands tend to have a patchy distribution of nutrient concentrations beneath shrub canopies (Allington and Valone 2010). The soil between shrub canopies is considered too nutrient poor to support establishment of grasses, leaving the ground exposed to erosional forces creating a feedback loop that leads to desertification (Allington and Valone 2010). This loop is amplified when livestock grazing is introduced as greater grazing pressure is placed on the grasses in the shrub interspaces (Thurow 2000). Allington and Valone (2010) placed a grazing exclusion fence on a shrub-dominated desertified rangeland site in Arizona in hopes to reduce grazing pressure on perennial grass communities and possibly even re-establish uniformity of soil nutrient distribution. It took almost 40 years for the plot within the exclusion fence to restore perennial grass communities to a state of being significantly greater than the surrounding land still under grazing conditions (Allington and Valone 2010). Once perennial grasses had re-established, the islands of soil nutrients beneath the shrub canopies no longer existed within the fenced plot, while the authors still observed those conditions outside the fence (Allington and Valone 2010). Allington and Valone (2010) determined that the exclusion of livestock increased soil aggregate stability and increased infiltration rates within the fence by 24% compared to the surrounding landscape under grazing conditions. The authors deduced that increased infiltration reduced erosion and helped promote nutrient accumulation in the bare spaces allowing for grass growth and further accumulated of nutrients by establishment of a grass litter layer (Allington and Valone 2010). Excluding livestock long term from a degraded rangeland site encouraged restoration by increasing nutrient retention through increased infiltration (Allington and Valone 2010).

Vegetation. Total vegetative cover and standing biomass are both positively correlated to infiltration rates and work to enhance soil conditions that keep erosion rates low (Blackburn et al. 1992). A decrease in vegetative cover has an opposite effect by decreasing the amount of rainfall interception which increases the force of raindrops to break up soil aggregates (Thurow 1991). In the bare interspaces, the soil infiltration capacity is maximized by smaller precipitation events and causes erosion rates to be much higher (Blackburn et al. 1992). A decrease in plant biomass will also decrease soil organic matter by decreasing litter inputs, and a decrease in above-ground biomass leads to a decrease in below-ground biomass which also decreases soil aggregation contributing to increased erosion rates (Thurow 1991). Most importantly, vegetation spatial distribution and growth form are primary factors influencing runoff and erosion rates on rangelands (Blackburn et al. 1992). The growth form of vegetation will affect the amount of rainfall interception (Thurow et al. 1987) as total aerial cover is more important than basal cover or total biomass when it comes to precipitation interception (Wilcox et al. 1988). On the Edwards Plateau of Texas, landowners are looking to vegetative community alterations to increase water yields on property by removing brush and allowing grasses to dominate (Thurow et al. 2000). Water yields from grass-dominated communities are greater than that of brush- dominated communities due to the lack of canopy interception, litter interception, and stem flow allowing more water to infiltrate into groundwater resources or overland flow to surface water resources (Thurow et al. 2000). Supportive studies from

Sonora, Texas show junipers intercept 73% of annual precipitation, live oaks intercept 46% of annual precipitation, and grasses intercept only 14% of annual precipitation (Thurow 1998). Conversely, in areas that receive less than 18 inches of precipitation a year, alteration of vegetative communities through brush clearing will not result in increased water yields due to evaporative effects from the soil (Thurow 1998).

The alterations of vegetative communities due to degradation of rangelands results in major reallocations in the water cycle that are not well understood (Wilcox and Thurow 2006). The loss of vegetation cover and cryptogamic crusts degrades rangeland condition over time resulting in a complete loss of topsoil and a decline in local freshwater resources (Wilcox and Thurow 2006). Research suggests that degradation begins by exhibiting a shift of water and nutrient resources, abandoning the grass interspaces and concentrating under shrub/tree canopies to the detriment of native grass communities (Wilcox and Thurow 2006). As conditions continue to degrade, water, nutrients and soil are transported out of the community, preventing recovery and accelerating the feedback loop of degradation (Wilcox and Thurow 2006).

Research Studies on Rangeland Restoration. Slimani et al. (2010) excluded sheep grazing using an exclusion fence around a 12 ha plot from 1976 to 2006 on a rangeland in Northern Africa, while allowing uncontrolled grazing on the surrounding rangeland (Slimani et al. 2010). Natural vegetative growth in the study area is in a patchy pattern of perennial grass tussocks with a range of annuals in the interspaces during wet years or a bare silty crust during dry years (Slimani et al. 2010). After 18 years, vegetative and soil analysis showed that native conditions at the time of exclusion were maintained within the plot, while conditions surrounding the plot became severely degraded (Slimani et al. 2010). Vegetative species composition within the enclosure did not change due to the mound growth form of native grass tussocks that provide advantageous micro-site features such as an erosive barrier to soil and additional water and nutrients; while outside the enclosure non-native species were identified that have never before been observed in this region (Slimani et al. 2010). Initial grazing effects outside the enclosure were observed to be biotic in nature (i.e. decreased vegetative cover and loss of perennial species); but in the last two years of the study, degradation effects were no longer considered to be biological (Slimani et al. 2010). The continual loss of perennials by grazing exposed more soil to wind erosion changing the compositional features of the native soil by loss of the lighter clay component and surface organic matter; consequently, loss of nutrients prevented vegetative recovery from abrasive forces of sediments in the wind, thus feeding the degradation processes in this study (Slimani et al. 2010). Wind erosion is a dominant factor at this research site in Northern Africa and effects can be compounded by drought; drought leads to sparse vegetation which exposes more soil to erosive forces and the eventual loss of soil organic matter inhibiting future plant growth during post-drought conditions (Slimani et al. 2010). The final two years of the study resulted in desertification within the fenced plot due to accumulating desertification and wind containing sediments from bare soil in the surrounding environment (Slimani et al. 2010). At the conclusion of the study, a decrease in perennial grass cover was observed inside and outside the enclosure leading the authors to conclude that, so long as rangeland degradation is local in extent, grazing exclosures maintain native ecosystems (Slimani et al. 2010). When rangeland desertification causes regional modifications, exclusion of grazing livestock is not enough to protect native ecosystems (Slimani et al. 2010).

Thurow et al. (1988) compared four livestock grazing strategies and their effects on rangeland infiltration and erosion: moderate stocking rate with continuous grazing, heavy stocking rate with continuous grazing, moderate stocking rate with high intensity/low frequency grazing, and heavy stocking rate with short duration grazing. The research site was an oak-grassland rangeland in south Texas with similar initial infiltration rates calculated using a drip-type rainfall simulator across all treatment plots (Thurow et al. 1988). The authors found that, in as little as two years, infiltration rates were significantly different across treatments with the moderate-continuous stocking rate maintaining the highest infiltration and the high-continuous stocking rate maintaining the lowest infiltration (Thurow et al. 1988). The local climate and the applied stocking rate treatment were the two most important agents of change affecting the infiltration rate measured on the plots over time (Thurow et al. 1988). The seasonal flux of plant species composition (a decline in winter and a rapid growth in the spring) influenced fluxes in infiltration rates as was the season of the onset of drought, which decreased infiltration the most when drought occurred just prior to the onset of spring growth (Thurow et al. 1988). Overall, vegetative cover was found to be positively related to infiltration rate and negatively related to interill erosion (Thurow et al. 1988). Unlike infiltration, there were no observed seasonal flux in interill erosion under any of the grazing management treatments, instead the rate of erosion increased in a "stair-step fashion" with sharp increases correlated with drought periods and plateaus correlated to wetter conditions (Thurow et al. 1988). The accumulation of litter was found to negatively correlate to interill erosion by litter biomass minimizing the energy of rainfall impact, thus reducing the initiation of erosion (Thurow et al. 1988).

On the Edwards Plateau of Texas, Blackburn et al. (1992) ran rainfall simulations on rangelands to determine the erosion and infiltration rates under various vegetative cover and species growth form. It is assumed that, in this region of Texas, soils are not exposed to extreme freeze-thaw erosional processes and are, therefore, primarily affected by rainfall impact (Blackburn et al. 1992). The authors conclude that bunchgrass growth form has greater mulch accumulation, encourage more microbial activity, and promote formation of stable soil aggregates more than sodgrass growth form (Blackburn et al. 1992). The amount of cover by living and dead grass biomass was found to be the dominant factor over variability in infiltration and erosion rates on this rangeland site (Blackburn et al. 1992). The authors note that as erosion rates increased on research plots, sodgrass cover increased to the detriment of bunchgrass cover (Blackburn et al. 1992).

Rainfall Simulation Studies on Rangelands. Since the 1950s, research in water erosion and its effects on rangelands has been centered around the use of rainfall simulators (Norton and Savabi 2010). Early designs used a pressure nozzle system to distribute the raindrops, but consistency was difficult to maintain over the entire plot and length of simulation; more current designs utilize capillary drop simulators that rely on acceleration due to gravity in order to simulate natural rainfall events and maintain homogeneity of water application (Norton and Savabi 2010). Early simulators had various other mechanical problems, difficulty in field applications, and lengthy soil recovery time in between passes of the water spray nozzles, all together making research difficult and inaccurate (Norton and Savabi 2010). After standardization of simulator technology, the impact of water quality used in simulations was investigated with dramatically different results on soil detachment from impact and consequent surface sealing observed with water applications of various qualities (Norton and Savabi 2010). Most recently, researchers have redefined the basic rainfall/runoff relationship by challenging the assumption that entire plots contribute to runoff during simulated rainfall studies (Stone et al. 2008). Using a variable intensity rainfall simulator, Stone et al. (2008) determined that plots are a variable matrix of points contributing to runoff and infiltration at the same time. Modeling small plot water budgets under this assumption, output performance is better matched to real-life conditions than under the assumption that the entire plot contributes to runoff at once (Stone et al. 2008). The variable matrix plot model is more evident in coarse textured soils than in fine textured soils, although it still outperforms whole plot runoff contribution models in all soil types (Stone et al. 2008). Langhans et al. (2010) also examined this phenomenon of variable infiltration rates within a single plot, but in sloped European agricultural fields (Langhans et al. 2010). The authors attempt to define a dynamic hydrologic conductivity within the model assumptions, best described as runoff occurring while there is visible ponding on some areas of the plot, i.e. runoff and infiltration occurring simultaneously at varying rates across the simulated plot (Langhans et al. 2010). Variable hydrologic conductivity in agricultural fields is caused by high macroporosity allowing high infiltration in some areas and lower infiltration due to trapped air inhibiting the downward movement of water in other areas (Langhans et al. 2010). Crop residue on the surface also can provide sites of preferential flow for water to infiltrate the soil faster than bare areas between crops (Langhans et al. 2010). The authors conclude that no realistic rainfall intensity could maximize the infiltration rate over an entire plot, where the whole plot is contributing to runoff as in past assumptions (Langhans et al. 2010).

In a montane region in northern Colorado, stem density and rainfall intensity were found to be the best predictors of runoff in simulated rainfall studies (Flenniken et al. 2001). Rainfall rates of 100mm/hr for 100 minutes were selected to insure infiltration rates would be exceeded and runoff would occur on these landscapes (Flenniken et al. 2001). Plot characteristics found to affect runoff the greatest include vegetative cover and microchannel characteristics, which was directly related to forb and grass stem density (Flenniken et al. 2001). Channel characteristics are important on rangelands because overland flow tends to concentrate in these depressions, further eroding and deepening the channel (Flenniken et al. 2001). The authors also observed effects of cattle grazing and concluded that consumption of aboveground biomass decreases stem density and increases runoff on these montane rangeland landscapes (Flenniken et al. 2001).

Wilcox et al. (1988) performed a simulated rainfall experiment across various percent mountain slopes to determine the factors affecting infiltration in these extreme environments. Similar to other studies, vegetative cover and total biomass were found to heavily influence infiltration rates with slope having a weaker positive correlation to infiltration (Wilcox et al. 1988). Other cited studies exhibit variable response of slope to infiltration ranging from positive to negative correlation, highlighting the importance of vegetative cover in these ecosystems (Wilcox et al. 1988). Soils in the study were very shallow, restricting the total amount of water storage and, thus, infiltration; consequently, small increases in soil depth correlated to big increases in observed infiltration rate (Wilcox et al. 1988). As expected, organic matter in the soil increased the water holding capacity through increased soil aggregation equating to negative correlation between bulk density and infiltration rate (Wilcox et al. 1988).

Application of Wastes as Rangeland Amendments

Environmental Concerns. As indicated previously, biosolids contain heavy metals, pathogens, and recalcitrant compounds that can be detrimental to environmental health (Haynes et al. 2009). Unlike other elements of concern, heavy metals do not undergo degradation through microbial or chemical processes, although when added to soils many become unavailable due to adsorption to soil particles (Haynes et al. 2009). As a general rule, Haynes et al. (2009) has found that 80% of added copper remains unavailable, 60-85% of lead remains unavailable, 60-70% nickel and cadmium remains unavailable, about half of cobalt remains unavailable, 90% chromium remains unavailable, and 80-90% of iron remains unavailable. On the other hand, 40-60% of added zinc becomes available and 70-80% of manganese becomes available (Haynes et al. 2009).

The flush of metals after biological waste application increases competition for soil adsorption sites with other soil nutrients and, when combined with a flush of organic matter and nutrients, many heavy metals become temporarily more available (Haynes et al. 2009). In some instances, a release of previously unavailable metals will cause a decrease in soil pH (Haynes et al. 2009). Over time, the metals get re-adsorbed to soil microsites where they remain in the topsoil, unless mixing (through plowing or other mechanism) occurs, and then adsorbed metals and soil particles can leach into lower soil profiles and even groundwater below (Haynes et al. 2009). If leaching does occur, metal contamination of groundwater tends to be found with high levels of dissolved organic matter due to the strong adsorption association (Haynes et al. 2009).

Plant species differ in their ability to accumulate heavy metals. Some plant species hyper-accumulate metals and continue to grow (and are good for metal contamination site restoration), while others are highly sensitive to low levels of metals in the soil (Haynes et al. 2009). Uptake of metals from the soil by plants tends to accumulate in the roots, so there is little risk of consumption by grazing animals. The highest risk of exposure to grazing livestock is through consumption of soil (sticking to plants post application); therefore, to minimize this risk, it is recommended that livestock are not allowed access to acreage that has been amended with wastes for a period of time (Haynes et al. 2009). Haynes et al. (2009) observed that when livestock diets contain metals they tend to not retain a high percentage of that which is ingested, therefore, there is low risk to production of their meat and consequent human consumption. The addition of heavy metals can change soil microbial communities through a decrease in diversity, which can alter vegetative communities that live in symbiosis with specific soil microbes (Haynes et al. 2009). Heavy metals have been observed to increase in soil metabolic activity and decrease soil biomass carbon, both changing soil microbial communities (Haynes et al.

2009). Overall, many integral soil microbe functions are unaffected due to the high diversity of microbes that perform the same functions, so the loss of one species due to metal toxicity does not necessarily mean a complete loss of the community function performed by that species (Haynes et al. 2009).

Sukkariyah et al. (2005) applied a single application of biosolids with extremely high heavy metal concentrations to agricultural land to determine the long-term incorporation of metals into ecosystems. Their study spanned over 17 years. The biosolids applied by the authors contained 21.5 mg/kg cadmium, 3650 mg/kg copper, 210 mg/kg nickel, 640 mg/kg lead, and 2980 mg/kg zinc, compared to the composted wastes applied in the Fort Hood experimental research analyzed in this thesis, which contained 1, 91, 4, 5, and 112 mg/kg respectively (Sukkariyah et al. 2005). Biosolids often contain organic matter and metal oxy-hydroxides that can increase the adsorptive capacity of native soil and bind heavy metals found in the biosolids, preventing loss through runoff or leaching (Sukkariyah et al. 2005). The design of the plots in the experiment by Sukkariyah et al. (2005) were constructed to ensure complete isolation of lateral loss of biosolids, forcing metals to either leach, bind to the soil, or uptake through vegetation (Sukkariyah et al. 2005). Plots were tilled to a depth of 15 cm yearly with a crop of corn planted/harvested every year for analysis of vegetative uptake (Sukkariyah et al. 2005). As expected, copper, zinc, and nickel concentrations increased as biosolid application rate increased (Sukkariyah et al. 2005). After 17 years copper, nickel, and zinc were found mainly in the top 20 cm of soil, and were not significantly different between treatments, but were recorded to be significantly higher than the control plot of no biosolids application (Sukkariyah et al. 2005). The corn crop uptake of metals amounted to a very small proportion of total applied with zinc uptake amounting to 0.52%-1.1%, uptake of copper amounting to 0.07%, and data is unavailable for nickel uptake (Sukkariyah et al. 2005). After 17 years, 95% of the total metal application through biosolids was accounted for either in the soil or vegetation, leading the authors to conclude minimal groundwater contamination risk in similar environments and soil conditions (Sukkariyah et al. 2005). Leaching of metals bound to clay-sized particles is an often cited concern, but Sukkariyah et al. (2005) found no significant effects on the downward dispersal of clay-sized content. Biosolid application increased the copper, cadmium, and zinc concentrations in the dispersible clay fraction of the test plots, but again there was little downward dispersal so sampled concentrations dropped sharply after 20 cm of depth (Sukkariyah et al. 2005).

