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Long-term Effects of Broiler Litter Application Rate on Runoff, Leaching, and Soil Respiration from a Captina Silt Loam
Long-term Effects of Broiler Litter Application Rate on Runoff, Leaching, and Soil Respiration from a Captina Silt Loam

A dissertation submitted in partial fulfillment of the requirements for a degree of Doctor of Philosophy in Crop, Soil, & Environment Sciences

by

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Abstract

Producers in regions with intense broiler (*Gallus gallus*) production take advantage of the plant nutrients contained in broiler litter (BL) to enhance yields of forage grasses. However, application of BL to pastures in karst regions, like the Ozark Highlands, can potentially reduce water quality if BL-derived contaminants enter surface or groundwater via runoff or drainage. Additionally, BL applications stimulate carbon dioxide (CO₂) release from the soil to the atmosphere and may contribute to global warming. The objectives of this study were to determine long-term trends in runoff and soil leachate water quality and to evaluate soil respiration under natural precipitation from a Captinasilt-loam soil (fine-silty, siliceous, active, mesic Typic Fragiudult) with a history of BL amendments under forage management amended annually with BL at three application rates (0, 5.6, and 11.2 Mg BL ha⁻¹). Runoff and leachate were collected for an 8-yr period and pH, electrical conductivity (EC), soluble plant nutrients (i.e., NO₃-N, NH₄-N, PO₄-P, Ca, K, Mg, Na, and P), trace metals (i.e., As, Cd, Cr, Cu, Fe, Mn, Ni, Se, and Zn), and dissolved organic carbon were measured. Similarly, soil respiration, temperature, and moisture were measured periodically for a 3-yr period. Litter application increased \( P < 0.01 \) average annual flow-weighted-mean (FWM) runoff Fe and leachate Na concentrations, while all other annual runoff and leachate concentrations and loads and 8-yr cumulative losses were unaffected \( P > 0.05 \) by BL rate. Eight-year cumulative runoff losses were < 1 % for all elements except Se, which ranged from 12 to 20 % of that applied in litter. Similarly, 8-year cumulative leaching losses of NH₄-N, C, N, P, Mn, and Cu represented < 2%, while Se and Cd exceeded 100% of that applied in BL. Soil respiration varied \( P < 0.01 \) with BL rate, sample date, and year. Litter increased respiration after application and again after rain events relative to the unamended control in all years. Multiple regression indicated that
respiration could be predicted using soil temperature at the 2-cm depth (T_{2cm}) and the product of T_{2cm} and soil volumetric water content (R^2 = 0.52; P < 0.01). Results indicate that pasturelands with a history of BL application may release BL-derived metals in runoff and drainage waters at concentrations harmful to health and that organic amendments, such as BL, can stimulate net release of CO_2 from soil to the atmosphere, potentially negatively affecting atmospheric greenhouse gas concentrations.
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To Alexandria, Isaac, and Valerie
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List of Abbreviations

AAC, ambient atmospheric concentration
AEC, anion exchange capacity
AETL, automated equilibrium tension lysimeter
ANCOVA, analysis of covariance
ANOVA, analysis of variance
AR, acid-recoverable
ATP, adenosine triphosphate
BL, broiler litter
CEC, cation exchange capacity
CFR, Code of Federal Regulations
CI, confidence interval
CI<sub>L</sub>, lower 95% confidence interval
CI<sub>U</sub>, upper 95% confidence interval
CO<sub>2</sub>-C, carbon dioxide-carbon
COD, chemical oxygen demand
DM, dry matter
DM-C, dry matter-carbon
DOC, dissolved organic carbon
DPBL, days-post broiler litter application
DRP, dissolved reactive phosphorus
EC, electrical conductivity
ETL, equilibrium tension lysimeter
FWM, flow-weighted-mean
ICP, inductively coupled, argon plasma mass spectrometry
LSD, protected least significant difference
M3, Mehlich III extractable
MCL, National Primary Drinking Water Regulations’ Maximum Contaminant Level
n, number of observations
NH<sub>4</sub>-N, ammonium-N
NO<sub>3</sub>-N, nitrate-N
ORP, oxidation-reduction potential
P, total phosphorus, including PO<sub>4</sub>-P but not particulate-P greater than 0.45 µm diameter
PO<sub>4</sub>-P, phosphate-phosphorus
PV, soil pore volume
PVC, polyvinyl chloride
r, correlation coefficient
R<sup>2</sup>, coefficient of determination
R<sub>s</sub>, soil respiration
SAR, sodium adsorption ratio
SOM, soil organic matter
$T_{2cm}$, soil temperature at 2-cm depth
$T_{10cm}$, soil temperature at 10-cm depth
VWC, soil volumetric water content
WE, water-extractable
WFPS, water-filled pore space
Introduction

Nationwide, the United States produced 8.4 billion broiler chickens (*Gallus gallus*) in 2012. Arkansas’ production of 1.0 billion birds in 2012 ranked third within the United States for number of birds produced (USDA-NASS, 2013). Broiler production generates between 1.2 and 1.7 Mg of boiler litter (BL) per 1000 birds (UADACES, 2014). Consequently, Arkansas’ broiler industry produced 1.2 to 1.7 million Mg of BL in 2012, the majority of which was concentrated in northwest Arkansas in a region called the Ozark Highlands (Major Land Resource Area 116A; Scott and Ward, 2002; Brye et al., 2013).

Broiler litter contains numerous plant nutrients, such as nitrogen (N) and phosphorus (P), and is land applied to pastures in the Ozark Highlands to enhance yields of tall fescue (*Lolium arundinaceum* Shreb.) and other forages. In addition to water-soluble nutrients, BL also contains trace metals such as arsenic (As), cadmium (Cd), copper (Cu), lead (Pb), selenium (Se), and zinc (Zn). Since BL is generally not incorporated into the soil during application, surface runoff during rain events has the potential to contaminate nearby surface waters. Likewise, vertical movement of nutrients and metals within the soil profile could also contaminate groundwater through leaching. Additionally, land applied BL can stimulate carbon release from the soil to the atmosphere and has the potential to increase atmospheric greenhouse gas concentrations.

The southern portion of the Ozark Highlands is characterized by well-developed karst made possible by the extensive presence of soluble carbonate rock (Scott and Ward, 2002). As water moves over and through the carbonate rock, the rock slowly dissolves creating cracks, fissures, sink holes and caves. Once created, these features can allow rapid preferential movement of surface water to groundwater. This preferential movement makes groundwater susceptible to contamination from anthropogenic activities (MacDonald et al., 1976; Scott and
Ward, 2002; Graening and Brown, 2003; USDA-NRCS, 2006), such as surface application of BL to pasture soil.

The Ozark Highlands, specifically northwest Arkansas, is also an area with increasing population. In northwest Arkansas, populations in Benton and Washington Counties have increased 127 and 79%, respectively, between 1990 and 2010 (USCB, 2013a, b, c). With population growth, the demand for water increases. In order to maintain high water quality within the Ozark Highlands, the long-term effects of land-applied BL on surface runoff and subsurface leaching must be studied.

Additionally, anthropogenic activities, including land application of BL (Jones et al., 2005), can increase CO$_2$ release to the atmosphere, thereby potentially increasing global warming (Forster et al., 2007) and promoting climate change (Trenberth et al., 2007). Soil respiration, the combined release of CO$_2$ from the soil originating from microorganisms and plant roots, is increased with amendments of BL to the soil (Roberson et al., 2008; Jones et al., 2006; Jones et al., 2005; Adams et al., 1997). Although models have used soil temperature and moisture to predict soil respiration, few researchers, if any, have attempted to identify if BL amendments should be addressed when using predictive models.

A long-term project was initiated in 2002 at the University of Arkansas in Fayetteville to explore the long-term effects of BL amendments to soil with a history of litter application in response to naturally occurring precipitation. Two years of leaching data (Pirani et al., 2006; Pirani et al., 2007), one season of plant uptake data (Brye and Pirani, 2006), four years of runoff data (Menjoulet et al., 2009), five years of soil storage data (Daigh et al., 2009), and soil-arsenic adsorption characteristics data (McDonald et al., 2009) have been gathered as part of this ongoing study. The project contained herein represents research activities initiated in 2002 and
focuses on summarizing temporal variations and linear trends in runoff and leaching losses of nutrients and trace metals as a function of BL application rate. In addition, carbon cycling was evaluated by investigating BL application rate effects on soil respiration. The results of this study will aide future researchers in mass balance analyses and evaluation of the fate and transport of BL-derived components.
References


Available at http://usda01.library.cornell.edu/usda/current/PoulProdVa/PoulProdVa-04-29-2013.pdf (verified 20 Jan 2014).


Chapter One

Literature Review
Literature Review

The Protein Industry

The current business model utilized by the poultry industry incorporates vertical integration (USDA-NASS, 2002) and outsourcing to maximize profits (Martinez, 1999). Vertical integration controls all aspects of production, from egg laying, to hatching, to growing, to processing, and finally to transporting finished products to market. Outsourcing is utilized as a risk management tool. This organizational structure results in what is referred to as “complexes”. A complex is a self-contained unit capable of all steps within the vertical process. Complexes intensify resources within relatively small geographic areas.

Once hatched, broilers are housed in buildings where continuous monitoring and manipulation maximizes growth. Each house contains a flock ranging in size from 22,000 to 23,000 birds (Patterson et al., 1998). Broiler house waste is commonly referred to as litter or broiler litter (BL). Litter is a mixture of excreta, fecal material, feed, and a bedding material, usually saw dust or rice (Oryza sativa L.) hulls. The nutrient content of litter makes it a useful organic soil amendment (Hileman, 1973; Patterson et al., 1998; Chamblee and Todd, 2002; Brye and Pirani, 2006). Litter is land applied to pastures in the Ozark Highlands to enhance yields of tall fescue (Lolium arundinaceum Shreb.) and other forages.

Consolidation within the poultry industry has grown to include other protein sources, such as pork and beef. Acquisitions by companies utilizing vertical integration have resulted in expanded use of vertical integration into pork and beef industries. Expanded use of vertical integration will result in more intensified use of resources within smaller geographic regions where pork and beef are produced. Similar to the phenomenon of large quantities of litter being produced in relatively small geographic regions, vertical integration within the pork and beef
industries will intensify pork and beef waste production in ever-smaller regions of the nation. Additionally, the steadily growing poultry industry (Figure 1-1) will produce even more waste. These additional waste products and quantities will also need to be managed.

**Broiler Production**

Chickens that have been raised specifically for meat production are termed “broilers.” Nationally, the United States produced 8.4 billion broiler chickens in 2012. This production was down 2% from 2011. However, the value of these birds was up 8% from 2011 to $24.8 billion in 2012 (USDA-NASS, 2013). The live weight of broilers produced also decreased 1% from 2011 to 2012 (USDA-NASS, 2013). The broiler industry has sustained prolonged growth due primarily to consolidation and expanded production for many years (USDA-NASS, 2002; Figure 1-1).

Vertical integration within the poultry industry has intensified resources within relatively small geographic areas. What use to be millions of small backyard flocks spread throughout the U.S. is now billions of birds produced by fewer than 50 agribusinesses in small regional zones (Martinez, 1999; USDA-NASS, 2002; Kemper et al., 2006). Northwest Arkansas, eastern Oklahoma, and southern Missouri, collectively part of the Ozark Highlands ecological region, represent one of these zones of concentrated broiler production.

Broiler Litter Production and Composition

The chemical composition and production rates of BL are quite variable. This variability is influenced by regional practices, company practices, type of storage, amount and type of bedding, feed and feed additives, type of flooring, and number of flocks raised between cleanouts (Kunkle et al., 1981; Patterson et al., 1998; Mitchell and Donald, 1999; Hatten et al., 2001; Chamblee and Todd, 2002; Applegate et al., 2003; Garbarino et al., 2003). In Mississippi, BL production rates were reported to vary based on the number of flocks produced between cleanouts, with five flocks (one year’s production) generating 1.2 Mg BL per 1000 birds and 10 flocks (2 year’s production) generating 0.7 Mg BL per 1000 birds (Chamblee and Todd, 2002). In Pennsylvania, a similar study reported that BL production rates varied by company, with the company producing the most flocks between cleanouts generating less BL per 1000 birds (Patterson et al., 1998). Patterson et al. (1998) also reported an overall mean BL production rate of 1 Mg BL per 1000 birds. Decreased BL production due to increased number of flocks produced on the bedding has been attributed to decomposition and volatilization of organic substrates in the BL (Chamblee and Todd, 2002). The University of Arkansas estimated BL production rates of 1.2 to 1.7 Mg of BL per 1000 birds (UADACES, 2014). Based on the UADACES estimates, Arkansas’ broiler industry produced 1.2 to 1.7 million Mg of BL in 2012, the majority of which was located in the Ozark Highlands.

Broiler litter is a source of plant macro- and micro-nutrients when used as a soil amendment (Wilson et al., 2001; Patterson et al., 1998; Mitchell and Donald, 1999; Chamblee et al., 2002; Harmel et al., 2008; van der Watt et al., 1994). Table 1-1 reports the typical chemical composition of BL from northwest Arkansas for an 8-yr period. In Arkansas, BL has been shown to improve rice (Oryza sativa L.) yields on precision-graded soils (Miller et al., 1990;
Miller and Wells, 1992), increase forage yields (Hileman, 1973; Huneycutt et al., 1988; Brye and Pirani, 2006), and even provide a residual N boost to corn (Zea mays L.) one year after application (Mozaffari et al., 2007).

Broiler litter also contains trace metals (Table 1-1; Garbarino et al., 2003; Pirani, 2005; Menjoulet, 2007; Franzluebbers et al., 2004; van der Watt et al., 1994). Trace metals like As originate from dietary supplements such as 3-nitro-4-hydroxylarsonic acid (roxarsone) and 4-aminophenylarsonic acid (p-arsanilic acid) that are used to enhance growth by controlling coccidiosis, a disease caused by coccidian, a single-celled parasite of the intestine (Garbarino et al., 2003; Garbarino et al., 2009). Broiler dietary additives are regulated by the United States Food and Drug Administration (21 CFR 558.60 and 21 CFR 556.530; Code of Federal Regulations). Roxarsone passes through broilers with little or no change and then undergoes biotic degradation into arsenate (Garbarino et al., 2003). O’Connor et al. (2005) reported finding As contaminated dust in 88% of homes tested near fields receiving BL applications at levels in excess of the Environmental Protection Agency’s industrial indoor workers cancer-endpoint of 3.8 mg kg⁻¹. The top 60% of homes that tested positive for As contained a mean As concentration of 34.2 mg kg⁻¹. The subset of homes selected for dust-As speciation (n = 31) revealed that 97% of homes tested positive for roxarsone (O’Connor et al., 2005).

Land Application of Litter

In most agricultural systems, N is the limiting nutrient for plant growth. In the past, BL application rates were based on the N requirement of the forage crop being grown and the amount of N supplied by the BL. However, this approach resulted in the application of other
nutrients and trace metals in excess of crop requirements and caused accumulation in the soil profile or loss to surface and/or groundwater through runoff and/or leaching.

**Phosphorus Index**

The Phosphorus Index is a resource management tool intended to be used by planners to communicate to land users the relative potential for P to move within a landscape and potentially enter waterways (USDA-NRCS, 1995). The P Index assesses the risk level of P leaving the site and also helps identify management alternatives to help minimize and reduce P loss in runoff from that site (USDA-NRCS, 1995). The P Index may be modified to reflect local or regional characteristics to better accommodate local conditions (USDA-NRCS, 1995). Once adapted to a local/regional zone, the modified index must be tested to confirm its validity (USDA-NRCS, 1995).

The Arkansas P Index for Pastures utilizes weighting factors based on simulated runoff studies (DeLaune et al., 2004; DeLaune, 2002). The court-mandated development of a P index specifically for the Eucha/Spavinaw watershed in north east Oklahoma and far northwest Arkansas was satisfied by the modification of the current P Index for Pastures to create the Eucha/Spavinaw P Index (DeLaune et al., 2006), which has decreased BL applications within the watershed (Sharpley et al., 2009). Sharpley et al. (2009) stated that despite decreased BL applications to soil within the watershed, surface waters will not see immediate decreases in P inputs due to the elevated residual P in the soils in the watershed and in the river sediments.
Potential Fate of Nutrients and Trace Metals in Litter Applied to Soil

Runoff

In general, the term runoff or overland flow refers to surface flow of water that has been delivered to the soil surface in excess of the soil’s water absorption ability. Two types of runoff are identified by soil scientists based on the mechanisms that create runoff: infiltration excess overland flow and saturated excess overland flow (Pilgrim and Cordery, 1993). Geologists and hydrologists identify a third type of runoff, throughflow (Pilgrim and Cordery, 1993). Infiltration excess overland flow, also referred to as Horton overland flow in honor of the hydrologist Robert E. Horton, is surface runoff that occurs when the water delivery rate (i.e., precipitation, irrigation or snow melt) to the soil surface exceeds the rate of infiltration and surface storage has been exceeded (Pilgrim and Cordery, 1993). Saturation excess overland flow is surface runoff that occurs when the near-surface soil has become saturated to the point where infiltration is limited and surface storage has been exceeded (Pilgrim and Cordery, 1993). Throughflow is the rapid movement of water into the soil via macropores (i.e., preferential flow), the subsequent lateral movement via a saturated zone within the soil or parent material, and finally the discharge to a receiving stream. Throughflow is a rapid near-surface subsurface phenomenon (Pilgrim and Cordery, 1993). For the purposes of this dissertation, runoff will refer to surface flow only and throughflow will be collected and referred to as leachate or drainage.

Runoff, infiltration capacity, and the subsequent movement and redistribution of water within the soil are closely related concepts that govern runoff. Water interception by plants or ground cover, evapotranspiration, and surface storage also influence runoff. Factors governing runoff include: soil surface slope and slope length; precipitation rate, duration, and type; soil infiltration rate; plant or cover residue and raindrop interception; soil textural class or proportion
of sand, silt, and clay; clay type; soil structure and plasticity; soil porosity; antecedent soil moisture condition; soil bulk density or compactness; soil hydraulic conductivity; plant root density; soil sodic condition; subsoil features that might limit water movement and redistribution such as a plow pan or fragipan; and land use (Horton, 1933; Pilgrim and Cordery, 1993; Rawls et al., 1993; Harper et al., 2008).

When the precipitation rate at the soil surface exceeds the rate of infiltration, surface water accumulates (Hillel, 2004a). Depending on soil surface slope, surface water accumulates in low areas creating puddles, depression storage or surface storage (Fetter, 2001a). Only after depression storage capacity is exceeded will runoff commence. Runoff moves down-slope along the soil surface in response to gravitational forces. Initially, movement takes the form of sheet flow, but as the velocity increases, the amount of energy contained within the water increases its ability to erode the soil surface (Hillel, 2004a). Erosion channels runoff and promotes the formation of rills, gullies and/or other similar surface features (Hillel, 2004a).

Plant nutrients and trace metals at the soil surface are potentially susceptible to off-site transport through runoff. Runoff may transport nutrients and trace metals in either of two forms: the dissolved form or the particulate-bound form. Runoff may contain dissolved nutrients and/or metals depending on numerous factors associated with dissociation constants of possible constituents. Runoff nutrient and metal concentrations, like in all aqueous solutions, are governed by: mineral crystallinity (i.e., more highly crystalline minerals are less likely to dissolve than poorly crystalline minerals), solution pH (i.e., a measure of hydrogen ions within the system), temperature, solution Eh (i.e., a measure of free electrons within the system), solution species activity (i.e., a measure of species concentration accounting for solution electrolyte concentration), the amount of time solution is in contact with the nutrient or metal
that is dissolving (i.e., some compounds dissolve more quickly than others), and the presence of chelates (i.e., compounds that interact with metals to increase solubility; Sposito, 2008a,b,c).

Runoff may also contain particulate-bound or adsorbed nutrients and metals. In order for a soil particle to be susceptible to movement by runoff, the particle must first be displaced or freed from the soil surface. Factors that influence particle detachment may include: raindrop impact where kinetic energy removes the particle from a stable position, soil texture where smaller particles require water with less kinetic energy to be moved, soil structure where well-structured soils are less likely to degrade, soil organic matter where increased organic matter improves soil structure, soil chemical properties (i.e., sodic or saline conditions), vegetative and/or ground cover interception where raindrop interception decreases raindrop kinetic energy associated with high velocity, and runoff and/or particle laden runoff impact (Hillel, 2004a). Detached soil particles will be moved by runoff when runoff velocity attains enough kinetic energy to move the particle. As moving water’s kinetic energy increases, larger particle sizes may be moved. Plant nutrients and trace metals carried by runoff may eventually enter surface waterways.

Volatilization

Volatilization from soil is the process of mass loss due to transformation of a solid-phase compound to a gaseous phase. For the purpose of this research, volatilization will focus on carbon dioxide (CO$_2$) production and loss from the soil surface. In soil, CO$_2$ is produced by soil macro- and micro-organisms and plant roots as carbon substrates, such as soil organic matter or carbon-based BL, are utilized as an energy source in cellular respiration:

$$C_6H_{12}O_6 + 6O_2 \rightarrow 6CO_2 + 6H_2O + \text{energy}$$

[1-1]
where sucrose is oxidized by oxygen to form CO$_2$, water, and energy. Cellular respiration occurs in the cell mitochondria where energy is captured for a net gain of two adenosine triphosphate (ATP) molecules in the Krebs cycle. The CO$_2$ produced is a by-product and can accumulate within the soil atmosphere at concentrations in excess of 10 times the atmospheric CO$_2$ concentration. The exchange of CO$_2$ at the soil surface between the soil atmosphere and the terrestrial atmosphere is driven by mass flow and diffusion, with the later being the more prominent process. Fick’s law describes gas diffusion in soil:

\[ J_g = -D(\partial c/\partial x) \]  

where \( J_g \) is the diffusive flux of CO$_2$ in this case (\( \mu \text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1} \)), \( D \) is the diffusion coefficient (\( \text{m}^2 \text{ s}^{-1} \)), \( c \) is CO$_2$ concentration (\( \mu \text{mol CO}_2 \text{ m}^{-3} \)), and \( x \) is the distance traveled (m). The concentration gradient is \( \partial c/\partial x \) and is the driving force of diffusion (Hillel, 2004b). The diffusion coefficient in Equation 1-2 is the most difficult variable to measure in situ because it is influenced by dynamic soil properties, mainly soil temperature and soil moisture. It should be noted that soil temperature and moisture are also important environmental factors governing soil microorganism activity, including reproduction and respiration rates.

**Leaching**

Water movement within soil may be quick as with preferential or mass flow or slow as when water interacts with the soil matrix. Mass flow occurs when water moves into and through the soil by way of preferential pathways, such as cracks in a Vertisol before sealing occurs or macro-organisms’ tunnels. Mass flow bypasses the soil matrix’s natural filtrating processes and thus may allow rapid delivery of contaminants to groundwater. On the other hand, when water moves through the soil profile, contaminants may be degraded or consumed by soil microbes or
interact with organic matter or soil particles. Soil solution moving below the root zone is drainage or leachate and will either move back into the root zone in response to matric potential gradients at a later time or enter the groundwater. Broiler litter contaminants in leachate can pollute groundwater. Polluted groundwater may be used for drinking. In the Ozark Highlands, topographic features allow relatively free movement and mixing of surface and groundwater.

**Soil Storage/Adsorption**

A soil’s ability to retain and store ions and compounds can be quite large. Factors that influence soil chemical retention are: parent material, soil depth, cation exchange capacity (CEC), anion exchange capacity (AEC), soil organic matter (SOM), soil pH, soil Eh, soil salinity, and amount of water movement. Ion/compound chemical characteristics, such as solubility, polarity, charge, and charge density, can also influence retention in soil. Broiler litter contains many constituents of varying chemical structures that can interact many ways with soil. For instance, P applied in BL to an alkaline soil may precipitate from soil solution forming calcium-bound P. The same P applied to an acidic soil may precipitate as iron- or aluminum-bound P. In both cases, P may be retained for many decades within a soil. In contrast, N contained in BL may take the form of nitrate, an anion. Nitrate and soil particle surfaces are negatively charged and thus repel one another. Therefore, nitrate moves with soil water and is seldom retained in the soil column.

