Landuse and Soil Property Effects on Infiltration and Soil Aggregate Stability in the Lower Mississippi River Valley

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Landuse and Soil Property Effects on Infiltration and Soil Aggregate Stability in the Lower Mississippi River Valley

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Crop, Soil, and Environmental Sciences

by

Rebecca Lynn Anderson
University of Arkansas
Bachelor of Science in Agricultural, Food and Life Sciences with a concentration in Environmental, Soil and Water Science, 2016

May 2019
University of Arkansas

This thesis is approved for recommendation to the Graduate Council.

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Abstract

Following European settlement of the Lower Mississippi River Valley (LMRV), agricultural expansion and unsustainable, agriculturally related practices have caused groundwater depletion, soil erosion, and surface water contamination by eroded sediments and sediment-bound nutrients to become major environmental threats to the region. The objective of this study was to evaluate the effects of common landuses [i.e., native prairie, deciduous forest, coniferous forest, Conservation Reserve Program (CRP) grassland, and conventional-tillage (CT) and no-tillage (NT) agriculture] on surface water infiltration and aggregate-stability-related properties [i.e., water-stable macroaggregate (WSA) size distribution, total water-stable macroaggregate (TWSA) concentration, and mean weight diameter (MWD)]. The overall infiltration rate for the deciduous forest (1.17 mm min$^{-1}$) was 9.25 times greater than the overall infiltration rate for the other five landuses, which did not differ and averaged 0.13 mm min$^{-1}$. The y-intercept characterizing the linear relationship between the natural logarithm of the infiltration rate and mid-point of time for the deciduous forest was more than 75% greater than the landuse with the next largest y-intercept, while the coniferous forest, native prairie, and CRP grassland had similar ($P > 0.05$) y-intercepts to each other. Total WSA concentrations in the top 10 cm in the native prairie, CRP, and coniferous forest were similar ($P > 0.05$) to each other, averaging 806 g kg$^{-1}$, and were 35% greater than that of the NT and CT agroecosystems, which did not differ and averaged 605 g kg$^{-1}$. Similarly, the MWD in the top 10 cm in the native prairie, CRP, and coniferous forest were similar ($P > 0.05$) to each other, averaging 2.15 mm, and were 70% greater ($P < 0.05$) than that of the NT and CT agroecosystems, which did not differ and averaged 1.27 mm. In the top 5 cm, the MWD and TWSA concentration were 17 and 8% greater, respectively, than that in the 5- to 10-cm depth interval. Results of this study demonstrated that
landuse affects surface infiltration, water-stable aggregation, and select near-surface soil physical and chemical properties in fine-textured, loessial and alluvial soils in the LMRV Delta region of eastern Arkansas.
Acknowledgements

I would like to thank Dr. Kristofor Brye for being an advisor and mentor I could depend on, throughout my graduate and undergraduate years. Dr. Kristofor Brye has inspired me to love soil science and has always motivated me towards academic excellence. I would also like to thank my committee members, Drs. Lisa Wood, David Miller, and Mike Richardson for their willingness to serve on my committee and for providing valuable advice on my research. Field assistance provided by Johan Desrochers, Marya McKee, Casey Rector, and Matt Thompson is gratefully acknowledged. Finally, funding for this research was provided by the Arkansas Natural Resource Conservation Service and is gratefully acknowledged.
Dedication

I wish to dedicate this thesis to my parents, who made many sacrifices to allow me to finish my education, always pushed me towards academic excellence, and made me believe I can do anything I set my mind to.
# Chapter 2: Landuse and Soil Property Effects on Infiltration in the Lower Mississippi River Valley

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Introduction
The Lower Mississippi River Valley (LMRV), one of the major river basins that comprise the Mississippi River Valley, is currently faced with several major issues: groundwater depletion, soil erosion, and surface water contamination by eroded sediments and sediment-bound nutrients (USDA-NRCS, 2013). Unsustainable groundwater withdrawals from the Alluvial Aquifer, the largest and shallowest aquifer in the LMRV, for agricultural irrigation purposes have been a main cause for the groundwater depletion issues the region faces (USGS, 2002). According to Arkansas Natural Resource Commission (ANRC), the amount of water that is annually withdrawn from the Alluvial Aquifer is approximately 226% of the estimated sustainable yield (ANRC, 2017). These extensive withdrawals have lowered aquifer levels and depleted groundwater reserves (USGS, 2002; Verkler et al., 2008). Furthermore, the conversion of native forests and grasslands to cultivated agricultural land has caused overall soil quality to decline, specifically surface infiltration capacity, and surface runoff and soil erosion to increase in the LMRV (Harper et al., 2008; USDA-NRCS, 2013). For instance, in 2013, almost one billion hectares of land in the LMRV were classified as highly erodible (USDA-NRCS, 2013). In addition, approximately 64 billion kilograms of sediment, 754 million kilograms of N, and 73 million kilograms of P are annually transported to rivers and streams within the LMRV (USDA-NRCS, 2013). Consequently, keeping excess sediment and potential aquatic pollutants in place on the terrestrial landscape is critical to reduce erosion and maintain water quality.

Groundwater can be naturally recharged in the LMRV through infiltration of rainfall and/or irrigation water into soil, where the excess gravitational water can eventually flow vertically downward to the aquifer (Alley, 2009). The ability of a soil to facilitate water infiltration is based primarily upon the physical and chemical properties associated with the soil surface, which are strongly influenced by landuse and land management practices (Harper et al.,
After reaching the soil surface, rainfall and/or irrigation water can either flow over the surface as runoff or can infiltrate into the soil in a process often referred to as runoff-infiltration partitioning (Horton, 1933; Harper et al., 2008).

Aggregate stability is one of the most important factors that control runoff-infiltration partitioning and soil erosion (Kemper and Rosenau, 1986; Graham et al., 1995). Soil aggregates are defined as secondary soil particles that are held together by soil organic matter (SOM), clay, or other cohesive materials (Smith et al., 2014). Consequently, aggregate stability is the degree to which soil aggregates can withstand disintegrating forces, including weathering, slaking, precipitation, and erosion (Kemper and Rosenau, 1986; Smith et al., 2014) and is an important indicator of soil structural stability and overall soil quality (Six et al., 2000). Generally, decreased aggregate stability causes soil crust formation and decreased soil pore continuity (Tisdall and Oades, 1982), which can lead to decreased infiltration and increased runoff and erosion (Le Bissonnais and Arrouays, 1997; Franzluebbers, 2002; Wuest et al., 2005; Stone and Schlegel, 2010; Alaoui et al., 2011).

Apart from severely limiting groundwater withdrawals for agricultural irrigation, a potentially effective way to increase Alluvial Aquifer recharge would be to identify landuses, land management practices, and soil surface properties that reduce runoff, improve infiltration-related, soil-surface properties to allow more rainfall and/or irrigation water to infiltrate to potentially recharge the Alluvial Aquifer. By increasing infiltration, the threat of soil erosion and sediment loading of nearby surface waters would also decrease, thereby increasing the future sustainability of the soil and water resources in the LMRV (Harper et al., 2008). Furthermore, identifying landuse and land management practices that improve aggregate stability could indirectly increase infiltration and reduce surface runoff and erosion (Le Bissonnais and
Arrouays, 1997; Franzluebbers, 2002; Bhattacharyya et al., 2008; Stone and Schlegel, 2010) and may lead to decreased contamination of surface waters by eroded sediments and sediment-bound nutrients as well as enhanced opportunity for groundwater recharge in the LMRV (USGS, 2002; USDA-NRCS, 2013). Therefore, the goal of this thesis is to evaluate the combination of common landuses, land management practices, and soil surface properties in the LMRV that could contribute to reduced runoff and improved infiltration to potentially enhance recharge of the Alluvial Aquifer.
Literature Cited


Chapter 1

Literature Review
Literature Review

Lower Mississippi River Valley

The Mississippi River Valley extends from the confluence of the Ohio and Mississippi Rivers in southern Illinois to the southern border of Louisiana (Oswalt, 2013). The Lower Mississippi River Valley (LMRV) is a sub-region of the Mississippi River Valley, which includes parts of Arkansas, Mississippi, and Louisiana (Rudis and Birdsey, 1986).

Cropland is the largest landuse in the LMRV, covering over 6.5 million hectares and accounting for 5% of the national cropland and 33% of the non-water area in the LMRV (USDA-NRCS, 2013). In the LMRV, the major non-water, non-urban landuses, where urban landuse accounts for 6% of the LMRV, in order of the greatest percentage landcover are: cropland (33%), wetlands (21%), forest (25%), and pasture/haylands (14%; USDA-NRCS, 2013; Figure 2). Cropland consists of approximately 28% no-tillage and 53% mulch-tillage (USDA-NRCS, 2013). Forests consist of approximately 48% coniferous, 39% deciduous, and 13% mixed forests (USDA-NRCS, 2013). Many land surface features in the LMRV are desirable for cultivated agriculture. For instance, the land in the LMRV is relatively flat. In fact, 92% of the cropland in the LMRV has slopes less than 2% (USDA-NRCS, 2013). In addition, the soil of the LMRV is dominated by loess and alluvial parent materials (Lindbo et al., 1994; Karstensen and Sayler, 2009), which tend to be fertile and highly agriculturally productive (Oswalt, 2013). The alluvial soils of the LMRV are particularly fertile because floodwaters previously deposited nutrients and organic matter along with soil sediments (Oswalt, 2013). The LMRV landscape has been altered by deforestation due to agricultural expansion (Oswalt, 2013). In 1930, forest covered approximately 4.8 million hectares, and was comprised of 89% bottomland hardwood forests. The main forest-type groups of the bottomland hardwood forests were oak (Quercus spp.)-gum
(Eucalyptus spp.)-cypress (Cupressus spp.) and elm (Ulmus spp.)-ash (Fraxinus spp.)-cottonwood (Aigeiros spp.) forests. The total forested land area (TFLA) was approximately 44.2% of the total land area (Oswalt, 2013). Other landuses that covered the remaining land area in 1930 included native prairies and non-forested wetlands (Brye and Pirani, 2005; The Trust for Public Land, 2016). By 2013, TFLA of the LMRV had decreased to only 28.5% (Oswalt, 2013).

Realizing how fertile the alluvial soil of the LMRV was for agriculture, producers, especially of soybean (Glycine max L.), quickly began converting forested land to cultivated agriculture (Oswalt, 2013). For example, TFLA ranged from 30.6 to 54.3% and averaged 42.8% in 1950, whereas by 2010, TFLA ranged from 20.5 to 44.4% and averaged only 30.2% throughout Arkansas, Lee, Monroe, and St. Francis Counties in eastern Arkansas (Oswalt, 2013).

Prior to European settlement, native prairie grasslands covered large areas of the LMRV (Brye and Pirani, 2005). Many native grasslands, however, were converted to cultivated agricultural use (Brye and Pirani, 2005). By 2000, grasslands covered only 0.4% of the LMRV (Karstensen and Sayler, 2009). Much of the Grand Prairie, which was once a large expanse of native prairie located in east-central Arkansas, was converted to cultivated cropland (Brye and Pirani, 2005). In 2005, less than 1% of the original Grand Prairie remained (ANHC, 2013). The majority of the land in the LMRV was converted for cultivated agriculture and, in 2000, the LMRV consisted of 65.6% agricultural land (Welch et al., 2009). Many studies have shown that the conversion of land from native landuses to cultivated agriculture causes overall soil quality to decline (Cambardella and Elliott, 1993; Martens, 2000; Six et al., 2000a; Bodhinayake and Si, 2004; Harper et al., 2008), which indicates that soil quality in the LMRV has likely declined since the expansion of agriculture. As soil quality declines, the soil becomes susceptible to large amounts of runoff and erosion. Tillage causes the soil surface to be bare and exposed to the
impact of raindrops, which impact the soil surface and cause soil particles to become dislodged (Pimentel et al., 1995). Because no vegetation is present on the soil surface to slow down runoff, loose soil particles are easily carried away with the water (Pimentel et al., 1995). Loess, one of the dominant parent materials of the LMRV (Lindbo et al., 1994), is greatly susceptible to water erosion (Fu, 1989) and has been shown to rapidly erode when the soil surface is exposed (Fu, 1989). Loess soil is mainly comprised of silt-sized soil particles that are relatively small compared to sand-sized particles and can, therefore, be more easily transported in water (Fu, 1989). In addition, silt particles do not have the adhering properties of clay, and therefore, tend to be attracted to other soil particles and organic matter to a lesser degree than clay particles are. This can result in silt particles dislodging from the soil surface and being transported relatively easily (Amézketa, 1999). Resultantly, loess soils in the LMRV can be extremely susceptible to erosion. In 2013, almost one million hectares of land in the LMRV were classified as highly erodible and over 130,000 hectares of land were classified as having large runoff potential (USDA-NRCS, 2013).

The Mississippi River Valley also contains the underlying Alluvial Aquifer, which lies below 80,000 km² of land surface in the LMRV region. The majority of the Alluvial Aquifer lies in the LMRV and is the largest and shallowest aquifer in the LMRV (USGS, 2002). The Alluvial Aquifer is mainly comprised of unconsolidated coarse-textured alluvial deposits and is overlain by a unit of fine-textured deposits called the Mississippi River Confining Unit (ANRC, 2017). The confining unit, also known as a “clay cap”, is less permeable than the Alluvial Aquifer itself below (USGS, 2002). The Alluvial Aquifer is used extensively for agricultural purposes. According to the 2016 Arkansas Groundwater Protection and Management Report, 98% of the water that was extracted from the Alluvial Aquifer was used for agricultural irrigation (ANRC,
Approximately 44% of cropped land in the LMRV is irrigated (USDA-NRCS, 2013). Rice (*Oryza sativa* L.) and soybean are the two most produced crops in eastern Arkansas. In 2016, approximately 615,000 hectares of rice (UA-DA-CES, 2017A) and 1.26 million hectares of soybean were harvested in Arkansas (UA-DA-CES, 2017B). Both rice and soybean require large amounts of irrigation for optimum productivity. For instance, in the Delta Region of Mississippi, which is a subsection of the LMRV, rice and soybean are irrigated with approximately 1.29 and 1.27 billion m$^3$ of water annually (Kebede et al., 2014).

The extensive annual withdrawals of groundwater from the Alluvial Aquifer for agricultural irrigation purposes in the LMRV have lowered aquifer levels and depleted groundwater (Verkler et al., 2008). In the LMRV region, the Alluvial Aquifer is shallow (AGS, 2008), with an average depth to the Alluvial Aquifer in Arkansas typically being between 15 to 45 m (USGS, 2002), thus making the aquifer relatively easily accessible with wells. Consequently, the Alluvial Aquifer is currently being overdrawn at unsustainable rates, where approximately 7,636 Mgal d$^{-1}$ (million gallons per day) of water are being extracted annually (ANRC, 2017). However, the Alluvial Aquifer is naturally recharged at a rate of only approximately 3,374 Mgal d$^{-1}$ (ANRC, 2017). Therefore, the amount of water that is discharged from the Alluvial Aquifer is 226% of the estimated sustainable yield. In the last decade, the average water level of the Alluvial Aquifer has declined by an average of 0.52 meters (ANRC, 2017). Cones of depression have formed in the Alluvial Aquifer due to overuse (USGS, 2002; ANRC, 2017; Figure 5). In areas where cones of depression have occurred, sustainable withdrawal rates have decreased to less than 0.15 Mgal d$^{-1}$, which is insufficient for supporting agriculture (USGS, 2002). The amount of water that is being discharged must be greatly reduced in the near future in order to avoid irreversible groundwater depletion of the Alluvial Aquifer. If
demand for use of the groundwater does not decrease, then increasing the recharge rate must occur.

*Groundwater Recharge*

Several major attempts to conserve water resources and alleviate groundwater depletion of the Alluvial Aquifer have been performed in the last several decades (Tacker et al., 2010). In particular, the US Army Corps of Engineers designed the Grand Prairie Area Demonstration Project (GPADP) that was authorized by US Congress in 1950 (Hill et al., 2006). The goal of the GPADP was to reduce Alluvial Aquifer withdrawals and increase irrigation efficiency by transporting water from the White River through a network of pipelines and canals to farms throughout the Grand Prairie (Hill et al., 2006). After being delivered, water is stored in tailwater recovery systems and used for irrigation proposes (Hill et al., 2006). However, many concerns and delays have occurred since the initiation of the GPADP, and the final cost and environmental impact of this project is still unknown (Tacker et al., 2010).

Groundwater can also be naturally recharged through infiltration of precipitation and/or irrigation water into soil, where the excess eventually flows vertically downward to an aquifer (Alley, 2009). Groundwater recharge is accelerated when the water table below an area of land is relatively shallow (Alley, 2009). Because the water table for the Alluvial Aquifer is relatively shallow (Ausbrooks and Prior, 2008), groundwater can be recharged through precipitation. The Mississippi River Confining Unit can restrict downward flow of water, especially in areas where the confining unit is relatively thick (USGS, 2002). A recent study suggested that approximately only 1 to 1.5% of total precipitation in the Mississippi River Valley is able to recharge the Alluvial Aquifer (Kresse et al., 2014). Consequently, if the infiltration rate of the land in the
LMRV is increased, groundwater recharge of the Alluvial Aquifer will likely increase. Water-level changes in the Alluvial Aquifer have been shown to be consistent with changes in precipitation (ANRC, 2017). Data collected from 1997 to 2016 revealed annual water level changes in the Alluvial Aquifer seem to be directly impacted by and follow the pattern of annual total precipitation. The ability of a soil to allow water to infiltrate is based on the physical properties associated mainly with the soil surface, which are closely tied to landuse management practices. Therefore, different soil physical properties or combinations of landuse management practices, which lead to increased infiltration, need to be identified to help increase groundwater recharge.

*Runoff-infiltration Partitioning*

After reaching the soil surface, precipitation or irrigation water can either flow over the surface as runoff or can infiltrate into the soil. The division of precipitation and/or irrigation water into these two separate paths is called runoff-infiltration partitioning (Horton, 1933). Two main mechanisms control runoff-infiltration partitioning: i) the infiltration-excess mechanism and ii) the saturation-excess mechanism (Horton, 1933; Harper et al., 2008). The infiltration-excess mechanism states that runoff is produced when precipitation/irrigation water is delivered to the soil surface at a faster rate than the soil is able to absorb the water. The saturation-excess mechanism states that runoff can occur when hydraulic conductivity is limited by a sub-surface restrictive layer (Horton, 1933; Harper et al., 2008). When the soil above a restrictive layer becomes saturated, the soil water has no place to flow and/or no more pores to fill, thus infiltration is no longer possible, forcing any remaining water delivered to the soil surface to become runoff or ponded water (Dunne and Black, 1970; Harper et al., 2008). Therefore, runoff
and infiltration are tandem processes, such that as infiltration decreases, runoff will increase and visa versa (Harper et al., 2008; Stone and Schlegel, 2010).

Infiltration occurs in soil due to differences in potential energy of water within the soil profile (Clark, 1990). Water always flows from a point of greater to a point of lower potential energy (Easton, 2016). The potential energy of soil water is influenced by a combination of matric and gravitational potentials (Clark, 1990). Gravitational forces can cause water to infiltrate deeper into the soil profile. Gravitational potential is determined through the soil profile’s position in the gravitational force field and occurs irrespective of soil properties (Clark, 1990). Matric potential, however, is influenced by adhesive and cohesive forces in the soil matrix (Easton, 2016). Adhesive forces cause water to be attracted to the inner surface of soil pores, and cohesive forces cause water molecules to be attracted to one another (Easton, 2016). Adhesive and cohesive forces work together to suck water in to the soil during infiltration and cause water to be held tightly in pores of the soil matrix (Easton, 2016). The smaller the diameter the pore is, the more tightly the water will be held inside the pore, which is why infiltration and internal redistribution generally occur at slower rates in soils with large clay contents, and thus smaller pore sizes, compared to soils with large sand contents, and thus larger pore sizes (Easton, 2016). Soil infiltration can be quantified multiple ways, including cumulative infiltration and infiltration rate. Cumulative infiltration is the total amount of water that infiltrates over some period of time (Chu, 1978). Infiltration rate is the velocity at which water travels after it has entered the soil (USDA-NRCS, 2014b).

Increased runoff can cause many detrimental effects, including increased soil erosion and contamination of nearby surface waters by eroded sediments and sediment-bound nutrients, like phosphorous, and/or chemicals, like pesticides (Harper et al., 2008). For instance 10% of lakes
and reservoirs and 14% of rivers assessed in 2008 in Arkansas were classified as impaired due to nutrient-related causes (EPA, 2017). Consequently, increased infiltration and decreased runoff results in multiple environmental benefits in addition to groundwater recharge.

Soil Property Effects on Infiltration

Many soil properties can affect infiltration, including soil texture, type of clay, antecedent soil moisture, bulk density, organic matter content, and aggregate stability (Harper et al., 2008; USDA-NRCS, 2014a). Soil texture is determined by the mixture of sand, silt, and clay particles in a soil (USDA-NRCS, 2014b). As the percentage of sand in a soil increases, infiltration will likely increase due to the sand particles, which are the largest of the three soil separates, creating relatively large pores between themselves to result in large hydraulic conductivity when wet (USDA-NRCS, 1998, 2014b). Conversely, as the percentage of clay in a soil increases, infiltration tends to decrease because the smaller clay particles create relatively smaller soil pores that restrict water flow (USDA-NRCS, 1998, 2014b).

The type of clay that makes up the clay fraction of a soil can influence infiltration. Clays that shrink when dry and swell when wet are referred to as shrink-swell clays. Widely varying infiltration rates can occur for shrink-swell clays. When shrink-swell clays dry and dehydrate, the crystalline clay structure collapses or shrinks, resulting in the formation of cracks in the soil and at the soil surface, which, when open, act as large conduits for water, therefore greatly increasing the infiltration rate of that soil. However, after becoming wet and rehydrated, the clay crystalline clay structure expands or swells, causing cracks to close and infiltration to greatly decrease or cease altogether (USDA-NRCS, 2014b).
Similar to texture and clay type, antecedent soil moisture can greatly impact infiltration (Harper et al., 2008). Soils with large or small moisture contents generally have low infiltration rates compared to soils with intermediate moisture contents (Harper et al., 2008). In general, as the moisture content of a soil increases, the hydraulic conductivity of the soil also increases (Harper et al., 2008). Therefore, soils that are dry generally do not conduct water well and, thus, have low infiltration rates (Harper et al., 2008). Conversely, soils with moisture contents near saturation have little pore space to store more infiltrating water, which also leads to low infiltration (Harper et al., 2008).

Bulk density of a soil can also significantly affect infiltration (Patel and Singh, 1981). As the bulk density of a soil increases, the soil becomes more compacted, and the pore space available to store water decreases, which causes the soil to not be able to transmit water as quickly and infiltration to decrease (USDA-NRCS, 2014b; Sajjadi, 2016).

Soil organic matter is also an important factor affecting the process of infiltration (Franzluebbers, 2002). Because the particle density of mineral soil is so much greater than that of organic matter, greater amounts of organic matter in a soil can cause the bulk density of the soil to decrease, which will result in greater pore space, which often leads to greater infiltration (Franzluebbers, 2002). In addition, the ability of a soil to hold and store water tends to increase when the soil organic matter (SOM) content increases, which can result in increased infiltration (USDA-NRCS, 2014b).

Aggregate stability is defined as the degree to which soil aggregates can withstand disintegrating forces (Kemper and Rosenau, 1986; Smith et al., 2014). Aggregate stability is measured by the concentration of water-stable macroaggregates in a soil. Macroaggregates are typically defined as aggregates with a diameter greater than 0.25 mm. (Oades, 1984). Water-
stable macroaggregates are measured by the concentration of macroaggregates that remain after the soil is mechanically disturbed by submergence in water. As soil aggregate stability decreases, soil aggregates are more likely to disintegrate, which will result in dispersed sediment filling pore spaces and blocking water from infiltrating through the pores (Harper et al., 2008). Therefore, increased soil aggregation generally leads to increased soil infiltration (Franzluebbers, 2002; Marquez et al., 2004; USDA-NRCS, 2014b).

Landuse and Management Practice Effects on Infiltration

Many landuse and management practices can affect infiltration, including soil disturbance (i.e., tillage, compaction) and vegetative cover (Harper et al., 2008; DeFauw et al., 2014). Generally, increased soil disturbance will result in decreased infiltration (USDA-NRCS, 2014b), and, therefore, increased runoff (Harper et al., 2008; Stone and Schlegel, 2010). Tillage is a common landuse management practice that disturbs the soil (Franzluebbers, 2002). Tillage can cause compaction of the soil, which reduces overall porosity and decreases infiltration (Azooz and Arshad, 1996; Sajjadi, 2016). Tillage also causes organic matter to be exposed and decompose at an accelerated rate, which will ultimately lead to decreased infiltration (Franzluebbers, 2002). In addition, the soil surface is typically left with little residue cover after tillage, which can facilitate the formation of soil surface crusts that can further impede infiltration (Andraski et al., 1985; USDA-NRCS, 1998).

Implementation of no-tillage (NT) agriculture can cause an increase in infiltration compared to conventional tillage (CT; Meek et al., 1992). Many soil physical and chemical interactions can result in increased infiltration into NT agroecosystems. For instance, earthworm and root channels, which function as natural conduits for water infiltration, are preserved in NT
agroecosystems, whereas earthworm and root channels are frequently destroyed by cultivation (Azooz and Arshad, 1996; USDA-NRCS, 2014b). No-tillage agriculture also maintains greater amounts of surface residue cover than cultivated agriculture (Andraski et al., 1985). Surface residue cover can cause precipitation to evaporate at a slower rate compared to bare soil, which allows water a greater amount of time to infiltrate (Mukhtar, 1985; Harper et al., 2008) and keeps the soil wetter and more conductive than dry soil. Surface residue can also prevent soil crusts from forming, which can restrict infiltration (Azooz and Arshad, 1996). Roth et al. (1988) compared infiltration in soils with and without residue cover using a rainfall simulation over a 60-minute period. One hundred percent of the water applied to the soil with the residue cover infiltrated, whereas only 20% of the water applied to the bare soil surface infiltrated (Roth et al., 1988).

Azooz and Arshad (1996) conducted a study in Alberta, Canada on the long-term effects of NT compared to CT agriculture on infiltration rate into Donnelly silt-loam (sandy-skeletal, mixed Typic Haplocrypts) soils. Results indicated that NT practices increased infiltration rates compared to CT (Azooz and Arshad, 1996). The NT caused an increase in mean steady-state infiltration rates, which ranged from approximately 27 to 225% for the silt-loam soils during the 2-yr study. Azooz and Arshad (1996) attributed the difference in infiltration rate to the fact that CT disrupted the continuity of soil pores, which allowed water to infiltrate into the soil. The NT sites had more old or decomposing root channels, which provided conduits for water flow, compared to the CT sites (Azooz and Arshad, 1996).

Andraski et al. (1985) compared runoff in moldboard-plow CT and NT agriculture on a Griswald silt loam (fine-loamy, mixed, mesic Typic Argiudoll) near Arlington in south-central WI. Simulated rainfall events were used to measure total runoff from each site. No-tillage was
shown to decrease runoff by an average of 60.8% compared to CT agriculture in four out of the six measurement periods over a total of four years. This decrease in runoff from NT sites caused an 80 to 90% decrease in erosion from NT sites compared to cultivated agriculture sites across all six measurement periods (Andraski et al., 1985).

Generally, native prairie soils, which have never been subject to tillage or agricultural production, have greater infiltration rates than soils that are used for agricultural production (Harper et al., 2008). Increased infiltration rates into native prairies results from typically greater aggregate stability, SOM content, hydraulic conductivity, and earthworm activity compared to cultivated soils (Lindstrom et al., 1994). Native prairie sites have been shown to have increased SOM contents and decreased bulk densities compared to cultivated agroecosystems, due to the fact that the aboveground biomass of cultivated agroecosystems is removed during harvest and not allowed to return to the soil and replenish the SOM (Bodhinayake and Si, 2004). In addition, the soils under native prairie sites are not disturbed, which allows root channels and macropores from the activity of soil biota to form and persist (Bodhinayake and Si, 2004). Macropores are large pores that are typically formed by the spaces between macroaggregates (Huang et al., 2002). As a result, preferential flow and infiltration are generally greater in native prairie soils compared to cultivated agricultural soils (Bodhinayake and Si, 2004).

Harper et al. (2008) compared water runoff from a native tallgrass prairie and two types of cultivated agroecosystems (i.e., ridge-tillage agriculture and CT agriculture) on Immanuel silt-loam (fine-silty, mixed, active, thermic, Oxyaquic Glossudalf) soils in the Mississippi River Delta region of east-central Arkansas. The two agricultural systems had greater runoff than the native prairie, where the percentage of total runoff from the ridge-tillage system and CT system was 47 and 34%, respectively, greater than that from the native prairie. This indicated that less
water was able to infiltrate into the agricultural systems than into the native prairie (Harper et al., 2008), presumably due to a combination of poorer soil surface structure and sub-surface compaction restricting infiltration.

Bodhinayake and Si (2004) compared steady-state infiltration rates into native prairies and a cultivated agroecosystem with clay-loam (fine-loamy, mixed, frigid, Typic Haplustolls) soils in the province of Saskatchewan in central Canada. The native prairie site had never been cultivated, and the cultivated agroecosystem was under a dryland wheat (*Triticum* spp.)-canola (*Brassica* spp.) rotation with fallow for the previous 50 years. Double-ring infiltrometers were used to measure the steady-state infiltration rate, which was used to estimate saturated hydraulic conductivity. The estimated saturated hydraulic conductivity of the native prairie was four-fold greater than that of the cultivated agroecosystem. The results of this study indicated that the native prairie was able to infiltrate water at a quicker rate than the cultivated agroecosystem over an extended period of time. The increased saturated hydraulic conductivity in the native prairie may be explained by the fact that the organic carbon content was also shown to be 64% greater in the native prairie compared to the agroecosystem. In addition, the native prairie had 70% greater macroporosity compared to the cultivated agroecosystem. The continuity of the macropore network was destroyed in the cultivated agroecosystem, which may be the reason the steady-state infiltration rate was decreased in the cultivated agroecosystem compared to the native prairie (Bodhinayake and Si, 2004).