As discussed previously, the most studied hormone found in animal wastes is estrogen which is known to rapidly degrade in aerated soils, therefore the risk of contamination from leaching or runoff is low (Haynes et al. 2009). The risk of estrogen contamination is highest if water is applied immediately post application of wastes, when the estrogen has not had time to degrade (Haynes et al. 2009). It has been shown that many hormones accumulate in the roots of plants, and are, therefore, low risk to consumption by grazing animals (Haynes et al. 2009). The highest risk of exposure to grazing livestock is through consumption of soil (i.e. staying on plants post topical application in the presence of vegetation), therefore, to minimize the risk of livestock consumption it is recommended that livestock is not allowed access to acreage that has been amended with wastes for a period of time (Haynes et al. 2009). Conversely, micro-organisms found in composted swine manure have been shown to speed up the time it takes hormones to degrade in the environment and could mitigate the threat of contamination (Parker et al. 2006).

Biological waste amendments to soil will increase the electroconductivity of a soil due to the increase in ions such as magnesium, calcium, and chloride, consequently increasing the salinity of the soil. Additional water application causes many of the salts to leach deeper into the soil profile returning the surface to prior electroconductive conditions (Haynes et al. 2009). Amendments of biosolids also change the pH, although inherent conditions in the parent soil determine whether it is an increase or a decrease in pH (Haynes et al. 2009). At pH 7 and above within the wastes applied, ammonia volatilization is favored; especially when there is ammonia prevalent in the parent soil (Haynes et al. 2009). If application of biosolids is not matched to community nitrogen needs, then excess nitrogen in the form of nitrate will leach out into local water sources in addition to ammonia volatilization losses (Haynes et al. 2009). The potential to lose a high amount of applied nitrogen highlights the need to apply waste amendments based on calculated nitrogen needs of the vegetative community in existence on the application site. Vegetative communities utilize higher amounts of nitrogen than applied phosphorous, therefore phosphorous is commonly accumulated in amended soils (Haynes et al. 2009). Phosphorous in soil has a greater chance of being lost through runoff than by leaching, unless the soil is sandy and of low pH or already phosphorous saturated due to repeated waste applications (Haynes et al. 2009). Management of runoff from waste-amended sites is recommended to control phosphorous accumulation in local water bodies. Some researchers suggest calculating application rates based on community phosphorous needs instead of nitrogen in order to reduce the amount of phosphorous lost to runoff. Oxygen-demanding compounds (i.e. Biological Oxygen Demand and Chemical Oxygen Demand) can also be transported through runoff from waste-amended sites and deposit in water resources where they can accumulate and lower water quality (Khaleel et al. 1981).

Even though biological wastes contribute nutrients in applied environments, they can also lead to a total loss of nutrients as well. One path of loss of nitrogen from application of biosolids is through ammonia volatilization. Harmel et al. (1997) found the volatilization is highest immediately post application, that volatilization fluxes occur on a diurnal pattern under warm temperatures, potential evapotranspiration influences nitrogen volatilization, and the rate of loss is significantly reduced in a matter of days (Harmel et al. 1997). Losses of nitrogen ranged from 9.5% to 16.6% of applied NH_3 at the low level of application of 6.7 Mg/ha, versus the high application rate of 17.9 Mg/ha losses of nitrogen ranged from 8.3% to 12.1% of applied NH_3 (Harmel et al. 1997). The biosolids in this study were surface applied; therefore, these observed rates of volatilization should not be anticipated if the biosolids are incorporated into the soil through tillage (Harmel et al. 1997). The authors argue that plant-available nitrogen is the more accurate measure of how much biosolid to apply versus the common solution using total nitrogen to determine application rate (Harmel et al. 1997). This will reduce the rapid loss of nitrogen post application and will increase the number of forms of nitrogen remaining available for plant uptake (Harmel et al. 1997).

Treatments, such as composting, effectively reduce the amount of pathogens present in biological wastes, but re-growth of those pathogens has been observed post-field application after wetting events through rainfall or irrigation (Parker et al. 2006). In particular, an increase in the measurable fecal coliform colonies and salmonella has been observed (Parker et al. 2006). Tanner et al. (2005) measured coliform and coliphages in air samples downwind of biosolid application sites distributed by common spray method. Spray application is usually performed using a large tank containing fluid biosolids being pulled behind a tractor with a pressurized hose dispersing the fluids in a fan pattern behind the tank (Tanner et al. 2005). The authors did not detect either of the two bacterial colonies in the air during application indicating very little risk of exposure (Tanner et al. 2005). It has been previously shown that viruses adsorb to particulate matter and Tanner et al. (2005) conclude that these microsites also provide adsorption for bacteria, thus limiting aerosolization. The highest risk of exposure to grazing livestock is through consumption of waste-amended soil (i.e. staying on plants post topical application in the presence of vegetation). Therefore, to minimize the risk of livestock consumption, it is recommended that livestock is not allowed access to acreage that has been amended with wastes for a period of time (Haynes et al. 2009).

Environmental Benefits. Application of biological wastes onto rangelands promotes land management by “increasing forage yield and quality, increasing carbon sequestration, improving wildfire mitigation management, reducing chemical fertilizer use, reducing soil erosion, increasing water resource protection and enhancing soil moisture, infiltration, and drainage” (McFarland et al. 2010). Many of these benefits originate from stewardship of the soil and beneficial changes in soil chemistry initiated by biological waste application. Applications of biological wastes to land increases the carbon content of soil; consequently, this has been linked to increased soil aggregation, increased total pore space and porosity, decreased bulk density, increased water holding capacity, and increased hydraulic conductivity (Khaleel et al. 1981; Haynes et al. 2009). Soil organic matter content and soil aggregation have additionally been observed to be conversely related to sediment loss and runoff volume (Khaleel et al. 1981). A major component of biosolid carbon is resistant to decay, and aids in maintenance of the observed increase in soil organic matter as well as an increase in cation exchange capacity of the soil (Haynes et al. 2009). These structural soil changes lead to an increase in infiltration rates and an increase in field capacity to hold water as a function of total capacity and capacity at wilting point (Khaleel et al. 1981; Haynes et al. 2009). Both long-term (18 to 85 years) and short-term (1 to 6 years) studies indicate an increase in carbon post biological waste application; although, utilization of liquid wastes will result in a smaller increase in organic carbon than utilizing solid wastes (Khaleel et al. 1981). The increase in soil carbon observed across many time scales in research is highly varied and therefore no overarching relationship between application rate and soil carbon increase has been developed (Khaleel et al. 1981). Supporting the lack of an application rate to carbon increase relationship is the indication that even carbon decomposition over time does not follow a linear relationship, further complicating attempts to establish a conjunction (Khaleel et al. 1981). Both long-term and short-term studies demonstrate a sustained decrease in soil bulk density post waste application as a result of organic matter diluting the dense mineral soil and, in spite of varying soil textures, a strong linear relationship can be indicated between waste application rate and percent reduction of bulk density (Khaleel et al. 1981). Effects of biological waste applications on hydraulic conductivity are much less studied and much more varied, but the general trend is an increase in hydraulic conductivity with waste applications (Khaleel et al. 1981). An exception occurs if the wastes are high in sodium. Highly saline wastes can cause the dispersion of aggregates and, therefore, retard the movement of water through the soil profile (Khaleel et al. 1981). Research on effects of biological waste application on infiltration rates are less common and indicate varying results from improvements in infiltration to increased runoff, highlighting a need for better understanding of this particular soil characteristic and its response to waste amendments (Khaleel et al. 1981).

Additions of organic matter have also been observed to increase microbial activity of soils (Haynes et al. 2009). Barbarick et al. (2004) applied biosolids onto rangelands in order to assess their affects on microbial communities within the soil over a span of six years. Much concern has been raised on biosolid alterations of rates of carbon and nitrogen mineralization in ecosystems, thus affecting microbial community dynamics directly and indirectly affecting vegetative communities as 85 to 95% of native grassland species are in symbiosis with arbuscular mycorrhizae (Barbarick et al. 2004). Biosolid application rates of 0, 30, and 40 Mg/ha were applied in this study and considered to be extreme amounts, even though rates at 45 Mg/ha have been suggested as beneficial rates of application by other studies. Contrary to stated hypothesis, biosolid applications increased the percentage of root samples that were colonized with arbuscular mycorrhizae, equating to a 33% increase of western wheatgrass root associations and 23% of blue gramma root associations (Barbarick et al. 2004). The observed colonization increase was correlated to an increase in total soil organic carbon from the biosolid application, as was an increase in metabolically-active biomass (Barbarick et al. 2004). The authors feared that trace elements (i.e., heavy metals) in the biosolids would reduce carbon dioxide respiration of the microbial communities, but they did not observe a decrease throughout the six-year study (Barbarick et al. 2004). Overall, these results indicate that biosolid-amended sites are able to maintain or possibly even promote microbial communities in rangeland soils (Barbarick et al. 2004).

As highlighted above, biological wastes contain vital plant nutrients like potassium, phosphorous, and nitrogen and can be utilized as agricultural fertilizers. Research has shown that adding organic matter and fertilizer together to a soil will produce greater yields than applying just fertilizer (Sikora and Yakovchenko 1996). Sikora and Yakovchenko (1996) attempt to quantify and compare soil organic matter mineralization of amendments of biosolid compost (acting as the organic matter) versus a mixture of biosolid compost with municipal solid wastes (acting as the organic matter plus fertilizer). The authors found that no stimulation of soil organic matter decomposition occurred by the addition of compost materials to municipal solid wastes (Sikora and Yakovchenko 1996). Decomposition of the mixture of the biosolid and the municipal waste was less than 5% after 1440 hours while decomposition of just the biosolid compost was 8% to 14% after 1440 hours (Sikora and Yakovchenko 1996). The authors hypothesize that often cited observations of increased soil organic matter decomposition are in fact a result of mineralization of the compost materials and not actual soil organic matter decomposition (Sikora and Yakovchenko 1996). This study demonstrates that application of composted biosolids contains enough nutritive value that nutritious amendments do not benefit environmental uptake.

Research Studies Utilizing Waste Amendments. Historical trends of abandonment of cultivated fields in the Mediterranean resulted in severe erosion, leading Walter and Calvo (2009) to study the application of wastes to restore ground cover in hopes of reducing degradation processes. Their study found no change in soil pH and not enough soluble salt accumulation to inhibit plant growth after applications of 0, 40, 80, and 120 megagrams of biosolids per hectare (Mg/ha), disputing often cited concerns of soil side effects from biosolid applications (Walter and Calvo 2009). The authors found a significant increase in percent plant cover for 40 and 80 Mg/ha application treatments in the first year, but not for the 120 Mg/ha application treatment due to remaining wastes on the soil surface preventing plant emergence (Walter and Calvo 2009). After the first year, total percent plant cover increased significantly for all treatments for all remaining years (Walter and Calvo 2009). Plants uptake heavy metals found in biosolids and can accumulate them; therefore, the authors measured accumulated heavy metals in the vegetative biomass and found all observations to be below phytotoxic levels (Walter and Calvo 2009). Walter and Calvo (2009) did report a decrease in species richness with waste amendments supporting plant species that grow in high nutrient levels and that grow rapidly. This resulted in a decrease in perennial species and an increase in annual species, leading the authors to hypothesize it is the shallower root systems of annuals that take advantage of the topically applied nitrogen and gain a competitive advantage over deeper rooted perennial species (Walter and Calvo 2009).

Similarly, Cuevas et al. (2000) applied composted municipal solid wastes to a semiarid shrubland in central Spain in hopes to improve plant establishment in a degraded area, to promote organic biomass additions to the soil, and to increase root production to minimize runoff and erosion. The authors analyzed soil chemical properties and evaluated vegetative growth one year post application of the municipal solid wastes (Cuevas et al. 2000). After one year, waste application was observed to decrease vegetative species diversity on all treated plots, but a significant increase in total vegetative biomass was also recorded (Cuevas et al. 2000). Municipal solid waste applications significantly increased inorganic nitrogen, phosphorous, potassium, and electrical conductivity of the soil after one year (Cuevas et al. 2000). This measured increase in vital plant nutrients is the primary factor causing increased vegetative production and caused the reduction in species richness due to sensitivity of some plant species to high nutrient conditions (Cuevas et al. 2000). Heavy metal concentrations in the soil increased with municipal solid waste application, but increases were found to be significant for only zinc, lead, and copper (Cuevas et al. 2000). Soil organic carbon, total nitrogen, cation exchange capacity, and pH did not significantly increase after one year post waste application (Cuevas et al. 2000). The authors hypothesized that the observed increase in nitrogen caused an increase in mineralization of the indigenous soil organic matter, therefore no significant increases in total organic matter were observed (Cuevas et al. 2000). Increased mineralization of inherent soil organic matter eventually leads to an observed decrease in total soil organic matter post municipal solid waste additions despite the addition of organic matter (Cuevas et al. 2000). The authors challenge that if short term responses in vegetation and nutrients are to be translated into long term gains on rangelands, then research studies on the effects of waste applications must extend past the three or four years usually examined (Cuevas et al. 2000).

Benton and Wester (1998) studied the residual effects on vegetation post application of biosolids over time, comparing one single biosolid application to yearly repeated applications. Biosolids were applied at rates of 0, 7, 18, 34, and 90 megagrams per hectare (Mg/ha) with total plant production and vegetative cover being the examined response variable (Benton and Wester 1998). The authors observed vegetative response to biosolid applications to be variable to the amount and seasonal timing of rainfall across all treatments (Benton and Wester 1998). In the case of a single application of biosolid amendment without regard to the application rate, there was a greater plant response when it was applied in the dormant season (Benton and Wester 1998). The authors did note an immediate growth response to biosolid application in the growing season, but in terms of total production, it was not as appreciable as the response from dormant season application (Benton and Wester 1998). This variable seasonal response is hypothesized to be due to the increased soil microbe stimulation over the time span from the dormant season application to the beginning of vegetative growth in the growing season, as well as less moisture loss from the soil over the same time period due to a mulch effect from the biosolids (Benton and Wester 1998). Benton and Wester (1998) observed little evidence of a carry-over effect of vegetative response into the second year after a single application of biosolids. Yearly repeated biosolid applications increased plant production compared to the single application plots over the same time interval only when adequate precipitation was received, in other words water had to not be a limiting factor (Benton and Wester 1998). When precipitation was limiting, less vegetative production was observed from all waste amendment treatments (Benton and Wester 1998). Finally, the authors found that all rates of biosolid applications increased vegetative production, with the highest increase in production observed under 7 Mg/ha and the least amount of increase in production observed under 90 Mg/ha (Benton and Wester 1998).

Jurado-Guerra et al (2006) applied low and moderate rates of biosolids onto desert rangelands in the Trans-Pecos region of Texas during the dormant season and during the growing season to compare vegetative response. Three levels of biosolid application rates at 0, 18, and 34 Mg/ha were utilized; concurrently, a non-organic mulch replicate made of nylon and polyester was applied as a fourth treatment to evaluate if chemical effects from the waste applications were contributing to observed results or if cover and moisture retention through protective mulch capabilities were contributing to observed results (Jurado-Guerra et al. 2006). Post biosolid application, surface nitrate nitrogen levels significantly increased under dormant season application across all rates compared to control plots and compared to growing season application (Jurado-Guerra et al. 2006). Subsurface nitrate nitrogen levels were not significantly different post biosolid application regardless of season of application and rate of application (Jurado-Guerra et al. 2006). Dormant season application of biosolids enabled an earlier onset of mineralization and nitrification, resulting in an overall higher soil nitrate nitrogen level compared to growing season application (Jurado-Guerra et al. 2006). Warmer temperatures at the time of application in the growing season also resulted in the loss of more nitrogen due to higher rates of volatilization (Jurado-Guerra et al. 2006). This observed seasonal effect of application diminished after one year post application and could no longer be observed in the top soil, but was still present in the subsoil (Jurado-Guerra et al. 2006). The authors attribute persistence of nitrogen in the subsoil to the longer residence time of dormant season application allowing for better incorporation into the soil profile (Jurado-Guerra et al. 2006). The nylon-polyester mulch-like substance did not have any effects on soil nitrate nitrogen, leading the authors to conclude that observed increases in nutrients were caused directly by the biosolid application and not indirectly through alteration of soil temperature, retention of soil moisture, and promotion of favorable microenvironmental conditions

(Jurado-Guerra et al. 2006). Jurado-Guerra et al. (2006) note that deficit spring and average summer precipitation resulted in no differential effect across the season of application of biosolids and hypothesize that under low rainfall conditions, lower rates of biosolid applications are more beneficial due to increased interception and retention by biosolids applied at higher rates (Jurado-Guerra et al. 2006).

