The Long-Term Impacts of Forest Removal on Floodplain Subsurface Hydrology

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by

Elliott Kellner

Dissertation Advisor: Jason A. Hubbart, Ph.D.

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The undersigned, appointed by the dean of the Graduate School, have examined the dissertation entitled

The Long-Term Impacts of Forest Removal on Floodplain Subsurface Hydrology

Presented by Elliott Kellner,

a candidate for the degree of Ph.D. Natural Resources

and hereby certify that, in their opinion, it is worthy of acceptance.

Dr. Jason A. Hubbart

Dr. Stephen Anderson

Dr. Keith Goyne

Dr. Martin Appold

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The Long-Term Impacts of Forest Removal on Floodplain Subsurface Hydrology

Elliott Kellner

Dissertation Advisor: Jason A. Hubbart

ABSTRACT

A study was implemented in fall 2010, in the Hinkson Creek Watershed, Missouri, USA, to improve quantitative understanding of the long-term impact of forest removal on floodplain hydrology. Automated volumetric water content (VWC) probes and piezometers equipped with pressure transducers to monitor shallow groundwater (SGW) temperature and level were installed in a gridded study design within a historic agricultural field (Ag) and a remnant bottomland hardwood forest (BHF). Groundwater was analyzed for 49 physiochemical metrics. Results showed VWC to be significantly different between sites ($p<0.01$) during the study, with site averages of 33.1 and 32.8% at the Ag and BHF sites, respectively. Semi-variogram analyses results suggest historic forest removal and cultivation of the Ag site facilitated the development of strong VWC spatial dependency. SGW temperature range at the Ag site was 72% greater than at the BHF site. BHF groundwater contained significantly ($p<0.05$) higher concentrations of nutrients, while Ag groundwater was characterized by significantly ($p<0.05$) higher concentrations of trace elements. Collective results highlight the greater extent to which BHF vegetation impacts subsurface hydrology, relative to grassland/agricultural systems, and point to the value of reestablishing floodplain forests for freshwater routing, water quality, aquatic ecosystem conservation, and flood mitigation in mixed-land-use watersheds.

CHAPTER I

INTRODUCTION

Background

Anthropogenic land use/land cover change has been occurring for centuries (Copeland et al., 1996), and has radically altered many landscapes. Land use changes like forest removal can impact a variety of physical, chemical, and biological natural processes. Studies have repeatedly and conclusively shown that land use change modifies hydrologic regimes in response to altered mass and energy flux (Twine et al., 2004; Karwan et al., 2007; Bulliner, 2011; Freeman, 2011). Land use impacts on hydrologic regimes are particularly important in floodplain landscapes. As a function of their location at the groundwater-surface water interface, floodplains can provide numerous ecosystem services such as biogeochemical transformation and remediation of nutrients and pollutants, runoff regulation, and flood water mitigation (Krause et al., 2007).

Floodplain forests in particular are important ecotones, regulating energy and nutrient fluxes between surface water and groundwater reservoirs (Sanchez-Pérez and Trémolières, 2003). Recent research highlighted the benefits of floodplain forests, including but not limited to filtering runoff/groundwater, reducing overbank flow velocity, functioning as a source of coarse woody debris to streams (an important component of in-stream habitat), and providing diverse habitat essential to biodiversity (Naiman et al., 1993; Piegay, 1997). Floodplain forests can also function as zones of nutrient (e.g. nitrate) retention, influencing groundwater chemistry and reducing nutrient loading to adjacent streams (Jordan et al., 1993). Additionally, floodplain soils and vegetation management are central to the flood water attenuation capacity of floodplains (Hubbart et al., 2011). Floodplains attenuate floods by retaining water and decreasing velocity, thereby slowing and reducing flood wave impacts (Wheater and Evans, 2009). Thus, downstream flood risk mitigation can be achieved by permitting the natural attenuation (i.e. inundation, infiltration, retention, evapotranspiration) afforded by floodplains (Wheater and Evans, 2009). However, research indicates that less than 25% of pre-Columbian bottomland hardwood forest remains in various regions of the U.S., and that the majority of forest removal has been due to agricultural conversion (Abernathy and Turner, 1987; Carter and Biagas, 2007).

Over time, land use/land cover changes (e.g. forest removal, agricultural cultivation) can result in alterations to subsurface hydraulic processes such as infiltration rate, percolation, and soil water retention (Hubbart et al., 2011; Wahren et al., 2009). Recent studies have identified numerous impacts of land use on subsurface hydrology. Scanlon et al. (2005) noted that rangeland cultivation resulted in an alteration of water flow direction, from primarily upward under rangeland, to primarily downward under cropland. Wahren et al. (2009) showed that afforestation resulted in increased infiltration, soil water retention, and hydraulic conductivity, particularly in the upper soil horizons. The researchers also noted that subsurface flow in forested sites can be dominated by preferential flow via macropores created by decayed roots (Wahren et al. 2009).

Similarly, Gonzalez-Sosa et al. (2010) identified land use impacts on topsoil hydraulic properties in France, showing that bulk density values were lowest and hydraulic conductivity values were highest for forest soils, relative to pasture and cropland. In a study comparing soil characteristics of a remnant bottomland hardwood forest and a former agriculture field, Hubbart et al. (2011) reported mean infiltration at the forest site to be 61% greater than the agricultural site, and showed that the upper one meter of the forest soil profile contained approximately 11% more soil water. In addition to influencing the distribution of water in the subsurface, hydrologic alterations can impact fluxes of mass and energy. For example, recharge rates can influence groundwater temperature regimes (Bundschuh, 1993). Similarly, studies by Soveri (1985), Katz et al. (1998), Adams et al. (2001), Böhlke (2002), and Kebede et al. (2005) reported the potential for recharge to either enrich or dilute groundwater with respect to a given constituent, depending on the chemical composition of atmospheric and vadose zone water relative to groundwater.

Vadose Zone Volumetric Water Content

Land use impacts on subsurface water are particularly critical in floodplain landscapes, where vadose zone water content can dramatically influence flood attenuation potential (Hubbart et al., 2011). Vadose zone water distribution and variability is a key issue in many areas of industry and research, including (but not limited to) agriculture, geology, and engineering (Benedetto et al., 2015), and is known to impact rates of energy and mass flux between the atmosphere, surface, and subsurface (Choi and Jacobs, 2007; Vereecken et al., 2007). Vadose zone volumetric water content (VWC) creates a critical interface that partitions precipitation, surface runoff, infiltration, and recharge (Heathman et al., 2012). Soil volumetric water content in surface soils exchanges mass and energy via the surface energy balance, thereby impacting soil heat flux (Phillip, 1957; Jackson, 1973; Hanks et al., 1980), albedo of the soil surface (Idso et al., 1975), and advection and conduction of energy through the soil profile (e.g. infiltration, percolation). Additionally, soil water content is a primary factor influencing soil microbial activity, community composition, soil respiration, and soil carbon sequestration processes (Skopp et al., 1990; Davidson et al., 1998; Drenovsky et al., 2004).

While precipitation regime typically drives soil volumetric water content, soil water is also influenced by land use practices and vegetative cover (Jipp et al., 1998). Specifically, vadose zone water is impacted by surface vegetation communities in terms of the spatial distribution of plants and species-specific rates of water use (Breshears et al., 1997). For example, as a result of canopy interception, stemflow, and throughfall, vegetation can produce spatiotemporal variability in infiltration and distribution of soil water (Martello et al., 2015). Land use/land cover changes such as forest removal can alter subsurface hydraulic processes, including infiltration, percolation, and soil water retention (Hubbart et al., 2011; Wahren et al., 2009), thereby reducing floodplain attenuation potential. Despite previous work on land use-soil water relationships, many studies have investigated the impacts of recent land use change, and little information is available regarding the long-term impacts (i.e. legacy effects) of historic land use change and established land use patterns on soil water regime. Considering the environmental importance of vadose zone water content, particularly in floodplain landscape positions, information is needed relating historic floodplain land use patterns and vadose zone hydrologic regimes.

Shallow Groundwater Temperature

Groundwater temperature is an important physical determinant of surface water quality in groundwater-fed surface water systems, influencing the metabolic rates, physiology, and life history traits of aquatic species, interstitial habitat suitability, and the nutrient cycling and productivity of aquatic communities (Poole and Berman, 2001; Sophocleous, 2002). Abrupt groundwater temperature change from background or reference conditions can alter biogeochemical reactions, eventually influencing groundwater quality (Mitchell and Ferris, 2005; Taniguchi et al., 2007; Figura et al., 2011). For example, temperature alterations can influence carbonate precipitation, salt solubility, dissolution of silicate minerals, and the mobilization of organic compounds. Prommer and Stuyfzand (2005) showed that oxidation reactions in groundwater depended significantly on spatial and temporal groundwater temperature variation.

Shallow groundwater temperature is influenced by numerous factors, including soil physical properties (e.g. thermal diffusivity), solar radiation, ground surface temperature, precipitation, water table depth, groundwater flow, and exchanges with surface water (Bundschuh, 1993; Saar, 2011; Wang et al., 2014). Any land use change that alters the magnitude or direction of any such influential components can subsequently impact the groundwater temperature regime. For example, a comparative modeling study by Ferguson and Beltrami (2006) suggested the potential for deforestation to produce subsurface temperature alterations. Removal of the forest canopy could increase surface heating due to increased solar radiation at the ground surface, and thus alter baseflow temperature via soil heat flux. In a case study comparing the surface energy balance at forest and agricultural sites, Zell and Hubbart (2013) reported greater average, maximum, and minimum daily change in energy storage ($W\ m^{-2}$) at the agricultural site, relative to the forest, thus suggesting greater surface warming at the exposed agricultural field. Considering the importance of groundwater temperature to a variety of surface and subsurface biogeochemical processes, studies are needed that accurately characterize groundwater temperature regimes. Despite progress of previous investigations, many of which focused on urban land use types and/or applied one-dimensional models, few studies have been published pertaining to rural land use impacts on shallow groundwater temperatures. Further, the majority of previous studies utilized episodic or opportunistic sampling methods, and involved the incorporation of critical assumptions (e.g. no horizontal groundwater flow [Taniguchi et al., 2005, 2007]). Few studies can be found in the literature utilizing continuous, automated, *in situ* sampling methods to analyze groundwater temperature relationships and dynamics. Such methods can produce high-resolution quantifications of groundwater temperature responses to land use altered environmental variables (e.g. air temperature, solar radiation) and subsurface characteristics and processes (e.g. hydraulic conductivity, groundwater flow).

Floodplain Groundwater Chemistry

In addition to the vadose zone water flow regime and shallow groundwater temperatures, hydrologic alterations due to land use/land cover change and floodplain management can impact the chemical quality of groundwater and surface water, a critical consideration for resource suitability and aquatic ecosystem conservation. For example, Scanlon et al. (2005) showed that vertical fluxes of water in the subsurface from evapotranspiration and recharge can enrich or dilute, respectively, the chloride and nitrate concentrations of groundwater. Land use change often includes altered vegetated canopy cover, which can impact the physiochemical processes and characteristics of the subsurface. For example, Addy et al. (1999) noted several vegetation-mediated impacts, including subsurface hydrology, spatial distribution of soil carbon, and land use legacies, all of which impacted groundwater nitrate removal rates. Yevenes and Mannaerts (2011) found that agricultural sub-catchments exported five times more nitrate than forest and range sub-catchments. Balestrini et al. (2011) showed that alteration of subsurface flow patterns due to consumptive water use by woody vegetation (i.e. evapotranspiration) was a primary factor contributing to riparian groundwater nitrate removal rates. Pionke and Urban (1985) reported concentrations of nitrate, chloride, and phosphate in groundwater that were five to seven times higher under cropland, relative to forest land use. Similarly, Jiang et al. (2008) reported increases of pH and pollutant concentrations in groundwater following the conversion of forest to cropland and urban areas. Many previous groundwater chemistry studies addressed land use impacts on specific chemical constituents (e.g. nitrate) or contaminants (e.g. chlorinated solvents); however, few previous studies were explicitly designed to characterize the chemical composition of groundwater at sites with contrasting land use histories. The impacts of land use changes such as forest removal are especially important in floodplains since riparian and alluvial forests serve a crucial function in regulating fluxes of energy and nutrients between surface and groundwater systems (Sanchez-Pérez and Trémolières, 2003). Given the recognized connections between groundwater and surface water (Hayashi and Rosenberry, 2002), and the location of floodplains at the groundwater-surface water interface, understanding patterns of environmental and anthropogenic influences on floodplain groundwater chemistry is important in order to effectively manage aquatic ecosystems, and maintain the suitability of groundwater for human uses.

Statement of Need

There has been insufficient attention devoted to long-term impacts of historic land use/land cover change and floodplain management by current land and water resource managers. Thus, research is needed that can provide decision-makers with quantitative information and thereby improve floodplain, groundwater, and aquatic ecosystem management (Hubbart, 2011). Furthermore, given the increasing pace of land use/land cover change worldwide and the importance of subsurface hydrology to floodplain attenuation potential, groundwater resource suitability, and aquatic ecosystems, information is needed concerning relationships between different floodplain land use types, subsurface hydrologic regime, and groundwater chemical composition.

Objectives

The overarching objective of this dissertation research was to improve understanding of the impacts of different rural land use types on floodplain subsurface water resources in mixed-land-use watersheds of the central U.S. Specific objectives were to a) utilize high-resolution, continuous, *in situ* soil water sensor data and geostatistical analyses to compare the spatiotemporal distribution of vadose zone volumetric water content of 100 year old agricultural and forested floodplain sites; b) characterize and compare shallow groundwater (SGW) temperatures 1-2 m below the water table at agricultural and forested floodplain sites, and relate observed SGW temperature contrasts to site differences in estimated groundwater flow and microclimate; c) quantitatively describe and compare groundwater chemical composition in an intensive gridded investigation of historic agricultural and forested floodplain sites.

Dissertation Structure

This dissertation is presented in the following chapters: in chapter two, "A Comparison of the Spatial Distribution of Vadose Zone Water in Forested and Agricultural Floodplains a Century after Harvest", gridded, multi-depth, continuous, automated, *in situ* soil water monitoring and geostatistical analytical methods were integrated to determine the spatiotemporal distribution and spatial correlation of vadose zone water content at forest and agricultural study sites during the 2011, 2012, and 2013 water years. In chapter three, "Agricultural and Forested Land Use Impacts on Floodplain Shallow Groundwater Temperature Regime", gridded, continuous, automated, groundwater monitoring during the 2011, 2012, 2013, and 2014 water years was used to investigate the long-term impact of contrasting land use on the thermal regime of shallow groundwater, and to relate observed differences to contrasting site microclimate and hydrology. In chapter four: "A Comparison of Forest and Agricultural Shallow Groundwater Chemical Status a Century after Land Use Change", data from a two year (June 2011 - June 2013) sampling campaign were analyzed for 49 different physiochemical metrics, in order to comprehensively characterize the chemical composition of floodplain groundwater at forest and agricultural sites. Grab samples from the adjacent stream were also analyzed, and results compared to those of floodplain groundwater to highlight potential contributing processes. Finally, in chapter five, "Conclusions and Synthesis", a summary of the key findings of this investigation is presented and future research directions are discussed that, if pursued, could lead to improved understanding of historic land use impacts on floodplain hydrology and floodplain, groundwater, and aquatic ecosystem management.

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CHAPTER II

A COMPARISON OF THE SPATIAL DISTRIBUTION OF VADOSE ZONE WATER IN FORESTED AND AGRICULTURAL FLOODPLAINS A CENTURY AFTER HARVEST

In Press:

Kellner, E., Hubbart, J. A. 2015. A Comparison of the Spatial Distribution of Vadose Zone Water in Forested and Agricultural Floodplains a Century after Harvest. *Science of the Total Environment*. x:xx.

Introduction

Background

Vadose zone water distribution and variability is a key issue in many areas of industry and research, including (but not limited to) agriculture, geology, and engineering (Benedetto et al., 2015), and is known to impact rates of energy and mass flux between the atmosphere, surface, and subsurface (Choi and Jacobs, 2007; Vereecken et al., 2007). Vadose zone volumetric water content (VWC) creates a critical interface that partitions precipitation, surface runoff, infiltration, and recharge (Heathman et al., 2012). Soil volumetric water content in surface soils exchanges mass and energy via the surface energy balance, thereby impacting soil heat flux (Phillip, 1957; Jackson, 1973; Hanks et al., 1980), albedo of the soil surface (Idso et al., 1975), and advection and conduction of energy through the soil profile (e.g. infiltration, percolation). Additionally, soil water content is a primary factor influencing soil microbial activity, community composition, soil respiration, and soil carbon sequestration processes (Skopp et al., 1990; Davidson et al., 1998; Drenovsky et al., 2004). Given the environmental importance of soil water, continued research on patterns of soil water distribution is needed to improve land and water resource management.

While precipitation regime typically drives soil volumetric water content, soil water is also influenced by land use practices and vegetative cover (Jipp et al., 1998). Specifically, vadose zone water is impacted by surface vegetation communities in terms of the spatial distribution of plants and species-specific rates of water use (Breshears et al., 1997). For example, as a result of canopy interception, stemflow, and throughfall, vegetation can produce spatiotemporal variability in infiltration and distribution of soil water (Martello et al., 2015). Jipp et al. (1998) reported significant differences between pasture and forest sites (Duncan's Multiple Range Test, α=0.05) in VWC down to eight meters. Vegetative impacts on soil water are particularly critical in floodplain landscapes, where vadose zone water content can dramatically influence flood attenuation potential (Hubbart et al., 2011). Typically, floodplains attenuate floods by retaining water and decreasing velocity, thereby slowing and reducing flood wave impacts (Wheater and Evans, 2009). However, downstream flood risk can also be mitigated by encouraging maximum floodplain infiltration capacity and retention of floodwater in the subsurface (Hubbart et al., 2011; Wheater and Evans, 2009). Land use/land cover changes such as forest removal can alter subsurface hydraulic processes, including infiltration rate, percolation, and soil water retention (Hubbart et al., 2011; Wahren et al., 2009), thereby reducing floodplain attenuation potential. In regions where land use changes lead to environmental degradation (Pacheco et al., 2014; Valle Junior et al., 2014a,b, 2015), consequences of flooding can be more severe due to enhanced soil erosion and subsequent impacts to water quality. Given the influence of vegetation on the soil water regime, and the importance of the soil water regime to floodplain hydrology, research is needed to determine the impact of floodplain land use types on the spatiotemporal distribution of soil water.

Soil VWC is a dynamic variable, displaying spatial (vertical and horizontal) and temporal (event, diurnal, annual) variations (Jackson, 1973; Breshears et al., 1997, Vereecken et al., 2007), making it difficult to characterize (Zhang et al., 2015). Soil water research based on episodic field sampling campaigns (e.g. Robertson et al., 1993, López-Vicente et al., 2015; Zhang et al., 2015) have often been unable to accurately describe the spatiotemporal variability of vadose zone water. A large volume of research in recent decades identifying this problem (e.g. Rhoades et al., 1976; Topp et al., 1980; Dalton et al., 1984) has resulted in the development of sensors capable of measuring VWC at small, relevant, and informative spatial and temporal scales. However, despite the availability of such high-resolution monitoring technology, few studies to date have employed continuous automated sensors in multi-depth, gridded sampling designs successfully reporting spatiotemporal relationships of vadose zone water regimes.

The use of geostatistical methods to analyze and describe soil water distribution has increased in recent decades (Yates and Warrick, 1987). Geostatistical methods such as semi-variogram analysis are capable of quantifying the spatial correlation of variables, information that can then be used for prediction of values in unsampled locations (e.g. kriging) (Yates and Warrick, 1987). Recent studies have applied these methods to research on land use impacts on soil properties (e.g. Nyberg, 1996). For example, Robertson et al. (1993) showed that the range of spatial correlation of several soil physical properties, including water content, was greater in an agricultural field, relative to an uncultivated field. Despite demonstrated usefulness of geostatistical techniques for describing spatial patterns of soil water content, few studies to date have applied the methods in investigations of forest removal impacts on soil water distribution and regime.

Objectives

Considering the environmental significance of soil water, there is a need for studies incorporating long-term, continuous soil water monitoring to characterize the spatial and temporal distribution of soil volumetric water content. Furthermore, given the increasing pace of land use/land cover change worldwide and the importance of subsurface hydrology to floodplain attenuation potential, information is needed concerning relationships between different floodplain land use types and vadose zone water distribution, and the long-term impacts of land use changes such as forest removal on soil water regime. Despite previous work on land use-soil water relationships, many studies have investigated the impacts of recent land use change, and little information is available regarding the long-term impacts (i.e. legacy effects) of historic land use change and established land use patterns on soil water regime. Therefore, the objective of the current work was to utilize high-resolution, continuous, *in situ* soil water sensor data and geostatistical analyses to compare the spatiotemporal distribution of vadose zone volumetric water content of 100 year old agricultural and forested floodplain sites. Results will improve understanding and, therefore, future management of land cover change on soil water regime in floodplains of mixed-land-use watersheds.

Methods

Site Description

This research was conducted in the Hinkson Creek Watershed (HCW) located in central Missouri, USA. The project was nested within a larger mixed-land-use experimental watershed study (approximately 240 km^2) implemented in 2008, with core work including environmental flows, water quality and climate (Hubbart et al., 2010). The HCW contains approximately 60% of the city of Columbia, a city with a population of approximately 116,000 (USCB, 2015). Land use in the HCW is approximately 34% forest, 38% agriculture, and 25% urban (Hubbart et al., 2011), making it a representative watershed for studying the effects of different land use types on water resources. According to a 64 year climate record of Columbia, Missouri, average annual temperature and precipitation within the watershed were 12.5 °C and 991 mm yr^{-1}, respectively (Missouri Climate Center, 2014).

Two floodplain reaches of Hinkson Creek with contrasting land use histories were selected for this investigation: a remnant bottomland hardwood forest (BHF), and a historic (i.e. currently fallow) agricultural field (Ag) (Figure 1). Sites were separated by a distance of approximately 900 m. Woody vegetation at the BHF site includes Silver maple (*Acer saccharinum*), Eastern cottonwood (*Populus deltoides*), Boxelder (*Acer negundo*), American elm (*Ulmus americana*), Black walnut (*Juglans nigra*), and Sycamore (*Platanus occidentalis*) (Hubbart et al., 2011). Forest vegetation was removed from the Ag site approximately 100 years ago. Current vegetation at the Ag site is primarily composed of gramminoids such as Johnsongrass (*Sorghum halepense*), and tall fescue (*Festuca arundinacea*), and the forbs goldenrod (*Solidago giganteana*), and Ironweed (*Vernonia altissima*) (Brown, 2013). Both sites contain Haymond silt loam alluvial soils (NRCS, 2015). Soils in the floodplain study area are underlain by Mississippian series limestones (Burlington formation) (Unklesbay, 1952; Miller and Vandike, 1997). Average site elevation is 174.4 and 176.3 m.a.s.l. for the Ag and BHF, respectively. Hubbart et al. (2011) reported statistical similarity (CI=0.05) of physical soil characteristics (i.e. bulk density and porosity) below 50 cm depth at the two sites. Above 50 cm, statistically significant differences ($p<0.05$) reported between study site soil characteristics reflected impacts of surface vegetation on soil hydraulic properties including infiltration capacity (23 and 38 cm hr^{-1} at the Ag and BHF, respectively; $p<0.05$). Thus, on the basis of findings of previous studies, sites were assumed similar in most respects; except land use history, which has been consistently different for more than 100 years.

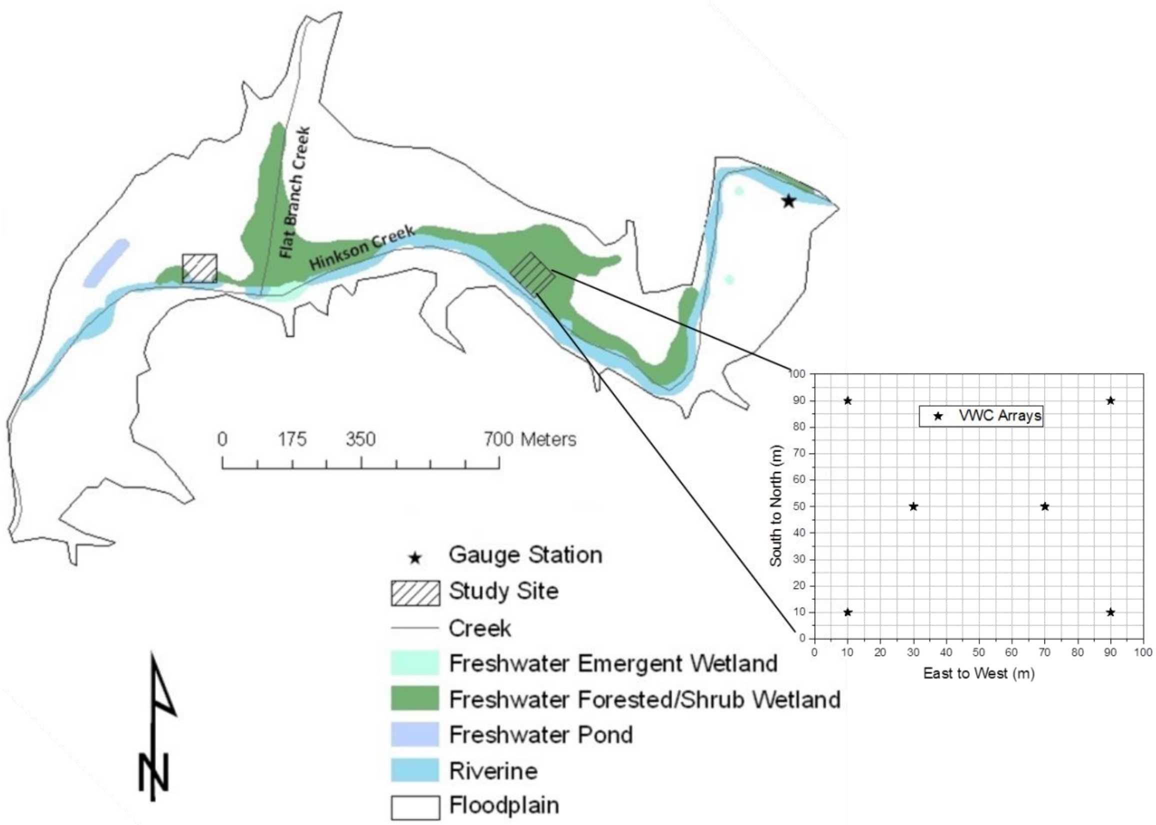

Figure 1. Shallow groundwater study sites in lower Hinkson Creek Watershed floodplain, located in central Missouri, USA.

Data Collection

Vadose zone volumetric water content was monitored using Decagon Devices, Inc. 10HS large volume soil moisture sensors at depths of 15, 30, 50, 75, and 100 cm. The 10HS determines volumetric water content (VWC, accuracy ±3%) by measuring the dielectric constant of the soil using capacitance-frequency domain technology (Decagon Devices Inc., 2014). For most mineral soils, particularly including low salinity and particularly sandy loamy alluvial soils as in the current work, 10HS sensors do not require calibration. Furthermore, considering high probe-to-probe agreement (Decagon Devices Inc., 2014), and similar soil texture at the study sites (Hubbart et al., 2011), specific calibration was unlikely to improve the accuracy of site VWC comparisons and was thus deemed unnecessary. VWC was monitored at six locations within each study site (Figure 1). The study design facilitated investigation of VWC spatial relationships in the horizontal and vertical directions. The gridded, multiple depth study design is similar to that used by Gish et al. (2002) to evaluate different soil water content measurement technologies and Guber et al. (2008) to investigate temporal stability of soil water content. Whereas the previous studies utilized random sampling, the current work applied a regular sampling distribution (Figure 1) in order to facilitate robust geostatistical analysis. Considering the spatiotemporal variance of subsurface hydrologic parameters (e.g. hydraulic conductivity [Dagan and Neuman, 2005]), a regular sampling distribution ensures that each sub-section of the site was sampled equally, thus avoiding potential over- or under-sampling (i.e. bias) of a given sub-section within the larger study site. All data were logged and recorded at 30 minute intervals for the duration of the 2011, 2012, and 2013 water years. Air temperature and precipitation were measured at Sanborn Field, located on the University of Missouri campus approximately 4 km northeast of the study sites, to obtain climate data representative of the HCW.