The majority of native prairies, which have never been cultivated, are still subject to multiple management practices that could affect the soil structure and infiltration rate (Daniel et al., 2002). For instance, burning, grazing, and mowing have been shown to decrease soil quality (Bharati et al., 2002; Brye, 2006). Periodic burning and/or mowing are common management
practices for native prairies. Burning helps to control weeds and woody encroachment, decrease the chance of wildfires, and promotes growth (Davison and Kindscher, 1999). Mowing also helps control weeds and undesired grasses, as well as prevent the invasion of shrubs or trees into the grassland (Davison and Kindscher, 1999). Grazing is a management practice occasionally used on native prairies for the purpose of stimulating plant growth, as well as economic benefits of livestock production (Davison and Kindscher, 1999). Over an extended period of time, native prairie management practices could cause the infiltration rate to decrease (Bharati et al., 2002).

Some agricultural lands have various limitations that restrict their productivity, such as poor drainage, steep slopes, shallow bedrock, among others. Converting highly erodible agricultural land to more natural, perennial vegetation, including grasses or trees, has been shown to increase infiltration and overall soil quality, and decrease soil erosion (Gebhart et al., 1994; Gilley and Doran, 1997; Lindstrom et al., 2004). In 1985 the Farm Service Agency (FSA) of the United States Department of Agriculture (USDA) established the Conservation Reserve Program (CRP), in which former agricultural land is converted to perennial grassland or trees (Wachenheim et al., 2014) with the aid of economic incentives. Farmers typically sign a 10-year contract agreeing to maintain permanent groundcover in exchange for financial benefits, often in the form of tax breaks and/or annual rent payments (Gilley and Doran, 1997; USDA-FSA, 2016). By 2016, a total of 9.6 million hectares of land were enrolled in the CRP program. The CRP seeks to enroll land that is highly susceptible to erosion. Approximately 47% of the agricultural land enrolled in the CRP is classified as highly erodible (USDA-NRCS, 2013). By enrolling land that is highly erodible and prone to runoff and soil erosion, the goal is to improve soil quality over time by decreasing runoff and erosion and increasing infiltration (USDA-NRCS, 2013).
Gilley and Doran (1997) compared runoff in a CRP grassland and moldboard-plow CT agroecosystem on a Granada silt loam (fine-silty, mixed, thermic Glossic Fragiudalf) in northern Mississippi. Runoff was measured using two, 60-minute rainfall simulations. Runoff from the undisturbed CRP grassland was 37 and 24% less than runoff from the long-term cultivated agroecosystem following the initial and subsequent rainfall simulations, respectively. This demonstrated that perennial grasslands may be able to significantly decrease runoff, and therefore increase infiltration, compared to cultivated agriculture (Gilley and Doran, 1997).

Bharati et al. (2002) compared infiltration among deciduous forest, grassland, and cultivated agriculture on Coland loam and sandy loam (fine-loamy, mixed superactive, mesic Cumulic Endoaquoll) soils in central Iowa. The deciduous forest and the grassland were established five years prior to infiltration measurements. The grassland was planted with switchgrass (Panicum virgatum L.) and was similar to a CRP grassland. The deciduous forest was similar to a CRP forest and included hybrid poplar (Populus X euramericana ‘Eugenei’), green ash (Fraxinus pennsylvanica Marsh), silver maple (Acer saccharinum L.), and black walnut (Juglans nigra L.). Cumulative infiltration was measured using a double-ring infiltrometer over a period of 60 minutes. There was a five-fold increase in cumulative infiltration in the less-disturbed deciduous forest and grassland compared to the cultivated agricultural sites (Bharti et al., 2002).

Other soil disturbances, such as compaction due to traffic, can affect infiltration (USDA-NRCS, 2014b). For example, Meek et al. (1992) measured infiltration rates into soil managed as a NT cotton (Gossypium spp.) agroecosystem with and without vehicle traffic in the western United States in a Wasco sandy loam (coarse-loamy, mixed, nonacid, thermic, Typic
Torriorthent). Results showed that water infiltration rate into the non-trafficked soil was approximately 73% greater than that of the trafficked soil (Meek et al., 1992).

Bai et al. (2009) also compared infiltration in various agroecosystems with different levels of vehicle traffic, including controlled-traffic and traditional-tillage agroecosystems, in silt-loam soils in the Loess Plateau in eastern China. Infiltration rates were measured over a 20-minute period using a double-ring infiltrometer. Controlled-traffic agriculture confines vehicles to specific traffic lanes, which are separated from crop zones. This disallows vehicles to compact the areas where the crop is growing. The controlled-traffic agriculture had a 67% greater final water infiltration rate than the traditional-tillage agriculture (Bai et al., 2009).

Infiltration is also influenced by the type of vegetation present. For example, studies have shown that deciduous forests can have greater infiltration rates than coniferous forests (Butzen et al., 2015; Wang et al., 2009). Coniferous forests typically have greater soil water repellency (SWR) than deciduous forests (Woche et al., 2005; Doerr et al., 2006), which likely contributes to decreased infiltration in coniferous compared to deciduous forests.

Butzen et al. (2015) compared SWR and runoff in coniferous and deciduous forests in the Hunstrück low mountain range in Germany in soil textures ranging from sand to sandy loam. The coniferous forest consisted of mainly spruce (Picea spp.) and the deciduous forest consisted of mainly beech (Fagus spp.). Soil water repellency was measured using the water-drop-penetration-time (WDPT) method. Field measurements of mean WDPT in seconds were 215% greater in the coniferous compared to the deciduous forest. Runoff coefficients were measured using a 60-minute rainfall simulation in both the coniferous and deciduous forest sites. A runoff coefficient (RC) is a unitless ratio that relates the amount of rainfall a soil receives compared to the amount of runoff over a given period of time. A greater RC indicates greater runoff and less
infiltration. Runoff coefficients in the coniferous forest were greater than that in the deciduous forest, which may be explained by the greater SWR in the coniferous forest. The median RC for highly water-repellant forest soils was 11.4% compared to 2.7% for less water-repellent soils. In addition, volumetric water content in the top 10 cm of soil was 31% less in the water-repellant forest soils compared to that of the less water-repellant soils, which indicated that less water was able to infiltrate into the water-repellent soils. Results of this study indicated that runoff was lower, and, therefore, infiltration rate was greater in deciduous compared to coniferous forests (Butzen et al., 2015).

Similar results for infiltration differences between forest types were reported by Wang et al. (2009) from residual and alluvial soils in the Simian Mountains of southwestern China. Wang et al. (2009) studied infiltration in coniferous and deciduous forests that were 25- and 20-years old, respectively. A double-ring infiltrometer was used to measure infiltration. The deciduous forest’s initial infiltration rate, stable infiltration rate, and cumulative infiltration were 62, 85, and 160%, respectively, greater than that of the coniferous forest (Wang et al., 2009).

The ability of a forest soil to allow infiltration also depends on the forest’s age (Bens et al., 2007). Generally, as the age of a forest increases, infiltration into the forest soil increases (Hümann et al., 2011). Infiltration rate in forested soils is in large part determined by the number and size of soil macropores (Aubertin, 1971). Many macropores are formed through the decomposition of dead tree roots, which either remain empty or are partially filled with porous organic matter. These root channels form a network of macropores under the soil surface in which water can infiltrate and flow through quickly (Aubertin, 1971). Generally, the larger and older the trees in a forest are, the larger and more abundant the root channels will be (Archer et al., 2015).
Archer et al. (2015) compared hydraulic conductivity and runoff under three forests of different ages [i.e., 6- and 48-yr-old Scots pine (*Pinus sylvestris*) plantations and a 4,000-yr-old ancient Caledonian pinewood (*P. sylvestris* var. *scotica*)] in the Cairngorm Mountains of the United Kingdom (UK). In-field hydraulic conductivity was calculated using a constant-head well permeameter. Median in-field hydraulic conductivity (mm h\(^{-1}\)) was approximately 15- and 97-fold greater in the 48- and 4,000-yr-old forests, respectively, than in the 6-yr-old forest. The 48- and 4,000-yr-old forests also had 67 and 246\% greater average total macroroots (> 1-mm diameter), respectively, as well as greater macroporosity (> 0.039-mm) compared to the 6-yr old forest. The soil of the 6-yr-old plantation had been disturbed prior to establishing the plantation. The previous forest that had existed before the 6-yr old forest was established had been cleared and the soil was disturbed by heavy machinery to construct drainage channels. Previous soil disturbance, in addition to a smaller number of macroroots and macropores, in the 6-yr-old forest may have explained the decreased hydraulic conductivity in the 6- compared to the 48- and 4,000-yr-old forests (Archer et al., 2015). Because hydraulic conductivity is directly affected by the infiltration rate of a soil, results of Archer et al. (2015) supported the notion that infiltration increases as forest age increases.

Hümann et al. (2011) studied the effects of forest age on runoff in southwest Germany. Rainfall simulations were used to calculate RCs for a recently established (1-yr-old) deciduous forest with a sandy-loam (Haplic Cambisol) surface texture and two deciduous forests, which were at least 30 years old and had clay-loam (Stagnic Cambisol) and silt-loam (Haplic Cambisol) surface textures. The RC from the recently established forest was 20\% compared to the more established forests, which had runoff coefficients ranging from 11.6 to 17.3\%. The greater RC in
the younger forest indicated that, as a forest ages, runoff decreases, thus infiltration increases (Hümann et al., 2011).

**Soil Aggregation**

Soil aggregates are defined as soil particles that are held together by organic matter, clay, or other cohesive materials (USDA-NRCS, 2014a). Consequently, aggregate stability is the degree to which soil aggregates can withstand disintegrating forces, including weathering, precipitation, and erosion (Kemper and Rosenau, 1986; Smith et al., 2014). Stable soil aggregates can be classified as macroaggregates (> 0.25 mm) or microaggregates (< 0.25 mm) (Oades, 1984). Soils typically have a hierarchical organization in which macroaggregates are formed through the cohesion of microaggregates (Tisdall and Oades, 1982; Puget et al., 2000). Generally, a greater age of macroaggregates than microaggregates in a soil is an indication of greater overall soil quality (Oades, 1984). Macroaggregates are essential for root growth, hydraulic conductivity, and the ability of the soil to resist erosion (USDA-NRCS, 2008). One method of measuring the size of aggregates is mean weight diameter (MWD), which is an index used to compare the size of water-stable aggregates between soils, taking into account the concentrations of varying aggregate size classes in a soil (Chaney and Swift, 1984). Aggregate stability, macroaggregation, and microaggregation are affected by physical and chemical properties of the soil, as well as landuse management practices (USDA-NRCS, 2008).

**Soil Property Effects on Soil Aggregation**

Many soil properties can affect soil aggregate stability, including SOM content, texture, fungal hyphae content, and root biomass (USDA-NRCS, 2008). The amount of organic matter in a soil greatly affects soil aggregate stability because organic matter functions as a binding agent
for soil aggregate formation (Tisdall and Oades, 1982; Puget et al., 2000). Generally, as SOM increases, soil aggregate stability also increases (Tisdall and Oades, 1982; Cambardella and Elliott, 1993; Puget, 2000). Organic matter assists in binding microaggregates together to form macroaggregates in the hierarchical scheme proposed by Tisdall and Oades (1982). Chaney and Swift (1984) measured the effects of SOM on aggregate stability in 26 British soils with varying textures used for agriculture. Results indicated that SOM significantly increased aggregate stability, as measured by MWD, indicating that SOM is a main factor involved in a soil’s structural stability (Chaney and Swift, 1984).

Soil texture can affect aggregate stability, as clay particles act as an adhesive to bind aggregates together (Oades, 1984). Generally, the greater the amount of clay, the greater the aggregate stability (Amézketa, 1999). Lado et al. (1992) measured aggregate stability in the 5- to 25-cm depth of six cultivated agricultural soils with clay contents ranging from 6 to 62% in Israel. On soils that were slowly wetted, MWD increased directly as clay content of the soil increased. As the clay content increased by 1%, the MWD increased by 0.02 mm. ($r^2 = 0.84$; Lado et al., 1992).

Roots and fungal hyphae, particularly mycorrhizal fungi, are major stabilizing factors of soil aggregates (Tisdall and Oades, 1982). Roots and fungal hyphae can bind microaggregates together to create macroaggregates (Wang et al., 2010) through the production of polysaccharides, which bind smaller aggregates together, creating larger aggregates (Tisdall and Oades, 1982). After death, fungal hyphae and roots can continue to stabilize soil aggregates during decomposition (Oades and Waters, 1991). Fungal hyphae and roots have also been shown to be the main binding agents for large macroaggregates ($> 2$-mm; Amézketa, 1999; Wang et al., 2010).
Grasses typically have high-density root systems that promote soil aggregation (Tisdall and Oades, 1982). Even after death, root systems still function to bind aggregates together as the roots decompose (Oades and Waters, 1991). Clarke et al. (1967) studied different types of pasture vegetation with varying root densities in loamy fine sand in South Australia. Aggregate stability in the top 8 cm of soil was correlated with the type of pasture vegetation that had the greatest root biomass. For the treatments in which grass with less root biomass (*Phalaris tuberosa* L.) was grown and no N was added, the water-stable macroaggregation (> 2 mm) ranged from 13 to 39%, while for the treatments in which grass with greater root biomass was grown (*Lolium rigidum* Gaud), the water-stable macroaggregation (> 2 mm) ranged from 25-52% (Clarke et al., 1967).

**Landuse Effects on Soil Aggregation**

Landuse management practices, including soil disturbance (i.e., tillage, vehicle traffic, and compaction), residue and vegetative cover, can affect soil aggregate stability (Six et al., 1998; Harper et al., 2008; DeFauw et al., 2014). Cultivation for agricultural purposes generally leads to decreased soil aggregate stability (Chaney and Swift, 1984; Cambardella and Elliott, 1993; Kasper et al., 2009; Smith et al., 2014). Cultivating the soil incorporates residues into the soil, allowing microorganisms to be in direct contact with the residue, which increases oxidation and decomposition rates of SOM (Wang et al., 2010; Smith et al., 2014). Cultivation also churns the soil, causing the subsoil to be brought to or near the soil surface where the sub-soil material is more susceptible to physical processes, such as slaking from precipitation or irrigation additions and freezing-thawing cycles, which may destroy soil macroaggregates (Beare et al., 1993; Paustian et al., 1997; Six et al., 1998). Cultivation causes fungal hyphae in the soil to be severed, which can decrease soil aggregate stability (Wang et al., 2010). Cultivation also alters
the size-class distribution of soil aggregates (Six et al., 1998), as cultivation has been shown to
decrease soil macroaggregate and increase microaggregate abundance (Tisdall and Oades, 1982;
Smith et al., 2014). Cultivation causes a decrease in macroaggregates by physically breaking
down macroaggregates into microaggregates (Smith et al., 2014).

Alternatively, NT agriculture has been shown to increase soil aggregate stability
compared to cultivated agriculture (Wood et al., 1991; Cambardella and Elliott, 1993; Wang et
al., 2010). No-tillage agricultural management practices often result in increased SOM, fungal
hyphae (Wang et al., 2010), and root biomass, which, in turn, can result in an overall increase in
soil aggregate stability and a greater proportion of macroaggregates compared to CT practices
(Tisdall and Oades, 1980; Six et al., 1998).

Franzluebbers (2002) evaluated the effects of landuse on infiltration, aggregate stability,
and soil organic carbon (SOC) on a Cecil sandy loam (fine, kaolinitic, thermic Typic
Kanhapludult) near Watkinsville, GA. Two landuses were evaluated: long-term CT agriculture
and long-term NT agriculture. The infiltration rate was measured on soil cores that were
collected from both landuses and left intact. On average, the infiltration rate (cm h\(^{-1}\)) was 227%
greater for intact soil under the NT agriculture than under cultivated agriculture. In addition, the
total SOC content was approximately double under NT than CT in the upper 12 cm of the soil.
Additional soil cores were collected, sieved to < 8 mm, and used to measure water-stable
macroaggregates and macroaggregate stability in the top 9 cm of soil. Water-stable
macroaggregate concentration (g g\(^{-1}\)) and macroaggregate stability [g (wet) g (dry)\(^{-1}\)] were
approximately 8.3 and 13% greater under NT than under CT, respectively (Franzluebbers, 2002).

Wang et al. (2010) evaluated the effects of cultivation on water-stable aggregates in the
top 15 cm of a sandy loam (Typic Endoaqepts) in Eastern China. Sites with cultivated wheat-
soybean (CWS) and NT wheat-soybean (NTWS) crop rotations were evaluated. The CWS sites had approximately 40% greater microaggregates (< 0.25 mm) compared to the NTWS sites. In addition, the NTWS sites had significantly greater fungal species richness and diversity in the 5- to 15-cm depth, suggesting that cultivated agriculture results in decreased fungal diversity and, therefore, decreased macroaggregation and increased microaggregation as a result of cultivation severing fungal hyphae (Wang et al., 2010).

Native prairies, which have never been subject to tillage or agricultural production, also tend to have greater aggregate stability than soils used for cultivated agricultural production (Cambardella and Elliott, 1993). Because the soil has never been disturbed by tillage, SOM, fungal hyphae, and soil biota have been allowed to increase in the soil, which, in turn, cause aggregation to increase (Chaney and Swift, 1984). Soils in native prairies have been shown to have greater macroporosity, which indicates more developed soil structure compared to soils of cultivated agroecosystems or contemporary grasslands that have been cultivated in the past (Bodhinayake and Si, 2004).

Martens (2000) compared soil aggregate stability of a native prairie to CT corn (Zea mays L.) and soybean agroecosystems in a Webster silty clay loam (fine-loamy, mixed, superactive, mesic Typic Endoaquolls) in Pocahontas County, IA. The percentage of water-stable macroaggregates (> 1 mm) in the top 15 cm of soil for the native prairie was three and 10 times greater than that in the corn and soybean agroecosystems, respectively. Alternatively, the native prairie had the lowest percentage of microaggregates (< 0.25 mm) in the top 15 cm of soil, where the percentage of water-stable microaggregates for the corn and soybean agroecosystems was four and five times, respectively, greater than that for the native prairie. This may be explained
by the fact that the SOC concentration in the top 15 cm of soil for the native prairie was 44 and 53\% greater than that in the corn and soybean agroecosystems, respectively (Martens, 2000).

Cambardella and Elliott (1993) measured soil aggregate stability and aggregate-size distribution in the top 20 cm of soil in native prairie and moldboard-plowed CT and NT agriculture on a Duroc loam (fine-silty, mixed, mesic, Pachic Haplustoll) in Sidney, NE. The soils samples were oven-dried before wet-sieving in order to simulate slaking. The native prairie had 130 and 36\% greater water-stable macroaggregates (> 0.25 mm) than the CT or NT agroecosystems, respectively. In addition, the CT agroecosystem had approximately 27 and 73\% greater water-stable microaggregates (< 0.25 mm) than the NT agroecosystem and native prairie, respectively. Conventional tillage caused the structural stability of the soil to decrease and caused greater destruction of macroaggregates compared to the NT and native prairie landuses. Less disturbance of the soil under native prairie allowed the soil’s structural stability and macroaggregation to increase compared to both agroecosystems (Cambardella and Elliott, 1993).

Some studies have shown that converting land formerly used for agricultural production to a native-species grassland, such as a CRP, can lead to increased aggregate stability (Tisdall and Oades, 1982). As the land remains undisturbed over a period of years, SOM begins to accumulate, the aggregate stability of the soil strengthens, and the number of macroaggregates in the soil increases (Elliott, 1986). Establishing grasslands also allows earthworm channels to form and persist, which can increase soil aggregation (Lindstrom et al., 1994). Over time, the grassland restoration begins to exhibit characteristics more similar to a native prairie than a cultivated agroecosystem (Jastrow, 1996; Bodhinayake and Si, 2004).

As the time since the establishment of a grassland increases, soil aggregate stability increases (Elliott, 1986). This was shown by Jastrow (1996), who measured aggregate stability in
the top 10 cm of soil in a native prairie and a chronosequence of native-vegetation grasslands that had been established 1, 4, 7, 10, and 13 years previously. Research sites were located near Chicago, IL and the soil series were a Mundelein silt loam (fine-silty, mixed, super-active, mesic Aquic Argiudolls), Drummer silty clay loam (fine-silty, mixed, super-active, mesic Typic Endoaquolls), and Wauconda silt loam (fine-silty, mixed, super-active, mesic Udollic Endoaqualfs). As time since cultivation increased, the percentage of macroaggregates (> 212 \( \mu m \)) increased exponentially (\( r^2 = 0.99 \)), indicating a strong, direct relationship between the two variables (Jastrow, 1996). Jastrow (1996) attributed the increase in soil aggregate stability to an increase in SOM in the grassland over time.

Huang et al. (2002) compared soil aggregate stability of CRP grasslands and cultivated agriculture in a Harney silt loam (fine, montmorillonitic, mesic Typic Argiustolls) in central Kansas. The CRP-grassland had been seeded with native grasses at least 10 years prior to the study. The cultivated agroecosystem was planted and cropped to a wheat-corn rotation five years prior to the study. The CRP grassland had approximately 50\% greater macropores (> 0.03-mm) in the top 10 cm of the soil than in the cultivated agroecosystem.

Zheng et al. (2004) compared soil aggregate stability in approximately the top 10 cm of soil and soil loss in an annually hayed CRP with two types of agroecosystems (i.e., CT and NT) in Morton County, North Dakota. The soils at each of the three landuse sites were a Williams loam (fine-loamy, mixed, super-active, frigid, Typic Argiustoll). The CRP grassland location had been previously under agriculture for at least 10 years and then converted to annually hayed CRP for at least five years. The annually hayed CRP-grassland contained approximately 9.1 and 6.5\% greater water-stable macroaggregates (> 2 mm) than the cultivated and NT agroecosystems, respectively. Soil aggregates in the annually hayed CRP grassland also had approximately 7 and
5% greater MWD than the cultivated and NT agroecosystems, respectively. Alternatively, the cultivated agroecosystem contained 17 and 40% greater water-stable microaggregates (<0.212 mm) than the NT agroecosystem and CRP grassland, respectively (Zheng et al., 2004). Increased microaggregation in the cultivated agroecosystem demonstrated how tillage can break down macroaggregates in the soil to form microaggregates. Increased microaggregation can cause the structural stability of a soil to decrease and the potential for soil erosion to increase (Zheng et al., 2004). In fact, soil erosion was 500% greater in the cultivated compared to the NT agroecosystem and annually hayed CRP grassland (Zheng et al., 2004).

Six et al. (2000b) studied the effects of landuse (i.e., CT and NT agriculture and deciduous forest) on aggregate distribution in a Wooster silt loam (fine-loamy, mixed, active, mesic Oxyaquic Fragiudalf) in Wooster, OH. The proportion of macroaggregates in the deciduous forest was approximately five and seven times greater than under NT and CT agriculture, respectively. The CT agriculture had the greatest proportion of microaggregates, which was approximately 33% greater than under the deciduous forest, and approximately 67% greater than under NT agriculture. Results suggested that CT agriculture leads to a decreased proportion of macroaggregates and an increased proportion of microaggregates compared to landuses that are less disturbed (i.e., deciduous forest and NT agriculture; Six et al., 2000b).

Soil aggregate stability is also influenced by the type of vegetation present. For example, studies have shown that deciduous forests can have greater soil aggregation than coniferous forests (Graham et al., 1995; Fang et al., 2015). One explanation for this involves the relative abundance of earthworms in coniferous and deciduous forests. The topsoil of deciduous forests generally have greater pHs than in coniferous forests (Graham et al., 1995; Ste-Marie and Paré, 1999), and earthworms are typically present in greater abundance in soils with more alkaline pHs.
Consequently, there are typically less earthworms in coniferous forests than deciduous forests (Terhivuo, 1989; Graham et al., 2005). For example, Terhivuo (1989) analyzed earthworm diversity and biomass in coniferous and deciduous forests in South Finland. A total of five earthworm species were present in the coniferous forest and earthworm biomass ranged from 4 - 11 g m$^{-2}$, while a total of 11 earthworm species were present in the deciduous forests and earthworm biomass ranged from 65 - 131 g m$^{-2}$. The activity of earthworms in soil has been shown to increase soil aggregate stability (Ketterings et al., 1997; Amézketa, 1999). This may explain why previous studies have shown greater soil aggregate stability in deciduous forests compared to coniferous forests. In addition, coniferous forests have been shown to have less SOM and microbial biomass (bacteria and fungi) than deciduous forests (Frey et al., 2004), which are major contributing factors to soil aggregation.

Graham et al. (1995) compared soil aggregate stability of coniferous and deciduous forests in a fine sandy loam in the San Gabriel Mountains near Los Angeles, CA. The deciduous and coniferous forests consisted of scrub oak (*Quercus dumosa* Nutt.) and Coulter pine (*Pinus coulteri* B. Don), respectively. The A horizon of the deciduous forest (0- to 6-cm depth) had approximately 90% water-stable aggregates and consisted almost entirely of worm casts, whereas the coniferous forest (0- to 1-cm depth) had only 78% water-stable aggregates and contained no worm casts (Graham et al., 1995).

Similar results were achieved by Fang et al. (2015), who analyzed soil aggregate stability in the top 15 cm of soil in coniferous and deciduous forests with clayey soils in Taihe County in Southern China. The deciduous forest had approximately 58% greater macroaggregates than in the coniferous forest. Furthermore, the MWD in the deciduous forest was 38% greater than that
in the coniferous forest. The coniferous forest, however, had approximately 33% greater microaggregates (< 0.25-mm) than the deciduous forest (Fang et al., 2015).

Aggregate Stability Effects on Infiltration

Soil aggregate stability is one of the most important factors that controls surface infiltration (Kemper and Rosenau, 1986; Graham et al., 1995). As soil aggregate stability decreases, soil aggregates are more likely to disintegrate, or slake, which will result in soil filling pore spaces and blocking water from infiltrating through the pores (Harper et al., 2008). Therefore, increased soil aggregate stability generally leads to increased soil infiltration (Franzluebbers, 2002; Marquez et al., 2004; USDA-NRCS, 2014b).

Another reason that aggregate stability and infiltration are positively correlated is the fact that both parameters tend to increase as the SOM content increases (Tisdall and Oades, 1982, Stone and Schlegal, 2010). Franzluebbers (2002) measured infiltration, aggregate stability, and SOC in two different landuses (i.e., long-term CT and NT agriculture) on a Cecil sandy loam (fine, kaolinitic, thermic Typic Kanhapludult) near Watkinsville, GA. The NT had the greatest soil aggregate stability (i.e., water-stable macroaggregates and macroaggregate stability) and also had the greatest infiltration. On average, the infiltration rate (cm h\(^{-1}\)) was 227% greater for intact soil under the NT than under CT agriculture. In addition, water-stable macroaggregate concentration (g g\(^{-1}\)) and macroaggregate stability [g (wet) g (dry)\(^{-1}\)] in the top 9 cm of soil were approximately 8.3 and 13% greater under NT than under CT, respectively (Franzluebbers, 2002). The cause was attributed to increased SOM because NT, which had greater soil aggregate stability and infiltration, also had greater SOC than CT. The total SOC content in the top 12 cm
of soil under NT was approximately double that under CT. Results supported the contention that soil aggregate stability and infiltration are positively correlated (Franzluebbers, 2002).

Similar results were obtained by Le Bissonnais and Arrouays (1997) who measured infiltration, aggregate stability, and SOC in loamy (Vermic Haplumbrepts), alluvial soils in the French Pyrenean piedmont in southwest France. The study included 10 agricultural sites that had been cultivated in the past for various durations ranging from 8 to 100 years. Soil samples were slowly wetted before aggregate breakdown into stable size fractions and MWD was measured. Cumulative infiltration was measured using a 2-hr rainfall simulation. Mean weight diameter and cumulative infiltration were strongly correlated ($r^2 = 0.93$; Le Bissonnais and Arrouays, 1997). In fact, aggregate stability and infiltration rate were so well correlated, Le Bissonnais and Arrouays (1997) suggested using soil aggregate stability as a predictor for infiltration, particularly because soil aggregate stability was much less difficult to measure than infiltration.

Aggregate stability is one of the most sensitive soil properties for assessing the impacts of landuse management practices as well as improvements in overall soil quality (Blanco and Lal, 2010; Bodhinayake and Si, 2004; Stone and Schlegel, 2010). Soil aggregate stability was reported to be correlated with soil hydraulic properties including macroporosity and saturated hydraulic conductivity, which controls overall infiltration rate (Bodhinayake and Si, 2004). Mean weight diameter was also shown to be highly correlated with infiltration rate (Stone and Schlegel, 2010). Soil aggregate stability has been described as less difficult to measure than infiltration rate and has the benefit of being able to be measured off-site in a laboratory setting (Le Bissonnais and Arrouays, 1997). Therefore, soil aggregate stability may be the most effective soil property for predicting infiltration rate (Bodhinayake and Si, 2004).
Justification

As an alternative to severely limiting groundwater withdrawals for agricultural irrigation, a potentially effective way to increase Alluvial Aquifer recharge would be to identify landuse management practices and soil surface properties that increase infiltration and improve infiltration-related properties to allow more rainfall and/or irrigation water to return to the Alluvial Aquifer. By increasing infiltration, runoff would decrease and the amount of water that recharges the Alluvial Aquifer could increase, thereby decreasing the threat of soil erosion, sediment loading of surface waters, and groundwater depletion and increasing the future sustainability of the soil and water resources in the LMRV.

Objectives and Testable Hypotheses

The main objective of this study is to evaluate the effects of landuse [i.e., native prairie, Conservation Reserve Program (CRP) grassland, coniferous forest, deciduous forest, cultivated agriculture, and no-tillage agriculture] on surface water infiltration and infiltration-related properties on silt-loam loessial and alluvial soils in the Delta region of eastern Arkansas in the LMRV. The secondary objective of this study is to evaluate the correlation among various soil surface physical and chemical properties and infiltration-related properties.