Park et al. (2010) compared the post-application effects of beef cow manure, swine effluent, and commercial fertilizers on a cropland planted with corn in Oklahoma. Both livestock wastes were found to produce significantly higher corn yields than commercial anhydrous ammonia, and were both evaluated by the authors to be reasonable substitutes for commercial fertilizer (Park et al. 2010). Swine effluent is in a liquid form and is, therefore, able to be applied to fields through existing irrigation infrastructure, leading to less costly application than beef cow manure and commercial fertilizer, which had to be applied using additional equipment and labor (Park et al. 2010). This capital savings from the use of swine effluent will not translate onto rangeland application as they are not irrigated and have no irrigation infrastructure in place (Park et al. 2010). The commercial fertilizer was the costliest of the three treatments and generated the lowest yield increase; beef cow manure generated the highest yield increase in number, but was not significantly different from yields under swine effluent application (Park et al. 2010). The treatment application rates were calculated to contain equivalent nitrogen loads so there was no significant difference between amount of nitrogen applied, therefore yield differences are due to other factors characteristic to the wastes (Park et al. 2010). Park et al. (2010) hypothesize that the increase of yield under waste application was due to the inclusion of micronutrients and organic matter, whereas commercial fertilizers do not include these elements (Park et al. 2010). The authors also note that repeated waste application as a fertilizer can maintain soil pH while repeated commercial fertilizer application can lead to acidification of soil (Park et al. 2010).

Ippolito et al. (2010) studied the application of biosolids as either one application or twice yearly applications and observed changes in soil and vegetative characteristics over time. Both application schedules resulted in an increase in perennial grass cover by second year to the expense of perennial forbes and annual grasses (Ippolito et al. 2010). Contrary to a previous study, invasive plant species cover at this site was found to decrease on many of the plots under biosolid application (Ippolito et al. 2010). The authors do concede that weather conditions in the second year were more favorable than the previous year in terms of precipitation during the growing season and that higher water availability influenced some of the vegetative growth observed (Ippolito et al. 2010). Observed soil chemistry changes in the study were expected by Ippolito et al. (2010) as elements contained in the wastes increased those found in the soil post application. Soil pH reduced reflecting the lower pH of the biosolid compared to the native soil, while electroconductivity, nitrate nitrogen, ammonium nitrogen, total carbon, total nitrogen, cadmium, copper, zinc, and phosphorous increased with increasing composted biosolid application rate (Ippolito et al. 2010). The authors observed that in plots of application rate 21 Mg/ha and above, total soil carbon was elevated after one single application of composted biosolid up to 14 years later (Ippolito et al. 2010). This finding along with the assumption that acidic waste amendments (like the one used by Ippolito et al. 2010) have total carbon associated with only organic matter, implies that soil organic matter status of rangelands can improve long term using application rates above 21 Mg/ha (Ippolito et al. 2010). This assumption is a valid one because acidic pH indicates low carbonate concentrations meaning low inorganic carbon amounts, leaving total carbon to be mainly organic in form (Ippolito et al. 2010).

McFarland et al. (2010) applied biosolids processed by aerobic bacteria, anaerobic bacteria, composting, or lime stabilization to rangelands at calculated agronomic rates based on nitrogen needs of vegetative communities and compared consequent vegetative quality for livestock grazing. Biosolid applications in all treatments increased vegetative production over that of the control plot 12 months post application, but vegetative communities were observed to be invasive species dominated (McFarland et al. 2010). Eight out of nine dominant plant species identified on amended plots are considered non-native to United States rangelands and of fair to poor livestock forage quality (McFarland et al. 2010). Despite the low forage value ranking of plant species, the various biosolid applications increased the average crude protein content of forage compared to the control plot, which is particularly important to livestock diets (McFarland et al. 2010). The authors state that biosolid amendments are beneficial for restoring forage productivity on rangelands both economically and environmentally, but are not valuable for the enhancement of native vegetative growth (McFarland et al. 2010).

Brenton and Fish (2007) studied biosolid waste leachate post application at 0, 7, 18, 34, and 90 Mg/ha on rangeland soil cores. The authors hypothesize that in water-limited rangelands, the chance of trace element contaminated runoff or leachate is low, and they found this to be true for all elements but nitrate (Brenton et al. 2007). Across all applied rates of biosolid application, nitrate in leachate was observed to exceed the United States Environmental Protection Agency (USEPA) guidelines for drinking water (Brenton et al. 2007). For the fine sandy soil sample, biosolid application did not affect the nitrate in leachate, as it was above drinking water standards across all application treatments; adversely, in the loamy soil sample nitrate leachate was observed to be similar across all application rates except 90 Mg/ha where it was significantly higher (Brenton et al. 2007). The authors state that the actual amount of water applied in order to obtain measurable amounts of leachate in this study are extremely rare in dry rangeland climates and, therefore, the actual occurrence of contaminated leachate reaching groundwater is equally as rare (Brenton et al. 2007).

Jaynes et al. (2003) studied the decomposition of biosolids applied once to the landscape and aged over a range from two years to seven years. The authors found a significant increase in soil nitrogen post-application with 80% decomposition over seven years that was hypothesized to reflect organic matter decomposition as decreases in both paralleled (Jaynes et al. 2003). The 175% increase in ash content of the biosolids corresponded to a high loss of organic matter over seven years with a large majority of the conversion within the first five years (Jaynes et al. 2003). Inorganic phosphorous solubility decreased over time as 18% was soluble in fresh biosolids and only 2% was soluble in biosolids aged five years or more (Jaynes et al. 2003). Even though soluble phosphates are sparse, they can still result in high concentrations in runoff as field conditions accumulate much less water from precipitation than that used in a lab analysis leading to dilution of true field concentrations (Jaynes et al. 2003). Out of all the heavy metals found in the applied biosolids, only zinc and copper were observed to leach out of the soil. No losses of lead, chromium, or mercury were measured as they all remained within the soil/biosolid complex (Jaynes et al. 2003).

Weindorf et al. (2006) studied composted organic material application onto urban soils to determine if it has any influence on the infiltration rate, soil water content, and/or the vertic properties of clay. This study occurred on an assortment of soils in the Dallas, Texas area many of which have a high shrink/swell capacity (Weindorf et al. 2006). The compost in this study was applied at depths of 0 cm, 2.5 cm, 5.0 cm, and 7.5 cm across plots of 3.34 m^2 and was tilled to a depth of 20.3 cm (Weindorf et al. 2006). The authors found that composted organic material amendments influenced infiltration rates variably along a ‘dry to wet’ continuum with one of two general results (Weindorf et al. 2006). If the soil was initially dry with large visible cracks, then the addition of compost resulted in decreased infiltration rates through reduction of the soil cracks. Soil cracks were reduced from compost characteristics that elevate soil moisture content (Weindorf et al. 2006). If the soil was initially moist, then the addition of compost resulted in increased infiltration rates through increased macroporosity and decreased bulk density (Weindorf et al. 2006). The authors determined that infiltration rates had a stronger association with native soil mineralogy and climatic conditions than actual compost applications; alternatively, the compost did increase soil moisture content significantly at a few sites and visibly reduced soil cracking at all the amended sites compared to controls (Weindorf et al. 2006).

White et al. (1997) studied a single application of biosolids at varying rates onto rangelands over nine years, longer than any previous study has followed biosolid applications. The authors argue that the high organic matter content characteristic of biosolids, relative to organic matter inherent to native soil, is a good source of nitrogen and phosphorous for vegetation and microbial communities (White et al. 1997). Despite the benefits of more organic matter with greater biosolid application, rates above 45 Mg/ha were found to not contribute significantly to long-term soil and vegetative benefits any more than rates below (White et al. 1997). This maximum rate of 45 Mg/ha is additionally recommended due to the minimization of heavy metal loading and the maximization of nutritional benefits provided from the biosolids (White et al. 1997). Conversely, soil microbial communities responded beneficially to high rates of biosolid application and were observed to increase significantly under rates of 45 Mg/ha and above after several growing seasons (White et al. 1997). The authors found that all biosolid-treated plots had reduced runoff compared to untreated plots under natural and simulated rainfall conditions (White et al. 1997). Nutrient concentrations calculated from plot runoff samples were greater from biosolid treated plots than untreated plots due to nutrients contained in biosolids, but the total mass of nutrients lost was less in biosolid treated plots due to a reduced volume of total runoff (White et al. 1997). This response is critical because runoff contributes to suspended sediments and nutrient accumulation in waterways and causes more degradation of water resources worldwide than point source polluters (White et al. 1997).

OBJECTIVES

The primary objective of this study is to determine if composted dairy waste applications have an effect on runoff and infiltration on degraded rangelands within the Fort Hood Western Training Grounds near Killeen, Texas. Small plot rainfall simulations will be utilized to compare seven different agronomic rates of application by measuring total runoff and infiltration; additionally, the runoff water quality will be analyzed to identify any non-point source pollution concerns for local water resources. We expect to determine a minimum application rate that causes a significant increase in infiltration and a maximum application rate where pollution concerns from runoff nutrients outweigh the beneficial increase in infiltration.

METHODS

Study Area

The Fort Hood military installation is located within Coryell County in central Texas in a sub-humid, temperate, continental climate. Coryell County geology consists of rolling plains underlain with hard limestone on the high ridges or soft limestone/marly clay on the lower hills and plateaus (McCaleb 1985). Precipitation is distributed relatively uniformly throughout the year averaging 34 inches (87 centimeters) per year. Average temperature ranges from 9.5°C in the winter to 28.3°C in the summer (McCaleb 1985). Coryell County occupies 676,249 acres of the Grand Prairie Region historically consisting of high quality tall grasses, mid grasses, and forbs. Current vegetative communities are dominated by short grasses, mid grasses, and poor quality forbs as well as invasion of oak and juniper tree species (McCaleb 1985). In 1983, rangeland was the most widespread land use covering 68% of the county, considerably higher than the second most widespread land use of cropland covering only 18% of the county. Urban land use in Coryell County accounts for only 2% of the total land area (McCaleb 1985).

The Fort Hood military installation covers 187,000 acres, approximately 28% of Coryell County, and leases many of its training areas to local farmers for livestock use (McCaleb 1985). The research site was located within the Fort Hood Western Training Grounds; an area used both for military training exercises and livestock production. The two composite soils within the research site are Brackett-Topsey and Slidell silty clay, which can be found over approximately 17% of Coryell County and are primarily in the lower formations of marly clay (McCaleb 1985). A major limitation of these two soils is their erodability, ranging from moderate to severe. The primary soil is the Slidell silty clay which is a fine, montmorillonitic, thermic Udic Pellusterts (McCaleb 1985). Generally, this soil is well drained with a high available water capacity but slow permeability which increases the drier the soil becomes. This soil consists of 1% – 3% slope with a moderate hazard of erosion and a high shrink/swell capacity limiting many urban applications (McCaleb 1985). Brackett-Topsey soil covers considerably less scope of the research site, being found only in a small locale of the 10 y^3/acre plot. It is a fine-loamy, carbonatic, thermic, shallow Typic Ustochrepts associated with three to eight percent slope. Generally, this soil has a slow permeability with moderate available water capacity and severe hazard of erosion. These soils experience high shrink/swell capacity limiting urban applications, are poorly suited for cropland and pasture land, and typically are utilized as rangelands (McCaleb 1985). Both the Brackett-Topsey and Slidell silty clay are moderately alkaline.

Vegetation communities observed within the treatment plots ranged from 87.2% to 63.2% perennial species and 36.8% to 12.8% annual species. Common grass species to over 85% of the treatment plots include Buffalograss (*Bouteloua dactyloides*), Texas Wintergrass (*Nassella leucotricha*), Oldfield Threeawn (*Aristida oligantha*), and King Ranch Bluestem (*Bothriochloa ischaemum*). Grass and forb species common to over 50% of the treatment plots include Meadow Dropseed (*Sporobolus composites var drummondii*), Broomweed (*Amphiachyris dracunculoides*), Prickly Sida (*Sida spinosa*), Hairy Grama (*Bouteloua hirsuta*), Purple Threeawn (*Aristida purpurea*), and Sideoats Grama (*Bouteloua curtipendula*). Tree species occurrences were infrequent and consisted of Mesquite (*Prosopis glandulosa*) and Flaming Leaf Sumac (*Rhus lanceolata*).

Methodology

The research site was selected within the Fort Hood Western Training Grounds in an area of homogenous soil type based on Natural Resources Conservation Service soil survey data. Seven plots over 100 acres total were delineated and assigned a dairy manure compost application rate of 0, 5, 10, 15, 20, 25, or 30 cubic yards per acre, 0 being the control. A GIS map of the research site with delineated treatment plots overlain with a Natural Resources Conservation Service soil survey map revealed that the control, 5 y^3/acre, 15 y^3/acre, 20 y^3/acre, 25 y^3/acre, and 30 y^3/acre plots are predominantly of the Slidell silty clay soil type, while the 10 y^3/acre plot was a mix of both the Slidell silty clay and Brackett-Topsey, a fine shallow loam (McCaleb 1985).

Composted Dairy Manure. Composted dairy manure from Organic Residual Reclamation, LLC was delivered with documentation from the United States Composting Council indicating nutrient, metal, and pathogen test results. Tests in August 2003 examined total nitrogen, phosphorous, potassium, calcium, magnesium, sulfate, moisture content, organic matter content, pH, soluble salts, particle size, fecal coliform under the USEPA Class A standard, trace metals under the USEPA Class A standard, carbon dioxide evolution, and maturity indicator through relative seedling emergence. Trace metals regulated under the USEPA Class A standard are arsenic, cadmium, chromium, copper, lead, mercury, molybdenum, nickel, selenium, and zinc (Walker et al. 1994). Trace metals included in the analysis but that are not on the USEPA Class A standard were iron, boron, and manganese. The nutrient specific amounts of the composted waste are located in appendix A. A ten cubic yard capacity tongue pulled spreader on 40 ton axels with 20 by 22 inch tires was designed and manufactured specifically for the rough terrain around the Fort Hood Western Training Grounds (Keating 2007). Unique specifications to this particular spreader include 31 inch ground clearance, stainless steel body, 31 inch discharge chain, and oversized distribution spinners all together enabling the machinery to apply compost at rates as high as 90 cubic yards per acre and compost under assorted moisture regimes (Keating 2007). The spreader was hitched to a 100 horsepower tractor and pulled across the designated plots distributing the compost treatments. All composted dairy manure treatments were applied following this procedure in May 2004.

Rainfall Simulations. The first round of rainfall simulations were performed July 12-14, 2004. Soil samples were collected prior to rainfall simulation with a scoop at a depth of 5 centimeters, and stored in a cooler until transferred to the Texas A&M Soil, Water, and Forage Testing Laboratory where they were analyzed for bulk density, percent soil moisture, and percent organic matter. Ocular estimates of foliar cover within the rainfall simulation operation was assessed pre-simulation by three separate individuals, with the average of the three assessments being recorded. Foliar cover categories were vegetation, litter, bare ground, and rock, with vegetation being divided into annual and perennial categories. The above ground vegetative biomass was clipped at the soil surface post-simulation and placed into individual brown paper bags by plot, transported to the lab, dried in an oven at 60°C for 48 hours, and weighed for estimates of total production. Microrelief was measured before each rainfall simulation using a microrelief board with 10 pens that measures the change in surface relief in centimeters. Microrelief was measured at three separate locations and the average was recorded. For each rainfall simulation, a drip type rainfall simulator was operated at a rate of six inches per hour (15.24 cm) for a total of 30 minutes. Tin edging was driven into the ground around the border of the rainfall simulation breadth that funneled runoff into a 20-liter collection bottle at one end to ensure all runoff was collected. The runoff in the bottle was weighed at five minute intervals for the entire 30 minute simulation, along with the initial time of runoff and the amount of runoff in pounds. In July 2004, all seven waste treatment plots had four 30 minute rainfall simulations performed. Runoff water samples were submitted to the Texas A&M Soil, Water, and Forage Testing Laboratory and analyzed for nutrient content. Current methods on nutrient analysis utilized by the lab are as follows: pH is determined with a hydrogen ion selective electrode; conductivity is determined with a conductivity probe; chloride levels are determined with a chloride ion selective electrode; calcium, magnesium, sodium, and potassium are determined through ICP analysis; carbonate, bicarbonate, and alkalinity are determined through acid titration with sulfuric acid; total dissolved salts and charge balance are calculated as the sum of ppm of anions and cations and the division of the sum of cations by the sum of anions respectively; and nitrate nitrogen is determined through the reduction of nitrite to nitrate using a cadmium column followed by spectrophotometric measurement (Methods and Method References 2010). A second round of rainfall simulations were begun January 14-16, 2005, but were never completed; therefore, only the 0, 5, 10,and 15 y^3/acre treatments had four 30 minute rainfall simulations performed. The runoff samples collected at this time were never sent in for analysis by the Texas A&M Soil, Water, and Forage Testing Laboratory. The lapse in time from the simulations in 2004/2005 to when I received the data in 2009 disabled the ability to go back out to the site and finish the second round of rainfall simulations.