Data Analysis

Descriptive statistics were calculated using VWC data collected at each depth, and averaged by land use type. Considering hydroclimate data are often non-normally distributed (McCuen, 2003), the Wilcoxon Signed-Rank test was used for statistical comparison of groundwater temperature data. Wilcoxon Signed-Rank test is an appropriate non-parametric choice for paired (e.g. temporally) non-normal data sets (Helsel and Hirsch, 1992). Wilcoxon Signed-Rank tests were run on daily VWC data averaged by land use. The spatial correlation of study site soil water content was analyzed by calculating the experimental semi-variogram, given by:

$$\gamma^* = \frac{\frac{1}{n}\sum[g(x)-g(x+h)]^2}{2} \qquad \text{Eq. 1}$$

where γ^* is the experimental semi-variogram ($\%^2$), g(x) is a VWC measurement at point x, and g(x + h) is a VWC measurement at a point located a distance (h) from point x (Clark, 1979). The semi-variogram quantifies the spatial correlation of separate measurements of a variable (i.e. different sampling locations) as a function of separation distance (Yates and Warrick, 1987). Semi-variogram analyses were performed on VWC data in the horizontal and vertical directions, and averaged by land use type. This analysis was performed on data aggregated to seasonal and study period time scales. Strength of spatial dependency was quantified according to the categorization presented by Kazemi et al. (2008), where the contribution of the nugget to the spatially dependent variability (nugget/sill*100%) is classified as strong (<25%), moderate (25 – 75%), and weak (>75%). Given the analysis of the experimental semi-variogram in the current work, the minimum observed separation distance (40 m and 15 cm in the horizontal and vertical directions, respectively) was used in place of the nugget for estimating strength of spatial correlation. Whereas many previous studies used the experimental semi-variogram to facilitate geostatistical modeling, the method was utilized in the current work only to identify and quantify the contribution of potential vadose zone hydrologic process-drivers (e.g. plant water use).

Results and Discussion

Climate During Study

Average air temperature (i.e. measured at Sanborn Field) during the study period (October 2010 – September 2013) was 13.9 °C. Precipitation during the first two water years of the study period (2011 and 2012) was uncharacteristically low in the watershed, with total annual precipitation of 762 and 739 mm, respectively. The period June 2012 – August 2012, was characterized by extreme (D3) to exceptional (D4) drought (USDM Drought Severity Classification; Svoboda et al., 2002), and was ranked one of the hottest and driest on record for the region (Kutta and Hubbart, 2014). Conversely, precipitation during the 2013 water year was 960 mm. Variable climate during the study period afforded the opportunity to measure and compare the response of soil water distribution under contrasting land use/land cover to a range of hydroclimatic conditions.

Soil Volumetric Water Content

Average volumetric water content in the top 100 cm of soil during the study period was 33.1 and 32.8% at the Ag and BHF sites, respectively. Wilcoxon Signed-Rank results indicated daily average VWC at the two study sites to be significantly different ($p<0.01$). Although a difference of 0.3% may seem small, it corresponds to a field-scale (6400 m^2; Figure 1) difference of 19.2 m^3, approximately 19,000 liters, of vadose zone water. Site average VWC differed by 1.0, 0.1, and 1.1% during 2011, 2012, and 2013 water years, illustrating the effect of climatic variation on VWC. While the differences between annual and study period VWC averages appear minor, differences were statistically significant (CI=0.05). The current work was designed to address the spatiotemporal distribution of VWC throughout the profile (i.e. vadose zone water regime), patterns that cannot be adequately described with a single number. Thus, depth and time averaged values can be misleading. Regardless, considering study sites are characterized by similar soils (i.e. silt loam), similar landscape position, and close geographical proximity (i.e. 900 m), it is notable that average site VWC differed. On average, the BHF site exhibited greater VWC at 15 and 30 cm depths (33.5 and 33.4%, respectively), relative to the Ag (32.5 and 33.0% at 15 and 30 cm, respectively) (Table 1). Conversely, the Ag site displayed greater VWC at deeper depths (32.9 and 34.6% at 50 and 100 cm, respectively), relative to the BHF (32.4 and 31.5% at 50 and 100 cm, respectively). The impact of climate variability on VWC regime is shown in Figure 2. Despite lower than average precipitation during the 2011 water year, the soil water contents of all measured depths at both sites were more similar, likely a residual of above average precipitation during the previous 2010 water year (1651 mm). Conversely, study site soils displayed more variability in VWC during the 2012 and 2013 water years, particularly during the drier post-drought (June 2012 – August 2012) period. For example, beginning in June 2012, Ag VWC at 100 cm did not follow the same drying trend as other depths, potentially due to groundwater table height (see below) (Kellner and Hubbart, 2015). Also, BHF VWC at 15 and 30 cm appeared to recover more quickly post-drought than other depths, potentially due to higher soil porosity (i.e. higher infiltration and hydraulic conductivity) relative to deeper depths (Hubbart et al., 2011). Figure 2 also illustrates the VWC variability at the event scale. Whereas episodic sampling could miss the vadose zone response to individual precipitation events, continuous monitoring facilitates the observation of wetting and drying phases of single events. Although flooding events occurred during the study period (e.g. May 31, 2013), Figure 2 does not show a definable influence of peak events on VWC patterns. A possible explanation is that the duration of floodplain inundation was insufficient to saturate the full profile, and produce a noticeable VWC peak. Inconsistent trends in site comparisons of VWC averages and standard deviations illustrate the spatiotemporal variability of soil water. Soil water content variability is further highlighted in Figure 3, with contrasting distributions between sites and soil depths. Statistics illustrated by Figure 3 were generated for all (i.e. not average) observed VWC data. Maximum VWC was observed at 15 cm depth at the BHF (57%) and 100 cm at the Ag site (56.5%). However, at both sites, minimum values were observed at the 50 cm depth (6.9 and 11.8% at the Ag and BHF, respectively). The effects of the historic 2012 drought on soil water content are clearly visible in VWC reductions at both sites, at all depths, from 2011 to 2012 (Figure 4). Notably, despite precipitation during the 2013 water year comparable to the 64 year average (960 and 991 mm, respectively), VWC did not appear to have recovered at either site by the end of the 2013 water year, a result that demonstrates the potential long-term effect of climatically dry periods on soil water content and distribution. Discussion regarding seasonal precipitation-VWC dynamics is included below.

Table 1. Average soil volumetric water content (%) at five measured depths during study period (October 2010 – September 2013) at agricultural (Ag) and bottomland hardwood forest (BHF) sites, Hinkson Creek Watershed, Missouri, USA. Parentheses contain standard deviations.

	Ag			
Depth (cm)	2011 WY	2012 WY	2013 WY	Avg.
15	33.5 (3.2)	31.6 (5.2)	32.4 (3.9)	32.5 (4.2)
30	34.3 (3.2)	32.2 (4.7)	32.4 (3.7)	33.0 (4.0)
50	35.4 (4.5)	32.4 (6.5)	31.1 (6.5)	32.9 (6.1)
75	34.6 (4.6)	31.2 (6.9)	31.4 (6.4)	32.4 (6.2)
100	36.0 (5.2)	34.1 (5.9)	33.6 (5.6)	34.6 (5.7)
	BHF			
Depth (cm)	2011 WY	2012 WY	2013 WY	Avg.
15	34.1 (4.0)	33.4 (7.2)	33.2 (4.6)	33.5 (5.4)
30	35.2 (3.7)	32.5 (5.1)	32.3 (4.5)	33.4 (4.7)
50	34.9 (4.5)	31.0 (6.5)	31.2 (6.6)	32.4 (6.2)
75	33.8 (4.1)	33.3 (4.8)	33.0 (5.6)	33.4 (4.8)
100	34.0 (4.2)	31.1 (5.9)	29.4 (6.1)	31.5 (5.8)

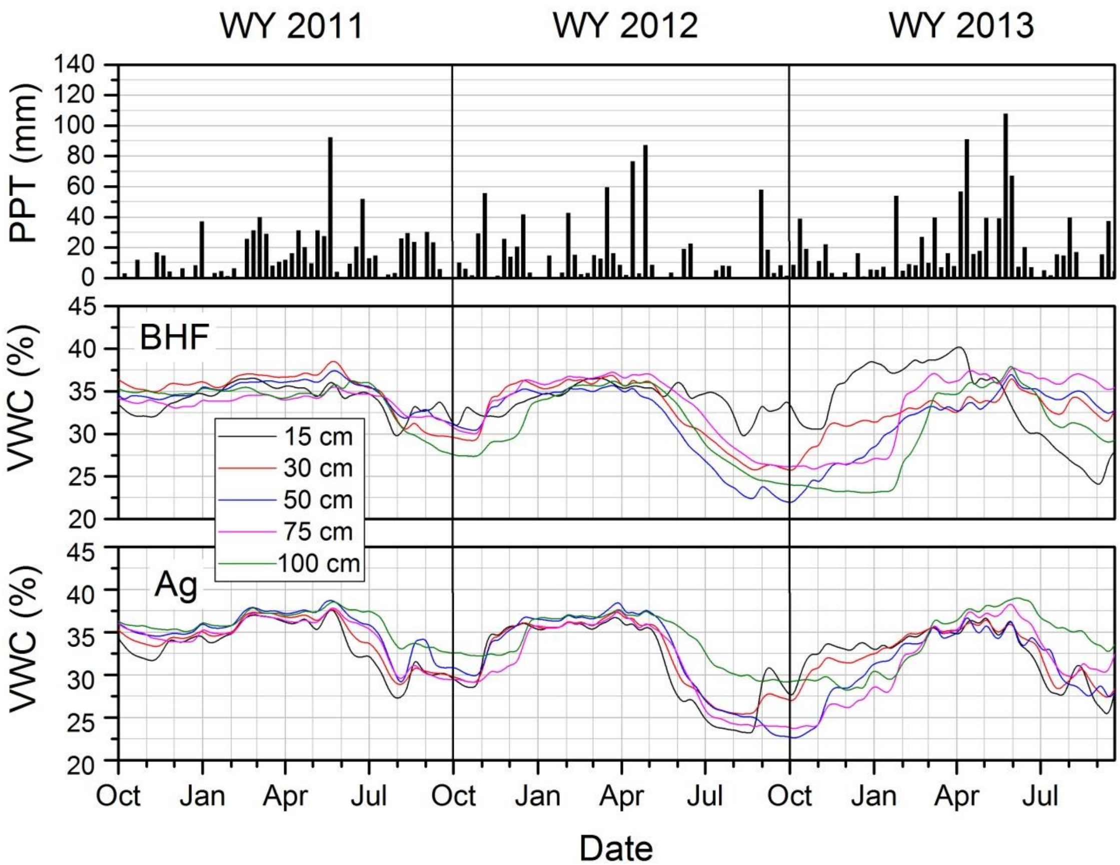

Figure 2. Weekly average precipitation (PPT) and soil volumetric water content (VWC) at five measured depths during 2011, 2012, and 2013 water years (WY) at agricultural (Ag) and bottomland hardwood forest (BHF) sites, Hinkson Creek Watershed, Missouri, USA. X-axis ticks and gridlines correspond to the first day of the month.

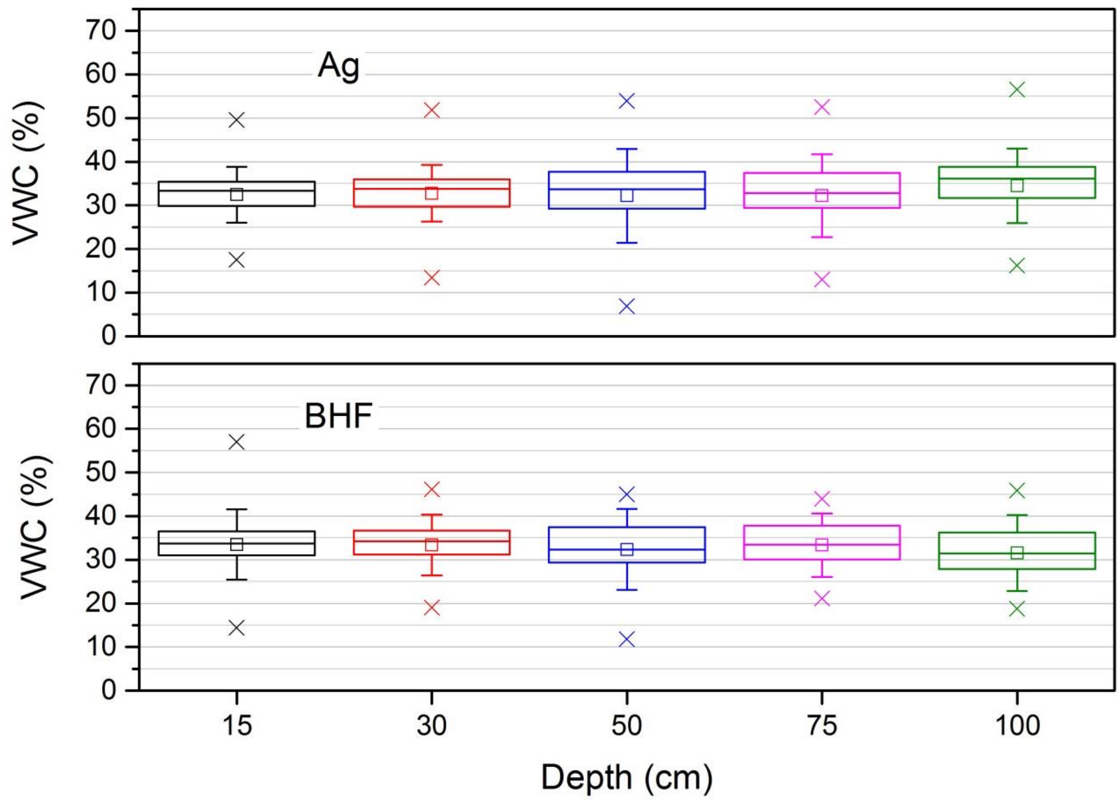

Figure 3. Box plots of soil volumetric water content (VWC) at five measured depths during 2011, 2012, and 2013 water years at agricultural (Ag) and bottomland hardwood forest (BHF) sites, Hinkson Creek Watershed, Missouri, USA. Boxes define standard deviation. Squares within boxes indicate mean. Midlines within boxes denote median. X's indicate maximum when above, minimum when below.

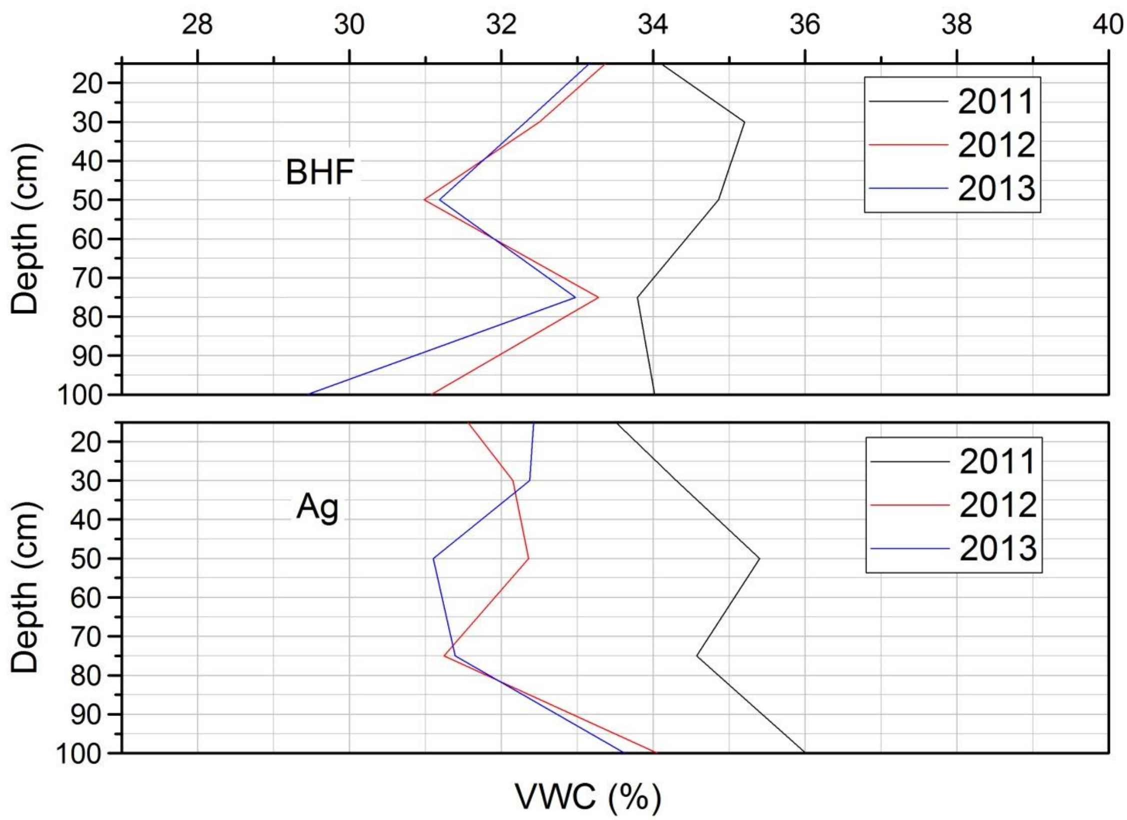

Figure 4. Average soil volumetric water content (VWC) from 15 to 100 cm during 2011, 2012, and 2013 water years at agricultural (Ag) and bottomland hardwood forest (BHF) sites, Hinkson Creek Watershed, Missouri, USA.

Significantly greater ($p<0.01$) average VWC at the Ag site (33.1%), relative to the BHF (32.8%) was largely attributed to the result of contrasting rates of plant water use between the sites. Woody species present at the BHF site such as cottonwood (*Populus deltoides*) are widely known for high potential transpiration (Pallardy and Kozlowski, 1981). For example, in a recent study conducted at the two study sites, results of a coupled vadose-phreatic zone model showed total evapotranspiration during 2011 and 2012 water years at the Ag and BHF sites to be 1603.3 and 1869.0 mm, respectively, a difference of more than 16% (Zell et al., 2015). Greater consumptive water use by woody vegetation at the BHF could explain lower average VWC at 50 and 100 cm depths (i.e.

tree rooting depth), relative to the Ag site. As Zell et al. (2015) showed greater evapotranspiration at the BHF site than the Ag, and the current work indicates a drier BHF subsurface at rooting depth, the results of the current work support the findings of Zell et al. (2015). Additionally, studies by Wahren et al. (2009) and Gonzalez-Sosa et al. (2010) showed that land use alterations to hydraulic conductivity can impact the residence time of subsurface water. Both studies noted greater subsurface hydraulic conductivity values at forested versus agricultural sites, with Wahren et al. (2009) noting the dominant influence of preferential flow paths in the forest subsurface resulting from decayed tree roots. Zell et al. (2015) reported higher values of measured and modeled estimated hydraulic conductivity, and Brooks-Corey Epsilon at the BHF relative to the Ag site. Lower conductivity of the Ag subsurface could further explain greater average VWC values at and below 50 cm, with slow-moving soil water accumulating as a result of subsequent infiltration events. Hubbart et al. (2011) reported statistical similarity of soil physical properties below 50 cm at the sites. However, considering the irregular spatial distribution of preferential flow paths in the subsurface, it is plausible that the Hubbart et al. (2011) study, which used a soil core method, would have missed preferential flow paths when collecting soil samples. Conversely, the current work use automated *in situ* monitoring, which is likely more capable, over time, of observing the effects of preferential flow paths in the subsurface (i.e. indirect observation). The greater average VWC observed at 15 and 30 cm depth at the BHF, relative to the Ag site, may be the result of surface soil physical characteristics. Hubbart et al. (2011) reported significantly ($p<0.05$) higher porosity and lower bulk density of BHF soil above 50 cm, relative to Ag site soil, presumably due to accumulation of organic matter in the shallow forest soil horizons. Additionally, solar radiation attenuation by forest canopy is known to reduce evaporative flux at the ground surface (Allen et al., 1998; Bulliner and Hubbart, 2013), another factor potentially contributing to greater average VWC in the BHF shallow subsurface. Hubbart et al. (2011) reported an average leaf area index (LAI) at the BHF of 3.11, indicating an average of 3.11 canopy layers per unit soil surface area. Conversely, since forest vegetation was removed from the Ag site, no forest canopy is present. Thus, the solar radiation attenuation potential of the BHF is likely greater than that of the Ag site due to forest canopy alone. Finally, the Ag site consistently displayed greater VWC at 100 cm. A recent study by Kellner and Hubbart (2015) identified a higher shallow groundwater table at the Ag site, relative to the BHF, with average water table depths of 2.66 and 3.13 m at the Ag and BHF sites, respectively. Although it is unlikely that capillary action would lift water more than a meter in sandy loam soils, a higher water table, shallower rooted plants, and a less hydraulically conductive subsurface could collectively explain consistently greater 100 cm VWC at the Ag site. Thus, patterns of vadose zone water content/distribution at the study sites suggest the influence of plant water use and subsurface hydrology, and highlight the long-term effects of forest removal on soil water regime.

The complex interactions between plant water use and subsurface hydrology are illustrated by the comparative VWC profiles in Figure 5. To better understand the impact of plant water use on VWC at the Ag and BHF, data were aggregated seasonally, with the period October – March representing the plant dormancy season, and April – September representing the period of plant growth. Despite more sum total precipitation received during the plant growth seasons (1475 mm) relative to the dormancy seasons (986 mm), both sites displayed reductions in VWC during plant growth from 15 - 50 cm depth, thereby demonstrating the primary influence of plant water use on shallow soil water content. It is of interest that the average BHF profile does not display a reduction in VWC at 75 cm during the plant growth season, as does the Ag profile. Considering greater typical rooting depth of woody vegetation than grass species (Scholes and Archer, 1997) and thus a potential to uptake water from greater depths (Eggemeyer et al., 2009), it is reasonable to expect seasonal reductions of VWC at greater depths at the BHF than the Ag site. However, that expectation is not supported by study results. The likely explanation is that the BHF subsurface is more hydraulically conductive than the Ag, and thus is capable of transmitting water more efficiently through the profile during the wet/plant growth season, a supposition supported by the findings of Zell et al. (2015). More efficient transmission of water, potentially due to routing by preferential flow paths (Wahren et al., 2009), could result in a wetting of the deeper profile (75 cm) during periods of increased precipitation (i.e. plant growth season), a process that highlights the potential for floodplain forests to more efficiently attenuate flood waters, relative to grassland/agricultural sites.

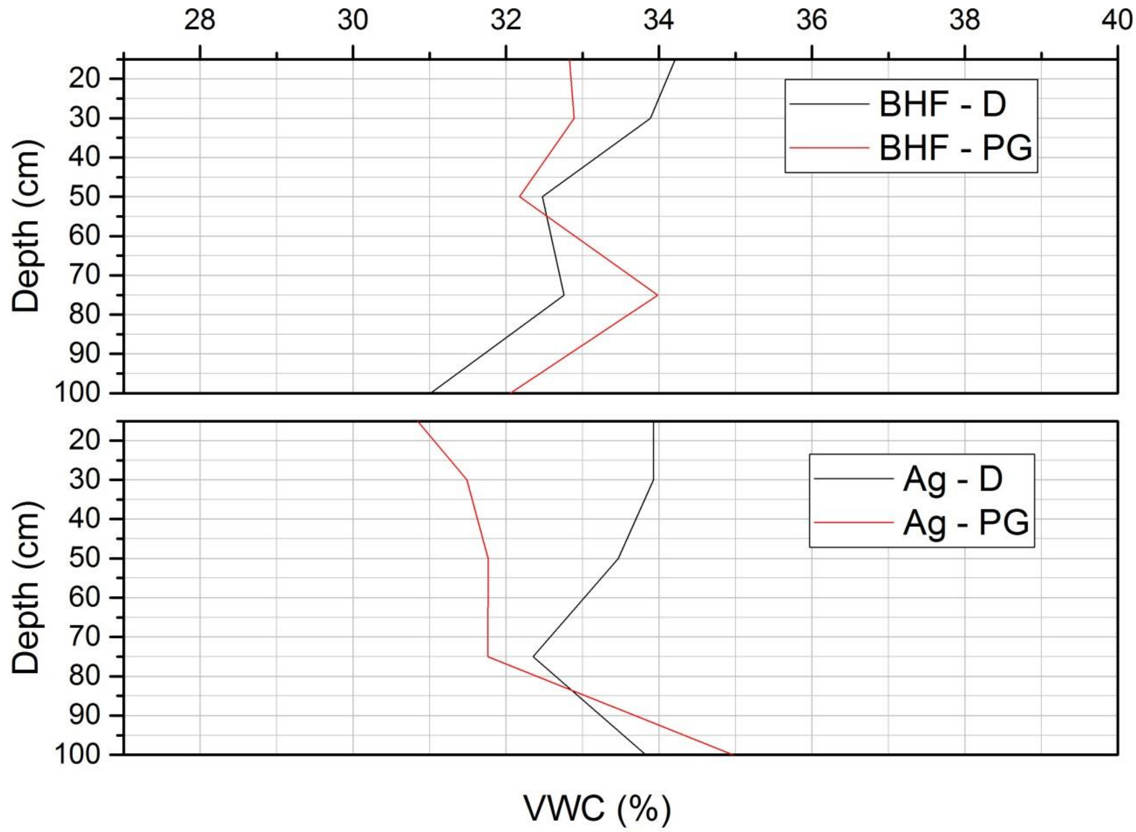

Figure 5. Average soil volumetric water content (VWC) from 15 to 100 cm during seasons of plant growth (PG) and dormancy (D) at agricultural (Ag) and bottomland hardwood forest (BHF) sites, Hinkson Creek Watershed, Missouri, USA.

VWC Spatial Correlation

Horizontal and vertical semi-variogram analyses were used to investigate the spatial correlation of soil water content at the study sites. As shown in Figure 6, the Ag site displayed stronger spatial correlation of VWC than the BHF in both the horizontal and vertical directions. Based on the criteria presented by Kazemi et al. (2008), Ag VWC is characterized by strong spatial dependency in both the horizontal (24%) and vertical (9%) directions. BHF soil water content did not exhibit a definable spatial correlation in the horizontal direction, but was characterized by a relatively short-range (25 cm), strong (12%) correlation in the vertical direction.

Results concerning the general spatial correlation and distribution of soil water are likely best explained by contrasting land use history. For example, canopy structure and preferential flow paths in the BHF subsurface likely contribute to spatial heterogeneity of hydrologic processes (e.g. stemflow, throughfall, infiltration, lateral flow, percolation) (Hagedorn et al., 1999; Bundt et al., 2000; Johnson and Lehmann, 2006), thus explaining the lack of consistent field-scale spatial dependence of soil water content. Results seemingly contrast with those of Nyberg (1996), who reported forest soil VWC spatial dependency to a range of 20 m. However, the current work applied a more coarse sampling scale, with a minimum horizontal separation of 40 m. Thus, it is likely that BHF VWC does exhibit horizontal spatial dependency, but at a shorter range than measured in the current work. Greater spatial dependency of Ag site VWC supports results of Robertson et al. (1993), who showed that field cultivation increased the range of spatial correlation for several soil physical properties, including water content. This is reasonable since conventional agricultural practices such as tillage often result in a mixing of the upper soil horizons (Hagedorn et al., 1999), and have been found to reduce spatial variation of VWC (Özgöz et al., 2007) and increase uniformity of soil physical properties (Tsegaye and Hill, 1998). A more physically uniform subsurface at the Ag site could permit the development of VWC spatial dependency resulting from field-scale variations in topography and texture. For example, Ag soil VWC generally displayed a positive trend with distance from the stream, with wetter soils located further from Hinkson Creek, and drier soils located adjacent to the creek (Figure 1). Therefore, results suggest removal of forest vegetation and subsequent cultivation of the Ag site lead to an effective homogenization of the upper soil profile, and facilitated the development of strong VWC spatial dependency.