**Plant Uptake**

Broiler litter contains nutrients and trace metals important to proper plant growth and maturity. Macronutrients, such as N, P, potassium (K), calcium (Ca), magnesium (Mg), and
sulfur (S), as well as micronutrients, such as iron (Fe), manganese (Mn), boron (B), Zn, Cu, and nickel (Ni), are contained in BL (Daigh et al., 2010; Menjoulet et al., 2009; Pirani et al., 2007; Sistani et al., 2006; Sistani et al., 2008). Although plant-available chemical forms of these nutrients and metals may not be readily available in fresh BL, biological and environmental processes can transform BL constituents into plant available ions/compounds to be taken up by plants and potentially removed from the system depending on the management practices being used (i.e., haying and/or grazing).

**Potential Environmental Consequences of Land-applied Litter**

Plant nutrients and trace metals contained in land-applied litter can undergo multiple biological and environmental processes. These processes may be classified into two groups, sinks, which are storage within the soil or organic compounds, and losses (Figure 1-2). Two of the more important losses, with regard to water quality within the Ozark Highlands, are runoff and leaching from soil.

**Water Quality Issues**

Water has many uses. Water quality is defined as the suitability of water for its intended use (Daniels et al., 2010). For example, water containing high concentrations of nitrate could be assessed as unimpaired if the intended use was agricultural irrigation. The same water body could be assessed as impaired if the intended use was for drinking water. Nonpoint source pollution is one form of pollution that impairs water (Daniels et al., 2010), and is today’s most important source of water quality problems (Hirsch et al., 2006). Nonpoint pollution comes from urban, suburban, and agricultural land uses; forest harvesting; mining activities; and the
atmosphere (Hirsch et al., 2006). Nonpoint pollutants include sediments, herbicides, pesticides and nutrients (Daniels et al., 2010).

Aquatic ecosystems, such as streams, lakes or coastal waters, may undergo eutrophication or nutrient enrichment when nutrients that are normally growth-limiting due to their naturally occurring scarcity in the environment become readily available in the aquatic system (Kalff, 2002a; Smith and Schindler, 2009). Phosphorus is the primary limiting nutrient in most water systems (Kalff, 2002a; Smith and Schindler, 2009; Schindler et al., 2008; Stumm and Morgan, 1996). Water-P concentration has been shown to be positively correlated to cyanobacterial biomass (Smith and Schindler, 2009) or chlorophyll-a concentration (Kalff, 2002b) in eutrophic waters. Stumm and Morgan (1996) stated that every P atom contained within a biogenic substance (i.e., a substance produced by a life-related process) entering a lake will be oxidized to form “phosphate(s) of oxidative origin” (i.e., bio-available P). Every P atom in this from will consume 342 oxygen atoms as algal protoplasm is produced or maintained (Stumm and Morgan, 1996). This increased oxygen demand in aquatic ecosystems is responsible for anoxic conditions (Stumm and Morgan, 1996) and may lead to fish kills and decreased water quality. Therefore, appropriate P management is important to maintaining water quality (Smith and Schindler, 2009). In Arkansas, nonpoint source P pollution associated with land-applied organic amendments, such as BL, is managed by the use of the P Index (DeLaune, 2002).

Ozark Highlands

As previously mentioned, the southern portion of the Ozark Highlands is characterized by well-developed karst made possible by the extensive presence of soluble carbonate rock (Scott and Ward, 2002; Brye et al., 2013). As unsaturated (with respect to calcite or dolomite) water
moves through the bedrock, caves are formed as the carbonate rock is dissolved (Fetter, 2001b). Karst spring discharge rates are positively correlated to precipitation within the recharge area (Fetter, 2001c; Graening and Brown, 2003; Stueber and Criss, 2005) indicating that runoff may enter groundwater directly (Fetter, 2001c). Karst springs can also store water between rain events (Fetter, 2001c), thus providing a source of well water. Stueber and Criss (2005) reported a residency time of 100 days in a karst cave in Illinois. Because of the unhindered movement of water between surface and groundwater, anthropogenic activities, including organic amendments to soil, may contaminate water resources (MacDonald et al., 1976, Scott and Ward, 2002; Fetter, 2001c; Graening and Brown, 2003; Stueber and Criss, 2005). Simard et al. (2000) stated that man-made and natural preferential flow pathways may be important mechanisms for P movement within and through soil during storm events following dry periods.

In Cave Springs Cave in Cave Springs, Benton County, Arkansas, Graening and Brown (2003) reported that the primary source of energy input into a phreatic cave [i.e., a cave where the bedding plane dips below the water table (Fetter, 2001b)] within the Ozark Highlands was from the cave’s stream ecosystem influx of dissolved organic matter. Stable carbon isotope comparisons between biota and possible biota food sources suggested that cavefish (*Amblyopsis roae*) and stygobitic isopods (*Caecidotea stiladacyyla*) primarily consumed sewage-derived organic matter. Graening and Brown (2003) reported the cave to be contaminated with excess nutrients, metals, and fecal bacteria whose concentrations peaked with storm events and summer growing seasons. Water concentrations of beryllium, Cu, Pb, Se and Zn were reported to exceed Arkansas state limits for chronic and acute toxicity to aquatic life (Graening and Brown, 2003).
Population Growth and Drinking Water

The Ozark Highlands is an area with increasing population. In northwest Arkansas, populations in Benton and Washington Counties increased 127 and 79%, respectively, between 1990 and 2010 (USCB, 2013a, b, c). Population growth increases demand for water resources, primarily with regard to drinking and recreational uses. In order to maintain high water quality within the Ozark Highlands, the effects of land-applied BL on surface runoff and subsurface leaching must be studied over a long period of time. Sustained population growth within northwest Arkansas mandates management practices that preserve the present natural resource base.

Litter-derived Nutrient and Metal Studies

Runoff Studies

The study of runoff nutrient concentrations and loads (or losses) is important because runoff normally empties into surface waters where water quality may be adversely affected by excess nutrient additions. The effect of BL application on runoff nutrient concentrations and losses has been widely explored within laboratory (Adeli et al., 2006; Vadas et al., 2004; Kleinman et al., 2002) and field studies (DeLaune et al., 2004; DeLaune, 2002; Pote et al., 1999; Moore et al., 1998; Edwards et al., 1997; Nichols et al., 1994; Edwards and Daniel, 1993) utilizing simulated rainfall. Such studies are important because researchers are able to manipulate dependent variables like rainfall intensity, duration, and frequency, which are difficult to control within natural environments. However, simulated rainfall studies often report rainfall rates that are infrequent (i.e., storm events with return rates greater than 5 years) in nature. Watershed studies (Sharpley et al., 2008; Sharpley et al., 1992) support the importance
of storm events on runoff, but few studies have monitored runoff from controlled small plots during naturally occurring rainfall (Menjoulet et al., 2009; Sistani et al., 2006; Wood et al., 1999) over extended periods of time.

Important results of simulated-rainfall-induced runoff studies include runoff concentrations and losses of BL constituents increasing with increased BL application rates (DeLaune et al., 2004; Moore et al., 1998; Edwards and Daniel, 1993). Edwards and Daniel (1993) reported linear relationships relating increased runoff concentrations and losses of total N, ammonium (NH$_4$-N), nitrate (NO$_3$-N), total P, dissolved-reactive-P (DRP), total-suspended solids and chemical oxygen demand (COD) to increased BL application rate. It should be noted that many researchers use similar terms to DRP, including soluble-reactive-P, to describe plant-available P, H$_2$PO$_4^-$ and HPO$_4^{2-}$, and is operationally defined by the method of determination, namely filtration with a 0.45-µm pore diameter filter and colorimetric determination, and will hereafter be referred to collectively as DRP. DeLaune et al. (2004) reported similar findings for DRP. Moore et al. (1998) reported similar BL effects for trace metal (i.e., As, Cu, Fe, K, Na, and Zn) concentrations within runoff.

Edwards and Daniel (1993) also reported a dilution effect pertaining to rainfall intensity and runoff. Runoff concentrations of total N, total P, DRP, and COD decreased as rainfall intensity increased, but losses increased. The increased runoff thus diluted concentrations, but carried more mass (Edwards and Daniel, 1993). Vadas et al. (2004) reported decreasing runoff dissolved-inorganic P concentrations as time progressed during the first 30 min of simulated runoff for three animal manures, including BL.

Adeli et al. (2006) reported that as the amount time between BL application and the runoff event increased total losses of N, NH$_4$-N, NO$_3$-N, and DRP in runoff decreased. Adeli et
al. (2006) also reported runoff concentrations of N, P, K, Ca, Mg, Cu and Zn decreased with increased number of runoff events. DeLaune et al. (2004) reported similar results for DRP and increased successive rainfall.

Broiler litter treated with aluminum sulfate prior to soil application has been reported to improve runoff quality during simulated (Moore et al., 1998; DeLaune et al., 2004) and naturally occurring (Sistani et al., 2006) rainfall. Additions of aluminum sulfate decreased runoff concentrations of As, Cu, Fe, Na (Moore et al., 1998), DRP (DeLaune et al., 2004; Sistani et al., 2006), NH₄-N, total P, and particulate P (Sistani et al., 2006). Shallow (i.e., 2-3 cm depth) incorporation of BL has been reported to not affect runoff quality (Nichols et al., 1994).

Results of studies exploring naturally occurring runoff from watersheds support results of simulation studies. In a study monitoring streams and discharges from karst aquifers in Illinois, Stueber and Criss (2005) reported increased concentrations of atrazine and K from agricultural land during high-flow conditions. Similarly, Sharpley et al. (2008) reported increased concentrations of total P and particulate P in stream water as storm size or storm return rate increased in a 39.5-ha watershed in Pennsylvania. Sharpley et al. (2008) suggested that as storm size increases, the area within the watershed contributing to runoff-related stream flow may increase.

Soil-test P has been reported to be positively correlated to runoff DRP (DeLaune et al., 2004; Pote et al., 1999). Pote et al. (1999) reported that water-extractable-P concentrations in the top 2 cm of the soil were positively correlated to DRP concentrations and loads in runoff. The relationships reported were different for different soil series, but normalization of the data by dividing DRP runoff concentrations by the volume of runoff prior to regression analyses resulted in soil-specific regression lines that did not differ from one another. Pote et al. (1999) suggested
that normalization of data corrected for site-specific hydrology differences associated with different soil series resulted in a more universally applicable relationship between water-extractable-P and runoff DRP concentrations and loads. Similarly, DeLaune et al. (2004) reported a positive correlation between soil Mehlich-III extractable-P concentration and DRP concentration in runoff. DeLaune et al. (2004) reported no correlation once BL was applied to the soil because the DRP contained in the BL overwhelmed the relationship.

Wood et al. (1999) conducted a 2-yr study of BL application rate at two levels, 9 and 18 Mg BL ha\(^{-1}\), and commercial fertilizer effects on seasonal runoff quality in Alabama in response to naturally occurring rainfall. Corn with winter rye (Secale cereal L.) was grown on a silty-clay soil with a 4% slope. Increased BL application rate was reported to increase flow-weighted-mean (FWM) runoff concentrations of NO\(_3\)-N and NH\(_4\)-N during the second season of corn production and NO\(_3\)-N during the second season of rye production (Wood et al., 1999). Wood et al. (1999) attributed the increased FWM runoff concentration of NO\(_3\)-N to be from increased NO\(_3\)-N contained within BL used during the second season, where the BL NO\(_3\)-N concentration increased from 12 to 363 mg NO\(_3\)-N kg\(^{-1}\) for year one to two, respectively. Total-P and DRP runoff concentrations and losses were reported to increase with increasing BL application rate during the second season of corn production as well (Wood et al., 1999). Wood et al. (1999) attributed these results to increased BL concentrations of P during the second year. Seasonal FWM runoff concentrations of Ca, K, Mg and Mn were reported to be greater in the 18 Mg BL ha\(^{-1}\) application rate treatment under corn production in either one or both corn seasons (Wood et al., 1999). In addition, seasonal runoff losses of K, Mg and Mn were reported to increase with increased BL application rate during corn production in either the first or second year of production, but results were not consistent for both years (Wood et al., 1999). Wood et al.
(1999) concluded by reporting FWM runoff concentrations of N, P, K, Mg, Mn, Cu and Zn were at levels that were adequate for algal growth in natural water bodies.

**Leaching Studies**

In general, any compound or ion that is soluble in water has the potential to be leached from soil as soil solution moves in response to matric and gravitational forces. Many compounds/ions contained in soil solution are thought to interact with the soil matrix as they move within soil and undergo numerous biological and chemical reactions that help protect groundwater from contamination. The following section will discuss techniques to collect soil solution or leachate followed by a section reviewing related studies pertaining to leaching.

**Leachate Collection Techniques**

Studies measuring solute and water fluxes through soil may use one of many techniques for leachate collection, each with its own merits and limitations. On one end of the spectrum, laboratory studies using undisturbed or artificially packed soil columns allow researchers maximum control of variables at relatively low cost. Soil column studies may be short in duration, ranging from a few hours to a few days. Variability of soil physical properties governing water movement may be high in some soils so many replications are important. Most recently, the use of monoliths, which are large soil columns with diameters > 0.3 m, have been used to more closely resemble natural soil and reduce spatial variability issues, but with increased expense. Soil columns are ideal for monitoring movements of environmentally hazardous materials, such as radioactive tracers, or for applying an annual amount of simulated rainfall within a short period of time.
Moving along the spectrum, studies using porous ceramic cup soil solution samplers may collect soil leachate if placed below the root zone in a field setting. Porous cup samplers collect soil solution in response to gravity under saturated conditions or may collect soil solution in response to a vacuum applied to the inside of the sampler and thus collect soil solution from saturation to the matric potential equivalent to the amount of vacuum applied. Although relatively inexpensive, porous cup samplers do not provide direct measurements of water and solute fluxes in soil due to the undefined collection area of the porous cup.

Nearing the other end of the spectrum, weighing and pan lysimeters, with better defined surface collection areas, are ideal for estimating water and solute fluxes, but can be expensive to install and maintain. Weighing lysimeters can be large and are many times filled with disturbed soil. As the name implies, weighing lysimeters use a scale to measure soil column changes in mass associated with precipitation and evapotranspiration. Pan lysimeters can be smaller, thus allowing more replications, and may be installed under undisturbed soil. As the name also implies, pan lysimeters use a reservoir for the collection and storage of soil water that passes through a porous plate with defined dimensions at the bottom of a soil column.

Brye et al. (1999) developed equilibrium tension lysimeters (ETL). By installing heat dissipation sensors in bulk soil adjacent to a pan lysimeter, Brye et al. (1999) were able to monitor the soil matric potential and then adjust suction within the lysimeter to reflect current soil conditions. Frequent, approximately twice per week, manual adjustments of ETLs were required to keep lysimeter suction within range of current soil matric potentials. With the addition of an automated control system, Masarik et al. (2004) were able to monitor and adjust lysimeter suction in real time and without manual intervention to more realistically mimic
naturally occurring drainage within undisturbed soil. Masarik et al. (2004) termed the device an automated equilibrium tension lysimeter (AETL).

**Related Leaching Studies**

Once surface applied to soil, BL may be rained upon. Assuming precipitation is not intercepted by plants, rain may react with BL in a manner as to extract soluble BL constituents. Gupta et al. (1997), in a BL toxicity study, extracted BL with deionized water according to the United States Environmental Protection Agency protocol for aquatic toxicity identification evaluations and reported that Cd, Cu, Fe, Pb, and Zn were extracted at concentrations great enough to be toxic to *Ceriodaphnia dubia*, the protocol’s indicator species. Gupta et al. (1997) reported, however, that the extractant was not toxic, possibly because organic matter within the BL may have formed complexes with the metals, thus making them less toxic. Karathanasis et al. (2005) reported that similar BL-extractable biosolids increased Cu, Zn, and Pb mobility through undisturbed subsoil monoliths (i.e., large soil columns). Karathanasis et al. (2005) also reported that monolith leachate metal losses were primarily colloidal-bound, with 61, 66 and 96% of Cu, Zn, and Pb, respectively, being associated with organic matter. Preliminary data of Karathanasis et al. (2005) indicated that Pb had a greater tendency to interact with BL-derived biosolids than with any of the soils used for the study.

Once precipitation has fallen and interacted with BL, the nutrient-laden water may be intercepted by a residue layer (i.e., mulch) above the soil surface where water may be absorbed and then infiltrate into the soil or be evaporated. Water may enter the soil surface directly, be evaporated, or runoff, in which runoff was addressed in the previous section.
Infiltration is the downward entry of water into soil at the soil surface and its subsequent movement and redistribution (Horton, 1933; Rode, 1965). If the precipitation rate is less than the maximum rate at which soil may absorb water, then infiltration is *supply controlled* (Hillel, 2004c). If, however, the rate of precipitation to the soil surface exceeds the infiltration rate for the soil, then the process is *soil controlled* (Hillel, 2004c). Soil controlled processes may be either *surface controlled* if the soil surface is responsible for the limited infiltration rate, or *profile controlled* if a lower soil horizon is responsible for the limited infiltration rate (Hillel, 2004c).

Infiltration is affected by many factors. Increased soil surface slope decreases water infiltration and increases runoff. Soil texture influences infiltration with coarse-textured or sandy soils having greater infiltration rates than fine-textured or clayey soils (Horton, 1933; Hillel, 2004c). Well-structured soils have greater infiltration rates than poorly structured soils. Initial soil moisture also affects infiltration, with initially dry soils generally having greater infiltration rates than initially wetter soils (Horton, 1933; Hillel, 2004c). Ground cover interception decreases raindrop velocity and increases infiltration (Hillel, 2004c). Shrink-swell clays present in soil may crack when dry allowing increased infiltration via preferential flow (Hillel, 2004c) or expand once wetted to decrease infiltration close to zero. The temperature of the water itself determines the water’s viscosity, which also affects infiltration. Warm water infiltrates soil at a greater rate than cold water. Soil profile heterogeneity affects infiltration, with homogeneous soil having greater infiltration rates than layered soils (Hillel, 2004c). Entrapped air within soil decreases infiltration. Soil crusting due to soil aggregate breakdown in saline and/or sodic conditions decreases infiltration (Hillel, 2004c). Soil surface crusting due to
amendments of dairy (Brock et al., 2007) and BL slurries (Adams et al., 1997) have been reported to decrease infiltration.

Once in the soil, water moves in response to gravitational, pressure, osmotic and matric gradients (Hillel, 2004d, e). Solutes move as they are carried by moving water either by mass flow or convection or in response to concentration gradients by diffusion (Hillel, 2004f). As previously mentioned, numerous chemical and biological processes may act on BL-derived constituents in soil solution. These processes may act as sinks (i.e., plant uptake or precipitation to the solid phase) or losses (i.e., volatilization or leaching).

Soil solution that drains from a soil of interest is referred to as leachate. Large leachate fluxes have been reported to occur during the spring in south-central Wisconsin (Brye et al., 2001; Brye et al., 2000) or during the winter in Germany (Harsch et al., 2009; Adams et al., 1994), when increased precipitation coincides with increased leachate fluxes (Harsch et al., 2009; Andraski et al., 2000; Adams et al., 1994) and decreased transpiration rates. In northwest Arkansas, the greatest fluxes of leachate have been reported in either the spring (Pirani et al., 2006) or winter (Pirani et al., 2006; Adams et al., 1994) depending on the year. During a 1-yr trace metal leaching study using artificially packed soil columns to explore possible reclamation of mine tailings, Zhu et al. (1999) reported that fescue reduced the leachate flux compared to bluestem (Andropogon gerardii) and control (i.e., no vegetation) columns. Zhu et al. (1999) also reported that Cd and Zn leachate concentrations were low during August and September, but increased in early October through late May for columns with growing vegetation. Leachate Pb concentrations were reported to be unaffected by the presence of vegetation (Zhu et al., 1999). Zhu et al. (1999) also reported that the presence of subsoil limited Cd and Zn leaching and helped to reduce vegetation-induced leaching of Cd and Zn in October through May. Topsoil
over mine tailings was reported to increase the mobility of Zn and Cd from mine tailings, but did not affect Pb (Zhu et al., 1999).

During a long-term leachate study in Germany, Harsch et al. (2009) reported that the leachate flux at a depth of 3.5 m from a grassland was greater than that from mature deciduous and coniferous forests. During a 40-yr observational study, three ecosystems were monitored as they matured (Harsch et al., 2009). Initially, similar leachate fluxes were reported for all ecosystems due to trees being small, but during the third year, as the coniferous trees matured more quickly, their water demands increased, which caused the leachate flux to decrease (Harsch et al., 2009). During the tenth year, the slower-growing deciduous trees deviated from the grassland leachate flux trend and started to decline (Harsch et al., 2009). By the 24th year of the study, the grassland leachate flux continued to be large, while the evergreen and hardwood forests were again equivalent to one another, but now at a lower flux than the grassland (Harsch et al., 2009). Harsch et al. (2009) reported that the 40-yr mean amount of leachate was 53, 37 and 26% of the received precipitation for the grassland, deciduous and coniferous ecosystems, respectively. Similar studies have been conducted in prairie and agricultural ecosystems (Brye and Norman, 2004; Brye et al., 2001, 2000, and 1999; Andraski et al., 2000).

During a 2.5-yr study using an ETL to collect leachate from a natural prairie ecosystem and two inorganically fertilized (189.1 kg N ha\(^{-1}\) yr\(^{-1}\)) agricultural ecosystems (i.e., no-tillage and chisel-plowed corn), Brye et al. (1999) reported drainage as 11, 31 and 44% of received precipitation for the prairie, no-tillage and chisel-plowed ecosystems, respectively. Brye et al. (2001) reported slightly increased drainage for a 4-yr period to be 16, 33 and 47% of received precipitation for the same prairie, no-tillage, and chisel-plowed ecosystems, respectively. Brye et al. (2000) reported decreased infiltration of precipitation in the prairie ecosystem due to a
residue layer on the soil surface. Brye et al. (2000) also demonstrated a relationship between the time since last cultivation and the amount of annual leachate, expressed as a fraction of the total annual precipitation. Brye et al. (2000) reported that as time increased from the last cultivation, annual drainage decreased, but not because of increased evapotranspiration demand. Brye et al. (2000) stated that land use in south-central WI had changed soil hydraulic properties and suggested that agricultural-related soil disturbances, such as chisel plowing, promoted extra drainage. Brye et al. (2000) suggested that drainage rates observed in the chisel-plowed corn agroecosystem could represent potential leaching losses of nitrate-N and pesticides during the spring season if above-normal precipitation occurred.

Nitrate-N is one of the most prominent groundwater pollutants originating from soil (Hillel, 2004g). Application of inorganic fertilizer to soil has been reported to increase NO$_3$-N concentrations (Brye and Norman, 2004; Domínguez et al., 2004; Brye et al., 2001; Andraski et al., 2000) and loads (Brye and Norman, 2004) in soil leachate. In a related ETL study, Brye et al. (2001) evaluated the denitrification potential of three ecosystems: a prairie restoration and a no-tillage and chisel-plowed, fertilized, continuous corn agroecosystem. Brye et al. (2001) reported that cumulative total dissolved carbon losses in leachate were greater for the agricultural ecosystems than the prairie ecosystem. However, dissolved organic carbon (DOC) leaching losses were similar between the two agricultural ecosystems (Brye et al., 2001). Brye et al. (2001) concluded that denitrification was limited by DOC supply in the agroecosystems and by NO$_3$-N supply in the prairie. Brye et al. (2001) stated that the prairie ecosystem was more efficient with N cycling than the agricultural systems.

In addition, Brye and Norman (2004) reported no difference between ecosystems for volume of leachate or cumulative leaching losses of B, Ca, Cl, K, Mg, Mn, P, Na, S, Zn, NH$_4$-N,
NO$_3$-N, and DOC. However, FWM leachate concentrations and cumulative losses of Al and Fe were greater from the prairie ecosystem than either the no-tillage or chisel-plowed agroecosystem (Brye and Norman, 2004). Brye and Norman (2004) reported increased FWM leachate concentrations and leaching losses of B, P, and NO$_3$-N in optimally N-fertilized (180 kg N ha$^{-1}$ yr$^{-1}$) corn compared to N-deficient (10 kg N ha$^{-1}$ yr$^{-1}$) corn. Nitrate-N concentrations in leachate have also been reported to be greater for inorganic than organic fertilizer sources in agricultural systems planted to corn (Dominguez et al., 2004).