It is expected that landuse in the Delta region of eastern Arkansas in the LMRV affects infiltration and infiltration-related properties. More specifically, it is hypothesized that landuse affects overall infiltration rate, as measured over a 20-minute time period with a double-ring infiltrometer, and the parameters (i.e., slope and y-intercept) characterizing the linear relationship between the natural logarithm of the infiltration and time. It was also hypothesized that soil aggregate stability in the top 10 cm differs among landuses. Finally, it was hypothesized
that various soil surface physical and chemical properties will be correlated with infiltration-related properties.
Literature Cited


Chapter 2

Landuse and Soil Property Effects on Infiltration in the Lower Mississippi River Valley
Abstract

Unsustainable, agriculturally related practices of water usage have caused increasing groundwater depletion in the Alluvial Aquifer in the Lower Mississippi River Valley (LMRV) of eastern Arkansas. To avoid further depletion, one method of decreasing drawdown of the Alluvial Aquifer would be to adopt agricultural management practices that increase surface infiltration, which would increase the amount of water that could potentially recharge the aquifer. The objective of this study was to evaluate the effects of landuse on surface water infiltration and infiltration-related properties in fine-textured, loessial and alluvial soils in the Delta region of eastern Arkansas in the LMRV. Research was conducted in six current major landuses [i.e., native prairie, deciduous forest, coniferous forest, Conservation Reserve Program (CRP) grassland, conventional-tillage (CT) agriculture, and no-tillage (NT) agriculture]. Soil particle-size analyses determined that 94% of all sites had a silt-loam surface texture. The remaining 6% of sites were identified as silt. Infiltration measurements were conducted over 20 minutes using a double-ring infiltrometer and the overall infiltration rate for the 20-minute time period was calculated for each landuse. The overall infiltration rate for the deciduous forest (1.17 mm min$^{-1}$) was 6.7 times greater than that for the other five landuses, which did not differ and averaged 0.17 mm min$^{-1}$. Overall infiltration rate was positively correlated ($P < 0.05$) with soil organic matter (SOM), total C, and total N contents and C:SOM ratio, while negatively correlated ($P < 0.05$) with estimated bulk density and extractable soil Na and Mg contents in the top 10 cm. The natural logarithm of the infiltration rate was calculated and linearly regressed against the mid-point of measurement time, which showed that the slope parameter was unaffected ($P > 0.05$), while the intercept parameter differed ($P < 0.05$) among landuses. The intercept for the deciduous forest was more than 75% greater than the landuse with the next largest intercept, while the coniferous forest, native prairie, and CRP grassland had similar ($P >$
0.05) intercepts to each other. The CRP and CT and NT agroecosystems had similar ($P > 0.05$) intercepts to each other, but smaller ($P < 0.05$) intercepts than the other three landuses. Results of this study demonstrated that surface water infiltration and infiltration-related properties, as well as select near-surface soil physical and chemical properties, differed among major landuses in fine-textured, loessial and alluvial soils in the LMRV Delta region of eastern Arkansas. Restoration of highly erodible agricultural lands in the LMRV should consider reforestation activities with deciduous species to contribute to improved infiltration capacity that can contribute to potentially greater groundwater recharge.
Introduction

The Lower Mississippi River Valley (LMRV), which includes parts of Arkansas, Louisiana, Mississippi, Missouri, Kentucky, and Tennessee, is a sub-region of the larger Mississippi River Valley watershed (Rudis and Birdsey, 1986). Prior to European settlement, the LMRV was dominated by native forests and grasslands (Oswalt, 2013). The landscape of the LMRV has since been altered extensively by deforestation due to agricultural expansion (Oswalt, 2013). Currently, cultivated agriculture constitutes 33% of the total non-water land area in the LMRV (USDA-NRCS, 2013).

The Alluvial Aquifer underlays approximately 80,000 km² of land surface in the LMRV region and is used extensively for agricultural irrigation purposes (USGS, 2002). Approximately 44% of cropland in the LMRV is irrigated (USDA-NRCS, 2013). The two most produced crops in eastern Arkansas, specifically, are rice (Oryza sativa L.) and soybean (Glycine max L.) and both require large amounts of irrigation water for optimum productivity (USGS, 2002; Kebede et al., 2014). However, the Alluvial Aquifer is currently being overdrawn at unsustainable rates, such that the amount of water that is annually withdrawn from the Alluvial Aquifer is approximately 226% of the estimated sustainable yield (ANRC, 2017). These extensive withdrawals have lowered aquifer levels and depleted groundwater reserves (USGS, 2002; Verkler et al., 2008). The average depth of the Alluvial Aquifer in Arkansas, which is typically between 15 and 45 m (USGS, 2002), has declined by an average of 0.52 m in the last decade (ANRC, 2017). Additionally, cones of depression have formed within the Alluvial Aquifer due to overuse (USGS, 2002; ANRC, 2017). In areas where cones of depression have occurred, sustainable withdrawal rates have decreased to less than 0.15 Mgal d⁻¹, which is insufficient for supporting current agriculture in the region (USGS, 2002). Consequently, the amount of water
that is being withdrawn from the Alluvial Aquifer must be greatly reduced in the near future in order to avoid completely irreversible groundwater depletion. If demand for the use of groundwater is not likely to decrease, then increasing the recharge rate must occur.

Groundwater can be naturally recharged in the LMRV through infiltration of rainfall and/or irrigation water into soil, where the excess gravitational water can eventually flow vertically downward to the aquifer (Alley, 2009). The ability of a soil to facilitate water infiltration is based primarily upon the physical and chemical properties associated with the soil surface, which are strongly influenced by landuse management practices (Harper et al., 2008). After reaching the soil surface, rainfall and/or irrigation water can either flow over the surface as runoff or can infiltrate into the soil in a process often referred to as runoff-infiltration partitioning (Horton, 1933; Harper et al., 2008).

Surface water infiltration can be quantified multiple ways, including cumulative infiltration and infiltration rate. Cumulative infiltration is the total amount of water that infiltrates over some period of time (Chu, 1978). Infiltration rate is the speed at which water enters the soil (USDA-NRCS, 2014). Furthermore, the process of infiltration can be divided into initial and steady-state infiltration phases (Wang et al., 2009). Initial infiltration occurs during a period of rapid infiltration as water begins to enter an unsaturated soil, whereas steady-state infiltration occurs as the soil approaches saturation and infiltration reaches a relatively constant rate (Wang et al., 2009).

Landuse and management practices can also affect infiltration through soil disturbance (i.e., tillage and/or surface compaction) and the maintenance of land cover (Bhattacharyya et al., 2008; Harper et al., 2008; DeFauw et al., 2014). Studies have shown that management practices that cause soil surface disturbance (i.e., tillage, compaction, deforestation, burning, mowing)
generally decrease infiltration and increase runoff (Andraski et al., 1985; Meek et al., 1992; Azooz and Arshad, 1996; Bharati et al., 2002; Bodhinayake and Si, 2004; Bai et al., 2009; Hümann et al., 2011; Archer et al., 2015). Repeated soil disturbance can cause increased bulk density if compaction occurs (Meek et al., 1992; Bai et al., 2009) as well as decreased soil organic matter (SOM) content from increased oxidation (Martens, 2000; Franzluebbers, 2002), often resulting in decreased infiltration (Meek et al., 1992; Azooz and Arshad, 1996; Bharati et al., 2002; Franzluebbers, 2002).

Soil disturbance and loss of SOM have also been shown to cause decreased aggregate stability (Six et al., 2000B). Aggregate stability is the degree to which soil aggregates can withstand disintegrating forces (Kemper and Rosenau, 1986), and is an important indicator of soil structural stability and overall soil quality (Six et al., 2000B). Generally, decreased aggregate stability causes decreased macroaggregation (> 0.25 mm diam.), macroporosity (> 0.50 mm diam.), and soil pore continuity (Tisdall and Oades, 1982; Six et al., 2000A; Bodhinayake and Si, 2004; Eusufzai and Fujii, 2012), all of which can ultimately lead to decreased infiltration (Le Bissonnais and Arrouays, 1997; Franzluebbers, 2002; Wuest et al., 2005; Stone and Schlegel, 2010; Alaoui et al., 2011).

Tillage is a common landuse management practice that disturbs the soil surface (Franzluebbers, 2002). Tillage causes SOM to be exposed and decompose at an above-average rate, which can ultimately lead to decreased infiltration (Azooz and Arshad, 1996; Franzluebbers, 2002). Furthermore, the soil surface is typically left with little residue cover after tillage, which can facilitate the formation of soil surface crusts that can also impede infiltration (Andraski et al., 1985; USDA-NRCS, 2014).
No-tillage (NT) agricultural management practices have been shown to decrease runoff (Andraski et al., 1985) and increase infiltration (Meek et al., 1992; Azooz and Arshad, 1996; Franzluebbers, 2002; Stone and Schlegel, 2010) compared to conventional-tillage (CT) management practices. Earthworm and root channels, which function as natural conduits for water infiltration, are preserved in NT agroecosystems, whereas earthworm and root channels are frequently destroyed by tillage (Azooz and Arshad, 1996; USDA-NRCS, 2014). No-tillage agriculture also maintains greater amounts of surface residue cover than CT agriculture (Andraski et al., 1985), which has been shown to increase infiltration (Mukhtar et al., 1985; Harper et al., 2008). Azooz and Arshad (1996) reported NT practices improved soil pore continuity, hydraulic conductivity, and infiltration, compared to CT practices in a Donnelly silt-loam and sandy-loam (Typic Haplocrypts) soils in Alberta, Canada.

In contrast to agriculturally managed soils, native prairies, which have never been subject to tillage or agricultural production, generally have greater infiltration rates (Bodhinayake and Si, 2004) and less runoff (Harper et al., 2008) than landuses with soils that are used for agricultural production. Increased infiltration rates into native prairies result from typically greater aggregate stability, SOM content, hydraulic conductivity, and earthworm activity compared to cultivated soils (Cambardella and Elliott, 1993; Martens, 2000; Bodhinayake and Si, 2004). However, many native prairies are still subject to multiple management practices, including burning, grazing, and mowing, which could negatively impact soil structure, infiltration rate, and overall soil quality (Weaver and Fitzpatrick, 1934; Weaver and Rowland, 1952; Davison and Kindscher, 1999).

Some agricultural lands have various limitations that restrict their productivity, such as poor drainage, steep slopes, or shallow bedrock. In addition, approximately 12% of cropland is
designated as highly erodible in the LMRV (USDA-NRCS, 2013). The conversion of highly erodible agricultural land to more natural, perennial vegetation, including grasses or trees, has been shown to increase infiltration and soil quality over time, as well as decrease soil erosion (Lindstrom et al., 1994; Gilley and Doran, 1997; Bharati et al., 2002; Huang et al., 2002; Sanderson et al., 2012). Bharati et al. (2002) reported that the conversion of formerly cultivated cropland to both deciduous forest and grassland resulted in a five-fold increase in cumulative infiltration and overall soil quality compared to cultivated agricultural sites in just six years in Coland loam and sandy-loam (Cumulic Endoaquolls) soils in central Iowa.

The Conservation Reserve Program (CRP), created by the Farm Service Agency in 1985, sought to convert highly erodible agricultural land to more natural, perennial vegetation in an effort to decrease erosion and improve soil quality (Lindstrom et al., 1994; Huang et al., 2002). By 2013, approximately 400,000 hectares of land were enrolled in the CRP in the LMRV (USDA-NRCS, 2013). Perennial grasses have extensive roots systems, which improve soil aggregation (Tisdall and Oades, 1982) and infiltration (Sanderson et al., 2012). In addition, perennial grasses decompose and annually provide organic matter to the soil, which decreases bulk density and increases infiltration (Elliott, 1986). In general, as the time since establishment of a grassland increases, soil aggregation and infiltration will increase (Elliott, 1986; Jastrow, 1996; Bharati et al., 2002). Bodhinayake and Si (2004) reported the average estimated saturated hydraulic conductivity ($K_{sat}$) of a native prairie and perennial grassland, which had been established 20 years previously, were four- and six-fold greater, respectively, than that of a CT agroecosystem in clay-loam (Typic Haplustolls) soils in Saskatchewan in central Canada.

Similar to grasslands, forests have extensive root systems which can facilitate the infiltration of water (Alaoui et al., 2011). However, the diameter of tree roots are typically much
larger than the roots of grasses, which could allow for greater preferential flow (Aubertin, 1971). Preferential flow can occur as water is conducted along the edges of roots or in macropores created from old, decayed tree roots (Aubertin, 1971; Sidle et al., 2001). Typically, as the time since establishment of a forest increases, the root system and macropore network will become more extensive (Archer et al., 2015). Greater macroporosity allows for more efficient transmission of water through the soil matrix, thereby increasing hydraulic conductivity and infiltration (Aubertin, 1971; Watson and Luxmoore, 1986; Archer et al., 2015). Alaoui et al. (2011) showed that forest soils contained more macropores that were able to preferentially transport water at a faster rate than grassland soils in the 5- to 15-cm depth of silt-loam and clay-loam soils in western Switzerland. Additionally, tree roots can loosen compacted soil, which in turn decreases soil bulk density (Hümann et al., 2011). Liao et al. (2006) showed that original grasslands that had transitioned into forests had decreased bulk density, increased SOM, and more stable aggregation over time in the 0- to 15-cm depth of sandy-loam and clay-loam soils in the Rio Grande Plains of Texas.

Other management practices that require vehicle traffic, which can disturb the soil and cause compaction, have been shown to decrease infiltration (Meek et al., 1992; Bai et al., 2009; USDA-NRCS, 2014). Meek et al. (1992) showed that the infiltration rate into non-trafficked soil was approximately 73% greater than that into trafficked soil in NT cotton (Gossypium spp.) agroecosystems on Wasco sandy-loam (Typic Torriorthents) soils in the western United States. Bai et al. (2009) reported that controlled-traffic agriculture exhibited decreased soil bulk density in the top 40 cm and a 45% increase in cumulative infiltration than CT agriculture in silt-loam soils in the Loess Plateau in eastern China.
Infiltration is also influenced by specific land cover type. For example, studies have shown that deciduous forests can have greater infiltration rates than coniferous forests (Wang et al., 2009; Butzen et al., 2015). Coniferous forests typically have greater soil water repellency (SWR) than deciduous forests (Woche et al., 2005; Doerr et al., 2006), which likely contributes to decreased infiltration. Additionally, deciduous forests are typically less acidic than coniferous forests (Graham et al., 1995; Ste-Marie and Paré, 1999), which has been shown to lead to a greater abundance of earthworms (Reich et al., 2005). Increased earthworm activity generally improves soil aggregation and macroporosity (Ketterings et al., 1997; Amézketa, 1999). Wang et al. (2009) reported the initial and stable infiltration rates and cumulative infiltration into a deciduous forest to be 62, 85, and 160% greater, respectively, compared to a coniferous forest in residual and alluvial soils in the Simian Mountains of southwestern China.

The Delta region of eastern Arkansas in the LMRV is highly agriculturally productive, partly due to the ability to irrigate crops. However, extensive, widespread irrigation has been a main cause for the groundwater depletion issues the region faces (USGS, 2002). Apart from severely limiting groundwater withdrawals for agricultural irrigation, a potentially effective way to increase Alluvial Aquifer recharge would be to identify landuse management practices and soil surface properties that reduce runoff, improve infiltration-related soil-surface properties to allow more rainfall and/or irrigation water to infiltrate to potentially recharge the Alluvial Aquifer. By increasing infiltration, the threat of soil erosion and sediment loading of nearby surface waters would also decrease, thereby increasing the future sustainability of the soil and water resources in the LMRV. Therefore, the main objective of this field study was to evaluate the effects of major, current landuses [i.e., native prairie, deciduous forest, coniferous forest, CRP grassland, and CT and NT agriculture] on surface water infiltration and infiltration-related,
near-surface soil properties of fine-textured, loessial and alluvial soils in the Delta region of eastern Arkansas in the LMRV. The secondary objective of this study was to evaluate the relationship among various soil surface physical and chemical properties and infiltration-related properties.

It was hypothesized that infiltration rate, as measured over a 20-minute time period with a double-ring infiltrometer, would differ among current, common landuses in the Delta region of eastern Arkansas in the LMRV. More specifically, it was hypothesized that the ranking of landuses and their associated overall infiltration rates would follow the intensity of soil disturbance caused by the various landuse management practices (i.e., tillage), such that the less intensely managed, or more naturally maintained, the landuse, the greater the associated infiltration rate. In particular, it was hypothesized that the native prairie landuse would exhibit the greatest infiltration rate because uncultivated grassland sites would, collectively, be the most naturally maintained. In addition, it was hypothesized that the CT agroecosystem would exhibit the slowest infiltration rate because annually-tilled soils would be subjected to the greatest degree of soil disturbance. Overall, it was hypothesized that, based on the intensity of landuse management practices which disturb the soil system, landuses and their associated infiltration rates would rank according to the following sequence: native prairie > deciduous forest > coniferous forest > CRP grassland > NT agriculture > CT agriculture. It was also hypothesized that landuse would affect the slope and intercept parameters characterizing the linear relationship between the natural logarithm (LN) of the infiltration rate and the mid-point of time over a 20-minute measurement interval. Finally, it was hypothesized that soil hydraulic properties [i.e., overall infiltration rate, slope and intercept parameters characterizing the linear relationship between the LN of the infiltration rate and the mid-point of time, and estimated $K_{sat}$] would be
correlated with various measured soil physical and chemical properties [i.e., antecedent soil moisture content (ASMC) in the top 6 cm and sand, silt, and clay concentrations, SOM content, estimated bulk density, pH, EC, extractable nutrient (P, K, Ca, Mg, Na, and Fe) contents, total C (TC) and N (TN) contents, and C:N, N:SOM, and C:SOM ratios in the top 10 cm.]

Materials and Methods

Site Descriptions

Research was conducted between Fall 2015 and Spring 2017 in six different landuses [i.e., native prairie, deciduous forest, coniferous forest, CRP grassland, and CT and NT agriculture] on fine-textured, loessial and alluvial soils in the LMRV’s Delta region of eastern Arkansas. These six landuses represent current, major landuses present throughout the Delta region of eastern Arkansas (USDA-NRCS, 2013).

This study included three native prairies (i.e., Kenneth Gray Prairie, Seidenstricker Prairie, and Roth Prairie; Table 1; Figure 1). These native tallgrass prairie sites are some of the few remnants of the Grand Prairie in eastern Arkansas that have been completely preserved from agricultural influence and soil disturbance, with the exception of periodic burning and occasional vehicle traffic (Brye and Pirani, 2005). The Kenneth Gray Prairie is located northeast of Stuttgart, AR (34°39'15" N, 91°24'47" W). At the time infiltration measurements were conducted, which are described in detail below, the prairie had been burned within the previous two weeks, and no vegetation, live or dead, was present. However, under normal vegetative conditions, major species present at the Gray Prairie are native tallgrasses including big bluestem (Andropogon gerardi), switchgrass (Panicum virgatum), Indiangrass (Sorghastrum nutans), and many forbs including purple coneflower [Echinacea purpurea (L.) Moench], black-eyed susan (Rudbeckia hirta L.), and goldenrod (Solidago spp.). A single soil series (Stuttgart silt loam; fine,
smectitic, thermic Albaquultic Hapludalfs; USDA-NRCS, 2017), which is in a udic soil moisture regime, is mapped throughout the Gray Prairie.

The Seidenstricker Prairie is located north of Stuttgart, AR (34°43'44" N, 91°33'17" W) and had been burned within six months prior to the time infiltration measurements were conducted. Similar to the Gray Prairie, major vegetative species present included big bluestem, switchgrass, Indiangrass, and many forbs including purple coneflower, black-eyed susan, and goldenrod. A single soil series (Dewitt silt loam; fine, smectitic, thermic Typic Albaqualfs; USDA-NRCS, 2017), which is in an aquic soil moisture regime, is mapped throughout the Seidenstricker Prairie.

The Roth Prairie is located southwest of Stuttgart, AR (34°27'15" N, 91°34'37" W). The major species present included little bluestem (Schizachyrium scoparium [Michx.] Nash), big bluestem, switchgrass, Indiangrass, and many forbs including purple coneflower, black-eyed susan, and goldenrod. In contrast to the Gray and Seidenstricker Prairies, two soil series (i.e., Stuttgart silt loam and Ethel silt loam) are mapped throughout the Roth Prairie. The Ethel soil series (fine-silty, mixed, active, thermic Typic Glossaqualfs) is located in an aquic soil moisture regime (USDA-NRCS, 2017). Natural prairie mounds are present at all three native prairie sites and are further evidence that the prairies have never been cultivated and that the soil has remained in its natural, undisturbed state (ANHC, 2013).

This study included four CRP-managed grasslands located at University of Arkansas System Division of Agriculture Pine Tree Research Station (PTRS) near Colt, AR (35°7'10.54" N, 90°45'51.56" W; Table 1; Figure 2). Major grass species present included big bluestem, switchgrass, and Indiangrass. Each CRP grassland was previously used for decades for cultivated agriculture; however, the areas were only marginally agriculturally productive for various
reasons and have subsequently been restored to managed grassland in order to increase soil quality. The CRP grasslands have not been cultivated for at least 10 years and have been only minimally disturbed by periodic mowing and removal of above-ground biomass. Three different soil series [Calloway silt loam (fine-silty, mixed, active, thermic Aquic Fraglossudalfs), Zachary silt loam (fine-silty, mixed, active, thermic Typic Albaqualfs), and Henry silt loam (coarse-silty, mixed, active, thermic Typic Fragiaqualfs); USDA-NRCS, 2017] are mapped throughout the four CRP grasslands. The Calloway silt loam is in a udic, while the Zachary and Henry silt loams are in an aquic soil moisture regime.

Four deciduous forest sites were included in this study (Table 1). Two different deciduous forest sites were located at PTRS (Figure 2). The major tree species present were oak (Quercus spp.), hickory (Carya spp.), gum (Eucalyptus spp.), and dogwood (Cornus spp.). It is likely that the deciduous forest areas had been cleared and used for cultivated agriculture in the past. However, based on the size of the trees (i.e., approximately 20 to 30 m tall), trees naturally regenerated and have not been disturbed for at least 30 years. Calloway silt loam was mapped at one site, while Calhoun silt loam (fine-silty, mixed, active, thermic Typic Glossaqualfs), which is in an aquic soil moisture regime (USDA-NRCS, 2017), was mapped at the second deciduous forest site at PTRS. The other two deciduous forest parcels were at the University of Arkansas System Division of Agriculture Lon Mann Cotton Research Station (LMCRS) near Marianna, AR (34°44′2.26″ N, 90°45′51.56″ W). One deciduous forest site at LMCRS was a historically managed pecan (Carya illinoinensis) grove on a Calloway silt-loam soil but has not been actively managed for pecan production for at least 15 years, other than periodic annual mowing for aesthetic purposes. The second deciduous forest site at LMCRS was a well-established deciduous forest stand on a Memphis silt loam (fine-silty, mixed, active, thermic Typic
Hapludalfs), which is in a udic soil moisture regime (USDA-NRCS, 2017), with growth indicating multiple decades of undisturbed development. Similar to the PTRS deciduous forest sites, the major tree species present were oak, hickory, gum, and dogwood. It is likely that the area had been cleared and used for cultivated agriculture in the past as well; however, based on the size of the trees (i.e., approximately 15 to 20 m tall), the forest parcel has been allowed to naturally regenerate for at least 40 years.

Four coniferous forest plantation sites at PTRS were included in this study (Table 1; Figure 2). Each coniferous forest site had previously been cleared and used for cultivated agriculture in the past. One of the plantation sites was planted to loblolly pine (*Pinus taeda*) at least 25 years ago, with an approximate row spacing of 3 m and an approximate tree spacing of 3 m. The other three plantation sites were planted to loblolly pine, with an approximate row spacing of 3 m and an approximate tree spacing of 3 m, at least 12 years ago. As of March 2016, the loblolly pine trees in the three young plantations varied in height between 10 and 15 m tall, while those in the older plantation varied in height from 20 to 25 m tall. The three young plantations had a surface pine needle residue layer that was approximately 3-cm thick, while the surface pine needle residue layer of the older plantation was approximately 0.5-cm thick. Two soil series were mapped among the three young plantations, Calloway silt loam and Loring silt loam (fine-silty, mixed, active, thermic Oxyaquic Fragiuudalfs), which is present in a udic soil moisture regime (USDA-NRCS, 2017), while Calloway silt loam is mapped throughout the older plantation.

A total of 11 CT agricultural sites were included in this study (Table 1). Two CT agricultural sites were located at PTRS in two different soil series (i.e., Calloway and Calhoun silt loams; Figure 2). The PTRS CT agricultural sites were fallow at the time of sampling, but
had been managed in a rice-soybean rotation for at least the past five years. A third CT agricultural site on a Memphis silt loam was located at LMCRS adjacent to the one deciduous forest site (Figure 3), which was also fallow at the time of sampling, but had been cropped to monoculture soybean for the past several years.

The remaining eight CT agricultural sites, as well as all eight NT agricultural sites, were 3 m wide by 6.1 m long replicated plots as part of a long-term, wheat (*Triticum aestivum*)-soybean, double-crop production system field study that was initiated in 2001 at LMCRS on a Calloway silt loam (Cordell et al., 2006; Figure 3). This field study was comprised of eight CT and eight NT treatments in a factorial combination with a wheat-residue-level (i.e., low and high achieved with differential N fertilization), residue burning (i.e., burning and non-burning), and irrigation (i.e., irrigated and dryland) treatments. The tillage, residue-level, and burn field treatments have been in place since 2002, while the irrigation treatment has been in place since 2005. In total, there were 48 plots, with three replications of each tillage-residue-level-burn-irrigation treatment combination. Each set of three plots of each treatment combination were used for the purposes of this study to represent an individual site for the CT and NT agricultural landuses. Cordell et al. (2006) and Smith et al. (2014) provided detailed descriptions of this long-term field study. Table 1 summarizes specific site information, dates visited for measurements, soil parent materials, and the mapped soil taxonomic descriptions for each landuse evaluated in this field study.

Throughout the region encompassing all study sites, the 30-year (1981-2010) mean annual air temperature was 16.6°C and the 30-year mean annual minimum and maximum air temperature ranged from 10.6 to 11.2°C and from 22.1 to 22.6°C, respectively (NOAA, 2013;
Table 2). The 30-year mean annual precipitation ranged from 125.6 to 128.5 cm throughout the region encompassing the study sites (NOAA, 2013; Table 2).

Infiltration Measurements

Infiltration measurements were conducted at the various sites described above on six dates between 7 November, 2015 and 6 July, 2016 (Table 1). Infiltration measurements were conducted using the same procedure at each site, as recently used and described by DeFauw et al. (2014) and Jacobs et al. (2015). A random location in each plot of the long-term, wheat-soybean, double-crop production system study at LMCRS or in triplicate locations at least 10 m apart from one another in the other landuses were chosen and any surface residue present was gently manually moved aside. A double-ring infiltrometer (model IN7-W, Turf-Tec International, Tallahassee, FL), with a 15-cm inner-ring diameter, was installed manually to a depth of approximately 2.5 cm so that no water leakage would occur from the perimeter of the outer ring. Once the infiltrometer was installed, the volumetric soil water content in the top 6 cm was measured in triplicate within the outer ring using a Theta Probe (model ML2x, Dynamax, Inc., Houston, TX). The outer ring of the infiltrometer was first filled with water, followed by filling the inner ring. All water used was tap water from a nearby building located at LMCRS or transported to the field from the University of Arkansas campus in Fayetteville. The height of the water column inside the inner ring was recorded immediately after the inner ring was filled to represent time zero and the height of the water column in the inner ring was subsequently recorded at 1, 2, 3, 4, 5, 8, 10, 12, 15, 18, and 20 minutes thereafter. If all water in the inner ring infiltrated before the 20-minute measurement period ended, then the time into the infiltration measurement when all the water had infiltrated was recorded.
The overall infiltration rate over the 20-minute measurement period, or over the time it took for complete infiltration to occur if the time was less than 20 minutes, was calculated for each infiltration measurement conducted. In addition, for each infiltration measurement conducted, the infiltration rate was calculated between each time interval, then the LN of the infiltration rate was calculated and linearly regressed against the mid-point of time (i.e., 0.5, 1.5, 2.5, 3.5, 4.5, 6.5, 9, 11, 13.5, 16.5, and 19 minutes) using Excel (version 2016, Microsoft Corporation, Redmond, WA). Similar to Desrochers et al. (2019), based on the resulting linear regression equations, the slope and intercept parameters were recorded for statistical analyses. The coefficient of determination (i.e., $r^2$ value) was also recorded for each linear regression equation.

Soil Sample Collection, Processing, and Analyses

At least 30 minutes after each infiltration measurement had been conducted, soil samples were collected from the top 10 cm within the infiltration measurement area using a 2-cm diameter push probe. Samples were dried in a forced-draft oven at 70°C for 48 hours, crushed, and then sieved through a 2-mm mesh screen for soil particle-size and chemical analyses.

Soil particle-size analyses were conducted according to a modified 12-hr hydrometer method (Gee and Or, 2002). Fifty (± 0.1)-gram subsamples of processed soil were combined with 50 mL of a 10% solution of sodium hexametaphosphate to disperse the particles. Soil suspensions were briefly mixed by manual swirling, transferred into 1-L sedimentation cylinders, and then diluted to a volume of 1 L with tap water. Cylinders were allowed to equilibrate to a uniform temperature overnight. Suspensions were then manually, vigorously mixed with a plunger and suspension densities were recorded using a hydrometer with a Bouyoucos scale after
40 seconds. This process was repeated three times. After the third 40-sec hydrometer reading was recorded, the cylinders were left standing until a hydrometer reading was recorded again six and 11 hours after the final mixing. Blank (i.e., no soil, just 50 mL of sodium hexametaphosphate and water) cylinders were also prepared to calibrate the hydrometer. The temperature of the blank was measured at the start of each set of 40-sec, 6-, and 11-hr hydrometer readings. The concentrations of sand, silt, and clay in each soil sample were calculated using standard equations (Gee and Or, 2002).