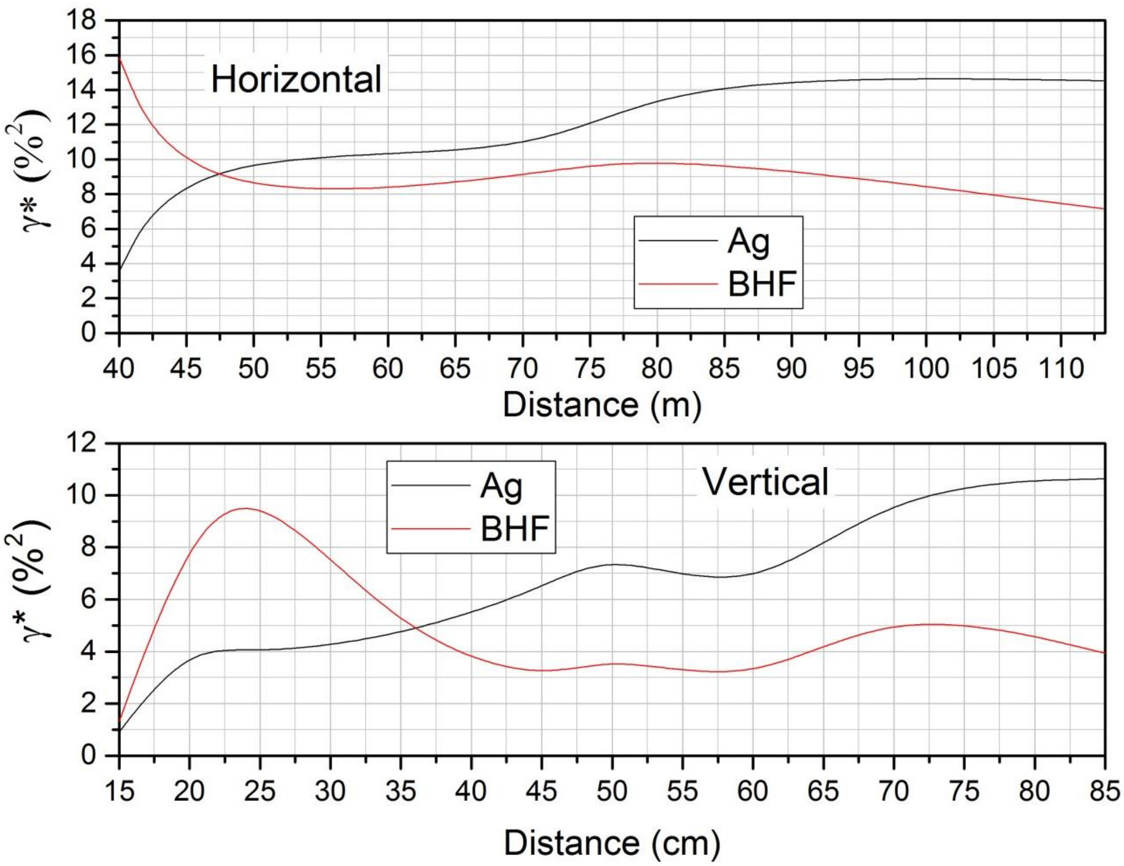

Figure 6. Experimental semi-variogram of soil volumetric water content in horizontal and vertical directions during 2011, 2012, and 2013 water years at agricultural (Ag) and bottomland hardwood forest (BHF) sites, Hinkson Creek Watershed, Missouri, USA.

To determine the influence of plant water use on the spatial correlation of VWC at the two sites, semi-variogram analyses were conducted on seasonally aggregated data, and plant growth and dormancy seasons (as described above) compared (Figure 7). Ag soil VWC exhibited stronger horizontal spatial dependency during the plant growth season (17%) than the dormancy season (39%), but similarly strong vertical spatial dependency during the two seasons (9%). BHF soil VWC displayed no horizontal spatial dependency in either season, but slightly stronger vertical spatial dependency during plant growth (11%) relative to dormancy (13%).

Seasonal contrasts in VWC spatial dependency illustrate the impact of plant water use on the distribution of soil water. López-Vicente et al. (2015) reported a reduction of VWC variation during wet conditions, relative to dry conditions. Therefore, climatically wet periods should correspond to *less* VWC variability. Results of the current work indicate lower horizontal and vertical semi-variogram values for Ag site VWC during the dormant season. Given the definition of the semi-variogram (equation 1), lower values correspond to smaller relative differences between observations (i.e. lower variation), thereby indicating that the dormant season was the “wet” period from a soil perspective. Furthermore, Ag site VWC standard deviation decreased from 5.9% during plant growth to 4.8% during the dormant season. Therefore, although more precipitation was received during the plant growth season, the spatial distribution of Ag site VWC suggests the plant growth season is the drier of the two seasons from a soil water perspective, given the rate(s) of plant water uptake during that time, consistent with observed results shown in Figure 5. In other words, results indicate that plant water use exerts a “drying” influence on the soil, thus counteracting precipitation influence on VWC distribution and spatial correlation. Ag site VWC standard deviation decreased from the plant growth season to the dormant season at all depths except 100 cm, where it remained the same; suggesting that the impacts of plant water use on soil water content do not extend to that depth. Results indicate less seasonal impact on BHF soil VWC, with no horizontal spatial dependency in either season, and only a slight decrease in vertical spatial dependency during the dormant season. In contrast to the Ag site, total BHF VWC standard deviation increased from the plant growth season (5.4%) to the dormant season (5.5%). However, at 30 and 50 cm specifically, BHF VWC standard deviation decreased from the plant growth season to the dormant season, consistent with Ag site results, suggesting that these two depths are most influenced by seasonal plant water use.

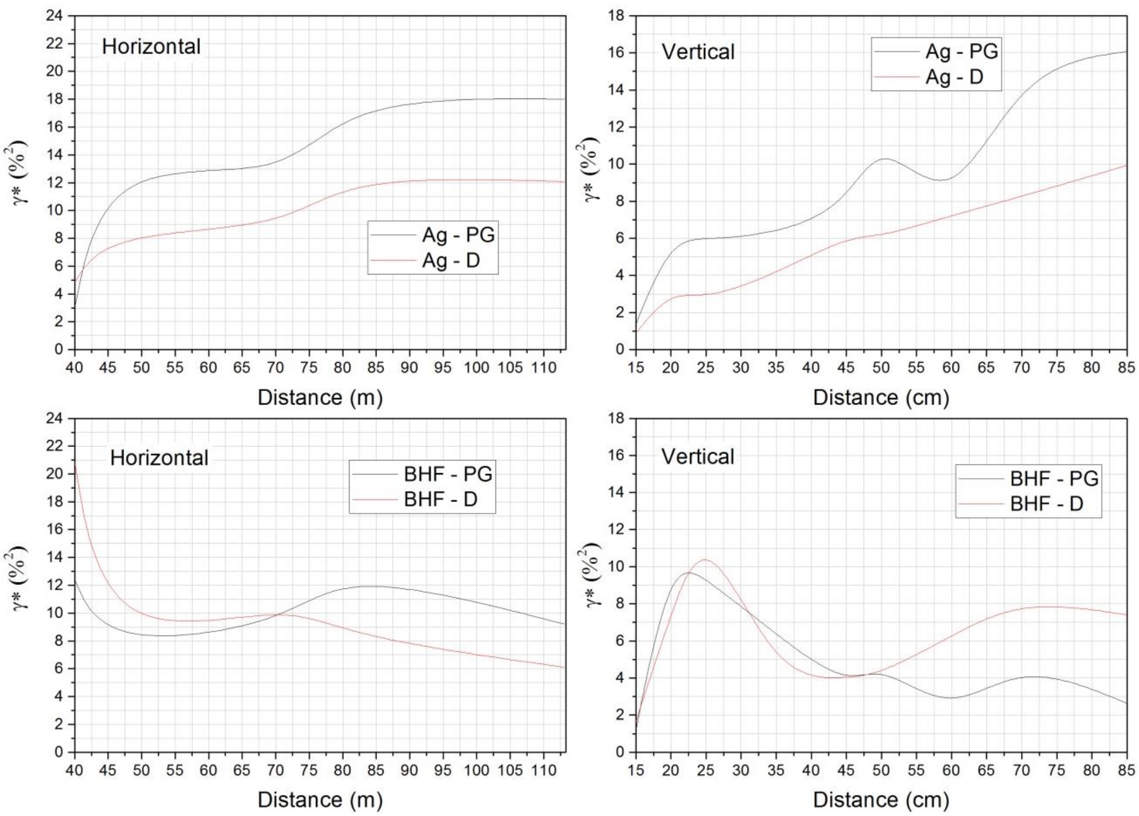

Figure 7. Experimental semi-variogram of soil volumetric water content in horizontal and vertical directions during seasons of plant growth (PG) and dormancy (D) at agricultural (Ag) and bottomland hardwood forest (BHF) sites, Hinkson Creek Watershed, Missouri, USA.

Results concerning the spatial correlation of soil VWC at specific depths are illustrated by Figures 8 and 9, and are likely best explained by contrasting vegetative influences on the soil profile. At both sites, spatial dependency of VWC was absent at 50 cm. However, excepting 50 cm, VWC spatial dependency increases with depth at the Ag site, from 23% at 15 cm to 2% at 100 cm. The strong spatial correlation observed in Ag site VWC at 100 cm could be the result of the higher shallow groundwater table identified by Kellner and Hubbart (2015). Notably, BHF VWC displayed strong spatial dependency at 15 cm (11%) and 30 cm (<1%), but no spatial dependency below 30 cm. Strong spatial correlation in BHF VWC at 15 and 30 cm could be the result of high average porosity (i.e. lateral hydraulic conductivity) in BHF shallow soil reported by Hubbart et al. (2001). The consistent absence of VWC spatial correlation at 50 cm is presumably due to the primary influence of plant rooting depth and water use at 50 cm. As shown in Figure 2, minimum VWC values at both sites were observed at 50 cm and both grassland and forest plant species are known to root at 50 cm (Scholes and Archer, 1997; Eggemeyer et al., 2009). However, the extension of vegetative influences past 50 cm at the BHF site suggests that impacts of forest vegetation on VWC extend to greater depths than that of agricultural/grassland vegetation. Thus, results suggest BHF vegetation contributes to a more heterogeneous vadose zone (presumably containing preferential flow paths at deeper depths), capable of more efficiently processing (i.e. infiltration, through-flow, percolation) floodwaters than the wetter, more homogenous, less conductive Ag site vadose zone.

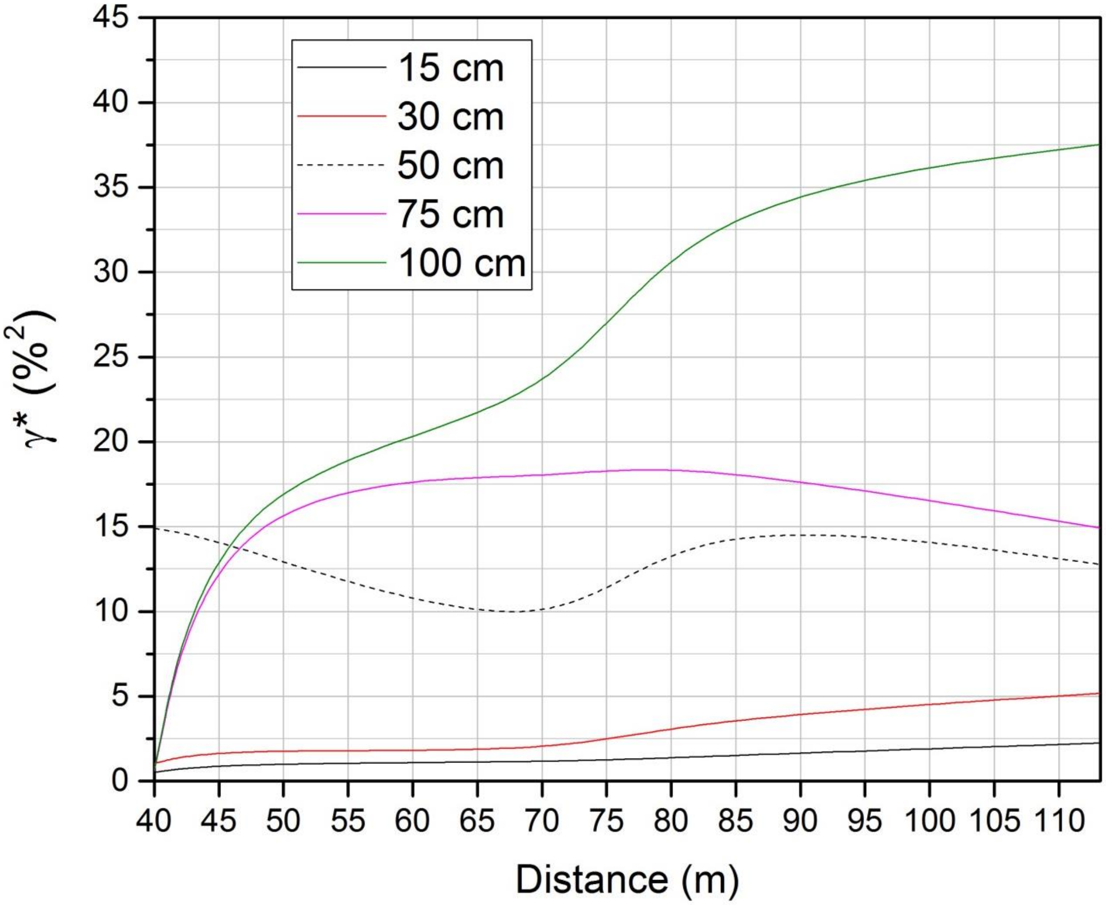

Figure 8. Experimental semi-variogram of soil volumetric water content in horizontal direction at five measured depths during 2011, 2012, and 2013 water years at agricultural (Ag) site, Hinkson Creek Watershed, Missouri, USA.

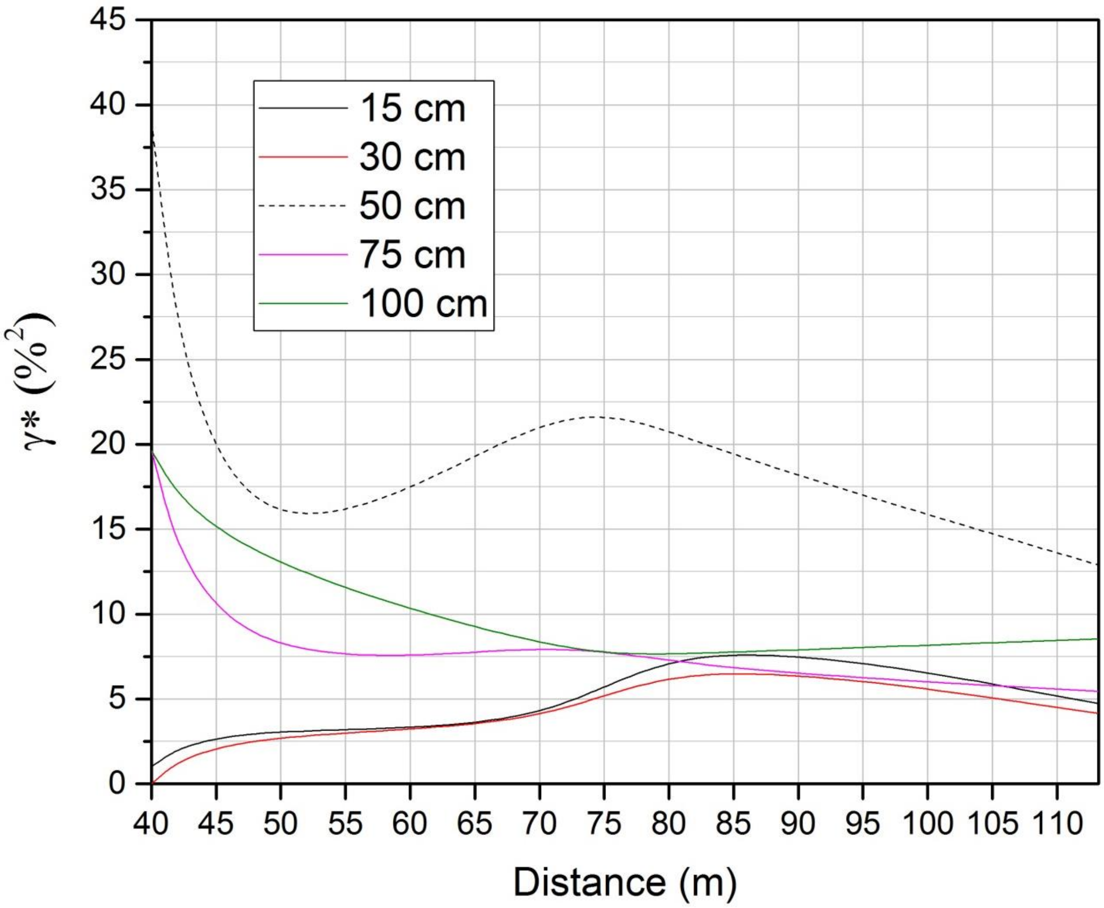

Figure 9. Experimental semi-variogram of soil volumetric water content in horizontal direction at five measured depths during 2011, 2012, and 2013 water years at bottomland hardwood forest (BHF) site, Hinkson Creek Watershed, Missouri, USA.

The use of continuous, automated, *in situ* monitoring in the current work provided the opportunity to characterize the soil water regime based on a temporally-rich dataset, as opposed to episodic sampling, which is often incapable of accurately capturing spatiotemporal variability of VWC. Moreover, the incorporation of semi-variogram analyses identified potential drivers (i.e. plant water use, preferential flow paths) of observed differences. Whereas previous VWC studies incorporating semi-variogram analyses predominately applied the method to the qualitative description of soil water spatial correlation (e.g. Nyberg, 1996) or as a preliminary step in the kriging process, the current work used the method to identify and quantify spatiotemporal VWC patterns and support mechanistic hypotheses, thereby making results of the analyses more informative. Additionally, by combining continuous automated sampling and semi-variogram analyses the study quantified and contrasted the extent (i.e. depth) of vegetative influence on soil water regime, important information for land and water resource managers seeking effective water quality and flood mitigation strategies.

Conclusions

Considering the increasing pace of global land use/land cover change, and the importance of soil water content to ecosystem biogeochemical processes, it is necessary to improve understanding of the long-term impacts of historic land cover change (e.g. forest removal) on the spatiotemporal distribution of soil water. Continuous, automated VWC sensors were installed at five depths (i.e. 15, 30, 50, 75, and 100 cm) at six locations within two floodplain study sites, a remnant bottomland hardwood forest (BHF) and historic agriculture site (Ag). VWC was measured at 30 minute intervals for the duration of the 2011, 2012, and 2013 water years. Results showed VWC to be significantly different between sites ($p<0.01$) during the study, with site averages of 33.1 and 32.8% at the Ag and BHF sites, respectively. While a difference of 0.3% may seem small, it corresponds to a field-scale (6400 m^2) difference of 19.2 m^3, or more than 19,000 liters, of vadose zone water. On average, the BHF site had greater VWC at 15 and 30 cm depths, while the Ag site displayed greater VWC at deeper depths (50 and 100 cm). However, inconsistent trends in site comparisons of VWC averages and standard deviations illustrate the spatiotemporal variability of soil water. Semi-variogram analyses indicated the presence of strong (<25%) horizontal and vertical spatial correlation at the Ag site, and a relatively short-range (25 cm) strong vertical spatial correlation at the BHF, but only indicated horizontal VWC spatial correlation in the top 30 cm of the BHF profile.

Considering similar soils, landscape position, topography, and close geographical proximity of the sites, it is reasonable to conclude that land use/land cover is the primary factor responsible for the observed differences in the current work. Historic (>100 years) removal of forest vegetation and subsequent cultivation of the Ag site lead to an effective homogenization of the upper soil profile, and facilitated the development of strong VWC spatial correlation. Moreover, the deep Ag soil profile does not exhibit evidence of preferential flow paths as does the BHF. Higher hydraulic conductivity of the BHF subsurface likely results in a wetting of the deeper profile (75 cm) during climatically wet periods, thus suggesting more efficient processing of water by the floodplain forest vadose zone. Collectively, results highlight the extent and degree to which forest vegetation impacts subsurface hydrology and the potential for floodplain forests to more effectively mitigate flood risk, as compared to grassland/agricultural sites. This study is one of the first to use continuous, automated, *in situ* monitoring technology and geostatistical analytical methods to describe relationships in vadose zone water regime between forest and agricultural floodplain land uses. The new information provided is of immediate use to natural resource managers and emphasizes the value of floodplain forest establishment as a strategy to mitigate flood risk, and is applicable to similar land use patterns across the landscape.

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CHAPTER III

AGRICULTURAL AND FORESTED LAND USE IMPACTS ON FLOODPLAIN SHALLOW GROUNDWATER TEMPERATURE REGIME

In Press:

Kellner, E., Hubbart, J.A. 2015. Agricultural and Forest Land Use Impacts on Floodplain Shallow Groundwater Temperatures. *Hydrological Processes* x:xx.

Introduction

Background

Groundwater temperature is an important physical determinant of surface water quality in groundwater-fed surface water systems, influencing the metabolic rates, physiology, and life history traits of aquatic species, interstitial habitat suitability, and the nutrient cycling and productivity of aquatic communities (Poole and Berman, 2001; Sophocleous, 2002). Relative to surface water, groundwater typically exhibits less seasonal temperature variability. Thus, groundwater discharge to streams can buffer surface water temperatures and contribute to the presence of in-stream areas of suitable habitat for temperature sensitive aquatic organisms (i.e. thermal refugia) (Kurylyk et al., 2014). Abrupt groundwater temperature change from background or reference conditions can alter biogeochemical reactions, eventually influencing groundwater quality (Mitchell and Ferris, 2005; Taniguchi et al., 2007; Figura et al., 2011). For example, temperature alterations can influence carbonate precipitation, salt solubility, dissolution of silicate minerals, and the mobilization of organic compounds. An increase in groundwater temperature of 10 °C can result in a two to fourfold increase in the rate of chemical reactions (Hähnlein et al., 2013). Prommer and Stuyfzand (2005) showed that oxidation reactions in groundwater depended significantly on spatial and temporal groundwater temperature variation. Yates et al. (1985) showed that temperature was the most important predictor of virus persistence in well water. Brielmann et al. (2009) reported that groundwater temperature significantly affected bacterial and faunal community composition, identifying it as the second most important driver of microbiological community variation after surface water influence. Groundwater bacterial diversity was shown to increase with temperature, while faunal diversity decreased. This is important considering bacterial roles in catalyzing certain geochemical processes (e.g. the oxidation of Fe^{+2} to Fe^{+3}) that are known to be temperature dependent (Bonte et al., 2011; Jesubek et al., 2013). Considering the importance of groundwater temperature to a variety of surface and subsurface biogeochemical processes, studies are needed that accurately characterize groundwater temperature regimes.

Two thermal groundwater zones exist: the upper or surficial zone, which typically includes the soil surface to 10-15 m below the surface; and the lower zone, below 10-15 m (Vandenbohede et al., 2011).While the temperature of groundwater in the lower zone is most often a function of the geothermal gradient, past climate, and deep groundwater flow, groundwater temperature in the surficial zone is primarily influenced by seasonal heating and cooling of the ground surface and recharge water (Kukkonen et al., 1994; Anderson, 2005; Vandenbohede et al., 2011). Shallow groundwater temperature is influenced by numerous factors, including soil physical properties (e.g. thermal diffusivity), solar radiation, ground surface temperature, precipitation, water table depth, groundwater flow, and exchanges with surface water (Bundschuh, 1993; Hillel, 1998; Saar, 2011; Wang et al., 2014). Any land use change that alters the magnitude or direction of any such influential components can subsequently impact the groundwater temperature regime. For example, analyzing the results of a number of transient groundwater flow simulations, Bense and Beltrami (2007) found that horizontal groundwater flow became a significant source of lateral heat transport (i.e. advection) above flow velocities of 10^{-8} m s^{-1}. A comparative modeling study by Ferguson and Beltrami (2006) suggested the potential for deforestation to produce subsurface temperature alterations. Hewlett and Fortson (1982) hypothesized that observed stream temperature increases were attributable to altered baseflow temperature resulting from upland clearcutting. Such an effect is plausible considering the impact of solar radiative heating on ground thermal regimes (Beltrami and Kellman, 2003) and solar radiation attenuation capacity of forest canopies (Bulliner and Hubbart, 2013). Removal of the forest canopy could increase surface heating due to increased solar radiation at the ground surface, and thus alter baseflow temperature via soil heat flux. In a case study comparing the surface energy balance at forest and agricultural sites, Zell and Hubbart (2013) reported greater average, maximum, and minimum daily change in energy storage (W m^{-2}) at the agricultural site, relative to the forest, thus suggesting greater surface warming at the exposed agricultural field. Similarly, land use changes have been shown to alter subsurface hydraulic properties such as infiltration rate, percolation, and soil water retention (Wahren et al., 2009; Hubbart et al., 2011), all of which can impact rates of thermal energy flow in the lithosphere. For example, studies by Wahren et al. (2009)

and Gonzalez-Sosa et al. (2010) found land use alterations to hydraulic conductivity, which can impact the residence time of groundwater and thus temperature regime. Specifically, both studies noted greater subsurface hydraulic conductivity values at forest versus agricultural sites, with Wahren et al. (2009) noting the dominant influence of preferential flow paths in the forest subsurface resulting from decayed tree roots.

Although many previous studies focused on the relationship between land use and groundwater chemical quality or baseflow hydrology, relatively little research has been conducted on the effect of land use on groundwater temperature. Taniguchi et al. (2007) compared subsurface temperature inversion depths in borehole profiles to determine surface warming effects of urban land use. They reported surface warming influence to depths of 140 m in Tokyo, Japan. Taylor and Stefan (2009) projected an increase in shallow groundwater temperature of approximately 3 °C as a result of watershed-scale urbanization. In Karlsruhe, Germany, Menberg et al. (2013) identified an increasing rate of urban heat flux into the shallow aquifer from ground surface warming and subsurface anthropogenic structures (e.g. basements and sewers) between 1977 (median=759 mW m^{-2}; standard deviation=89 mW m^{-2}) and 2011 (median=828 mW m^{-2}; standard deviation=143 mW m^{-2}). Eggleston and McCoy (2014), studying the impact of urbanization on coastal aquifer thermal regimes, reported a negative correlation between canopy cover and groundwater temperatures at 30 m depth. Kurylyk et al. (2015), in a comparative modeling study using several analytical solutions to the one-dimensional equation for subsurface heat transport, found that land cover disturbances (i.e. forest removal) could result in warming of shallow aquifers. Henriksen and Kirkhusmo (2000) identified shallow groundwater warming following forest clearing in Norway, where in an aquifer with a temperature ranging from 1–10 °C, 89% of observations of groundwater temperatures greater than 7 °C occurred after forest removal. Similarly, Guenther et al. (2014) also noted increased groundwater temperatures after forest clearing in coastal headwater catchment in British Columbia, Canada.

Despite progress of previous investigations, many of which focused on urban land use types and/or applied one-dimensional models, few studies have been published pertaining to rural land use impacts on shallow groundwater temperatures. Further, the majority of previous studies utilized episodic or opportunistic sampling methods, and involved the incorporation of critical assumptions (e.g. no horizontal groundwater flow by Taniguchi et al. [2005, 2007]). Few studies can be found in the literature utilizing continuous, automated, *in situ* sampling methods to analyze groundwater temperature relationships and dynamics. Such methods can produce high-resolution quantifications of groundwater temperature responses to land use altered environmental variables (e.g. air temperature, solar radiation) and subsurface characteristics and processes (e.g. hydraulic conductivity, groundwater flow).