In Ohio, zero-tension lysimeters were placed at a depth of 45 cm, just above a fragipan, to evaluate the effects of fertilizer type [ammonium nitrate, cow manure-straw mixture, and legume-rye mixture (Vicia villosa Roth-Secale cereal L.), all applied at a rate of 150 kg N ha$^{-1}$] and earthworm (primarily Lumbricus terrestris L., 44-61% of total; but also Aporrectodea trapezoids Dúges, Aporrectodea tuberculata Eisen and Lumbricus rubellus) population densities (control, increased or decreased) on soil drainage over a 6-yr period (Dominguez et al., 2004). Dominguez et al. (2004) reported increased leachate fluxes from plots with increased earthworm population densities compared to decreased populations. Earthworm population did not affect leachate NO$_3$-N concentrations (Dominguez et al., 2004). Total-N drainage losses were reported to be 33.9 and 13.5 kg N ha$^{-1}$ from increased and decreased earthworm populations, respectively (Dominguez et al., 2004). Dominguez et al. (2004) stated that total-N losses were primarily due to the greater leachate fluxes associated with the increased earthworm population. Simple linear regression indicated that earthworm population density, measured in counts m$^{-2}$, explained 50% of the variability in total-N fluxes at the 45-cm depth (Dominguez et al., 2004). Dominguez et al. (2004) concluded that earthworms had influenced soil surface hydrology and direct visual
observations further suggested that earthworms may have increased preferential flow to the fragipan.

Preferential flow has also been identified as a mechanism of P movement within intact soil columns (Brock et al., 2007; Sinaj et al., 2002; Jensen et al., 1998). Jensen et al. (1998) applied tracers, $^{32}$P isotope (as $H_2^{32}PO_4^-$) and tritium ($^3$H$_2$O), as a one-time pulse of $8.17 \times 10^{-3}$ pore volumes (PV), temporarily replacing influent waters of a saturated soil core at steady state using the constant-head method. Prior to saturation, the soil’s Ap horizon was removed. Later, Brilliant Blue dye was added to stain preferential pathways. The P isotope was first detected in leachate at 0.012 PV and peaked at 0.04 PV (Jensen et al., 1998). Tritium was first detected in leachate at 0.02 PV and peaked at 0.32 PV (Jensen et al., 1998). Total leachate recovery of $^{32}$P and $^3$H$_2$O, were reported to be 1.1 and 88.8%, respectively (Jensen et al., 1998). Other leaching studies have reported minimal P loss in soil leachate (Brock et al., 2007; Sinaj et al., 2002).

Jensen et al. (1998) reported that most of the $^{32}$P was retained in the upper few millimeters of the column. At soil depths greater than 14.8 cm, $^{32}$P was detected at background levels. Exposure of hyperfilm MP, a radioactive sensitive film, to cross-sectional layers of the soil column after leaching suggested that $^{32}$P was adsorbed to biopore (i.e., a tubular macropore reported as mainly earthworm burrows), walls with diameters greater than 3 mm (Jensen et al., 1998). In support of this report, in a study comparing preferential pathways to matrix soil flow, Sinaj et al. (2002) reported increased oxalate-extractable-P concentrations in preferential pathways. Jensen et al. (1998) reported complete or total Brilliant Blue staining to a depth of 2 cm (Jensen et al., 1998). Jensen et al. (1998) also reported that in the upper half of the column, staining occurred in and around biopores. Below 10 cm, only 10-40% of biopores were stained (Jensen et al., 1998).
Preferential flow as a mechanism of P transport into soil has also been studied in soils with histories of liquid dairy manure and BL application (Brock et al., 2007). Brock et al. (2007) applied dairy manure to soil columns at field capacity and then monitored soil leachate. Leaching losses of DRP were increased with the addition of manure in all soils and FWM DRP concentrations increased 1.5 to 10.5 times the initially observed leachate concentrations (Brock et al., 2007). Brock et al. (2007) reported that P losses were not related to soil-P-saturation levels of surface or sub-surface soil except for one field where more than 40 years of BL application had resulted in excessive P accumulation (2400 mg P kg\(^{-1}\)) in the soil to a depth of 50 cm. Brock et al. (2007) suggested that subsoil P saturation may increase P losses in soil leachate.

Nitrate leaching with the addition of BL to soil has also been reported. Application of BL was reported to increase NO\(_3\)-N leachate concentrations (Adams et al., 1994). Adams et al. (1994) applied BL at three rates (0, 10 and 20 Mg ha\(^{-1}\)) and broiler manure (17.7 Mg ha\(^{-1}\), adjusted to equal the same N application rate for the high BL treatment) to a Captina silt loam in northwest Arkansas and then used pan lysimeters and suction cup samplers to collect leachate using a suction of -8 to -12 kPa at 60 and 120-cm depths. Leachate NO\(_3\)-N concentrations at the 60-cm depth peaked at concentrations > 10 mg L\(^{-1}\) about 30 days post-BL-application, then decreased to control levels during a dry period for all non-control treatments (Adams et al., 1994). Nitrate concentrations peaked a second time at 54 and 41 mg L\(^{-1}\) for high treatments of BL and manure, respectively, at 70 to 90 days post-treatment at (Adams et al., 1994). The low BL treatment peaked a second time at 13 mg L\(^{-1}\) at 140 to 150 days post-BL application (Adams et al., 1994). Leachate NO\(_3\)-N concentrations did not differ between the 20 Mg BL ha\(^{-1}\) and 17.7 Mg manure ha\(^{-1}\) treatments. The 20 Mg BL ha\(^{-1}\) and 17.7 Mg manure ha\(^{-1}\) treatments had increased leachate NO\(_3\)-N concentrations compared to the low BL treatment of 10 Mg BL ha\(^{-1}\).
Adams et al. (1994) reported NO$_3$-N losses of 1.7, 2.9 and 2.2% of total N applied during the first 60 days post-application at a depth of 60-cm for the 10 and 20 Mg BL ha$^{-1}$ and 17.7 Mg manure ha$^{-1}$ treatments, respectively. Adams et al. (1994) reported NO$_3$-N leachate concentrations at the 120-cm depth to peak approximately 120 days post-treatment for all conditions, with the low BL rate being below the drinking water standard of 10 mg L$^{-1}$ and the high application rates exceeding 10 mg L$^{-1}$.

As previously discussed, BL-extracted biosolid chelates have been reported to increase trace metal (Cu, Pb and Zn) leaching from soil (Karathanasis et al., 2005). Other trace metal leaching studies have used radioactive isotope tracers (Jones and Belling, 1967), lead-arsenate-contaminated orchard soil (Peryea and Kammereck, 1997), and metal-spiked sewage sludge (McLaren et al., 2004) to document trace metal leaching from soil. Chromium, Cu, and Pb have been reported to be retained by soil (Karathanasis et al., 2005; McLaren et al., 2004; Peryea and Kammereck, 1997; Jones and Belling, 1967). McLaren et al. (2004) reported increased leachate concentrations of Cd, Ni, and Zn with the application of metal-spiked sewage sludge. Although the amounts of Cd, Ni, and Zn leached were small relative to the amounts added, McLaren et al. (2004) reported that leachate concentrations of Ni and Zn exceeded New Zealand’s environmental drinking standards. Karathanasis et al. (2005) reported the retention of Zn within soil to be equivalent to that of Cu. An isotope tracer study by Jones and Belling (1967) reported that Cu was retained at the surface in soils with exchange capacities greater than 2 meq (100 g soil)$^{-1}$ and still remained within 3 cm of the soil surface when exchange capacities were less than 2 meq (100 g soil)$^{-1}$. Jones and Belling (1967) also reported that calcareous soils retained Se, primarily at or near the soil surface. Jones and Belling (1967) reported the greatest metal movement within soil to occur with Mo in high pH soils. Peryea and Kammereck (1997)
reported that the addition of monoammonium phosphate to lead-arsenate-contaminated topsoil increased As leaching into and through the subsoil, while Pb did not move or leach.

**Soil Respiration Studies**

Since BL application to pasture soil has occurred for extended periods of time in the Ozark Highlands and other regions where intense broiler production occurs, and since continued applications of BL will likely continue to occur in these regions, and since organic amendments to soil have been shown to increase CO$_2$ release to the atmosphere, which increases radiative forcing, it is imperative to study the effects of long-term BL application rate effects on soil respiration in non-cultivated soil. Furthermore, prediction of soil respiration based on measurable environmental factors is needed for larger-scale models that are used to predict global climate changes.

Anthropogenic activities increase CO$_2$ release to the atmosphere, which in turn can increase global warming (Forster et al., 2007) and promote climate change (Trenberth et al., 2007). Soil surface CO$_2$ flux, also referred to as soil respiration, is influenced by agricultural management practices (Risch and Frank, 2010; Roberson et al., 2008; Brye et al., 2006b; Al-Kaisi and Yin, 2005; Yamulki and Jarvis, 2002; Wagai et al., 1998; Linn and Doran, 1984) and is temporally (Ding et al., 2010; Pingintha et al., 2010; Risch and Frank, 2010; Ruehr et al., 2010; Brown et al., 2009; Brye and Riley, 2009; Jones et al., 2006; Davidson et al., 1998) and spatially (Aiken et al., 1991) variable. Soil moisture (Brown et al., 2009) and temperature (Reth et al., 2009; Brye et al., 2006b; Fierer et al., 2006; Fang and Moncrieff, 2001) have also been shown to influence CO$_2$ flux from soil. Additions of mineral fertilizers (Ding et al., 2010) and animal manures (Jones et al., 2005), including turkey (*Meleagris gallopavo*; Penghamkeerati et al.,
have been reported to increase CO$_2$ flux. In general, additions of animal wastes result in increased carbon sequestration (i.e., carbon storage) within soil (Roberson et al., 2008; Jones et al., 2006; Jones et al., 2005; Adams et al., 1997). Researchers have attempted to model CO$_2$ flux (Pinginththa et al., 2010; Brye et al., 2006b; Šimůnek and Suarez, 1993), but few, if any, have attempted to account for changes in CO$_2$ flux in response to varying BL amendment rates.

Soil moisture has been reported to be positively correlated (Pinginththa et al., 2010; Brown et al., 2009), negatively correlated (Brye et al., 2006b; Jones et al., 2006), and uncorrelated (Ding et al., 2010; Brye et al., 2006a; Al-Kaisi and Yin, 2005) to observed soil surface CO$_2$ fluxes. Brye and Riley (2009) demonstrated a quadratic relationship between water-filled pore space (WFPS) and soil respiration with respiration being greatest at 50% WFPS. Jones et al. (2006) reported two linear regressions relating soil surface CO$_2$ flux and soil moisture level. A positively correlated ($R^2 = 0.30$) regression line was used for volumetric water contents between 0.1 and 0.3 m$^3$ m$^{-3}$, while a negatively correlated ($R^2 = 0.12$) regression line was used for volumetric water contents greater than 0.3 m$^3$ m$^{-3}$ (Jones et al., 2006). In general, soil microbial activities, including respiration, have an optimal soil moisture range. If soil moisture conditions are too dry, soil microbes may slow respiration or even die, thus decreasing soil surface CO$_2$ flux. On the other hand, if soil moisture conditions are too wet, soil microbes may be forced to utilize other terminal electron receivers due to the lack of oxygen or die, thus again resulting in reduced soil surface CO$_2$ flux. In addition, as the soil water content approaches saturation, gas movement (i.e., diffusion) within a porous media becomes limited (refer to diffusion coefficient term in Equation 1-2).
Soil temperature has been reported to be positively correlated to soil surface CO$_2$ flux (Ding et al., 2010; Ruehr et al., 2010; Brown et al., 2009; Brye et al., 2006a; Jones et al., 2006; Fang and Moncrieff, 2001) and is variable by season (Ruehr et al., 2010; Brown et al., 2009) and time of day (Ruehr et al., 2010). Diurnal hysteresis effects, similar to wetting and drying curves within soil, have been observed (Pingintha et al., 2010; Ruehr et al., 2010) and differ by time of year (Ruehr et al., 2010) and soil depth (Pingintha et al., 2010). Many nonlinear relationships relating soil respiration and soil temperature have been proposed and are reviewed by Fang and Moncrieff (2001). More recently, exponential relationships have been used to explain variations in CO$_2$ fluxes with soil temperature variations (Ruehr et al., 2010; Brown et al., 2009; Jones et al., 2006).

Additions of BL to soil have increased soil surface CO$_2$ flux (Roberson et al., 2008; Jones et al., 2006; Jones et al., 2005; Adams et al., 1997). Adams et al. (1997) explored the feasibility of using BL slurry to enhance closed-crop canopies with elevated CO$_2$ concentrations to increase photosynthesis. Five BL slurry treatments and a control receiving de-ionized water were imposed on artificially packed soil columns. Adams et al. (1997) reported a 3- to 8.5-hr lag time between when BL slurry was added and when elevated respiration commenced, although no statistical support was supplied. Lag time increased as successive applications of slurry increased, possibly due to a soil-crusting effect (Adams et al., 1997). Stored litter slurry inoculated with fresh BL slurry aged for seven days had the shortest lag time of 3 hr with the greatest CO$_2$ flux and the greatest total amount of CO$_2$ released when compared to all other treatments.

In Scotland, CO$_2$ production from a grassland was monitored in response to soil surface amendments of three organic manures, BL, cattle slurry and sewage sludge pellets, and two
inorganic fertilizers, urea and a compound containing ammonium nitrate, on a sandy-clay-loam soil (Jones et al., 2006; Jones et al., 2005). During the 3-yr study, soil surface CO$_2$ flux was reported to be similar for inorganic fertilizers and the unamended control (Jones et al., 2006; Jones et al., 2005). During the first two years, cattle slurry and BL were reported to have greater cumulative soil respiration compared to control plots (Jones et al., 2005). Sewage sludge pellets were reported to have increased cumulative soil respiration, but only during the second year of the study (Jones et al., 2005). During the third year, cumulative soil respiration was similar for all amendments and the control (Jones et al., 2006). During the first two years, the greatest CO$_2$ fluxes from BL and sludge pellets occurred in July, within a month of treatment application (Jones et al., 2005). These results suggest that BL amendment effects on soil surface CO$_2$ flux: 1) may be variable from year to year, 2) may peak later than the hourly scale suggested by Adams et al. (1997) and/or, 3) may initially peak on an hourly scale, as reported by Adams et al. (1997), and then later peak a second time on a weekly or monthly scale.

Brye et al. (2006a) evaluated the effects of BL type (fresh and pelletized) and application rate (five levels) on soil surface CO$_2$ flux from two silt-loam soils used for rice production in eastern Arkansas. No difference in CO$_2$ flux was reported for BL type (Brye et al., 2006a). Soil surface CO$_2$ flux differed by location with greater fluxes being associated with greater soil temperatures at the location sampled later in the day (Brye et al., 2006a). Brye et al. (2006a) reported that BL rate had no effect on CO$_2$ flux, except for the first sample date at 15 d post-litter application where increased BL application rate appeared to increase CO$_2$ flux based on visual interpretation of presented data. Brye et al. (2006a) suggested the one-time BL application rate effect could have been related to tillage. Tillage effects on soil respiration have been reported for time periods between 12 and 20 d post tillage (Al-Kaisi and Yin, 2005). Brye et al. (2006a)
also reported that, when soil temperatures at the 2.5 and 10 cm depths and soil volumetric water content from the 0- to 6-cm depth interval were included in a multiple regression model, the whole model was non-significant.

Brye and Riley (2009) evaluated the effects of prairie restoration age on near-surface soil properties, including CO\textsubscript{2} flux, within the Ozark Highlands in Benton County, Arkansas. Important results demonstrated a temporal effect on CO\textsubscript{2} flux, although CO\textsubscript{2} flux did not vary by location (i.e., restoration age of prairie or native prairie). Soil surface CO\textsubscript{2} flux was reported to be positively linearly correlated to soil temperature at two soil depths (2 and 10 cm), but unrelated to soil WFPS or soil volumetric water content (VWC; Brye and Riley, 2009). A quadratic model predicting soil surface CO\textsubscript{2} flux from WFPS was developed ($P < 0.001; R^2 = 0.11$) and predicted maximum CO\textsubscript{2} fluxes between 40 and 60 \% WFPS (Brye and Riley, 2009). Multiple regressions using soil temperature at the 2-cm depth and the linear and quadratic soil moisture terms explained 19\% of the observed variability in soil surface CO\textsubscript{2} flux (Brye and Riley, 2009).

**Important Results from Various Aspects of the Original Study**

As was alluded to in the Introduction, a study was initiated in Summer 2002 at the University of Arkansas’ Agricultural Research and Extension Center in Fayetteville to evaluate the effects of BL application rate on the fate and transport of BL-derived plant nutrients and trace metals. Pirani (2005) summarized the first two years (i.e., 2003 and 2004) of BL application rate effects on seasonal and annual plant nutrient and trace metal leaching from the root zone of tall fescue. Important results from the initial 2-yr plant nutrient leaching study indicated no consistent BL treatment effects. However, FWM concentrations of P, Ca, Mg, Na and
ammonium (NH$_4$-N) were reported to differ by BL treatment either seasonally and/or annually at some point during the study. In the first year (i.e., May 2003 to April 2004), FWM concentrations of P in soil leachate were twice as large in the low litter treatment (5.6 Mg ha$^{-1}$) than the control and high-litter treatments (0 and 11.6 Mg ha$^{-1}$, respectively) in the spring and for the entire year (Pirani et al., 2007). However, the observed BL rate effect in the spring was great enough to influence the annual P leaching trend for the entire first year of the study. This suggests that leachate P concentrations may occasionally increase due to BL application rate and coupled with the underlying karst development in the region, may impair groundwater quality.

Important results from the initial 2-yr trace metal leaching study included seasonal BL rate effects on FWM leachate Mn, chromium (Cr), and Fe concentrations as well as annual effects for FWM Ni and Cu concentrations (Pirani et al., 2006). Seasonal mass losses were observed in 2004 for As, Fe, and Zn. Overall results, however, were not consistent and Pirani et al. (2006) concluded that the initial two years of BL application to soil with a history of BL application did not influence trace metal leaching from soil.

Brye and Pirani (2006) evaluated trace metal uptake by tall fescue throughout the growing season following the first BL application. Prior to litter application, Brye and Pirani (2006) reported above-ground dry matter (DM) production to be similar for all plots. Likewise, plant tissue concentrations of Fe, Mg, Mn, Se, Cu, Zn and Al were reported to be similar between treatments, but temporal effects demonstrating a decrease in tissue Cu concentrations and an increase in Zn and Al tissue concentrations were reported prior to initial BL application. After BL treatment, Brye and Pirani (2006) reported increased DM production in plots receiving BL when compared to unamended control plots, but there was no difference between the two BL treatments that received BL. Fescue tissue concentrations of Al, Cu, Fe, Mg, Mn, Se and Zn
were unaffected by BL application rate. The 7.5 month cumulative uptake of Cu, Mg, Mn, Se, and Zn by fescue was shown to increase with BL application rate, and was attributed to increased DM production associated with increased BL rates (Brye and Pirani, 2006).

Menjoulet (2007) summarized the first four years (i.e., 2003 to 2007) of BL application rate effects on seasonal and annual plant nutrient and trace metal runoff from the tall-fescue-dominated pasture soil. Important results from this initial 4-yr runoff study showed a lower than expected 4-yr cumulative runoff, 6 mm, < 0.1 % of cumulative rainfall for the same time period, which was attributed to a moderately high hydraulic conductivity of the soil that allowed greater infiltration than runoff (Menjoulet et al., 2009). The FWM concentration of total dissolved P in runoff was reported to increase with increased BL application rate and FWM runoff concentration of Fe was greater for the high- compared to the low-litter treatment. Menjoulet et al. (2009) also reported that the annual FWM As concentration in runoff exceeded the Environmental Protection Agency’s maximum contamination level for As in drinking water. The 4-year cumulative runoff mass loss of Cd was greater from the low-litter treatment than the unamended control or high-litter treatments. Also, increased BL rate increased Fe cumulative runoff loss. Menjoulet et al. (2009) indicated that non-regulated non-point runoff P concentrations from BL-amended soils may attain concentration levels of the same order of magnitude as regulated point sources. Runoff concentrations and loads of all other plant nutrients (i.e., NO$_3$-N, NH$_4$-N, dissolved-P, DOC, Ca, K, Mg, and Na), runoff concentrations of trace metals (i.e., Cr, Cu, Mn, Ni, Se, and Zn) and runoff loads of trace metals (i.e., Cd, Cr, Cu, Mn, Ni, and Zn) were unaffected by BL application rate (Menjoulet et al., 2009).

Daigh (2009) investigated BL application rate effects on changes in soil storage of plant nutrients and trace metals over a 5-yr period (i.e., 2003 to 2008). Important general soil
chemical property results from this initial soil-storage-change study included an increase in soil organic matter within the 0- to 10-cm depth interval due to increased BL application rate. Soil pH was also observed to increase with increased BL application rate. Daigh et al. (2009) proposed that the observed decreases in dissolved organic carbon within all BL treatments may have been due to losses associated with soil respiration.

Daigh et al. (2009) also explored acid-recoverable (AR), Mehlich III extractable (M3) and water-extractable (WE) nutrient and trace metal content changes. With the exception of Na and Cu, AR nutrient and trace metal content changes were influenced primarily by soil profile depth, rather than BL application rate. Soil depth x BL rate interactions were reported for AR Na and Cu soil content changes over the 5-yr study. Soil AR Cu contents increased with increasing BL application rate in the top 10 cm, but showed no difference at depths from 10 to 90 cm. Soil AR Na contents increased in plots receiving BL and decreased in the control. Acid-recoverable soil P and Ca contents were reported to have increased with the largest BL application rate compared to the low BL treatment and unamended control (Daigh et al., 2009).

Daigh et al. (2009) reported significant differences in M3-extractable soil Na contents as BL application rate increased. The unamended control lost Na, while the BL-treated plots were accumulating M3-extractable Na. Likewise, M3-extractable soil Zn and Cu were reported to increase with BL application, although that from the two litter rates did not differ from one another.

Daigh et al. (2009) also reported that WE soil contents of P, Mg, Fe, and Al increased in the high-litter treatment (11.2 Mg ha\textsuperscript{-1}), but were equivalent for the unamended control and low-litter (5.6 Mg ha\textsuperscript{-1}) treatments. In a similar manner, WE soil Na content increased with increasing BL rate, where WE soil Na content differed among all treatments.
In a separate study, Daigh et al. (2010) evaluated the effects of water extractant, dilution ratio, and extraction time on the chemical properties of BL extracts and observed no difference between rainwater and deionized water as an extractant for BL constituents. Daigh et al. (2010) also reported extractant concentrations generally increased as extraction time increased from 5 minutes to 24 hours. However, DOC concentrations were observed to decrease at extraction times greater than six hours, which was inferred to be due to microbial activity and consumption. Daigh et al. (2010) recommended at least 30 minutes, but less than 6 hours, for extraction times of BL, while no optimal dilution ratio was suggested.

McDonald et al. (2009) conducted a soil sorption study to evaluate the effects of BL application history and soil depth on As sorption characteristics. McDonald et al. (2009) reported that As adsorption decreased as BL application rate increased. Additional results included greater As adsorption deeper in the profile (20-50 cm depth interval) than in surface (0-20 cm depth interval) soil. Also, soil clay content had a greater effect on As adsorption than did soil organic matter concentration. McDonald et al. (2009) reported As adsorption doubled when P was present in the supernatant solution when compared to supernatant solution without P. It was proposed that the supernatant solution Ca$^{2+}$ cation may have precipitated with As and/or P thus removing As and/or P from the solution phase. McDonald et al. (2009) concluded that soil adsorption of As was dependent on BL application history and thus a single adsorption isotherm would not be adequate to explain As adsorption within soil with a history of BL application.

Most recently, a study was conducted to investigate trends in above-ground DM yield following differential BL application (Brye et al., 2010). Brye et al. (2010) reported that although cessation of BL application immediately decreased DM production, yields from the unamended control remained relatively high compared to other forage yields reported by other
investigators. Brye et al. (2010) concluded that soils with a long history of BL applications can continue to maintain adequate DM yields even after six years without BL amendments.

**Justification**

To ascertain the effects of BL application rate on the fate and transport of litter-derived plant nutrients and trace metals, a long-term study needs to be conducted where similar measurements are taken over an extended period of time (i.e., > 5 years) and are summarized annually and cumulatively over the entire study period. This type of long-term study can examine the effect of inter-annual climatic variability, such as changes in rainfall and air temperature from year to year on the fate and transport of BL-derived nutrients and metals. These data should provide scientists, policy makers, and regulators with the best assessment of the long-term effects, or possible risks, of BL application to soil.