Soil electrical conductivity (EC) and pH were determined potentiometrically using an electrode in a 1:2 (mass/volume) soil-to-water mixture. Soil organic matter concentration was determined by loss-on-ignition, in which the soil was combusted in an oven for 2 h at 360°C. Total C and N concentrations were determined by high-temperature combustion with an Elementar VarioMAX Total C and N Analyzer (Elementar Americas Inc., Mt. Laurel, NJ). Upon treatment with dilute hydrochloric acid, none of the soil from any sites effervesced, thus all measured soil C was assumed to be organic C. The soil C:N, N:SOM, and C:SOM ratios were calculated from measured TC, TN, and SOM concentrations. Soil was extracted with Mehlich-3 extractant solution in a 1:10 (mass/volume) soil-to-solution ratio (Tucker, 1992) and analyzed for extractable nutrients (i.e., P, K, Ca, Mg, S, Fe, Na, Mn, Cu, and Zn) by inductively coupled argon-plasma spectrophotometry (CIROS CCD model; Spectro Analytical Instruments, MA).

Using the measured sand, silt, clay, and SOM concentrations from each infiltration measurement, the $K_{sat}$ and soil bulk density were estimated based on regression relationships (Saxton and Rawls, 2006). This approach has been used previously by Brye et al. (2006) to derive additional soil properties after initial soil sampling had occurred. Based on the regression-
estimated soil bulk density and measured soil nutrient concentrations (mass-per-mass units), soil nutrient contents were calculated and reported as kg or Mg ha\(^{-1}\).

Statistical Analyses

Based on a completely random design, a one-factor analysis of variance (ANOVA) was conducted using the PROC MIXED procedure in SAS (version 9.4, SAS Institute, Inc., Cary, NC) to evaluate the effects of landuse on overall infiltration rate, slope and intercept parameters characterizing the linear relationship between the LN of the infiltration rate and the mid-point of time, estimated \(K_{sat}\), ASMC in the top 6 cm, sand, silt, and clay concentrations, SOM content, estimated bulk density, pH, EC, extractable soil nutrient (i.e., P, K, Ca, Mg, S, Na, Fe, Mn, Zn, and Cu) contents, TC and TN contents, and C:N, N:SOM, and C:SOM ratios in the top 10 cm of soil. When appropriate, means were separated by least significant difference (LSD) at the 0.05 level.

Linear correlation analyses were conducted using Minitab (version 18, Minitab, Inc., State College, PA) to analyze the relationship between soil hydraulic properties (i.e., overall infiltration rate, slope and intercept parameters characterizing the linear relationship between the LN of the infiltration rate and the mid-point of time, and estimated \(K_{sat}\)) and soil physical and chemical properties [i.e., ASMC in the top 6 cm; sand, silt, and clay concentrations; SOM content; estimated bulk density; pH and EC; extractable nutrient (i.e., P, K, Ca, Mg, S, Na, Fe, Mn, Zn, and Cu) contents; TC and TN contents; and C:N, N:SOM, and C:SOM ratios in the top 10 cm]. Significance was judged at the \(P < 0.05\) level.
**Results and Discussion**

*Initial Soil Properties*

Particle-size Distribution and Texture

Since infiltration is known to be affected by many different near-surface soil properties, it was necessary to evaluate the degree of similarity in particle-size distribution among the landuses, as similarly textured soils were targeted to justify infiltration comparisons among landuses. Among individual replicate samples, sand content ranged from 0.01 g g\(^{-1}\) at a CT site at LMCRS to 0.42 g g\(^{-1}\) at another CT agricultural site at LMCRS. Silt content ranged from 0.54 g g\(^{-1}\) at a CT site at LMCRS to 0.82 g g\(^{-1}\) at a NT agricultural site at LMCRS. Clay content ranged from 0.02 g g\(^{-1}\) at a NT site at LMCRS to 0.23 g g\(^{-1}\) at a CRP site at PTRS.

Sand, silt, and clay concentrations in the top 10 cm differed among landuses \((P < 0.01\); Table 3). The native prairie had the largest mean sand content, which differed \((P < 0.05)\) from that of the other five landuses. The deciduous and coniferous forests and CT and NT agroecosystems had mean sand contents that were similar \((P > 0.05)\) to each other. The CRP had the lowest mean sand content, which differed \((P < 0.05)\) from that of the other five landuses. The CT and NT agroecosystems and deciduous forest had mean silt contents that were similar \((P > 0.05)\) to each other and greater than that of the other three landuses. The CT agroecosystem, deciduous forest, and CRP had mean silt contents that were similar \((P > 0.05)\) to each other and less than that of the NT agroecosystem. In addition, the deciduous and coniferous forests and CRP had mean silt contents that were similar \((P > 0.05)\) to each other and less than that of the CT and NT agroecosystems. The native prairie had the lowest mean silt content, which differed \((P < 0.05)\) from that of the other five landuses. The CRP had the largest mean clay content, which differed \((P < 0.05)\) from that of the other five landuses. The coniferous and deciduous forests, native prairie, and CT agroecosystem had mean clay contents that were similar \((P > 0.05)\).
0.05) to each other. The CT and NT agroecosystems and deciduous forest also had mean clay contents that were similar ($P > 0.05$) to each other, but were less ($P < 0.05$) than that of the other three landuses.

The distribution of sand, silt, and clay particles in a soil determine soil texture and can greatly affect infiltration rate (USDA-NRCS, 2014). Soils that contain large sand contents generally have greater infiltration rates than soils than contain large clay contents because sandy soils have larger soil particles, which creates larger pore spaces for water to move through (USDA-NRCS, 2014). Alternatively, soils with large clay contents have smaller pore spaces, causing water flow and infiltration to be restricted (Alaoui et al., 2011; USDA-NRCS, 2014). Consequently, since soil texture can affect infiltration (USDA-NRCS, 2014), it was important that the near-surface soil texture was relatively uniform throughout all soils where infiltration was measured in order to appropriately compare infiltration rates between locations. Despite several differences in particle-size distribution among landuses included in this study, 94% of the sites where infiltration measurements were conducted had a silt-loam texture in the top 10 cm, while the remainder had a silt texture. Since a large majority of the soils were silt-loam textured, it was concluded that there was relative uniformity among the sites sampled across the six landuses and therefore any potential effects of differential particle-size distributions on infiltration were considered small enough to be ignored.

Antecedent Soil Moisture Content

Antecedent soil moisture content is a dynamic soil property that can greatly influence infiltration into the soil (USDA-NRCS, 2014). When measuring infiltration rate, the impact of ASMC can be minimized by conducting the infiltration measurements in soils with similar
moisture contents. In general, soil moisture content has an inverse effect on infiltration rate. As
the soil approaches saturation, the infiltration rate tends to decrease because the available pore
space to store more water and the hydraulic gradient are decreasing (Shukla et al., 2003; Wang et
al., 2009), unless there is sufficient redistribution to remove infiltrated water away from the
surface.

To minimize potential differences in moisture contents among the sites and soils included
in this study, infiltration measurements were conducted approximately three to five days after a
rainfall event occurred at each respective study site. Although attempts were made to maintain
uniform ASMC by conducting all infiltration measurements shortly after a rainfall event had
occurred, the spatial and temporal variability in infiltration-related, near-surface soil properties,
other than particle-size distribution, present at each site (i.e., water-holding capacity, available
pore space, etc.) made it difficult to maintain an identical ASMC at each site. Among all
measured sites, ASMC in the top 6 cm ranged from 0.03 cm$^3$ cm$^{-3}$ at the Roth Prairie to 0.62 cm$^3$
cm$^{-3}$ at a deciduous forest site at PTRS. Consequently, the ASMC in the top 6 cm differed among
landuses ($P < 0.05$). Antecedent soil moisture contents in the top 6 cm for the deciduous forest,
CRP, and native prairie were similar ($P > 0.05$), but were greater ($P < 0.05$) than that for the
other three landuses (Table 3). Antecedent soil moisture contents in the top 6 cm for the CRP,
native prairie, and CT and NT agroecosystems were similar ($P > 0.05$) to each other and less ($P$
$< 0.05$) than that for the deciduous forest. Antecedent soil moisture contents in the top 6 cm for
the CT and NT agroecosystems and coniferous forest were similar ($P > 0.05$) to each other, but
less ($P < 0.05$) than that for the other three landuses.

Despite the variability of the ASMC prior to the start of infiltration measurements, it is
likely that the soil moisture contents somewhat equalized within the first few minutes of the 20-
minute measurement period. It has been observed that the initial rate at which water is able to enter soil decreases rapidly as the soil becomes wetted and nearly saturated (Horton, 1933), eventually reaching a stable infiltration rate referred to as steady-state infiltration (Císlerová et al., 1988). As water continually infiltrates into soil over time and approaches a steady-state infiltration rate, the soil moisture content increases, and the impact of the ASMC decreases (Lal and Shukla, 2004; Sajjadi et al., 2016). Within several minutes after infiltration began, the soil moisture content in the top few centimeters likely equalized and no longer strongly influenced the infiltration rate. Sajjadi et al. (2016) demonstrated that the ASMC has the greatest impact on the infiltration rate during the first few minutes of a measurement period. However, as infiltration was measured in this study over a 20-minute period rather than just during the first few minutes of infiltration, it was likely that any differences in infiltration caused by ASMC were negligible and could therefore be ignored.

**Landuse Effects on Infiltration**

**Overall Infiltration Rate**

Infiltration over the 20-minute measurement period followed an expected pattern, where infiltration rates were initially large, but tended to decrease exponentially over time in all landuses (Figure 4A). However, large variability was encountered among replicate measurements within and between landuses, which was also somewhat expected, as soil hydraulic properties are known to have large spatially variability.

The overall infiltration rates among individual replicate measurements ranged from 0.003 mm min\(^{-1}\) at a CT agricultural site at LMCRS to 5.75 mm min\(^{-1}\) at a deciduous forest site at LMCRS. The overall infiltration rate for the deciduous forest was nearly seven times greater
(1.17 mm min\(^{-1}\); \(P < 0.05\)) than that for the other five landuses, which, in contrast to that hypothesized, did not differ and averaged 0.17 mm min\(^{-1}\) (Table 3). Though the ASMC in the top 6 cm differed among the landuses (\(P < 0.01\)) at the time infiltration measurements began (Table 3), it does not appear that the overall infiltration rate was influenced by the ASMC, as there was no correlation between overall infiltration rate and the ASMC in the top 6 cm (\(P > 0.05\); Table 4). However, despite the lack of correlation between overall infiltration rate and ASMC across all data, the deciduous forest had the numerically largest mean ASMC in the top 6 cm, while also exhibiting the largest overall infiltration rate (Table 3). Similar to the results of this study, Bharati et al. (2002) measured infiltration and gravimetric soil moisture among different landuses on Coland loam and sandy-loam (Cumulic Endoaquolls) soils in central Iowa and reported no correlation between cumulative infiltration and ASMC in the top 7 cm.

One explanation for the greater infiltration into the deciduous forest compared to the other landuses is that the deciduous forest may have had a more expansive root system. Forested areas typically have many macroroots (> 2-mm diameter) that facilitate soil water infiltration and hydraulic conductivity (Aubertin, 1971; Watson and Luxmoore, 1986). Water can preferentially flow along the edges of macroroots and in the macropores created by old, decayed root systems (Aubertin, 1971; Meek et al., 1992; Sidle et al., 2001; Alaoui et al., 2011). A greater abundance of macroroots and preferential flow may have resulted in greater infiltration in the deciduous forest compared to the other landuses. Alaoui et al. (2011) compared porosity of forested and grassland soils in the 5- to 15-cm depth of silt-loam and clay-loam soils in western Switzerland and reported greater total porosity in forested compared to grassland soils. In addition, macropores in the forested soil were able to transmit water more effectively than macropores in the grassland soil (Alaoui et al., 2011).
The greater infiltration in the deciduous forest could also be explained by an observed trend between SOM, estimated bulk density, and overall infiltration rate. Data indicated a strong positive correlation between overall infiltration rate and SOM content \( (P < 0.01) \) and a strong negative correlation between overall infiltration rate and estimated bulk density in the top 10 cm \( (P < 0.01; \text{Table 4}) \). Furthermore, landuses with greater SOM contents and lower bulk densities generally exhibited greater overall infiltration rates (Table 3). One property influencing the bulk density of the soil is the SOM content because SOM and bulk density share an inverse relationship (Gonzalez-Sosa et al., 2010). Consequently, it follows that, as the SOM content of the soil increases and bulk density decreases, more pore space is available to transmit water and, therefore, overall infiltration rate increases (Meek et al., 1992; Bharati et al., 2002). The deciduous forest had the numerically largest SOM content and lowest bulk density, as well as the greatest overall infiltration rate, compared to the other five landuses (Table 3). Conversely, the CT and NT agroecosystems and CRP grassland, which had the numerically lowest SOM contents and greatest bulk densities, exhibited the numerically lowest overall infiltration rates.

Similar results to this study were reported by Bharati et al. (2002) in Coland loam and sandy-loam (Cumulic Endoaquolls) soils in central Iowa, where cumulative infiltration measured over a 60-minute period was greater in a deciduous forest compared to other landuses included in the study (i.e., deciduous forest, grassland, grazed pasture, and cultivated agriculture). The greater infiltration in the deciduous forest was attributed to a greater number of macropores formed from a combination of tree roots and soil biota (Bharati et al., 2002). Additionally, at most sites, decreased bulk density was also related to increased infiltration (Bharati et al., 2002).

Similar to overall infiltration rate, estimated \( K_{sat} \) in the top 10 cm was also greater \( (P < 0.05) \) in the deciduous forest compared to the other five landuses (Table 3). Saturated hydraulic
conductivity can be used as an index for overall soil quality (Bodhinayake and Si, 2004). Saturated hydraulic conductivity and infiltration rate are related to one another and are both influenced by soil aggregation and SOM content (Bodhinayake and Si, 2004). Similar results to this study were reported by Gonzalez-Sosa et al. (2010) from coarse-textured soils in multiple landuses in southeastern France, where it was reported that the deciduous forest had a greater $K_{sat}$, mean SOM content, and total porosity, as well as a lower mean bulk density in the 0- to 5-cm depth interval compared to coniferous forest, pastureland, and cultivated agricultural landuses.

Similar to overall infiltration rate, TC and TN contents in the top 10 cm were also numerically greater in the deciduous forest compared to the other five landuses (Table 3). One explanation for this is that SOM content is a large source for soil organic C (SOC) and N (SON) (Gregorich et al., 1994). Both organic and inorganic forms of C and N are included in TC and TN, respectively (Gregorich et al., 1994). Greater amounts of SON and SOC, and therefore SOM, in the topsoil can result in increased soil aggregation (Elliott, 1986; Six et al., 2000B; Fang et al., 2015). Studies have shown that greater amounts of TC or TN are related to increased SOM and, as a result, increased aggregation and infiltration (Chaney and Swift, 1984; Elliott, 1986; Wuest et al., 2005; Stone and Schlegel, 2010).

The deciduous forest landuse also had the numerically largest extractable soil P and K contents compared to the other five landuses (Table 3). Studies have shown that increased amounts of P or K in soil can result in improved soil aggregation (Elliott, 1986; Levy and Torrento, 1995; Fang et al., 2015). In general, greater soil P results from increased SOM because SOM contains organic P (Elliott, 1986). As SOM begins to accumulate, the aggregate stability of the soil strengthens, and the number of macroaggregates in the soil increases (Elliott, 1986). The
results of this study indicated that extractable soil P content was positively correlated with SOM content in the top 10 cm \((P < 0.01)\). Similar to the results of this study, Fang et al. (2015) reported greater soil P contents and macroaggregation in the top 15 cm of deciduous forests compared to coniferous forests in clayey soils in Taihe County in Southern China. Whereas the effect of K on aggregation is less well known, some studies have shown that extractable soil K can limit clay dispersion and increase aggregate stability (Levy and Torrento, 1995; Amézketa, 1999).

In contrast to that hypothesized, overall infiltration rate into the native prairie ranked second numerically behind the deciduous forest and was similar to that in the coniferous forest, CRP, and CT and NT agroecosystems (Table 3). Native prairies, which have never been cultivated, have been shown to have increased infiltration and decreased runoff that result from factors such as greater aggregate stability, SOM content, hydraulic conductivity, and earthworm activity compared to cultivated soils (Lindstrom et al., 1994; Schwartz et al., 2003; Bodhinayake and Si, 2004; Harper et al., 2008; Stone and Schlegel, 2010). One explanation was that the overall infiltration rate of the native prairie sites may have been affected by infrequent management practices other than cultivation. For instance, all native prairies included in this study were periodically burned. Burning is a common management practice that controls weeds and woody encroachment, decreases the risk of wildfires, and promotes growth in prairies (Davison and Kindscher, 1999). Residue burning permanently removes vegetation that would eventually return to the soil as organic matter (Razafimbelo et al., 2006; Virto et al., 2007). As SOM content decreases, aggregate stability, available pore space, and the capacity for water infiltration decrease (Franzluebbers, 2002). In addition, the hydrophobic properties of ash, produced through the burning of above-ground biomass, may lead to increased surface runoff.
and decreased infiltration (Johansen et al., 2001). Furthermore, vehicles occasionally traveled across all native prairie sites that were included in this study, at least slightly increasing compaction. Soil compaction can result in a decreased infiltration rate, which is due to the destruction of soil pores through which water can travel (Meek et al., 1992; Sajjadi et al., 2016).

The deciduous forest sites were much less impacted by any management practices, with the exception of periodic mowing in the pecan grove, which may explain why, in contrast to that hypothesized, the deciduous forest exhibited greater overall infiltration than the native prairie. The negative impact of management practices on native prairie soils is evidenced by the greater \( (P < 0.05) \) bulk densities and smaller \( (P < 0.05) \) SOM contents in the top 10 cm of the native prairie compared to the deciduous forest (Table 3), which were both shown to be strongly correlated with overall infiltration rate \( (P < 0.01; \) Table 4). Furthermore, the native prairie exhibited numerically smaller TC and TN contents in the top 10 cm of soil compared to deciduous forest, likely resulting from the smaller SOM contents in the native prairie compared to the deciduous forest (Table 3). Jastrow (1996) demonstrated the adverse effect of management practices other than cultivation by showing that native prairies that were subject to periodic management practices, including burning and grazing over a period of several decades, had similar macroaggregation, which is a product of SOM, in the top 10 cm compared to grasslands that were formerly cultivated in Mundelein silt loam (Aquic Argiudolls) and Wauconda silt loam (Udollic Endoaqualfs) soils near Chicago, IL.

The CRP landuse was expected to have a greater overall infiltration rate than the agricultural landuses, but measurement results did not support this hypothesis (Table 3). The CRP results could be explained by the periodic disturbance (i.e., mowing and aboveground biomass removal) the CRP sites experienced for approximately 10 years prior to this study.
Possibly as a result these management practices, the CRP exhibited a relatively small SOM content and relatively large bulk density in the top 10 cm compared to the other five landuses, which may have caused the overall infiltration rate of the CRP to be smaller than was originally expected (Table 3). However, long periods of time without soil disturbance have been shown to increase SOM and soil aggregation, decrease bulk density, and increase infiltration (Jastrow, 1996; Martens, 2000; Franzluebbers, 2002; Bodhinayake and Si, 2004; Wang et al., 2010). Therefore, it is possible that, given enough time, the overall infiltration rate of the CRP could increase beyond what was observed in this study. Similar results were reported by Schwartz et al. (2003) from fine-textured soils in the Southern Great Plains, where the hydraulic conductivity of CRP grasslands that had been established for 10 years were generally smaller than that of native prairies and CT agroecosystems. Schwartz et al. (2003) concluded that soils required more than 10 years under CRP-grassland management for the damaging effects of tillage on soil structure and hydraulic conductivity to be mitigated.

Similar to that hypothesized, the coniferous forest had relatively low overall infiltration rates, which was likely due to greater soil water repellency (SWR) from hydrophobic pine needle accumulation than in the deciduous forest, whose forest floor material is generally more base-cation rich leading to more rapid decomposition (Woche et al., 2005; Doerr et al., 2006). Large SWR under coniferous forest landuse likely contributed to decreased infiltration compared to the deciduous forest landuse (Butzen et al., 2015).

Studies have also shown that deciduous forests can have greater soil aggregation and macroporosity than coniferous forests (Graham et al., 1995; Wang et al., 2009; Fang et al., 2015), which can lead to greater surface infiltration (Wang et al., 2009). One explanation for greater aggregation and macroporosity involves the relative abundance of earthworms in
coniferous and deciduous forests. Earthworms are typically present in greater abundance in deciduous compared to coniferous forests (Terhivuo, 1989; Graham et al., 1995) due to the base-cation-rich nature of the deciduous forest litter and the typically greater abundance of fungal hyphae in deciduous forest soils compared to that under conifers (Graham et al., 1995; Ste-Marie and Paré, 1999; Reich et al., 2005).

Graham et al. (1995) compared soil aggregate stability of coniferous and deciduous forests in a fine sandy loam in the San Gabriel Mountains near Los Angeles, CA. The deciduous and coniferous forests consisted of scrub oak (Quercus dumosa Nutt.) and Coulter pine (Pinus coulteri B. Don), respectively. The A horizon (i.e., top 6 cm) of the deciduous forest had approximately 90% water-stable aggregates and consisted almost entirely of worm casts, whereas the A horizon (i.e., top 1 cm) in the coniferous forest had only 78% water-stable aggregates and contained no worm casts (Graham et al., 1995). The absence of earthworms within the coniferous forest could therefore be an additional explanation for why the coniferous forest exhibited a reduced overall infiltration rate compared to the deciduous forest.

There are several additional plausible explanations to consider in the differences in overall infiltration rates among landuses, particularly for the coniferous forest results. The coniferous forest sites used in this study consisted of once-cultivated plantations that were periodically mowed and burned, creating an environment similar to that of the native prairies. In addition, the deciduous forest sites were older (i.e., ≥ 30 years old) than the coniferous forest sites (i.e., ≥ 12 years old). Generally, as the age, or time since establishment, of a forest increases, the size and abundance of the root channels will increase (Archer et al., 2015). The root channels can form a network of macropores under the soil surface through which water can infiltrate and flow quickly (Aubertin, 1971). An extensive macropore network in the deciduous
forest is evidenced by the greater \((P < 0.05)\) estimated \(K_{\text{sat}}\) in the deciduous compared to the coniferous forest landuse (Table 3). Saturated hydraulic conductivity is largely determined by soil pore size, particularly macroaggregation, because in saturated soils water flows almost entirely through macropores (Cameira et al., 2003; Jarvis, 2007; Eusufzai and Fujii, 2012; Karahan and Erşahin, 2017). Therefore, an older forest with a well-developed root system and macropore network would be expected to have a greater \(K_{\text{sat}}\) than a younger forest. Similar to the results of this study, Wang et al. (2009) reported that deciduous forests exhibited greater initial and steady-state infiltration rates and cumulative infiltration compared to coniferous forests in residual and alluvial soils in the Simian Mountains of southwestern China. In addition, the deciduous forest had a lower bulk density and greater macroporosity and estimated \(K_{\text{sat}}\) in the top 20 cm compared to the coniferous forest (Wang et al., 2009).

The deciduous and coniferous forests of this study exhibited numerically greater soil C:N ratios in the top 10 cm than the CRP grassland and CT and NT agroecosystems (Table 3). Differing C:N ratios likely resulted from dissimilar SOM decomposition rates among landuses. Greater decomposition of SOM can lead to less N present and larger C:N ratios in soil. Lauber et al. (2008) showed that deciduous and coniferous forest soils generally have greater C:N ratios than CT agricultural or grassland soils because tree litter typically contains greater amounts of lignin, which has a large C:N ratio, compared to other types of landuses.

In contrast to that hypothesized, the overall infiltration rate for the NT did not differ from that in the CT agricultural landuse. Many studies have reported an increase in infiltration rate into NT compared to CT agricultural soils (Azooz and Arshad, 1996; Bhattacharyya et al., 2008; Stone and Schlegel, 2010). It is known that tillage can cause soil compaction, which can reduce overall porosity and decrease infiltration (Azooz and Arshad, 1996; Sajjadi et al., 2016).
However, on average, neither SOM content nor soil bulk density in the top 10 cm differed ($P > 0.05$) between CT and NT agroecosystems (Table 3), thus indicating that infiltration should not differ between the two current agricultural landuses. Additionally, the CT and NT agroecosystems included in this study exhibited similar ($P > 0.05$) extractable soil P contents in the top 10 cm (Table 3), which has been shown to be related to SOM and aggregation (Elliott, 1986; Fang et al., 2015).

Eusufzai and Fujii (2012) reported that the incorporation of crop residue into tilled clay-loam soils in northeastern Japan resulted in increased aggregation and overall soil quality compared to soils which were not amended with crop residue. Therefore, it is possible that, in addition to similar SOM and bulk density in the top 10 cm on average across the agricultural landuses, the degree of aggregation may also be similar between the CT and NT sites evaluated in this study, resulting in similar overall infiltration rates. Similar soil aggregation in CT and NT landuses is supported by the similarity ($P > 0.05$) of electrical conductivities (EC) and extractable Ca contents in the top 10 cm between CT and NT agroecosystems (Table 3). Electrical conductivity is related to the concentrations of cations in soil solution (Friedman, 2005). Cations, such as Ca, have been shown to augment clay flocculation, which improves soil aggregation (Amézketa, 1999; Lado et al., 2004). Larger EC values are associated with clay flocculation, whereas smaller EC values are associated with clay dispersion (Miller, 1987; Amézketa, 1999). Lado et al. (2004) reported increased soil EC was related to decreased clay dispersion and increased pore volume in the 5- to 15-cm depth of sandy-loam (Humic Dystrudepts) soils in northwestern Spain.
Infiltration Rate Trends over Time

Infiltration rates were expected to be relatively large initially and decline over time. This trend results as water infiltrates throughout the soil matrix and progressively fills the available soil pore space (Hillel, 2004). As the soil approaches saturation, the infiltration rate begins to decline, eventually reaching a more constant steady-state infiltration rate (Hillel, 2004), as occurred for most landuses. However, though the infiltration rate over the 20-minute measurement period was shown to be affected by landuse (Table 3), infiltration trends over time during the 20-minute measurement period could also yield insight into the effects of landuse on infiltration. Therefore, measured infiltration rates were calculated between each consecutive measurement times and LN transformed to linearize the relationship (Figure 4) in order to facilitate additional statistical analyses. Based on the resulting linear regression equations, the slope and intercept parameters were used to represent the trend in infiltration rate over time.

The LN transformation of infiltration rates between measurements times resulted in $r^2$ values ranging from < 0.1 to 1.0 across all infiltration measurements. Approximately 51% of the regression equations had $r^2$ values exceeding 0.6, indicating that a majority of infiltration measurements exhibited a trend of decreasing infiltration rate over time as infiltration approached steady state, reinforcing what has been observed in previous studies (Sajjadi et al., 2016). A total of 105 infiltration measurements were included in this study. However, $r^2$ values were calculated for only 95 of the infiltration measurements, with 10 measurements containing insufficient data for calculating a $r^2$ value. This lack of data was a result of either rapid infiltration during the measurement period resulting in too few data points, such as in the case with one native prairie site and one deciduous forest site, or the water in the inner ring infiltrated
so slowly that measurable infiltration did not occur during the 20-minute measurement interval, as occurred for several of the current agricultural sites.

Slope Parameter Characterizing the LN of Infiltration Rate and Mid-Point of Time

The slope parameter characterizing the linear relationship between the LN of the infiltration rate and the mid-point of time relates to the rate at which the infiltration rate declines from the initial infiltration rate towards steady-state infiltration. The steady-state infiltration rate is largely affected by the hydraulic conductivity of the soil, which is related to soil pore size (Hillel, 2004; Archer et al., 2015; Sajjadi et al., 2016). Soil pore size, in turn, is determined by sand, silt, and clay distribution and soil structure (Hillel, 2004).

The slope parameter ranged from -0.76 for a deciduous forest site at LMCRS to 0.20 at a deciduous forest site at PTRS. In contrast to results for overall infiltration rate, the slope parameter was unaffected ($P > 0.05$) by landuse. The fact that the slopes were similar among the six landuses may indicate that the mechanism of infiltration, after infiltration began, was similar among landuses. These results may be due to the purposely targeted similarities in texture, parent material, and geography among the various landuses included in this study. For instance, 94% of the sites where infiltration measurements were conducted had a silt-loam texture. Additionally, all soils included in this study originated from fine-textured loess or alluvium parent materials in the Delta region of the LMRV. These results indicate the process of infiltration represented by the slope parameter may have been influenced by intrinsic soil characteristics more than by current, aboveground landuse and/or management practices.

Alternatively, the similarity of the slope parameters could be due to the combined effects of ecosystem age and management practices on soil characteristics among the six landuses. For
instance, the native prairies included in this study were managed through periodic burning, which could have altered soil conditions through the removal of organic matter and the creation of hydrophobic conditions near the soil surface. In addition, the deciduous forest sites included in this study were non-native and may have been cultivated at some point in previous decades. The coniferous forests had been planted as plantations less than 15 years previously and the inter-row areas were also mowed periodically. Combinations of management practices and soil properties may have caused the process of infiltration represented by the slope parameter to be similar among the landuses.

The similarity in the slope parameter among the six landuses may also indicate that the slope parameter is more resilient to changes created by landuse than was originally hypothesized. The effect of landuse on the rate of change of the infiltration rate over time, represented by the slope parameter, is not fully understood. Further research is warranted into factors that affect the slope parameter of the linear relationship characterizing the LN of the infiltration rate over time.