Statistical Analysis. The statistical software SAS and JMP were used to generate analysis graphs and models for determination of any correlations or significance between composted waste treatments and infiltration rates including effects of soil and vegetative characteristics. Individual variable distribution was determined using the Kolmogorov-Smirnov goodness of fit statistic and was assessed to have an exponential distribution for the majority of measured variables. All statistical comparisons utilized a Tukey's Honestly Significant Difference (HSD) comparative analysis for determination of significant differences. Soil, cover, and treatment variable interactions to infiltration responses are determined through all first order and main effects interactions under a stepwise approach for model selection using a backward selection technique. All data was analyzed at α = 0.05, under a 95% confidence level.

RESULTS AND DISCUSSION

For every rainfall simulation performed, the weight of the collected runoff was recorded at 5 min intervals for the entire 30 min simulation. The weight was converted to a depth across the 1m² plot and further revised into a rate of runoff observed at the collected point in time during the simulation. The basic equation used for this conversion is depth in centimeters divided by the time interval in minutes at which that depth was recorded, yielding a cm/min runoff rate. Knowing the rate of rainfall application and the runoff rate allowed for calculation of the rate of plot infiltration. Parallel to other studies, the infiltration rate was calculated as only the rate of precipitation minus the rate of runoff due to the difficulty in measurement and minor extent of other factors within water budgets (Wilcox et al. 1988).

All Data Averaged

First, the data was examined across the whole experiment in order to determine any overarching patterns in infiltration rate versus composted-waste application rate. This initial analyses averaged all calculated infiltration rates together from plots under the same compost application treatment without regard to season of application. Figure 1 graphically represents the averaged infiltration rates at five minute intervals for the entire 30 min simulation for all compost-application treatments. As expected, infiltration from

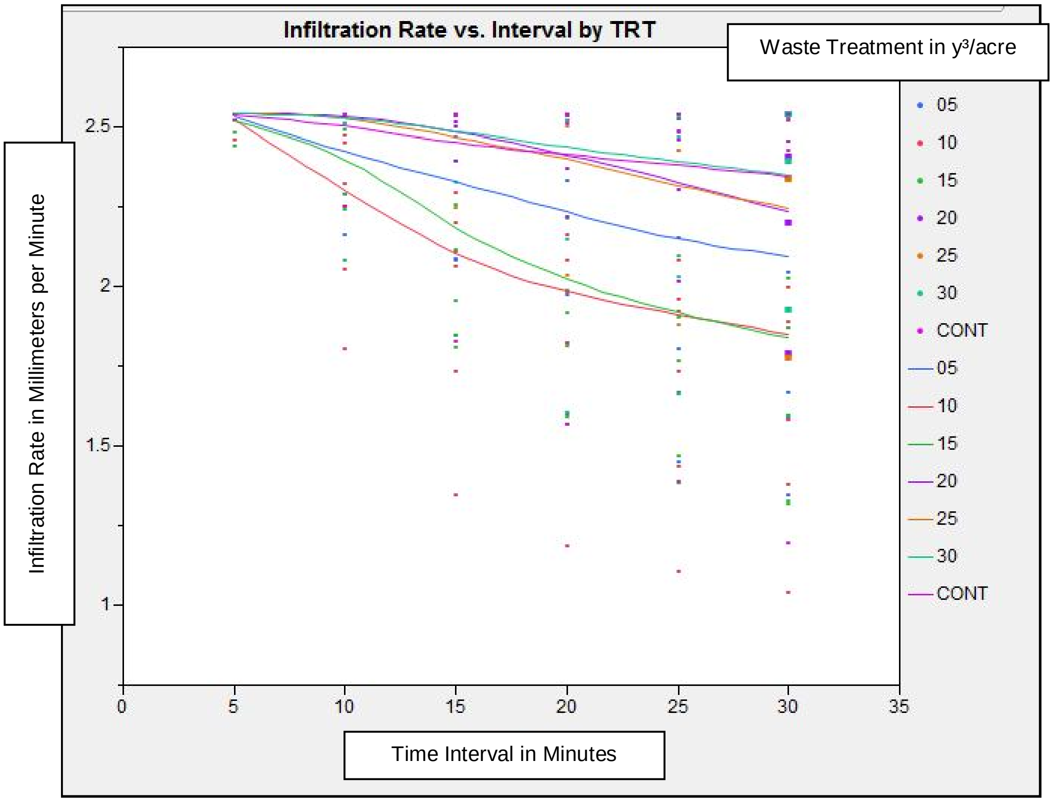


Figure 1. Average of infiltration rates by waste application treatment graphed at five minute intervals for the entire 30 min rainfall simulation.

0 - 5 mins of rainfall simulation matched the rainfall rate. Percent soil moisture before rainfall simulation was below 35% for all plots, leaving lots of pore space to accumulate water at the onset of precipitation. Infiltration from 5 - 10 min reduced on the 5, 10, and 15 y^3/acre treatment plots while 20, 25, and 30 y^3/acre treatment plots maintained infiltration at precipitation rates. After 10 mins until the end of the simulation at 30 mins there exists a clear difference between the 5, 10, and 15 y^3/acre treatment plots and the control, 20, 25, and 30 y^3/acre treatment plots where the latter maintain higher infiltration rates. This makes sense as addition of composted waste has many beneficial effects including observable increases in soil aggregation and increases in total pore space both contributing to an increase in infiltration rate (Haynes et al. 2009). Beneficial effects from composted-waste applications do not explain the maintenance of high infiltration rates on

the control treatment plot which received no soil amendments. Vegetative cover for the control plot was high compared to many other treatment plots, and review of previous literature indicates that vegetative cover can support high infiltration rates. Total vegetative cover is positively correlated to infiltration rate (Blackburn et al. 1992) and is important for maintenance of soil aggregates that promote infiltration (Thurow 2000). A Tukey's HSD analysis indicates that litter cover and forb cover for all the treatment plots were statistically similar; average control plot forb cover values range from 1 – 50% and average control plot litter cover values range from 5 – 61%. Comparatively, percent grass cover was found to differ significantly across treatment plots where the 30 y^3/acre treatment was the highest in mean value but not significantly different from that of the control treatment plot. Figure 2 graphically displays the percent grass cover by treatment and shows that the control plot

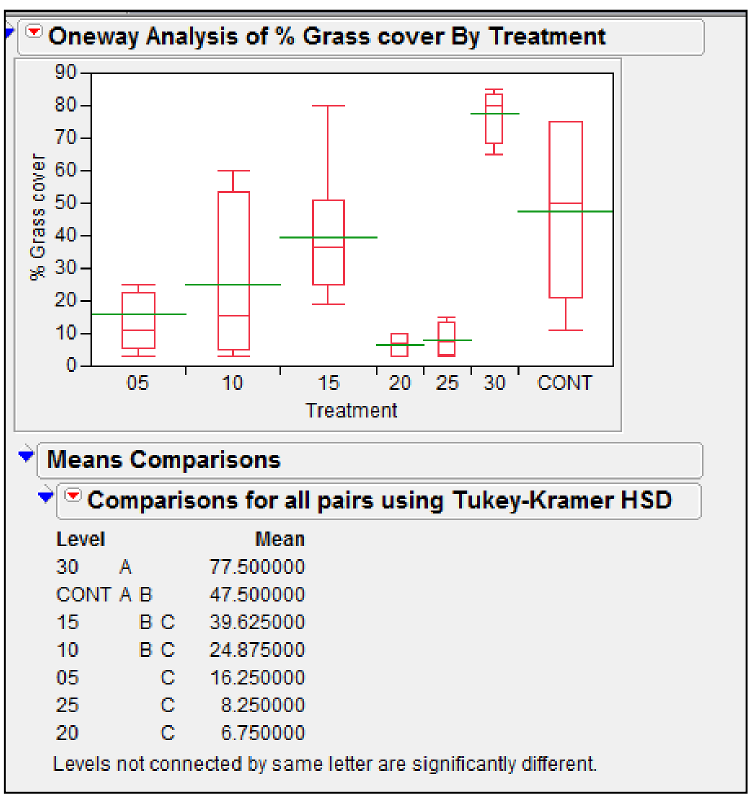


Figure 2. Box plot comparison of averaged percent grass cover for all simulations graphed by treatment with a Tukey's HSD comparison and green bars representing each treatment mean.

had the second-highest mean value. Grass cover values for the control plot range from 11% to 75%. Good vegetative cover on the control plot could have promoted high infiltration rates above the 5, 10, and 15 y³/acre treatment plots that all had significantly less grass cover.

To determine statistical differences in treatments, Figure 3 compares infiltration rates by treatment plot with confidence intervals at 5 minute periods when infiltration rates were calculated from collected runoff. As demonstrated by the graph, many of the confidence intervals overlap indicating no significant difference between those overlapping

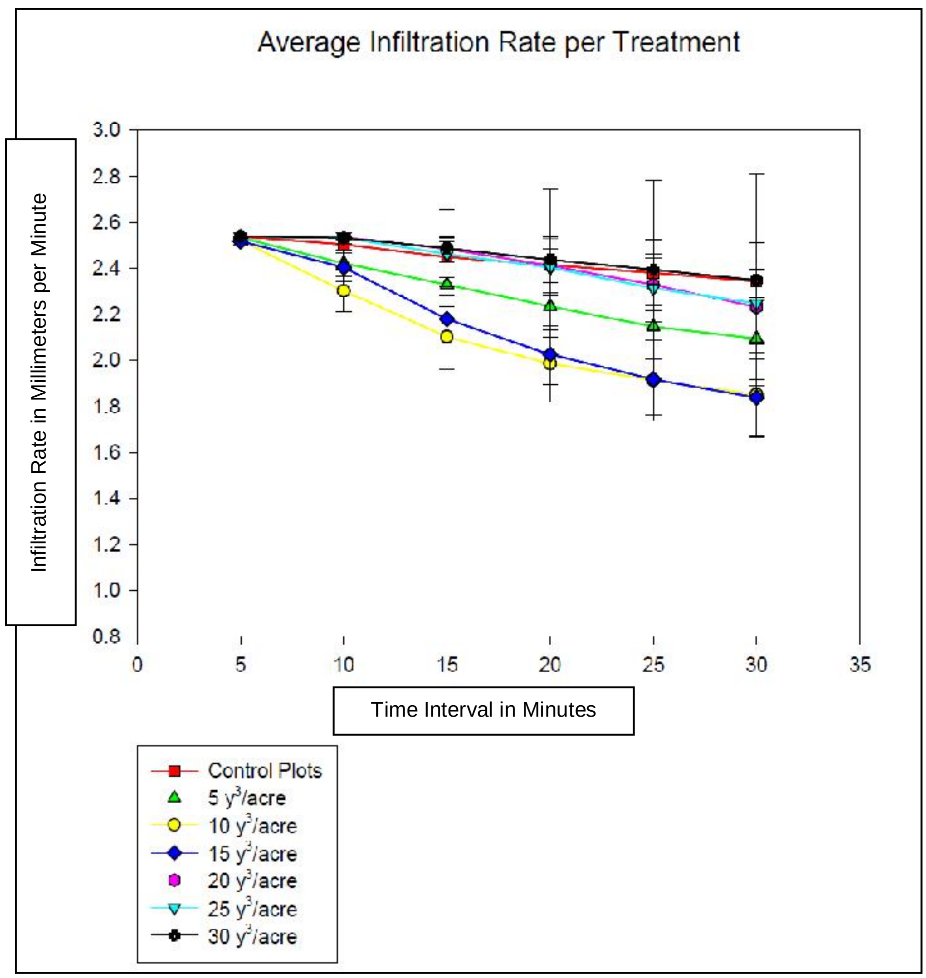
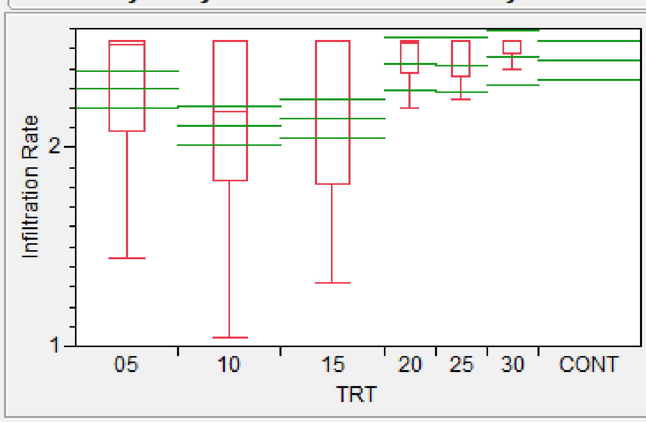


Figure 3. Average of infiltration rates by waste application treatment graphed at five minute intervals for the entire 30 min simulation including calculated 95% confidence intervals.

treatments. Individual confidence intervals are difficult to distinguish in Figure 3 due to the high variability of calculated infiltration rates, therefore infiltration rates are graphically represented by box plots for easier visual comparison in Figure 4. The 10 and 15 y³/acre treatment plots are graphically very similar, while overlapped confidence intervals indicate that the 5, 10, and 15 y³/acre treatment plots are statistically similar. The Tukey's HSD comparative analysis supports this conclusion as well, indicating that the 10 and 15 y³/acre application treatments are statistically different from the control treatment and the 20, 25, and 30 y³/acre application treatments, with the 10 and 15 y³/acre treatments having significantly lower infiltration rates.

Oneway Analysis of Infiltration Rate By TRT



Means Comparisons

Comparisons for all pairs using Tukey-Kramer HSD

Level		Mean
30	A	2.4562500
CONT	A	2.4387222
20	A	2.4223472
25	A	2.4166667
05	A B	2.2938264
15	B	2.1472917
10	B	2.1124028

Levels not connected by same letter are significantly different.

Figure 4. Box plot comparison of averaged infiltration rates graphed by treatment with a Tukey's HSD comparison and green bars representing the upper limit of a 95% confidence interval, the mean, and the lower limit of a 95% confidence interval.

Further statistical analysis is unavailable due to the unrecognizable distribution of the response variable ‘infiltration rate.’ Statistical models require an assumption of distribution be met for utilization of that particular model to analyze a data set, but no common distribution fits the variable infiltration rate. Neither the distributions of normal, exponential, nor gamma described the distribution of infiltration rate preventing statistical analysis through Analysis of Variance, Analysis of Covariance, and General Linear Model. More accurate statistical comparisons can be made by dividing the data up between seasons of rainfall simulation instead of lumping all plots together in an average analysis. Vegetative cover changes from season to season, as well as the utilization of water within the soil by living vegetation affecting soil moisture conditions at the onset of precipitation. Winter precipitation and occasional snow fall (occurrence of snow is rare at this location in central Texas, but can vary from year to year (McCaleb 1985)) increases percent soil moisture due to less uptake by vegetation in the winter and can reduce the total amount of pore space available to absorb water in the next rain event, thus increasing runoff rates.

Summer Rainfall Simulations

This first round of rainfall simulations were performed two months post waste-treatment application, recall that wastes were applied in May 2004. All seven composted waste application treatments received four rainfall simulation runs in July 2004, and the calculated infiltration rate of all the simulations averaged per treatment are graphically shown in Figure 5. In the summer rainfall simulations, the 5 y^3/acre treatment maintained the highest infiltration rates throughout the entire 30 min rainfall simulation interval; in fact, two out of the four individual simulation plots did not generate any runoff until well after the 30 min time interval of runoff collection was over.