**Objectives and Hypotheses**

The objectives of the proposed study are to determine the long-term effects of BL application rate to a silt-loam pasture soil on 1) drainage, soil leachate chemistry, and elemental leaching losses; 2) runoff, runoff chemistry, and elemental runoff losses and; 3) soil respiration and carbon losses. It was hypothesized that continued annual additions of BL would 1) increase mean annual leachate pH and EC, as well as annual FWM leachate concentrations and loads of BL-derived DOC, NO$_3$-N, Ca, Mg, Na, Cd, Cu, Fe, Ni, and Zn relative to an unamended control over the 8-yr study period; 2) have no effect on annual drainage and annual FWM leachate concentrations and loads of BL-derived NH$_4$-N, PO$_4$-P, K, As, Mn, and Se relative to an unamended control over the 8-yr period; 3) increase FWM leachate concentrations and
cumulative loads of DOC, NO₃-N, Ca, Mg, Na, Cd, Cu, Fe, Ni, and Zn relative to an unamended control over the 8-yr period; 4) increase mean annual runoff pH and EC, as well as annual FWM runoff concentrations and loads of BL-derived DOC, PO₄-P, As, Ca, Cd, Cu, Cr, Fe, Na, Ni, P, and Se relative to an unamended control during the 8-yr period; 5) have no effect on annual FWM runoff concentrations and loads of BL-derived NO₃-N, NH₄-N, Mg, Mn, and Zn relative to an unamended control over the 8-yr period; 6) decrease mean annual runoff and annual FWM runoff K concentrations and loads relative to an unamended control over the 8-yr period and; 7) increase surface soil respiration and annual CO₂-C emissions relative to an unamended control over a 3-yr period. Additionally, it was hypothesized that temporal variations in soil respiration would be related to soil moisture and temperature and that these relationships may vary by BL application rate.
References


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Table 1-1. Mean annual broiler litter composition averaged over an 8-yr period and annual mean maxima and minima. Litter was applied to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas.

<table>
<thead>
<tr>
<th>Litter Property</th>
<th>Mean Annual Composition</th>
<th>Mean Maximum</th>
<th>Mean Minimum</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moisture (kg kg⁻¹)</td>
<td>0.24</td>
<td>0.27</td>
<td>0.21</td>
</tr>
<tr>
<td>pH</td>
<td>8.4</td>
<td>8.8</td>
<td>8.0</td>
</tr>
<tr>
<td>EC† (dS m⁻¹)</td>
<td>11.9</td>
<td>14.8</td>
<td>9.8</td>
</tr>
<tr>
<td>NO₃-N (mg kg⁻¹)</td>
<td>207</td>
<td>513</td>
<td>38</td>
</tr>
<tr>
<td>NH₄-N (mg kg⁻¹)</td>
<td>4640</td>
<td>7183</td>
<td>2877</td>
</tr>
</tbody>
</table>

Total Elements

<table>
<thead>
<tr>
<th>Element</th>
<th>Mean Annual Composition</th>
<th>Mean Maximum</th>
<th>Mean Minimum</th>
</tr>
</thead>
<tbody>
<tr>
<td>C (%)</td>
<td>37.1</td>
<td>39.5</td>
<td>33.9</td>
</tr>
<tr>
<td>N (%)</td>
<td>4.4</td>
<td>5.3</td>
<td>4</td>
</tr>
<tr>
<td>P (%)</td>
<td>2.2</td>
<td>2.6</td>
<td>1.6</td>
</tr>
<tr>
<td>K (%)</td>
<td>3.5</td>
<td>4.4</td>
<td>2.9</td>
</tr>
<tr>
<td>Ca (%)</td>
<td>3.7</td>
<td>4.4</td>
<td>2.9</td>
</tr>
<tr>
<td>Mg (%)</td>
<td>0.7</td>
<td>0.8</td>
<td>0.6</td>
</tr>
<tr>
<td>S (%)</td>
<td>1.1</td>
<td>1.6</td>
<td>0.6</td>
</tr>
<tr>
<td>Na (mg kg⁻¹)</td>
<td>9098</td>
<td>16094</td>
<td>3857</td>
</tr>
<tr>
<td>Al (mg kg⁻¹)</td>
<td>347</td>
<td>558</td>
<td>243</td>
</tr>
<tr>
<td>Fe (mg kg⁻¹)</td>
<td>413</td>
<td>613</td>
<td>197</td>
</tr>
<tr>
<td>Mn (mg kg⁻¹)</td>
<td>568</td>
<td>751</td>
<td>421</td>
</tr>
<tr>
<td>Zn (mg kg⁻¹)</td>
<td>510</td>
<td>645</td>
<td>395</td>
</tr>
<tr>
<td>Cu (mg kg⁻¹)</td>
<td>496</td>
<td>678</td>
<td>298</td>
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<tr>
<td>B (mg kg⁻¹)</td>
<td>52.6</td>
<td>60.9</td>
<td>46.5</td>
</tr>
<tr>
<td>Ni (mg kg⁻¹)</td>
<td>10.4</td>
<td>16.1</td>
<td>5.9</td>
</tr>
<tr>
<td>Cd (mg kg⁻¹)</td>
<td>0.19</td>
<td>0.60</td>
<td>0.05</td>
</tr>
<tr>
<td>Cr (mg kg⁻¹)</td>
<td>7.7</td>
<td>15.6</td>
<td>3.1</td>
</tr>
<tr>
<td>As (mg kg⁻¹)</td>
<td>26.8</td>
<td>39.9</td>
<td>19</td>
</tr>
<tr>
<td>Se (mg kg⁻¹)</td>
<td>3.5</td>
<td>7.3</td>
<td>1.6</td>
</tr>
</tbody>
</table>

† EC, electrical conductivity.
Figure 1-1. United States commercial broiler production by year. Broiler production within the United States has steadily increased over the last half of the 20th century. Data were extracted from USDA-NASS (2010).
Figure 1-2. Soil mass balance flow diagram.
Appendix 1-1. Local geology of the study site located at the University of Arkansas Agricultural Research and Extension Center in Fayetteville, Arkansas. Plots were established on the weathered Cane Hill Member of the Hale Formation. The Cane Hill Formation is the lower of two members of the Hale Formation and its lower contact boundary marks the Mississippian-Pennsylvanian boundary in the Ozark Plateaus region. The Cane Hill Member is composed of fine-grained sandstone in the research area (figure modified from AGS, 2014b).

<table>
<thead>
<tr>
<th>Period</th>
<th>Mississippian</th>
<th>Pennsylvania</th>
<th>Hale</th>
<th>Cane Hill</th>
<th>Pitkin</th>
<th>Fayetteville</th>
<th>Batesville</th>
<th>Moorefield</th>
<th>Boone</th>
</tr>
</thead>
</table>


Appendix 1-2. Research area plot diagram and broiler litter application treatments (control, 0; low, 5.6; and high, 11.2 Mg dry litter ha\(^{-1}\)). Taken from Pirani (2005).
Chapter Two

Long-term Drainage and Leachate Water Quality Trends from a
Broiler Litter-amended Udult in the Ozark Highlands
Abstract

Producers in regions with intense broiler (*Gallus gallus*) production take advantage of the plant nutrients contained in broiler waste products like broiler litter (BL) to enhance yields of forage grasses. However, application of BL to pasturelands in karst regions like the Ozark Highlands can potentially reduce water quality due to leaching of BL-derived nutrients and trace metals. The objective of this study was to determine long-term linear trends in drainage and soil leachate water quality under natural precipitation from a Captina silt-loam soil (fine-silty, siliceous, active, mesic Typic Fragiudult) with a history of litter applications under forage management amended annually with BL at three application rates [0 (control), 5.6 (low), and 11.2 (high) Mg BL ha\(^{-1}\)]. Automated equilibrium tension lysimeters were used to continuously monitor and collect leachate from an undisturbed soil profile at a depth of 0.9 m for the 8-yr period from May 2003 through April 2011. Leachate pH, oxidation-reduction potential (ORP), electrical conductivity (EC), and soluble plant nutrients (i.e., NO\(_3\)-N, NH\(_4\)-N, PO\(_4\)-P, Ca, K, Mg, Na, and P), trace metals (i.e., As, Cd, Cr, Cu, Fe, Mn, Ni, Se, and Zn), and dissolved organic carbon (DOC) were measured. Annual flow-weighted-mean (FWM) concentrations and annual loads were determined. Average annual drainage and leachate pH, EC, FWM concentrations, and loads of NO\(_3\)-N, PO\(_4\)-P, Cd, Cr, K, P, Zn, and DOC did not vary over the 8-yr period and were unaffected by BL application rate. Average annual FWM concentrations and loads of NH\(_4\)-N, As, Mn, and Ni decreased, while Cu and Se increased during the 8 years, but were also unaffected by BL rate. Continued annual additions of BL increased average annual FWM leachate Na concentrations relative to the unamended control. Eight-year cumulative leaching loads of NH\(_4\)-N, C, N, P, Mn, and Cu represented less than 2% of that applied in litter treatments, while cumulative leaching loads of NO\(_3\)-N, K, Ca, Na, Mg, Zn, Fe, As, Ni, and Cr represented
between 9 and 99% of that applied in BL. Cumulative leaching loads of Se and Cd exceeded 100% of that applied in BL. Results indicate that pasturelands with a history of BL application may continue to release BL-derived metals, such as As and Se, at concentrations harmful to health regardless of current management practice long after litter application has ceased. Results for Na, a relatively mobile cation in soil, required 8 yrs to identify BL-induced leaching changes, suggesting that less mobile cations, like Ca, will require longer observational periods, perhaps decades, to document BL-induced leaching changes.
Introduction

Nationwide, the United States produced 8.4 billion broiler chickens (*Gallus gallus*) in 2012, with the top five states, Georgia, Alabama, Arkansas, North Carolina, and Mississippi, producing 4.9 billion broilers (USDA-NASS, 2013). Broiler production at this scale generates large quantities of waste in relatively small geographic regions that must be managed. Referred to as broiler litter (BL), the waste is a mixture of excreta, feathers, feed, and bedding material, such as rice (*Oryza sativa* L.) hulls, saw dust, or straw, and is generated at a rate of 1.1 to 1.5 Mg BL per 1000 birds (UADACES, 2002). In 2012, Arkansas produced 1.0 billion broilers and an estimated 1.1 to 1.5 million Mg of BL, the majority of which was concentrated in the Ozark Highlands region (Major Land Resource Area 116A) of northwest Arkansas.

Broiler litter contains numerous plant nutrients, such as nitrogen (N) and phosphorus (P), and is often land applied to pastures in the Ozark Highlands to increase yields of tall fescue (*Lolium arundinaceum* Shreb.) and other forages (Hileman, 1973; Huneycutt et al., 1988; Brye et al., 2010). In addition to water-soluble nutrients, and until July 2011, BL also contained trace metals, such as arsenic (As), cadmium (Cd), copper (Cu), selenium (Se), and zinc (Zn) (Kunkle et al., 1981; van der Watt et al., 1994; Daigh et al., 2010; D’Angelo et al., 2012). Because of economic limitations associated with BL transportation, some pasture soils have received annual BL amendments for many decades, resulting in excessive accumulation of some nutrients and metals in the soil (van der Watt et al., 1994; Daigh et al., 2009). If BL-derived nutrients and metals were to become mobile and subsequently leave the site of application, detrimental environmental effects could occur.

The southern portion of the Ozark Highlands, which encompasses all of northwest Arkansas, is characterized by well-developed karst topography made possible by the extensive
The formation of caves and preferential water pathways in the bedrock allows free exchange of water between surface water and groundwater. For example, soil water may drain into a preferential flow pathway where it moves laterally, hindered only by slope, then discharge as a spring into a losing stream where the process repeats. Because of the unhindered movement of water between surface water and groundwater, anthropogenic activities, including BL amendments to pasturelands, may contaminate surface and groundwater resources (MacDonald et al., 1976; Fetter, 2001; Peterson et al., 2002; Scott and Ward, 2002; Graening and Brown, 2003; Stueber and Criss, 2005; USDA-NRCS, 2006).

The Ozark Highlands, specifically northwest Arkansas, is also an area with increasing population. In northwest Arkansas, populations in Benton and Washington Counties have increased 127 and 79%, respectively, between 1990 and 2010 (USCB, 2013a, b, c). As the population grows, water demands also increase. Domestic drinking water in northwest Arkansas is not supplied from surface water alone; there are many rural residents that use groundwater as their primary source of household and drinking water. Therefore, to maintain high water quality among all sources of water within the Ozark Highlands, where significant amounts of BL are land applied, the effects of land-applied BL on the potential leaching of BL-derived, soluble constituents needs to be studied.

Many previous studies examining the potential leaching of BL-derived nutrients and trace metals have generally been limited in scope or collection techniques, have been rainfall simulations (Brock et al., 2007), have been relatively short in duration (i.e., < 3 yr; Pirani et al., 2006; Pirani et al., 2007; D’Angelo et al., 2012), and have been in response to natural precipitation. Many studies have been limited to only a few BL-derived plant nutrients,
particularly N and P. Nitrogen, in the form of nitrate-N (NO$_3$-N), is important to study because NO$_3$-N is highly mobile in soil, is susceptible to leaching, and at elevated concentrations in groundwater used for drinking is the cause of methemoglobinemia in human infants and ruminant animals, which in rare cases can lead to death. Both inorganic fertilizer (Brye et al., 2001; Agele et al., 2004; Brye and Norman, 2004; Dominquez et al., 2004) and BL amendments (Adams et al., 1994; Agele et al., 2004) to soil have been reported to increase NO$_3$-N leaching. Phosphorous has also been extensively studied because P can contribute to nutrient enrichment or eutrophication of surface water when P-laden overland flow (i.e., runoff) enters waterways. In general, P is considered immobile in soil and has been reported to accumulate in soils amended with animal wastes (Brock et al., 2007), including BL (Daigh et al., 2009). However, in the past few decades, preferential flow of P has been reported (Simard et al., 2000) as well as increased P leaching associated with increased subsoil P-saturation (Brock et al., 2007) suggesting that groundwater quality could be jeopardized in areas with histories of BL application. Other BL-derived nutrients and trace metals have not been as extensively studied as N and P and could potentially pose risks to water quality. Karathanasis et al. (2005) reported that BL-extracted, biosolid-chelates increased Cu and Zn leaching from soil. In addition, trace metals have even been reported to leach from non-BL-amended soil (Jones and Belling, 1967; Peryea and Kammereck, 1997; McLaren et al., 2004).

Soil leachate collection techniques have generally used zero- or fixed-tension to collect soil water from either a defined or undefined collection area. Because soil matric potentials are dynamic, soil water samples collected under zero- or fixed-tension may not accurately represent natural drainage patterns. Additionally, sampling from an undefined collection area does not allow accurate calculation of nutrient and metal loads. However, equilibrium tension lysimeters
(ETLs; Brye et al., 1999) alleviate some of the weaknesses of previous measurement techniques. By installing heat dissipation sensors in bulk soil adjacent to a pan lysimeter with a known collection area, Brye et al. (1999) were able to monitor the bulk-soil matric potential and then adjust suction within a pan lysimeter to reflect fluctuating soil moisture conditions and hydraulic gradients. With the addition of an automated control system, Masarik et al. (2004) were able to monitor and adjust lysimeter suction in real time to more realistically mimic naturally occurring hydraulic gradients and resulting drainage within undisturbed soil. The only studies to use automated ETLs (AETLs) to examine drainage from BL-amended soil focused on BL application rate effects on seasonal and annual plant nutrient (Pirani et al., 2007) and trace metal (Pirani et al., 2006) leaching within the first two years of initiating new BL applications to soil that had high soil test-P originating from historic BL additions.

Since soils with a history of BL amendments continue to receive BL in karst areas, like the Ozark Highlands, it is necessary to identify long-term leaching trends of nutrients and metals to protect surface and groundwater resources. However, only short-term seasonal and annual leaching patterns associated with BL application rate have been documented for nutrients (Pirani et al., 2007) and heavy metals (Pirani et al., 2006) using AETLs, and no study has evaluated long-term (i.e., > 5 years) leaching trends from BL-amended soil in response to natural precipitation using AETLs. Therefore, the objective of this study was to determine long-term linear trends in drainage and soil leachate water quality under natural precipitation from a Captina silt-loam soil (fine-silty, siliceous, active, mesic Typic Fragiudult) under forage management amended annually with BL at three application rates [0 (control), 5.6 (low), and 11.2 (high) Mg BL ha$^{-1}$] after having a history of BL amendments. It was hypothesized that continued annual additions of BL would increase mean annual leachate pH and EC, as well as
annual FWM concentrations and loads of BL-derived DOC, NO₃-N, Ca, Mg, Na, Cd, Cu, Fe, Ni, and Zn relative to an unamended control over the 8-yr study period. Similarly, it was hypothesized that continued annual additions of BL would have no effect on annual drainage and annual FWM concentrations and loads of BL-derived NH₄-N, PO₄-P, K, P, As, Mn, and Se relative to an unamended control. In addition, it was hypothesized that eight years of continued BL application would increase 8-yr FWM concentrations and cumulative loads of DOC, NO₃-N, Ca, Mg, Na, Cd, Cu, Fe, Ni, and Zn relative to an unamended control.

Materials and Methods

Site Description

Research was initiated in 2002 (Pirani et al., 2006) at the Agricultural Research and Extension Center in Fayetteville, Arkansas (36°05’49.18”N 94°10’44.65”W; elevation: 394.7 m). Six plots, 6-m long by 1.5-m wide, were selected on a Captina silt loam (fine-silty, siliceous, active, mesic Typic Fragiudult; USDA-NRCS, 2013), with a 5% west-to-east slope (Pirani et al., 2006). All plots had a history of land-applied BL prior to 2002 and were initially chosen based on similar soil pH [6.2 (standard error = 0.5)] and high Mehlich-3 extractable P [210 (24) mg kg⁻¹] in the top 5 cm (Pirani et al., 2006). Soil particle-size distribution was determined by Pirani (2005) prior to the study’s initiation to a depth of 85 cm with the soil textural class in the 0 to 10 cm depth interval confirmed to be silt loam with 63 % silt and 5.5 % clay. Pirani et al. (2006) also reported increasing clay content with increasing soil depth to 85 cm and a significant textural class change from silt loam to clay loam in the 65 to 85 cm depth interval. Plots had previously been used in runoff studies and were equipped with steel edging to prevent surface water runon as well as to channel runoff from within the plots to aluminum collection gutters.
positioned on the down-slope end of each plot. Initially, ground cover was predominately tall fescue (*Lolium arundinaceum* Shreb.; Pirani, 2005), but in recent years other species have become increasingly common {i.e., clover (*Trifolium spp.*), Johnson grass [*Sorghum halepense* (L.) Pers.] and Bermuda grass (*Cynodon dactylon* L.)}.

The 30-year mean annual air temperature and precipitation in Fayetteville, AR is 13.9 °C and 123 cm, respectively (NOAA, 2013). The average date of the first frost is Oct 17 and the average date of the last frost is April 15 (NOAA, 2013).

**Experimental Design**

The six field plots were arranged in a randomized complete block design with two replications to evaluate BL application rate effects on drainage and soil leachate chemistry. The field treatments in this study included three BL application rates imposed annually as a single application. A control treatment received no annual BL or inorganic fertilizer. A low (5.6 Mg dry litter ha\(^{-1}\)) and high (11.2 Mg dry litter ha\(^{-1}\)) BL rate treatment were established based on the current University of Arkansas Cooperative Extension Service’s litter application recommendations when the study began in 2002 (Pirani et al., 2006). However, BL application recommendations in Arkansas have since changed and are now based on the Phosphorus Index (DeLaune et al., 2004a, b; DeLaune et al., 2006). Despite the change in recommended BL application rate, BL treatment application rates have remained unchanged throughout the study in order to maintain treatment consistency over time.
Lysimeter Installation

Automated, stainless steel, equilibrium tension lysimeters (76.2-cm long by 25.4-cm wide; Masarik et al., 2004) were installed under each of the six plots in late summer 2002 (Pirani et al., 2006). The stainless steel, 0.2-μm, porous collection plates were positioned for a soil interface at a depth of 90 cm under an undisturbed soil column (Pirani et al., 2006). Soil matric potentials were automatically monitored every 10 minutes via heat dissipation sensors (229-L; Campbell Scientific, Logan, UT) placed in the bulk soil at the 90-cm depth. A vacuum pump (TD-2N; Brailsford and Company, Rye, NY) was installed to remove leachate from the soil column in response to the natural fluctuations of the monitored soil matric potentials. Similar to Brye et al. (1999), the vacuum applied to remove leachate was equivalent to 2 kPa less than the measured matric potential in the bulk soil to avoid ponding above the porous plate. Additional information regarding lysimeter installation (Brye et al., 1999; Pirani et al., 2006; and Pirani et al., 2007) and datalogger programming (Masarik et al., 2004) have been previously reported.

Broiler Litter Analyses and Application

Broiler litter was manually applied to plots once annually starting 30 April 2003. Application occurred approximately the first week of May each year. The BL used in this study had been collected from a single chicken house after production of 6 to 8 flocks, had an age ranging from 12 to 18 months, and had bedding material composed of an equal mixture of sawdust and rice hulls (Pirani et al., 2006). Prior to application, BL moisture was determined so dry-weight-equivalent amounts of BL could be calculated for each plot receiving BL. Three BL sub-samples were collected each year prior to application and characterized using procedures for manure analysis (Peters, 2003). Litter pH and EC were determined potentiometrically using a
1:2 BL mass to water volume mixture. Litter NO$_3$-N and NH$_4$-N concentrations were
determined using a Skalar San Plus automated wet chemistry analyzer (Skalar Analytical B.V.,
The Netherlands) after extraction with 2 M potassium chloride. Total C and N were determined
by high-temperature combustion using a LECO CN-2000 analyzer (LECO Corp., St. Josseph,
MI). Total Ca, Cu, Fe, K, Mg, Mn, Na, P, S, and Zn were determined by inductively coupled
argon plasma mass spectrometry (ICP; CIROS CCD model, Spectro Analytical Instruments,
MA) after nitric acid digestion and treatment with hydrogen peroxide. Similarly, ICP was used
to determine total recoverable Al, As, Cd, Cr, Ni, and Se after digestion with nitric and
hydrochloric acid, hydrogen peroxide, and heat (USEPA, 1996).

**Leachate Collection and Analyses**

Beginning immediately after the 2003 BL application, leachate solution was collected
from lysimeters using a separate vacuum pump approximately every two weeks during dry
periods or more frequently as needed. The volume of leachate, pH, EC, and oxidation-reduction
potential (ORP) were measured after collection. Leachate sub-samples were filtered using a 1.6-
µm glass microfiber filter. Once filtered, three, 20-mL aliquots were acidified and three aliquots
were left unacidified. Samples were then stored at 4 °C until chemical analyses could be
performed.

Total dissolved As, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Se and Zn concentrations
were determined by ICP on acidified aliquots. Total DOC was determined using a Shimadzu
Total Organic Carbon Analyzer (Model TOC-CSH, Shimadzu Scientific Instruments, Columbia,
MD) on unacidified aliquots. Acidified aliquots were also used to determine ammonium-N
(NH$_4$-N) and PO$_4$-P and unacidified aliquots were used to determine NO$_3$-N concentrations using a Skalar San Plus automated wet chemistry analyzer.

**Plot Management**

Plots were regularly monitored and maintained since 2002. Above-ground biomass was removed using a bagging push mower eight times in 2003 and 2004 and four times (i.e., first week of May, June, July, and September) annually thereafter to a height of 9 cm as to mimic hay harvesting. Prior to mowing each year, two randomly selected, 0.25-m$^2$ subsamples were hand collected and combined from each plot. Samples were dried at 55 °C for 5 d in a forced-air drier and weighed for dry matter (DM) determination. Above-ground DM production was summed annually.

Precipitation was also monitored by two on-site rain gauges. A simple funnel-reservoir system collected precipitation and a micrometeorological weather station monitored wind speed, air temperature, relative humidity, total solar radiation, photosynthetically active radiation, and rainfall via a tipping bucket every 30 min.