Intercept Parameter Characterizing the LN of Infiltration Rate and Mid-Point of Time

In contrast to the slope, the intercept parameter characterizing the linear relationship between the LN of the infiltration rate and the mid-point of time differed ($P < 0.05$) among landuses. The intercept parameter ranged from -0.94 at a CT agricultural site at LMCRS to 3.99 at a deciduous forest site at LMCRS. Specifically, the intercept parameter relates to the instantaneous infiltration rate when time was zero, which is a theoretical value representing the soil conditions before infiltration first occurred. The instantaneous, or initial, infiltration rate is mainly controlled by the ASMC and the porosity at the immediate soil surface (Lal and Shukla, 2004). When water is applied to the soil, matric potential gradients cause water to be sucked into
empty pores at the soil surface. Drier soil pores create stronger suction gradients, causing the instantaneous infiltration rate to increase as soil moisture decreases (Lal and Shukla, 2004). In addition, more abundant and larger pores at the soil surface allow water to immediately enter the soil at a faster rate than soils with fewer or smaller pores at the soil surface (Lal and Shukla, 2004). Therefore, the instantaneous infiltration rate will tend to be greater in soils with greater porosity at the soil surface (Lal and Shukla, 2004). Soil surface porosity is an indicator of overall soil quality and can be affected by SOM, bulk density, and degree of soil aggregation (Bhattacharyya et al., 2008).

Bhattacharyya et al. (2008) measured initial infiltration rate and porosity from CT and NT agroecosystems on sandy-clay-loam (Typic Haplaquepts) soils in the Indian Himalayas. The mean initial infiltration rate during the first five minutes of the measurement period was approximately 14% greater in the NT compared to the CT agroecosystem (Bhattacharyya et al., 2008). Additionally, the NT contained approximately 19% more transmission pores (i.e., 30 to 150 μm in diameter) in the top 15 cm of soil than the CT agroecosystem (Bhattacharyya et al., 2008). The greater initial infiltration rate and abundance of transmission pores in the topsoil of the NT likely resulted from the greater soil organic C content in the top 15 cm in the NT compared to the CT agroecosystem (Bhattacharyya et al., 2008).

The effect that landuse had on soil conditions prior to infiltration may explain why the intercepts differed among the six landuses. The coniferous forest, native prairie, and CRP had similar ($P > 0.05$) intercepts to each other, but lower intercepts ($P < 0.05$) than for the deciduous forest (Table 3). The coniferous forest, native prairie, and CRP were generally more managed than the deciduous forest and less managed than the agroecosystems, thus the coniferous forest, native prairie, and CRP were expected to have a smaller intercept than that of the deciduous
forest, but a larger intercept than that of the current agroecosystems. In addition, the intercepts for the CRP and current CT and NT agroecosystems had similar ($P > 0.05$) intercepts to each other and smaller ($P < 0.05$) intercepts than the other three landuses (Table 3). The CRP grassland and current CT and NT agroecosystems were the three most intensely managed landuses, which may explain why the intercept parameters for these three were numerically lower than for the remaining three, less-managed landuses. Greater soil disturbance due to more intensive management practices likely caused decreased soil aggregation and porosity, as evidenced by the numerically smaller SOM contents and numerically larger bulk densities compared to the three less-managed landuses (Table 3). In addition, the CRP and CT and NT agroecosystems exhibited numerically smaller TC and TN contents compared to the remaining three, less-managed landuses. Studies have shown that greater TC and TN resulted in greater soil aggregation (Chaney and Swift, 1984; Wuest et al., 2005; Stone and Schlegel, 2010). The results of this study indicated that the intercept parameter was positively correlated with SOM, TC, and TN contents ($P < 0.01$) and negatively correlated with bulk density in the top 10 cm ($P < 0.01$; Table 4). The CRP and CT and NT agroecosystems also exhibited numerically smaller extractable Fe contents in the top 10 cm compared to the remaining three, less-managed landuses. Studies have shown that greater amounts of Fe can cause flocculation of clay particles and increased soil aggregate stability (Shainberg et al., 1987; Amézketa, 1999; Mbagwu and Auerswald, 1999).

The deciduous forest had the largest intercept parameter ($P < 0.05$) among the other five landuses (Table 3), which indicated that, collectively, the near-surface soil conditions in the deciduous forest sites were more conducive to infiltration than the other five landuses, as evidenced by the relatively large SOM content and relatively small bulk density in the deciduous
forest compared to the other five landuses (Table 3). Additionally, the deciduous forest exhibited the numerically largest TC and TN contents among the six landuses (Table 3). The deciduous forest was hypothesized to have the second greatest intercept parameter because the deciduous forest was less managed than the other landuses, with the exception of the native prairie. Soils that are less disturbed through management practices have been shown to have greater initial infiltration rates than more intensely managed soils (Lal et al., 1989; Shukla et al., 2003; Bhattacharyya et al., 2008).

Since the deciduous forest had the largest mean ASMC in the top 6 cm (Table 3), the near-surface soil in the deciduous forest was likely more conductive for water than the other landuses at the time infiltration measurements were conducted, which could also explain the deciduous forest having the largest intercept parameter (Table 3). Consequently, one might expect initial soil water content differences to contribute to the explanation for the intercept differences among landuses. However, among all landuse data, the intercept parameter was uncorrelated with ASMC in the top 6 cm ($P > 0.05$; Table 4), indicating that the intercept parameter was not solely influenced by the ASMC of the soil. It is possible that ASMC had less of an effect on the early infiltration process than was originally expected, but rather more complex relationships among soil properties and landuse characteristics appear to exist that control infiltration.

**Soil Property Correlations with Infiltration-related Parameters**

Linear correlation analyses were conducted in order to examine the relationship between infiltration-related properties (i.e., overall infiltration rate, slope and intercept parameters characterizing the linear relationship between the LN of the infiltration rate and the mid-point of time, and estimated $K_{sat}$) and various soil physical and chemical properties among landuses [i.e.,
ASMC in the top 6 cm; sand, silt, and clay concentrations; SOM content; estimated bulk density; pH and EC; extractable nutrient (P, K, Ca, Mg, S, Na, Fe, Mn, Zn, and Cu) contents; TC and TN contents; and C:N, N:SOM, and C:SOM ratios in the top 10 cm] (Table 4). These analyses were performed to identify factors which may strongly influence the infiltration process.

Understanding and identifying soil physical and/or chemical properties that are related to infiltration may allow for more accurate, future prediction of infiltration in a given landuse without the need to directly perform extensive infiltration measurements.

Numerous near-surface soil physical and chemical properties were correlated with infiltration-related parameters (Table 4). As expected, SOM, TC and TN contents, and C:SOM ratio were positively correlated \( (P < 0.01) \) with overall infiltration rate, the intercept parameter, and estimated \( K_{sat} \). Clay concentration, extractable soil S and Mn contents, and C:N ratio also were positively correlated \( (P < 0.05) \) with the intercept parameter, while sand concentration, EC, extractable soil P, K, S, Fe, and Zn contents, and N:SOM ratio were positively correlated \( (P < 0.05) \) with estimated \( K_{sat} \).

Soil organic matter is an important factor affecting infiltration (Franzluebbers, 2002). Because the particle density of mineral soil is so much greater than that of organic matter, greater amounts of SOM can cause the bulk density of the soil to decrease, resulting in greater pore space, and leading to greater infiltration (Franzluebbers, 2002). Increased SOM, which acts as a binding agent for soil aggregates (Tisdall and Oades, 1982; Puget et al., 2000), generally leads to increased aggregate stability (Amézketa, 1999; Franzluebbers, 2002; Márquez et al., 2004), and, as a result, increased infiltration (Le Bissonnais and Arrouays, 1997; Franzluebbers, 2002).

Soil organic matter is also a major source of SON and SOC, with humic matter composing 30 to 55% of organic matter, which, in turn, is responsible for 50 to 58% of total
SOC (Gregorich et al., 1994). Both inorganic and organic forms of N are included in TN, however approximately 95% of TN has been shown to be contained within SOM (Walworth, 2013). Studies have shown that greater amounts of SON and SOC, and therefore SOM, in topsoil result in increased soil aggregation (Elliott, 1986; Six et al., 2000B; Fang et al., 2015). Wuest et al. (2005) observed that infiltration rate was correlated with total soil C and N, aggregate stability, and percolation ($r = 0.67$ to 0.95).

Saturated hydraulic conductivity is related to infiltration rate and can be used to assess the impacts of landuse management practices as well as overall soil quality (Bodhinayake and Si, 2004; Gonzalez-Sosa et al., 2010). Many studies have shown SOM to have a positive effect on $K_{sat}$ (Lado et al., 2004; Eusufzai and Fujii, 2012) because greater amounts of SOM can cause macroaggregation and macroporosity to increase, which can result in greater $K_{sat}$ (Lado et al., 2004; Jarvis, 2007). Eusufzai and Fujii (2012) showed that macroporosity and field-$K_{sat}$ increased in clay-loam soils with greater SOM contents. Lado et al. (2004) reported that soils with greater SOM contents had less clay dispersion and greater aggregation and $K_{sat}$ compared to soils with smaller SOM contents in the 5- to 15-cm depth of sandy-loam soils in northwestern Spain. Furthermore, it has been documented that increased SOM content will result in increased TC and TN (Gregorich et al., 1994; Walworth, 2013). Studies have shown that greater TC and/or TN in the topsoil result in improved soil aggregation (Chaney and Swift, 1984; Wuest et al., 2005; Liao et al., 2006; Stone and Schlegel, 2010). As soil aggregation increases, soil pore space also increases, and $K_{sat}$ will, generally, increase as a result (Bodhinayake and Si, 2004; Wang et al., 2009). Fang et al. (2015) measured soil properties among different landuses (coniferous and deciduous forests) with clayey soils in Taihe County in Southern China and reported that the
landuse with the greater total P content in the top 15 cm also had a greater macroaggregation to contribute to increased $K_{sat}$.

Greater amounts of Fe, especially in combination with Al and SOM, can cause clay particles to flocculate (Shainberg et al., 1987; Mbagwu and Auerswald, 1999). Amézketa (1999) conducted a comprehensive literature review of aggregate stability, which revealed that Fe has been shown to be positively correlated with the stability of clay particles within soil aggregates. Flocculation of clay particles leads to increased aggregation (Amézketa, 1999; Mbagwu and Auerswald, 1999) and $K_{sat}$ (Bagarello et al., 2006). Shainberg et al. (1987) showed that greater soil Fe concentrations resulted in greater $K_{sat}$ in sandy-loam soils.

The potential effect of soil K on clay flocculation is currently not well understood (Levy and Torrento, 1995; Amézketa, 1999), but has been shown to be intermediate between that of Ca and Na (Shainberg et al., 1987; Levy and Van Der Watt, 1990; Heil and Sposito, 1993). Calcium causes clay particles to flocculate (Heil and Sposito, 1993), whereas Na causes clay particles to disperse (Cresimanno et al., 1995; Amézketa, 1999). Results of studies on the effect of K on soil have been varied and contradictory (Shainberg et al., 1987; Amézketa, 1999). Quirk and Schofield (1955) reported that extractable soil K caused dispersion in clays similar to Na. However, Levy and Torrento (1995) showed that K can limit clay dispersion and increase aggregate stability, potentially increasing $K_{sat}$.

As would be expected, estimated soil bulk density was negatively correlated ($P < 0.01$) with overall infiltration rate, the intercept parameter, and estimated $K_{sat}$. As soil bulk density increases, the soil becomes more compacted, and the macropore space available to transmit water decreases (Meek et al., 1992; Wang et al., 2009), resulting in decreased infiltration (Meek et al., 1992; Bharati et al., 2002; Franzluebbers, 2002). Similar to this study, Meek et al. (1992)
reported a negative correlation between bulk density and infiltration rate \( (r^2 = 0.60) \) in cotton 
\( (Gossypium \text{ spp.}) \) agroecosystems in the western US on a Wasco sandy loam (Typic Torriorthent). Meek et al. (1992) also reported an increase in bulk density from 1.6 to 1.8 Mg m\(^{-3}\) resulted in a 53% decrease in infiltration. Furthermore, greater bulk densities generally result in lower \( K_{\text{sat}} \) (Jabro, 1992; Rawls et al., 1998) because, as soil bulk density increases, porosity also decreases, which limits the volume of conductive pores (Rawls et al., 1998). Similar results were reported by Wang et al. (2009) from residual and alluvial forest soils in the Simian Mountains of southwestern China, where \( K_{\text{sat}} \) was negatively correlated with bulk density \((P < 0.01)\) and positively correlated with macroporosity \((P < 0.01)\) in the top 60 cm.

Similar to bulk density, extractable soil Na and Mg contents were also negatively correlated \((P < 0.05)\) with overall infiltration rate and the intercept parameter. The large ionic radii of sodium cations can cause negatively charged clay particles to repel one another (Schoenau and Karamanos, 1993) and, consequently, become dispersed (Rengasamy and Olsson, 1991; Cresimanno et al., 1995; Amézketa, 1999), which can clog soil pores and result in decreased infiltration (Suarez et al., 2006). Kazman et al. (1983) measured infiltration rates of various soils using a rainfall simulator and reported that infiltration rates were significantly reduced by even low exchangeable Na concentrations.

Magnesium has been shown to be a relatively weak binding agent for soil aggregates (Heil and Sposito, 1993; Dontsova and Norton, 2002). However, the flocculation effect of Mg on clay is considered to be less than that of Ca (Zhang and Norton, 2002) because the ionic radius of Mg is much smaller than that of Ca (Heil and Sposito, 1993; Dontsova and Norton, 2002; Zhang and Norton, 2002). Therefore, aggregates with relatively large Mg concentrations may disintegrate more readily than soils with less Mg (Dontsova and Norton, 2002). Disintegration of
soil aggregates may result in clogging soil pores (Zhang and Norton, 2002). Donsova and Norton (2002) measured infiltration in Mg- and Ca-saturated soils and discovered that increased Mg caused clay particles bound in aggregates to disperse at a rate up to 14 times greater than systems containing Ca alone. This type of dispersive effect could easily lead to the clogging of available pore space, thereby reducing infiltration and soil hydraulic conductivity (Yousaf et al., 1987; Bagarello et al., 2006).

Soil pH, EC, and extractable soil Ca were negatively correlated ($P < 0.05$) with the intercept parameter, while clay concentration was negatively correlated ($P < 0.01$) with estimated $K_{sat}$, which was also expected. Total soil N content was also negatively correlated ($P < 0.05$) with the slope parameter, which was the only significant correlation among the slope parameter and any soil property evaluated in this study.

Calcium ions have a relatively large hydrated radius (Zhang and Norton, 2002) and typically cause the opposite effect of Na (Amézketa, 1999). Calcium causes clay particles to flocculate, which limits the swelling of clays (Heil and Sposito, 1993). The flocculation of clay particles forms the basis for soil aggregation and structural stability (Six et al., 2000A). Therefore, increased amounts of Ca in a soil can result in increased aggregate stability and, as a result, increased infiltration (Quirk and Schofield, 1955; Amézketa, 1999). However, in contrast to that hypothesized, extractable soil Ca content in the top 10 cm was uncorrelated with overall infiltration rate ($P > 0.05$; Table 4), which may be related to the relatively large Ca contents and relatively low variability measured among the landuses (Table 3).

Antecedent soil moisture content in the top 6 cm and silt concentration and extractable soil Cu content in the top 10 cm were uncorrelated ($P > 0.05$) with overall infiltration rate, slope and intercept parameters, or estimated $K_{sat}$ (Table 4). In contrast to the results of this study, other
studies have shown that initial infiltration rate was impacted by the ASMC (Shukla et al., 2003; Sajjadi et al., 2016). Sajjadi et al. (2016) measured infiltration into soil cores with varying gravimetric moisture contents and discovered that, as the ASMC of soil cores decreased by 2.5%, the initial infiltration decreased by approximately 50%.

**Implications**

Identifying landuse and land management practices that increase SOM content and reduce bulk density may lead to greater surface water infiltration and, ultimately, aquifer recharge in the Delta region of eastern Arkansas in the LMRV. This could be accomplished by increasing the amount of surface residue left on the soil surface after a crop harvest, as well as facilitating the incorporation of that residue in soil. For example, in the CRP and native prairie landuses, this could be accomplished by increasing the amount of time between controlled burns, as well as eliminating the off-site transport of aboveground biomass and surface residue. In addition, any reduction in vehicular traffic across any landuse could reduce or prevent surface compaction and increase infiltration rates. Improving infiltration, hence enhancing the opportunity for groundwater recharge, will help maintain agricultural productivity and future sustainability of soil and water resources.

In addition to the current, major landuses evaluated in this study, future research should include additional landuses, such as pasturelands and haylands, which are also important landuses in the LMRV and comprise 14% of the total land cover (USDA-NRCS, 2013). Furthermore, pastureland and hayland occupy 73 million ha throughout the US (Sanderson et al., 2012). Pasturelands are a type of grassland used for animal forage production, which may be mowed or grazed (Allen et al., 2011). Pasturelands provide both economic and ecological
services (Sanderson et al., 2012). Unlike cultivated agroecosystems, pasturelands maintain year-round grass cover, as well as dense root systems, which may reduce runoff and erosion and increase infiltration (Sanderson et al., 2012). More research should be conducted on the effectiveness of converting highly erodible agricultural land to pastureland with the goal of improving surface water infiltration, soil quality, and, ultimately, groundwater recharge in the LMRV.

In order for groundwater recharge to occur, water must first infiltrate into the soil and flow downward through the soil profile, before eventually reaching the aquifer (Alley, 2009). During the 20-minute time period, or over the time it took for complete infiltration to occur if the time was less than 20 minutes, it was likely that the infiltrated water did not percolate deeper than approximately the top 10 cm of the soil profile. Therefore, it can be assumed that the overall infiltration rate and infiltration-related parameters measured in this study were mainly affected by near-surface rather than subsurface soil properties. Subsurface soil features, such as the argillic horizons in many of the soils of the LMRV, can affect the hydraulic conductivity of the soil and may reduce groundwater recharge rates (USGS, 2002; ANRC, 2017). However, the effect of subsurface soil properties was not measured in this study. Therefore, more research should be conducted on the effect of subsurface soil properties on hydraulic conductivity and groundwater recharge in the LMRV.

In addition, groundwater recharge rates can be impacted by the type of vegetation in a landuse (Acharya et al., 2018). Different types of plants have different root densities, ground cover, and biomass and, therefore, transpire water at different rates (Schlesinger and Jasechko, 2014). For instance, larger plants with more extensive roots systems typically have greater transpiration rates than plants with smaller or less extensive root systems (Acharya et al., 2018).
The amount of water that is absorbed by roots and removed from the soil through transpiration can affect the amount of water that is able to recharge groundwater (Acharya et al., 2018). As a result, conversion of land from an agroecosystem to a grassland or forest may also result in increased transpiration without a substantial increase in groundwater recharge (Acharya et al., 2018). Conservation efforts should consider the effects of land cover and transpiration on potential groundwater recharge. Additionally, more research should be conducted on the effect of transpiration rate, percolation, and groundwater recharge. Results of this study suggest that forest restoration, specifically initiation reforestation activities with deciduous tree species, of agricultural lands, rather than initiating grassland restorations, such as the CRP landuse evaluated in this study, may provide another potential alternative to increase infiltration and potential groundwater recharge, indirectly reducing soil erosion, in the LMRV. Deciduous forest restoration may provide more ecological benefits than grassland restoration activities in loessial and alluvial soils in the Delta region of eastern Arkansas. Consequently, further study on the broader impact of landuse on ecological health in the LMRV is advised. However, despite the fact that overall infiltration rates into the forested systems evaluated in this study were shown to be greater than other current, common landuses, the fate of water after infiltration is not well-understood. Considering that different tree species uptake and transpire water at different rates (Hacke et al., 2000), reforestation may increase the amount of water absorbed from the soil after infiltration and ultimately transpired (Alaoui et al., 2011), which may impact the amount of water that is able to percolate downward and eventually recharge groundwater supplies.
Conclusions

This field study evaluated the effects of major, current landuses on surface water infiltration and infiltration-related properties on fine-textured, loessial and alluvial soils in the Delta region of eastern Arkansas in the LMRV, which is presently plagued by groundwater depletion from extensive groundwater withdrawals for irrigated crop production. Results of this study showed that landuse and various land management practices on loessial and alluvial soils in the Delta region of eastern Arkansas significantly affected surface infiltration and infiltration-related parameters. Contrary to that hypothesized, the deciduous forest had the largest overall infiltration rate, while the remaining landuses (i.e., native prairie, coniferous forest, CRP, and CT and NT agriculture) had similar overall infiltration rates. A strong positive correlation was shown between overall infiltration rate and estimated K\text{sat}, indicating estimated K\text{sat} is related to infiltration rate and can be used to assess the impacts of landuse management practices, as well as overall soil quality. As hypothesized, the more-managed landuses (i.e., CRP grassland and CT and NT agroecosystems) generally exhibited lower intercept values, representing the theoretical infiltration rate at time zero, compared to the less-managed landuses (i.e., deciduous/coniferous forests and native prairie). Overall infiltration rate was positively correlated with SOM, TC, and TN contents, and C:SOM ratio, while negatively correlated with bulk density, and extractable soil Na and Mg contents in the top 10 cm. Results indicated that SOM content and soil bulk density are specific near-surface landuse characteristics that have a large effect on overall infiltration rate and other infiltration-related parameters. Identifying landuse management practices and soil surface properties that improve infiltration-related, soil-surface properties to allow more rainfall and/or irrigation water to infiltrate to potentially recharge the Alluvial Aquifer is a potentially effective way to increase Alluvial Aquifer recharge. Specifically,
restoration of highly erodible agricultural lands in the LMRV should target incorporation of deciduous species to improve infiltration capacity that can potentially contribute to potentially greater groundwater recharge.

Acknowledgements

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Literature Cited


## Appendix

Table 1. Summary of landuses, specific site information, dates visited for measurements, soil parent material, and the mapped soil taxonomic descriptions for research locations included in this study.

<table>
<thead>
<tr>
<th>Landuse</th>
<th>Specific Site</th>
<th>Location</th>
<th>DateMeasured</th>
<th>Soil Parent Material</th>
<th>Mapped Soil Series (Taxonomic Description)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Native Prairie</td>
<td>Seidenstricker Prairie</td>
<td>Arkansas County, Monroe County</td>
<td>23 March, 2016</td>
<td>Alluvium</td>
<td>Dewitt (fine, smectitic, thermic Typic Albaqualfs)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Stuttgart (fine, smectitic, thermic Albaquultic Hapludalfs)</td>
</tr>
<tr>
<td></td>
<td>Gray Prairie</td>
<td></td>
<td>23 March, 2016</td>
<td>Alluvium</td>
<td>Stuttgart (fine, smectitic, thermic Albaquultic Hapludalfs)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>6 July, 2016</td>
<td>Alluvium</td>
<td>Ethel (fine-silty, mixed, active, thermic Typic Glossaqualfs)</td>
</tr>
<tr>
<td></td>
<td>Roth 1</td>
<td>Arkansas County</td>
<td>6 July, 2016</td>
<td>Alluvium</td>
<td>Ethel (fine-silty, mixed, active, thermic Typic Glossaqualfs)</td>
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<tr>
<td></td>
<td>Roth 2</td>
<td>Arkansas County</td>
<td>6 July, 2016</td>
<td>Alluvium</td>
<td>Ethel (fine-silty, mixed, active, thermic Typic Glossaqualfs)</td>
</tr>
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<td></td>
<td>CRP</td>
<td></td>
<td>22 March, 2016</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
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<td></td>
<td>CRP 2</td>
<td>PTRS</td>
<td>25 May, 2016</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
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<td></td>
<td>CRP 3</td>
<td>PTRS</td>
<td>6 July, 2016</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
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<td>CRP 4</td>
<td>PTRS</td>
<td>6 July, 2016</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
</tr>
<tr>
<td>Deciduous Forest</td>
<td>Pecan Grove</td>
<td>LMRCS</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
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<td></td>
<td>Forest 1</td>
<td>PTRS</td>
<td>22 March, 2016</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
</tr>
<tr>
<td></td>
<td>Forest 2</td>
<td>PTRS</td>
<td>22 March, 2016</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
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<td>Forest 3</td>
<td>LMRCS</td>
<td>24 May, 2016</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
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<td>Coniferous Forest</td>
<td>Forest 1</td>
<td>PTRS</td>
<td>25 May, 2016</td>
<td>Loess</td>
<td>Loring (fine-silty, mixed, active, thermic Oxyaqua Fragiudalfs)</td>
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<td>Forest 2</td>
<td>PTRS</td>
<td>25 May, 2016</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
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<td></td>
<td>Forest 3</td>
<td>PTRS</td>
<td>25 May, 2016</td>
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<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
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<td>Forest 4</td>
<td>PTRS</td>
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<td>Loess</td>
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<td>Conventional-Tillage Agriculture</td>
<td>CT 1</td>
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<td>LMRCS</td>
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Table 1 (Cont.)

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<th>Date Measured</th>
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<th>Mapped Soil Series (Taxonomic Description)</th>
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<td>Loess</td>
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<td>thermic Aquic Fraglossudalfs)</td>
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<td>LMRCS</td>
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<td>Loess</td>
<td>Calloway (fine-silty, mixed, active,</td>
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<td>7 November, 2015</td>
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<td>Calloway (fine-silty, mixed, active,</td>
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<td>thermic Aquic Fraglossudalfs)</td>
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<td>NT 7</td>
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<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active,</td>
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<td>thermic Aquic Fraglossudalfs)</td>
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<tr>
<td>NT 8</td>
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<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active,</td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td>thermic Aquic Fraglossudalfs)</td>
</tr>
</tbody>
</table>

† CRP, Conservation Reserve Program
‡ PTRS, University of Arkansas System Division of Agriculture Pine Tree Research Station, near Colt, AR
§ LMCRS, University of Arkansas System Division of Agriculture Lon Mann Cotton Research Station, near Marianna, AR
§ CT, conventional tillage (CT 1: no-Burn, low residue, irrigated; CT 2: burned, low residue, irrigated; CT 3: no-Burn, low Residue, irrigated; CT 4: burned, high residue, irrigated; CT 5: burned, low residue, irrigated; CT 6: no-burn, high residue, irrigated; CT 7: no-burn, low residue, irrigated; CT 8: burned, high residue, non-irrigated; CT 9: burned, low residue, non-irrigated; CT 10: no-burn, high residue, non-irrigated; CT 11: no-burn, low residue, non-irrigated)
§§ NT, no-tillage (NT 1: burned, high residue, irrigated; NT 2: burned, low residue, irrigated; NT 3: no-burn, high residue, irrigated; NT 4: no-burn, low residue, irrigated; NT 5: burned, high residue, non-irrigated; NT 6: burned, low residue, non-irrigated; NT 7: no-burn, high residue, non-irrigated; NT 8: no-burn, low residue, non-irrigated)
Table 2. Summary of the 30-year (1981 to 2010) mean annual precipitation and air temperature, as well as the 30-year minimum and maximum monthly air temperatures for each location included in this study (NOAA, 2013).

<table>
<thead>
<tr>
<th>Landuse</th>
<th>Specific Site</th>
<th>Location</th>
<th>Precipitation (cm)</th>
<th>Mean Temp. (°C)</th>
<th>Min Temp. (°C)</th>
<th>Max Temp. (°C)</th>
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<tbody>
<tr>
<td>Native Prairie</td>
<td>Seidenstricker Prairie</td>
<td>Arkansas County</td>
<td>125.6</td>
<td>16.6</td>
<td>10.6</td>
<td>22.6</td>
</tr>
<tr>
<td></td>
<td>Gray Prairie</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Roth 1</td>
<td>Arkansas County</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Roth 2</td>
<td>Arkansas County</td>
<td></td>
<td></td>
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<td>CRP†</td>
<td>PTRS‡</td>
<td></td>
<td>128.5</td>
<td>16.6</td>
<td>11.2</td>
<td>22.1</td>
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<tr>
<td>Deciduous Forest</td>
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<tr>
<td></td>
<td>PTRS</td>
<td></td>
<td>128.5</td>
<td>16.6</td>
<td>11.2</td>
<td>22.1</td>
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<tr>
<td></td>
<td>LMCRS¶</td>
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<tr>
<td>Coniferous Forest</td>
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<td>PTRS</td>
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<td>128.5</td>
<td>16.6</td>
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<td>CT§</td>
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<tr>
<td></td>
<td>LMCRS</td>
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<td>128.5</td>
<td>16.6</td>
<td>11.2</td>
<td>22.1</td>
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<tr>
<td>NT§§</td>
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</tbody>
</table>

† CRP, Conservation Reserve Program
‡ PTRS, University of Arkansas System Division of Agriculture Pine Tree Research Station, near Colt, AR
¶ LMCRS, University of Arkansas System Division of Agriculture Lon Mann Cotton Research Station, near Marianna, AR
§ CT, conventional-tillage agriculture
§§ NT, no-tillage agriculture
Table 3. Summary of the effect of landuse on overall infiltration rate (OIR), slope and intercept parameters characterizing the linear relationship between the natural logarithm of the infiltration rate and the mid-point of time, estimated saturated hydraulic conductivity (K_{sat}), antecedent soil moisture content (ASMC) in the top 6 cm, and sand, silt, and clay concentrations, estimated bulk density (BD), soil organic matter (SOM) content, pH, EC, extractable soil nutrient (P, K, Ca, Mg, S, Na, Fe, Mn, Zn, and Cu) contents, total C (TC) and N (TN) contents, and C:N, N:SOM, and C:SOM ratios in the top 10 cm of various landuses [i.e., native prairie (NP), deciduous forest (DF), coniferous forest (CF), Conservation Reserve Program grassland (CRP), conventional-tillage agriculture (CT), and no-tillage agriculture (NT)] in the Delta region of eastern Arkansas.