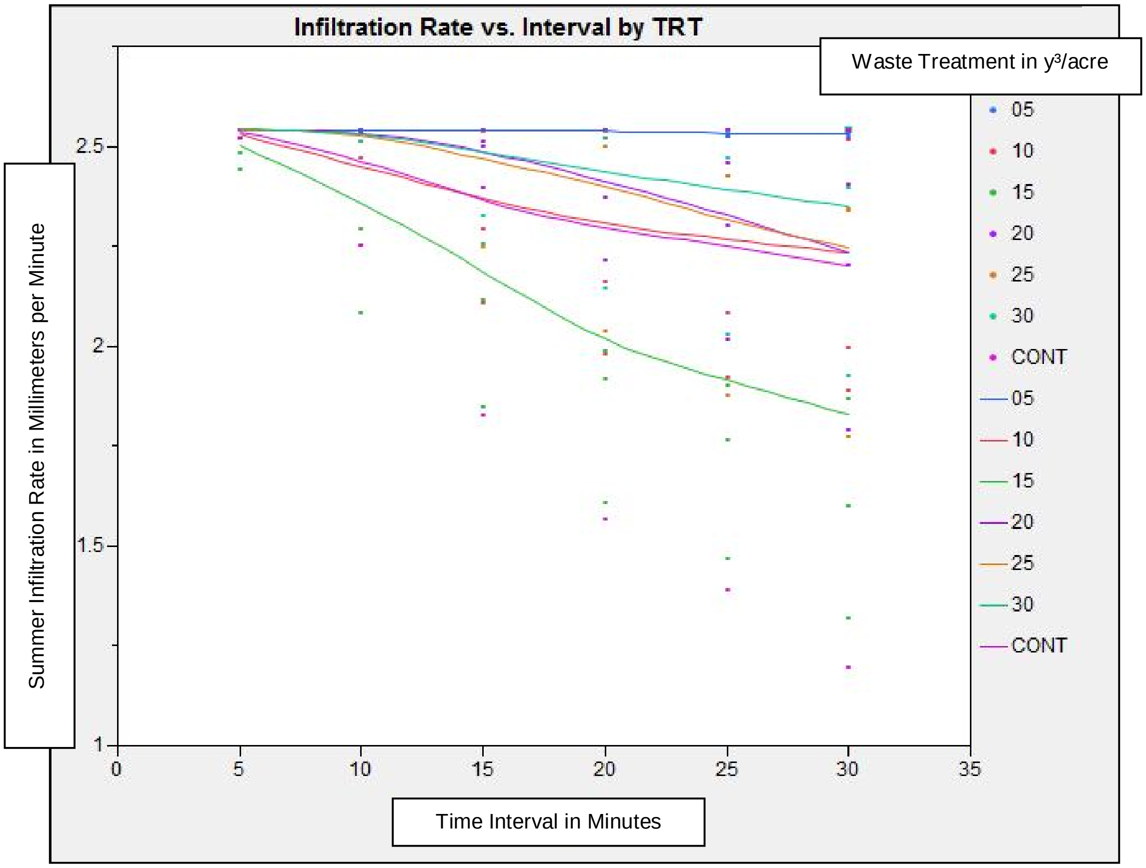


Figure 5. Average of July 2004 infiltration rates by waste application treatment graphed at five minute intervals for the entire 30 min rainfall simulation.

For the first 10 min of rainfall simulation the 5, 20, 25, and 30 y³/acre treatments maintained infiltration rates at rainfall application rates, after 10 min the 20, 25, and 30 y³/acre treatments started generating runoff and exhibited falling infiltration rates. The control treatment and the 10 y³/acre treatment exhibited very similar infiltration behavior throughout the entire 30 min simulation maintaining rates well above the 15y³/acre treatment and below the 5, 20, 25, and 30 y³/acre treatment grouping. The 15 y³/acre treatment maintained the lowest infiltration rates with two out of four individual simulation plots generating runoff in under five minutes. When averaged infiltration rates by treatment are graphed in box plots and analyzed with a Tukey's HSD comparison, the output follows this same logic. As seen in Figure 6, the 5, 20, 25, and 30 y³/acre treatments are statistically grouped and identified by a Tukey's HSD comparative analysis as being significantly different from the 15 y³/acre treatment. The control treatment and the 10 y³/acre treatment are statistically similar to both groupings by the Tukey's HSD analysis demonstrating the high variability in the time these plots took to generate runoff, with both treatments having values in under 5 min and values of over 30 min.

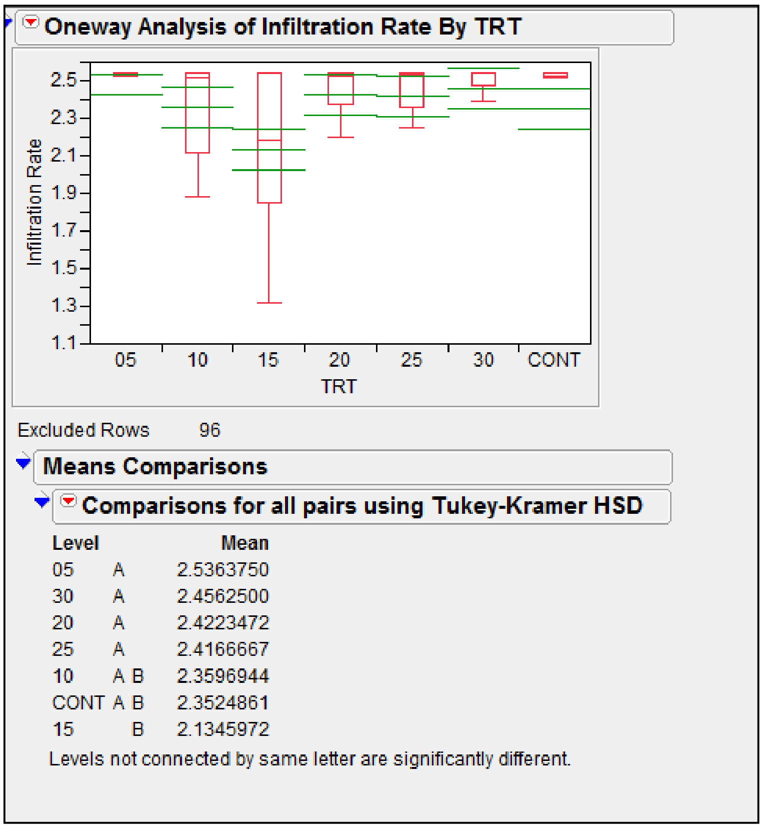


Figure 6. Box plot comparison of July 2004 averaged infiltration rates by treatment with a Tukey's HSD comparison and green bars representing the upper limit of a 95% confidence interval, the mean, and the lower limit of a 95% confidence interval.

As reviewed previously, the addition of composted waste should increase soil aggregation and total porosity both contributing to an increase in infiltration rates (Haynes et al. 2009); theoretically applied to this experiment, results should indicate increased rates of waste application treatments having increased infiltration rates. This is not the case for rainfall simulations performed in July 2004 when just the waste-application rate and infiltration rate are analyzed, indicating that other variables are affecting observed infiltration rates in these statistical models. One reason for unexpected analysis results from the variable ‘infiltration rate’ is the inability to accurately compare simulation plots under this variable. Runoff collection stopped at 30 min on every single plot, regardless of whether runoff had actually started or not; therefore, there are many values of zero runoff for entire 30 min simulations. Converted to an infiltration rate, this is equal to that of rainfall applied which does not really yield a large amount of information about the infiltration capacity of a plot. Essentially, an analysis of a plot of known water infiltration capacity (because it met its limit through generation of runoff) is being compared to an analysis of a plot with unknown infiltration capacity (because its limit was never met) and considering them equal measures. In order to account for a plots ability to infiltrate more water post 30 min, a new response variable of ‘maximum infiltration’ was calculated by multiplying the maximum infiltration rate available (rainfall rate) by the time it took for a plot to generate runoff. This new variable characterizes the maximum amount of water a plot can hold before runoff began and infiltration rates start to decline. Although runoff collection was ended at 30 min, all the rainfall simulations were run until runoff was generated, with that time being recorded and calculated in this new variable. The distribution of the new variable maximum infiltration for the summer data set is exponential, allowing for statistical analysis not available with the variable infiltration rate.

In order to determine what input variables have a significant influence on the new response variable the next analysis ran the waste-application treatments with the soil characteristic variables of average bulk density, average percent organic matter, average percent moisture, and mean microtopography in a stepwise approach for model selection to the response variable maximum infiltration. All variables were entered into the model and significance was determined at the α = 0.05 significance level. Insignificant interactions between input variables and the response variable were removed one at a time using a backward selection technique, until only significant variables remain in a simple model. The first stepwise model output for summer rainfall simulations is shown in Table 1.

Table 1. Statistical output table from July 2004 stepwise model one comparing all measured soil characteristics and waste application treatment to the response variable maximum infiltration.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
treatment	6	17	0.63	0.7014
b.d. avg	1	17	0.05	0.832
moist	1	17	0.02	0.8808
o. m.	1	17	3.5	0.0787
micro	1	17	1.49	0.2392

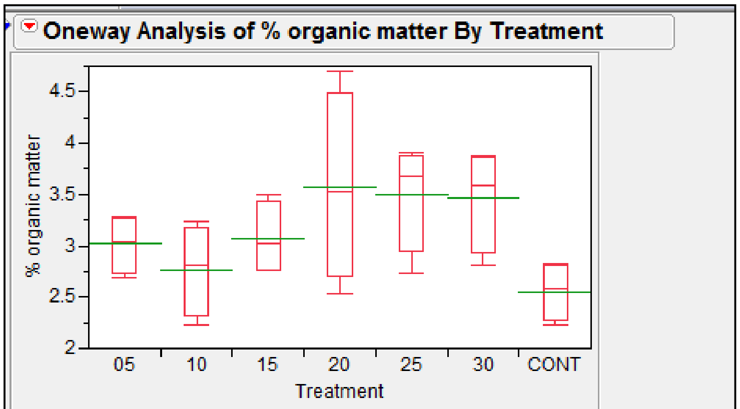
Stepwise model one determined that no main effects variables input had a significant effect on the response variable of maximum infiltration, so the least-significant variable of percent soil moisture was removed and the analysis was run again as stepwise model two. Similarly, stepwise model two found no variables to be significant to the response variable under an α = 0.05 significance level with the least significant being the bulk density of the soil. Stepwise model three was then run comparing only the waste treatments, percent organic matter, and mean microtopography against the response variable of maximum infiltration and again found no significant output, although percent organic matter comes close at a p value of 0.0639 (remember α = 0.05). For stepwise model four, mean microtopography was removed and only waste treatment and percent organic matter were analyzed to the response variable maximum infiltration with the final output shown in Table 2. It is only in stepwise model four that a significant variable at

Table 2. Statistical output table from July 2004 stepwise model four comparing the soil characteristic percent organic matter and the waste application treatment to the response variable maximum infiltration.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
treatment	6	20	1.12	0.3855
o. m.	1	20	6.04	0.0232

$\alpha = 0.05$ significance level is finally found, and it is organic matter. That is to say that percent organic matter is the only soil characteristic variable that has a significant influence on the response variable of maximum infiltration. As noted in Table 2 even the waste-application treatments do not have a significant effect on the response variable maximum infiltration, but this was not removed from the model analysis because waste application treatments are the basis of my experimental research. Having identified organic matter as the only soil characteristic to be significant to maximum infiltration, surface cover characteristic variables of percent bareground, percent litter cover, percent grass cover, and percent forb cover were next analyzed using the same stepwise approach for model selection. Surface and soil characteristics were analyzed separately because if analyzed together then there would be too many variables in the model and none would be found significant. Unfortunately, these surface-cover characteristics are all collinear to each other and therefore could not be analyzed in a model together. These variables are measured as a percent of the same 100% total; therefore one measurement will affect how large the rest of the measurements can amount. In statistics, when model-input *X* variables have a relationship to each other this situation is known as collinearity and prevents accuracy of models. When cover characteristics were run individually with waste treatment in the stepwise model against the response variable maximum infiltration, no cover variable was found to have significance. First-order interactions between all main effects variables and waste-application treatments were included in the initial stepwise model analysis, but were not found to contribute to model significance.

After determining that percent organic matter is the only variable to have a significant effect on the maximum infiltration of a plot in the summer, organic matter was plotted by waste application treatment in box plots and compared using a Tukey's HSD comparative analysis with results in Figure 7. The Tukey's HSD test indicates that there is



Excluded Rows 16

Means Comparisons

Comparisons for all pairs using Tukey-Kramer HSD

Level		Mean
20	A	3.5710000
25	A	3.5015000
30	A	3.4632500
15	A	3.0732500
05	A	3.0160000
10	A	2.7680000
CONT	A	2.5542500

Levels not connected by same letter are significantly different.

Figure 7. Box plot comparison of July 2004 percent organic matter graphed by treatment with a Tukey's HSD comparison and green bars representing the mean.

no significant difference in percent organic matter between treatments, but by comparing the calculated means a pattern emerges. The mean observations follow a general pattern of increasing percent organic matter as waste-application rate increases: the control treatment has the lowest mean, followed by the 10 y^3/acre treatment, 5 y^3/acre treatment, 15 y^3/acre treatment, 30 y^3/acre treatment, 25 y^3/acre treatment, and finally the 20 y^3/acre treatment has the highest percent organic matter mean. This general trend does not indicate significant differences in the treatments, all observed values are within a similar range of each other indicating no significant effect of the waste amendment at any rate. A trend of increasing percent organic matter with increasing composted waste application is to be expected as biological wastes are 40-70% organic matter (Haynes et al. 2009) and increasing rates of application will increase the total amount of organic matter applied and incorporated into the soil. Unfortunately, the waste amendments applied in May 2004, just two months prior to the first run of rainfall simulations, did not have an effect on the amount of organic matter measured within the soil. The composted waste amendments may not have had enough time to incorporate into the native soil affecting aggregation and porosity enough to significantly affect infiltration rates in just two months time.

Runoff Nutrient Analysis. Collected runoff samples from the rainfall simulations performed in July 2004 were sent to the Texas A&M Soil, Water, and Forage Testing Laboratory where they were analyzed for nutrient content. Runoff samples were collected for only those plots that generated runoff in under the 30 min time interval, and therefore only 16 plots out of 28 total were included in this analysis. The control treatment only had one plot generate runoff in under 30 min; when statistically analyzed, this amounts to zero degrees of freedom (calculated by sample size $n - 1$) and generates abnormal box plots as seen in Figures 8 and 9. Every other waste application treatment had either two or three instances of runoff generation in less than 30 min and exhibit slightly better graphical comparison ability.

Figure 8 is a box plot graph and Tukey's HSD comparative analysis of nitrate-nitrogen (nitrate-N) in runoff by treatment and indicates that most waste application treatments generate similar amounts of nitrate-N in runoff. The Tukey's HSD analysis

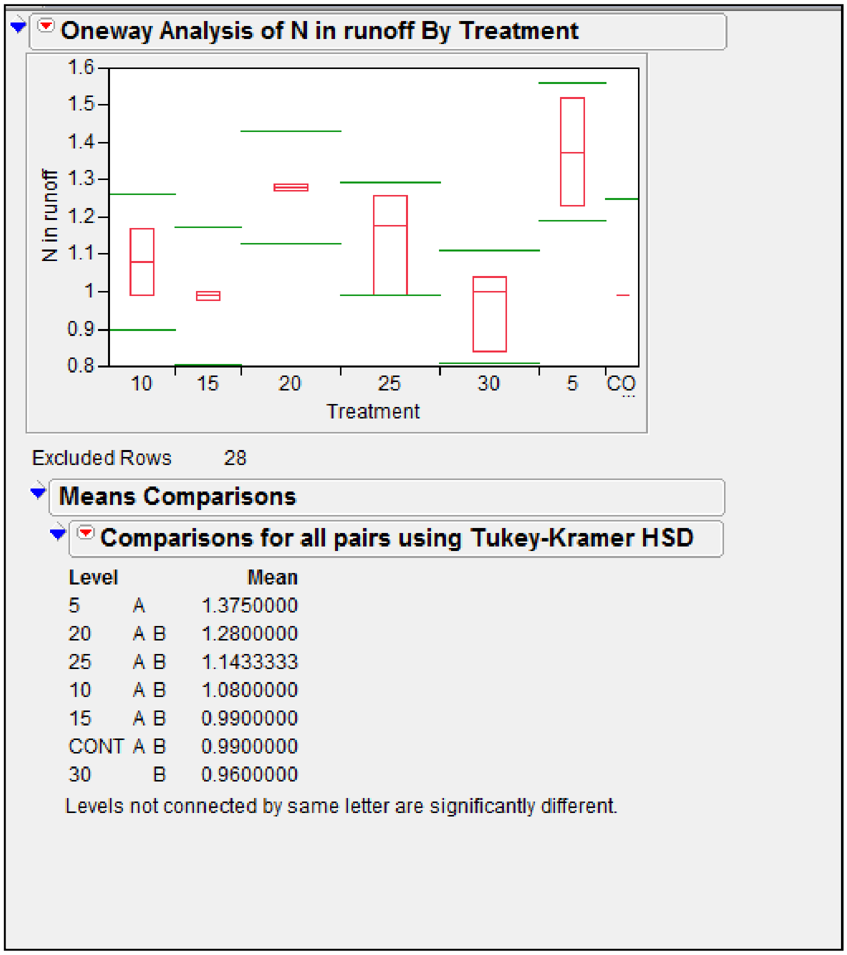


Figure 8. Box plot comparison of nitrate-N in runoff in mg/L graphed by treatment with a Tukey's HSD comparison and green bars represent the upper limit of a 95% confidence interval and the lower limit of a 95% confidence interval.

concludes that only the 5 y³/acre treatment is different from the 30 y³/acre treatment, but visual comparison of the box plots indicates that the 5 y³/acre treatment lost more nitrate-N. This is counterintuitive because there was less total nitrogen applied in the 5 y³/acre treatment application than the 30 y³/acre treatment application. Additionally, the control treatment (without waste amendment) generated measurable nitrate-N in runoff despite have no additional nitrogen applied. The similarity between observations from the control plot and observations among treatment plots indicates that the addition of composted wastes does not significantly affect nitrate-N in runoff on this landscape. Observed values of nitrate-N in runoff ranged from 0.84 – 1.52 mg/L, all well below the Texas drinking water quality standard of 10 mg/L for nitrate-N (Garcia et al. 2008). Recall that rainfall was applied at a rate of 6 in/hr (15.24 cm/hr). This equates to 18% of the yearly average precipitation for Coryell County (which is 34 in or 86.36 cm) and rainstorms of this intensity are unlikely to be observed (McCaleb 1985). The heaviest rain event over the duration of one day was 8.35 inches of rainfall in 1964 in the city of Gatesville, just north of the Fort Hood military installation (McCaleb 1985). Therefore, nitrate-N water contamination in runoff from composted-waste applications on the Fort Hood Western Training Grounds is unlikely due to low amounts calculated in experimental plot runoff and the rare occurrence of a precipitation event intense enough to generate measurable runoff.