Objectives

Given the lack of information in the primary literature, and the importance of groundwater temperature to the biogeochemical health of aquatic ecosystems, there is a need for studies incorporating long-term, continuous groundwater monitoring to characterize the shallow groundwater (SGW) temperature regimes (i.e. annual and seasonal trends/patterns) at sites with contrasting land use histories, thereby supporting or refuting previous conclusions based on theory alone or episodic sampling techniques.

Such information will help decision-makers more effectively manage freshwater resources. Therefore, the objectives of the following work were to 1) characterize and compare groundwater temperatures 1-2 m below the water table at agricultural and forest sites, and 2) relate observed SGW temperature contrasts to site differences in estimated groundwater flow and microclimate.

Methods

Site Description

This research was conducted in the Hinkson Creek Watershed (HCW) located in central Missouri, USA (Figure 1). The project is nested within an experimental watershed study implemented in 2008. The HCW contains approximately 60% of the city of Columbia, a community of 108,000 (USCB, 2011). Land use in the HCW is approximately 34% forest, 38% agriculture, and 25% urban (Hubbart et al., 2011), making it a regionally representative location for studying the effects of different land use types on groundwater resources. According to a 64 year climate record of Columbia, Missouri (station ID #231790, 231791), average annual temperature and precipitation within the watershed were 12.5 °C and 991 mm yr^{-1}, respectively (Missouri Climate Center, 2014). Two floodplain reaches of Hinkson Creek with contrasting land use histories were selected for data collection: a remnant (i.e. no previous harvest) bottomland hardwood forest (BHF), and a historic (i.e. currently fallow) agricultural field (Ag). Woody vegetation at the BHF site includes Silver maple (*Acer saccharinum*), Eastern cottonwood (*Populus deltoides*), Boxelder (*Acer negundo*), American elm (*Ulmus americana*), Black walnut (*Juglans nigra*), and Sycamore (*Platanus occidentalis*)

(Hubbart et al., 2011). Vegetation at the Ag site is primarily gramminoids such as Johnsongrass (*Sorghum halepense*), and tall fescue (*Festuca arundinacea*), and the forbs goldenrod (*Solidago giganteana*), and Ironweed (*Vernonia altissima*) (Brown, 2013). Both sites contain Haymond silt loam alluvial soils (NRCS, 2015). Average site elevation is 174.4 and 176.3 m.a.s.l. for the Ag and BHF, respectively. Hubbart et al. (2011) reported statistical similarity ($p>0.05$) of physical soil characteristics (i.e. bulk density and porosity) below 50 cm depth at the two sites. Above 50 cm, statistical differences ($p<0.05$) between study site soil characteristics indicated impacts of surface vegetation on soil hydraulic properties, specifically infiltration capacity and volumetric water content (Hubbart et al., 2011). Given both study sites contain alluvial silt loam floodplain soils as per the results of Hubbart et al. (2011), it is reasonable to assume that the thermal properties (e.g. thermal diffusivity) of study site soils are comparable.

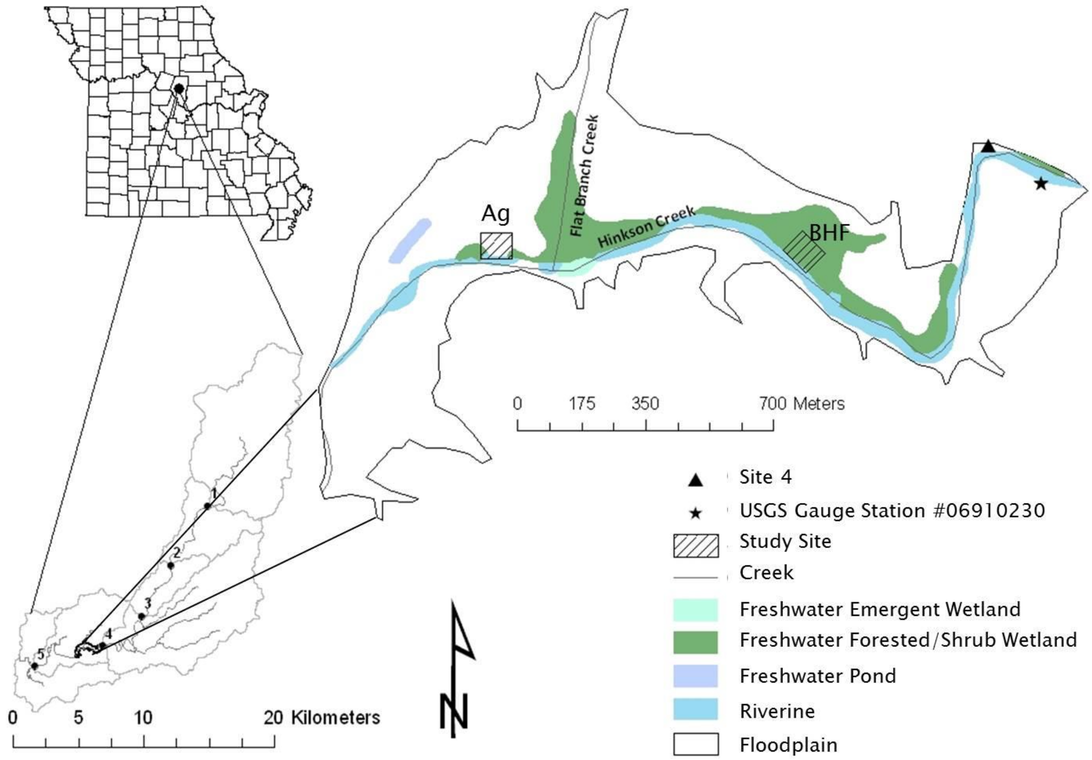

Figure 1. Shallow groundwater piezometer sites in lower Hinkson Creek Watershed floodplain, central Missouri, USA.

Data Collection

To monitor and characterize SGW at the study sites, an 80 x 80 m grid of nine equidistant piezometers (i.e. each 40 m apart) was installed at each site (Figure 1). The grid approach provided the opportunity for spatial analysis of SGW temperature. A similar design was used by Haycock and Burt (1993), who investigated nitrate concentration in floodplain groundwater in a 24 x 16 m grid of 24 boreholes. In the current study, each site comprised three rows of three piezometers located 10, 50, and 90 m from the stream, often referred to as rows 1, 2, and 3, respectively, throughout the proceeding text. Steel drive-point piezometers 5.49 m long with a 4 cm inner diameter and a 0.76 m screened lower segment (i.e. drive point) were driven into the unconfined alluvial aquifer. On average, the top of the well casing extended 24 cm above the ground surface. Alexander and MacQuarrie (2005) showed that steel standpipes do not preferentially conduct heat to the subsurface, and that temperatures within steel piezometers accurately reflect temperatures in surrounding groundwater. Each piezometer was equipped with a Solinst Levelogger Gold (Solinst Canada, Ltd.), which includes a pressure transducer to measure water level (error: $\pm$0.003 m), and a temperature sensor (error: $\pm$0.05 °C). Sensors were placed at the bottom of the piezometers, at a depth of approximately 5.49 m below ground surface. Data were logged at 30 minute intervals for the duration of the 2011, 2012, 2013, and 2014 water years (October 1, 2010 – September 30, 2014). Water table depth and water table elevation were calculated using the results of study site surveys to correct for differences in piezometer elevation. Water table depth was included considering the study's focus on shallow groundwater responsiveness to changing surface conditions.

Streamflow data for Hinkson Creek were collected at US Geological Survey gauging station (#06910230), located approximately 2000 m upstream of the sites, and a project-related monitoring site located approximately 8000 m downstream of the study sites. Stream temperature data for Hinkson Creek were collected at 5 minute intervals for the duration of the 2011, 2012, 2013, and 2014 water years at a monitoring site located 1500 m upstream from the two study sites (i.e. Site 4 shown in Figure 1). Air temperature and precipitation were measured at Sanborn Field, located approximately 4 km northeast of the study sites on the University of Missouri campus, to obtain climate data representative of the Hinkson Creek Watershed. Additionally, microclimate stations were installed in 2011 adjacent to central piezometers of each row to monitor soil temperature. Each station included a Hobo Onset S-TMB soil temperature probe installed at 30 cm depth. Microclimate stations collected data at 30 minute intervals for the duration of the 2012, 2013, and 2014 water years. The one water year (i.e. 2011) discrepancy between SGW data and microclimate data was the result of project financial limitations.

Data Analysis

Descriptive statistics and percent differences were calculated based on SGW temperature data collected at each piezometer, and averaged by row and land use type. SGW temperature lag time was calculated by comparing observed seasonal peak SGW temperatures with corresponding observed peak air temperatures as per the methods of Keery et al. (2007). Considering hydroclimate data are often non-normally distributed (McCuen, 2003), the Wilcoxon Signed-Rank test was used for statistical comparison of groundwater temperature data. Wilcoxon Signed-Rank test is an appropriate non-parametric choice for paired (e.g. temporally) non-normal data sets (Helsel and Hirsch, 1992).

Site hydraulic conductivity was previously determined by Zell et al. (2015) from the results of a series of falling head slug tests. Slug test results indicated average hydraulic conductivity to be 1.0 and 3.5 m day^{-1} at the Ag and BHF sites, respectively (Zell et al., 2015). Values for site hydraulic conductivity, hydraulic head measured at each piezometer, and stream stage were used to estimate flow as per the method presented by Devlin (2003). Flow was estimated according to Darcy's equation:

$$q = -K\nabla H \qquad \text{Eq. 1}$$

where q is the bulk flow rate (m day^{-1}), K is hydraulic conductivity (m day^{-1}), and H is hydraulic head (m) (Darcy, 1856).

Soil and groundwater temperature data were used to estimate thermal damping depth and thermal diffusivity as per the equation:

$$A_0 = A_z e^{-z/d} \qquad \text{Eq. 2}$$

where A is the temperature amplitude (°C) at a given depth z (m), and d is damping depth (m) (Hillel, 1998). Damping depth is a function of soil thermal properties, and is given by the equation:

$$d = \sqrt{\frac{2D}{\omega}} \qquad \text{Eq. 3}$$

where D is thermal diffusivity (m^2 sec^{-1}) and ω is the radial frequency of the temperature fluctuation (sec^{-1}) (Hillel, 1998). Equations 2 and 3 incorporate several assumptions, including one-dimensional vertical heat transport, negligible convective heat transport, a homogenous soil profile, and temporally constant soil thermal properties. Nonetheless, they are useful tools for interpreting subsurface thermal regime, especially in a field situation where soils are relatively homogenous, as in the current work.

Results

Climate During Study

Average air temperature (i.e. measured at Sanborn Field) during the study period was 13.4 °C (Table 1). Relative to the 64 year precipitation annual average of 991 mm yr^{-1} (Missouri Climate Center, 2014), precipitation during the first two years of the study period was uncharacteristically low in the watershed, with total annual precipitation of 762 and 739 mm during the 2011 and 2012 water years, respectively (Figure 2). Conversely, precipitation during the 2013 and 2014 water years was 960 and 867 mm, respectively. The range of hydrologic conditions during the study period was further observed in streamflow statistics of Hinkson Creek. During the four water years, average, maximum, and minimum flows in Hinkson Creek were 1.52, 314.13, and 0.00 $m^3 s^{-1}$, respectively (Figure 3). Average stream temperature during the study period was 14.2 °C. Average water table depth for the four water years was 2.66 and 3.13 m at the Ag and BHF sites, respectively, an 18% difference. Average hydraulic head for the four water years was 177.10 and 178.48 m.a.s.l. at the Ag and BHF sites, respectively. Considering greater average water table depth at the BHF site, higher average head at the BHF site is likely a result of higher elevation relative to the Ag site (i.e. 174.4 and 176.3 m.a.s.l. for the Ag and BHF, respectively). Maximum streamflow (314.13 $m^3 s^{-1}$) and minimum water table depth at both sites (-3.18 and -2.74 m at the Ag and BHF sites, respectively, indicating flood conditions) all occurred on May 31, 2013, as shown in Figures 2 and 3.

Table 1. Descriptive statistics of climate variables and shallow groundwater of agricultural (Ag) and bottomland hardwood forest (BHF) sites during 2011, 2012, 2013, and 2014 water years, Hinkson Creek Watershed, central Missouri, USA.

Climate Variable	Avg.	Std. Dev.	Max.	Min.
Ta (°C)	13.4	10.9	33.8	-20.1
PPT (mm)	3327.9*	7.6	130.8	0.0
Streamflow (m^3 s^{-1})	1.5	8.8	314.1	0.0
Tw (°C)	14.2	9.2	31.0	-0.3
Ag Water Table Depth (m)	2.7	0.4	5.4	-2.4
BHF Water Table Depth (m)	3.1	0.5	5.4	-1.9

Ta = air temperature
PPT = precipitation
* = cumulative
SGW = shallow groundwater
Tw = stream temperature
Note: Ta, PPT, and Tw statistics are for daily average data; flow statistics are for 15 min. data; and SGW level statistics are for 30 min. data.

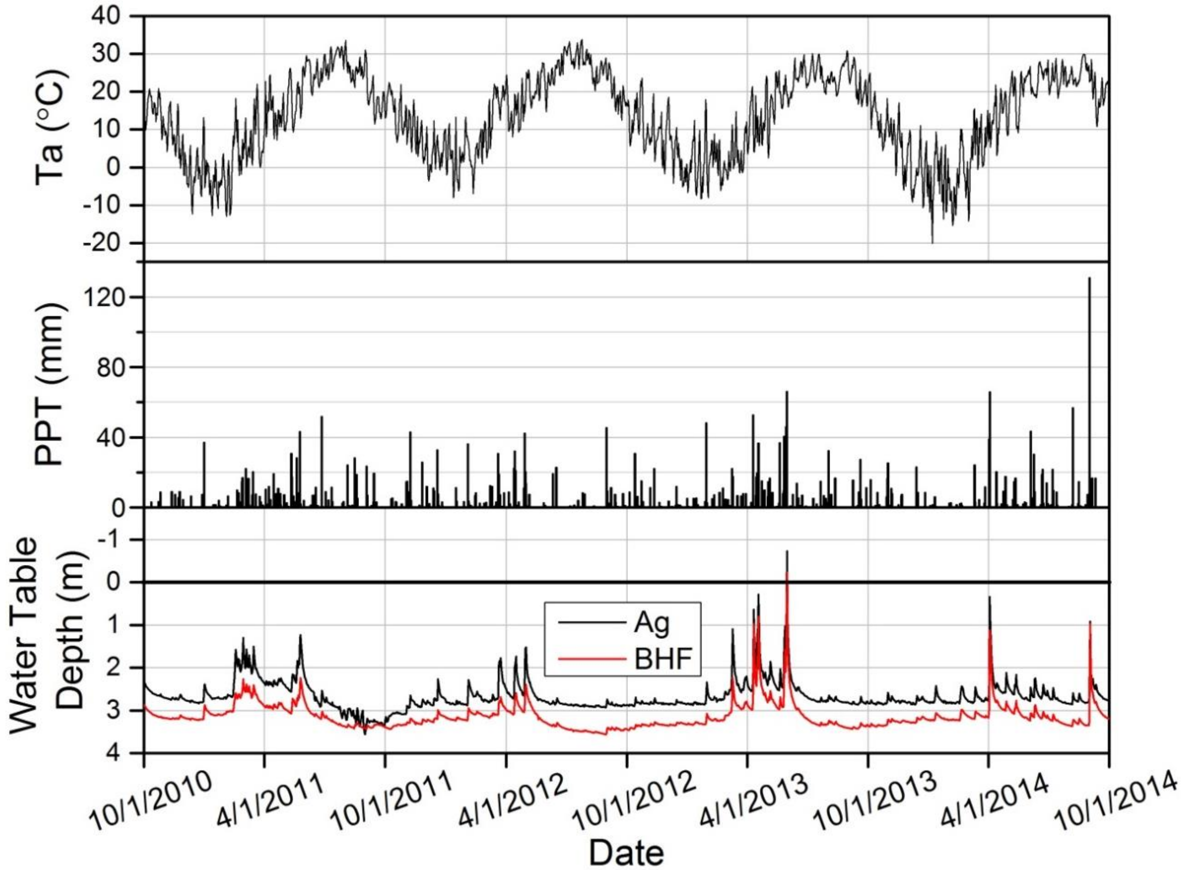

Figure 2. Average daily air temperature (Ta), precipitation (PPT), and water table depth of agricultural (Ag) and bottomland hardwood forest (BHF) sites during 2011, 2012, 2013, and 2014 water years, Hinkson Creek Watershed, central Missouri, USA. The bold line at 0 m on the water table plot indicates the ground surface.

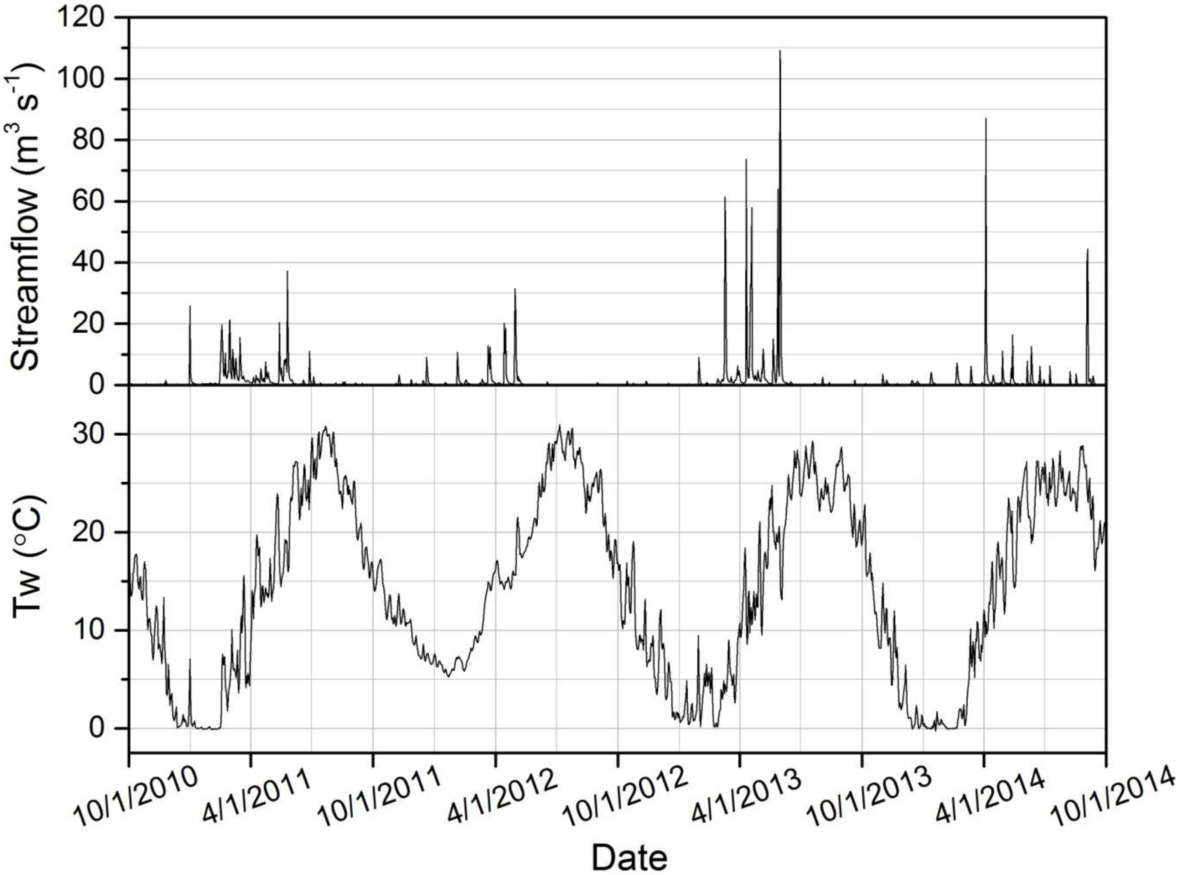

Figure 3. Streamflow and temperature (Tw) of Hinkson Creek during 2011, 2012, 2013, and 2014 water years, central Missouri, USA.

Groundwater Flow

Results of the Devlin (2003) method indicate average groundwater flow toward Hinkson Creek during the study period was 0.009 m day^{-1} at the BHF site and 0.011 m day^{-1} at the Ag site, a relatively small difference (Figure 4). Low average groundwater flow rates are reasonable considering the low hydraulic gradient of the topographically flat floodplain study sites. However, the flow regime at the BHF site was more seasonally dynamic, illustrated by the higher maximum groundwater flow rate of 0.024 m day^{-1}, versus the Ag site maximum of 0.018 m day^{-1}. High average groundwater flow rates observed at both sites during the beginning of the study period (i.e. fall 2010), and the subsequent rapid decline in flow rate, are likely the result of above average precipitation during the 2010 water year, which was 1651 mm, more than 66% greater than the 64 year average of 991 mm yr^{-1} (Missouri Climate Center, 2014). Application of the Devlin (2003) method to two-dimensional hydraulic head data also produces information regarding flow direction, via an angle of the principle flow path relative to the x-axis, represented in this case by Hinkson Creek. The average angle of groundwater flow relative to the x-axis was 41° at the BHF site, and 76° at the Ag site. Therefore, during the study period, groundwater at the Ag site flowed more directly towards Hinkson Creek, relative to the BHF site. During September and October, 2011, average groundwater flow at the Ag site was directed *into* the floodplain from Hinkson Creek, an effective reversal of baseflow direction. Likewise, baseflow reversals occurred at the BHF site during August and September, 2012, and August 2014, both of which correspond with climatically dry periods.

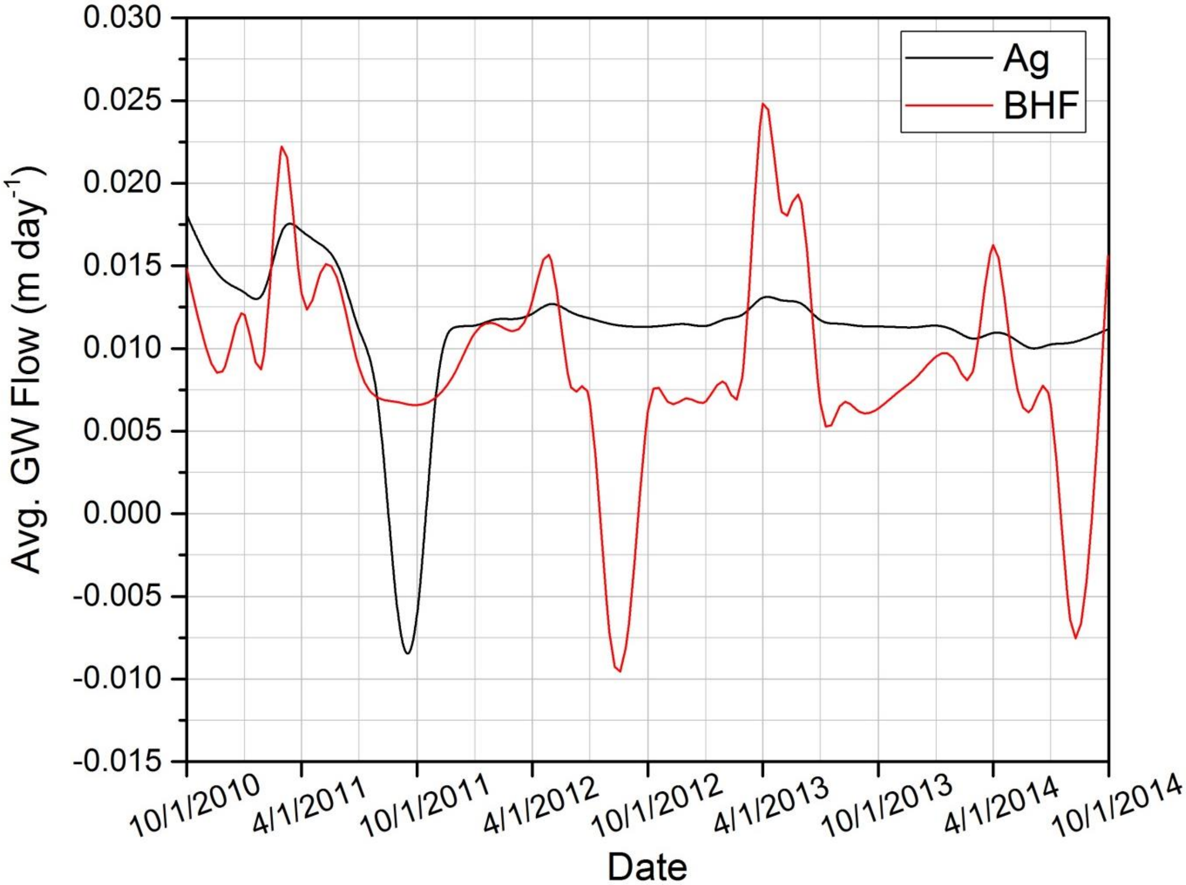

Figure 4. Average groundwater (Gw) flow at agricultural (Ag) and bottomland hardwood forest (BHF) sites during 2011, 2012, 2013, and 2014 water years, Hinkson Creek Watershed, central Missouri, USA, using the Devlin (2003) method. Negative groundwater flow indicates baseflow reversal (i.e. flow from the creek *into* the floodplain).

Intra-Site Comparisons of SGW Temperature

Average observed SGW temperatures in piezometers grouped according to distance from the stream (10, 50, and 90 m for rows 1, 2, and 3, respectively) at the Ag site were 11.7, 10.9, and 10.8 °C, for rows 1, 2, and 3, respectively (Table 2). Results from Wilcoxon Signed-Rank tests showed average row temperature values to be significantly different ($p<0.001$) from one another. Average SGW temperatures of the three Ag rows were within 1.0 °C. However, row 1 exhibited a greater temperature range

(8.5 °C) than row 2 (7.0 °C), and row 3 (5.3 °C). The contrast is observable in Figure 5 by the greater peak-to-peak amplitude (range) of row 1 relative to rows 2 and 3. Average SGW temperatures of the three piezometer monitoring rows at the BHF site were 10.9, 11.6, and 11.1 °C, for rows 1, 2, and 3, respectively (Table 2). Despite values within 1.0 °C, average SGW temperatures of the three rows at the BHF site were all significantly different ($p<0.001$) from one another. Temperature range at the BHF (5.1, 4.6, and 4.5 °C, for rows 1, 2, and 3, respectively) showed a decreasing trend with increasing distance from Hinkson Creek similar to that at the Ag site. However, mean temperature did not show a decreasing trend with increasing distance from the stream, with row 2 exhibiting the greatest average SGW temperature during the study period. While the rows at the Ag site showed temperature ranges contrasting by as much as 3.2 °C, SGW temperature trends of rows at the BHF site were more similar, with ranges differing by less than 1.0 °C (Figure 5).

Table 2. Descriptive statistics of shallow groundwater temperature (°C) at piezometers grouped according to distance from stream (10, 50, and 90 m for rows 1, 2, and 3, respectively) at agricultural (Ag) and bottomland hardwood forest (BHF) sites during 2011, 2012, 2013, and 2014 water years, Hinkson Creek Watershed, central Missouri, USA.