**Calculations**

Flow-weighted mean leachate concentrations (mg L$^{-1}$) and leachate loads (kg ha$^{-1}$) were determined annually and cumulatively for the entire 8-yr period for each dissolved soil leachate constituent measured. For the purpose of this study, a year was designated as starting the day BL was applied in late April or early May of one year and ending the day before BL was re-applied in the following calendar year. Flow-weighted mean concentrations were calculated by dividing the total elemental mass collected from the lysimeters (i.e., leached) during the time period of
interest from each plot by the total drainage for the same time period from the same plot. Loads
were calculated by dividing the total elemental mass leached for a given plot during the time
period of interest by the lysimeter collection area (0.1935 m²). Similarly, annual and 8-yr mean
pH, EC, and ORP were calculated for each plot. In order to ascertain the effects of BL treatment
on the Na status of annual leachate at the beginning and end of the study, annual leachate sodium
adsorption ratio (SAR) was calculated for each plot for the 2003 and 2010 study years using Eq.
2-1:

\[
\text{SAR} = \frac{[\text{Na}^+]}{0.5[\text{Ca}^{2+}] + 0.5[\text{Mg}^{2+}]}^{0.5}
\]  
[Eq. 2-1]

where [Na⁺], [Ca²⁺], and [Mg²⁺] are the annual measured FWM leachate concentrations
expressed in mEq L⁻¹ of Na, Ca, and Mg, respectively (Bresler et al., 1982).

Statistical Analysis

Analysis of variance (ANOVA) was used to identify BL application rate effects on
annual and 8-yr cumulative DM production using the PROC MIXED procedure in SAS (version
9.2; SAS Institute Inc., Cary, NC) while treating blocks as a random variable. When appropriate,
means were then separated using a protected least significant difference (LSD) at \( \alpha = 0.05 \).

Analysis of covariance (ANCOVA) was used to identify BL application rate (covariate)
effects on the relationship between drainage; mean annual leachate pH, EC, and ORP; and
annual FWM leachate concentrations and leachate loads of NO₃-N, NH₄-N, PO₄-P, DOC, As,
Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Se, and Zn (dependent variables) over the 8-year
period (time, independent variable) using the PROC MIXED procedure in SAS while treating
blocks as a random variable. Initially, a full model was used to test for different slopes among
BL treatments. If slopes were similar, a second, reduced model was used to test for different y-
intercepts among BL treatments. When appropriate, slopes and y-intercepts were estimated and then separated using contrast statements at $\alpha = 0.05$. In cases where BL and time had no effect, treatment means, overall grand mean, and standard error of the mean were calculated for informational purposes. The blocking variance was also determined and expressed as a percentage of total variance by dividing the blocking estimate for a given soil leachate property by the sum of the blocking and error estimates for that soil leachate property and then multiplying by 100. In addition, relationships between annual precipitation, drainage, DM, and leachate dependent variables were assessed by Pearson’s correlation analysis using the PROC CORR procedure in SAS.

Analysis of variance was also used to identify BL application rate effects on 8-yr cumulative drainage; 8-yr mean pH, EC, and ORP; and 8-yr FWM leachate concentrations and cumulative leachate loads and the percent of leached parameter relative to the amount applied in BL for NO$_3$-N, NH$_4$-N, PO$_4$-P, DOC, As, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Se, and Zn using the PROC MIXED procedure in SAS while treating blocks as a random variable. When appropriate, means were separated using a protected LSD at $\alpha = 0.05$. In cases where BL had no effect, treatment means, overall grand mean, and standard error of the mean were calculated. Similarly, ANOVA was used to identify BL application rate effects on leachate annual SAR separately for study years 2003 and 2010.

Results and Discussion

Pre-treatment Uniformity

Since field plots used in this study had received organic amendments prior to 2002 when AETLs were installed, it was necessary to address pre-treatment plot uniformity. Three months
prior to the initial BL application in 2003, precipitation, runoff, drainage, and DM were monitored and soil samples were collected \cite{Pirani2006}. During this 3-mo period, 179 mm of precipitation fell, which was 98 mm below the 30-yr normal for the area during February, March, and April \cite{Pirani2006}. The mean 3-mo cumulative runoff \cite{Menjoulet2009} and drainage \cite{Pirani2006} prior to the first BL application did not differ among pre-assigned BL treatments. In addition, mean runoff EC and FWM runoff concentrations of NO$_3$-N, NH$_4$-N, DOC, As, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Se, and Zn did not differ among pre-assigned BL treatments during the 3-mo period \cite{Menjoulet2009}. Similarly, mean leachate pH, EC, and ORP \cite{Pirani2006} as well as FWM concentrations and loads of DOC \cite{Pirani2006}, NO$_3$-N, NH$_4$-N, Ca, K, Mg, Na, and P \cite{Pirani2007} and FWM leachate concentrations of Mn, Ni, and Zn \cite{Pirani2006} at the 90-cm depth did not differ among pre-assigned BL treatments. In the three months prior to initial BL application, total DM did not differ among pre-assigned BL treatments \cite{Pirani2006}. Soil pH, EC, and organic matter concentration did not differ among pre-assigned BL treatments for any 10-cm soil depth interval to a depth of 90 cm \cite{Pirani2006}. Similarly, total recoverable soil Cd, Cu, and Zn and Mehlich-3 extractable soil P, K, Ca, Mg, and Na did not differ among pre-assigned BL treatments for any 10-cm depth interval prior to the first BL application in 2003 \cite{Pirani2007}.

Based on the number of measured parameters that did not differ among pre-assigned BL treatments during the 3-mo period prior to the initial litter application in 2003, the plots in this study were assumed to be as uniform as reasonably could be expected \cite{Brye2006, Pirani2006, Pirani2007, Daigh2009, McDonald2009, Menjoulet2009}. Therefore, it was also assumed that any subsequent observed differences were due to the
response to newly imposed BL treatments rather than to inherent differences among experimental plots (Pirani et al., 2006; Pirani et al., 2007; Menjoulet et al., 2009).

**Broiler Litter Composition**

The mean annual composition of the BL used throughout this study was 37.1% C, 4.4% N, 3.7% Ca, 3.5% K, and 2.2% P on a dry-weight basis and had 24% moisture by mass when applied (Table 2-1). Similar BL compositions have been previously reported in Pennsylvania (Kleinman et al., 2005), New York (Brock et al., 2007), Arkansas (Adams et al., 1994), and Nigeria (Agele et al., 2004). The mean annual BL C:N ratio was 8.4 averaged over the 8-yr period and suggested that BL decomposition by soil microorganisms would have been relatively quick with a likely net increase in soil N levels that would have promoted plant growth. The mean annual BL N:P ratio was 2.0 averaged over the 8-yr period and suggested that the BL used in this study supplied P in excess of plant growth requirements further suggesting that soil P concentrations would have also increased during this time period. Daigh et al. (2009) reported acid-recoverable, Mehlich-3-extractable, and water-soluble soil-P increased with the addition of BL over the first 5 years of this 8-yr study period. Mean annual inputs of nutrients and other constituents associated with BL treatments are summarized in Table 2-1.

**Precipitation**

During the 8-yr period, annual precipitation at the study site averaged 1178 mm [standard error (SE) = 78] which was 4.4% below the 30-yr mean annual precipitation for Fayetteville, AR (1232 mm; NOAA, 2013). Annual precipitation ranged from a low of 739 mm in 2005 to a high of 1508 mm in 2010 (Figure 2-1). Study years 2003, 2006, 2007, 2008, and 2009 were within ±
7% of the 30-yr mean precipitation, while study years 2004 and 2005 were 13 and 40% below the 30-yr mean precipitation, respectively. Study year 2010 exceeded the 30-yr mean by 22%. During the current study, Fayetteville, AR set three record highs for monthly precipitation totals: March 2008 (study year 2007, 255 mm), Oct 2009 (study year 2009, 272 mm), and April 2011 (study year 2010, 388 mm). Similarly, record least monthly total precipitation records were also set in Nov 2007 (study year 2007, 9 mm) and Aug 2010 (study year 2010; 0.5mm; NOAA, 2013).

Above-ground Dry Matter

Similar to previous studies (Huneycutt et al., 1988; Brye et al., 2010) and as would be expected, additions of BL increased DM relative to the unamended control (Table 2-2). Annual above-ground DM production ranged from a low of 4.9 Mg ha\(^{-1}\) in the unamended control in 2003 to a high of 21.6 Mg ha\(^{-1}\) in the high-litter treatment in 2010 (Table 2-2). Brye et al. (2010) reported that, during the first 6 years of this study, DM production increased over time for both the low- and high-BL treatments, while DM in the unamended control did not change over time. From 2008 to 2010, annual DM did not differ \((P > 0.05)\) among BL treatments because plant speciation shifts increased DM variability. In June 2006, Johnson grass started to encroach into the research area, but contributed little to DM. In July 2007, Johnson grass had become a prominent species contributing to DM in one of the six plots and was observed in two other plots. Encroachment of Johnson grass continued throughout the remainder of the study. In September 2008, Johnson grass was observed in five of six plots, and by November 2010 Johnson grass was observed throughout the entire research area. Eight-year cumulative DM production in the high-BL treatment was greater \((P < 0.05)\) than the unamended control, but did
not differ \((P > 0.05)\) from the low-BL treatment, which was also similar to the unamended control (Table 2-2).

As would be expected, annual DM and annual precipitation were positively correlated \((r = 0.45, P < 0.01)\) during the 8-yr study period, indicating that approximately 20% of the observed variability in annual DM could be attributed to changes in annual precipitation. Dry matter production and yield responses to irrigation and rainfall have been well-documented for forage grasses and crops (Jensen et al., 2001; Fay et al., 2003). In contrast, annual DM and annual drainage were not correlated \((r = -0.01, P = 0.96)\) during the 8-yr period.

**Drainage and Soil Leachate pH, EC, and ORP Trends over Time**

Broiler litter application rate effects on annual soil drainage and soil leachate chemical properties were analyzed by ANCOVA to determine if annual soil leachate properties changed over time. Annual drainage, leachate pH, and EC were unaffected by BL application rate \((P > 0.05)\), time \((P > 0.05)\), or their interaction \((P > 0.05; \text{Table 2-3; Figure 2-2})\) and were summarized by averaging over all treatment conditions (i.e., grand means; Table 2-4). Annual drainage averaged 471 \((SE = 51)\) mm yr\(^{-1}\), which represented 40% of the annual mean precipitation and ranged from a low of 0.9 mm in the low-litter treatment in 2005 to a high of 1003 mm in the unamended control in 2006. In 2005, annual drainage did not exceed 2.9 mm for any treatment because of low precipitation (i.e., 739 mm; Figures 2-1 and 2-2). Unexpectedly, annual drainage was not correlated \((P = 0.08, \text{Table 2-5})\) to annual precipitation. A possible explanation may be related to increased water demands associated with increased DM production observed in later years of the study causing reduced drainage even though precipitation was above normal during the same time period. Similar to Pirani et al. (2006),
mean annual leachate pH and EC were unaffected by BL amendments. Averaged over litter treatment and time, mean annual leachate pH was 6.15 (SE = 0.05) and EC was 190 (SE = 10) µS cm⁻¹ (Table 2-4). Since soil pH has been reported to increase with additions of BL (Sharpley et al., 1993; Kingery et al., 1994; Daigh et al., 2009) and because BL contains base-forming cations (Table 2-1), it was hypothesized that annual leachate pH would increase over time in BL-amended soil relative to the unamended control. Although results reported here were unexpected, decreases in leachate pH have been reported at a depth of 70 cm in pasture and forest soils amended once with metal-spiked sewage sludge (McLaren et al., 2004).

Mean annual leachate ORP decreased 7.1 mV yr⁻¹ (Table 2-6) during the 8-yr period, was unaffected ($P > 0.05$) by BL application rate (Table 2-3, Figure 2-2), and negatively correlated to annual precipitation ($r = -0.34$, $P = 0.02$, Table 2-5). Pirani et al. (2006) also reported no BL rate effect on annual leachate ORP during the first two years of monitoring after reintroducing BL to soil with a history of organic amendments.

**Leachate Concentration Trends over Time**

Broiler litter application rate effects on annual FWM leachate nutrient and metal concentrations were analyzed by ANCOVA to determine if FWM leachate concentrations changed over time. Annual FWM leachate Na concentration was the only monitored leachate parameter in the study to demonstrate an interaction between BL rate and time ($P < 0.01$, Table 2-3) indicating that the relationship between annual FWM leachate Na concentration and time differed among BL treatment conditions (Table 2-3). The slope of the linear relationship between annual FWM leachate Na concentration and time for the unamended control (-0.6 mg Na L⁻¹ yr⁻¹) did not differ from zero ($P = 0.18$), but was different than the significantly positive
slope \((P < 0.01)\) for both treatments that received BL, which did not differ from each other (low- and high-litter slopes were 1.6 and 1.5 mg L\(^{-1}\) yr\(^{-1}\), respectively; Figure 2-3). In addition, annual FWM leachate Na concentrations were positively correlated to annual precipitation \((r = 0.29, P = 0.05, \text{ Table 2-5})\). As hypothesized, long-term annual additions of BL-derived Na (Table 2-1) increased annual FWM Na concentrations in leachate relative to the unamended control.

Although Pirani et al. (2007) reported no treatment effect on annual FWM leachate Na concentrations during the first two years of BL amendments, Daigh et al. (2009) reported that acid-recoverable, Mehlich-3-extractable, and water-soluble soil Na increased in all 10-cm depth intervals to a depth of 80-cm during the first 5 years of continued BL amendments. Results indicate that the accumulation of BL-derived Na within the soil profile increased soil leachate Na concentrations observed in this study only after multiple consecutive annual applications of BL. Shepherd and Bennet (1998) reported similar, but slightly greater, FWM leachate Na concentrations of 22 and 57 mg L\(^{-1}\) at a depth of 1.5 m following a total application of 125 t BL ha\(^{-1}\) over a 3-yr period to sandy soil.

Since high exchangeable Na in conjunction with low salt concentrations may cause swelling, dispersion, and reduced permeability in soil, which in turn may increase translocation of dispersed clay into lower soil horizons where clay films and clay pans may develop, which in turn could limit water movement, it was necessary to characterize annual leachate exchangeable Na. To do this, the annual SAR was calculated and subjected to ANOVA to separately evaluate the first (2003) and last (2010) years of this study. In 2003, SAR did not differ among BL treatments \((P > 0.05)\); thus averaged over litter treatments, the mean annual SAR was 0.83 (SE = 0.07). In contrast, SAR in 2010 was affected by litter treatment \((P < 0.05)\). The unamended control had a mean SAR of 0.43, which differed \((P < 0.05)\) from the high- and low-litter
treatments of 1.13 and 1.08, respectively, which did not differ from each other \((P > 0.05)\).

Although annual leachate SAR was relatively low compared to some irrigation waters in use, the low annual leachate EC of 190 \(\mu\text{S cm}^{-1}\) (Table 2-4) suggested that leachate water was relatively pure and, in conjunction with the observed SAR, suggests that soil structure would not be detrimentally affected (Bresler et al., 1982) at a depth of 90 cm with long-term BL applications. These results highlight the importance of long-term observational studies regarding BL amendments to soil because Na is a relatively mobile cation in soil and one of the first to be leached from a soil profile, but may require more than two years before deep leaching losses are observed in finer-textured soils amended with BL.

In contrast to annual FWM leachate Na concentrations, annual FWM leachate Fe concentration was affected by BL \((P = 0.04)\) and time \((P < 0.01)\), but not their interaction \((P > 0.05, \text{Table 2-3})\) indicating similar slopes for the linear relationship between annual FWM Fe concentration and time across BL treatments, but different y-intercepts for the same relationship across BL treatments (Figure 2-3). Averaged across all BL treatments, annual FWM leachate Fe concentrations decreased \((-0.3 \text{ mg L}^{-1} \text{ yr}^{-1}; \ P < 0.01)\) over time (Figure 2-3). The y-intercepts were 0.24, 0.27, and 0.19 \text{ mg L}^{-1} for the control, low, and high BL treatments, respectively. While the y-intercept for the unamended control did not differ from that for the low- \((P = 0.26)\) or the high-BL \((P = 0.22)\) treatments, the y-intercepts for the low- and high-BL treatments differed from one another \((P = 0.02)\). During the first two years of this study, Pirani et al. (2006) reported no BL-treatment effect on annual FWM leachate Fe concentrations, but observed a seasonal effect for BL. During the spring of 2004, FWM leachate Fe concentrations from the low- and high-litter amended treatments were 0.09 and 0.07 \text{ mg L}^{-1}, respectively, which were similar to each other, but greater than that for the unamended control \((0.03 \text{ mg L}^{-1}; \ P = 0.04);\)
Pirani et al., 2006). Pirani et al. (2006) attributed the spring effect in part to wet soil conditions that increased Fe mobility and thus increased Fe concentrations in the leachate.

In contrast to annual FWM Na concentrations, annual FWM leachate NH$_4$-N, As, Mn, and Ni concentrations decreased over time ($P \leq 0.05$) during the 8-yr period (Figure 2-4), but neither the slopes nor the y-intercepts for the linear relationships between these annual FWM concentrations and time were affected by BL treatment ($P > 0.05$; Table 2-3). Annual FWM leachate NH$_4$-N and Ni concentrations had greater concentrations and variances in 2005 than other years, presumably due to low drainage associated with below normal precipitation in 2005. Annual precipitation was negatively correlated ($P < 0.01$) to annual FWM leachate NH$_4$-N ($r = -0.73$) and Ni ($r = -0.77$, Table 2-5) concentrations. Similarly, annual FWM leachate Ni concentrations and annual drainage were negatively correlated ($r = -0.43$, $P < 0.01$). Averaged across litter treatments, annual FWM leachate NH$_4$-N concentration decreased 0.03 mg L$^{-1}$ yr$^{-1}$ during the 8-yr study. Similar to a 4-yr study by Brye et al. (2001) who used ETLs to monitor nitrogen and carbon leaching from a Plano silt loam soil in a tallgrass prairie restoration and two N-fertilized maize (Zea may L.) agroecosystems (no-tillage and chisel-plowed), leachate NH$_4$-N concentrations did not exceed 1 mg NH$_4$-N L$^{-1}$. Pirani et al. (2007) also reported annual FWM leachate NH$_4$-N concentrations to be unaffected by BL treatment during the first year after BL application. However, during the second year, Pirani et al. (2007) reported that annual FWM leachate NH$_4$-N concentration was greater in the low-litter treatment (0.04 mg L$^{-1}$) than in the unamended control (0.02 mg L$^{-1}$).

Averaged across litter treatments, annual FWM leachate As concentrations decreased ($P < 0.01$) 0.01 mg L$^{-1}$ yr$^{-1}$ during the 8-yr period (Figure 2-4). The first three years after BL application had greater annual FWM leachate As concentrations and increased variances than the
last 5 years of the study (Figure 2-4). During the first three years for the litter-amended treatments and the first two years for the unamended control, annual FWM leachate As concentrations exceeded the National Primary Drinking Water Regulations’ Maximum Contaminant Level (MCL) of 0.01 mg L\(^{-1}\) (USEPA, 2009). The explanation for the observed increase in leachate As concentrations and related variances during the first three years of the study is unclear. If the reintroduction of BL amendments to soil with a history of BL amendments that had not received BL for some time was responsible, one would expect the unamended control to maintain a relatively low annual FWM leachate As concentration compared to the BL-amended treatments, similar to that observed in 2005 (Figure 2-4).

However, because the unamended control also demonstrated increased leachate As concentrations in 2003 and 2004 relative to the last 5 years of the study, the elevated leachate As concentrations may be due to the change in management practices associated with the start of the study. After three annual spring applications of 5 Mg BL ha\(^{-1}\) to a Maury silt-loam soil and 28 drainage events between July 2007 and April 2009, D’Angelo et al. (2012) reported soil leachate collected at a 0.9-m depth using zero-tension pan lysimeters to have As concentrations below the level of detection (i.e., 0.005 mg L\(^{-1}\)) and concluded that As was strongly retained by the soil.

Similar to Pirani et al. (2006), annual soil FWM leachate As concentrations were unaffected by BL treatment throughout the entire 8-yr study. Pirani et al. (2006) also reported a spike in leachate As concentration in 2003, 55 d after BL application in the high-litter treatment, which coincided with a leachate concentration spike in DOC, Se, Mn, and Fe. In contrast to the results of the current study, Jackson et al. (2006) suggested that BL amendments to soil increased As solubility via increased competition for soil adsorption sites and increased complexation with BL-derived DOC. Similarly, McDonald et al. (2009) reported increased As sorption in an
unamended control soil compared to soil receiving four consecutive years of BL amendments. In addition, Peryea and Kammereck (1997) reported an interaction between application of monoammonium phosphate and number of pore-volume displacements (i.e., the amount of leaching or drainage) to increase soil leachate As concentrations from an arsenate-contaminated, loam topsoil collected from an apple orchard.

During the course of the current study, annual FWM leachate As concentrations exceeded the National Primary Drinking Water Regulations’ Maximum Contaminant Level (MCL) of 0.01 mg L\(^{-1}\) (USEPA, 2013) in all years except 2006 and 2007. In 2005, only the annual FWM leachate As concentration for the unamended control remained below the MCL, while that for the BL-amended treatments exceeded the MCL (Figure 2-4). Because the unamended control exceeded the MCL for As long after BL amendments had ceased, it is possible that soil with a history of BL amendments may continue to pose environmental risks even after BL amendments have stopped.

Similar to annual FWM leachate NH\(_4\)-N and As concentrations and averaged across litter treatments, annual FWM leachate Mn and Ni concentrations decreased (\(P < 0.05\)) 0.002 and 0.005 mg L\(^{-1}\) yr\(^{-1}\), respectively, during the 8-yr period (Figure 2-4, Table 2-6). Similar to Pirani et al. (2006), annual FWM leachate Mn and Ni concentrations were unaffected by BL application rate (Table 2-3). The y-intercept estimates for the relationships between annual FWM leachate Mn and Ni concentrations and time averaged over BL treatments both differed from zero (\(P < 0.01\), Table 2-6). Annual FWM leachate Mn concentrations did not exceed the secondary MCL level of 0.05 mg L\(^{-1}\) (USEPA, 2009) for drinking water at anytime during the 8-yr study period. Annual FWM leachate Ni concentrations were negatively correlated (\(P < 0.01\)) to annual precipitation (\(r = -0.77\)) and drainage (\(r = -0.43\), Table 2-5), which explains why the greatest
annual concentrations were observed in the driest study year (2005) when drainage was low (Figure 2-1). In 2005, annual FWM leachate Ni concentrations exceeded the National Recommended Water Quality Criteria for the protection of freshwater aquatic life of 0.052 mg L⁻¹ (USEPA, 2013). This occurrence suggests that during dry years, when soil drainage and baseflow to streams is low, surface waters may receive baseflow waters with elevated Ni concentrations precisely when aquatic life may be experiencing other stresses, such as low dissolved oxygen levels or elevated water temperatures.

Similar to mean annual leachate ORP, annual FWM leachate Ca, Cu, Mg, and Se concentrations were unaffected by BL application rate ($P > 0.05$, Table 2-3), but increased ($P < 0.05$) during the 8-yr period (Table 2-6, Figure 2-5). The $y$-intercept estimates for the linear relationships between these variables and time differed from zero ($P < 0.05$) for FWM Ca, Mg, and Se leachate concentrations, while the $y$-intercept for FWM Cu did not differ from zero ($P > 0.05$, Table 2-6). Pirani et al. (2007) also reported no BL treatment effect on annual FWM leachate Ca concentrations during the first two years of this drainage study. In contrast, Agele et al. (2004) reported a BL treatment effect using a zero-tension collection method from under a disturbed soil, where it was reported that the average annual leachate Ca concentrations were 80 and 30 mg L⁻¹ from BL-amended (10 t BL ha⁻¹) and unamended treatments, respectively. Similar to Agele et al. (2004), Shepherd and Bennett (1998) reported numeric increases in average leachate Ca concentrations with increasing BL application rate. Similar to the current study, Shepherd and Bennett (1998) also reported numeric increases in leachate Ca concentrations over time, although not formally analyzed. In a concurrent and related study, Daigh et al. (2009) reported increased acid-recoverable soil Ca after five consecutive years of annual BL amendments of 11.2 Mg BL ha⁻¹ relative to an unamended control. Daigh et al.
(2009) also reported no BL-treatment effect on Mehlich-3-extractable or water-soluble soil Ca during the same time period, although water-soluble soil Ca decreased over time regardless of BL treatment. Results of Daigh et al. (2009), in conjunction with the observed increase in annual FWM leachate Ca concentrations reported in the current study, suggest that additions of BL-derived Ca (Table 2-1) to soil either entered the acid-recoverable-soil-Ca fraction or was removed from the top 90 cm of soil in the water-soluble-soil-Ca fraction by means of leaching.