Phosphate in runoff followed a more expectable pattern despite have no significant difference between treatments according to Tukey's HSD comparative analysis, shown in Figure 9.

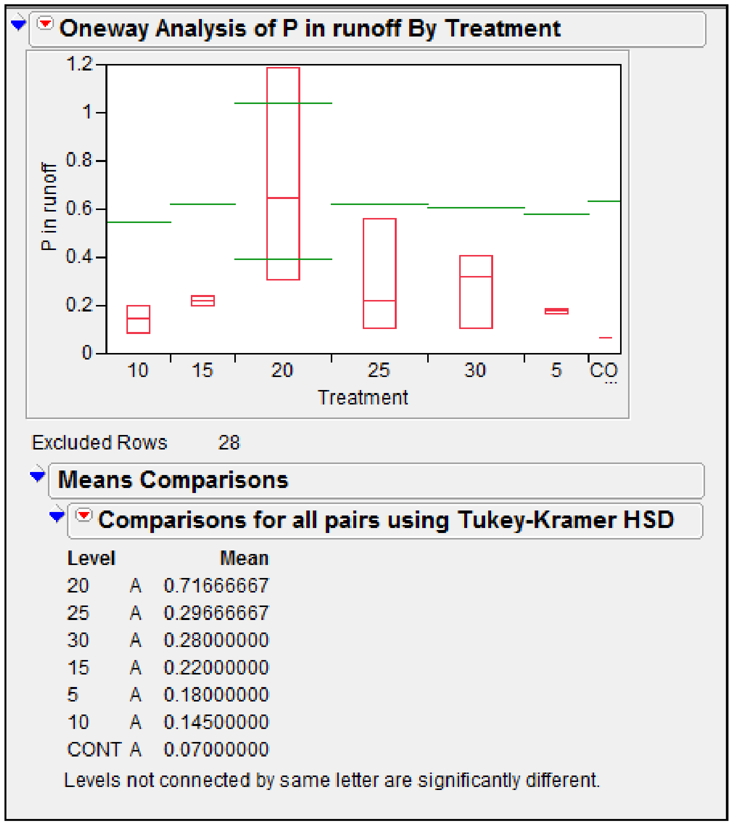


Figure 9. Box plot comparison of phosphate in runoff in mg/L graphed by treatment with a Tukey's HSD comparison and green bars representing the upper limit of a 95% confidence interval and the lower limit of a 95% confidence interval.

In much the same fashion as percent organic matter, phosphate in runoff followed a general pattern of increased loss of phosphorous with increased application of waste treatment. The control plot generated the least amount of phosphate in runoff and is followed by the 10 y^3/acre treatment which generated a loss equivalent to twice the control's mean. The 5, 15, 30, and 25 y^3/acre treatments were all within a mean of 0.1 mg/L of phosphate in runoff in increasing order, followed by the greatest generation of phosphate in runoff by a large amount is the 20 y^3/acre treatment. Again, the observed values between the control plot and the treatment plots range within a similar set of values indicating no significant increase in phosphate loss through runoff on this landscape. According to the above results, phosphate in runoff from lands without waste addition is similar to lands that have been amended with up to 30 y^3/acre waste amendments. Measured phosphate values in collected runoff ranged from 0.05 – 1.19 mg/L in this experiment, although there are no drinking water quality standards to compare this to for phosphate in Texas. Excess phosphorus in surface water bodies is a pollutant concern for its contribution to algal growth and not necessarily a concern by simple presence of phosphorus in water. Algal growth conditions are dependent on many other factors, therefore phosphorus contamination is determined by a screening level test set at a value where phosphorus could potentially begin to support excessive algal growth. This value is 0.80 mg/L for streams (McDonald et al. 2004), although levels as low as 0.02 mg/L have been known to be problematic in surface waters (Aitkenhead-Peterson 2009). Observed phosphate in collected runoff exceeded the 0.02 mg/L value cited for creating known problematic conditions in surface waters, and only the 20 y^3/acre treatment was observed to exceed the TCEQ screening level. This indicates that phosphate in runoff could potentially increase algal growth in local water bodies. Again, the occurrence of a precipitation event intense enough to generate measurable runoff is rare and reduces the possibility of local water source contamination from waste applied rangelands within the Fort Hood military installation.

A more accurate analysis of nutrients in runoff could be completed on the Fort Hood Western Training Grounds rangeland if a larger sample size were available, as each treatment in this analysis had a maximum of three samples and one treatment had only one sample.

Winter Data Analysis

The second round of rainfall simulations are performed eight months post waste-amendment applications, recall that waste treatments were applied May 2004. The control treatment and the 5, 10, and 15 y^3/acre composted- waste application treatments had rainfall simulations run in January 2005, with each treatment having four individual simulations performed. Graphically represented in Figure 10 is the average of these four simulations per treatment across 5 min intervals over the entire 30 min simulation.

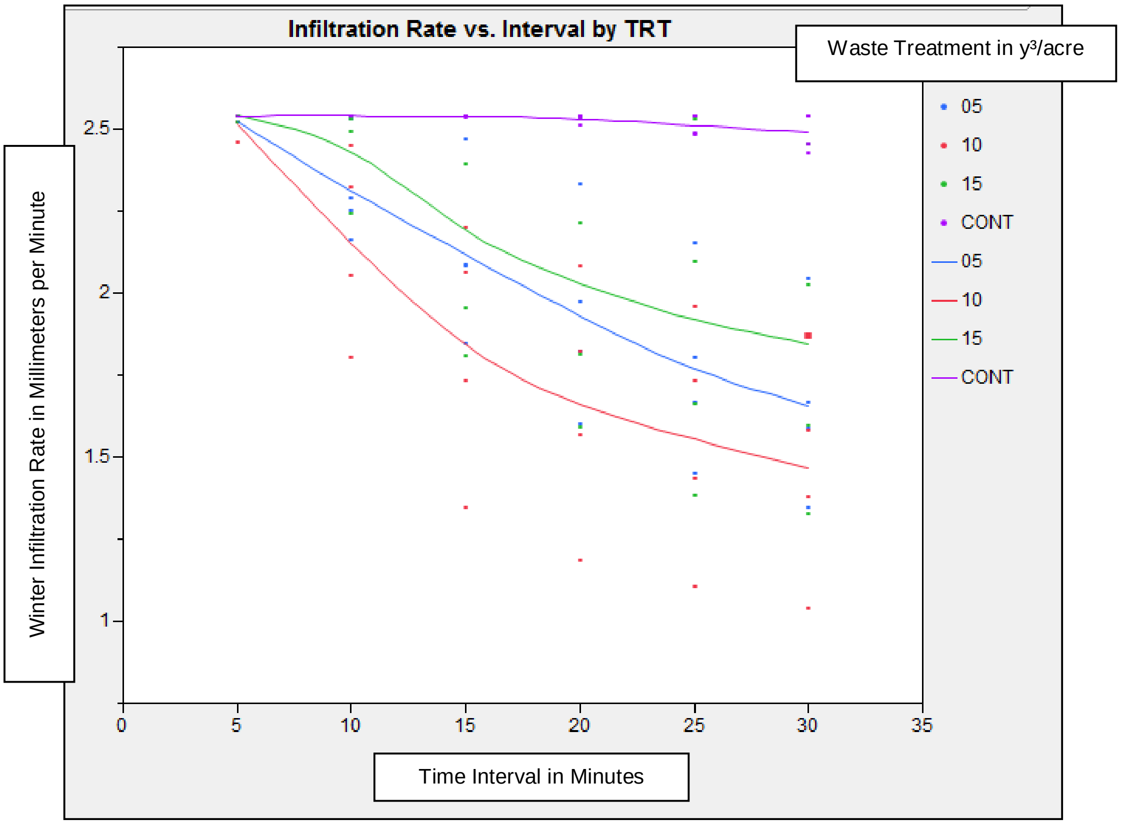


Figure 10. Average of January 2005 infiltration rates by waste application treatment graphed at five minute intervals for the entire 30 min simulation.

Similar to summer, the winter infiltration rates do not follow the expected pattern of increasing infiltration rates by increasing application of wastes determined in the literature review. The control treatment maintained infiltration rates close to the rate of rainfall application, while the 5, 10, and 15 y³/acre treatment infiltration rates fell over the 30 min simulation. Unlike summer infiltration rates there is a very clear distinction between each of the waste application treatments, supported in Figure 11 by box plot comparison and a Tukey's HSD comparative analysis. Here, the control treatment is significantly different

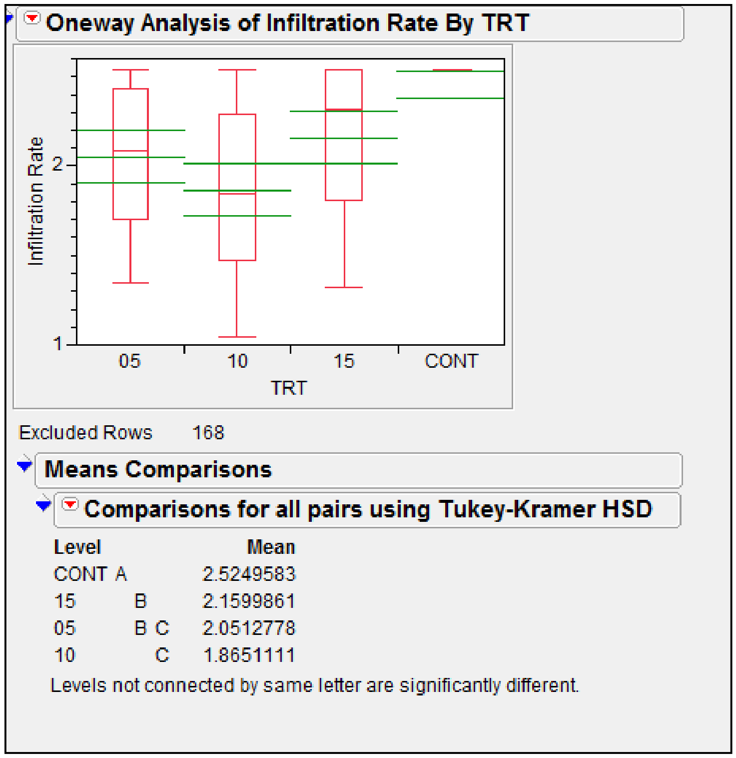


Figure 11. Box plot comparison of January 2005 averaged infiltration rates by treatment with a Tukey's HSD comparison and green bars represent the upper limit of a 95% confidence interval, the mean, and the lower limit of a 95% confidence interval.

from the other three plots of waste application treatments with a higher infiltration rate on average. This seems to indicate that composted waste amendments reduced infiltration rates on treatment plots in January 2005, but other environmental variables could be influencing observed low infiltration rates on the treatment plots. In the same fashion for summer data analysis, in order to account for a plots ability to infiltrate more water post 30 min in the winter, the new response variable of 'maximum infiltration' was calculated by multiplying the maximum-infiltration rate by the time a plot took to generate runoff. This new variable characterizes the maximum amount of water a plot can hold before runoff begins and infiltration rates start to decline. Although runoff collection was ended at 30 min, all the rainfall simulations were run until runoff was generated, with that time being recorded and calculated in this new variable. The distribution of the variable maximum infiltration for the winter data set is also exponential, allowing for statistical analysis not available with the variable infiltration rate.

In order to determine what input variable had a significant influence on the new response variable this next analysis ran the waste-application treatments with the soil characteristic variables of average bulk density, average percent organic matter, average percent moisture, and mean microtopography in a stepwise approach for model selection to the response variable maximum infiltration. All variables were entered into the model and significance was determined at the $\alpha = 0.05$ significance level. Insignificant interactions between variables and the response were removed one at a time using a backward selection technique, until only significant variables remained in a simple model. The first stepwise model output is shown in Table 3.

Table 3. Statistical output table from January 2005 stepwise model one comparing all measured soil characteristics and waste application treatment to the response variable maximum infiltration.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Treatment	3	8	2.8	0.1088
B.D. Avg	1	8	0.16	0.6989
moist	1	8	0.21	0.6603
o. m.	1	8	0.3	0.5961
micro	1	8	1.12	0.3203

Stepwise model one determined that no variables input had a significant effect on the response variable of maximum infiltration, so the least-significant variable of average bulk density was removed and the analysis was run again as stepwise model two.

Similarly, stepwise model two found no variables to be significant to the response variable under an α = 0.05 significance level with the least significant being the percent moisture of the soil. Stepwise model three was then run comparing only the waste treatments, percent organic matter, and mean microtopography against the response variable of maximum infiltration and again found no significant output. For stepwise model four, percent organic matter was removed and only waste treatment and mean microtopography were analyzed to the response variable maximum infiltration with the output shown in Table 4.

Table 4. Statistical output table from January 2005 stepwise model four comparing the soil characteristic mean microtopography and the waste application treatment to the response variable maximum infiltration.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
TRTMNT	3	11	3.27	0.063
micro	1	11	1.49	0.2472

In stepwise model four, there is still no significant variable at the α = 0.05 significance level, although the waste-application treatment variable is close with a value of 0.063. For the final model, stepwise model five, mean microtopography was removed and the waste-application treatment variable is found to be significant. The basic model of significant output is shown in Table 5.

Table 5. Statistical output table from January 2005 stepwise model five comparing the waste application treatment to the response variable maximum infiltration.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
TRTMNT	3	12	3.67	0.0437

That is to say that the waste-application treatment variable is the only variable that has a significant influence on the response variable of maximum infiltration. Not having identified any soil characteristics to be significant, surface-cover characteristic variables of percent bareground, percent litter cover, percent grass cover, and percent forb cover were next analyzed using the same stepwise approach for model selection. As stated in the summer analysis, these cover characteristics are all collinear to each other and therefore could not be analyzed in a model together. When surface-cover characteristics were run individually with waste treatment in models against the response variable maximum infiltration, no cover variable was found to have significance. Again, first-order interactions between all main effects variables and waste-application treatments were included in the initial stepwise model analysis, but were not found to contribute to model significance.

Despite its non-significance in winter model analysis, percent organic matter in the winter followed a similar pattern as the summer percent organic matter measurements. A Tukey's HSD comparative analysis indicates that there is no difference in percent organic matter between treatments, but by comparing the calculated means a pattern of increasing percent organic matter with increasing composted-waste application emerges. As seen in Figure 12, the control treatment has the lowest mean, followed by the 5 y^3/acre treatment, 10 y^3/acre treatment, and finally the 15 y^3/acre treatment has the highest mean percent organic matter. Again, these observed values are all within a similar range and are not significantly different across the control plot and the composted waste amendment treatment plots indicating no effect of the amendment addition.