<table>
<thead>
<tr>
<th>Variable</th>
<th>P</th>
<th>NP</th>
<th>DF</th>
<th>CF</th>
<th>CRP</th>
<th>NT</th>
<th>CT</th>
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</thead>
<tbody>
<tr>
<td>OIR (mm min^{-1})</td>
<td>&lt; 0.01</td>
<td>0.49 b</td>
<td>1.17 a</td>
<td>0.23 b</td>
<td>0.07 b</td>
<td>0.05 b</td>
<td>0.04 b</td>
</tr>
<tr>
<td>Slope</td>
<td>0.65</td>
<td>-0.11 a</td>
<td>-0.15 a</td>
<td>-0.10 a</td>
<td>-0.10 a</td>
<td>-0.07 a</td>
<td>-0.11 a</td>
</tr>
<tr>
<td>Intercept</td>
<td>&lt; 0.01</td>
<td>0.93 b</td>
<td>1.65 a</td>
<td>0.94 b</td>
<td>0.38 bc</td>
<td>0.05 c</td>
<td>-0.12 c</td>
</tr>
<tr>
<td>K_{sat} (mm hr^{-1})</td>
<td>&lt; 0.01</td>
<td>29.4 b</td>
<td>40.3 a</td>
<td>27.8 b</td>
<td>16.6 c</td>
<td>24.3 bc</td>
<td>23.3 bc</td>
</tr>
<tr>
<td>ASMC (cm^3 cm^{-3})</td>
<td>&lt; 0.01</td>
<td>0.25 ab</td>
<td>0.32 a</td>
<td>0.17 c</td>
<td>0.29 ab</td>
<td>0.22 bc</td>
<td>0.22 bc</td>
</tr>
<tr>
<td>Sand (g g^{-1})</td>
<td>&lt; 0.01</td>
<td>0.26 a</td>
<td>0.19 b</td>
<td>0.17 b</td>
<td>0.13 c</td>
<td>0.18 b</td>
<td>0.16 b</td>
</tr>
<tr>
<td>Silt (g g^{-1})</td>
<td>&lt; 0.01</td>
<td>0.64 d</td>
<td>0.73 abc</td>
<td>0.71 bc</td>
<td>0.70 c</td>
<td>0.76 a</td>
<td>0.75 ab</td>
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<tr>
<td>Clay (g g^{-1})</td>
<td>&lt; 0.01</td>
<td>0.10 bc</td>
<td>0.08 bc</td>
<td>0.11 b</td>
<td>0.17 a</td>
<td>0.07 c</td>
<td>0.09 bc</td>
</tr>
<tr>
<td>BD (g cm^{-3})</td>
<td>&lt; 0.01</td>
<td>1.34 ab</td>
<td>1.24 c</td>
<td>1.29 bc</td>
<td>1.35 ab</td>
<td>1.41 a</td>
<td>1.38 a</td>
</tr>
<tr>
<td>SOM (Mg ha^{-1})</td>
<td>&lt; 0.01</td>
<td>40.2 ab</td>
<td>45.1 a</td>
<td>43.3 a</td>
<td>33.0 c</td>
<td>32.7 c</td>
<td>35.3 bc</td>
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<tr>
<td>pH</td>
<td>&lt; 0.01</td>
<td>6.28 ab</td>
<td>5.60 c</td>
<td>5.61 c</td>
<td>5.71 bc</td>
<td>6.90 a</td>
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<tr>
<td>EC (dS m^{-1})</td>
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<td>0.09 a</td>
<td>0.10 a</td>
<td>0.06 b</td>
<td>0.06 b</td>
<td>0.11 a</td>
<td>0.10 a</td>
</tr>
<tr>
<td>P (kg ha^{-1})</td>
<td>&lt; 0.01</td>
<td>20.4 b</td>
<td>136 a</td>
<td>52.7 b</td>
<td>50.1 b</td>
<td>31.9 b</td>
<td>31.8 b</td>
</tr>
<tr>
<td>K (kg ha^{-1})</td>
<td>&lt; 0.01</td>
<td>93.1 c</td>
<td>159 a</td>
<td>140 ab</td>
<td>130 ab</td>
<td>117 bc</td>
<td>119 bc</td>
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<tr>
<td>Ca (kg ha^{-1})</td>
<td>0.02</td>
<td>1628 ab</td>
<td>1250 b</td>
<td>1369 b</td>
<td>1241 b</td>
<td>1894 a</td>
<td>1670 ab</td>
</tr>
<tr>
<td>Mg (kg ha^{-1})</td>
<td>0.01</td>
<td>268 b</td>
<td>200 b</td>
<td>278 b</td>
<td>241 b</td>
<td>438 a</td>
<td>430 a</td>
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<tr>
<td>S (kg ha^{-1})</td>
<td>0.01</td>
<td>21.8 a</td>
<td>22.0 a</td>
<td>19.9 a</td>
<td>20.1 a</td>
<td>13.7 b</td>
<td>19.3 a</td>
</tr>
<tr>
<td>Na (kg ha^{-1})</td>
<td>0.01</td>
<td>31.5 b</td>
<td>28.4 b</td>
<td>18.2 c</td>
<td>28.2 b</td>
<td>47.4 a</td>
<td>41.3 a</td>
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<tr>
<td>Fe (kg ha^{-1})</td>
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<td>334 a</td>
<td>333 a</td>
<td>324 ab</td>
<td>274 bc</td>
<td>197 d</td>
<td>254 c</td>
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<tr>
<td>Mn (kg ha^{-1})</td>
<td>0.01</td>
<td>350 a</td>
<td>340 a</td>
<td>393 a</td>
<td>316 a</td>
<td>207 b</td>
<td>314 a</td>
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<tr>
<td>Zn (kg ha^{-1})</td>
<td>0.01</td>
<td>2.4 b</td>
<td>5.8 a</td>
<td>5.5 a</td>
<td>2.4 b</td>
<td>2.2 b</td>
<td>2.7 b</td>
</tr>
<tr>
<td>Cu (kg ha^{-1})</td>
<td>0.77</td>
<td>1.5 a</td>
<td>1.8 a</td>
<td>2.0 a</td>
<td>2.2 a</td>
<td>1.7 a</td>
<td>1.7 a</td>
</tr>
<tr>
<td>TC (Mg ha^{-1})</td>
<td>&lt; 0.01</td>
<td>20.0 ab</td>
<td>22.6 a</td>
<td>21.2 a</td>
<td>14.3 c</td>
<td>15.0 c</td>
<td>16.2 bc</td>
</tr>
<tr>
<td>TN (Mg ha^{-1})</td>
<td>&lt; 0.01</td>
<td>1.41 ab</td>
<td>1.69 a</td>
<td>1.44 ab</td>
<td>1.06 c</td>
<td>1.25 bc</td>
<td>1.29 bc</td>
</tr>
<tr>
<td>C:N</td>
<td>&lt; 0.01</td>
<td>14.5 a</td>
<td>14.2 a</td>
<td>14.7 a</td>
<td>13.6 ab</td>
<td>12.0 c</td>
<td>12.6 bc</td>
</tr>
<tr>
<td>N:SOM</td>
<td>&lt; 0.01</td>
<td>0.033 bc</td>
<td>0.036 ab</td>
<td>0.033 bc</td>
<td>0.031 c</td>
<td>0.038 a</td>
<td>0.037 ab</td>
</tr>
<tr>
<td>C:SOM</td>
<td>0.01</td>
<td>0.48 a</td>
<td>0.49 a</td>
<td>0.49 a</td>
<td>0.42 b</td>
<td>0.46 ab</td>
<td>0.46 ab</td>
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</tbody>
</table>

† Means in a row with the same letter do not differ (P > 0.05).
Table 4. Summary of the linear correlations ($r$) between infiltration-related parameters [i.e., overall infiltration rate, slope and intercept parameters characterizing the linear relationship between the natural logarithm of the infiltration rate and the mid-point of time, and estimated saturated hydraulic conductivity ($K_{sat}$)] and soil physical and chemical properties [i.e., antecedent soil moisture content (ASMC) in the top 6 cm, and sand, silt, and clay concentrations, soil organic matter (SOM) content, estimated bulk density, pH, EC, extractable soil nutrient (P, K, Ca, Mg, S, Na, Fe, Mn, Zn, and Cu) contents, total C (TC) and N (TN) contents, and C:N, N:SOM, and C:SOM ratios] in the top 10 cm of various landuses in the Delta region of eastern Arkansas.

<table>
<thead>
<tr>
<th>Soil Property</th>
<th>Overall Infiltration Rate</th>
<th>Slope</th>
<th>Intercept</th>
<th>$K_{sat}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>ASMC</td>
<td>-0.15</td>
<td>0.19</td>
<td>-0.18</td>
<td>-0.02</td>
</tr>
<tr>
<td>Sand</td>
<td>0.05</td>
<td>0.11</td>
<td>-0.01</td>
<td>0.44***</td>
</tr>
<tr>
<td>Silt</td>
<td>-0.03</td>
<td>-0.06</td>
<td>-0.22</td>
<td>0.11</td>
</tr>
<tr>
<td>Clay</td>
<td>-0.01</td>
<td>-0.06</td>
<td>0.22*</td>
<td>-0.49***</td>
</tr>
<tr>
<td>SOM</td>
<td>0.47***</td>
<td>-0.17</td>
<td>0.40***</td>
<td>0.59***</td>
</tr>
<tr>
<td>Bulk density</td>
<td>-0.59***</td>
<td>0.19</td>
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* Significant ($P < 0.05$) correlation between variables
** Significant ($P < 0.01$) correlation between variables
*** Significant ($P < 0.001$) correlation between variables
Figure 1. Geographic distribution of the three native prairie sites included in this study located near Stuttgart, AR (Google Earth, 2019).
Figure 2. Geographic distribution of the 12 sites included in this study at the University of Arkansas System Division of Agriculture Pine Tree Research Station (PTRS) near Colt, AR representing four different landuses [i.e., conventional-tillage agriculture, Conservation Reserve Program (CRP) grassland, coniferous forest, and deciduous forest] (Google Earth, 2019).
Figure 3. Geographic distribution of four sites included in this study at the University of Arkansas System Division of Agriculture Lon Mann Cotton Research Station (LMCRS) near Marianna, AR representing three different landuses (i.e., conventional-tillage agriculture, no-tillage agriculture, and deciduous forest). The upper-left-most yellow star marks the location of the long-term, wheat-soybean double-crop field study whose 16 treatment combinations, replicated three times each, were included in this study (Google Earth, 2019).
Figure 4. Raw infiltration rates over time (A) and the natural logarithm of the infiltration rate over time (B) for the infiltration measurements conducted at the Roth Prairie. Different symbols on a panel represent different replicate infiltration measurements.
Chapter 3

Landuse and Soil Property Effects on Soil Aggregate Stability in the Lower Mississippi River Valley
Abstract

Surface water contamination and groundwater depletion are currently major environmental issues facing the Lower Mississippi River Valley (LMRV). Improving aggregate stability and, indirectly, reducing surface runoff and soil erosion may lead to multiple environmental benefits in the LMRV, including decreased contamination of surface waters by eroded sediments and sediment-bound nutrients as well as increased infiltration and enhanced opportunity for groundwater recharge. The objective of this study was to evaluate the effects of common landuses [i.e., native prairie, deciduous forest, coniferous forest, Conservation Reserve Program (CRP) grassland, and conventional-tillage (CT) and no-tillage (NT) agriculture], aggregate-size class and soil depth on aggregate-stability-related properties [i.e., water-stable macroaggregate (WSA) size distribution, total water-stable macroaggregate (TWSA) concentration, and mean weight diameter (MWD)] on fine-textured, loessial and alluvial soils in the Arkansas Delta region of the LMRV. When separated into five aggregate-size classes (i.e., 0.25- to 0.5-mm, 0.5- to 1.0-mm, 1.0- to 2.0-mm, 2.0- to 4.0-mm, and > 4.0-mm), WSA concentrations in the top 10 cm generally increased as aggregate size increased within the grassland and forested landuses, with the exception of in the deciduous forest. Total WSA concentrations in the top 10 cm in the native prairie, CRP, and coniferous forest were similar (P > 0.05) to each other, averaging 806 g kg⁻¹, and were 35% greater (P < 0.05) than that of the NT and CT agroecosystems, which did not differ and averaged 605 g kg⁻¹. The MWD in the top 10 cm in the native prairie, CRP, and coniferous forest were similar (P > 0.05) to each other, averaging 2.15 mm, and were 70% greater (P < 0.05) than that of the NT and CT agroecosystems, which did not differ and averaged 1.27 mm. In the top 5 cm, the MWD and TWSA concentration were 17 and 8% greater (P < 0.05), respectively, than that in the 5- to 10-
cm depth interval. Results of this study demonstrated that soil depth, landuse, and various landuse management practices significantly affect water-stable aggregation of loessial and alluvial soils in the Delta region of eastern Arkansas. Grassland and forest restoration, specifically of highly erodible agricultural land, should be considered to contribute to improvements in soil structural stability and the future sustainability of soil and water resources in the LMRV.
Introduction

The Lower Mississippi River Valley (LMRV) is one of the major river basins that comprise the Mississippi River Valley (USDA-NRCS, 2013). The LMRV covers over 250,000 km² of land and extends over parts of Arkansas, Louisiana, Mississippi, Missouri, Kentucky, and Tennessee (USDA-NRCS, 2013). Since European settlement of the region, the landscape of the LMRV has been greatly altered by agricultural expansion (USDA-NRCS, 2013). Whereas native forests and grasslands once dominated the landscape of the LMRV, cultivated agriculture now comprises approximately 33% of the total land area (Karstensen and Sayler, 2009; Welch et al., 2009; Oswalt, 2013). Many studies have shown that the conversion of native landuses to agricultural land causes overall soil quality to decline and soil erodibility to increase (Cambardella and Elliott, 1993; Martens, 2000; Six et al., 2000; Bodhinayake and Si, 2004; Harper et al., 2008).

Loess, one of the dominant soil parent materials of the LMRV (Lindbo et al., 1994), is particularly susceptible to water erosion and has been shown to rapidly erode when the soil surface is exposed (Fu, 1989). Loessial soils are mainly comprised of silt-sized soil particles that are easily transported by wind and water (Fu, 1989). Silt particles do not have the adhering properties of clay, and therefore, tend to be attracted to other soil particles and organic matter to a lesser degree than clay particles. This can result in silt particles dislodging from the soil surface and being transported relatively easily (Amézketa, 1999). Consequently, loessial soils in the LMRV can be extremely susceptible to erosion. As a result, soil erosion and contamination of surface waters by eroded sediments and sediment-bound nutrients are major environmental issues currently impacting the LMRV (USDA-NRCS, 2013). For instance, in 2013, almost one billion hectares of land in the LMRV were classified as highly erodible (USDA-NRCS, 2013).
addition, approximately 64 billion kg of sediment, 754 million kg of N, and 73 million kg of P are annually transported to rivers and streams within the LMRV (USDA-NRCS, 2013). Consequently, keeping excess sediment and potential aquatic pollutants in place on the terrestrial landscape is critical to reduce erosion and maintain water quality.

Aggregate stability is one of the most important factors that control soil erosion and surface runoff (Kemper and Rosenau, 1986; Graham et al., 1995). Soil aggregates are defined as secondary soil particles that are held together by soil organic matter (SOM), clay, or other cohesive materials (USDA-NRCS, 2014). Consequently, aggregate stability is the degree to which soil aggregates can withstand disintegrating forces, including weathering, precipitation, and erosion (Kemper and Rosenau, 1986; Smith et al., 2014) and is an important indicator of soil structural stability and overall soil quality (Six et al., 2000). Generally, decreased aggregate stability causes decreased soil pore continuity (Tisdall and Oades, 1982), which can lead to decreased infiltration and increased runoff and erosion (Le Bissonnais and Arrouays, 1997; Franzluebbers, 2002; Wuest et al., 2005; Stone and Schlegel, 2010; Alaoui et al., 2011).

Stable soil aggregates can be classified as macroaggregates (> 0.25 mm) or microaggregates (< 0.25 mm; Oades, 1984). Soils typically have a hierarchical organization in which macroaggregates are formed through the cohesion of microaggregates (Tisdall and Oades, 1982; Puget et al., 2000). Generally, a greater abundance of macro- than microaggregates in a soil is an indication of greater overall soil quality (Oades, 1984). Macroaggregates are essential for root growth, hydraulic conductivity, and the ability of the soil to resist erosion (USDA-NRCS, 2008). Another method of measuring aggregate stability is mean weight diameter (MWD), which is an index used to compare the size of aggregates among soils, by taking into account the concentrations of aggregates in varying size classes (Chaney and Swift, 1984).
Aggregate stability, and soil aggregation in general, can be affected by many different soil physical and chemical properties including SOM, clay, fungal hyphae, root biomass, and earthworm activity (Amézketa, 1999). Clay particles and SOM are important binding agents for aggregate formation and can improve aggregate stability (Tisdall and Oades, 1982; Chaney and Swift, 1984; Elliot, 1986; Lado et al., 1992; Amézketa, 1999). Roots and fungal hyphae are also major stabilizing agents of soil aggregates (Tisdall and Oades, 1982), and, through the production of polysaccharides, can bind smaller into larger aggregates (Wang et al., 2010). Even after death, fungal hyphae and roots can continue to stabilize soil aggregates during decomposition (Oades and Waters, 1991; Amézketa, 1999). The movement or burrowing of soil fauna, especially earthworms, through the soil can facilitate the formation of macropores and macroaggregates (Hillel, 2004).

Aggregate stability can also be affected by landuse or agricultural management practices, such as vehicle traffic, burning, mowing, grazing, and tillage (Chan et al., 2002; Jarvis, 2007; Malhi and Kutcher, 2007; Bai et al., 2009; Smith et al., 2014). Vehicle traffic can cause soil compaction and reduce porosity and aggregation (Meek et al., 1992; Jarvis, 2007; Bai et al., 2009). Burning, grazing, and mowing management practices decrease the return of organic matter to the soil and can, therefore, negatively impact aggregate stability (Chan et al., 2002; Malhi and Kutcher, 2007; USDA-NRCS, 2008; Smith et al., 2014). Tillage for agricultural purposes can physically break apart soil aggregates, sever fungal hyphae, increase SOM decomposition rate, and result in decreased aggregate stability (Jastrow et al., 1998; Six et al., 2000; Kasper et al., 2009; Stone and Schlegel, 2010; Bhattacharyya et al., 2013).

Agricultural conservation methods, such as no-tillage (NT) agriculture, have been shown to reduce soil disturbance and lead to increased aggregate stability compared to conventional-
tillage (CT) agriculture (Cambardella and Elliott, 1993; Franzluebbers, 2002; Malhi and Kutcher, 2007; Wang et al., 2010; Bhattacharyya et al., 2013). Native, undisturbed landuses, such as native prairies or forests, which have never been subject to tillage or agricultural production, also tend to have soils with greater aggregate stability than soils used for CT agricultural production (Cambardella and Elliott, 1993; Martens, 2000; Bodhinayake and Si, 2004). Organic matter, root density, soil biota, and aggregate stability generally increase over time in soils that remain undisturbed by tillage (Chaney and Swift, 1984; Jastrow, 1996; Martens, 2000; Harper et al., 2008).

Many studies have shown that converting agricultural land to perennial grasslands or forests can lead to increased aggregate stability and decreased erosion (Lindstrom et al., 1994; Six et al., 2000; Huang et al., 2002; Evrendilek et al., 2004; An et al., 2010). The Conservation Reserve Program (CRP), created by the Farm Service Agency in 1985, seeks to convert highly erodible agricultural land to more natural, perennial vegetation in an effort to decrease erosion and improve overall soil quality in targeted areas on the landscape (Lindstrom et al., 1994; Huang et al., 2002). Over time, grassland restorations often begin to exhibit characteristics more similar to a native prairie than a cultivated agroecosystem (Jastrow, 1996; Huang et al., 2002; Udawatta et al., 2008).

Improvements in aggregate stability may indirectly lead to increased infiltration and groundwater recharge in the LMRV (USGS, 2002; ANRC, 2017). The Alluvial Aquifer, the largest and shallowest aquifer located in the LMRV, is currently plagued by groundwater depletion from extensive withdrawals for irrigated agricultural production (USGS, 2002; ANRC, 2017). These withdrawals have lowered aquifer levels and depleted groundwater reserves (USGS, 2002; Verkler et al., 2008). Consequently, in order to avoid completely irreversible
groundwater depletion of the Alluvial Aquifer, withdrawal rates must be greatly reduced, or
recharge rates must be greatly increased in the near future (USGS, 2002).

Identifying landuse and management practices that improve aggregate stability could help maintain the future sustainability of soil and water resources of the LMRV (Harper et al., 2008; Smith et al., 2014). Improvements in aggregate stability can indirectly increase infiltration and reduce surface runoff and erosion (Le Bissonnais and Arrouays, 1997; Franzluebbers, 2002; Bhattacharyya et al., 2008; Stone and Schlegel, 2010) and may lead to multiple environmental benefits in the LMRV, including decreased contamination of surface waters by eroded sediments and sediment-bound nutrients as well as enhanced opportunity for groundwater recharge (USGS, 2002; USDA-NRCS, 2013). Therefore, the main objective of this study was to evaluate the effects of common landuses [i.e., native prairie, deciduous forest, coniferous forest, CRP grassland, and CT and NT agriculture], aggregate-size class, and soil depth on aggregate-stability-related properties [i.e., water-stable macroaggregate (WSA) size distribution, total water-stable macroaggregate (TWSA) concentration, and MWD] on fine-textured, loessial and alluvial soils in the Arkansas Delta region of the LMRV. The secondary objective of this study was to evaluate the relationship among aggregate-stability-related properties, infiltration-related parameters, and various soil physical and chemical properties.

Landuse management practices that result in soil disturbance (i.e., tillage) were hypothesized to decrease the overall concentration of TWSA and impede the formation of aggregates with larger diameters. Therefore, it was hypothesized, that more naturally maintained landuses (i.e., grasslands and forests) would exhibit greater TWSA concentrations and MWD as well as a more well-developed aggregate hierarchy, with WSA concentrations increasing with increasing aggregate-size class. In contrast, it was hypothesized that the more intensely managed
landuses (i.e., agroecosystems) would exhibit smaller TWSA concentrations and MWD as well as a less well-developed aggregate hierarchy, with WSA concentrations decreasing with increasing aggregate-size class. In particular, it was hypothesized that the TWSA concentration, MWD, and aggregate hierarchy development would be greatest in the native prairie landuse because uncultivated grassland sites would, collectively, be the most naturally maintained and lowest in the CT agroecosystem because annually tilled sites would be subjected to the greatest degree of soil disturbance. Overall, it was hypothesized that, based on the intensity of landuse management practices disturbing the soil system, landuses and their associated aggregate stability would rank according to the following sequence: native prairie > deciduous forest > coniferous forest > CRP grassland > NT agriculture > CT agriculture. It was also hypothesized that aggregate-stability-related properties would differ between soil depths (i.e., 0- to 5- and 5- to 10-cm). Since soil biotic activity and organic matter contents are typically greater nearer to the surface, the top 5 cm of soil was hypothesized to exhibit a greater TWSA concentration, MWD, and aggregate hierarchy development than the 5- to 10-cm depth interval. Finally, it was hypothesized that TWSA concentration and MWD in the top 10 cm would be correlated with infiltration-related properties (i.e., overall infiltration rate and slope and intercept parameters characterizing the linear relationship between the natural logarithm of the infiltration rate and mid-point of time) and soil physical and chemical properties [i.e., sand, silt, and clay concentrations, SOM content, estimated bulk density, total C (TC) and N (TN) contents, and C:N, N:SOM, and C:SOM ratios] in the top 10 cm.
Materials and Methods

Site Descriptions

This study included six landuses [i.e., native prairie, deciduous forest, coniferous forest, CRP grassland, and CT and NT agriculture] common in the Delta region of eastern Arkansas in the LMRV. Table 1 summarizes specific site information, dates visited for measurements, soil parent materials, and the mapped soil taxonomic descriptions for each landuse evaluated in this field study. Table 2 summarizes soil physical and chemical properties in the top 10 cm of the landuses included in this study. Details of the soil sample collection, processing, and analyses are given in Anderson (2019).

Three native prairies (i.e., Kenneth Gray Prairie, Seidenstricker Prairie, and Roth Prairie) were included in this study (Table 1; Figure 1). The Kenneth Gray Prairie is located northeast of Stuttgart, AR (34°39'15" N, 91°24'47" W), the Seidenstricker Prairie is located north of Stuttgart, AR (34°43'44" N, 91°33'17" W), and the Roth Prairie is located southwest of Stuttgart, AR (34°27'15" N, 91°34'37" W). These native tallgrass prairie sites are some of the few remnants of the Grand Prairie in eastern Arkansas that have been completely preserved from agricultural influence and soil disturbance, with the exception of periodic burning and occasional vehicle traffic (Brye and Pirani, 2005). Natural prairie mounds are present at all three native prairie sites and provide further evidence that the prairies have never been cultivated and that the soil has remained in its natural, undisturbed state (ANHC, 2013).

At the time soil samples were collected, which is described in detail below, all three prairies had been burned within the previous six months. However, under normal vegetative conditions, major species present at these native tallgrasses include big bluestem (*Andropogon gerardi*), little bluestem (*Schizachyrium scoparium* [Michx.] Nash), switchgrass (*Panicum
virgatum), Indiangrass (*Sorghastrum nutans*), and many forbs including purple coneflower
(*Echinacea purpurea* (L.) Moench), black-eyed susan (*Rudbeckia hirta* L.), and goldenrod
(*Solidago* spp.). The Gray Prairie contained a single soil series (Stuttgart silt loam; fine,
smectitic, thermic Albaquultic Hapludalfs; USDA-NRCS, 2017), which is in a udic soil moisture
regime. The Seidenstricker Prairie contained a single soil series (Dewitt silt loam; fine, smectitic,
thermic Typic Albaqualfs; USDA-NRCS, 2017), which is in an aquic soil moisture regime. The
Roth Prairie contained two soil series (i.e., Stuttgart silt loam and Ethel silt loam). The Ethel soil
series (fine-silty, mixed, active, thermic Typic Glossaqualfs) is located in an aquic soil moisture
regime (USDA-NRCS, 2017).

Additional landuse sites were located at the University of Arkansas System Division of
Agriculture Pine Tree Research Station (PTRS) near Colt, AR (35°7'10.54" N, 90°45'51.56" W;
Table 1; Figure 2). The PTRS included four CRP-grassland sites, which had not been cultivated
for at least 10 years and had been only minimally disturbed by periodic mowing and removal of
above-ground biomass (Figure 2). Each CRP grassland was previously used for decades for
cultivated agriculture; however, the areas were only marginally agriculturally productive for
various reasons and have subsequently been converted to managed grassland in order to increase
soil quality. Major grass species present in the CRP grasslands included big bluestem,
switchgrass, and Indiangrass. Three different soil series [Calloway silt loam (fine-silty, mixed,
active, thermic Aquic Fraglossudalfs), Zachary silt loam (fine-silty, mixed, active, thermic Typic
Albaqualfs), and Henry silt loam (coarse-silty, mixed, active, thermic Typic Fragiaqualfs);
USDA-NRCS, 2017] were mapped throughout the four CRP grasslands. The Calloway silt loam
is in a udic, while the Zachary and Henry silt loams are in an aquic soil moisture regime.
The PTRS included four coniferous forest plantation sites that had previously been cleared and used for cultivated agriculture in the past (Figure 2). One of the plantation sites was planted to loblolly pine (*Pinus taeda*) at least 25 years ago, with an approximate row spacing of 3 m and an approximate tree spacing of 3 m. The other three plantation sites were planted to loblolly pine, with an approximate row spacing of 3 m and an approximate tree spacing of 3 m, at least 12 years ago. As of March 2016, the loblolly pine trees in the three young plantations varied in height between 10- and 15-m tall, while those in the older plantation varied in height from 20- to 25-m tall. The three young plantations had a surface pine needle residue layer that was approximately 3-cm thick, while the surface pine needle residue layer of the older plantation was approximately 0.5-cm thick. Two soil series were mapped among the three young plantations, Calloway silt loam and Loring silt loam (fine-silty, mixed, active, thermic Oxyaquic Fragiudalfs), which are present in a udic soil moisture regime (USDA-NRCS, 2017), while Calloway silt loam is mapped throughout the older plantation.

The PTRS included two deciduous forest sites (Figure 2). These deciduous forest areas had likely been cleared and used for cultivated agriculture in the past. Major tree species present at the two deciduous forest parcels at PTRS were oak (*Quercus* spp.), hickory (*Carya* spp.), gum (*Eucalyptus* spp.), and dogwood (*Cornus* spp.). However, based on the size of the trees (i.e., approximately 20- to 30-m tall), trees naturally regenerated and have not been disturbed for at least 30 years. Calloway silt loam was mapped at one site, while Calhoun silt loam (fine-silty, mixed, active, thermic Typic Glossaqualfs), which is in an aquic soil moisture regime (USDA-NRCS, 2017), was mapped at the second deciduous forest site at PTRS. The PTRS also included two CT agricultural sites in two different soil series (i.e., Calloway and Calhoun silt loams; Figure 2). The CT agricultural sites at the PTRS were fallow at the time of sampling, but had
been managed in a rice (*Oryza sativa* L.)-soybean (*Glycine max* L.) rotation for at least the past five years.

Additional landuse sites were located at the University of Arkansas System Division of Agriculture Lon Mann Cotton Research Station (LMCRS) near Marianna, AR (34°44’2.26” N, 90°45’51.56” W; Table 1; Figure 3). Two deciduous forest sites were at the LMCRS. One deciduous forest site at LMCRS was a historically managed pecan (*Carya illinoinensis*) grove on a Calloway silt-loam soil, but has not been actively managed for pecan production for at least 15 years, other than periodic annual mowing for aesthetic purposes. The second deciduous forest site at LMRCS was a well-established deciduous forest stand on a Memphis silt loam (fine-silty, mixed, active, thermic Typic Hapludalfs), which is in a udic soil moisture regime (USDA-NRCS, 2017), with growth indicating multiple decades of undisturbed development. Similar to the PTRS deciduous forest sites, the major tree species present were oak, hickory, gum, and dogwood. It is likely that the area had been cleared and used for cultivated agriculture in the past as well; however, based on the size of the trees (i.e., approximately 15- to 20-m tall), the forest parcel has been allowed to naturally regenerate for at least 40 years.