Similar to annual FWM leachate Ca and Mg, annual FWM leachate Cu and Se increased ($P < 0.05$) during the 8-yr period, but were unaffected by litter treatment ($P > 0.05$, Table 2-3). Annual FWM leachate Cu concentrations never exceeded the action level of the National Primary or Secondary Drinking Water Regulations of 1.3 or 1.0 mg L$^{-1}$, respectively (USEPA, 2009). In contrast, annual FWM leachate Se concentrations exceeded the MCL of 0.05 mg L$^{-1}$ for study years 2007 through 2010 (USEPA, 2009). Annual precipitation was positively ($P < 0.01$) correlated to annual FWM Cu ($r = 0.44$) and Se ($r = 0.60$, Table 2-5) concentrations. The relationship between leachate Se concentration and precipitation partially explains the pattern in Figure 2-5. Because the soil parent material does not naturally contain appreciable Se and because there was not a litter treatment effect, the Se leached from the soil profile must have originated from the organic amendments that were added prior to the initiation of the current study. Additionally, improved analytical instrumentation with lower detection limits for trace metals during the duration of the study may have contributed to the observed changes over time for annual FWM Se concentrations.

Similar to annual leachate pH and EC, annual FWM leachate NO$_3$-N, PO$_4$-P, DOC, Cd, Cr, K, P, and Zn concentrations did not change over time ($P > 0.05$), were unaffected by BL treatment ($P > 0.05$) or the interaction between BL treatment and time ($P > 0.05$, Table 2-3), and
were summarized by their grand means (Table 2-4) for the 8-yr period. During the study, annual FWM concentrations did not exceed 0.06 mg L\(^{-1}\) for Cd and Cr (Figure 2-6), 1.2 mg L\(^{-1}\) for NO\(_3\)-N, PO\(_4\)-P, P, and Zn (Figures 2-6 and 2-7), or 24 mg L\(^{-1}\) for DOC and K (Figure 2-6). Similar to annual FWM leachate NH\(_4\)-N, Cu, Mn, and Ni concentrations, annual FWM leachate Cr and Zn had elevated concentrations and increased variances in study year 2005 when precipitation and drainage were low (Figure 2-1). Annual precipitation was correlated (\(P < 0.01\)) with annual FWM leachate Cd (\(r = 0.41\)), Cr (\(r = -0.46\)), and Zn (\(r = -0.60\), Table 2-5) concentrations. Similarly, annual drainage was negatively correlated (\(P < 0.05\)) with annual FWM leachate DOC (\(r = -0.42\)), P (\(r = -0.33\)), and Zn (\(r = -0.45\)) concentrations. Averaged over BL treatments and time, the annual FWM leachate NO\(_3\)-N concentration was 0.11 (SE = 0.02, Table 2-4).

Additionally, the annual FWM leachate NO\(_3\)-N concentrations never exceeded the primary MCL of 10 mg L\(^{-1}\) for drinking water (USEPA, 2009). Similarly, the annual FWM leachate Zn concentration was 0.28 (SE = 0.03) mg L\(^{-1}\) and never exceeded the recommended secondary MCL of 5 mg L\(^{-1}\) (USEPA, 2009).

**Leachate Load Trends over Time**

Broiler litter application rate effect on annual leachate nutrient and metal loads were analyzed by ANCOVA to determine if leachate loads changed over time. In general, annual leachate nutrient and metal loads were lowest during study year 2005 (Figures 2-8 through 2-11) when drainage was low and were positively correlated (Table 2-7) to annual drainage. Similar to annual FWM leachate NH\(_4\)-N, As, Mn, and Ni concentrations, annual leachate NH\(_4\)-N, As, Fe, Mn, and Ni loads decreased over time (\(P < 0.05\), Tables 2-6 and 2-8) during the 8-yr period (Figure 2-8) and were unaffected by litter treatment (\(P > 0.05\)). Annual leachate NH\(_4\)-N, As, Fe,
Mn, and Ni loads decreased 0.03, 0.04, 0.16, 0.01, and 0.01 kg ha\(^{-1}\) yr\(^{-1}\), respectively (Table 2-6). Pirani et al. (2007) reported no difference among litter treatments in annual leachate NH\(_4\)-N loads during the first two years of a concurrent study. Similarly, Pirani et al. (2006) reported no difference among litter treatments in annual leachate As, Fe, Mn, and Ni loads during the same two years. However, Pirani et al. (2006) reported BL treatment affects during study year 2004 for seasonal As loads in the winter and Fe loads in the fall, suggesting that the temporal leaching losses associated with BL applications may occur on timescales shorter than the annual timescale used in the current study.

Similar to annual FWM leachate Ca, Cu, Mg, and Se concentrations, annual leachate Cu and Se loads increased \((P < 0.01)\) during the 8-yr period and were unaffected by BL treatment \((P > 0.05)\) or the interaction between BL and time \((P > 0.05, \text{ Table 2-8})\). Annual leachate Cu and Se loads increased at a rate of 0.01 and 0.36 kg ha\(^{-1}\) yr\(^{-1}\), respectively (Table 2-6, Figure 2-9). Although the y-intercept estimate for the linear relationship between annual leachate Cu loads across time was negative, it did not differ from zero \((P > 0.05, \text{ Table 2-6})\). A similar pattern was observed for the y-intercept estimate for the linear relationship between annual FWM Cu concentrations across time (Table 2-6). Annual leachate Cu loads ranged from a low of zero in the low-litter treatment in 2005 to a high of 0.08 kg ha\(^{-1}\) in the unamended control in 2009 (Figure 2-9). Similarly, annual leachate Se loads ranged from a low of zero in all BL treatment conditions in both 2005 and 2006 to a high of 3.6 kg ha\(^{-1}\) in the unamended control in 2009 (Figure 2-9). Annual leachate Cu and Se loads were positively \((P < 0.01)\) correlated to annual precipitation \((r = 0.57 \text{ and } 0.46, \text{ respectively, Table 2-7})\). Similarly, annual Cu loads were correlated to annual drainage \((r = 0.39, P = 0.01)\).
Identical to annual leachate FWM NO\textsubscript{3}\textendash N, PO\textsubscript{4}\textendash P, DOC, Ca, Cd, Cr, K, Mg, Na, P, and Zn concentrations, annual leachate loads for these same nutrients and metals were unchanged over time ($P > 0.05$), were unaffected by BL treatment ($P > 0.05$) or the interaction between BL and time ($P > 0.05$, Table 2-8), and were summarized by their grand means (Table 2-4) for the 8-yr period. In general, the leachate load ranges for these nutrients and metals occurred at similar times during the study with the low end of the range occurring in 2005 when drainage was low and the high end of the range occurring in either 2004 or 2006 when drainage was high (Figures 2-1, 2-10 and 2-11). In addition, annual drainage was positively ($P \leq 0.01$) correlated to all annual nutrient and metal loads, with the exception of PO\textsubscript{4}\textendash P, P, and Se loads, and annual precipitation was positively ($P < 0.05$) correlated to annual leachate DOC ($r = 0.32$), Ca ($r = 0.29$), and Na ($r = 0.34$, Table 2-7) loads. In general, coefficient of correlations between annual drainage and annual leachate loads were positive, while correlations between annual drainage and annual FWM leachate concentrations tended to be negative suggesting a dilution effect in leachate that is similar to the one observed in simulated runoff by Edwards and Daniel (1993).

**Annual Blocking Variance**

Experimental blocks were treated as a random variable during statistical analyses (ANCOVA) of annual leachate concentrations and loads in order to gain insight regarding spatial variability at the research location. Variability among blocks accounted for approximately 38\% of the total observed variability in annual mean drainage (Table 2-9) and suggested that spatial variability in drainage may be relatively high at the 90-cm depth at the study site. Similarly, variability among blocks accounted for approximately 53, 35, 13, and 62\% of the total variability in mean annual FWM leachate concentrations of PO\textsubscript{4}\textendash P, DOC, Fe, and P (Table 2-9),
respectively, and indicated that the soil’s natural spatial variability may warrant an increase in the number of replications in future studies if inferences are to be made with regard to annual FWM leachate PO$_4$-P or P concentrations. Although reasonable pre-treatment plot uniformity was demonstrated, the observed block variability associated with annual FWM PO$_4$-P, DOC, Fe, or P leachate concentrations may also be related to runoff studies conducted on-site prior to the initiation of the current study in which organic soil amendments were used. However, Pirani et al. (2006) reported no pre-treatment effects for FWM leachate DOC or P concentrations for the 3-month period prior to the initiation of this study.

In contrast to mean annual FWM leachate PO$_4$-P, DOC, Fe, and P concentrations, mean annual leachate pH, ORP, EC, and FWM leachate NO$_3$-N, NH$_4$-N, As, Ca, Cr, Cu, Mg, Ni, Se, and Zn concentrations had negative block variance estimates (Table 2-9) indicating two possible explanations. First, variability among blocks contributed little to total variability in mean annual leachate pH, ORP, EC and FWM concentration of these nutrients and metals. This explanation is most likely for annual leachate pH, ORP, and EC. Second, block variance estimates actually estimated correlation coefficients between drainage and leachate concentrations. Because variance is calculated by dividing the sum of squared deviations by the sample size, variance must be positive. The discrepancy of negative variances arises because SAS estimates covariance parameters. It is possible that these estimates are small and do not differ from or approach zero, which may be the case for some parameters, such as mean annual leachate pH, ORP, and EC, but it is also possible that these estimates are correlation coefficient estimates. Because the correlation coefficient is the covariance divided by the standard deviation of the variable, it is possible to have a negative correlation estimate. Because annual FWM leachate concentration was calculated by dividing the total elemental mass collected in leachate during a
study year by the total drainage for that year, the FWM concentration is a quotient of two
dependant variables (i.e., annual drainage and loads). In support of this interpretation, annual
drainage and annual FWM leachate NO$_3$-N, NH$_4$-N, Ca, Cr, Cu, Mg, Ni, Se, and Zn
concentrations had negative correlation coefficients ($r$, Table 2-5).

In contrast to annual FWM leachate nutrient and metal concentrations and with the
exception of annual leachate Fe loads, all blocking variances expressed as a percent of total
variance for annual load parameters were positive (Table 2-10). Blocking variance for leachate
loads ranged from a low of 6% for annual leachate As and Mn loads to a high of 47% for annual
leachate Cu loads suggesting that blocking within the current study was successful with regards
to number of blocks used. However, future research designs at this location may attain greater
sensitivity to statistical analyses if a minimum of three blocks are used. If annual leachate Cu or
Na loads or annual FWM leachate PO$_4$-P or P concentrations are to be studied, four blocks
would be recommended.

**Eight-year Mean Drainage Chemistry, FWM Concentrations, and Cumulative Loads**

After eight years of annual BL amendments, 8-yr cumulative drainage; 8-yr mean pH,
EC, and ORP; and 8-yr FWM leachate concentrations (Table 2-11) and leachate loads (Table 2-
12) of nutrients and trace metals (NO$_3$-N, NH$_4$-N, PO$_4$-P, DOC, As, Ca, Cd, Cr, Cu, Fe, K, Mg,
Mn, Na, Ni, P, Se, and Zn) did not differ ($P > 0.05$) among litter treatments. Similar results were
reported for 2-yr cumulative leachate loads during the first two years of a concurrent study
(Pirani et al., 2006; Pirani et al., 2007). In addition, cumulative 8-yr leaching of total nutrients
and trace metals expressed as a percentage of that applied in BL did not differ ($P > 0.05$)
between the low- and high-litter treatments (Table 2-13). Cumulative 8-yr leaching of NH$_4$-N,
C, N, P, Mn, and Cu represented less than 2% of that applied in litter treatments, while cumulative 8-yr leaching of NO$_3$-N, K, Ca, Na, Mg, Zn, Fe, As, Ni, and Cr represented between 9 and 99% of that applied in BL. However, cumulative 8-yr leaching of Se and Cd exceeded 170% of that applied in BL. Pirani et al. (2007) reported similar results for cumulative 2-yr leaching loads for the same nutrients reported in the current study with the exception of NO$_3$-N, which Pirani et al. (2007) reported as 104 and 52% for the low- and high-litter treatments, respectively. In addition, Pirani et al. (2006) reported similar results for cumulative 2-yr leaching loads expressed as a percentage of that applied in BL for metals with the exception of As and Se. Pirani et al. (2006) reported 2-yr cumulative As loads from low- and high-litter treatments of 757 and 378%, respectively, compared to the 60 and 43% (Table 2-13) reported in the current study covering an 8-yr period. Pirani et al. (2006) also reported 2-yr cumulative loads for Se of 159 and 80% compared to the 1463 and 1347% reported here. The elevated mass losses (i.e., > 100%) for Se reported in the current study indicated that leaching losses of Se from BL-amended pasture land with a history of BL applications, that contained trace metals like Se, may eventually release Se into the environment via leaching below 0.9 m. The observed positive correlations between precipitation and leachate Se concentrations and loads (Tables 2-5 and 2-7) suggest that Se may be retained in soil during drier years and then released during wetter years, with an overall net accumulation of Se in the soil profile.

Considering BL treatment had no effect (P > 0.05) on annual nutrient and metal leaching (Tables 2-3 and 2-8) or 8-yr cumulative leaching (Tables 2-11 and 2-12), but Se leaching percents exceeded 100%, percent losses corrected for leaching losses from the unamended control were calculated (corrected losses, Table 2-13) by subtraction of the control treatment’s 8-yr cumulative load from the litter-treated 8-yr cumulative loads within the same experimental
block. Similar to 8-yr cumulative percent losses, cumulative 8-yr leaching of total nutrients and trace metals corrected for control losses and expressed as a percentage of that applied in BL did not differ ($P > 0.05$) between the low- and high-litter treatments (Table 2-13). Negative corrected percent losses would indicate nutrient and metal losses originated from organic soil amendments that occurred before the current study began in 2002, but variability within the low-litter treatment increased uncertainty and limited the inferences that could be made. Averaging corrected percent losses over low- and high-litter treatments and calculating 95% confidence intervals (CI) allowed comparison of the mean to zero. Results indicated that zero fell within the CI for all nutrients and metals, with the exception of NO$_3$-N (mean = 17.8, CI$_{upper} = 29.7$, CI$_{lower} = 6.0\%$) and total-N (mean = 0.65, CI$_{upper} = 1.3$, CI$_{lower} = 0.001\%$). This was strong evidence that BL-derived total-N leached from the soil profile as NO$_3$-N. Similar results have been previously reported (Adams et al., 1994; Agele et al., 2004). In addition, because CIs for NH$_4$-N, C (as DOC), P, Mn, and Cu were small and contained zero, these BL-derived nutrients and metals are not leaching from the soil.

**Summary of Environmental Concerns**

Based on the current 8-yr observational study and with the exception of annual FWM leachate Na and Fe concentrations, annual leaching trends were influenced primarily by time and not BL application rate, with annual FWM leachate Ca, Cu, Mg, and Se concentrations and annual Cu and Se loads increasing with time. Correlation coefficients indicated a dilution effect occurring with increased annual drainage and/or precipitation for annual FWM leachate NH$_4$-N, DOC, Cr, Ni, P, and Zn concentrations. Similarly but with the exceptions of annual PO$_4$-P, P, and Se loads, annual nutrient and metal loads increased with increased annual drainage.
During the course of the current study, annual FWM leachate As concentrations exceeded the National Primary Drinking Water Regulations’ MCL of 0.01 mg L$^{-1}$ (USEPA, 2013) in all years except 2006 and 2007. Similarly, annual FWM leachate Se concentrations exceeded the primary MCL of 0.05 mg L$^{-1}$ for study years 2007 through 2010 (USEPA, 2009). However, annual leachate concentrations of NO$_3$-N, Cd, and Cu never exceeded the primary MCL’s of 10, 0.005, and 1.3 mg L$^{-1}$, respectively, and annual leachate concentrations of Cu, Mn, and Zn never exceeded secondary MCLs (USEPA, 2009). However, in 2005, annual FWM leachate Ni concentrations exceeded the National Recommended Water Quality Criteria for the protection of freshwater aquatic life of 0.052 mg L$^{-1}$ (USEPA, 2013), which is important in karst regions where surface and groundwater intermingle freely. Although leachate at the 0.9-m depth may have environmentally sensitive concentrations of nutrients or metals, at many locations the distance to groundwater is great enough for further removal by adsorption of contaminants before leachate enters groundwater.

Because repeated application of BL to pastureland over many years has resulted in the accumulation of nutrients and metals within soil, it is possible that future environmental changes could potentially release these nutrients or metals to move towards groundwater. For example, in the current study, annual FWM As concentrations were elevated during the first three years of the study, but were similar during the 3-month pre-litter observation period suggesting that management practices may have increased As release from the soil. Similarly, annual Se concentrations and loads were positively correlated to annual precipitation and exceeded the primary MCL during wetter years suggesting that as atmospheric carbon dioxide concentrations increase in the future, rain may become more acidic, thus potentially increasing mobility of some metals.
Additionally, some metals had leaching patterns that required eight years to identify. If a relatively mobile cation like Na requires three or more years before BL-induced leaching changes are observed, then less mobile cations, such as Ca, may require observational studies longer than eight years to identify BL-induced changes in leaching. Similarly, some leaching patterns, such as that of P, may require decades of observation.

**Summary and Conclusions**

Based on continuous monitoring for 8-yr of nutrient and metal losses for BL-amended pasture land with a history of organic amendments and under naturally occurring precipitation using AETLs, annual FWM leachate Na and Fe concentrations were the only water quality parameters affected by litter application rate. Similarly, 8-yr cumulative leaching loads corrected for leaching losses from the unamended control provided evidence that BL-derived N leached from the soil profile in the form of NO$_3$-N. Correlation coefficients between annual drainage and annual FWM leachate concentrations during an 8-yr period were mostly negative, and in conjunction with the positive correlations between annual drainage and annual leachate loads suggested a dilution effect similar to those observed previously in runoff.

Leaching trends over time were the most common result reported for most nutrients and metals. Mean annual drainage, leachate pH and EC were unchanged during the 8-yr period. Mean annual FWM concentrations and loads of NH$_4$-N, As, Mn, and Ni decreased and mean annual FWM concentrations and loads of Cu and Se increased during the 8 years. In contrast, mean annual FWM concentrations and loads of NO$_3$-N, PO$_4$-P, Cd, Cr, K, P, Zn, and DOC did not vary over time.
The lack of evidence to support BL-induced leaching effects in this study can be attributed to four factors. First, the timescale used in the current study may be too long to identify relatively short-term leaching patterns like that of NO$_3$-N. Similarly, the timescale used in this study may be too short to identify leaching patterns that may require decades to identify. Second, leachate was collected in response to naturally occurring precipitation instead of simulated rainfall, which is commonly applied in quantities greater than normal precipitation. Third, pre-2002 soil amendments to the research plots controlled leachate composition to a greater degree than did the current study’s imposed BL treatments. Fourth, limited replication in association with increased variability associated with drainage reduced the strength of most statistical calculations.
References


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Table 2-1. Mean annual broiler litter (BL) composition and constituent added in low- (5.6 Mg ha\(^{-1}\)) and high- (11.2 Mg ha\(^{-1}\)) litter treatments over an 8-yr period to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas. Annual mean maxima and minima are provided as an indication of parameter range.

<table>
<thead>
<tr>
<th>Litter Property</th>
<th>Mean Annual Composition</th>
<th>Mean Maximum</th>
<th>Mean Minimum</th>
<th>Litter Rate Low kg ha(^{-1})</th>
<th>Litter Rate High kg ha(^{-1})</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moisture (kg kg(^{-1}))</td>
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<td>0.27</td>
<td>0.21</td>
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</tr>
<tr>
<td>pH</td>
<td>8.4</td>
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</tr>
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<td>EC(^{\dagger}) (dS m(^{-1}))</td>
<td>11.9</td>
<td>14.8</td>
<td>9.8</td>
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<td></td>
</tr>
<tr>
<td>NO(_3)-N (mg kg(^{-1}))</td>
<td>207</td>
<td>513</td>
<td>38</td>
<td>1.1</td>
<td>2.31</td>
</tr>
<tr>
<td>NH(_4)-N (mg kg(^{-1}))</td>
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<td>7183</td>
<td>2877</td>
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<td>52.0</td>
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<tr>
<td>Total Elements</td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>C (%)</td>
<td>37.1</td>
<td>39.5</td>
<td>33.9</td>
<td>2078</td>
<td>4155</td>
</tr>
<tr>
<td>N (%)</td>
<td>4.4</td>
<td>5.3</td>
<td>4.0</td>
<td>246</td>
<td>493</td>
</tr>
<tr>
<td>P (%)</td>
<td>2.2</td>
<td>2.6</td>
<td>1.6</td>
<td>123</td>
<td>246</td>
</tr>
<tr>
<td>K (%)</td>
<td>3.5</td>
<td>4.4</td>
<td>2.9</td>
<td>196</td>
<td>392</td>
</tr>
<tr>
<td>Ca (%)</td>
<td>3.7</td>
<td>4.4</td>
<td>2.9</td>
<td>207</td>
<td>414</td>
</tr>
<tr>
<td>Mg (%)</td>
<td>0.7</td>
<td>0.8</td>
<td>0.6</td>
<td>39.2</td>
<td>78.4</td>
</tr>
<tr>
<td>S (%)</td>
<td>1.1</td>
<td>1.6</td>
<td>0.6</td>
<td>61.6</td>
<td>123.2</td>
</tr>
<tr>
<td>Na (mg kg(^{-1}))</td>
<td>9098</td>
<td>16094</td>
<td>3857</td>
<td>50.9</td>
<td>101.9</td>
</tr>
<tr>
<td>Al (mg kg(^{-1}))</td>
<td>347</td>
<td>558</td>
<td>243</td>
<td>1.9</td>
<td>3.9</td>
</tr>
<tr>
<td>Fe (mg kg(^{-1}))</td>
<td>413</td>
<td>613</td>
<td>197</td>
<td>2.3</td>
<td>4.6</td>
</tr>
<tr>
<td>Mn (mg kg(^{-1}))</td>
<td>568</td>
<td>751</td>
<td>421</td>
<td>3.2</td>
<td>6.4</td>
</tr>
<tr>
<td>Zn (mg kg(^{-1}))</td>
<td>510</td>
<td>645</td>
<td>395</td>
<td>2.9</td>
<td>5.7</td>
</tr>
<tr>
<td>Cu (mg kg(^{-1}))</td>
<td>496</td>
<td>678</td>
<td>298</td>
<td>2.8</td>
<td>5.6</td>
</tr>
<tr>
<td>B (mg kg(^{-1}))</td>
<td>52.6</td>
<td>60.9</td>
<td>46.5</td>
<td>0.29</td>
<td>0.59</td>
</tr>
<tr>
<td>Ni (mg kg(^{-1}))</td>
<td>10.4</td>
<td>16.1</td>
<td>5.9</td>
<td>0.058</td>
<td>0.116</td>
</tr>
<tr>
<td>Cd (mg kg(^{-1}))</td>
<td>0.19</td>
<td>0.60</td>
<td>0.05</td>
<td>0.001</td>
<td>0.002</td>
</tr>
<tr>
<td>Cr (mg kg(^{-1}))</td>
<td>7.7</td>
<td>15.6</td>
<td>3.1</td>
<td>0.04</td>
<td>0.09</td>
</tr>
<tr>
<td>As (mg kg(^{-1}))</td>
<td>26.8</td>
<td>39.9</td>
<td>19</td>
<td>0.15</td>
<td>0.30</td>
</tr>
<tr>
<td>Se (mg kg(^{-1}))</td>
<td>3.5</td>
<td>7.3</td>
<td>1.6</td>
<td>0.019</td>
<td>0.039</td>
</tr>
</tbody>
</table>

\(^{\dagger}\)EC, electrical conductivity.
Table 2-2. Broiler litter application rate (control, 0 Mg ha\(^{-1}\); low, 5.6 Mg ha\(^{-1}\); and high, 11.2 Mg ha\(^{-1}\)) effects on mean annual above-ground dry matter and 8-yr cumulative production.