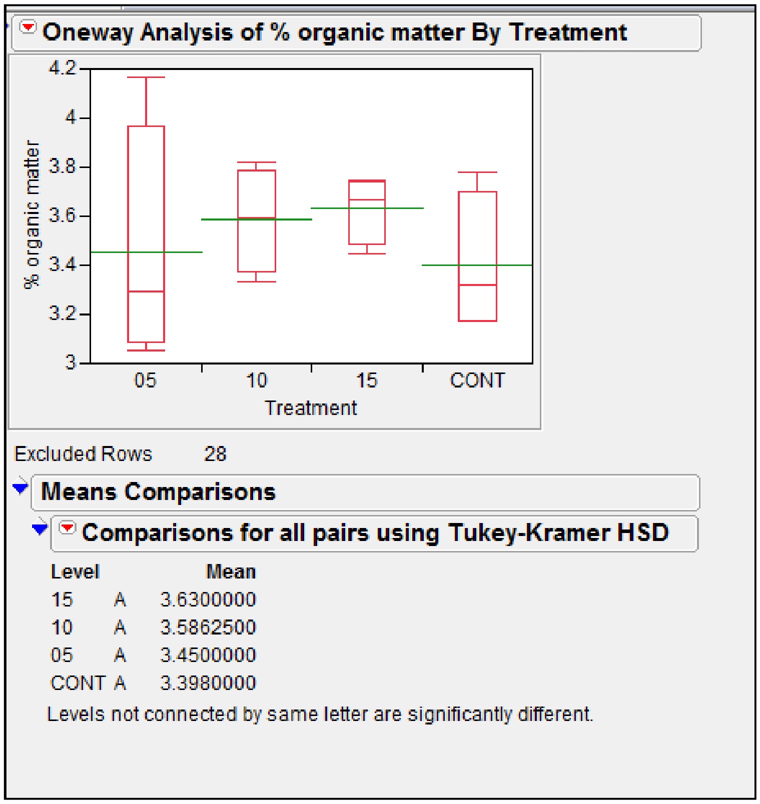
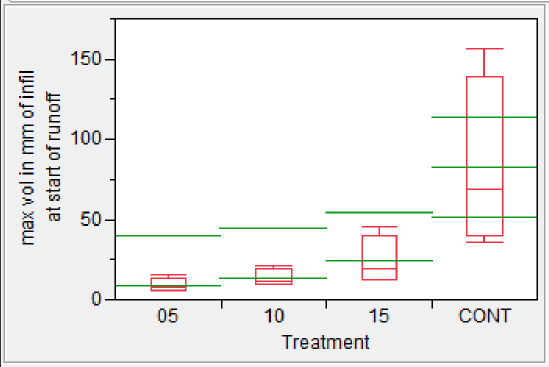


Figure 12. Box plot comparison of January 2005 percent organic matter by treatment with a Tukey's HSD comparison and green bars representing the mean.

Interestingly composted-waste application (as seen in Figure 11) does not follow this similar pattern in spite of being identified as significant by our model, although in Figure 11 it is graphed to the response variable average infiltration rate. Again, this is due to the 30 min time constraint of the rainfall simulation runoff collection and the misassumption that plots that did not generate runoff in under 30 minutes can be evenly compared to plots where maximum infiltration has been met and runoff has begun. When a comparison of the created response variable of maximum infiltration by treatment is made, the control plot still maintains superior maximum infiltration compared to the treated plots. As seen in Figure 13, the 5, 10, and 15 y^3/acre treatments follow the expected pattern of increasing maximum infiltration with increasing waste application though are not found to be significantly different from each other.

Oneway Analysis of max vol in mm of infil at start of runoff By Treatment



Excluded Rows 28

Means Comparisons

Comparisons for all pairs using Tukey-Kramer HSD

Level		Mean
CONT	A	82.575400
15	A B	24.053800
10	B	13.703300
05	B	9.124950

Levels not connected by same letter are significantly different.

Figure 13. Box plot comparison of January 2005 maximum infiltration by treatment with a Tukey's HSD comparison and green bars representing the upper limit of a 95% confidence interval, the mean, and the lower limit to a 95% confidence interval.

As previously indicated, the total cover of the control plot was high compared to most other treatment plots, but a significant relationship between vegetative cover and maximum infiltration was not determined through the stepwise model analysis. The data does not support the conclusion that vegetative cover could have increased maximum infiltration on the control plot despite observed control plots means of higher cover values. One factor that may be to blame for insignificant differences in many variables, including vegetative cover, is spatial auto correlation. This statistical concept describes geographical relationships of observations to each other and can be described in two effectual circumstances (Ott and Longnecker 2001). One effect of spatial auto correlation is when an event at one location causes an event at another nearby location (Ott and Longnecker 2001). The second effect of spatial auto correlation is when an event in one location is similar to an event at another nearby location by shear closeness (Ott and Longnecker 2001). Observations cannot be assumed independent if spatial auto correlation has been determined to be in effect; therefore, regression analysis (like that used here) cannot account for spatial relationships as a core assumption to these models is that all observations are independent (Ott and Longnecker 2001). In much the same way as collinearity prevented accurate analysis of surface-cover characteristics, spatial auto correlation prevents accurate analysis of rainfall-simulation events performed on a waste-application treatment plot. Unfortunately, geographical data of the specific location of individual rainfall simulations is not available for either summer or winter data sets and the ability to account for spatial auto correlation is lost. Another factor contributing to counterintuitive results is high variability in many of the measured values on all treatment plots. Noticeable in Figures 11, 12, and 13 are large box plots per waste application treatment indicating that all observed values fall within the large range covered by the box plot. The high variability in observed values and the overlapping of observations between treatments reduces that ability to distinguish differences between the treatment groups.

Comparison of Summer to Winter. The control plot and the 5, 10, and 15 y^3/acre waste treatment plots had rainfall simulations run both in July 2004 and January 2005; therefore, these plots were included in a seasonal analysis comparing infiltration rates observed in the summer and later observed in the winter. Graphed in Figure 14 is the average of infiltration rates of these four treatment plots in the summer simulation compared to the winter simulation at 5 min intervals over the entire 30 min simulation.

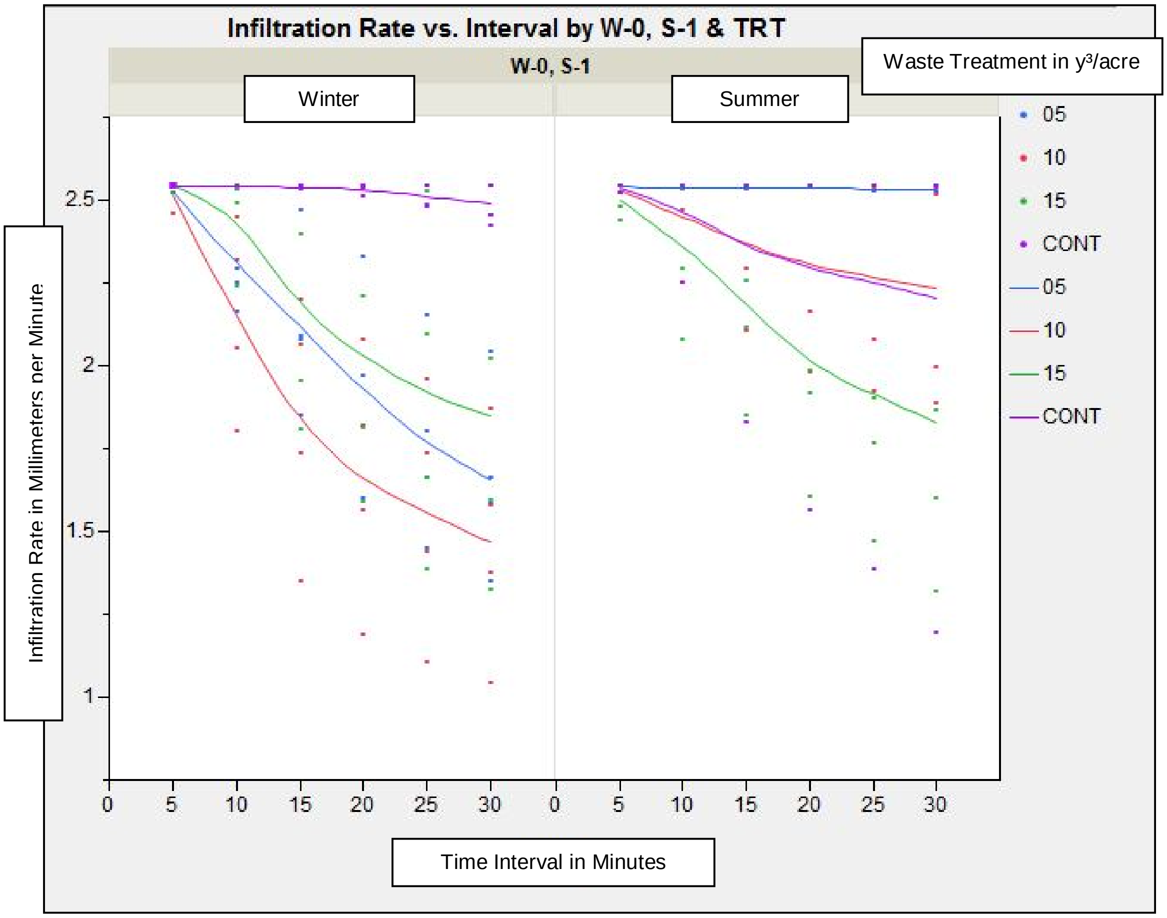


Figure 14. Average infiltration rate by treatment and by season of simulation, graphed at five minute intervals for the entire 30 min simulation.

Infiltration rates fell from the summer simulations to the winter simulations for all composted-waste treatments plots, but increased for the control plot. A decrease in infiltration rates should be expected from summer to winter due to a loss of vegetative activity and growth in the winter, reducing the positive effects of vegetative cover on infiltration rates and the root binding effects to soil aggregates. The literature review indicates that composted-waste amendments will incorporate into the soil over time increasing aggregates, increasing soil organic matter, and decreasing mean bulk density all working to increase infiltration rates, but this has not been observed on the Fort Hood Western Training Grounds landscape. To analyze if any of these changes occurred due to natural processes, first the maximum amount of infiltration by a plot was averaged by season of rainfall simulation without considering the effects of the treatment amendments. The two seasons are compared by a Tukey's HSD test, shown in Figure 15.

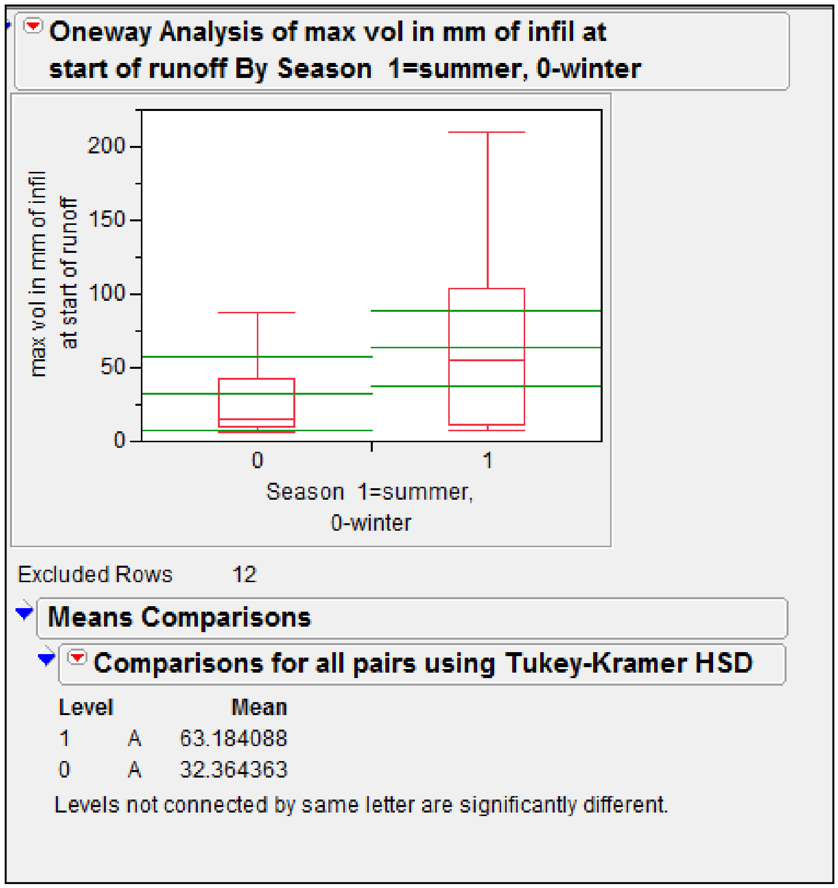


Figure 15. Box plot comparison of maximum volume of infiltration by season of rainfall simulation with a Tukey's HSD comparison and green bars representing the upper limit of a 95% confidence interval, the mean, and the lower limit of a 95% confidence interval.

Evident is the mean amount of maximum infiltration decreased from summer to winter but not by a significant amount. According to the Tukey's HSD comparative analysis, the maximum infiltration by a plot is not different from the summer simulation studies to the winter simulations studies. As determined by another study, a seasonal flux of infiltration rate is expected due to a coordinating seasonal flux in plant species composition (Thurow et al. 1988). Percent organic matter in the soil is next analyzed to determine if it changed from summer to winter without considering the effects of the treatment amendments, and the output is shown in Figure 16.

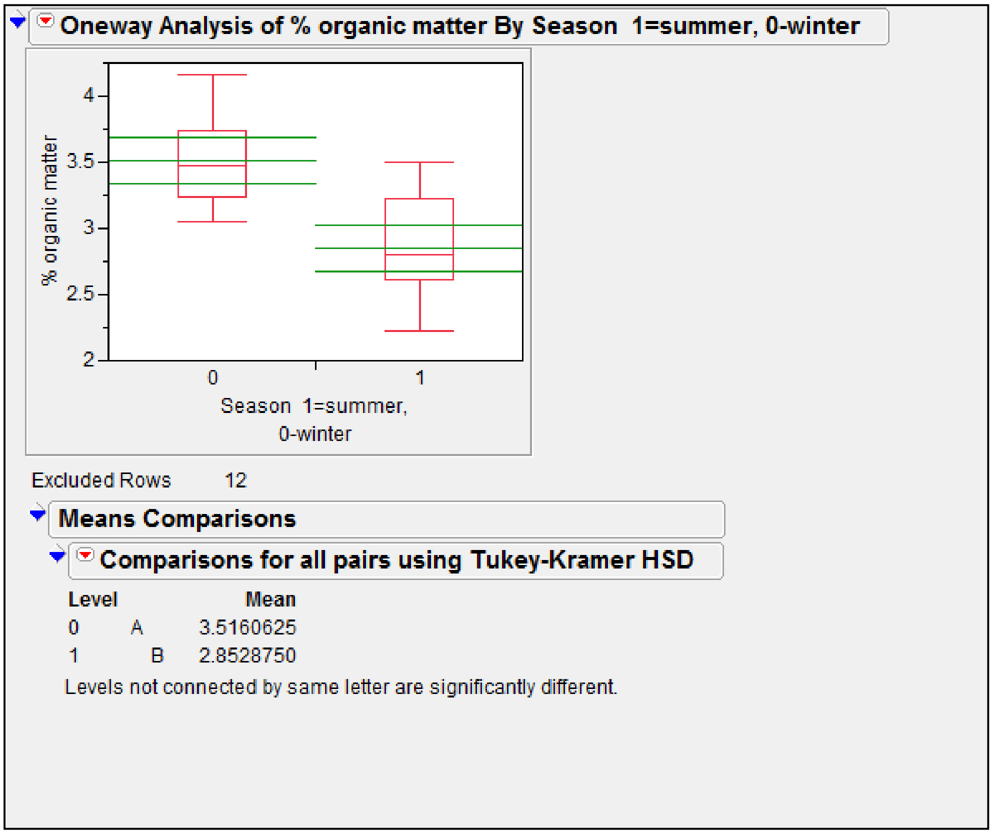


Figure 16. Box plot comparison of percent organic matter in the soil by season of rainfall simulation with a Tukey's HSD comparison and green bars representing the upper limit of a 95% confidence interval, the mean, and the lower limit of a 95% confidence interval.

Percent organic matter increased significantly in winter observations compared to summer observations by a Tukey's HSD comparative analysis. This increase can be contributed to the incorporation of newly deceased vegetation into the soil and the reduction of organic matter decomposition rates due to lower temperatures. An increase in organic matter in the winter could also help to elevate the amount of water a plot can hold, contributing to the insignificant differential comparison of maximum infiltration seen in

Figure 15 when a difference should be expected. All plots were individually found to experience a significant increase in organic matter from the summer to the winter except for the 5 y³/acre treatment plot. Figure 17 compares the percent organic matter for the 5 y³/acre waste application treatment in the summer to the winter. The mean percent

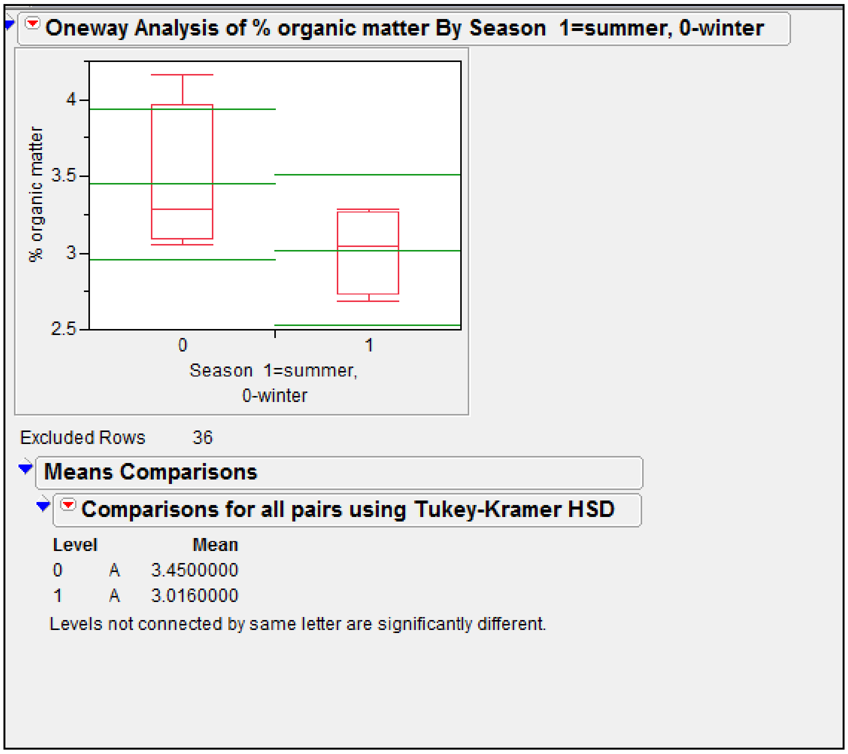


Figure 17. Box plot comparison of percent organic matter for the 5 y³/acre treatments by season of rainfall simulation with a Tukey's HSD comparison and green bars represent the upper limit of a 95% confidence interval, the mean, and the lower limit of a 95% confidence interval.

organic matter per season did increase from summer to winter for the 5 y³/acre treatment, but a Tukey's HSD comparative analysis determined that it was not significant. A significant increase in the control treatment's percent organic matter from summer to winter despite no amendment of composted waste could be contributed to the higher than average vegetative cover observed. More living vegetative matter in the summer equates to more organic matter available in the winter.