Site	Row	Avg.	Std. Dev.	Max.	Min.	Range
Ag	Row 1	11.7	1.7	16.8	8.3	8.5
	Row 2	10.9	1.4	14.5	7.5	7.0
	Row 3	10.8	1.2	13.5	8.3	5.3
BHF	Row 1	10.9	1.2	13.9	8.8	5.1
	Row 2	11.6	1.0	14.2	9.6	4.6
	Row 3	11.1	1.0	13.7	9.1	4.5

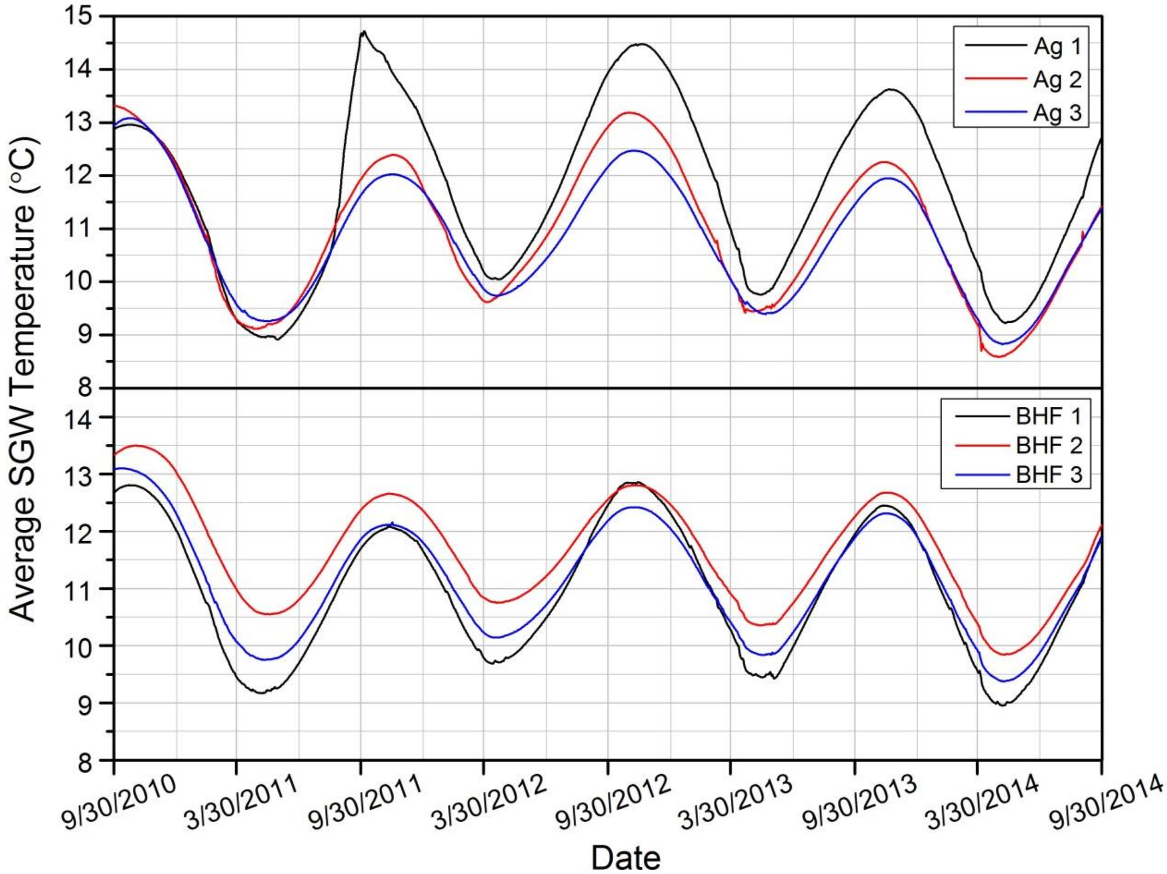

Figure 5. Average shallow groundwater (SGW) temperatures of three piezometer rows at agricultural (Ag) and bottomland hardwood forest (BHF) sites during 2011, 2012, 2013, and 2014 water years, Hinkson Creek Watershed, central Missouri, USA.

Ag and BHF SGW Temperature

Average, standard deviation, maximum, and minimum temperatures of SGW during the four water years (2011, 2012, 2013, and 2014) at the Ag site were 11.1, 1.5, 16.8, and 7.5 °C, respectively (Table 3). Average, standard deviation, maximum, and minimum temperatures of SGW during the four water years at the BHF site were 11.2, 1.1, 14.2, and 8.8 °C, respectively. Median SGW temperature was 11.0 and 11.2 °C for the Ag and BHF site, respectively. Wilcoxon Signed-Rank results indicated SGW temperatures at the two study sites to be significantly different ($p<0.001$) at 30 minute time intervals. Observed data were tested for autocorrelation by reducing average SGW temperature data from 30 minute to daily resolution and retested using Wilcoxon Signed-Rank. Results indicated daily SGW temperatures at the Ag and BHF were also significantly different ($p<0.001$). The total observed range over the study period between maximum and minimum temperatures at the Ag site (9.3 °C) was 72% greater than the BHF (5.4 °C) (Figure 6). The greater peak-to-peak amplitude (range) of Ag site SGW temperature relative to the BHF is shown in Figure 7.

Table 3. Descriptive statistics of shallow groundwater temperature (°C) at agricultural (Ag) and bottomland hardwood forest (BHF) sites during 2011, 2012, 2013, and 2014 water years, Hinkson Creek Watershed, central Missouri, USA.

	WY	Avg.	Std. Dev.	Max.	Min.	Range
Ag	2011	10.9	1.5	16.8	8.3	8.5
	2012	11.4	1.3	16.8	9.3	7.6
	2013	11.4	1.5	14.9	8.6	6.2
	2014	10.8	1.5	14.3	7.5	6.8
	Total	11.1	1.5	16.8	7.5	9.3
BHF	2011	11.3	1.3	14.2	9.0	5.2
	2012	11.3	0.9	13.2	9.3	3.9
	2013	11.3	1.0	13.7	9.2	4.5
	2014	10.9	1.1	13.0	8.8	4.3
	Total	11.2	1.1	14.2	8.8	5.4

WY = water year

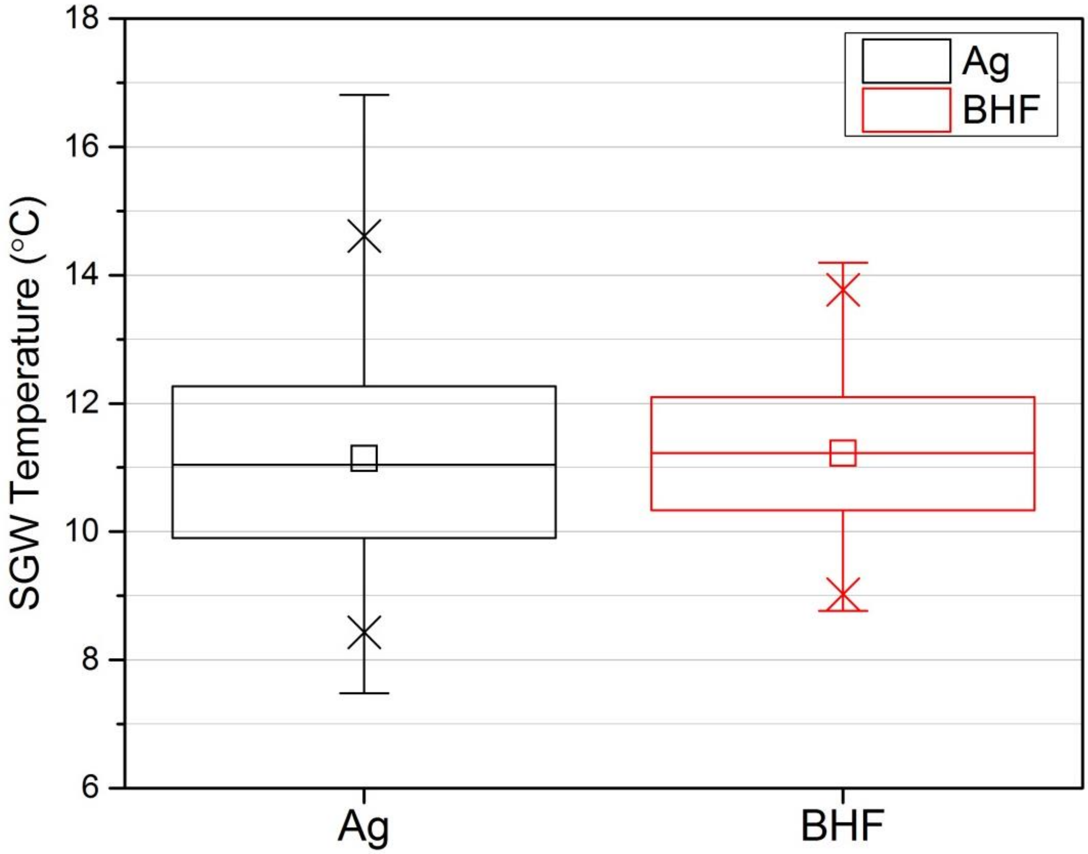

Figure 6. Box and whisker of plot of shallow groundwater (SGW) temperature of agricultural (Ag) and bottomland hardwood forest (BHF) sites during 2011, 2012, 2013, and 2014 water years, Hinkson Creek Watershed, central Missouri, USA. Squares within boxes indicate mean. X's indicate 99th percentile when above, 1st percentile when below. Minus signs indicate maximum when above, minimum when below.

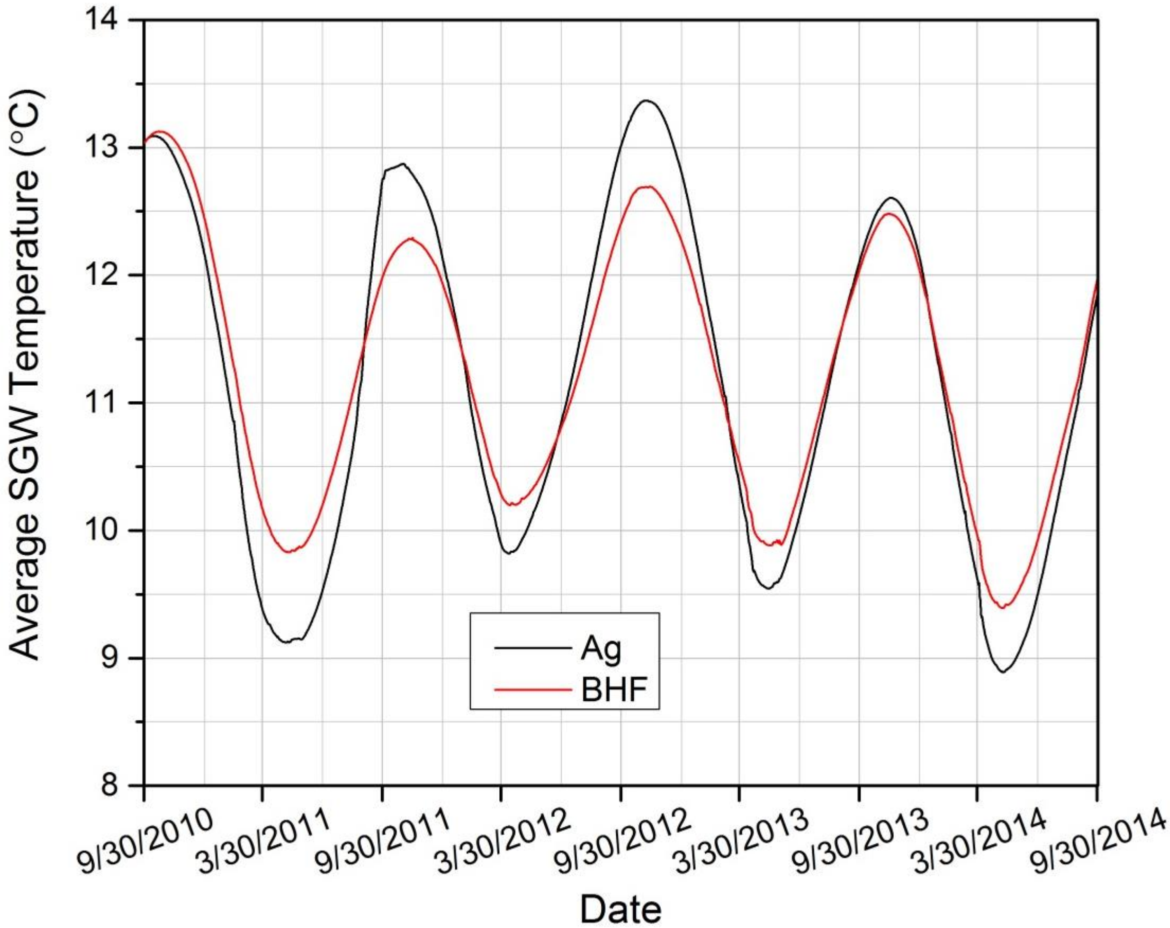

Figure 7. Average shallow groundwater (SGW) temperature at agricultural (Ag) and bottomland hardwood forest (BHF) sites during 2011, 2012, 2013, and 2014 water years, Hinkson Creek Watershed, central Missouri, USA.

Average soil temperature at 30 cm was 8.8 and 9.8 °C at the Ag and BHF sites, respectively (Table 4). 30 cm soil temperature data from each microclimate station and groundwater temperature data were used to calculate subsurface thermal diffusivity as per equations 2 and 3. Average thermal diffusivity during the study was 4.72 x 10^{-7} and 4.43 x 10^{-7} m^2 sec^{-1} at the Ag and BHF sites, respectively (Table 4). As shown in Table 4, the average amplitude of 30 cm soil temperature at study sites showed similar values for each water year, with inconsistent differences of less than 0.5 °C. Average damping depth was 2.11 and 2.18 m at the BHF and Ag sites, respectively. Average observed lag time between groundwater temperature and air temperature at the Ag and BHF sites was 98 and 101 days, respectively, or approximately three months. Although this difference may seem small, the Ag site displayed a lower average lag time, consistent with a higher estimated subsurface thermal diffusivity and lower damping depth.

Table 4. Average thermal diffusivity (m^2 sec^{-1}) and 30 cm soil temperature amplitude (°C) at agricultural (Ag) and bottomland hardwood forest (BHF) sites during 2012, 2013, and 2014 water years, Hinkson Creek Watershed, central Missouri, USA.

	Site	D (m^2 sec^{-1})	30 cm Ts Amp. (°C)
2012 WY	BHF	4.64E-07	11.1
	Ag	5.62E-07	10.7
2013 WY	BHF	5.06E-07	10.6
	Ag	5.50E-07	10.6
2014 WY	BHF	5.16E-07	10.5
	Ag	5.26E-07	10.9
Total	BHF	4.43E-07	12.4
	Ag	4.72E-07	12.9

WY = water year
D = thermal diffusivity
Ts = soil temperature
Amp. = amplitude

Discussion

Figure 7 illustrates the greater (72%) response of the shallow groundwater (SGW) temperature at the Ag site, relative to the BHF, to seasonal soil temperature change. This responsiveness is most clearly evident at the end of each water year in the steep warming trend of the groundwater at the Ag site relative to the BHF. Considering the similarity of the two sites geographically, and in terms of physical soil characteristics (i.e. bulk density and porosity) below 50 cm (Hubbart et al., 2011), it is reasonable to conclude that the ultimate determinant of the observed differences is land use type (i.e. vegetative cover). However, given the number of influential factors, a mechanistic explanation of groundwater temperature differences requires investigating the potential contribution of several different variables.

Groundwater flow differences can impact groundwater temperature regimes (Bundschuh, 1993; Taniguchi et al., 1999; Bense and Kooi, 2004; Reiter, 2005). However, according to Bundschuh (1993), who modeled shallow groundwater temperature seasonality under a variety of conditions, there is a minimum horizontal flow magnitude necessary for thermal advection to become the dominant force contributing to groundwater temperature, approximately 100 m yr^{-1}, or 0.3 m day^{-1}, depending on aquifer properties. The highest horizontal groundwater flow value estimated in the current work was 0.024 m day^{-1}, an order of magnitude less than the threshold value, and average groundwater flow during the study was much less (0.009 m day^{-1} and 0.011 m day^{-1} at the BHF and Ag sites, respectively). Therefore, it is unlikely that the subsurface thermal regimes of the study sites are predominately influenced by horizontal convective heat transport (Anderson, 2005).

Results indicated small inconsistent differences between soil temperature amplitude at the study sites. If the primary mechanistic cause of the larger observed Ag site SGW temperature range was increased radiative heating of the ground surface resulting from forest removal, then considering similar average SGW temperatures (11.1 and 11.2 °C at the Ag and BHF, respectively), heat transport theory would predict a larger temperature amplitude at the Ag surface, relative to the BHF (Hillel, 1998). However, such an expectation is not consistent with results. As shown in Table 4, the average 30 cm soil temperature amplitude was actually larger at the BHF site in water year 2012 and the same at the two sites during 2013, both years in which the average temperature range of Ag SGW was greater than that of the BHF (Table 3). Therefore, it is unlikely that observed SGW temperature differences can be attributed to increased radiative heating of the Ag site ground surface. This conclusion contrasts with those of previous studies (Hewlett and Fortson, 1982; Henriksen and Kirkhusmo, 2000; Beltrami and Kellman, 2003; Ferguson and Beltrami, 2006; Eggleston and McCoy, 2014; Guenther et al., 2014). However, the majority of such studies investigated the impact of recent forest clearing on groundwater temperatures. In the current work, forest removal occurred over a century ago; and although the field was cropped continuously for several decades, it was left fallow in the mid 1980's. While the grassland vegetative cover currently present in the Ag field presumably does not attenuate incoming solar radiation to the extent of the forested site canopy, neither is the Ag ground surface bare. Thus, observations indicate the potential for grassland species to reduce incident solar radiation, thereby mitigating the impacts of woody vegetation removal on the energy balance of the shallow subsurface.

Results showed average water table depth during the study period was 2.66 and 3.13 m at the Ag and BHF sites, respectively, an 18% difference. Contrasting water table position relative to the ground surface is likely the result of woody vegetation at the BHF site (e.g. increased water use, decreased recharge). Despite greater average infiltration capacity at the BHF, relative to the Ag site (Hubbart, 2011), a recent study conducted at the two sites (Zell et al., 2015) showed that groundwater recharge was 400% greater at the Ag site, potentially due to greater plant water use at the BHF. Specifically, results of coupled vadose-phreatic zone modeling showed total evapotranspiration during 2011 and 2012 water years at the Ag and BHF sites to be 1603.3 and 1869.0 mm, respectively, a difference of more than 16% (Zell et al., 2015). Increased recharge can result in a subsurface temperature regime more closely resembling that of the surface due to vertical convective heat transport (Anderson 2005, Keery et al., 2007). Thus, it is possible that the greater observed Ag SGW temperature range is primarily the result of contrasting rates of plant water use and groundwater recharge at the study sites, a hypothesis further supported by comparing results from individual water years. For example, as shown in Figure 7, differences in SGW temperature range varied from year to year, with greater percent differences in 2011 and 2012 (63 and 92%, respectively), and smaller percent differences in 2013 and 2014 (37 and 59%, respectively). The temporal pattern of SGW temperature range differences is inconsistent with that of percent difference site groundwater flow magnitude, average air temperature, and air temperature range. Rather, the variable that corresponds most closely with the pattern of SGW temperature range differences is precipitation, illustrating the importance of surface water dynamics to floodplain groundwater temperature regimes. During drier years (i.e. 2011 and 2012), higher rates of plant water use at the BHF site may have contributed to a drier, less thermally conductive subsurface, relative to the Ag site, thereby resulting in larger differences in SGW temperature. Conversely, in wetter years (i.e. 2013 and 2014) smaller differences in subsurface water content between the two sites could have produced more similar SGW temperatures. Considering the lack of evidence supporting influence of surface heating and horizontal groundwater flow, study results suggest that observed SGW temperature differences are due to contrasting groundwater recharge rates, a condition presumably resulting from greater plant water use at the BHF, relative to the Ag site.

Despite low average horizontal groundwater flow magnitude at the study sites, flow results could help explain patterns of seasonal temperature differences between piezometer rows. SGW at Ag site piezometer rows showed temperature ranges contrasting by as much as 3.2 °C, with temperature range decreasing with increasing distance from the creek (Figure 5). The three BHF rows were more similar, with SGW temperature ranges differing by less than 1.0 °C. Seasonal periods of higher horizontal groundwater flow velocity at the BHF site may result in greater SGW temperature homogeneity (Figure 4), while consistently lower flow velocity at the Ag site could result in greater spatial heterogeneity of SGW temperature. Additionally, groundwater flow results indicated a reversal of flow direction at the Ag site during the fall of 2011, the period of time corresponding with the rapid warming of groundwater at row 1 (10 m from stream) (Figure 5). Results suggest that during this period (September and October, 2011) baseflow was reversed at the Ag site, and Hinkson Creek became, for a time, a losing stream within that reach. Average SGW temperature along the row 1 transect during the two month period was 13.9 °C, compared to 11.9 and 11.6 °C for rows 2 and 3, respectively. Average stream temperature in Hinkson Creek during this period was 16.5 °C. Thus, during the period corresponding to baseflow reversal, SGW temperature at Ag row 1 showed a closer association with stream temperature than rows farther from the stream. Similarly, flow results show a reversal of baseflow direction at the BHF site in August and September, 2012, and August, 2014. Importantly, these two periods correspond with warmer average BHF groundwater temperatures (Figure 7), suggesting that baseflow reversal at the BHF during dry months could help explain warmer average BHF groundwater temperatures during the summer of 2014, relative to the Ag site, and differences in groundwater temperature between piezometer rows at the BHF site (i.e. row 1 average temperature exceeding that of rows 2 and 3 only during summer of 2012) (Figure 5). Additionally, as shown in Table 1, mean streamwater temperature (14.2 °C) was higher than mean SGW temperature (11.1 and 11.2 °C at the Ag and BHF, respectively). Thus, regardless of baseflow, Hinkson Creek functions as a heat source via conduction during portions of the year, which could further explain the higher mean temperature and temperature range of groundwater in piezometers located nearer the creek.

In addition to distance from the creek, infiltration/recharge rate could help explain observed SGW temperature differences between rows. For example, row 2 at the BHF site displayed the highest mean groundwater temperature (Table 2). Specifically, piezometer 5, located in the middle of the site, displayed highest mean, maximum, and minimum groundwater temperature during the study. Hubbart (2011) showed a higher infiltration rate around piezometer 5, relative to the rest of the site, likely related to higher leaf area index and tree diameter. Thus, greater vertical convective heat transport in the vicinity of BHF piezometer 5 could be responsible for the greater response of groundwater temperature in that area to seasonally changing surface temperature.

While results suggest the influence of multiple mechanistic processes (i.e. conduction, recharge, and infiltration) on SGW temperatures at the Ag and BHF sites, more work is needed to determine the extent to which each process is contributing to observed differences. For example, considering its importance to subsurface thermal regime, future work should include measured ground surface temperature, which could provide further information on the impacts of contrasting surface vegetation on subsurface temperatures. Likewise, coupling surface to subsurface heat flux and hydrologic modeling, and incorporating measured microclimate and hydrologic data (e.g. incident ground surface shortwave radiation, ground surface temperature, volumetric soil water content), could help quantify the impact of vertical water flow on the temperature regime of floodplain shallow groundwater.

Conclusions

Considering the lack of information in the primary literature and the importance of groundwater temperature to the biogeochemical health of aquatic ecosystems, there is a need for improved understanding of rural land use impacts on SGW temperature regime. This work was conducted to compare the temperature of shallow groundwater (SGW) at agricultural and bottomland hardwood forest floodplain sites. An 80 x 80 m grid of nine piezometers was installed 10 m from Hinkson Creek at each site (total n=18) to monitor temperature and hydraulic head during the 2011, 2012, 2013, and 2014 water years. The study is one of the first to utilize long-term, continuous, automated, *in situ* monitoring to investigate rural land use impacts on shallow groundwater temperatures. Average water table depth was 18% greater at the Ag site, relative to the BHF. Wilcoxon Signed-Rank results indicated SGW temperature at the two sites to be significantly different ($p<0.001$) at both 30 minute and daily resolutions. Although average SGW temperature at the two sites was similar, the Ag site showed a temperature range 72% greater than the BHF. Considering similar surface soil temperature amplitudes and low average horizontal groundwater flow values at both sites, results suggest contrasting groundwater recharge rate, possibly resulting from greater plant water usage at the BHF, is the likely mechanistic cause for the observed SGW temperature differences. Moreover, patterns of intra-site groundwater temperature differences at both study sites illustrate the influence of stream-aquifer thermal conduction and occasional baseflow reversals.

The 72% greater SGW temperature range at the Ag site indicates a greater responsiveness of the groundwater to seasonally changing climate variables, relative to the BHF. While mean SGW temperatures at the two sites were similar, the greater temperature range of Ag site SGW suggests the potential for higher baseflow temperatures, especially during the late summer/fall, a condition that could negatively impact temperature sensitive aquatic organisms by altering thermal refugia. Although more work is needed to determine the extent to which each process is contributing to observed differences, results suggest that removal of woody vegetation has altered the hydrology (e.g. groundwater recharge rate) of the Ag site, thereby impacting the temperature regime of shallow groundwater. Results of the study point to the impact of bottomland hardwood forest clearing and agricultural development on SGW temperatures and floodplain processes. Considering the importance of groundwater temperature to the biogeochemical health of aquatic ecosystems, these results hold important implications for natural resource managers regarding the impacts of rural land use on freshwater resources and rural floodplain management.

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CHAPTER IV

A COMPARISON OF FOREST AND AGRICULTURAL SHALLOW GROUNDWATER CHEMICAL STATUS A CENTURY AFTER LAND USE CHANGE

Revised version of earlier publication, in print:

Kellner, E., Hubbart, J.A., Ikem, A. 2015. Comparing Forest and Agricultural Shallow Groundwater Chemical Status a Century after Harvest. *Science of the Total Environment* **529**: 82-90.

Introduction

Background

It is generally accepted that chemical composition is a critical concern for groundwater quality. Groundwater chemistry is a function of numerous interacting factors, including geology, groundwater flow, recharge, residence time, and anthropogenic impacts such as land use. For example, Scanlon et al. (2005) showed that vertical fluxes of water in the subsurface from evapotranspiration and recharge can enrich or dilute, respectively, the chloride and nitrate concentrations of groundwater. Similarly, studies by Soveri (1985), Katz et al. (1998), Adams et al. (2001), Böhlke (2002), and Kebede et al. (2005) reported the potential for recharge to either enrich or dilute groundwater with respect to a given constituent, depending on the chemical composition of atmospheric and vadose zone water relative to groundwater. Stuyfzand (1999) noted the potential for groundwater flow to reduce or increase spatial variations in groundwater chemical composition via the counteracting influences of mixing and dispersion, respectively. Given the recognized connections between groundwater and surface water (Hayashi and Rosenberry, 2002), understanding patterns of environmental and anthropogenic influences on groundwater chemistry is important in order to effectively manage aquatic ecosystems, and maintain the suitability of groundwater for human consumption.

Studies repeatedly and conclusively show that land use change impacts groundwater quality. For example, Trojan et al. (2003) showed that groundwater chemistry was significantly different ($p<0.05$) between land use types (e.g. agricultural, residential, and industrial), concluding that land use was the dominant factor influencing shallow groundwater quality. Squillace et al. (1996) found increased levels of shallow groundwater contamination by a common fuel oxygenate (methyl tert-butyl ether, or MTBE) in urban areas relative to rural areas. Pionke and Urban (1985) reported concentrations of nitrate, chloride, and phosphate in groundwater that were five to seven times higher under cropland, relative to forest land use. Similarly, Jiang et al. (2008) reported increases of pH and pollutant concentrations in groundwater following the conversion of forest to cropland and/or urban areas.

Land use change often includes altered vegetated canopy cover, which can impact the physiochemical processes and characteristics of the subsurface. For example, Addy et al. (1999) noted several vegetation-mediated impacts, including subsurface hydrology, spatial distribution of soil carbon, and land use legacies, all of which impacted groundwater nitrate removal rates. Osborne and Kovacic (1993) reported that forested buffer strips were more effective at reducing nitrate concentrations in shallow groundwater than grass buffers, but that grass buffers more efficiently retained phosphorus. Yevenes and Mannaerts (2011) found that agricultural sub-catchments exported five times more nitrate than forest and range sub-catchments. Balestrini et al. (2011) showed that alteration of subsurface flow patterns due to consumptive water use by woody vegetation (i.e. evapotranspiration) was a primary factor contributing to riparian groundwater nitrate removal rates. In addition to nitrate, land use has been shown to influence the biogeochemistry of various chemical species, including phosphorus (Hedley et al., 1982), carbon (Guo and Gifford, 2002), sulfur (Lilienfein et al., 2000), calcium (Grigal, 2000), silica (Carey and Fulweiler, 2012), and mercury (Lacerda et al., 2004), among others.