<table>
<thead>
<tr>
<th>Time Period†</th>
<th>P-value</th>
<th>Mean Dry Matter (Mg ha(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Control</td>
</tr>
<tr>
<td>2003‡</td>
<td>0.04</td>
<td>4.9a</td>
</tr>
<tr>
<td>2004‡</td>
<td>0.01</td>
<td>5.6a</td>
</tr>
<tr>
<td>2005§</td>
<td>0.04</td>
<td>5.3a</td>
</tr>
<tr>
<td>2006§</td>
<td>0.02</td>
<td>5.3a</td>
</tr>
<tr>
<td>2007¶</td>
<td>&lt; 0.01</td>
<td>5.6a</td>
</tr>
<tr>
<td>2008</td>
<td>0.19</td>
<td>9.5a</td>
</tr>
<tr>
<td>2009</td>
<td>0.11</td>
<td>10.4a</td>
</tr>
<tr>
<td>2010</td>
<td>0.18</td>
<td>12.0a</td>
</tr>
<tr>
<td>8-yr Cumulative</td>
<td>0.05</td>
<td>58.7a</td>
</tr>
</tbody>
</table>

† Study years are designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. For example 2003 represents the time period from May 2003 to April 2004.

‡ Data for 2003 and 2004 were taken from Pirani (2005).
§ Data for 2005 and 2006 were taken from Menjoulet et al. (2009).
¶ Data for 2007 were taken from Daigh et al. (2009).
# Means in the same row followed by different letters are significantly different (P < 0.05).
Table 2-3. Analysis of covariance summary of the effects of broiler litter (BL) application rate, time (Year), and their interaction on the linear relationship between select soil leachate properties and flow-weighted mean concentrations and time.

<table>
<thead>
<tr>
<th>Leachate Property</th>
<th>Source of Variance</th>
<th>BL</th>
<th>Year</th>
<th>BL x Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Drainage</td>
<td></td>
<td>0.59</td>
<td>0.51</td>
<td>0.88</td>
</tr>
<tr>
<td>pH</td>
<td></td>
<td>0.98</td>
<td>0.82</td>
<td>0.91</td>
</tr>
<tr>
<td>ORP†</td>
<td></td>
<td>0.98</td>
<td>&lt; 0.01 ‡‡</td>
<td>0.84</td>
</tr>
<tr>
<td>EC‡</td>
<td></td>
<td>0.60</td>
<td>0.61</td>
<td>0.22</td>
</tr>
</tbody>
</table>

Concentrations

<table>
<thead>
<tr>
<th>Leachate Property</th>
<th>BL</th>
<th>Year</th>
<th>BL x Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO₃⁻-N</td>
<td>0.91</td>
<td>0.55</td>
<td>0.81</td>
</tr>
<tr>
<td>NH₄⁺-N</td>
<td>0.71</td>
<td><strong>0.05</strong></td>
<td>0.88</td>
</tr>
<tr>
<td>PO₄³⁻-P</td>
<td>0.46</td>
<td>0.14</td>
<td>0.74</td>
</tr>
<tr>
<td>DOC§</td>
<td>0.13</td>
<td>0.53</td>
<td>0.18</td>
</tr>
<tr>
<td>As</td>
<td>0.98</td>
<td>&lt; 0.01</td>
<td>0.99</td>
</tr>
<tr>
<td>Ca</td>
<td>0.36</td>
<td>&lt; 0.01</td>
<td>0.35</td>
</tr>
<tr>
<td>Cd</td>
<td>0.08</td>
<td>0.10</td>
<td>0.22</td>
</tr>
<tr>
<td>Cr</td>
<td>0.27</td>
<td>0.26</td>
<td>0.32</td>
</tr>
<tr>
<td>Cu</td>
<td>0.14</td>
<td>&lt; 0.01</td>
<td>0.20</td>
</tr>
<tr>
<td>Fe</td>
<td><strong>0.04</strong></td>
<td>&lt; 0.01</td>
<td>0.14</td>
</tr>
<tr>
<td>K</td>
<td>0.31</td>
<td>0.86</td>
<td>0.42</td>
</tr>
<tr>
<td>Mg</td>
<td>0.38</td>
<td><strong>0.02</strong></td>
<td>0.34</td>
</tr>
<tr>
<td>Mn</td>
<td>0.23</td>
<td><strong>0.03</strong></td>
<td>0.15</td>
</tr>
<tr>
<td>Na</td>
<td>0.08</td>
<td>&lt; 0.01</td>
<td>&lt; 0.01</td>
</tr>
<tr>
<td>Ni</td>
<td>0.74</td>
<td><strong>0.01</strong></td>
<td>0.80</td>
</tr>
<tr>
<td>P</td>
<td>0.32</td>
<td>0.28</td>
<td>0.78</td>
</tr>
<tr>
<td>Se</td>
<td>0.92</td>
<td>&lt; 0.01</td>
<td>0.80</td>
</tr>
<tr>
<td>Zn</td>
<td>0.29</td>
<td>0.20</td>
<td>0.53</td>
</tr>
</tbody>
</table>

† ORP, oxidation-reduction potential.
‡ EC, electrical conductivity.
§ DOC, dissolved organic carbon.
‡ Test for different y-intercepts among BL treatments with common slope.
# Test if common slope is different than zero.
†† Test for different slopes among BL treatments.
‡‡ P ≤ 0.05 are indicated in bold.
Table 2-4. Summary of annual broiler litter (BL) treatment means and grand means across all BL treatments for soil leachate properties that were unaffected by BL or time (Tables 2-3 and 2-8) during an 8-yr period as determined by analysis of covariance. Broiler litter was hand applied once annually to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas at three application rates (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively). Standard errors of treatment means and grand means are provided in parenthesis as estimates of variability.

<table>
<thead>
<tr>
<th>Leachate Property</th>
<th>Annual Broiler Litter Treatment Mean (^$)</th>
<th>Grand Mean (^$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Drainage (mm)</td>
<td>532 (106)</td>
<td>471 (51)</td>
</tr>
<tr>
<td>pH</td>
<td>6.09 (0.09)</td>
<td>6.15 (0.05)</td>
</tr>
<tr>
<td>EC(^\dagger) (µS cm(^{-1}))</td>
<td>175 (18)</td>
<td>190 (10)</td>
</tr>
</tbody>
</table>

Concentrations (mg L\(^{-1}\))

<table>
<thead>
<tr>
<th>Property</th>
<th>Control</th>
<th>Low</th>
<th>High</th>
<th>Grand Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO(_3)-N</td>
<td>0.07 (0.02)</td>
<td>0.11 (0.05)</td>
<td>0.14 (0.03)</td>
<td>0.11 (0.02)</td>
</tr>
<tr>
<td>PO(_4)-P</td>
<td>0.19 (0.05)</td>
<td>0.12 (0.02)</td>
<td>0.10 (0.03)</td>
<td>0.13 (0.02)</td>
</tr>
<tr>
<td>DOC(^\dagger)</td>
<td>3.4 (0.3)</td>
<td>4.2 (0.3)</td>
<td>3.4 (0.3)</td>
<td>3.7 (0.2)</td>
</tr>
<tr>
<td>Cr</td>
<td>0.01 (&lt; 0.01)</td>
<td>0.01 (0.01)</td>
<td>0.01 (&lt; 0.01)</td>
<td>0.01 (&lt; 0.01)</td>
</tr>
<tr>
<td>K</td>
<td>18.2 (2.1)</td>
<td>16.6 (1.2)</td>
<td>19.9 (1.2)</td>
<td>18.2 (0.9)</td>
</tr>
<tr>
<td>P</td>
<td>0.24 (0.06)</td>
<td>0.12 (0.01)</td>
<td>0.12 (0.03)</td>
<td>0.16 (0.02)</td>
</tr>
<tr>
<td>Zn</td>
<td>0.33 (0.08)</td>
<td>0.28 (0.04)</td>
<td>0.23 (0.02)</td>
<td>0.28 (0.03)</td>
</tr>
</tbody>
</table>

Loads (kg ha\(^{-1}\))

<table>
<thead>
<tr>
<th>Property</th>
<th>Control</th>
<th>Low</th>
<th>High</th>
<th>Grand Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO(_3)-N</td>
<td>0.25 (0.06)</td>
<td>0.41 (0.25)</td>
<td>0.74 (0.17)</td>
<td>0.47 (0.10)</td>
</tr>
<tr>
<td>PO(_4)-P</td>
<td>0.65 (0.27)</td>
<td>0.42 (0.09)</td>
<td>0.37 (0.13)</td>
<td>0.48 (0.10)</td>
</tr>
<tr>
<td>DOC</td>
<td>14.6 (2.7)</td>
<td>16.3 (2.9)</td>
<td>15.5 (2.2)</td>
<td>15.5 (1.5)</td>
</tr>
<tr>
<td>Ca</td>
<td>94.8 (23)</td>
<td>50.0 (9)</td>
<td>91.5 (17)</td>
<td>78.7 (10)</td>
</tr>
<tr>
<td>Cd</td>
<td>0.01 (&lt; 0.01)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
</tr>
<tr>
<td>Cr</td>
<td>0.04 (0.02)</td>
<td>0.02 (0.01)</td>
<td>0.03 (0.01)</td>
<td>0.03 (0.01)</td>
</tr>
<tr>
<td>K</td>
<td>107 (25)</td>
<td>61.7 (11)</td>
<td>103 (19)</td>
<td>90.4 (11)</td>
</tr>
<tr>
<td>Mg</td>
<td>31.7 (8)</td>
<td>18.4 (3)</td>
<td>31.8 (6)</td>
<td>27.3 (3)</td>
</tr>
<tr>
<td>Na</td>
<td>62.0 (15)</td>
<td>50.4 (8)</td>
<td>74.6 (11)</td>
<td>62.3 (7)</td>
</tr>
<tr>
<td>P</td>
<td>0.80 (0.3)</td>
<td>0.43 (0.1)</td>
<td>0.44 (0.1)</td>
<td>0.55 (0.1)</td>
</tr>
<tr>
<td>Zn</td>
<td>1.1(0.2)</td>
<td>0.8 (0.1)</td>
<td>1.2 (0.2)</td>
<td>1.0 (0.1)</td>
</tr>
</tbody>
</table>

\(^\dagger\) EC, electrical conductivity.
\(^\dagger\) DOC, dissolved organic carbon.
\(^\$\) Treatment means, n = 16.
\(^\$\) Grand means, n = 48.
Table 2-5. Correlation coefficients ($r$) for both annual precipitation and annual drainage with annual leachate properties and flow-weighted mean leachate concentrations during an 8-yr period ($n = 48$).

<table>
<thead>
<tr>
<th>Leachate Property</th>
<th>Precipitation</th>
<th>Drainage</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>$r$</td>
<td>$P$-value</td>
</tr>
<tr>
<td>Drainage</td>
<td>0.26</td>
<td>0.08</td>
</tr>
<tr>
<td>pH</td>
<td>-0.07</td>
<td>0.65</td>
</tr>
<tr>
<td>ORP$^\dagger$</td>
<td>-0.34</td>
<td>0.02#</td>
</tr>
<tr>
<td>EC$^\ddagger$</td>
<td>0.11</td>
<td>0.47</td>
</tr>
</tbody>
</table>

Concentrations
- NO$_3$-N: -0.14 0.33 -0.09 0.53
- NH$_4$-N: -0.73 < 0.01 -0.44 < 0.01
- PO$_4$-P: -0.10 0.50 -0.27 0.06
- DOC$^\S$: 0.22 0.13 -0.42 < 0.01
- As: -0.26 0.07 0.18 0.21
- Ca: 0.12 0.42 -0.07 0.64
- Cd: 0.41 < 0.01 0.24 0.11
- Cr: -0.46 < 0.01 -0.26 0.08
- Cu: 0.44 < 0.01 -0.17 0.25
- Fe: -0.12 0.41 0.01 0.97
- K: 0.01 0.96 0.21 0.15
- Mg: 0.07 0.66 -0.06 0.68
- Mn: -0.07 0.62 -0.06 0.70
- Na: 0.29 0.05 -0.05 0.72
- Ni: -0.77 < 0.01 -0.43 < 0.01
- P: 0.15 0.30 -0.33 0.02
- Se: 0.60 < 0.01 -0.12 0.41
- Zn: -0.60 < 0.01 -0.45 < 0.01

$^\dagger$ ORP, oxidation-reduction potential.
$^\ddagger$ EC, electrical conductivity.
$^\S$ DOC, dissolved organic carbon.
$^\|$ Pearson test for correlation.
# $P \leq 0.05$ are indicated in bold.
Table 2-6. Summary of common intercept and slope estimates for the linear relationship between the annual soil leachate property and time. Soil was amended once annually with broiler litter (BL) at three application rates (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) for an 8-yr period.

<table>
<thead>
<tr>
<th>Leachate Property</th>
<th>Intercept</th>
<th>( P )-value( ^\dagger )</th>
<th>Slope</th>
<th>( R^2 )( ^\S )</th>
</tr>
</thead>
<tbody>
<tr>
<td>ORP( ^\dagger ) (mV)</td>
<td>75.3</td>
<td>&lt; 0.01</td>
<td>-7.1</td>
<td>0.32</td>
</tr>
</tbody>
</table>

Concentrations (mg L\(^{-1}\))

<table>
<thead>
<tr>
<th></th>
<th>Intercept</th>
<th>( P )-value( ^\dagger )</th>
<th>Slope</th>
<th>( R^2 )( ^\S )</th>
</tr>
</thead>
<tbody>
<tr>
<td>NH(_4)-N</td>
<td>0.23</td>
<td>&lt; 0.01</td>
<td>-0.03</td>
<td>0.09</td>
</tr>
<tr>
<td>As</td>
<td>0.05</td>
<td>&lt; 0.01</td>
<td>-0.01</td>
<td>0.26</td>
</tr>
<tr>
<td>Ca</td>
<td>10.6</td>
<td>&lt; 0.01</td>
<td>1.4</td>
<td>0.17</td>
</tr>
<tr>
<td>Cu</td>
<td>-0.002</td>
<td>0.11</td>
<td>0.002</td>
<td>0.53</td>
</tr>
<tr>
<td>Mg</td>
<td>4.0</td>
<td>&lt; 0.01</td>
<td>0.42</td>
<td>0.12</td>
</tr>
<tr>
<td>Mn</td>
<td>0.024</td>
<td>&lt; 0.01</td>
<td>-0.002</td>
<td>0.09</td>
</tr>
<tr>
<td>Ni</td>
<td>0.044</td>
<td>&lt; 0.01</td>
<td>-0.005</td>
<td>0.16</td>
</tr>
<tr>
<td>Se</td>
<td>-0.17</td>
<td>&lt; 0.01</td>
<td>0.09</td>
<td>0.72</td>
</tr>
</tbody>
</table>

Loads (kg ha\(^{-1}\))

<table>
<thead>
<tr>
<th></th>
<th>Intercept</th>
<th>( P )-value( ^\dagger )</th>
<th>Slope</th>
<th>( R^2 )( ^\S )</th>
</tr>
</thead>
<tbody>
<tr>
<td>NH(_4)-N</td>
<td>0.38</td>
<td>&lt; 0.01</td>
<td>-0.03</td>
<td>0.11</td>
</tr>
<tr>
<td>As</td>
<td>0.29</td>
<td>&lt; 0.01</td>
<td>-0.04</td>
<td>0.21</td>
</tr>
<tr>
<td>Cu</td>
<td>-0.01</td>
<td>0.16</td>
<td>0.01</td>
<td>0.41</td>
</tr>
<tr>
<td>Fe</td>
<td>1.24</td>
<td>&lt; 0.01</td>
<td>-0.16</td>
<td>0.25</td>
</tr>
<tr>
<td>Mn</td>
<td>0.13</td>
<td>&lt; 0.01</td>
<td>-0.01</td>
<td>0.13</td>
</tr>
<tr>
<td>Ni</td>
<td>0.11</td>
<td>&lt; 0.01</td>
<td>-0.01</td>
<td>0.11</td>
</tr>
<tr>
<td>Se</td>
<td>-0.69</td>
<td>0.04</td>
<td>0.36</td>
<td>0.41</td>
</tr>
</tbody>
</table>

\( ^\dagger \) ORP, oxidation-reduction potential.
\( ^\dagger \) Test if common intercept is different than zero.
\( ^\S \) Coefficient of determination (\( R^2 \)) is provided as a measure of the strength of the relationship.
Table 2-7. Correlation coefficients ($r$) for annual precipitation and drainage with annual leachate loads during an 8-yr period (n = 48).

<table>
<thead>
<tr>
<th>Leachate Property</th>
<th>Precipitation</th>
<th>Drainage</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>$r$</td>
<td>$P$-value$^\dagger$</td>
</tr>
<tr>
<td>NO$_3$-N</td>
<td>0.12</td>
<td>0.41</td>
</tr>
<tr>
<td>NH$_4$-N</td>
<td>0.07</td>
<td>0.65</td>
</tr>
<tr>
<td>PO$_4$-P</td>
<td>-0.02</td>
<td>0.87</td>
</tr>
<tr>
<td>DOC$^\dagger$</td>
<td>0.32</td>
<td><strong>0.03$^\S$</strong></td>
</tr>
<tr>
<td>As</td>
<td>-0.11</td>
<td>0.47</td>
</tr>
<tr>
<td>Ca</td>
<td>0.29</td>
<td><strong>0.05$^\S$</strong></td>
</tr>
<tr>
<td>Cd</td>
<td>0.14</td>
<td>0.35</td>
</tr>
<tr>
<td>Cr</td>
<td>0.21</td>
<td>0.15</td>
</tr>
<tr>
<td>Cu</td>
<td>0.57</td>
<td>$&lt;0.01$</td>
</tr>
<tr>
<td>Fe</td>
<td>-0.10</td>
<td>0.51</td>
</tr>
<tr>
<td>K</td>
<td>0.19</td>
<td>0.20</td>
</tr>
<tr>
<td>Mg</td>
<td>0.28</td>
<td>0.06</td>
</tr>
<tr>
<td>Mn</td>
<td>0.05</td>
<td>0.73</td>
</tr>
<tr>
<td>Na</td>
<td>0.34</td>
<td><strong>0.02$^\S$</strong></td>
</tr>
<tr>
<td>Ni</td>
<td>0.07</td>
<td>0.63</td>
</tr>
<tr>
<td>P</td>
<td>0.10</td>
<td>0.49</td>
</tr>
<tr>
<td>Se</td>
<td>0.46</td>
<td>$&lt;0.01$</td>
</tr>
<tr>
<td>Zn</td>
<td>0.25</td>
<td>0.08</td>
</tr>
</tbody>
</table>

$^\dagger$ DOC, dissolved organic carbon.

$^\dagger$ Pearson test for correlation.

$^\S$ $P \leq 0.05$ are indicated in bold.
Table 2-8. Analysis of covariance summary of the effects of broiler litter (BL) application rate, time (Year), and their interaction on the linear relationship between soil leachate loads and time.

<table>
<thead>
<tr>
<th>Leachate Property</th>
<th>Source of Variance</th>
<th>BL</th>
<th>Year</th>
<th>BL x Year</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>‡</td>
<td>§</td>
<td>¶</td>
</tr>
<tr>
<td>NO₃-N</td>
<td></td>
<td>0.47</td>
<td>1.00</td>
<td>0.93</td>
</tr>
<tr>
<td>NH₄-N</td>
<td></td>
<td>0.44</td>
<td>0.02</td>
<td>0.73</td>
</tr>
<tr>
<td>PO₄-P</td>
<td></td>
<td>0.61</td>
<td>0.12</td>
<td>0.88</td>
</tr>
<tr>
<td>DOC†</td>
<td></td>
<td>0.98</td>
<td>0.59</td>
<td>0.95</td>
</tr>
<tr>
<td>As</td>
<td></td>
<td>0.59</td>
<td>&lt; 0.01</td>
<td>0.75</td>
</tr>
<tr>
<td>Ca</td>
<td></td>
<td>0.66</td>
<td>0.31</td>
<td>0.96</td>
</tr>
<tr>
<td>Cd</td>
<td></td>
<td>0.15</td>
<td>0.69</td>
<td>0.46</td>
</tr>
<tr>
<td>Cr</td>
<td></td>
<td>0.55</td>
<td>0.96</td>
<td>0.20</td>
</tr>
<tr>
<td>Cu</td>
<td></td>
<td>0.87</td>
<td>&lt; 0.01</td>
<td>0.29</td>
</tr>
<tr>
<td>Fe</td>
<td></td>
<td>0.74</td>
<td>&lt; 0.01</td>
<td>0.93</td>
</tr>
<tr>
<td>K</td>
<td></td>
<td>0.45</td>
<td>0.40</td>
<td>0.89</td>
</tr>
<tr>
<td>Mg</td>
<td></td>
<td>0.63</td>
<td>0.41</td>
<td>0.95</td>
</tr>
<tr>
<td>Mn</td>
<td></td>
<td>0.42</td>
<td>0.01</td>
<td>0.64</td>
</tr>
<tr>
<td>Na</td>
<td></td>
<td>0.45</td>
<td>0.68</td>
<td>0.60</td>
</tr>
<tr>
<td>Ni</td>
<td></td>
<td>0.11</td>
<td>0.01</td>
<td>0.49</td>
</tr>
<tr>
<td>P</td>
<td></td>
<td>0.40</td>
<td>0.70</td>
<td>0.74</td>
</tr>
<tr>
<td>Se</td>
<td></td>
<td>0.91</td>
<td>&lt; 0.01</td>
<td>0.51</td>
</tr>
<tr>
<td>Zn</td>
<td></td>
<td>0.53</td>
<td>0.60</td>
<td>0.90</td>
</tr>
</tbody>
</table>

† DOC, dissolved organic carbon.
‡ Test for different y-intercepts among BL treatments with common slope.
§ Test if common slope is different than zero.
¶ Test for different slopes among BL treatments.
# P ≤ 0.05 are indicated in bold.
Table 2-9. Blocking variance expressed as a percentage of total variance for annual mean leachate properties and flow-weighted mean concentrations. Percents were calculated by dividing the blocking estimate for a given soil leachate property by the sum of the blocking and error estimates for that soil leachate property and then multiplying by 100.

<table>
<thead>
<tr>
<th>Leachate Property</th>
<th>Percent of Total Variance</th>
</tr>
</thead>
<tbody>
<tr>
<td>Drainage</td>
<td>38.2</td>
</tr>
<tr>
<td>pH</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>ORP(^{†})</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>EC(^{‡})</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>NO(_3)-N</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>NH(_4)-N</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>PO(_4)-P</td>
<td>53.4</td>
</tr>
<tr>
<td>DOC(^{§})</td>
<td>34.8</td>
</tr>
<tr>
<td>As</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Ca</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Cd</td>
<td>1.7</td>
</tr>
<tr>
<td>Cr</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Cu</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Fe</td>
<td>12.8</td>
</tr>
<tr>
<td>K</td>
<td>0.1</td>
</tr>
<tr>
<td>Mg</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Mn</td>
<td>1.0</td>
</tr>
<tr>
<td>Na</td>
<td>1.1</td>
</tr>
<tr>
<td>Ni</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>P</td>
<td>61.8</td>
</tr>
<tr>
<td>Se</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Zn</td>
<td>&lt; 0.1</td>
</tr>
</tbody>
</table>

\(^{†}\) ORP, oxidation-reduction potential.
\(^{‡}\) EC, electrical conductivity.
\(^{§}\) DOC, dissolved organic carbon
Table 2-10. Blocking variance expressed as a percentage of total variance for annual leachate loads. Percents were calculated by dividing the blocking estimate for a given soil leachate property by the sum of the blocking and error estimates for that soil leachate property and then multiplying by 100.