Bulk density is another soil characteristic positively affected by the percent organic matter in the soil through a negative relationship, and experienced a significant decrease from summer to winter as seen in Figure 18. Less bulk density means less soil compaction,

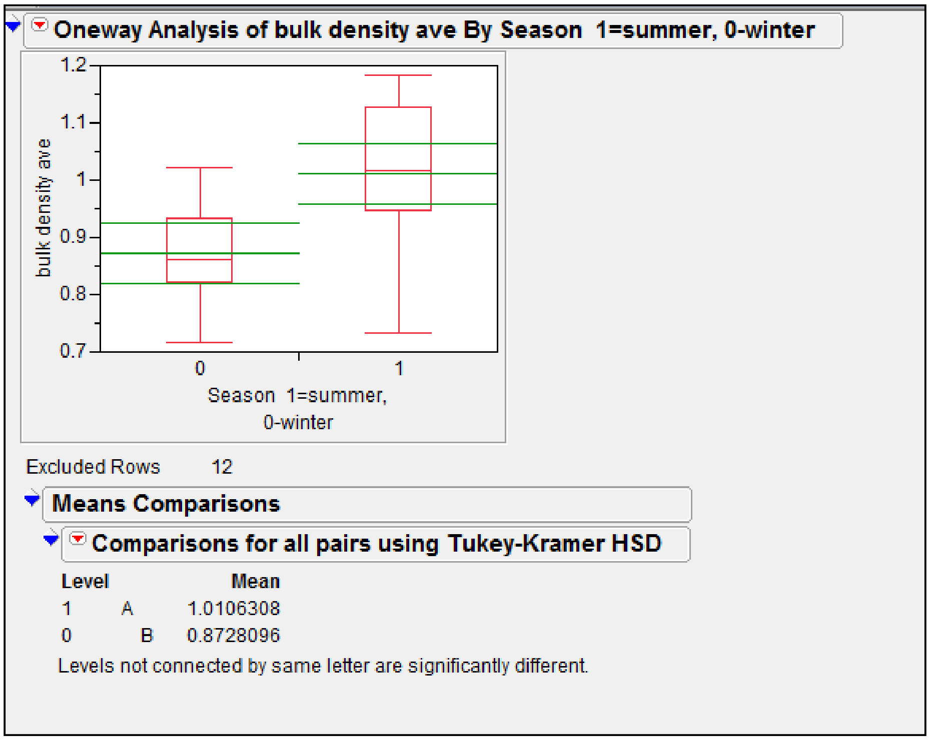


Figure 18. Box plot comparison of mean bulk density of the soil by season of rainfall simulation with a Tukey's HSD comparison and green bars representing the upper limit of a 95% confidence interval, the mean, and the lower limit of a 95% confidence interval.

more pore spaces, better maintenance of soil aggregates, and higher infiltration rates. Therefore, the decrease in bulk density from summer to winter highlights the incorporation of more organic matter into soil and a promotion of good soil qualities. This decrease in bulk density can also contribute to the response variable of maximum infiltration in the winter not being significantly different from summer maximum infiltration.

Questionable in the results here is the increase in infiltration rate on the control plots, as no amendments were applied to instigate changes. Bulk density decreased significantly from summer to winter on the control plot, although the reason is unknown as there were no amendments to explain it. The exclusion of the research area from military training exercises (a source of soil compaction) over the course of the entire experiment may have enabled initiation of restoration of soil characteristics. High vegetative cover mean values may have helped to influence infiltration rates as well as observed winter organic matter increases; but again, vegetative cover was found to be insignificant to the response of maximum infiltration and cannot be concluded as a significant factor. As was evident in the individual season analysis, the seasonal comparison analysis contained high variability in observations. Box plots covering large areas in Figures 15, 16, 17, and 18 indicate a wide range of values need in order to capture all observations. Even though many variables are found to be significantly different from summer to winter, there still exists overlapping observations found in one season that would be more expected in the other season. This high variability on the landscape naturally, remember Figures 15, 16, and 18 removed the effects of the waste application treatments, makes it difficult to determine any treatment effects. The high variability of this landscape contributes to the confounding results in the individual seasonal analyses.

CONCLUSION

Comparing the data by treatment without regard to season of application exposed the inability to capture observed relationships through the calculated variable infiltration rate. The essential flaw to the data set is the cessation of runoff collection at 30 min without consideration to whether runoff had even initiated. This prevented accurate comparison by pitting plots of known infiltration capacity versus plots of unknown infiltration capacity. Additionally, the distribution of the variable infiltration rate in this study does not fit widely utilized models and is unfit for analysis by traditional statistical techniques. Through division of data into summer and winter rainfall simulation sets and the creation of a new variable able to capture a plots maximum adsorption capacity, more accurate analysis was completed.

The statistical analysis of rainfall simulations in July 2004 indicates that composted treatment applications do not significantly affect the infiltration rate of a plot, and that other variables may be influencing the infiltration rate on this landscape. A stepwise model of regression identified percent organic matter as the only variable of significance to influence maximum infiltration for a plot when all measured soil characteristics, surface cover characteristics, and first order interactions between main effects and treatments were included. A Tukey's HSD comparative analysis did not identify any statistical differences among the control plot and all rates of treatment amendments indicating that composted waste additions did not significantly affect the percent organic matter within the soil of the research area. High variability in all variables observed for the summer data set unrelated to the rate of composted waste application effectively camouflages possible effects of the treatments.

Runoff nutrient analysis occurred only for the summer rainfall simulations and only for those plots that generated runoff in under the 30 min time constraint. A Tukey's HSD comparative analysis does identify a significant difference of nitrate-N in runoff between the 30 y^3/acre treatment plot and the 5 y^3/acre treatment plot, but for most treatment applications there was no significant difference from the control plot indicating little treatment effect. All recorded values of nitrate-N were well below TCEQ standards for drinking water and pose little risk for local water resource contamination. A Tukey's comparative analysis does not identify any significant differences of phosphate in runoff among the control plot and all treatment plots implying no treatment relationship to the amount of phosphate loss on this landscape. Phosphorous has no water quality standard in Texas; it is instead evaluated by a screening level at which it has the potential to influence excessive algal growth. One runoff sample (out of 16 total for all treatments) from the 20 y^3/acre treatment plot exceeded the screening level for phosphate, indicating a small risk of surface water quality contamination from runoff. The occurrence of a precipitation event comparable to intensities applied in this experiment is rare and reduces the possibility of local water source contamination from waste applied rangelands within the Fort Hood military installation.

The statistical analysis of rainfall simulations in January 2005 again indicates that composted-treatment applications do not significantly affect the infiltration rate, and that other variables besides treatment-application rate are influencing the maximum infiltration capacity of a plot. A stepwise model selection analysis results in the determination of a significant relationship between treatments of waste application and maximum infiltration, but a box plot comparison illustrates a loss of maximum infiltration for all treatment plots compared to the control treatment. Similar to the summer data set, high variability in all observed variables across all treatment-application rates effectively masks potential relationships between waste application and infiltration rate. Comparison of simulations performed in the summer to those performed in the winter indicates a significant increase of percent soil organic matter without considering the treatment application of the composted wastes. An additional significant decrease in bulk density from summer to winter further illustrates a very high natural variability on this landscape before treatments were applied.

Accurate analysis of both the summer and winter data sets was inhibited by the incomplete data collection across all treatments in both seasons. Inconsistent sampling techniques from the summer rainfall simulations to the winter rainfall simulations prevented a full analysis of all treatment-applied plots. The 20, 25, and 30 y^3/acre treatment plots were left out of the winter data analysis due to lack of data. This reduced the ability to compare changes in measured soil characteristics and infiltration rates over time, disabling the application of observed patterns at lower rates of application to higher rates of application. The inability to account for variable relationships like spatial auto correlation and collinearity may have contributed to statistical error through violation of model assumptions in many analyses performed. Additionally, plot specific soil classification is unavailable preventing accountability of significant changes in clay/loam/sand percentages affecting changes in infiltration rates. The lack of site specific soil characterization forces an assumption that county wide soil data is accurate down to the m^2 plot level, which very well may be incorrect. Degradation of the rangeland from military training exercises most likely has lead to a greater loss of specific components of the top soil, further negating the assumption of accurate large scale soil classification. Due to all of the above constraints in the data and a high variability naturally on this landscape, the conclusion that composted waste amendments increased infiltration rates on the Fort Hood Western Training Grounds cannot be confirmed. These results do confirm the conclusion that runoff from waste amended sites do not pose additional threat to nutrient levels in local water resources due to low levels of nutrients observed in runoff and the rarity of natural rainfall events as intense as water applied in this research to generate measurable runoff.

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APPENDIX A

Compost Technical Data from Organic Residual Reclamation, LLC

Compost Parameters	**Reported as**	**Test Results**	**Test Results**
Plant Nutrients:	% weight basis	% wet weight basis	% dry weight basis
Nitrogen	Total N	0.93	2.1
Phosphorus	P2O5	0.34	0.76
Potassium	K2O	0.41	0.92
Calcium	Ca	2.5	5.7
Magnesium	Mg	0.22	0.49
Moisture Content	% wet weight basis	55.3	
Organic Matter Content	% dry weight basis	54	
pH	units	7.59	
Soluble Salts	dS/m (mmhos/cm)	3.95	
Particle Size	% under 9.5 mm, dw basis	100	
Select Pathogens	PASS/FAIL: USEPA Class A standard	Pass: Fecal Coliform	
Trace Metals	PASS/FAIL: USEPA Class A standard	Pass: As, Cd, Cr, Cu, Pb, Hg, Mo, Ni, Se, Zn	

Metals and Coliform Bacteria	**Units, dry weight**
Arsenic (As)	2 mg/kg
Cadmium (Cd)	< 1 mg/kg
Chromium (Cr)	5 mg/kg
Copper (Cu)	91 mg/kg
Lead (Pb)	5 mg/kg
Mercury (Hg)	< 1 mg/kg
Molybdenum (Mo)	3 mg/kg
Nickel (Ni)	4 mg/kg
Selenium (Se)	3 mg/kg
Zinc (Zn)	112 mg/kg
Total Solids	44.70%
Fecal Coliform	< 2 MPN/g

APPENDIX B

Recorded simulation plot characteristics data table

Treatment	bulk density ave	% soil moisture ave	% organic matter	Mean MicroTopo standard deviation	% Grass cover	% forb cover	% baregound	% litter cover	Season 1=summer, 0-winter	N in runoff (mg/L)	P in runoff (mg/L)	start time of runoff (min)	max infil at start of runoff (mm)
CONT	1.18	24.40	2.41	0.86	24	1	45	30	1	0.99	0.07	4.33	10.9982
CONT	0.86	18.00	2.234	0.72	11	5	69	15	1	0.00	0.00	33	83.82
CONT	1.00	15.92	2.826	0.64	75	1	14	10	1	0.00	0.00	28.05	71.247
CONT	1.04	26.82	2.747	0.89	20	50	20	10	1	0.00	0.00	31.91	81.0514
5	0.73	18.31	2.687	0.9	25	1	59	15	1	1.23	0.17	7.06	17.9324
5	0.94	15.00	3.287	0.97	10	15	10	65	1	0.00	0.00	47.83	121.4882
5	0.98	12.04	2.886	0.67	50	7	8	35	1	0.00	0.00	52.2	132.588
5	1.02	11.67	3.204	0.52	4	20	42	34	1	1.52	0.19	20.16	51.2064
10	0.84	28.50	3.234	1.2	60	1	24	15	1	0.00	0.00	82.66	209.9564
10	1.02	24.16	2.223	0.74	60	1	19	20	1	0.00	0.00	4	10.16
10	1.17	21.13	2.587	0.47	21	3	26	50	1	0.99	0.09	23.33	59.2582
10	1.12	20.76	3.028	0.53	35	1	14	50	1	1.17	0.20	4.33	10.9982
15	1.13	21.86	2.763	0.78	55	3	27	15	1	0.00	0.00	2.75	6.985
15	1.01	18.99	2.765	0.7	38	7	40	15	1	1.00	0.24	3.41	8.6614
15	0.97	22.71	3.494	0.77	25	5	35	35	1	0.98	0.20	9.66	24.5364
15	1.15	19.76	3.271	0.88	80	3	5	12	1	0.00	0.00	43.33	110.0582
20	0.96	24.39	3.838	0.7	10	20	15	55	1	1.29	1.19	14.61	37.1094
20	0.99	22.55	4.71	0.63	10	40	15	35	1	0.00	0.00	50.66	128.6764
20	0.94	16.63	2.528	0.72	4	3	88	5	1	1.28	0.31	7.56	19.2024
20	0.84	34.01	3.208	0.58	3	6	81	10	1	1.27	0.65	9.65	24.511
25	0.82	20.38	3.557	0.94	3	7	75	15	1	0.00	0.00	20.62	52.3748
25	0.77	44.70	2.734	0.72	10	25	25	40	1	1.18	0.22	13.75	34.925
25	0.70	34.06	3.909	0.72	15	7	20	58	1	1.26	0.56	14.84	37.6936
25	0.72	29.41	3.806	0.63	5	1	24	70	1	0.99	0.11	9	22.86
30	0.86	28.11	3.838	0.78	80	3	7	10	1	0.00	0.00	43.87	111.4298
30	0.88	25.70	2.807	0.38	85	3	4	8	1	1.04	0.32	7.66	19.4564
30	1.03	32.27	3.335	0.72	65	3	20	12	1	1.00	0.41	16.01	40.6654
30	0.88	35.78	3.873	0.68	80	10	2	8	1	0.84	0.11	24.5	62.23

Treatment	bulk density ave	% moisture ave	% organic matter	Mean MicroTopo standard deviation	% Grass cover	% forb cover	% baregound	% litter cover	Season 1=summer, 0-winter	N in runoff (mg/L)	P in runoff (mg/L)	start time of runoff (min)	max infil at start of runoff (mm)
CONT	0.79	30.56	3.176	1.04	75	5	15	5	0			61.56	156.3624
CONT	0.84	30.93	3.174	0.51	65	5	5	25	0			14.16	35.9664
CONT	0.87	31.05	3.462	0.52	35	1	3	61	0			19.66	49.9364
CONT	0.84	35.39	3.78	0.62	75	3	12	10	0			34.66	88.0364
5	1.02	27.11	3.052	0.4	16	7	72	5	0			2.3	5.842
5	0.85	31.02	3.201	0.72	3	7	70	20	0			2.35	5.969
5	0.81	32.47	4.164	0.76	10	15	65	10	0			6.16	15.6464
5	0.92	30.63	3.383	0.56	12	5	33	50	0			3.56	9.0424
10	0.91	29.93	3.689	0.54	10	20	40	30	0			5.58	14.1732
10	0.72	32.97	3.336	0.75	5	7	73	15	0			3.89	9.8806
10	0.94	30.15	3.498	0.75	3	10	57	30	0			3.87	9.8298
10	0.82	30.47	3.822	0.9	5	5	50	40	0			8.24	20.9296
15	0.82	24.99	3.611	0.65	35	35	10	20	0			5.77	14.6558
15	0.88	35.58	3.747	0.63	25	20	30	25	0			17.85	45.339
15	0.95	35.34	3.716	0.62	40	25	15	20	0			9.35	23.749
15	0.98	28.93	3.446	0.42	19	1	40	40	0			4.91	12.4714

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