Objectives

Many previous groundwater chemistry studies addressed land use impacts on specific chemical constituents (e.g. nitrate) or contaminants (e.g. chlorinated solvents); however, few previous studies were explicitly designed to characterize the chemical composition of groundwater at sites with contrasting land use histories. Given the importance of groundwater resources for sustaining human populations and aquatic ecosystems, and considering the increasing pace of land use/land cover change worldwide (in particular urbanization), studies are critically needed that investigate the effects of rural anthropogenic land uses on the physiochemical quality of groundwater. Thus, the objectives of the current work were to quantitatively describe and compare groundwater chemical composition in an intensive gridded investigation of historic agricultural and forested floodplain sites.

Methods

Site Description

This research was conducted in the Hinkson Creek Watershed (HCW) located in central Missouri, USA. The project was nested within a larger experimental watershed study (approximately 240 km^2) implemented in 2008, with core work including environmental flows, water quality, and climate (e.g. urban heat island) (Figure 1). The HCW contains approximately 60% of the city of Columbia, community of 108,000 (USCB, 2011). Land use in the HCW is approximately 34% forest, 38% agriculture, and 25% urban (Hubbart et al., 2011), making it a representative location for studying the effects of different land use types on groundwater resources. According to a 64 year climate record of Columbia, Missouri, average annual temperature and precipitation within the watershed were 12.5 °C and 991 mm yr^{-1}, respectively (Missouri Climate Center, 2014). Two floodplain reaches of Hinkson Creek with contrasting land use histories were selected for data collection: a remnant bottomland hardwood forest (BHF), and a historic (i.e. currently fallow) agricultural field (Ag). Geographical coordinates of the study sites are as follows: Ag 38.927092, -92.359043; BHF 38.926658, -92.348443. Woody vegetation at the BHF site includes Silver maple (*Acer saccharinum*), Eastern cottonwood (*Populus deltoides*), Boxelder (*Acer negundo*), American elm (*Ulmus americana*), Black walnut (*Juglans nigra*), and Sycamore (*Platanus occidentalis*) (Hubbart et al., 2011). Forest vegetation was removed from the Ag site approximately 100 years ago. Current vegetation at the Ag site is primarily composed of gramminoids such as Johnsongrass (*Sorghum halepense*), and tall fescue (*Festuca arundinacea*), and the forbs goldenrod (*Solidago giganteana*), and Ironweed (*Vernonia altissima*) (Brown, 2013). Both sites contain Haymond silt loam alluvial soils (NRCS, 2015). Soils in the floodplain study area are underlain by Mississippian series limestones (Burlington formation) (Unklesbay, 1952; Miller and Vandike, 1997). Average site elevation is 174.4 and 176.3 m.a.s.l. for the Ag and BHF, respectively. Hubbart et al. (2011) reported statistical similarity ($p>0.05$) of physical soil characteristics (i.e. bulk density and porosity) below 50 cm depth at the two sites. Above 50 cm, statistical differences ($p<0.05$) between study site soil characteristics reflected impacts of surface vegetation on soil hydraulic properties such as infiltration capacity. Thus, while the alluvial soils at the two sites are similar in composition, there are clear land use related lithosphere alterations that suggest the potential for surface impacts on subsurface processes.

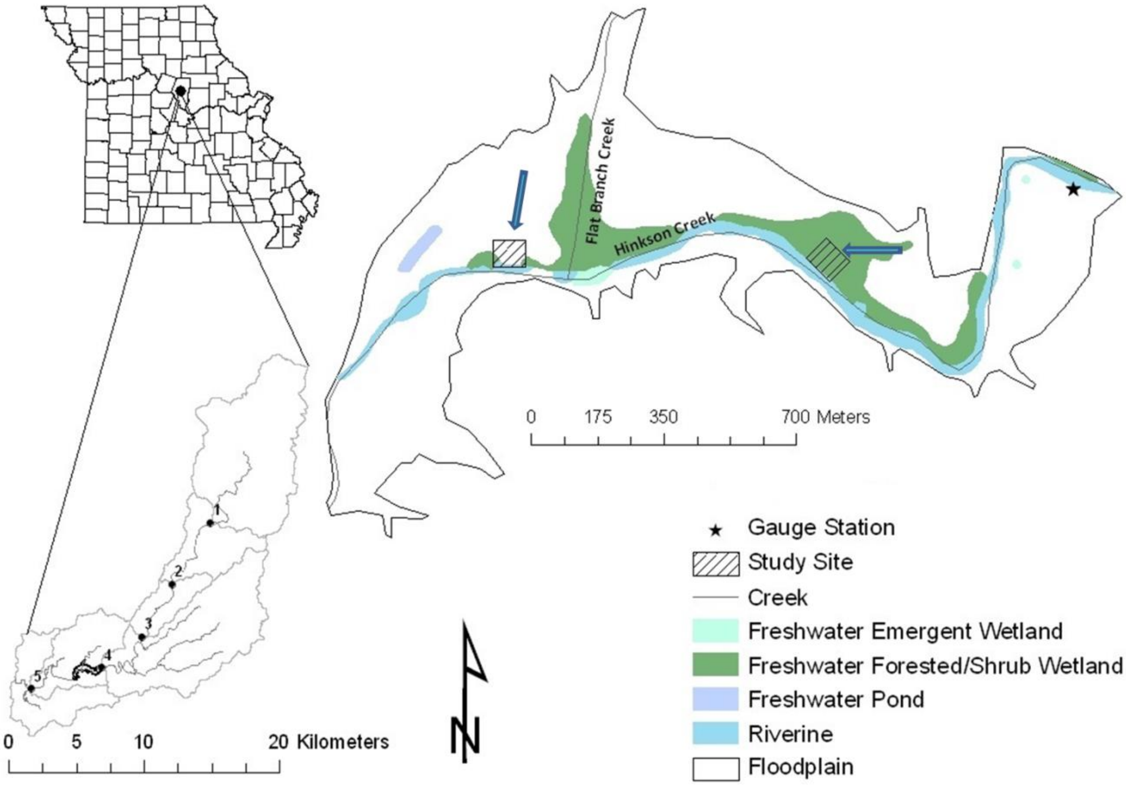

Figure 1. Shallow groundwater study sites in lower Hinkson Creek Watershed floodplain, located in central Missouri, USA. Arrows indicate average monthly groundwater flow direction during study period at each site.

Data Collection

To monitor and characterize shallow groundwater at the study sites, an 80 x 80 m grid of nine equidistant piezometers (i.e. each 40 m apart) was installed at each site (Figure 1). The grid approach provided the opportunity for spatial analysis of shallow groundwater hydrology and chemical composition. A similar design was used by Haycock and Burt (1993), who investigated nitrate concentration in floodplain groundwater in a 24 x 16 m grid of 24 boreholes. In the current work, steel drive-point piezometers 5.49 m long with a 4.0 cm inner diameter and a 0.76 m screened bottom segment (i.e. drive point) were driven into the unconfined alluvial aquifer. Each piezometer was equipped with a Solinst Levelogger Gold (Solinst Canada, Ltd.) with integrated pressure transducer to measure water level (error: $\pm$0.003 m) and temperature sensor (error: $\pm$0.05 °C). Data were logged at 30 minute intervals for the duration of the study period (June 2011 – June 2013). Water table depth and water table elevation were calculated using the results of study site surveys to correct for differences in piezometer elevation. Streamflow and stage data for Hinkson Creek were collected at US Geological Survey gauging station (#06910230), located approximately 2000 m upstream of the sites, and a project-related monitoring site located approximately 8000 m downstream of the study sites. Air temperature and precipitation were measured at Sanborn Field, located on the University of Missouri campus approximately 4 km from the study sites, to obtain climate data representative of the Hinkson Creek Watershed.

Monthly groundwater sampling campaigns were conducted at both sites from June 2011 - June 2013 (n = 24) within the first few days of each month. Previous groundwater chemistry studies utilized similar monthly sampling frequencies (e.g. Banaszuk et al., 2005). Groundwater samples were drawn from the eighteen piezometers using a battery operated peristaltic pump. Monthly grab samples were collected at the same time from Hinkson Creek adjacent (approximately 10 m) to the BHF and Ag sites for comparison with groundwater data. Samples were analyzed for pH, electrical conductivity (EC), total dissolved solids (TDS), total dissolved carbon (TC), dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), total dissolved nitrogen (TDN), reactive phosphorus (orthophosphate), chloride (Cl^-), nitrate (NO_3^-), nitrite (NO_2^-), ammonia (NH_3^+), ammonium (NH_4^+), gallium (Ga), boron (B), indium (In), uranium (U), bismuth (Bi), lithium (Li), rubidium (Rb), silicon (Si), mercury (Hg), phosphorus (P), sulfur (S), potassium (K), sodium (Na), magnesium (Mg), calcium (Ca), silver (Ag), aluminum (Al), arsenic (As), barium (Ba), beryllium (Be), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe), manganese (Mn), molybdenum (Mo), nickel (Ni), lead (Pb), antimony (Sb), selenium (Se), tin (Sn), strontium (Sr), titanium (Ti), thallium (Tl), vanadium (V), and Zinc (Zn). In total, 49 separate physiochemical metrics were monitored and analyzed.

Groundwater and surface water samples were stored in coolers during transport from the field to the laboratory. Upon arrival, samples were placed in a refrigeration unit and stored at a temperature of approximately 4 °C until analysis. The pH, EC and TDS of water samples were measured within 24 hours of collection using a Thermo Orion pH/ORP/conductivity meter (model 555A; Fisher Scientific) and electrodes (pH combination electrode model: Orion 910600; electrical conductivity electrode model: Orion 013005A). Appropriate buffer solutions (pH approximately 4, 7 and 10 at 25 °C) were utilized to calibrate the meter before measurements. For EC measurements, the meter was calibrated using two EC solutions (100 and 1413 $\mu S\ cm^{-1}$ at 25 °C) supplied by Fisher Scientific. Spectrophotometric analyses were performed using a Hach DR 5000 spectrophotometer. The variables Cl^-, NO_3-N, NO_2-N, NH_3-N, and orthophosphate were quantified spectrophotometrically using the mercuric thiocyanate (Hach method 8113), cadmium reduction (Hach method 8039), ferrous sulfate (Hach method 8153), ammonia salicylate (Hach method 8155) and molybdovanadate (Hach method 8114) procedures, respectively (Hach, 2005). NO_3-N, NO_2-N, NH_3-N values were converted to NO_3, NO_2, and NH_3, respectively, using published conversion coefficients (Hach, 2015).

A portion of each sample was passed through a 0.45 micron filter. Samples were placed in 40 mL clean vials and analyzed for total dissolved carbon (TC), dissolved inorganic carbon (DIC), and total dissolved nitrogen (TDN) with the Shimadzu TOC/TN analyzer (Shimadzu Scientific Instruments). Dissolved organic carbon (DOC) was calculated from the difference between TC and DIC. Carbon and nitrogen concentrations were estimated by nondispersive infrared (NDIR) and chemiluminescence detection techniques respectively. Carbon and nitrogen results were validated with known standards during each batch run. The analytical recoveries obtained for carbon and nitrogen standards ranged from 90 to 105 %.

For elemental analysis, ground and surface waters samples were passed through 0.45 micron filters and nitric acid digested using the Ethos EZ microwave digester (Milestone Inc.). The sample digestion process was initiated by the addition of 5 mL nitric acid to 45 mL of sample, and the mixture was subjected to microwave digestion process (Step 1: 160 ^{o}C for 10 min at 1050 W; Step 2: 165 ^{o}C for 10 min at 1050 W). The cooled digests were then placed in acid washed LDPE bottles and aspirated into a Varian Vista-Pro CCD simultaneous inductively coupled plasma–optical emission spectrometer (ICP–OES) (Varian Inc.). The ICP–OES was calibrated with diluted mixed standards containing all the elements of interest. A certified standard reference material (SRM 1643e–trace elements in water) purchased from the National Institute of Standards and Technology (NIST) was used to validate the accuracy of the analytical method. The recoveries of the elements in SRM 1643e ranged from 80 to 108 %. Method blanks were also analyzed for every sample batch and recalibration was conducted every 10 samples.

Data Analysis

Site hydraulic conductivity was determined from the results of a series of falling head slug tests, according to Hvorslev's method (Fetter, 2001). Slug test results indicated average hydraulic conductivity to be 1.0 and 3.5 m day^{-1} at the Ag and BHF sites, respectively. Values for site hydraulic conductivity, hydraulic head measured at each piezometer, and stream stage were used to calculate flow as per Devlin (2003).

Considering hydroclimate data are often non-normally distributed (McCuen, 2003), the Mann-Whitney U test was used for statistical comparison of groundwater chemical data. The Mann-Whitney U test is an appropriate comparison test for samples that violate the parametric assumption of normality (Mann and Whitney, 1947). Site level statistical comparisons were performed on groundwater chemical data averaged by land use type.

Results and Discussion

Climate During Study

Hydroclimatic conditions in the Hinkson Creek Watershed during the period of study (June 2011 – June 2013) were predictably variable. Average, maximum, and minimum air temperatures were 14.8, 33.8, and -8.3 °C, respectively (Table 1). Total precipitation for the two years was 1791 mm, below average considering the HCW 64 year average annual precipitation of 991 mm yr^{-1}. In particular, only 659 mm of precipitation was received in 2012. The period June 2012 – August 2012, was characterized by extreme (D3) to exceptional (D4) drought (USDM Drought Severity Classification; Svoboda et al., 2002), and was ranked one of the hottest and driest on record for the region (Kutta and Hubbart, 2014). Average flow in Hinkson Creek was 1.55 $m^3 s^{-1}$. Maximum observed precipitation (66 mm day^{-1}) and streamflow (278.76 $m^3 s^{-1}$) both occurred during a storm event on May 31, 2013. Average water table depth was 2.72 and 3.17 m at the Ag and BHF sites, respectively. Differences in water table depth likely include the effect of greater consumptive water use by woody vegetation at the BHF, relative to the Ag site. Average monthly groundwater flow velocity was 0.012 and 0.010 m day^{-1} at the Ag and BHF sites, respectively (Figure 2). Such low average groundwater flow values are reasonable considering the low hydraulic gradient at the topographically flat floodplain sites. Average groundwater flow direction relative to Hinkson Creek was 72° and 40° at the Ag and BHF sites, respectively. Average groundwater temperature from June 2011 to June 2013 was 11.2 and 11.3 °C at the Ag and BHF sites, respectively. Similar average groundwater temperature at the study sites is important to the current work, since temperature alterations can influence carbonate precipitation, salt solubility, dissolution of silicate minerals, and the mobilization of organic compounds (Hahnlein et al., 2013).

Table 1. Descriptive statistics of climate variables and shallow groundwater of monitoring sites during study period (June 2011 - June 2013), Hinkson Creek Watershed, central Missouri, USA.

Climate Variable	Avg.	Std. Dev.	Max.	Min.
Ta (°C)	14.8	10.4	33.8	-8.3
PPT (mm)	1791.2	7.2	66.0	0.0
Streamflow (m^3 s^{-1})	1.6	9.6	278.8	0.0
Ag SGW Temp. (°C)	11.3	1.5	16.8	8.3
BHF SGW Temp. (°C)	11.2	1.0	13.7	9.0
Ag Water Table Depth (m)	2.7	0.5	5.4	-2.4
BHF Water Table Depth (m)	3.2	0.5	5.4	-1.9

Ta=air temperature

PPT=precipitation

SGW=shallow groundwater

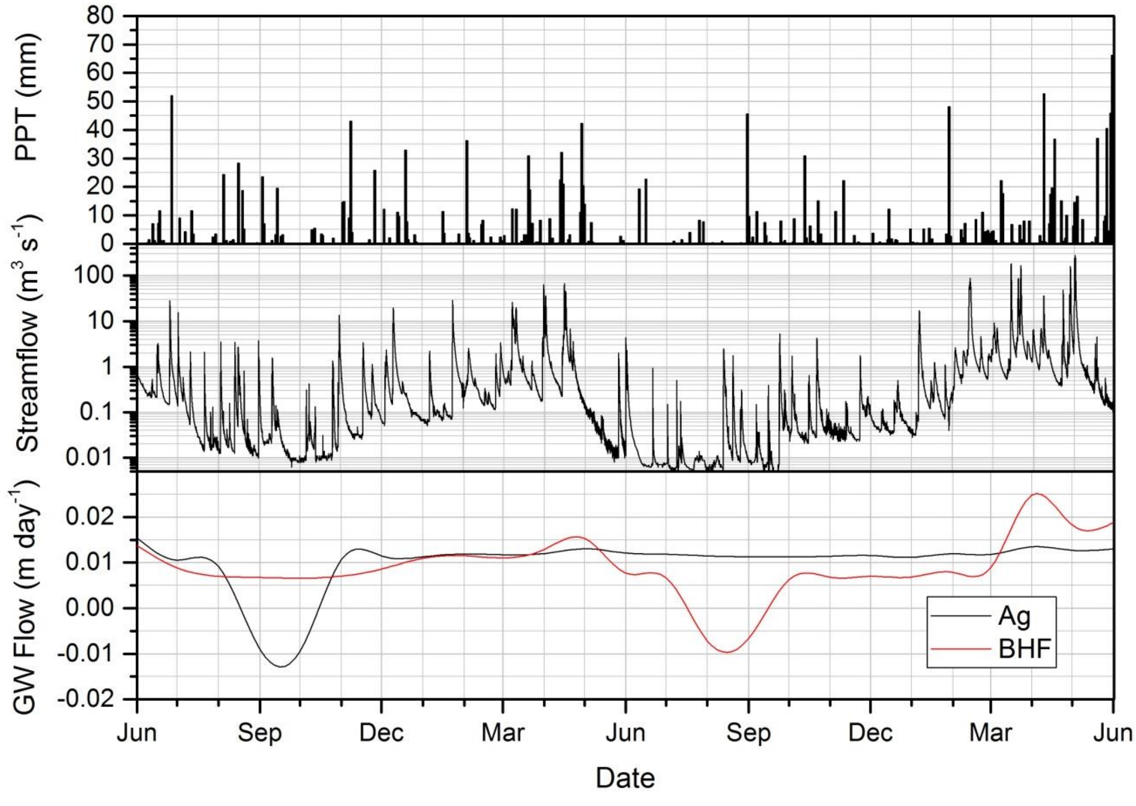

Figure 2. Precipitation (PPT), Hinkson Creek streamflow, and average monthly groundwater (GW) flow of floodplain monitoring sites during study period (June 2011-June 2013), Hinkson Creek Watershed, central Missouri, USA. Negative groundwater flow values indicate baseflow reversal (i.e. flow *into* study site from creek).

Groundwater Chemical Composition

Mann-Whitney results indicated statistically significant differences ($p<0.05$) between the Ag and BHF sites for 32 out of 49 physiochemical metrics (Table 2). For example, pH of Ag site groundwater (average=pH 7.08) was significantly ($p<0.001$) higher than BHF groundwater pH (average=pH 6.72). The electrical conductivity of groundwater at the BHF site (average=800 $\mu S\ cm^{-1}$) was significantly ($p<0.001$) higher than that of Ag groundwater (average=523 $\mu S\ cm^{-1}$). Total dissolved solids (TDS) concentration in BHF groundwater (average=400 $mg\ L^{-1}$) was significantly ($p<0.001$) higher than Ag site groundwater (average=261 $mg\ L^{-1}$). BHF groundwater also displayed significantly ($p<0.001$) higher concentrations of total dissolved nitrogen (TDN), and ammonia; whereas Ag groundwater showed significantly higher concentrations of nitrate. Although dissolved organic carbon (DOC) was not significantly different between the two sites ($p=0.1919$), BHF groundwater exhibited significantly ($p<0.001$) higher concentrations of dissolved inorganic carbon (DIC) and total dissolved carbon (TC). Generally, groundwater samples from the BHF site were characterized by significantly ($p<0.001$) higher concentrations of nutrients (e.g. S, K, Ca, Mg, Na), while samples from the Ag site showed significantly ($p<0.01$) higher concentrations of trace elements (e.g. As, Cd, Co, Cu, Ni, Mo).

Table 2. Results of statistical comparison (Mann-Whitney U Test) of average agricultural (Ag) and bottomland hardwood forest (BHF) site physiochemical parameters and corresponding average concentrations (mg L^{-1}) during study period (June 2011 - June 2013), Hinkson Creek Watershed, central Missouri, USA.

	Parameter	Ag Average	BHF Average
$p<0.05$	pH	7.08	6.72
	Conductivity (uS cm^{-1})	523	800
	Total Dissolved Solids	261	400
	Total Dissolved Carbon	73.91	100.46
	Dissolved Inorganic Carbon	66.61	93.61
	Total Dissolved Nitrogen	0.58	1.04
	Nitrate	3.0	2.5
	NH3	0.30	0.75
	Chloride	15.6	13.0
	Al	0.0073	0.0048
	As	0.0047	0.0036
	B	0.0717	0.0375
	Bi	0.002	0.001
	Ba	0.2041	0.2414
	Ca	73.9149	120.1247
	Cd	0.0002	0.0001
	Co	0.0009	0.0006
	Cu	0.0003	0.0002
	Fe	0.0549	0.1990
	Hg	0.0006	0.0007
	K	1.3296	2.5156
	Li	0.0032	0.0021
	Mg	6.6820	9.4946
	Mo	0.0010	0.0007
	Na	11.6965	14.5494
	Ni	0.0017	0.0008
	Rb	0.002	0.001
	S	5.449	15.862
	Sb	0.002	0.002
	Ti	0.0003	0.0002
	U	0.1038	0.1332

*Unless otherwise noted, parameter values are reported in mg L^{-1}.

Table 2. Continued.

	Parameter	Ag Average	BHF Average
$p>0.05$	Dissolved Organic Carbon	7.37	6.96
	Ag †	0.0003	0.0002
	Be	1.87E-05	1.67E-05
	Cr	0.0004	0.0004
	Ga	0.0010	0.0009
	In	0.0016	0.0014
	Mn	2.0183	1.9032
	Nitrite †	0.04	0.06
	Mean Orthophosphate †	0.3	0.3
	P	0.017	0.009
	Pb	0.0011	0.0012
	Se	0.004	0.004
	Si	5.8752	6.5350
	Sn	0.006	0.003
	Sr	0.2243	0.2478
	Tl	0.004	0.004
	V	0.0008	0.0010
	Zn	9.8144	9.5632

*Unless otherwise noted, parameter values are reported in mg L^{-1}.
† Denotes constituent for which, although significant differences were found, average concentrations for both sites fell below method accuracy limits.

BHF groundwater samples had higher concentrations of TC and DIC relative to the Ag site. Considering the primary importance of the carbonate system to the buffering capacity of natural waters (Faust and Aly, 1981; Deutsch, 1997), results suggest a greater potential for BHF groundwater to buffer changes in pH. Although the BHF had higher concentrations of both TC and DIC, DOC was not significantly different between the two sites. This distinction could be explained by previously noted trends of decreasing organic carbon with increasing profile depth, due to processes such as sorption to soil material. For example, Blazejewski et al. (2009) noted that most subsurface organic carbon lenses occurred within 1 m of the soil surface and that organic carbon masses below 2 m were scarce. Jardin et al. (1989) identified increasing sorption of dissolved organic carbon to soil material with increasing profile depth. Therefore, although the BHF site likely contains greater stocks of aboveground organic carbon in woody biomass, it is reasonable to infer that given average water table depth (2.7 and 3.2 m at the Ag and BHF sites, respectively), differences in groundwater dissolved carbon concentrations would be predominately evident in the inorganic fraction.

Perhaps unsurprisingly, results indicate higher groundwater concentrations of nitrogen species except nitrate at the BHF relative to the Ag site. Average groundwater nitrate concentrations were 3.0 and 2.5 mg L^{-1} at the Ag and BHF sites, respectively. These results illustrate the impact of land use type on the nitrogen biogeochemical cycle, and support extensive previous research (e.g. Osborne and Kovacic, 1993; Balestrini et al., 2011; Yevenes and Mannaerts, 2011) that indicated decreased subsurface nitrate concentrations in forest land use sites.

Higher groundwater concentrations of nutrients (e.g. S, K, Ca, Mg, Na) (Figure 3) at the BHF, relative to the Ag site, are presumably attributable to processes of nutrient recycling and accumulation by forest vegetation. For example, in their seminal biogeochemical study at Hubbard Brook, Likens et al. (1977) showed that of the total annual calcium uptake by forest vegetation, 87% was eventually returned to the forest floor, and 13% was accumulated in standing vegetation; noting that the undisturbed forest efficiently cycled and retained calcium. Numerous studies have detailed nutrient recycling by forest ecosystems via throughfall, stemflow, and litterfall (Carlisle et al., 1966; Parker, 1983; Tobón et al., 2004), and accumulation of nutrients in forest ecosystems over time (Switzer and Nelson, 1972; Cole and Rapp, 1981; Yavitt and

Fahey, 1986). Moreover, forest canopies are understood to intercept atmospheric particulates, thereby increasing the net input of elements such as nitrogen and sulfur (Likens et al., 1977), both of which exhibited significantly ($p<0.05$) higher concentrations in BHF groundwater relative to Ag in the current work. Thus, it is plausible that the presence of woody vegetation has enriched the nutrient budget of the BHF, as compared to the Ag site, resulting in higher concentrations of nutrients in shallow groundwater (Carlisle et al., 1966; Switzer and Nelson, 1972; Cole and Rapp, 1981; Parker, 1983; Tobón et al., 2004). Conversely, long-term agricultural activity (e.g. crop removal) can result in soil nutrient depletions, if nutrient budgets are not carefully managed (Grant et al., 2002). Thus, historic (>100 years) cultivation and cropping of the Ag site could further explain observed differences in groundwater nutrient concentrations. Another possible explanation for observed differences in groundwater nutrient concentrations is contrasting rates of groundwater recharge at the study sites. Zell et al. (2015) utilized a coupled series of surface and subsurface hydrologic models to estimate cumulative groundwater recharge during the 2011 and 2012 water years. Recharge estimates were 400 and 77 mm at the Ag and BHF sites, respectively, a difference of more than 400%. Considering previous research (e.g. Scanlon et al., 2005) demonstrating the potential for increased recharge to dilute groundwater nutrient concentrations, it is likely that, in addition to recycling and accumulation by forest vegetation, observed nutrient concentration patterns are the result of contrasting site hydrology. Exacting these relationships is beyond the scope of the current research, but supplies impetus for future investigations.