Eight NT and eight CT agricultural sites were included in long-term, wheat (*Triticum aestivum*)-soybean (*Glycine max* L.), double-crop production system field study located at LMRCS (Figure 3). This field study was initiated in 2001 on a Calloway silt loam and was comprised of CT and NT treatments in a factorial combination with a wheat-residue-level (i.e., low and high achieved with differential N fertilization), residue burning (i.e., burning and non-burning), and irrigation (i.e., irrigated and dryland) treatments (Cordell et al., 2006). The tillage, residue-level, and burn field treatments have been in place since 2002, while the irrigation treatment has been in place since 2005. In total, there were 48, 3-m wide by 6.1-m long plots,
with three replications of each tillage-residue-level-burn-irrigation treatment combination. Each set of three replications of each treatment combination were used for the purposes of this study to represent an individual site for the CT and NT agricultural landuses. Cordell et al. (2006) and Smith et al. (2014) provided detailed descriptions of this long-term field study. An additional CT agricultural site on a Memphis silt loam was located at LMRCS adjacent to the one deciduous forest site (Figure 3), which was bare, with no vegetation at the time of sampling, but had been cropped to monoculture soybean the previous fall.

Throughout the region encompassing all study sites, the 30-year (1981-2010) mean annual air temperature was 16.6°C and the 30-year mean annual minimum and maximum air temperature ranged from 10.6 to 11.2°C and from 22.1 to 22.6°C, respectively (NOAA, 2013; Table 3). The 30-year mean annual precipitation ranged from 125.6 to 128.5 cm throughout the region encompassing the study sites (NOAA, 2013; Table 3).

**Aggregate Stability**

Soil samples were collected for aggregate stability assessment from all sites on 19 and 20 March, 2017. Three soil cores were collected from the top 10 cm of each site at each site/landuse or one core per small plot in each agricultural landuse in the long-term, wheat-soybean, double-crop field study using a 7.3-diameter, stainless steel core chamber and slide hammer. Each of the three cores per site/agricultural treatment combination were separated into the 0- to 5- and 5- to 10-cm depth intervals. The separated 0- to 5- and 5- to 10-cm portions were combined to create one composite sample per depth per site/treatment combination. The composite samples were gently hand-crushed, pushed through a 6-mm mesh screen, and then air-dried for approximately 7 days.
An aggregate-stability analysis, similar to the procedure described by Yoder (1936), Kemper and Rosenau (1986), and Smith et al. (2014), was performed using approximately 200 g of air-dried soil. A mechanical wet-sieving apparatus, filled with tap water, was used to oscillate soil aggregates in water at 30 cycles per minute for 5 minutes and allowed to pass through a stack of progressively smaller sieves (i.e., 4.0-, 2.0-, 1.0-, 0.5-, and 0.25-mm mesh sizes). After the oscillating process, the remaining soil aggregates on each mesh screen were washed from the sieves into small aluminum trays, dried in a forced-draft oven at 70°C for 24 hours, and weighed separately by aggregate-size class (i.e., 0.25- to 0.5-mm, 0.5- to 1.0-mm, 1.0- to 2.0-mm, 2.0- to 4.0-mm, and > 4.0-mm). The wet-sieving procedure was performed twice using sub-samples from each composite soil sample per site/agricultural treatment combination. The mass of the soil in each aggregate size class was divided by the mass of the original sample to calculate the WSA fractions. The MWD was calculated for each aggregate-stability analysis performed (Kemper and Rosenau, 1986; Stone and Schlegel, 2010). The MWD was equal to the sum of the WSA fraction in each aggregate-size class multiplied by the mean diameter of the particle-size range within that size class (i.e., 5.0-, 3.0-, 1.5, 0.75, and 0.125-mm). The dry mass of soil aggregates retained on each of the individual sieves was summed and then divided by the original sample mass to calculate the TWSA.

**Statistical Analyses**

A three-factor analysis of variance (ANOVA) was conducted using the PROC MIXED procedure in SAS (version 9.4, SAS Institute, Inc., Cary, NC) to analyze the effects of landuse, soil depth, aggregate-size class, and their interactions on WSA concentrations using a split-split plot design, in which landuse was the whole plot, soil depth was the first split-plot, and
aggregate-size class was the second split-plot. The whole-plot portion of the study was a completely random design. Additional two-factor ANOVAs were conducted using the PROC MIXED procedure in SAS to analyze the effects of landuse, soil depth, and their interactions on TWSA concentration and MWD using a split-split plot design, in which landuse was the whole plot and soil depth was the split-plot. When appropriate, means were separated by least significant difference (LSD) at the 0.05 level.

Linear correlation analyses were conducted using Minitab (version 18, Minitab, Inc., State College, PA) to analyze the relationship between TWSA concentration and MWD in the top 10 cm and infiltration-related parameters (i.e., overall infiltration rate and the slope and intercept parameters characterizing the linear relationship between the natural logarithm of the infiltration rate and the mid-point of time) and various soil physical and chemical properties (i.e., sand, silt, and clay concentrations, SOM content, estimated bulk density, TC and TN contents, and C:N, N:SOM, and C:SOM ratios in the top 10 cm.) Significance was judged at the $P < 0.05$ level.

Results and Discussion

Water-stable Aggregation

Soil erosion and contamination of surface waters by eroded sediments and sediment-bound nutrients are major environmental issues currently facing the LMRV (USDA-NRCS, 2013). For instance, on an annual basis, approximately 64 billion kg of sediment, 754 million kg of N, and 73 million kg of P are transported to rivers and streams within the LMRV (USDA-NRCS, 2013). Improvements in soil structural stability, particularly WSA formation, can lead to decreased surface runoff and soil erosion (Six et al., 2002; Harper et al., 2008; Blanco and Lal,
Therefore, it is important to identify and understand soil properties and management practices that can affect WSA formation and aggregate-size distribution of soils within the LMRV.

Aggregate-size Distribution

According to the model of aggregate hierarchy presented by Tisdall and Oades (1982), macroaggregates are formed through the cohesion of microaggregates by various organic and inorganic binding agents (i.e., clay, SOM, plant roots, and fungal hyphae). When separated into five aggregate-size classes (i.e., 0.25- to 0.5-mm, 0.5- to 1.0-mm, 1.0- to 2.0-mm, 2.0- to 4.0-mm, and > 4.0-mm), WSA concentration (g aggregates kg⁻¹ soil) differed by landuse among aggregate-size classes (P < 0.01) and differed among size classes between soil depths (P < 0.01) and (Table 4).

Among all landuse-size-class combinations, averaged across soil depth, WSA concentration was greatest (P < 0.05) in the 2.0- to 4.0-mm size class in the native prairie (284 g kg⁻¹) and numerically lowest in the 0.25- to 0.5-mm size class in the native prairie (68.0 g kg⁻¹; Figure 4). Within grassland and forested landuses, WSA concentrations generally increased as aggregate size increased, with the exception of in the deciduous forest (Figure 4). In the native prairie, WSA concentrations in the two largest (2.0- to 4.0-mm and > 4.0-mm) aggregate-size classes averaged 243 g kg⁻¹ and were 159% greater (P < 0.05) than in the two smallest (0.25- to 0.5-mm and 0.5- to 1.0-mm) aggregate-size classes, which averaged 94 g kg⁻¹ (Figure 4). In addition, in the CRP grassland, WSA concentrations in the two largest (2.0- to 4.0-mm and > 4.0-mm) aggregate-size classes averaged 215 g kg⁻¹ and were 115% greater (P < 0.05) than in the two smallest (0.25- to 0.5-mm and 0.5- to 1.0-mm) aggregate-size classes, which averaged 100 g
kg$^{-1}$ (Figure 4). Furthermore, in the coniferous forest, WSA concentrations in the two largest (2.0- to 4.0-mm and > 4.0-mm) aggregate-size classes averaged 205 g kg$^{-1}$ and were 85% greater ($P < 0.05$) than in the two smallest (0.25- to 0.5-mm and 0.5- to 1.0-mm) aggregate-size classes, which averaged 111 g kg$^{-1}$ (Figure 4). The relatively large proportion of $> 2$-mm-diameter WSA is to be expected in naturally maintained landuses because, as land remains undisturbed over a period of time, SOM, earthworm activity, and fungal hyphae generally increase (Cambardella and Elliott, 1993; Lindstrom et al., 1994; Jastrow, 1996; Li et al., 2013; Archer et al., 2015) and, as a result, the size and stability of macroaggregates in the soil may also increase (Tisdall and Oades, 1982; Chaney and Swift, 1984; Elliott, 1986; Cambardella and Elliott, 1993; Martens, 2000; Wuest et al., 2005). In addition, grasslands and forests develop extensive root systems over time that bind soil aggregates together and promote the formation of macroaggregates (Tisdall and Oades, 1982; Oades and Waters, 1991). Similar to the results of this study, Celik (2005) measured WSA size distribution among grasslands and coniferous forests and reported greater concentrations of large ($> 2$-mm) than small ($< 1$-mm) WSA in both landuses in the top 10 cm of clay and silty-clay-loam (Typic Haploxerolls) soils in southern Mediterranean Turkey.

In contrast to that hypothesized, averaged across soil depths, the coniferous forest had relatively greater ($P < 0.05$) concentrations of large WSA ($> 2$-mm), but similar ($P > 0.05$) concentrations of small WSA (0.25- to 0.5-mm) compared to that of the deciduous forest (Figure 4). Soil aggregate hierarchy was expected to be more developed in the deciduous than in the coniferous forest because the deciduous forest sites were older (i.e., $\geq 30$ years old) than the coniferous forest sites ($\geq 12$ years old). Generally, as the age, or time since establishment, of a forest increases, the size and abundance of WSA will increase (Li et al., 2013; Archer et al., 2015). In addition, deciduous forest litter is base-cation rich and has been shown to promote an
abundance of earthworms and fungal hyphae (Terhivuo, 1989; Ste-Marie and Paré, 1999; Reich et al., 2005) and facilitate WSA formation (Graham et al., 1995). The general decrease in WSA concentration with decreasing size class in the coniferous forest may be a result of decreasing SOC stability with decreasing size class that has been shown to be present in coniferous forests (Fang et al., 2015). In contrast, SOC stability has been shown to be relatively constant throughout all aggregate-size classes in deciduous forests (Fang et al., 2015). However, Fang et al. (2015) identified a lack of literature pertaining to the effects of forest vegetation on soil aggregate hierarchy and suggested these effects should continue to be examined in further research.

In contrast to the grasslands and forests, averaged across soil depths, WSA concentrations within the smallest (0.25- to 0.5-mm) aggregate-size class in the NT (149 g kg\(^{-1}\)) and CT agriculture (134 g kg\(^{-1}\)) were 22 and 26% numerically greater, respectively, than WSA concentrations within the four larger aggregate-size classes (0.5- to 1.0-mm, 1.0- to 2.0-mm, 2.0- to 4.0-mm, and > 4.0-mm), which averaged 122 g kg\(^{-1}\) and 106 g kg\(^{-1}\), respectively (Figure 4). The relatively large proportion of small (0.25- to 0.5-mm) WSA in the agroecosystems, which were frequently subjected to soil disturbance by agricultural management practices, suggests that large aggregates are more fragile than smaller aggregates. Agricultural management practices, including various soil disturbances (i.e., tillage, vehicle traffic, and compaction), have been shown to increase SOM decomposition rate, sever plant roots and fungal hyphae, and cause the physical breakdown of macro- into microaggregates (Tisdall and Oades, 1982; Cambardella and Elliott, 1993; Shukla et al., 2003; Kasper et al., 2009; Smith et al., 2014).

As hypothesized, averaged across soil depth, the native prairie, CRP, and coniferous forest had greater \((P < 0.05)\) concentrations of large WSA (> 2.0 mm) than the NT and CT

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agroecosystems, whereas the NT and CT agroecosystems had at least numerically greater WSA concentrations in the smallest aggregate-size class (0.25- to 0.5-mm) compared to that of the native prairie, CRP, and coniferous forest (Figure 4). These results suggest that agricultural management practices caused the breakdown of larger into smaller WSA and resulted in fewer WSA in the NT and CT agroecosystems than in the less-managed (i.e., grassland and forest) landuses. Similar to the results of this study, Jastrow et al. (1996) reported a greater abundance of large (> 1.0 mm) WSA and less small (< 0.5 mm) WSA in native-vegetation grasslands compared to CT agroecosystems in the top 10 cm of silt-loam and silty-clay-loam soils near Chicago, IL.

In contrast to that hypothesized, averaged across soil depths, the NT and CT agroecosystems exhibited similar aggregate-size distributions to each other. The WSA concentration in each aggregate-size class under NT was similar ($P > 0.05$) to that in the same aggregate-size class under CT agricultural landuse (Figure 4). However, many studies have reported a greater abundance of large WSA and fewer small WSA and microaggregates in NT compared to CT agroecosystems (Cambardella and Elliott, 1993; Bossuyt et al., 2002; Malhi and Kutcher, 2007; Stone and Schlegel, 2010; Wang et al., 2010). Bhattacharyya et al. (2012) showed that continuous NT agriculture resulted in greater concentrations of large WSA (> 2.0 mm) and lower concentrations of small (0.25- to 2.0 mm) WSA compared to continuous CT agriculture in the top 5 cm of sandy-clay-loam (Typic Haplaquept) soils in the Indian Himalayas. Bhattacharyya et al. (2012) attributed the improved aggregation under NT to greater amounts of intra-aggregate SOM and, in particular, SOC.

When separated into five aggregate-size classes and averaged across landuse, WSA concentrations generally increased as aggregate-size class increased in both depth intervals
In the top 5 cm, WSA concentrations were greatest ($P < 0.05$) in the two largest (2.0- to 4.0-mm and > 4.0-mm) size classes and lowest ($P < 0.05$) in the smallest (0.25- to 0.5-mm) size class (Figure 5). In the 5- to 10-cm depth interval, WSA concentrations were at least numerically greater in the > 4.0-mm (144 g kg$^{-1}$) and 2.0- to 4.0-mm (164 g kg$^{-1}$) size classes than in the other size classes, which did not differ ($P > 0.05$) from each other (Figure 5).

Among all depth-size-class combinations, WSA concentrations of larger (> 1.0 mm) aggregate-size classes were greater ($P < 0.05$) in the top 5 cm than in the 5- to 10-cm depth interval (Figure 5). However, in the 5- to 10-cm depth, the WSA concentration of the smallest (0.25- to 0.5-mm) aggregate-size class was greater ($P < 0.05$) than that in the top 5 cm (Figure 5). The relative abundance of large WSA and fewer small WSA in the top 5 cm likely resulted from greater SOM contents and biotic activity that is typical of near-surface soil (Anders et al., 2010). Similar to the results of this study, Bossuyt et al. (2002) reported water-stable aggregate (> 53 μm) percentages generally increased with increasing aggregate-size class in the top 5 cm of soils, whereas water-stable aggregate percentages were similar among all aggregate-size classes in the 5- to 15-cm depths in agroecosystems on Hiwassee sandy-clay-loam (Rhodic Kanhapludults) soils in the southern Appalachian Mountains.

**Mean Weight Diameter**

Aggregate MWD (mm-diameter) is a common index used to compare soil structure among soils by taking into account the concentrations of varying aggregate-size classes in a soil (Chaney and Swift, 1984). Greater concentrations of large aggregates result in a larger MWD and indicates greater aggregate stability (Chaney and Swift, 1984). In the current study, MWD was independently affected by both landuse ($P < 0.01$) and soil depth ($P < 0.01$; Table 4).
Averaged across soil depths, MWD differed among the six landuses included in this study (Table 4). The MWD of the native prairie, CRP, and coniferous forest were similar ($P > 0.05$) to each other, averaging 2.15 mm, and were 68% greater than that of the deciduous forest and NT and CT agroecosystems, which did not differ and averaged 1.28 mm (Figure 6). In addition, the forests and grasslands generally exhibited greater SOM and TC contents as well as lower bulk densities than the agroecosystems (Table 2). Forests and grasslands typically contain more extensive and developed root systems than agricultural soils (Aubertin, 1971; Alaoui et al., 2011). Plant roots are able to hold large macroaggregates together and can create large macropores for water to flow through (Tisdall and Oades, 1982; Alaoui et al., 2011). Therefore, the forests and grasslands were expected to be more structurally stable and exhibit a greater MWD than that of the more intensely managed agroecosystems. Many studies have reported greater MWD in grasslands or forests compared to agroecosystems (Martens, 2000; Shukla et al., 2003; Evrendilek et al., 2004; Zheng et al., 2004; Celik, 2005). Similar to the results of this study, Evrendilek et al. (2004) reported the MWD of grasslands and coniferous forests was 276 and 238% greater, respectively, than that of a continuous barley (Hordeum vulgare L.) and wheat agroecosystem in the top 10 cm of clay and silty-clay-loam (Typic Haploxerolls) soils in the southern Mediterranean region of Turkey. In addition, the grasslands and forests had greater SOM and SOC contents as well as lower bulk densities compared to the agroecosystem (Evrendilek et al., 2004).

In contrast to that hypothesized, averaged across soil depth, the MWD of the NT and CT agroecosystems were similar ($P > 0.05$) to each other (Figure 6). Although non-significant, aggregate MWD was 6% greater in the NT (1.30 mm) than in the CT agroecosystem (1.23 mm; Figure 6). Many studies have shown that NT can result in greater MWD than CT agricultural
management practices (Malhi and Kutcher, 2007; Virto et al., 2007; Stone and Schlegel, 2010; Bhattacharyya et al., 2013). Malhi and Kutcher (2007) reported NT resulted in a 61% increase in MWD compared to CT agriculture in the top 5 cm of sandy-loam and clay-loam soils in Saskatchewan, Canada.

Similar MWD between NT and CT agroecosystems may have been the result of the interactive effects of multiple agricultural practices (i.e., irrigation, residue burning, differential residue amounts, and fertilizer treatments) among the agroecosystems included in this study (Cordell et al., 2006). The interaction of agricultural management practices can have a greater effect on water-stable aggregation than tillage practices alone (Mikha and Rice, 2004). Smith et al. (2014) demonstrated that combinations of irrigation and residue level/fertilizer treatments significantly affected WSA formation in many of the same NT and CT treatments included in this study.

In contrast to that hypothesized, averaged across soil depths, MWD was greater ($P < 0.05$) in the coniferous (2.06 mm) than in the deciduous forest (1.32 mm) (Figure 6). The deciduous forest was expected to exhibit a greater MWD because the age, or time since establishment, of the deciduous forest sites (i.e., ≥ 30 years) was greater than that of the coniferous forest sites (≥ 12 years old). As the age of a forest increases, the size and stability of soil aggregates generally increase (Li et al., 2013; Quintero and Comerford, 2013). In addition, deciduous forest soils have been shown to contain a greater abundance of soil microbes, bacteria, and earthworms than coniferous forest soils (Graham et al., 1995; Frey et al., 2004; Li et al., 2013), presumably due to the generally more alkaline and nutrient-rich upper soil profile as a result of decomposition of the generally more nutrient-rich deciduous leaf litter, all of which may lead to greater WSA formation and MWD (Graham et al., 1995; Jarvis, 2007). Soils with larger
divalent cation concentrations tend to promote flocculation followed by aggregation than soils with lower divalent cation concentration, which commonly occur between deciduous and coniferous forests (Graham et al., 1995; Li et al., 2013). However, coniferous forest soils have been shown to have greater SOC stability than deciduous forest soils, due to the accumulation of needle litter and relatively slow decomposition rate that is typical of coniferous forest soils (Frey et al., 2004; Berg, 2014). Increased SOC stability can also facilitate WSA formation and lead to greater MWD (Fang et al., 2015). Moreover, Fang et al. (2015) identified a lack of literature pertaining to the effects of WSA formation under forest landuse and suggested these effects continue to be studied in further research.

Averaged across landuse, MWD also differed ($P < 0.01$) between soil depths (Table 4). The MWD in the top 5 cm (1.85 mm) was 17% greater than that in the 5- to 10-cm depth interval (1.58 mm), which may have resulted from the greater SOM contents and biotic activity that is typical of near-surface soils (Anders et al., 2010). Similar to the results of this study, Bhattacharyya et al. (2008) reported the MWD in the top 15 cm was greater than that in the 15- to 30-cm depth interval among rice (*Oryza sativa* L.) and wheat agroecosystems on sandy-clay-loam (Typic Haplaquepts) soils in the Indian Himalayas.

Total Water-stable Macroaggregation

Total WSA concentration (> 0.25-mm) was determined as the sum of the WSA concentrations in all aggregate-size classes within a soil (Smith et al., 2014). Similar to MWD, TWSA concentration in the top 10 cm was independently affected by landuse ($P < 0.01$) and soil depth ($P < 0.01$) (Table 4).
Averaged across soil depths, TWSA concentrations differed among the six landuses included in this study (Figure 7). Specifically, TWSA concentrations in the top 10 cm were similar among the native prairie, coniferous forest, and CRP, averaging 806 g kg\(^{-1}\), and were 45% greater than that under CT agriculture (Figure 7). Total WSA concentrations in the top 10 cm were similar \((P > 0.05)\) among the coniferous and deciduous forests, CRP, and NT agroecosystems and numerically smaller than that of the native prairie (Figure 7). In addition, TWSA concentrations were also similar \((P > 0.05)\) among the deciduous forest and NT and CT agroecosystems and numerically smaller than in the other three landuses (Figure 7).

As hypothesized, TWSA concentration in the top 10 cm of the native prairie (864 g kg\(^{-1}\)) was at least numerically greater than that in all other landuses included in this study (Figure 7). Native prairies, which have never been disturbed by tillage or agricultural production, have been shown to exhibit greater aggregate stability than more intensely managed landuses (Cambardella and Elliott, 1993; Martens, 2000; Schwartz et al., 2003; Bodhinayake and Si, 2004). In addition, averaged across soil depths, the TWSA concentration of the CRP (768 g kg\(^{-1}\)) was similar to that of the native prairie (Figure 7). These results suggest that, after at least 10 years under CRP-grassland management, the aggregate stability of formerly cultivated agroecosystems was assumed to improve to levels comparable to that of a native prairie and greater than that of current agroecosystems. Over time, the SOM content, earthworm activity, and aggregate stability of grassland soils has been shown to increase (Chaney and Swift, 1984; Jastrow, 1996; Martens, 2000; Bodhinayake and Si, 2004). Grasses typically have dense root systems that promote soil aggregation (Tisdall and Oades, 1982; Elliott, 1986). Grass root systems develop over time, and even after death, still function to bind aggregates together (Oades and Waters, 1991). Jastrow (1996) measured TWSA (> 0.21 mm) percentage in the top 10 cm of soil in a native prairie and
chronosequence of native-vegetation grasslands on silt-loam and silty-clay-loam soils near Chicago, IL. As the age, or time since cultivation, of a grassland increased, TWSA percentage increased exponentially \( (r^2 = 0.99) \), indicating a strong, direct relationship between the two variables (Jastrow, 1996). In the same grassland chronosequence, Jastrow et al. (1998) reported that fine root length, fungal hyphae length, and SOC also generally increased as grassland age increased, all of which can contribute to soil aggregation. Averaged across soil depths, the TWSA concentrations of the native prairie \( (864 \text{ g kg}^{-1}) \), coniferous forest \( (786 \text{ g kg}^{-1}) \), and CRP \( (768 \text{ g kg}^{-1}) \) were 55, 36, and 24% greater \( (P < 0.05) \), respectively, than that of the CT agroecosystem \( (557 \text{ g kg}^{-1}) \), which had the numerically lowest TWSA concentration among all landuses included in this study (Figure 7). Furthermore, the native prairie and coniferous forest had at least numerically greater SOM and TC contents in the top 10 cm compared to the CT agroecosystem (Table 2). The decreased TWSA in the CT agroecosystem may be attributed to the negative effects of tillage, including physical breakdown of aggregates as well as SOM and C losses (Tisdall and Oades, 1982). Many studies have reported greater TWSA in grasslands and forests than CT agroecosystems (Cambardella and Elliot, 1993; Six et al., 2000; Huang et al., 2002; Celik, 2005). Similar to the results of this study, Harper et al. (2008) reported greater TWSA \( (> 0.25 \text{ mm}) \) and TC content as well as a decreased bulk density in the top 15 cm of a native prairie compared to a CT agroecosystem on silt-loam (Oxyaquic Glossudalfs) soils in the Mississippi River Delta region of eastern Arkansas. Celik (2005) reported greater TWSA \( (> 0.5 \text{ mm}) \) in the top 10 cm of coniferous forests and grasslands than CT agroecosystems on clay and silty-clay-loam (Typic Haploxerolls) soils in southern Mediterranean Turkey.

In contrast to that hypothesized, averaged across soil depths, TWSA concentrations in the NT and CT agroecosystems were similar \( (P > 0.05) \) to each other (Figure 7). Tillage for
agricultural purposes has been shown to increase SOM decomposition, disrupt plant roots, fungal hyphae, and the soil pore network, and result in decreased aggregate stability (Cambardella and Elliott, 1993; Six et al., 2000; Bossuyt et al., 2002; Franzluebbers, 2002; Bhattacharyya et al., 2012, 2013). Franzluebbers (2002) reported greater TWSA (> 0.25 mm) concentrations in the top 9 cm under long-term NT compared to CT on sandy-loam (Typic Kanhapludult) soils near Watkinsville, GA. In addition, the SOC content of the top 12 cm was approximately double in the NT than in the CT agroecosystems (Franzluebbers, 2002).

Although non-significant, the TWSA concentration was 14% numerically greater under NT (634 g kg\(^{-1}\)) than CT (557 g kg\(^{-1}\)) agriculture in the top 10 cm (Figure 7). Even after 15 years under continuous NT management, the expected long-term effects of NT on soil aggregation may still require more time to become evident. Similar to the results of this study, Motschenbacher et al. (2011) and Smith et al. (2014) measured TWSA concentrations under NT and CT agroecosystems and did not consistently show significant differences between NT and CT, even after > 9 years of consistent management.

Averaged across landuse, TWSA concentration also differed \((P < 0.01)\) between soil depths (Table 4). Total WSA concentration was 8% greater in the top 5 cm (734 g kg\(^{-1}\)) than in the 5- to 10-cm depth interval (677 g kg\(^{-1}\)). Generally, SOM and biotic activity is greatest in near-surface soils and decreases at lower soils depths (Anders et al., 2010). Soil biota produce important binding agents for aggregate formation (Hillel, 2004). For instance, earthworms, plant roots, fungal hyphae, and soil microorganisms, particularly bacteria and fungi, produce sticky organic substances than can bind soil particles and small aggregates to one another (Hillel, 2004). In addition, the movement of plant roots and burrowing of soil animals (i.e., earthworms and insects) through the soil can facilitate the formation of macropores and macroaggregates.
(Hillel, 2004). As a result, surface soils typically have greater macroaggregation than subsurface soils (Anders et al., 2010; Bhattacharyya et al., 2013; Smith et al., 2014). Similar to the results of this study, Motschenbacher et al. (2011) reported greater TWSA (> 0.25 mm) concentrations in the top 5 cm than in the 5- to 10-cm depth interval under NT and CT landuse in a Dewitt silt loam (Aquic Fraglossudalf) in the LMRV Delta region of eastern Arkansas.

Aggregate Stability Correlations with Soil Properties and Infiltration-related Parameters

Linear correlations were conducted to examine the relationships among MWD and TWSA concentration in the top 10 cm, infiltration-related parameters (i.e., overall infiltration rate and slope and intercept parameters characterizing the linear relationship between the natural logarithm of the infiltration rate and mid-point of time), and various soil physical and chemical properties (i.e., sand, silt, and clay concentration, SOM content, estimated bulk density, TC and TN contents, and C:N, N:SOM, and C:SOM ratios) in the top 10 cm (Table 5). These analyses were performed to identify factors that may strongly influence WSA formation under various commons landuses on the loessial and alluvial soils of the LMRV portion of eastern Arkansas. Since aggregate stability is one of the most sensitive soil properties for assessing the impacts of landuse management practices as well as improvements in overall soil quality (Bodhinayake and Si, 2004; Blanco and Lal, 2010; Stone and Schlegel, 2010), identifying soil physical and/or chemical properties that are related to water-stable aggregation may allow for determination and implementation of future methods of improving aggregate stability and soil quality in areas, such as the LMRV Delta region of eastern Arkansas, that have experienced surface erosion and groundwater depletion due to the lack of infiltration capacity.
Total WSA concentration and MWD are important properties related to aggregate stability (Franzluebbers, 2002). As would be expected, across all landuse-soil depth combinations, TWSA concentration and MWD in the top 10 cm were strongly, positively correlated \((r = 0.92; P < 0.001; \text{Table 5})\). However, few significant correlations were identified among TWSA concentration, MWD, and infiltration-related parameters or various soil physical and chemical properties (Table 5).

Silt concentration (g g\(^{-1}\)) was negatively correlated with TWSA \((r = -0.36; P < 0.05)\) and MWD \((r = -0.49; P < 0.01)\). One explanation for this is that soils that contain large amounts of silt and relatively low amounts of SOM, such as those included in this study (Table 2), predominate the Arkansas Delta region of the LMRV (DeLong et al., 2003; Smith et al., 2014) and tend to be susceptible to slaking and dispersion, which may hinder WSA formation (Lal and Shukla, 2004; USDA-NRCS, 2008). Slaking is the process by which the sudden saturation of an aggregate with water can cause air to become entrapped and compressed within the aggregate-pore network and result in many small explosions that release the entrapped air, and, ultimately, cause the complete disintegration of the aggregate (Hillel, 2004; Lal and Shukla, 2004). Furthermore, the processes of slaking in silty soils may be exaggerated by wet-sieving laboratory procedures, in which soils are initially air-dried before being sieved, such as those used in this study (USDA-NRCS, 2008). As a result, the aggregate stability of soils with large silt contents, and especially those with small SOM contents, may be relatively low (Lal and Shukla, 2004; USDA-NRCS, 2008).