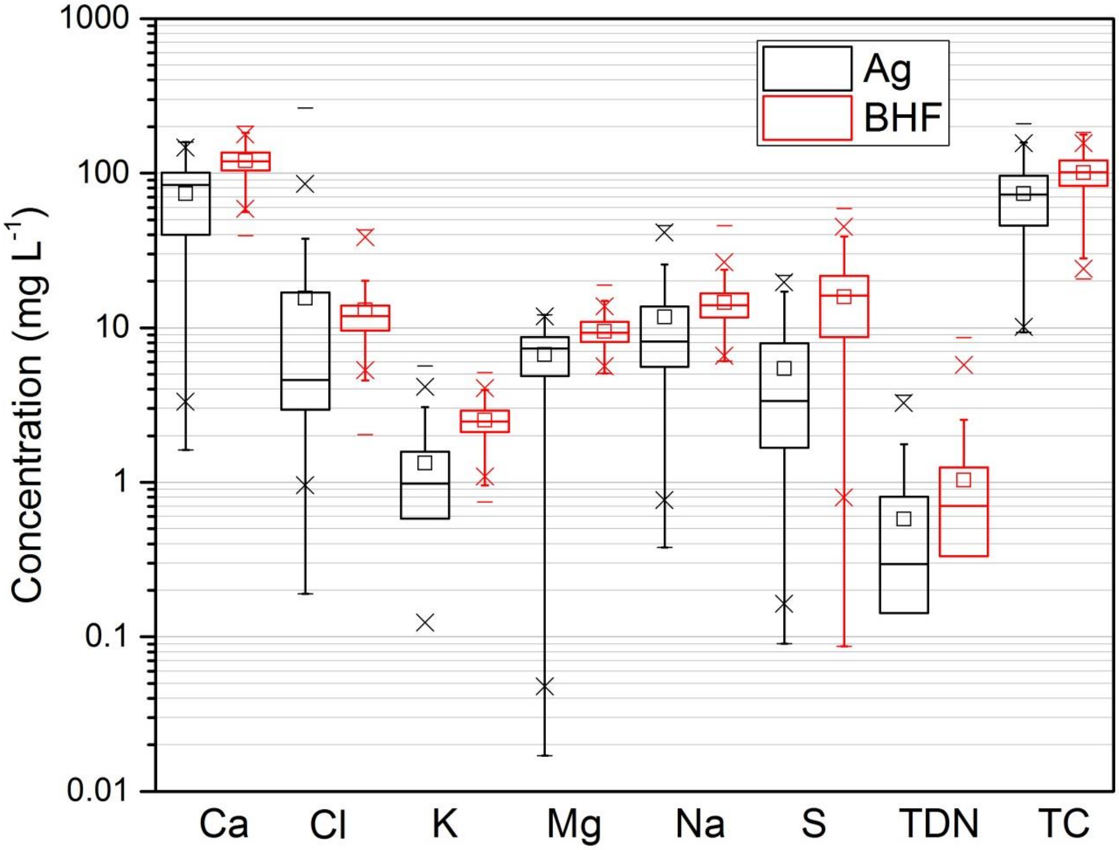

Figure 3. Box and whisker plot of groundwater nutrient concentrations at agricultural (Ag) and bottomland hardwood forest (BHF) sites during study period (June 2011 - June 2013), Hinkson Creek Watershed, central Missouri, USA, where Ca = Calcium, Cl = Chloride, K = Potassium, Mg = Magnesium, Na = Sodium, S = Sulfur, and TDN and TC = total dissolved nitrogen and total carbon, respectively. Squares within boxes indicate mean. X's indicate 99th percentile when above, 1st percentile when below. Minus signs indicate maximum when above, minimum when below.

There are several possible explanations for higher groundwater concentrations of trace elements (Figure 4) at the Ag site, relative to the BHF. For example, observed differences could be the result of plant uptake at the BHF site. Numerous studies have demonstrated the potential for woody vegetation to reduce subsurface concentrations of various trace elements (e.g. Cd, Co, Cu, Mo, Ni) via stabilization, sequestration in plant tissues, and volatilization to the atmosphere (Pulford and Watson, 2003; Pilon-Smits, 2005). Additionally, although average groundwater flow rates at the two sites were similar (0.012 and 0.010 m day^{-1} at the Ag and BHF sites, respectively), average direction of groundwater flow, relative to Hinkson Creek, was 72° and 40° at the Ag and BHF sites, respectively. While baseflow reversals occurred at both sites (September and October, 2011 at the Ag; August and September, 2012 at the BHF), groundwater flow direction became nearly parallel to Hinkson Creek at the BHF during both summer seasons of the study, suggesting greater stream-aquifer exchange (e.g. flow-through) at the BHF site. Thus, it is reasonable to infer that lower groundwater concentrations of trace elements at the BHF site are the result of seasonal flushing of the aquifer by Hinkson Creek. Regardless of the specific mechanism(s), the identification of which are beyond the scope of the current work, differences in trace element concentrations illustrate the potential for land use/land cover change to impact hydrologic and biogeochemical processes, thereby altering the biogeochemical cycles of various elements.

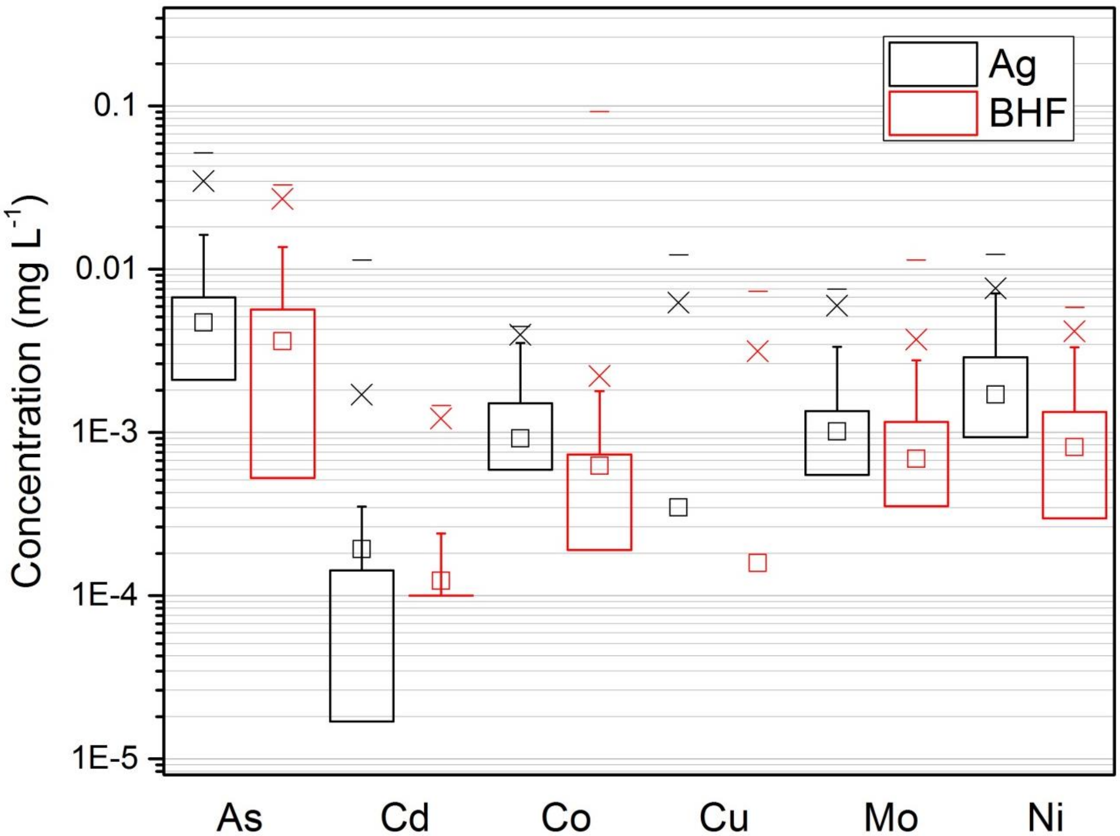

Figure 4. Box and whisker plot of groundwater trace element concentrations at agricultural (Ag) and bottomland hardwood forest (BHF) sites during study period (June 2011 - June 2013), Hinkson Creek Watershed, central Missouri, USA, where As = Arsenic, Cd = Cadmium, Co = Cobalt, Cu = Copper, Mo = Molybdenum, Ni = Nickel. Squares within boxes indicate mean. X's indicate 99th percentile when above, 1st percentile when below. Minus signs indicate maximum when above, minimum when below.

Shallow Groundwater-Surface Water Comparisons

Shallow groundwater chemical concentrations were compared to surface water chemical concentrations analyzed in Hinkson Creek grab samples, collected adjacent to each study site (approximately 10 m). Considering the significant differences ($p<0.05$) observed between study site groundwater nutrient concentrations, two-dimensional subsurface concentration patterns were created to investigate sink/source potential between shallow groundwater and surface water. Results illustrate contrasting patterns at the Ag and BHF sites (Table 3, Figures 5 and 6). In both figures, Hinkson Creek nutrient concentrations are shown at the bottom of the plot, along the x-axis, where y distance equals zero. At the Ag site (Figure 5), average concentrations of K, S, and Mg were higher in the creek (5.36, 30.03, and 9.50 mg L^{-1}, respectively), relative to floodplain shallow groundwater (1.33, 5.45, and 6.68 mg L^{-1}, respectively). At the BHF, while average concentrations of K and S were greater in the creek (5.06 and 30.83 mg L^{-1}, respectively) than the floodplain (2.52 and 15.86 mg L^{-1}, respectively), Mg showed higher concentrations in site groundwater (average=9.49 mg L^{-1}), relative to the creek (average=9.18 mg L^{-1}). Similarly, BHF groundwater had higher levels of Ca (average=120.12 mg L^{-1}) relative to the creek (average=72.72 mg L^{-1}) (Figure 6). As explained above, this observed pattern is likely attributable to recycling and accumulation of nutrients by woody vegetation (Carlisle et al., 1966; Switzer and Nelson, 1972; Cole and Rapp, 1981; Parker, 1983; Tobón et al., 2004) at the BHF site.

Table 3. Average nutrient concentrations (mg L^{-1}) in agricultural (Ag) and bottomland hardwood forest (BHF) shallow groundwater, and adjacent Hinkson Creek grab samples, during study period (June 2011 - June 2013), Hinkson Creek Watershed, central Missouri, USA.

	Ag	Hinkson	BHF	Hinkson
Ca	73.9149	72.8779	120.1247	72.0733
Mg	6.6820	9.5035	9.4946	9.1009
K	1.3296	5.3646	2.5156	5.0563
S	5.4494	30.0345	15.8617	30.8346
Nitrate	3.0	2.6	2.5	2.1
TC	73.91	40.26	100.46	40.75

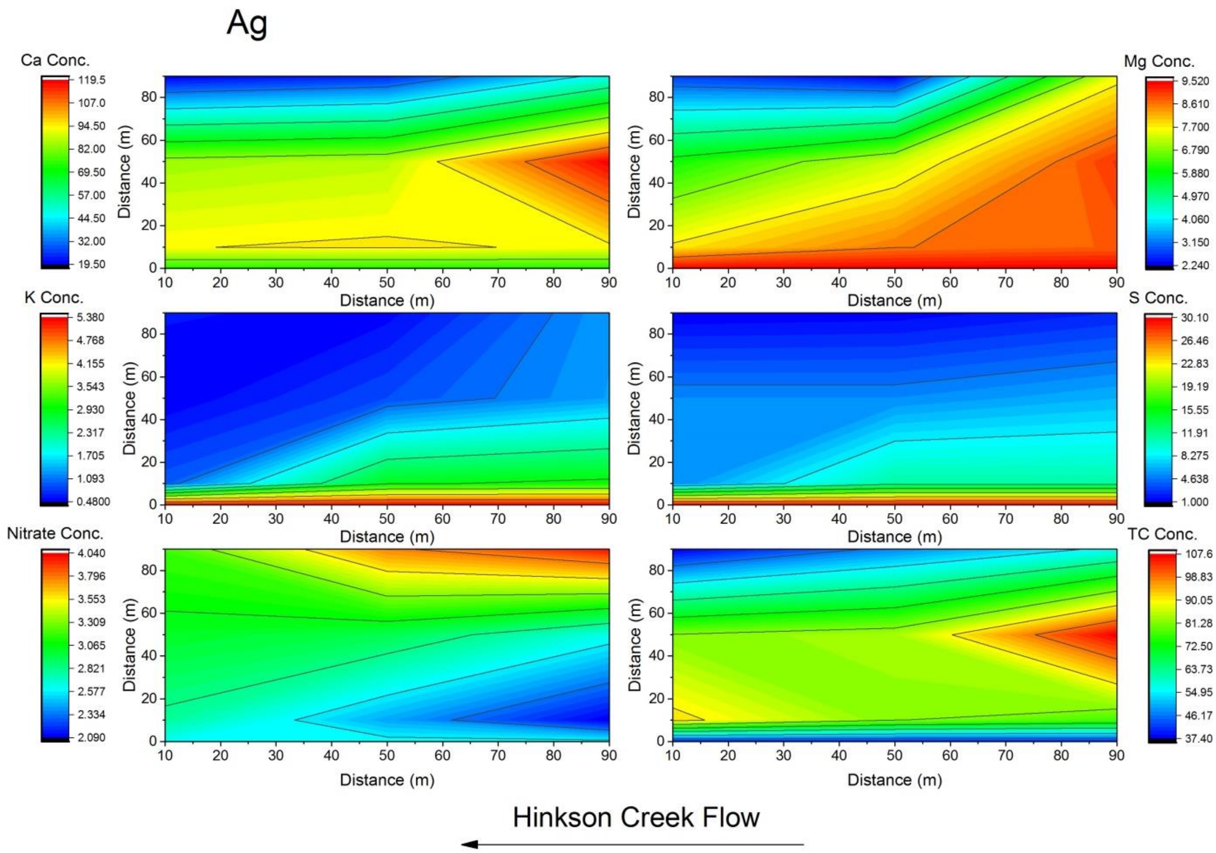

Figure 5. Comparison of average agricultural (Ag) site shallow groundwater and Hinkson Creek nutrient concentrations (mg L^{-1}) during study period (June 2011 - June 2013), Hinkson Creek Watershed, central Missouri, USA. Ca = calcium, K = potassium, Mg = Magnesium, S = sulfur, TC = total dissolved carbon.

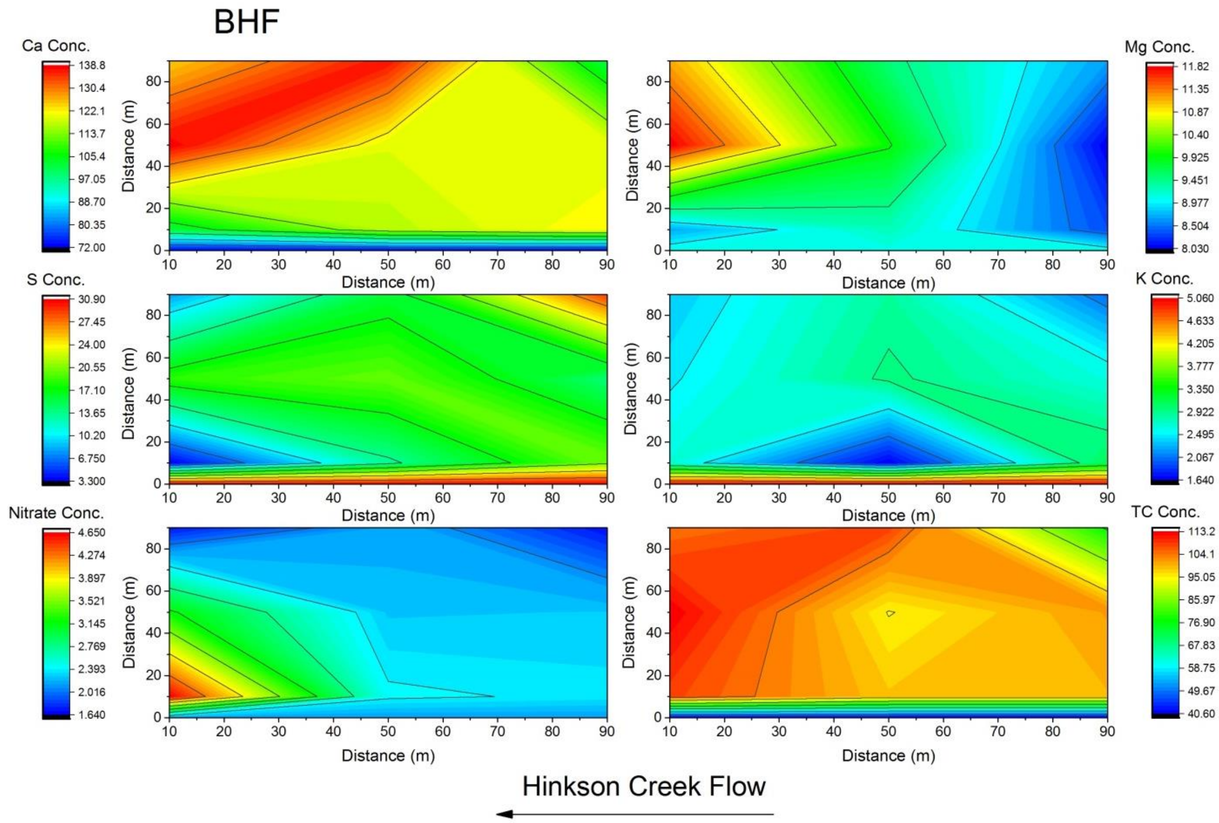

Figure 6. Comparison of average bottomland hardwood forest (BHF) site shallow groundwater and Hinkson Creek nutrient concentrations (mg L^{-1}) during study period (June 2011 - June 2013), Hinkson Creek Watershed, central Missouri, USA. Ca = calcium, S = sulfur, Mg = magnesium, K = potassium, TC = total dissolved carbon.

Sulfur distribution in the Ag and BHF shallow groundwater may, at least in part, be attributable to contrasting groundwater flow patterns given average groundwater flow direction relative to Hinkson Creek was 72° and 40° at the Ag and BHF sites, respectively. Moreover, baseflow reversal occurred at the BHF site in August and September, 2012, during a period of low groundwater flow/drought, and groundwater flow direction was nearly parallel to the creek during summer seasons, suggesting the potential for hyporheic exchange/flow-through with Hinkson Creek. Thus, the flow of creek/hyporheic water into the BHF floodplain, could explain the high average concentration of S in the northeastern corner of the site (average=29.61 mg L^{-1}), comparable to S average concentration in Hinkson Creek (30.83 mg L^{-1}) (Figure 6). In contrast, the direction of groundwater flow at the Ag site was generally more orthogonal relative to the creek. However, in September and October, 2011, average groundwater flow at the Ag site was -78° and -89°, respectively, indicating flow into the floodplain from the creek. The reversal of groundwater flow direction was the likely the result of a period of high streamflow in the creek. Thus, episodic flow from the creek into the Ag floodplain could help further explain the patterns of K, S, and Mg groundwater concentration visible in Figure 5. Contrasting patterns of total dissolved carbon distribution at the two sites illustrate the effect of woody vegetation on subsurface carbon distribution. Total dissolved carbon at the BHF site (average=100.46 mg L^{-1}) was more homogenously distributed and at higher levels than Hinkson Creek (average=40.75 mg L^{-1}). Conversely, total dissolved carbon at the Ag site (average=73.91 mg L^{-1}) was more heterogeneously distributed, but still greater than levels within the creek (average=40.26 mg L^{-1}).

The apparent correlation of concentrations of Ca, Mg, and TC visible in Figures 5 and 6 (i.e. zones of high concentration in the northwest corner of the BHF and the east side of the Ag site) could indicate the influence of bedrock dissolution. As noted, both sites are underlain by Mississippian series limestones (Unklesbay, 1952; Miller and Vandike, 1997). The dissolution of carbonates contained in limestone (e.g. calcite and dolomite) can release Ca, Mg, and CO_3 (Chou et al., 1989). Thus, increased concentrations of those three constituents in specific locations could be the result of bedrock weathering.

Nitrate concentration patterns at the two sites highlight two important features. First, the zone of higher nitrate levels on the western side of the BHF site (4.6 mg L^{-1}, as compared to site average of 2.5 mg L^{-1}) could be the result of nitrate introduction to the subsurface from the adjacent small tributary (Figure 1). The confluence of the tributary and Hinkson Creek is located near the area of high nitrate concentration. Thus, the nitrate enriched zone could be an effect of surface water/groundwater exchange processes, thereby illustrating the potential for the BHF subsurface to act as a sink for surface water quality concerns such as nitrate. Second, shallow groundwater nitrate concentration at the Ag site is oriented in a high-to-low pattern from the back of the site toward the creek, a direction consistent with the prevailing direction of groundwater flow. Thus, although average nitrate levels were significantly ($p<0.05$) higher at the Ag site (3.0 mg L^{-1}), relative to the BHF (2.5 mg L^{-1}), the decreasing nitrate concentration gradient with respect to groundwater flow direction suggests potential nitrate remediation within the Ag subsurface. Notably, and despite the altered condition of the Ag subsurface resulting from historic forest removal, the floodplain retained a certain degree of functionality in its role as a shallow groundwater biogeochemical sink at the time of this work. Despite this observation, observed differences in subsurface chemical concentration patterns/distributions are likely the result of vegetative differences (e.g. Ca, Mg, TC) and contrasting site hydrology (e.g. S, K, nitrate).

General Study Observations

Study results suggest that historic forest removal has altered the biogeochemistry and hydrology of the Ag site, resulting in quantifiable contrasts in shallow groundwater chemical composition that persist after nearly a century. Such conclusions are crucial to improved understanding of land use/land cover change impacts on groundwater chemistry and highlight the importance of floodplain forests for the effective management of water resources. Considering the coupling of groundwater and surface water quality, study results support the use of riparian forest buffers to protect groundwater connected aquatic ecosystems by the effective recycling of macronutrients and uptake of trace elements. Considering forest harvest occurred approximately a century ago, the current work compares two stable sites, thereby providing quantitative evidence highlighting the long-term benefit of managed riparian forests to improve water quality. Future research should incorporate precipitation chemistry data, redox information, and/or isotope analysis to facilitate a process-based investigation of biogeochemical dynamics, thereby elucidating the specific mechanisms responsible for observed differences in study site groundwater chemical composition.

Conclusions

To address knowledge gaps concerning groundwater chemistry impacts after a century of land use change, a study was initiated in 2011 to compare the chemical composition of groundwater from two study sites with contrasting land use histories. An equidistant grid of nine piezometers was installed in a remnant bottomland hardwood forest and a historic (i.e. fallow) agricultural field. Groundwater was sampled on a monthly basis and analyzed for 49 physiochemical metrics. Results highlight the extensive impacts of land use history on groundwater chemistry. Mann-Whitney tests indicated significant differences ($p<0.05$) between the study sites for 32 out of 49 parameters. Compared to the Ag site, BHF groundwater was characterized by significantly ($p<0.05$) lower pH, higher electrical conductivity, and higher concentrations of total dissolved solids and inorganic carbon. BHF groundwater contained significantly ($p<0.05$) higher concentrations of nitrogen species except nitrate, which was higher in Ag groundwater. Furthermore, BHF groundwater contained significantly ($p<0.05$) higher concentrations of nutrients such as sulfur, potassium, magnesium, calcium, and sodium, relative to the Ag site. Conversely, Ag groundwater was characterized by significantly ($p<0.05$) higher concentrations of trace elements such as arsenic, cadmium, cobalt, copper, molybdenum, nickel, and titanium. Potential explanations for the observed differences include contrasting site hydrology (i.e. infiltration rates, groundwater recharge and flow), greater uptake of trace elements by woody vegetation, agriculturally-driven nutrient depletions, and the accumulation and effective recycling of nutrients by forest vegetation. Comparison of shallow groundwater chemical composition with that of Hinkson Creek suggests that subsurface concentration patterns are the result of contrasting site hydrological connectivity between shallow groundwater and the creek and vegetated cover type. Results detail potential impacts of surface vegetation alteration on subsurface chemistry and groundwater quality, thereby illustrating land use impacts on the lithosphere and hydrosphere. This study is one of the first to comprehensively characterize and compare the chemical composition of groundwater from sites with contrasting land use histories, approximately a century after initial land use change. Findings hold important implications for land and water managers, and support the use of floodplain and riparian forests for the conservation of both groundwater and surface resource quality.

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CHAPTER V

CONCLUSIONS AND SYNTHESIS

Summary

Studies have repeatedly and conclusively shown that land use change modifies hydrologic regimes, thus altering rates of mass and energy flux (Twine et al., 2004; Karwan et al., 2007; Bulliner, 2011; Freeman, 2011). Similar to other freshwater resources, subsurface water is likewise impacted by land use. For example, vadose zone water is impacted by surface vegetation communities in terms of the spatial distribution of plants and species-specific rates of water use (Breshears et al., 1997). As a result of canopy interception, stemflow, and throughfall, vegetation can produce spatiotemporal variability in infiltration and distribution of soil water (Martello et al., 2015). Additionally, removal of the forest canopy could increase surface heating due to increased solar radiation at the ground surface, and thus alter baseflow temperature via soil heat flux. Alterations of groundwater thermal regimes are an issue of concern considering the importance of groundwater temperature to a variety of surface and subsurface biogeochemical processes, such as the metabolic rates, physiology, and life history traits of aquatic species, interstitial habitat suitability, and the nutrient cycling and productivity of aquatic communities (Poole and Berman, 2001; Sophocleous, 2002); and rates of groundwater biogeochemical reactions (Mitchell and Ferris, 2005; Figura et al., 2011). Furthermore, land use/land cover changes can impact subsurface water chemistry. For example, Pionke and Urban (1985) reported concentrations of nitrate, chloride, and phosphate in groundwater that were five to seven times higher under cropland, relative to forest land use. Similarly, Balestrini et al. (2011) showed that alteration of subsurface flow patterns due to consumptive water use by woody vegetation (i.e. evapotranspiration) was a primary factor contributing to riparian groundwater nitrate removal rates.

Subsurface water distribution, variability, and quality are particularly critical in floodplain landscapes, which serve as the interface between subsurface and surface water systems. Floodplains can provide numerous ecosystem services such as biogeochemical transformation and remediation of nutrients and pollutants, runoff regulation, and flood water mitigation (Krause et al., 2007). Typically, floodplains attenuate floods by retaining water and decreasing velocity, thereby slowing and reducing flood wave impacts (Wheater and Evans, 2009). However, downstream flood risk can also be mitigated by encouraging maximum floodplain infiltration capacity and retention of floodwater in the subsurface (Hubbart et al., 2011; Wheater and Evans, 2009). Land use/land cover changes such as forest removal can alter subsurface hydraulic processes, including infiltration rate, percolation, and soil water retention (Hubbart et al., 2011; Wahren et al., 2009), thereby reducing floodplain attenuation potential. Thus, vadose zone water capacity can dramatically influence flood attenuation potential (Hubbart et al., 2011). Considering the importance of floodplains to the mitigation of flood risk, the maintenance of groundwater and surface water quality, and the support of aquatic ecosystems, research is needed to quantitatively characterize the impacts of different land use/land cover types on the floodplain subsurface hydrology.

The general objective of this dissertation research was to improve understanding of the impacts of different rural land use types on floodplain subsurface water resources in mixed-land-use watersheds of the central U.S. Specific objectives were to a) utilize high-resolution, continuous, *in situ* soil water sensor data and geostatistical analyses to compare the spatiotemporal distribution of vadose zone volumetric water content of 100 year old agricultural and forested floodplain sites; b) characterize and compare shallow groundwater (SGW) temperatures 1-2 m below the water table at agricultural and forested sites, and relate observed SGW temperature contrasts to site differences in estimated groundwater flow and microclimate; c) quantitatively describe and compare groundwater chemical composition in an intensive gridded investigation of historic agricultural and forested floodplain sites.

Soil Volumetric Water Content

This study was one of the first to integrate continuous, automated, *in situ* monitoring in a multi-depth, gridded sampling design with geostatistical analytical methods to characterize the spatiotemporal distribution of vadose zone water in floodplains with contrasting land use histories. Results showed VWC to be significantly different between sites ($p<0.01$) during the study, with site averages of 33.1 and 32.8% at the Ag and BHF sites, respectively. Semi-variogram analyses indicated the presence of strong (<25%) horizontal and vertical spatial correlation of VWC at the Ag site, and a relatively short-range (25 cm) vertical spatial correlation at the BHF, but only indicated horizontal VWC spatial correlation in the top 30 cm of the BHF profile. Likely mechanisms contributing to patterns of observed differences include contrasting rates and depths of plant water use, and the presence of preferential flow paths in the BHF subsurface. Results suggest historic forest removal and cultivation of the Ag site lead to an effective homogenization of the upper soil profile, and facilitated the development of strong VWC spatial dependency. Conversely, higher hydraulic conductivity of the more heterogeneous BHF subsurface likely results in a wetting of the deeper profile (75 cm) during climatically wet periods (a hypothesis supported by results of seasonal analyses), and thus a potentially more effective processing of hydrologic inputs (e.g. floodwaters). The use of automated monitoring in the current work provided a temporally-rich dataset, more capable of characterizing VWC spatiotemporal variability than traditional episodic sampling methods. Furthermore, whereas previous VWC studies incorporating semi-variogram analyses applied the method qualitatively or as a preliminary modeling step, the current work used the method to identify and quantify spatiotemporal VWC patterns and support mechanistic hypotheses, thereby making results of the analyses more informative. Study results indicate the detailed information the integration of automated, gridded, multi-depth monitoring and geostatistical analysis can produce and its value for vadose zone water research. Collective results highlight the greater extent and degree to which forest vegetation impacts subsurface hydrology, relative to grassland/agricultural systems, and point to the value of reestablishing floodplain forests for freshwater routing, water quality, and flood mitigation in mixed-land-use watersheds.