In contrast to that hypothesized, SOM and clay concentration were uncorrelated with TWSA concentration or MWD (Table 5), which may be related to the relatively low variability of SOM contents and clay concentrations measured among the landuses (Table 2). Soil organic
matter and clay particles are binding agents that can facilitate soil aggregate formation (Amézketa, 1999). According to the hierarchical scheme proposed by Tisdall and Oades (1982), flocculated clay particles form the basis of soil aggregates and SOM assists in binding particles and aggregates together. It is well-documented that increases in SOM and/or clay result in improved aggregate stability (Tisdall and Oades, 1982; Chaney and Swift, 1984; Elliot, 1986; Lado et al., 1992; Amézketa, 1999). Evrendilek et al. (2004) reported a strong, positive correlation ($r = 0.82; P < 0.001$) between MWD and SOM content in the top 20 cm across multiple landuses (i.e., cropland, forests, grasslands) on silty-clay-loam and clay (Typic Haploxerolls) soils in the southern Mediterranean region of Turkey.

Total C content was also uncorrelated with TWSA or MWD, which was unexpected (Table 5). In contrast to the results of this study, many studies have reported that greater C concentrations resulted in increased aggregate stability (Six et al., 2000; Wuest et al., 2005; Stone and Schlegel, 2010; Bhattacharyya et al., 2012). Soil organic C, which is a component of TC, and characterizes all the TC measured in this study due to the lack of inorganic C presence in all soils sampled in this study, has been shown to actively stabilize aggregates (Amézketa, 1999; Evrendilek et al., 2004). Therefore, increased TC likely results in increased SOC and can result in improved soil aggregation (Franzluebbers, 2002; Wuest et al., 2005; Stone and Schlegel, 2010; Fang et al., 2015).

Similar to SOM, clay, and TC, in contrast to that hypothesized, overall infiltration rate was also uncorrelated with TWSA concentration or MWD (Table 5). However, many studies have shown increased infiltration resulting from increased aggregate stability (Le Bissonnais and Arrouays, 1997; Franzluebbers, 2002; Shukla et al., 2003; Bhattacharyya et al., 2008; Stone and Schlegel, 2010). Generally, as the stability of soil aggregates increase, the size and continuity of
soil pores, through which water can flow, also increase (Tisdall and Oades, 1982), resulting in increased infiltration and decreased runoff and erosion (Le Bissonnais and Arrouays, 1997; Franzluebbers, 2002; Shukla et al., 2003; Bhattacharyya et al., 2008; Alaoui et al., 2011). In contrast, disintegration of unstable aggregates can clog soil pores and reduce infiltration capacity (Harper et al., 2008). Stone and Schlegel (2010) reported steady-state infiltration rate and MWD in the top 10 cm were strongly, positively correlated \((r = 0.91)\) across multiple landuses on silt-loam (Aridic Argiustolls) soils in western Kansas. Similarly, Le Bissonnais and Arrouays (1997) reported cumulative infiltration rate and MWD in the top 30 cm were strongly, positively correlated \((r = 0.97)\) among agroecosystems on loamy (Vermic Haplumbrepts) soils in the French Pyrenean piedmont in southwest France. In fact, aggregate stability and infiltration rate were so well correlated that Le Bissonnais and Arrouays (1997) suggested using aggregate stability as a predictor for infiltration, particularly because aggregate stability was much less difficult to measure than infiltration. The lack of significant correlations among aggregate-stability-related properties and various soil physical, chemical, and hydraulic properties measured in this study may be attributed to the narrow range of soil physical and chemical properties characterizing the soils of the landuses evaluated in this study (Table 2), which was likely due to the similarities in soil parent material and soil texture among the landuses (Table 1) that were specifically targeted for this study.

Despite the deciduous forest having the greatest \((P < 0.05)\) overall infiltration rate (Table 2), the deciduous forest exhibited a relatively low TWSA concentration \((\text{g kg}^{-1})\) and MWD among the six landuses (Figure 6). One explanation for the greater infiltration in the deciduous forest may be a greater presence of macropores compared to the other landuses. Forested soils typically have an abundance of macropores that form from decomposed macroroots and biotic
activity (Bharati et al., 2002). Preferential flow can occur through these macropores and along
the edges of macroroots and improve the infiltration capacity and hydraulic conductivity of
forested soils (Aubertin, 1971; Bharati et al, 2002; Alaoui et al., 2011). Bharati et al. (2002)
reported 60-minute cumulative infiltration was approximately 2.5-fold greater in deciduous
forests than switchgrass grasslands on Coland loam and sandy loam (fine-loamy, mixed
superactive, mesic Cumulic Endoaquoll) soils in central Iowa. The greater infiltration into
forested than grassland soils was attributed to a greater presence of macropores in the deciduous
forest (Bharati et al., 2002).

**Implications**

Surface water contamination and groundwater depletion are currently major
environmental issues facing the LMRV (USDA-NRCS, 2013). Consequently, identifying
landuse and landuse management practices that improve soil structural stability could help
maintain agricultural productivity and future sustainability of soil and water resources in the
LMRV. Improving aggregate stability, and indirectly reducing surface runoff and soil erosion,
may lead to multiple environmental benefits in the LMRV, including decreased contamination of
surface waters by eroded sediments and sediment-bound pollutants, as well as increased
infiltration and enhanced opportunity for groundwater recharge. Future research needs to be
conducted on additional methods of improving aggregate stability and decreasing the risk of soil
erosion in the LMRV. In addition, future research should continue to examine the relationship
between aggregate stability and infiltration in soils of the LMRV.

The results of this study indicated that grasslands and forests generally had greater
aggregate stability than agroecosystems on fine-textured, loessial and alluvial soils in the
Arkansas Delta region of the LMRV. However, agricultural production, which is worth over $9 billion annually, is currently a necessary practice in the LMRV and cannot be eliminated (USDA-NRCS, 2013). Therefore, identifying combinations of landuse attributes and agricultural management practices that can lead to increased aggregate stability, while still maintaining sufficient crop quality and yield, may be necessary to address the environmental issues currently facing the LMRV. Currently, there is insufficient literature on the interactive effects of various agricultural management practices (i.e., irrigation, burning, crop residue level, and fertilizer treatments) on soil quality properties, particularly aggregate stability (Smith et al., 2014). In addition to the effects of tillage evaluated in this study, future research should be conducted on the interactive effects of agricultural management practices, which can vary widely in the LMRV and may significantly impact soil structural stability (USDA-NRCS, 2013).

Additionally, agricultural conservation methods that may improve aggregate stability and reduce erosion, such as manure application and the use of cover crops, should be identified and implemented in the LMRV. For instance, only approximately 1% of cropland in the LMRV has cover crops included in the crop rotation, despite model simulations suggesting that the adoption of cover crops in the LMRV could reduce sediment loss to nearby surface waters by as much as 70% (USDA-NRCS, 2013). In addition, although applications of manure to agricultural soils have been shown to improve aggregate stability, especially in combinations with NT practices (Shukla et al., 2003; Mikha and Rice, 2004), manure applications are made to only approximately 1% of the cropland area in the LMRV (USDA-NRCS, 2013).

The results of this study suggest that grassland and forest restoration may improve aggregate stability and, indirectly, decrease the risk of soil erosion and surface water sedimentation and contamination in the LMRV. Specifically, CRP-grassland restoration efforts
appeared to be successful at improving water-stable aggregation of fine-textured, loessial and alluvial soils in the Arkansas Delta region of the LMRV. Currently, over nine million hectares of land, 47% of which are classified as highly erodible, are enrolled in the CRP program in the LMRV (USDA-NRCS, 2013). In addition, approximately 12% of cropland in the LMRV is classified as highly erodible (USDA-NRCS, 2013). Grassland and forest restoration, especially on agricultural land classified as highly erodible, should continue to be considered in order to improve aggregate stability and decrease soil erosion in the LMRV.

Conclusions

This field study evaluated the effects of major landuses and soil depth on aggregate-stability in fine-textured, loessial and alluvial soils in the Arkansas Delta region of the LMRV. Results of this study suggest that landuse and various landuse management practices significantly affect water-stable aggregation of loessial and alluvial soils in the Delta region of eastern Arkansas. As hypothesized, the less-managed landuses (i.e., grasslands and forests) generally exhibited greater TWSA concentrations and MWD than current agroecosystems, with the exception of the deciduous forest. The TWSA concentration and MWD of the CRP grassland was similar to that of the native prairie and at least numerically greater than the agroecosystems, indicating that CRP-management practices were effective in improving aggregate stability of former agroecosystems to levels comparable to that of a native prairie and greater than that of current agroecosystems. The NT and CT agroecosystems exhibited similar aggregate-size distributions, TWSA concentrations, and MWD to each other. The long-term benefits of NT, even after 16 years, were not yet evident in this study. Based on the results of this study, implementation of NT practices alone may be insufficient to improve soil structure and decrease
the risk of soil erosion in fine-textured, loessial and alluvial soils in the Arkansas Delta region of the LMRV. Results indicated that soil in the top 5 cm had greater aggregate stability than soil in the 5- to 10-cm depth in fine-textured, loessial and alluvial soils of the Arkansas Delta region of the LMRV. Identifying landuse management practices and soil surface properties that improve water-stable aggregation may decrease the risk of soil erosion and contamination of surface waters by eroded sediment and sediment-bound nutrients. Grassland and forest restoration, specifically of highly erodible agricultural land should continue to be considered to contribute to improvements in soil structural stability and the future sustainability of soil and water resources in the LMRV.

**Acknowledgement**

This research project was funded by the Arkansas Natural Resource Conservation Service. Field and laboratory assistance from Johan Desrochers, Marya McKee, Casey Rector, and Matt Thompson are gratefully acknowledged.
Literature Cited


Archer, N., W. Otten, S. Schmidt, A. Bengough, N. Shah, and M. Bonell. 2015. Rainfall infiltration and soil hydrological characteristics below ancient forest, planted forest and grassland in a temperate northern climate. Ecohydrol. 9:585-600.


Appendix

Table 1. Summary of landuses, specific site information, dates visited for measurements, soil parent material, and the mapped soil taxonomic descriptions for research locations included in this study.

<table>
<thead>
<tr>
<th>Landuse</th>
<th>Specific Site</th>
<th>Location</th>
<th>Date Measured</th>
<th>Soil Parent Material</th>
<th>Mapped Soil Series (Taxonomic Description)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Native Prairie</td>
<td>Seidenstricker Prairie</td>
<td>Arkansas County</td>
<td>23 March, 2016</td>
<td>Alluvium</td>
<td>Dewitt (fine, smectitic, thermic Typic Albaqualfs)</td>
</tr>
<tr>
<td></td>
<td>Gray Prairie</td>
<td>Monroe County</td>
<td>23 March, 2016</td>
<td>Alluvium</td>
<td>Stuttgart (fine, smectitic, thermic Albaqualtic Hapludalfs)</td>
</tr>
<tr>
<td></td>
<td>Roth 1</td>
<td>Arkansas County</td>
<td>6 July, 2016</td>
<td>Alluvium</td>
<td>Stuttgart (fine, smectitic, thermic Albaqualtic Hapludalfs)</td>
</tr>
<tr>
<td></td>
<td>Roth 2</td>
<td>Arkansas County</td>
<td>6 July, 2016</td>
<td>Alluvium</td>
<td>Ethel (fine-silty, mixed, active, thermic Typic Glossaqualfs)</td>
</tr>
<tr>
<td>CRP</td>
<td>CRP† 1</td>
<td>PTRS‡</td>
<td>22 March, 2016</td>
<td>Loess</td>
<td>Henry (coarse-silty, mixed, active, thermic Typic Fragiqualfs)</td>
</tr>
<tr>
<td></td>
<td>CRP 2</td>
<td>PTRS</td>
<td>25 May, 2016</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
</tr>
<tr>
<td></td>
<td>CRP 3</td>
<td>PTRS</td>
<td>6 July, 2016</td>
<td>Alluvium</td>
<td>Zachary (fine-silty, mixed, active, thermic Typic Albaqualfs)</td>
</tr>
<tr>
<td></td>
<td>CRP 4</td>
<td>PTRS</td>
<td>6 July, 2016</td>
<td>Loess</td>
<td>Henry (coarse-silty, mixed, active, thermic Typic Fragiqualfs)</td>
</tr>
<tr>
<td>Deciduous Forest</td>
<td>Pecan Grove</td>
<td>LMRC§</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
</tr>
<tr>
<td></td>
<td>Forest 1</td>
<td>PTRS</td>
<td>22 March, 2016</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
</tr>
<tr>
<td></td>
<td>Forest 2</td>
<td>PTRS</td>
<td>22 March, 2016</td>
<td>Loess</td>
<td>Calhoun (fine-silty, mixed, active, thermic Typic Glossaqualfs)</td>
</tr>
<tr>
<td></td>
<td>Forest 3</td>
<td>LMRC§</td>
<td>24 May, 2016</td>
<td>Loess</td>
<td>Memphis (fine-silty, mixed, active, thermic Typic Hapludalfs)</td>
</tr>
<tr>
<td>Coniferous Forest</td>
<td>Forest 1</td>
<td>PTRS</td>
<td>25 May, 2016</td>
<td>Loess</td>
<td>Loring (fine-silty, mixed, active, thermic Oxyaqua Fragiudalfs)</td>
</tr>
<tr>
<td></td>
<td>Forest 2</td>
<td>PTRS</td>
<td>25 May, 2016</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
</tr>
<tr>
<td></td>
<td>Forest 3</td>
<td>PTRS</td>
<td>25 May, 2016</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
</tr>
<tr>
<td></td>
<td>Forest 4</td>
<td>PTRS</td>
<td>25 May, 2016</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
</tr>
<tr>
<td>Conventional-Tillage Agriculture</td>
<td>CT§ 1</td>
<td>LMRC§</td>
<td>24 May, 2016</td>
<td>Loess</td>
<td>Memphis (fine-silty, mixed, active, thermic Typic Hapludalfs)</td>
</tr>
<tr>
<td></td>
<td>CT 2</td>
<td>PTRS</td>
<td>22 March, 2016</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
</tr>
<tr>
<td></td>
<td>CT 3</td>
<td>PTRS</td>
<td>22 March, 2016</td>
<td>Loess</td>
<td>Calhoun (fine-silty, mixed, active, thermic Typic Glossaqualfs)</td>
</tr>
<tr>
<td></td>
<td>CT 4</td>
<td>LMRC§</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
</tr>
<tr>
<td></td>
<td>CT 5</td>
<td>LMRC§</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
</tr>
<tr>
<td></td>
<td>CT 6</td>
<td>LMRC§</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
</tr>
<tr>
<td></td>
<td>CT 7</td>
<td>LMRC§</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
</tr>
</tbody>
</table>
Table 1 (Cont.)

<table>
<thead>
<tr>
<th>Landuse</th>
<th>Specific Site</th>
<th>Location</th>
<th>Date Measured</th>
<th>Soil Parent Material</th>
<th>Mapped Soil Series (Taxonomic Description)</th>
</tr>
</thead>
<tbody>
<tr>
<td>CT 8</td>
<td>LMRCS</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
<td></td>
</tr>
<tr>
<td>CT 9</td>
<td>LMRCS</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
<td></td>
</tr>
<tr>
<td>CT 10</td>
<td>LMRCS</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
<td></td>
</tr>
<tr>
<td>CT 11</td>
<td>LMRCS</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
<td></td>
</tr>
<tr>
<td>No-tillage</td>
<td>Agriculture</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NT§§ 1</td>
<td>LMRCS</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
<td></td>
</tr>
<tr>
<td>NT 2</td>
<td>LMRCS</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
<td></td>
</tr>
<tr>
<td>NT 3</td>
<td>LMRCS</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
<td></td>
</tr>
<tr>
<td>NT 4</td>
<td>LMRCS</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
<td></td>
</tr>
<tr>
<td>NT 5</td>
<td>LMRCS</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
<td></td>
</tr>
<tr>
<td>NT 6</td>
<td>LMRCS</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
<td></td>
</tr>
<tr>
<td>NT 7</td>
<td>LMRCS</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
<td></td>
</tr>
<tr>
<td>NT 8</td>
<td>LMRCS</td>
<td>7 November, 2015</td>
<td>Loess</td>
<td>Calloway (fine-silty, mixed, active, thermic Aquic Fraglossudalfs)</td>
<td></td>
</tr>
</tbody>
</table>

† CRP, Conservation Reserve Program
‡ PTRS, University of Arkansas System Division of Agriculture Pine Tree Research Station, near Colt, AR
§ LMCRS, University of Arkansas System Division of Agriculture Lon Mann Cotton Research Station, near Marianna, AR
§§ NT, no-tillage (NT 1: burned, high residue, irrigated; NT 2: burned, high residue, irrigated; NT 3: no-burn, high residue, irrigated; NT 4: no-burn, low residue, irrigated; NT 5: burned, high residue, non-irrigated; NT 6: burned, low residue, non-irrigated; NT 7: burned, high residue, non-irrigated; NT 8: no-burn, low residue, non-irrigated)
Table 2. Summary of the effect of landuse on sand, silt, and clay concentrations, estimated bulk density (BD), soil organic matter (SOM) and total carbon (TC) contents in the top 10 cm, and overall infiltration rate (OIR) of various landuses [i.e., native prairie (NP), deciduous forest (DF), coniferous forest (CF), Conservation Reserve Program grassland (CRP), conventional-tillage agriculture (CT), and no-tillage agriculture (NT)] in the Delta region of eastern Arkansas. Data and results are reproduced from Anderson (2019).

<table>
<thead>
<tr>
<th>Landuse</th>
<th>Sand (g g⁻¹)</th>
<th>Silt (g g⁻¹)</th>
<th>Clay (g g⁻¹)</th>
<th>BD (g cm⁻³)</th>
<th>SOM (Mg ha⁻¹)</th>
<th>TC (Mg ha⁻¹)</th>
<th>OIR (mm min⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>NP</td>
<td>0.26 a</td>
<td>0.64 d</td>
<td>0.10 bc</td>
<td>1.34 ab</td>
<td>40.2 ab</td>
<td>20.0 ab</td>
<td>0.49 b</td>
</tr>
<tr>
<td>DF</td>
<td>0.19 b</td>
<td>0.73 abc</td>
<td>0.08 bc</td>
<td>1.24 c</td>
<td>45.1 a</td>
<td>22.6 a</td>
<td>1.17 a</td>
</tr>
<tr>
<td>CF</td>
<td>0.17 b</td>
<td>0.71 bc</td>
<td>0.11 b</td>
<td>1.29 bc</td>
<td>43.3 a</td>
<td>21.2 a</td>
<td>0.23 b</td>
</tr>
<tr>
<td>CRP</td>
<td>0.13 c</td>
<td>0.70 c</td>
<td>0.17 a</td>
<td>1.35 ab</td>
<td>33.0 c</td>
<td>14.3 c</td>
<td>0.07 b</td>
</tr>
<tr>
<td>NT</td>
<td>0.18 b</td>
<td>0.76 a</td>
<td>0.07 c</td>
<td>1.41 a</td>
<td>32.7 c</td>
<td>15.0 c</td>
<td>0.05 b</td>
</tr>
<tr>
<td>CT</td>
<td>0.16 b</td>
<td>0.75 ab</td>
<td>0.09 bc</td>
<td>1.38 a</td>
<td>35.3 bc</td>
<td>16.2 bc</td>
<td>0.04 b</td>
</tr>
</tbody>
</table>

† Means in a column with the same letter do not differ (P > 0.05).
Table 3. Summary of the 30-year (1981 to 2010) mean annual precipitation and air temperature, as well as the 30-year minimum and maximum monthly air temperatures for each location included in this study (NOAA, 2013).

<table>
<thead>
<tr>
<th>Landuse</th>
<th>Specific Site</th>
<th>Location</th>
<th>Precipitation (cm)</th>
<th>Air Temperature Mean (°C)</th>
<th>Air Temperature Minimum (°C)</th>
<th>Air Temperature Maximum (°C)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Native Prairie</td>
<td>Seidenstricker Prairie</td>
<td>Arkansas County Monroe County</td>
<td>125.6</td>
<td>16.6</td>
<td>10.6</td>
<td>22.6</td>
</tr>
<tr>
<td></td>
<td>Gray Prairie</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Roth 1</td>
<td>Arkansas County</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Roth 2</td>
<td>Arkansas County</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>CRP†</td>
<td>PTRS‡</td>
<td>Arkansas County Monroe County</td>
<td>128.5</td>
<td>16.6</td>
<td>11.2</td>
<td>22.1</td>
</tr>
<tr>
<td>Deciduous Forest</td>
<td>PTRS‡</td>
<td>Arkansas County</td>
<td>128.5</td>
<td>16.6</td>
<td>11.2</td>
<td>22.1</td>
</tr>
<tr>
<td></td>
<td>LMCRS¶</td>
<td>Arkansas County</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Coniferous Forest</td>
<td>PTRS</td>
<td></td>
<td>128.5</td>
<td>16.6</td>
<td>11.2</td>
<td>22.1</td>
</tr>
<tr>
<td>CT§</td>
<td>LMCRS</td>
<td></td>
<td>128.5</td>
<td>16.6</td>
<td>11.2</td>
<td>22.1</td>
</tr>
<tr>
<td></td>
<td>PTRS</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NT§§</td>
<td>LMCRS</td>
<td></td>
<td>128.5</td>
<td>16.6</td>
<td>11.2</td>
<td>22.1</td>
</tr>
</tbody>
</table>

† CRP, Conservation Reserve Program
‡ PTRS, University of Arkansas System Division of Agriculture Pine Tree Research Station, near Colt, AR
¶ LMCRS, Lon Mann Cotton Research Station, near Marianna, AR
§ CT, conventional-tillage agriculture
§§ NT, no-tillage agriculture
Table 4. Analysis of variance summary of the effects of landuse, soil depth, aggregate-size class, and their interactions on water-stable aggregate (WSA) concentration, mean weight diameter (MWD), and total water-stable aggregate (TWSA) concentration on fine-textured, loessial and alluvial soils in the Delta region of eastern Arkansas.

<table>
<thead>
<tr>
<th>Source of Variation</th>
<th>WSA</th>
<th>MWD</th>
<th>TWSA</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>P</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Landuse</td>
<td>&lt; 0.01</td>
<td>&lt; 0.01</td>
<td>&lt; 0.01</td>
</tr>
<tr>
<td>Soil depth</td>
<td>&lt; 0.01</td>
<td>&lt; 0.01</td>
<td>&lt; 0.01</td>
</tr>
<tr>
<td>Landuse x soil depth</td>
<td>0.26</td>
<td>0.06</td>
<td>0.26</td>
</tr>
<tr>
<td>Size class</td>
<td>&lt; 0.01</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Landuse x size class</td>
<td>&lt; 0.01</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Depth x size class</td>
<td>&lt; 0.01</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Landuse x soil depth x size class</td>
<td>0.18</td>
<td>–</td>
<td>–</td>
</tr>
</tbody>
</table>
Table 5. Summary of the linear correlations ($r$) between total water-stable aggregate (TWSA) concentration, mean weight diameter (MWD), infiltration-related parameters (i.e., overall infiltration rate and slope and intercept parameters characterizing the linear relationship between the natural logarithm of the infiltration rate and mid-point of time), and soil physical and chemical properties [i.e., sand, silt, and clay concentrations, soil organic matter (SOM) content, estimated bulk density, total C and N contents, and C:N, N:SOM, and C:SOM ratios] in the top 10 cm of various landuses in the Delta region of eastern Arkansas. Data and results are reproduced from Anderson (2019).

<table>
<thead>
<tr>
<th>Soil Property</th>
<th>TWSA</th>
<th>MWD</th>
</tr>
</thead>
<tbody>
<tr>
<td>MWD (mm diameter)</td>
<td>0.92***</td>
<td>--</td>
</tr>
<tr>
<td>Overall infiltration rate (mm min$^{-1}$)</td>
<td>0</td>
<td>-0.18</td>
</tr>
<tr>
<td>Slope</td>
<td>-0.03</td>
<td>0.14</td>
</tr>
<tr>
<td>Intercept</td>
<td>0.11</td>
<td>0.03</td>
</tr>
<tr>
<td>Sand (g g$^{-1}$)</td>
<td>0.37*</td>
<td>0.25</td>
</tr>
<tr>
<td>Silt (g g$^{-1}$)</td>
<td>-0.36*</td>
<td>-0.49**</td>
</tr>
<tr>
<td>Clay (g g$^{-1}$)</td>
<td>-0.02</td>
<td>0.19</td>
</tr>
<tr>
<td>SOM (Mg ha$^{-1}$)</td>
<td>-0.05</td>
<td>-0.03</td>
</tr>
<tr>
<td>Bulk density (g cm$^{-3}$)</td>
<td>0.07</td>
<td>0.05</td>
</tr>
<tr>
<td>Total C (Mg ha$^{-1}$)</td>
<td>0.01</td>
<td>0.01</td>
</tr>
<tr>
<td>Total N (Mg ha$^{-1}$)</td>
<td>0.02</td>
<td>0.02</td>
</tr>
<tr>
<td>C:N</td>
<td>-0.10</td>
<td>-0.10</td>
</tr>
<tr>
<td>N:SOM</td>
<td>0.08</td>
<td>0.08</td>
</tr>
<tr>
<td>C:SOM</td>
<td>0.08</td>
<td>0.08</td>
</tr>
</tbody>
</table>

* Significant ($P < 0.05$) correlation between variables
** Significant ($P < 0.01$) correlation between variables
*** Significant ($P < 0.001$) correlation between variables
Figure 1. Geographic distribution of the three native prairie sites included in this study located near Stuttgart, AR (Google Earth, 2019).
Figure 2. Geographic distribution of the 12 sites included in this study at the Pine Tree Research Station (PTRS) near Colt, AR representing four different landuses [i.e., conventional-tillage agriculture, Conservation Reserve Program (CRP) grassland, coniferous forest, and deciduous forest] (Google Earth, 2019).
Figure 3. Geographic distribution of four sites included in this study at the Lon Mann Cotton Research Station (LMCRS) near Marianna, AR representing three different landuses (i.e., conventional-tillage agriculture, no-tillage agriculture, and deciduous forest). The upper-left-most yellow star marks the location of the long-term, wheat-soybean, double-crop field study whose 16 treatment combinations, replicated three times each, were included in this study (Google Earth, 2019).
Figure 4. Landuse [native prairie (NP), deciduous forest (DF), coniferous forest (CF), Conservation Reserve Program grassland (CRP), conventional-tillage agriculture (CT), and no-tillage agriculture (NT)] and aggregate-size class effects on water-stable aggregate (WSA) concentration. Different letters atop bars indicate significant differences separated by the most conservative least significant difference (42 g kg\(^{-1}\)).
Figure 5. Soil depth (0-5 cm and 5-10 cm) and aggregate-size class effects on water-stable aggregate (WSA) concentrations. Different letters atop bars indicate significant differences separated by the most conservative least conservative difference (16.6 g kg⁻¹).
Figure 6. Landuse [native prairie (NP), deciduous forest (DF), coniferous forest (CF), Conservation Reserve Program grassland (CRP), conventional-tillage agriculture (CT), and no-tillage agriculture (NT)] effect on (A) mean weight diameter and (B) total water-stable aggregate (TWSA) concentration. Different letters atop bars indicate significant differences between landuses ($P < 0.05$).
Conclusions
This field study evaluated the effects of major, current landuses on surface water infiltration and aggregate stability on fine-textured, loessial and alluvial soils in the Delta region of eastern Arkansas in the Lower Mississippi River Valley (LMRV). Results of this study showed that landuse and various landuse management practices affected infiltration, infiltration-related parameters, water-stable aggregation, and certain physical and chemical soil properties in the Delta region of eastern Arkansas in the LMRV. Contrary to that hypothesized, the deciduous forest had the largest overall infiltration rate, while the remaining landuses (i.e., native prairie, coniferous forest, Conservation Reserve Program (CRP) grassland, and conventional-tillage (CT) and no-tillage (NT) agriculture) had similar overall infiltration rates. A strong positive correlation was shown between overall infiltration rate and estimated saturated hydraulic conductivity ($K_{\text{sat}}$), indicating estimated $K_{\text{sat}}$ is related to infiltration rate and can be used to assess the impacts of landuse management practices, as well as overall soil quality. The CRP grassland and CT and NT agroecosystems generally exhibited lower y-intercept values, representing the theoretical infiltration rate at time zero, compared to the deciduous and coniferous forests and native prairie. Overall infiltration rate was positively correlated with soil organic matter (SOM), total carbon (TC), and total nitrogen (TN) contents, and C:SOM ratio, while negatively correlated with bulk density and extractable soil Na and Mg contents in the top 10 cm. Results indicated that SOM content and soil bulk density are specific near-surface landuse characteristics that have a large effect on overall infiltration rate and other infiltration-related parameters.

As hypothesized, the less-managed landuses (i.e., grasslands and forests) generally exhibited greater total water-stable macroaggregate (TWSA) concentrations and mean weight diameter (MWD) than current agroecosystems, with the exception of the deciduous forest. The
TWSA concentration and MWD of the CRP grassland was similar to that of the native prairie and at least numerically greater than the agroecosystems, indicating that CRP-management practices were effective in improving aggregate stability of former agroecosystems to levels comparable to that of a native prairie and greater than that of current agroecosystems. The NT and CT agroecosystems exhibited similar aggregate-size distributions, TWSA concentrations, and MWD to each other. The long-term benefits of NT, even after 16 years, were not yet evident in this study. Based on the results of this study, implementation of NT practices alone may be insufficient to improve soil structure and decrease the risk of soil erosion in fine-textured, loessial and alluvial soils in the Arkansas Delta region of the LMRV. Results indicated that soil in the top 5 cm had greater aggregate stability than soil in the 5- to 10-cm depth in fine-textured, loessial and alluvial soils of the Arkansas Delta region of the LMRV.

Identifying landuse management practices and soil surface properties that improve infiltration-related soil-surface properties and water-stable aggregation may allow more rainfall and/or irrigation water to infiltrate to potentially recharge the Alluvial Aquifer and decrease the risk of soil erosion and contamination of surface waters by eroded sediment and sediment-bound nutrients. Grassland and forest restoration, specifically of highly erodible agricultural land, should continue to be considered to contribute to improvements in infiltration capacity, soil structural stability, and the future sustainability of soil and water resources in the LMRV.