Shallow Groundwater Temperature

This study was one of the first to utilize long-term, continuous, automated, *in situ* monitoring to investigate rural land use impacts on shallow groundwater temperatures. Results confirmed that forest removal can influence groundwater thermal regimes, thereby supporting the results of previous studies based primarily on episodic sampling and reductive process modeling. Average SGW temperature during the study period was 11.1 and 11.2 °C at the Ag and BHF sites, respectively. However, temperature range at the Ag site was 72% greater than at the BHF site. Results indicate a greater responsiveness to seasonal climate fluctuations in Ag site SGW temperature related to absence of forest canopy. Patterns of intra-site groundwater temperature differences at both study sites illustrate the influence of stream-aquifer thermal conduction and occasional baseflow reversals. Previous studies have indicated increased vertical heat conduction as a primary mechanism for forest removal impacts on groundwater temperature regimes. However, considering similar observed surface soil temperature amplitudes and low average groundwater flow values at both sites, results of the current work suggest that rates of plant water use, groundwater recharge, and subsurface hydraulic conductivity are likely mechanistic causes for the observed SGW temperature differences. Contrasting findings are likely related to the time scales involved in the studies. For example, whereas previous studies have investigated the effects of recent canopy removal on groundwater resources, the current research addressed groundwater thermal regimes more than 100 years after forest removal. Thus, results of the study highlight the long-term impact of land use/land cover change on subsurface hydrology and groundwater temperature regime, a valuable contribution to the existing body of research on groundwater temperature. Moreover, given the importance of groundwater temperature to the biogeochemical health of aquatic ecosystems, study results point to the potential for floodplain forest removal to adversely impact both groundwater and surface water quality.

Shallow Groundwater Chemical Composition

This study is among the first to comprehensively characterize and compare shallow groundwater chemical composition at sites with contrasting land use histories. Statistical tests indicated significant differences ($p<0.05$) between the study sites for 32 out of 49 parameters. Compared to the Ag site, BHF groundwater was characterized by significantly ($p<0.05$) lower pH, higher electrical conductivity, and higher concentrations of total dissolved solids and inorganic carbon. BHF groundwater contained significantly ($p<0.05$) higher concentrations of nitrogen species except nitrate, which was higher in Ag groundwater. BHF groundwater contained significantly ($p<0.05$) higher concentrations of nutrients such as sulfur, potassium, magnesium, calcium, and sodium, relative to the Ag site. Higher groundwater concentrations of nutrients at the BHF, relative to the Ag site, are presumably attributable to processes of nutrient recycling and accumulation by forest vegetation, and agriculturally-driven nutrient depletions. Conversely, Ag groundwater was characterized by significantly ($p<0.05$) higher concentrations of trace elements such as arsenic, cadmium, cobalt, copper, molybdenum, nickel, and titanium. Possible explanations for lower groundwater concentrations of trace elements at the BHF site, relative to the Ag, include phytoremediation (i.e. uptake, stabilization, sequestration, and/or volatilization) of shallow groundwater by BHF vegetation and seasonal flushing of the BHF site aquifer by Hinkson Creek via baseflow reversals. Comparison of shallow groundwater chemical composition with that of nearby receiving water suggests that subsurface concentration patterns are the result of contrasting site hydrological connectivity between shallow groundwater and the creek and vegetated cover type. Study results suggest that historic forest removal has altered the biogeochemistry and hydrology of the Ag site, resulting in quantifiable contrasts in shallow groundwater chemical composition that persist after nearly a century. Such conclusions are crucial to improved understanding of land use/land cover change impacts on groundwater chemistry and highlight the importance of floodplain forests for the effective management of water resources. Given the coupling of groundwater and surface water quality, study results support the use of riparian forest buffers to protect groundwater connected aquatic ecosystems by the effective recycling of macronutrients and uptake of trace elements. Considering forest harvest occurred approximately a century ago, the work compares two stable sites, thereby providing quantitative evidence highlighting the long-term benefit of managed floodplain forests to improve water quality.

Synthesis

In each of the preceding investigations, results indicated the influence of forest vegetation as a primary factor controlling observed site differences, thereby highlighting the potential benefits of maintaining and restoring floodplain forests. Contrasting rates and depths of plant water use, and the presence of preferential flow paths in the BHF subsurface were identified as likely explanations for observed spatiotemporal patterns of vadose water distribution. Greater rates of groundwater recharge at the Ag site, relative to the BHF, were proposed as the primary mechanism contributing to observed differences in shallow groundwater temperature regimes. This hypothesis is supported by the results of Zell et al. (2015), who estimated 400% greater groundwater recharge at the Ag, relative to the BHF, during the 2011 and 2012 water years. Contrasting rates of groundwater recharge were, in turn, found to be related to 16% greater transpiration at the BHF, and higher BHF subsurface hydraulic conductivity (i.e. more effective lateral transmission of water) (Zell et al., 2015). Similarly, plant uptake and nutrient recycling by forest vegetation, and greater hydraulic connectivity of the BHF aquifer to Hinkson Creek, relative to the Ag, were shown to be the most likely processes contributing to observed differences in shallow groundwater chemical composition at the study sites. Thus, in each study, greater plant water use by forest vegetation and a more physically heterogeneous and transmissive BHF subsurface, likely due to preferential flow paths created by woody roots, were identified as the primary factors explaining observed differences between the two floodplain sites.

Results and conclusions of this work are critically important for land and water resource managers seeking more effective strategies for the protection and conservation of both groundwater and surface water. Considering the demonstrated potential for floodplain forests to more effectively attenuate floodwaters, better protect the temperature regime of groundwater and adjacent surface water systems, more efficiently recycle and accumulate nutrients in the subsurface, and promote surface water processing via seasonal aquifer-stream exchange, results point to the reestablishment of floodplain forests as a valuable alternative/addition to traditional engineering methods of watershed management (e.g. built structures, hydrologic routing, and surface water impoundments). Despite initially rich soils, many floodplains in the central U.S. are currently marginally productive due to agriculture-driven nutrient depletion; and late spring-early summer inundation from stream backwatering and flooding can result in long periods of saturated soil, which can adversely impact crop production (Stanturf et al., 1998). As opposed to remaining fallow or continuing to be cultivated despite low profitability, such marginally productive floodplains could be of greater use if converted to bottomland hardwood forests. In other words, a natural resource commodities valuation could indicate floodplain forests to be more socioeconomically valuable than marginally productive floodplain agricultural fields.

Future Work

Results of the VWC study suggested the influence of contrasting rates of plant water use and subsurface hydraulic conductivity (i.e. preferential flow paths) on the spatiotemporal distribution of vadose zone water. However, the primary influence of those two factors remains a hypothesis. Future work should quantitatively characterize both factors, in order to support or refute such conclusions. For example, vadose zone hydraulic conductivity and vadose zone-groundwater-surface water connectivity could be determined via a tracer study. Regardless, given the detailed VWC information provided by the current work, future studies should incorporate a more process-based approach to characterize the subsurface hydrology of the sites. Although groundwater temperature results suggested the influence of multiple mechanistic processes (i.e. conduction, recharge, and infiltration), more work is needed to determine the extent to which each process is contributing to observed differences. For example, considering its importance to subsurface thermal regime, future work should include measured ground surface temperature, which could provide further information on the impacts of contrasting surface vegetation on subsurface temperatures. Likewise, coupling surface to subsurface heat flux and hydrologic modeling, and incorporating measured microclimate and hydrologic data (e.g. incident ground surface shortwave radiation, ground surface temperature, volumetric soil water content), could help quantify the impact of vertical water flow on the temperature regime of floodplain shallow groundwater. Future shallow groundwater chemical composition research should incorporate precipitation chemistry data, redox information, and/or isotope analysis to facilitate a process-based investigation of biogeochemical dynamics, thereby elucidating the specific mechanisms responsible for observed differences in study site groundwater chemical composition. Specifically, the measurement of dissolved oxygen and the inclusion of speciation modeling could provide information regarding the toxicity and biological availability of the chemical compounds within floodplain shallow groundwater.

Closing Comments

Globally, forests are being cleared, primarily in order to expand agricultural land uses (Foley et al., 2005; Rudel et al., 2005). In the central U.S., the historic removal of the majority of floodplain forests (Stanturf et al., 1998) has resulted in alterations to flows of mass and energy at the terrestrial-aquatic interface, culminating in radical changes to the hydrology and biogeochemistry of mixed-land-use watersheds (Hubbart, 2011). Flood risks, groundwater quality issues, and aquatic ecosystem degradation continue, regardless of persistent efforts to address subsequent water resource management problems via engineering methods. Research suggests that natural resource problems stemming from anthropogenic disturbance cannot be adequately mitigated by further anthropogenic disturbance (e.g. built structures, hydrologic routing, and surface water impoundments) (De Laney, 1995; Hubbart, 2011). Instead, it may be advisable to allow the landscape to function according to hydrologic processes evolved over geologic time. The results of this dissertation research identify several potential benefits to floodplain forest reestablishment; among them are effective processing of hydrologic inputs and potential mediation of downstream flood risk, conservation of shallow groundwater and thus aquatic ecosystem thermal regimes, and protection of groundwater and surface water chemical quality via plant uptake and nutrient recycling. Fittingly, these potential benefits correspond to issues currently facing the Hinkson Creek Watershed. Thus, the current work overwhelmingly supports the use of floodplain forests as a land and water resource management practice in mixed-land-use watersheds of the central U.S.

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APPENDIX

Experimental semi-variogram analyses included in chapter two were performed according to the following equation:

$$\gamma^* = \frac{\frac{1}{n}\sum[g(x) - g(x+h)]^2}{2} \qquad \text{Eq. 1}$$

where γ^* is the experimental semi-variogram ($\%^2$), g(x) is a VWC measurement at point x, and g(x + h) is a VWC measurement at a point located a distance (h) from point x. Thus, the analysis was conducted by calculating the differences between measurements at every sampling location. Given the six volumetric water content (VWC) monitoring locations at each study site, there were 15 unique comparison pairs available at each depth for the horizontal analysis. Given the five measured depths (i.e. 15, 30, 50, 75, and 100 cm) at each location, there were 10 unique comparison pairs at each sampling location for the vertical analysis. Unique comparison pairs for both horizontal and vertical analyses, with associated lag distance, are presented in Table 1. In semi-variogram analysis, statistical robustness is a function of the numbers of unique pairs (i.e. sample size). Thus, to increase the robustness of the analyses, semi-variogram results from all five measurement depths were integrated (i.e. summed; see equation 1) for the site average horizontal analysis (Figures 1 and 2), thereby bringing the number of unique pairs to 75. Similarly, semi-variogram results from all six sampling locations were integrated for the site average vertical analysis (Figures 1 and 2), thereby bringing the number of unique pairs to 60. Figures containing semi-variogram results are presented here with scatter points and straight (i.e. unsmoothed) lines to clearly note semi-variogram values at specific lag distances.

While integrating sampling locations in the vertical analysis is intuitive, integrating semi-variogram values from different depths for the horizontal analysis may, to some, appear problematic. Accordingly, it is acknowledged that VWC at different depths exhibited distinct spatial relationships, presumably due to contrasting patterns of plant water use and preferential flow paths. However, the semi-variogram method analyzes relative differences between comparison pairs. VWC measurements at separate depths *and* separate sampling locations were not compared. Rather, the relative differences of measurements at the same depth were integrated (i.e. summed; see equation 1) to produce an estimate of horizontal spatial correlation across all measured depths. Moreover, contrasting spatiotemporal VWC relationships at separate depths were the impetus for the inclusion of Figures 3 and 4 in chapter two. Geostatistical analyses were intended to describe the large-scale (i.e. "big picture") spatiotemporal trends of soil water content at the study sites. The analysis was successful in that this objective was met, thereby providing new information concerning the field-scale spatial correlation of VWC at sites with contrasting land use histories. Future work should include a greater number of sampling locations (i.e. >6), thereby increasing the number of unique pairs and facilitating more statistically robust semi-variogram analyses, free of the requirement to integrate values from separate depths. Such analyses could provide more reliable results concerning VWC spatial relationships at separate depths. Furthermore, applying a richer sampling frequency could provide more information regarding horizontal spatial correlation, if the minimum lag distance was reduced. In the current work, the minimum lag distance was 40 m. Although the bottomland hardwood forest (BHF) site did not display definable VWC horizontal spatial correlation at the site level (Figure 1), it is possible that correlation trends existed, but at a shorter range. Thus, increasing the spatial sampling frequency and reducing the minimum lag distance should be considered for future work.

Experimental semi-variogram analyses applied in chapter two were performed on VWC data averaged over the duration of the study period (Figures 1, 3, and 4) and seasonally (Figure 2). For study period duration, this required averaging approximately 52,500 data points. As the seasons (i.e. plant growth and dormancy) described in chapter two were each six months long (i.e. April - September and October - March, respectively), approximately 26,000 data points were averaged for each period. Undoubtedly, data aggregation of such a scale produces coarse results, lacking in the degree of variation that a finer resolution analysis could provide (thus, providing impetus for future work). However, as noted, the analyses performed in chapter two were intended to describe the large-scale (i.e. “big picture”) spatiotemporal patterns of soil water content at the study sites. Results achieve this objective. Future work could include similar analyses performed on a smaller temporal scale, for example, event-based analyses of wetting and drying processes, and thus describe spatiotemporal VWC trends at a finer resolution.

Table 1. Volumetric water content (VWC) semi-variogram comparison pairs for horizontal and vertical analyses, with associated lag distances.

	Distance	# of Pairs
Horizontal	40 m	1
	44.7 m	4
	72.1 m	4
	80 m	4
	113.1 m	2
	Total	15
Vertical	15 cm	1
	20 cm	1
	25 cm	2
	35 cm	1
	45 cm	1
	50 cm	1
	60 cm	1
	70 cm	1
	85 cm	1
	Total	10

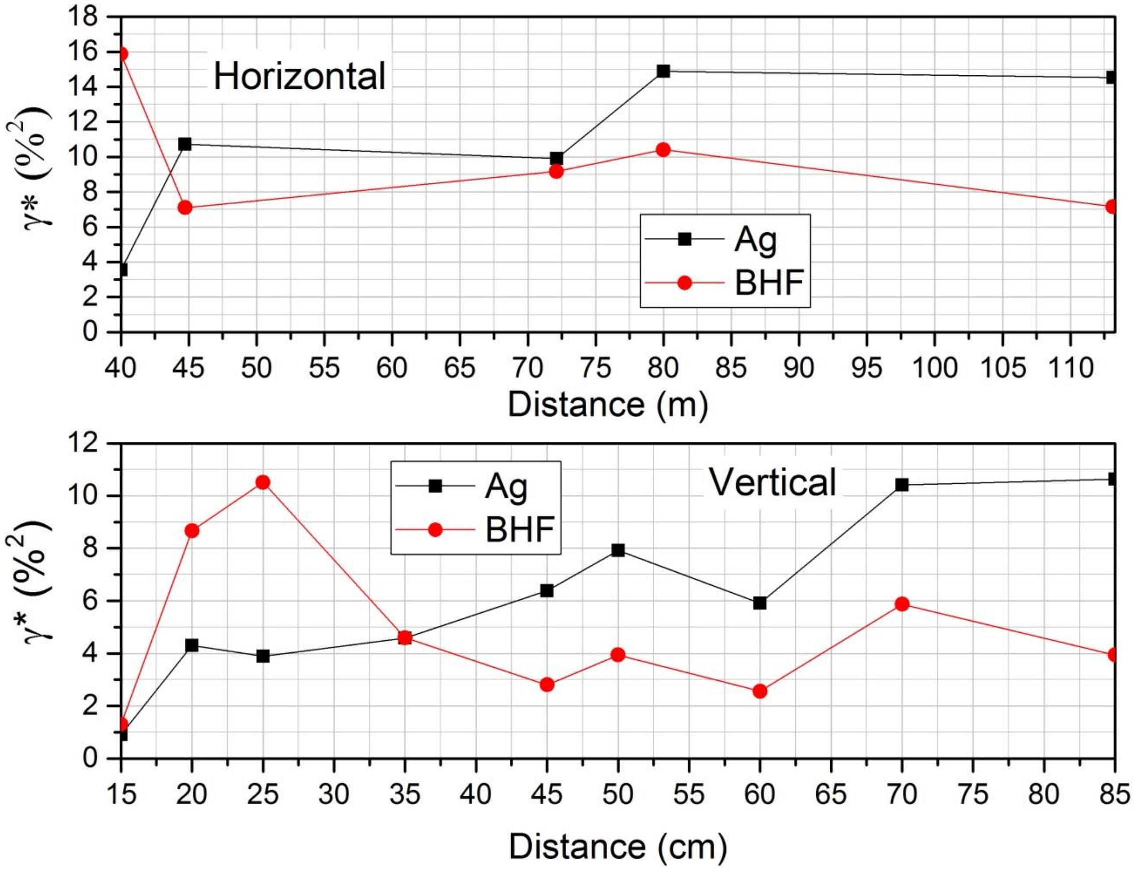

Figure 1. Experimental semi-variogram of soil volumetric water content in horizontal and vertical directions during 2011, 2012, and 2013 water years at agricultural (Ag) and bottomland hardwood forest (BHF) sites, Hinkson Creek Watershed, Missouri, USA.

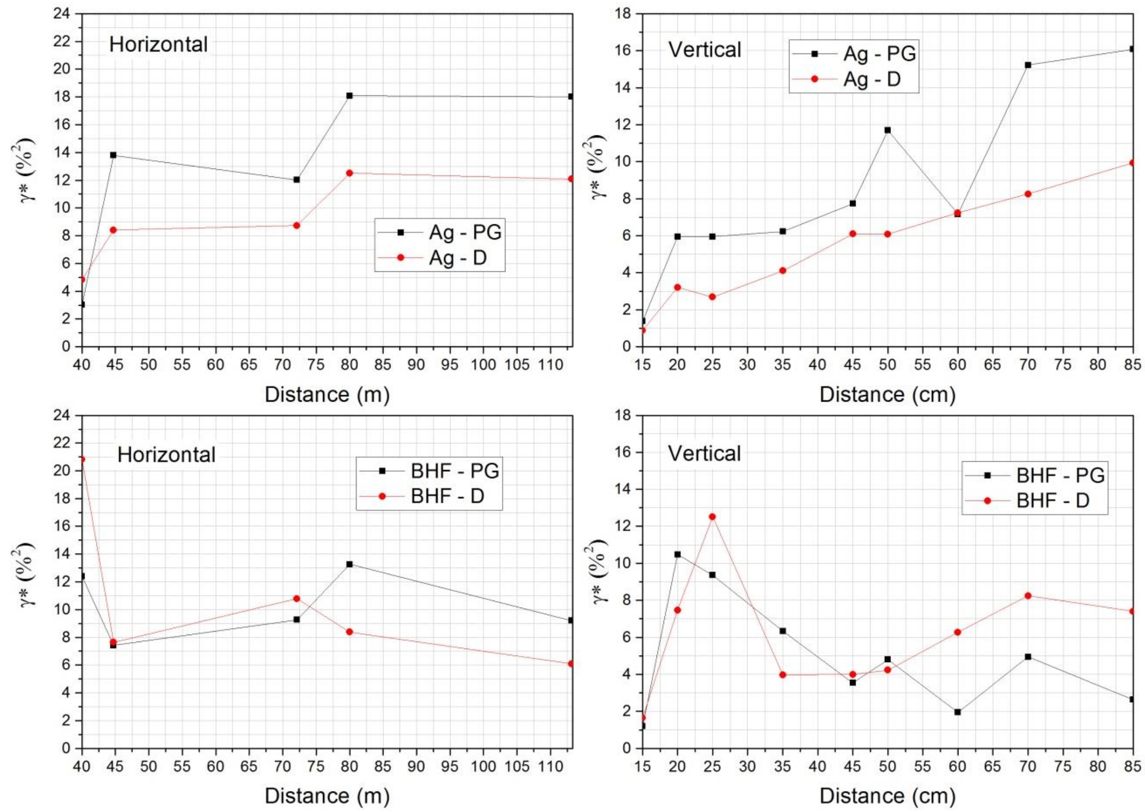

Figure 2. Experimental semi-variogram of soil volumetric water content in horizontal and vertical directions during seasons of plant growth (PG) and dormancy (D) at agricultural (Ag) and bottomland hardwood forest (BHF) sites, Hinkson Creek Watershed, Missouri, USA.

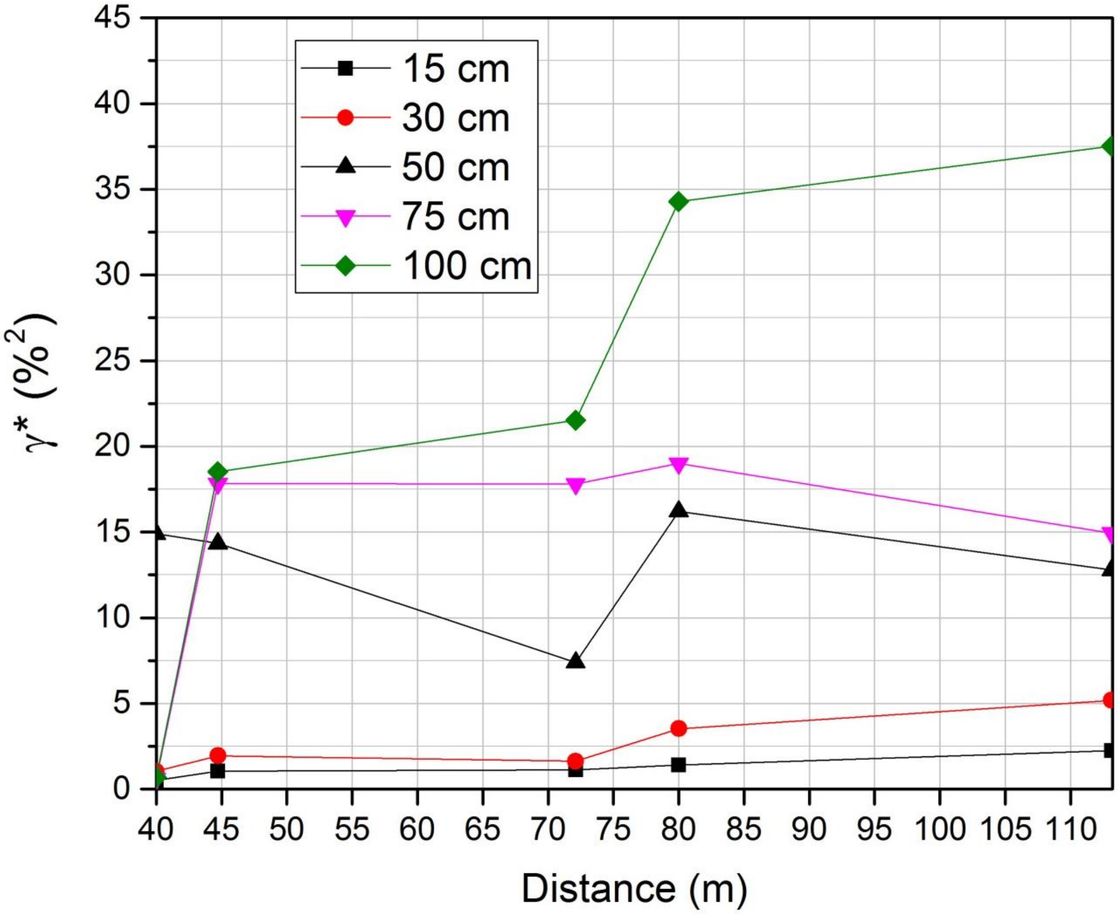

Figure 3. Experimental semi-variogram of soil volumetric water content in horizontal direction at five measured depths during 2011, 2012, and 2013 water years at agricultural (Ag) site, Hinkson Creek Watershed, Missouri, USA.

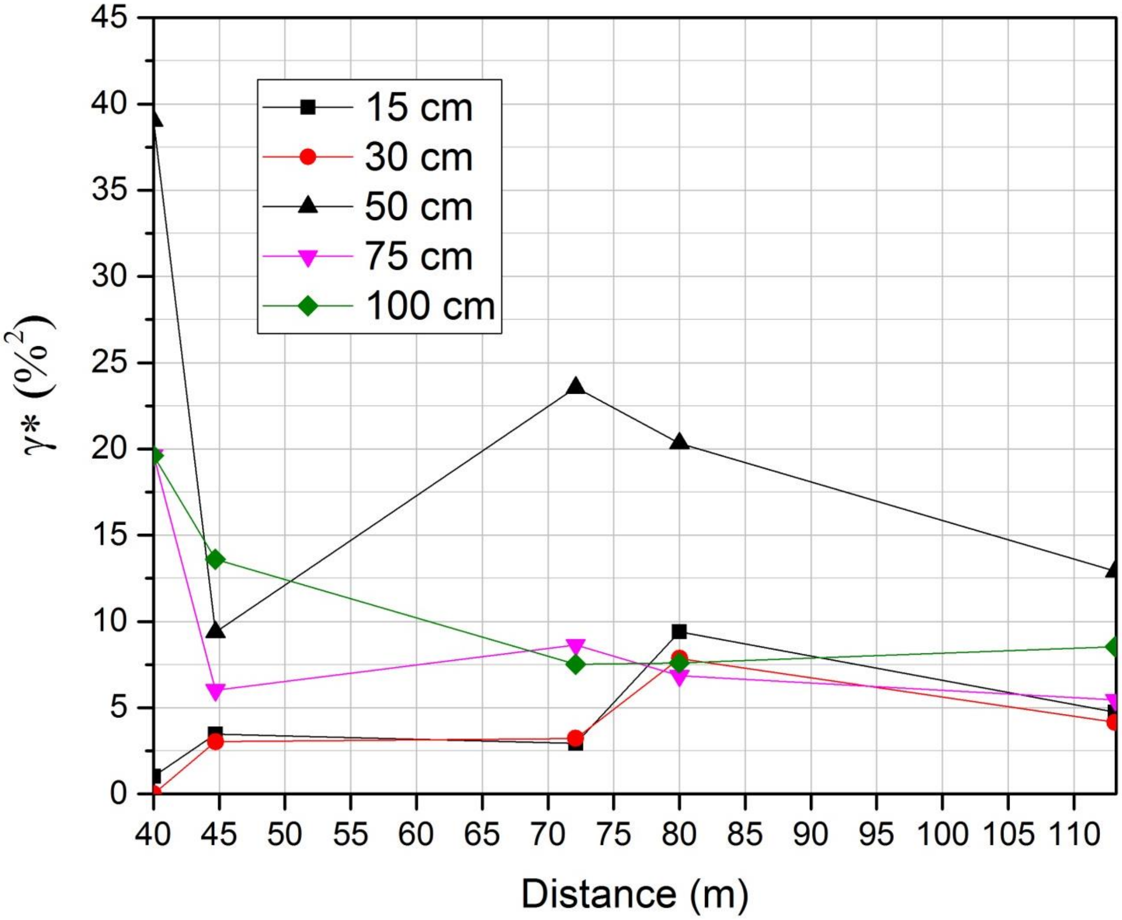

Figure 4. Experimental semi-variogram of soil volumetric water content in horizontal direction at five measured depths during 2011, 2012, and 2013 water years at bottomland hardwood forest (BHF) site, Hinkson Creek Watershed, Missouri, USA.

VITA

Elliott Kellner grew up in Springfield, Missouri. He graduated from the University of Missouri in 2005 with a Bachelor of Arts. He returned to the University of Missouri in 2011 to pursue a Master's degree in Forestry, graduating in 2013. Following completion of his doctoral program, he intends to continue full-time hydrologic research.