<table>
<thead>
<tr>
<th>Leachate Property</th>
<th>Percent of Total Variance</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO₃-N</td>
<td>13.4</td>
</tr>
<tr>
<td>NH₄-N</td>
<td>23.8</td>
</tr>
<tr>
<td>PO₄-P</td>
<td>20.4</td>
</tr>
<tr>
<td>DOC†</td>
<td>9.5</td>
</tr>
<tr>
<td>As</td>
<td>6.0</td>
</tr>
<tr>
<td>Ca</td>
<td>23.8</td>
</tr>
<tr>
<td>Cd</td>
<td>24.1</td>
</tr>
<tr>
<td>Cr</td>
<td>13.4</td>
</tr>
<tr>
<td>Cu</td>
<td>46.7</td>
</tr>
<tr>
<td>Fe</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>K</td>
<td>33.1</td>
</tr>
<tr>
<td>Mg</td>
<td>27.3</td>
</tr>
<tr>
<td>Mn</td>
<td>6.0</td>
</tr>
<tr>
<td>Na</td>
<td>42.3</td>
</tr>
<tr>
<td>Ni</td>
<td>34.6</td>
</tr>
<tr>
<td>P</td>
<td>19.7</td>
</tr>
<tr>
<td>Se</td>
<td>34.5</td>
</tr>
<tr>
<td>Zn</td>
<td>34.0</td>
</tr>
</tbody>
</table>

† DOC, dissolved organic carbon.
Table 2-11. Analysis of variance summary of the effects of broiler litter (BL) application rate on 8-yr cumulative drainage; 8-yr mean leachate pH, EC, and ORP; and 8-yr flow-weighted mean leachate concentrations. Broiler litter was hand applied once annually to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas at three application rates (0, 5.6, and 11.2 Mg BL ha⁻¹; control, low, and high, respectively) for an 8-yr period. Standard errors of treatment means and grand means are provided in parenthesis as estimates of variability.

<table>
<thead>
<tr>
<th>Leachate Property</th>
<th>P-value</th>
<th>Broiler Litter Treatment Mean†</th>
<th>Grand Mean#</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Control</td>
<td>Low</td>
</tr>
<tr>
<td>Drainage (mm)</td>
<td>0.52</td>
<td>4254 (2168)</td>
<td>2851 (529)</td>
</tr>
<tr>
<td>pH</td>
<td>0.63</td>
<td>6.09 (0.09)</td>
<td>6.19 (0.02)</td>
</tr>
<tr>
<td>ORP† (mV)</td>
<td>0.53</td>
<td>48.0 (5.9)</td>
<td>42.9 (0.5)</td>
</tr>
<tr>
<td>EC‡ (µS cm⁻¹)</td>
<td>0.69</td>
<td>175 (48)</td>
<td>173(27)</td>
</tr>
<tr>
<td>Concentrations (mg L⁻¹)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NO₃-N</td>
<td>0.17</td>
<td>0.04 (0.01)</td>
<td>0.10 (0.04)</td>
</tr>
<tr>
<td>NH₄-N</td>
<td>0.13</td>
<td>0.04 (&lt;0.01)</td>
<td>0.05 (&lt;0.01)</td>
</tr>
<tr>
<td>PO₄-P</td>
<td>0.67</td>
<td>0.23 (0.21)</td>
<td>0.12 (0.02)</td>
</tr>
<tr>
<td>DOC§</td>
<td>0.75</td>
<td>3.44(1.35)</td>
<td>4.45 (0.59)</td>
</tr>
<tr>
<td>As</td>
<td>0.82</td>
<td>0.03 (&lt; 0.01)</td>
<td>0.03 (&lt;0.01)</td>
</tr>
<tr>
<td>Ca</td>
<td>0.79</td>
<td>15.7 (4.1)</td>
<td>14.5 (2.8)</td>
</tr>
<tr>
<td>Cd</td>
<td>0.50</td>
<td>&lt;0.01 (&lt; 0.01)</td>
<td>&lt;0.01 (&lt; 0.01)</td>
</tr>
<tr>
<td>Cr</td>
<td>0.67</td>
<td>0.01 (&lt;0.01)</td>
<td>0.01 (&lt;0.01)</td>
</tr>
<tr>
<td>Cu</td>
<td>0.57</td>
<td>0.01 (&lt;0.01)</td>
<td>0.01 (&lt;0.01)</td>
</tr>
<tr>
<td>Fe</td>
<td>0.64</td>
<td>0.16 (0.13)</td>
<td>0.18 (&lt;0.01)</td>
</tr>
<tr>
<td>K</td>
<td>0.87</td>
<td>18.1 (3.9)</td>
<td>17.7 (1.9)</td>
</tr>
<tr>
<td>Mg</td>
<td>0.77</td>
<td>5.30 (1.30)</td>
<td>5.24 (0.52)</td>
</tr>
<tr>
<td>Mn</td>
<td>0.74</td>
<td>0.01 (&lt;0.01)</td>
<td>0.02 (0.01)</td>
</tr>
<tr>
<td>Na</td>
<td>0.36</td>
<td>10.6 (2.0)</td>
<td>14.0 (0.9)</td>
</tr>
<tr>
<td>Ni</td>
<td>0.60</td>
<td>0.01 (&lt;0.01)</td>
<td>0.01 (&lt;0.01)</td>
</tr>
<tr>
<td>P</td>
<td>0.65</td>
<td>0.28 (0.26)</td>
<td>0.12 (0.01)</td>
</tr>
<tr>
<td>Se</td>
<td>0.92</td>
<td>0.19 (0.03)</td>
<td>0.18 (0.04)</td>
</tr>
<tr>
<td>Zn</td>
<td>1.00</td>
<td>0.22 (0.05)</td>
<td>0.22 (0.01)</td>
</tr>
</tbody>
</table>

† ORP, oxidation-reduction potential.
‡ EC, electrical conductivity.
§ DOC, dissolved organic carbon.
¶ Treatment means, n = 2.
# Grand means, n = 6.
Table 2-12. Analysis of variance summary of the effects of broiler litter (BL) application rate on 8-yr cumulative soil leachate loads. Broiler litter was hand applied once annually to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas at three application rates (0, 5.6, and 11.2 Mg BL ha$^{-1}$; control, low, and high, respectively) for an 8-yr period. Standard errors of treatment means and grand means are provided in parenthesis as estimates of variability.

<table>
<thead>
<tr>
<th>Leachate Property</th>
<th>$P$-value</th>
<th>Broiler Litter Treatment Mean$^{\dagger}$</th>
<th>kg ha$^{-1}$</th>
<th>Grand Mean$^{\S}$</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Control</td>
<td>Low</td>
<td>High</td>
</tr>
<tr>
<td>NO$_3$-N</td>
<td>0.06</td>
<td>2.0 (1.3)</td>
<td>3.3 (1.8)</td>
<td>5.9 (2.3)</td>
</tr>
<tr>
<td>NH$_4$-N</td>
<td>0.53</td>
<td>1.9 (1.1)</td>
<td>1.4 (0.3)</td>
<td>2.2 (0.9)</td>
</tr>
<tr>
<td>PO$_4$-P</td>
<td>0.73</td>
<td>5.2 (4.0)</td>
<td>3.4 (&lt;0.1)</td>
<td>3.0 (2.3)</td>
</tr>
<tr>
<td>DOC$^{\ddagger}$</td>
<td>0.86</td>
<td>117 (17)</td>
<td>130 (40)</td>
<td>124 (10)</td>
</tr>
<tr>
<td>As</td>
<td>0.61</td>
<td>1.1 (0.5)</td>
<td>0.7 (0.1)</td>
<td>1.0 (0.5)</td>
</tr>
<tr>
<td>Ca</td>
<td>0.64</td>
<td>758 (514)</td>
<td>400 (2)</td>
<td>731 (146)</td>
</tr>
<tr>
<td>Cd</td>
<td>0.51</td>
<td>0.05 (0.04)</td>
<td>0.03 (&lt;0.01)</td>
<td>0.01 (0.02)</td>
</tr>
<tr>
<td>Cr</td>
<td>0.58</td>
<td>0.3 (0.2)</td>
<td>0.1 (&lt;0.1)</td>
<td>0.2 (0.1)</td>
</tr>
<tr>
<td>Cu</td>
<td>0.40</td>
<td>0.2 (0.1)</td>
<td>0.2 (&lt;0.1)</td>
<td>0.3 (0.1)</td>
</tr>
<tr>
<td>Fe</td>
<td>0.70</td>
<td>4.1 (1.9)</td>
<td>5.2 (1.0)</td>
<td>3.1 (0.4)</td>
</tr>
<tr>
<td>K</td>
<td>0.63</td>
<td>856 (560)</td>
<td>493 (39)</td>
<td>821 (266)</td>
</tr>
<tr>
<td>Mg</td>
<td>0.63</td>
<td>254 (170)</td>
<td>147 (13)</td>
<td>254 (56)</td>
</tr>
<tr>
<td>Mn</td>
<td>0.78</td>
<td>0.5 (0.4)</td>
<td>0.4 (0.1)</td>
<td>0.7 (0.3)</td>
</tr>
<tr>
<td>Na</td>
<td>0.56</td>
<td>496 (317)</td>
<td>404 (101)</td>
<td>596 (190)</td>
</tr>
<tr>
<td>Ni</td>
<td>0.51</td>
<td>0.4 (0.3)</td>
<td>0.3 (&lt;0.1)</td>
<td>0.7 (0.5)</td>
</tr>
<tr>
<td>P</td>
<td>0.71</td>
<td>6.4 (4.9)</td>
<td>3.4 (0.3)</td>
<td>3.5 (2.5)</td>
</tr>
<tr>
<td>Se</td>
<td>0.44</td>
<td>8.5 (5.2)</td>
<td>5.5 (2.2)</td>
<td>8.4 (4.1)</td>
</tr>
<tr>
<td>Zn</td>
<td>0.45</td>
<td>8.5 (2.8)</td>
<td>6.4 (1.5)</td>
<td>9.5 (4.4)</td>
</tr>
</tbody>
</table>

$^{\dagger}$ DOC, dissolved organic carbon.
$^{\ddagger}$ Treatment means, n = 2.
$^{\S}$ Grand means, n = 6.
Table 2-13. Leaching mass balance for broiler litter-derived nutrients from a pasture soil amended with broiler litter (BL). Litter was hand applied once annually to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas at three application rates (0, 5.6 and 11.2 Mg BL ha\(^{-1}\); control, low and high, respectively) for an 8-yr period. Leachate was continuously monitored and collected by automated equilibrium tension lysimeters. Values represent 8-yr totals.

<table>
<thead>
<tr>
<th>Leachate Property</th>
<th>Mass Added(^{†})</th>
<th>Mass Leached(^{‡})</th>
<th>Percent Loss(^§)</th>
<th>Corrected Loss(^¶)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Low (kg ha(^{-1}))</td>
<td>High (kg ha(^{-1}))</td>
<td>Low (%)</td>
<td>High (%)</td>
</tr>
<tr>
<td>NO(_3)N</td>
<td>8.8</td>
<td>18.5</td>
<td>3.3</td>
<td>5.9</td>
</tr>
<tr>
<td>NH(_4)N</td>
<td>208</td>
<td>416</td>
<td>1.4</td>
<td>2.2</td>
</tr>
<tr>
<td>Total Elements</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>C</td>
<td>16624</td>
<td>33248</td>
<td>130</td>
<td>124</td>
</tr>
<tr>
<td>N</td>
<td>217</td>
<td>435</td>
<td>4.7</td>
<td>8.1</td>
</tr>
<tr>
<td>P</td>
<td>984</td>
<td>1968</td>
<td>3.4</td>
<td>3.5</td>
</tr>
<tr>
<td>K</td>
<td>1568</td>
<td>3136</td>
<td>493</td>
<td>821</td>
</tr>
<tr>
<td>Ca</td>
<td>1656</td>
<td>3312</td>
<td>400</td>
<td>731</td>
</tr>
<tr>
<td>Na</td>
<td>407</td>
<td>814</td>
<td>404</td>
<td>596</td>
</tr>
<tr>
<td>Mg</td>
<td>314</td>
<td>627</td>
<td>147</td>
<td>254</td>
</tr>
<tr>
<td>Mn</td>
<td>26</td>
<td>51</td>
<td>0.4</td>
<td>0.7</td>
</tr>
<tr>
<td>Zn</td>
<td>23</td>
<td>46</td>
<td>6.4</td>
<td>9.5</td>
</tr>
<tr>
<td>Cu</td>
<td>22</td>
<td>45</td>
<td>0.2</td>
<td>0.3</td>
</tr>
<tr>
<td>Fe</td>
<td>18</td>
<td>37</td>
<td>5.2</td>
<td>3.1</td>
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\(^{†}\) Cumulative mass of nutrient added from BL amendments.
\(^{‡}\) Cumulative mass of nutrient leached.
\(^§\) Percent loss equal to mass leached minus mass added, times 100.
\(^¶\) Corrected percent loss was corrected for leaching loss from unamended control before calculating percent loss. Negative values represent control treatment within same experimental block loss was greater that littered treatment.
Figure 2-1. Annual precipitation and drainage from broiler litter (BL) amended pasture soil during eight years. Broiler litter was hand applied once annually to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas at three application rates (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively). Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. The horizontal dashed line represents the 30-yr mean precipitation of 1232 mm (NOAA, 2013).
Figure 2-2. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effect over an 8-yr period on annual drainage [standard error (SE) = 51], annual leachate pH (SE = 0.05), annual electrical conductivity (EC; SE = 10), and annual oxidation-reduction potential (ORP). Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. The solid line for ORP represents a significant \((P < 0.05)\) change over time averaged across BL treatments.
Figure 2-3. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effect over an 8-yr period on annual flow-weighted-mean (FWM) leachate sodium (Na) and iron (Fe) concentrations. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. The alternating long-short dashed line represents changes in annual soil FWM leachate Na concentrations over time averaged over the low- and high-BL treatments. The solid, small-dashed, and long-dashed lines represent changes in annual soil FWM leachate Fe concentrations over time for the control, low, and high BL treatments, respectively.
Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha⁻¹; control, low, and high, respectively) effect over an 8-yr period on annual flow-weighted-mean leachate ammonium-nitrogen (NH₄-N), arsenic (As), manganese (Mn), and nickel (Ni) concentrations. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. Solid lines represent changes over time averaged across all BL treatments.
Figure 2-5. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effect over an 8-yr period on annual flow-weighted-mean leachate calcium (Ca), copper (Cu), magnesium (Mg), and selenium (Se) concentrations. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. Solid lines represent changes over time averaged across all BL treatments.
Figure 2-6. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha⁻¹; control, low, and high, respectively) effect over an 8-yr period on annual flow-weighted-mean leachate nitrate-nitrogen [NO₃-N; standard error (SE) = 0.02], phosphate-phosphorus (PO₄-P; SE =0.02), dissolved organic carbon (DOC; SE = 0.2), cadmium (Cd; SE < 0.01), chromium (Cr; SE < 0.01), and potassium (K; SE = 0.9) concentrations. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year.
Figure 2-7. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effect over an 8-yr period on annual flow-weighted-mean leachate phosphorus [P; standard error (SE) = 0.02] and zinc (Zn; SE = 0.03) concentrations. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year.
Figure 2-8. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha$^{-1}$; control, low, and high, respectively) effect over an 8-yr period on annual leachate ammonium-nitrogen (NH$_4$-N), iron (Fe), arsenic (As), nickel (Ni), and manganese (Mn) loads. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. Solid lines represent changes over time averaged across all BL treatments.
Figure 2-9. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effect over an 8-yr period on annual leachate copper (Cu) and selenium (Se) loads. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. Solid lines represent changes over time averaged across all BL treatments.
Figure 2-10. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effect over an 8-yr period on annual leachate nitrate-nitrogen [NO\(_3\)-N; standard error (SE) = 0.01], phosphate-phosphorus (PO\(_4\)-P; SE = 0.01), dissolved organic carbon (DOC; SE = 1.5), calcium (Ca; SE = 78.7), cadmium (Cd; SE < 0.01), and chromium (Cr; SE = 0.01) loads. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year.
Figure 2-11. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effect over an 8-yr period on annual leachate potassium [K; standard error (SE) = 11], sodium (Na; SE = 7), magnesium (Mg; SE = 3), zinc (Zn; SE = 0.1), and phosphorus (P; SE = 0.1) loads. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year.
Appendix 2-1. Example SAS program for analysis of covariance run as a macro and related data.

options mprint;

title 'SEASONAL LEACHATE LOADS and FWM -- Richard McMullen --';
title2 'All elements Leachate Model Estimates pH, Redox, EC and DOC etc.';

data loads;
infile 'F:\McMullen\Leachate\SAS\LeachateAnnualSums.csv' firstobs=2 DLM=',' truncover LRECL = 600 DSD;
input Year BL $ Block Lys Vol pH redox EC NO3 NH3 PO4 DOC Al As B Ca Cd Co Cr Cu Fe K Mg Mn Mo Na Ni TP Pb S Se Ti Zn fwmNO3 fwmNH3 fwmPO4 fwmDOC fwmAl fwmAs fwmB fwmCa fwmCd fwmCo fwmCr fwmCu fwmFe fwmK fwmMg fwmMn fwmMo fwmNa fwmNi fwmP fwmPb fwmS fwmSe fwmTi fwmZn volmm;
* year = 2002 + year;
label BL = 'Broiler Litter';
run;

proc sort data=loads; by year block; run; quit;

title3 'INITIAL DATA LISTING';
proc print data=loads noobs; by year block;
id year block;
var lys bl Vol pH redox EC NO3 NH3 PO4 DOC Al As B Ca Cd Co Cr Cu Fe K Mg Mn Mo Na Ni TP Pb S Se Ti Zn fwmNO3 fwmNH3 fwmPO4 fwmDOC fwmAl fwmAs fwmB fwmCa fwmCd fwmCo fwmCr fwmCu fwmFe fwmK fwmMg fwmMn fwmMo fwmNa fwmNi fwmP fwmPb fwmS fwmSe fwmTi fwmZn volmm;
run cancel;
quit;

%Macro xxx(var=);

title3 'Parameter ESTIMATES & CONTRASTS for 1st TEST- Diff Slopes, Diff Intercepts';
title4 "-------- &var --------";
proc mixed data=loads method=type3 ;
class block bl ;
* model &var = BL Year BL*Year / ddfm=kr residual outpm=resid;    /* original model */
model &var = BL BL*Year / ddfm=kr noint solution residual outpm=phresid;    /* run to get estimates in original model */
random Block ;
contrast 'Slope C vsH' bl*year 1 -1 0;
contrast 'Slope CvsL' bl*year 1 0 -1;
contrast 'Slope HvsL' bl*year 0 1 -1;
contrast 'Intercept CvsH' bl 1 -1 0;
contrast 'Intercept CvsL' bl 1 0 -1;
contrast 'Intercept HvsL' bl 0 1 -1;
run;
quit;

title3 'Parameter ESTIMATES for 2nd TEST - common slope, different intercept';
title4 "---------- &var ----------";
proc mixed data=loads method=type3 ;
  class block bl ;
  * model &var = BL Year         / ddfm=kr residual outpm=ph2resid;                   /* run when
    BL*year NS to test common slope = 0 */
  model &var = BL Year         / ddfm=kr noint solution residual outpm=phresid;    /* run to get
    estimates in common slope model */
  random Block ;
  * estimate 'Intercept C' intercept 1 bl 1 0 0;
  * estimate 'Intercept H' intercept 1 bl 0 1 0;
  * estimate 'Intercept L' intercept 1 bl 0 0 1;
  estimate 'Common Slope' year 1;
  contrast 'Intercept CvsH' bl 1 -1 0;
  contrast 'Intercept CvsL' bl 1 0 -1;
  contrast 'Intercept HvsL' bl 0 1 -1;
run;
%MEnd xxx;

%xxx(var=Vol );
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%xxx(var=redox );
%xxx(var=EC );
%xxx(var=NO3 );
%xxx(var=NH3 );
%xxx(var=PO4 );
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Appendix 2-2. Example SAS program for analysis of variance run as a macro and related data.

options mprint;

title '8 YEAR LEACHATE LOADS and FWM -- Richard McMullen --';
title2 'All elements Leachate';

data loads;
  infile 'F:\McMullen\Leachate\SAS\8yearANOVA\8yearTotals.csv' firstobs=2 DLM=,';
  truncover LRECL = 600 DSD;
  input BL $ Block Lys Vol pH redox EC NO3 NH3 PO4 DOC Al As B Ca
  Cd Co Cr Cu Fe K Mg Mn Na Ni TP Pb S Se Ti Zn fwmNO3 fwmNH3
  fwmPO4 fwmDOC fwmAl fwmAs fwmB fwmCa fwmCd fwmCo fwmCr fwmCu
  fwmFe fwmK fwmMg fwmMn fwmMo fwmNa fwmNi fwmP fwmPb fwmS fwmSe fwmTi
  fwmZn ;
  label BL = 'Broiler Litter';
run;

proc sort data=loads; by block lys;
run;
quit;

title3 'INITIAL DATA LISTING';
proc print data=loads noobs; by block;
id block;
  var lys bl Vol pH redox EC NO3 NH3 PO4 DOC Al As B Ca
  Cd Co Cr Cu Fe K Mg Mn Na Ni TP Pb S Se Ti Zn fwmNO3 fwmNH3
  fwmPO4 fwmDOC fwmAl fwmAs fwmB fwmCa fwmCd fwmCo fwmCr fwmCu
  fwmFe fwmK fwmMg fwmMn fwmMo fwmNa fwmNi fwmP fwmPb fwmS fwmSe fwmTi
  fwmZn ;
run ;
quit;

%Macro xxx(var=);

title3 'ANALYSIS OF VARIANCE';
title4 "---------- &var -----------";
proc mixed data=loads method=type3 ;
class block bl;
model &var = BL ;
random block;
lsmeans BL;
contrast 'Control vs High' bl 1 -1 0 ;
contrast 'Control vs Low' bl 1 0 -1 ;
contrast 'Low vs High' bl 0 1 -1 ;
run;

%MEnd xxx;

%xxx(var=Vol   );
%xxx(var=pH    );
%xxx(var=redox );
%xxx(var=EC    );
%xxx(var=NO3   );
%xxx(var=NH3   );
%xxx(var=PO4   );
%xxx(var=DOC   );
%xxx(var=Al    );
%xxx(var=As    );
%xxx(var=B     );
%xxx(var=Ca    );
%xxx(var=Cd    );
%xxx(var=Co    );
%xxx(var=Cr    );
%xxx(var=Cu    );
%xxx(var=Fe    );
%xxx(var=K     );
%xxx(var=Mg    );
%xxx(var=Mn    );
%xxx(var=Mo    );
%xxx(var=Na    );
%xxx(var=Ni    );
%xxx(var=TP    );
%xxx(var=Pb    );
%xxx(var=S     );
%xxx(var=Se    );
%xxx(var=Ti    );
%xxx(var=Zn    );
%xxx(var=fwmNO3 );
%xxx(var=fwmNH3 );
%xxx(var=fwmPO4 );
%xxx(var=fwmDOC );
%xxx(var=fwmAl );
%xxx(var=fwmAs );
%xxx(var=fwmB  );
%xxx(var=fwmCa );
%xxx(var=fwmCd );
%xxx(var=fwmCo );
%xxx(var=fwmCr );
%xxx(var=fwmCu );
%xxx(var=fwmFe );
%xxx(var=fwmK);
%xxx(var=fwmMg);
%xxx(var=fwmMn);
%xxx(var=fwmMo);
%xxx(var=fwmNa);
%xxx(var=fwmNi);
%xxx(var=fwmP);
%xxx(var=fwmPb);
%xxx(var=fwmS);
%xxx(var=fwmSe);
%xxx(var=fwmTi);
%xxx(var=fwmZn);
Note: The 8-yr data set is presented as transposed.

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<td>133.738</td>
<td>83.270</td>
<td>423.869</td>
<td>309.702</td>
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<tr>
<td>SumB kg/ha</td>
<td>6.166</td>
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<td>4.131</td>
<td>6.009</td>
<td>2.163</td>
<td>3.541</td>
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<td>401.905</td>
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<td>0.015</td>
<td>0.042</td>
<td>0.126</td>
<td>0.098</td>
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<td>0.386</td>
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<td>0.104</td>
<td>0.110</td>
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<td>0.048</td>
<td>0.048</td>
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<td>FWM Ca mg/L</td>
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<tr>
<td>FWM Cd mg/L</td>
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<td>0.001</td>
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<td>0.004</td>
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<td>0.005</td>
<td>0.006</td>
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<td>FWM Cu mg/L</td>
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<td>0.004</td>
<td>0.005</td>
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<td>8.589</td>
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<td>0.009</td>
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<td>0.004</td>
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<td>FWM Se mg/L</td>
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<td>0.003</td>
<td>0.011</td>
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<td>FWM Ti mg/L</td>
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Appendix 2-3. Example SAS program for Pearson’s correlation analysis.

options mprint;

title 'Annual Runoff LOADS and FWM -- Richard McMullen --';
title2 'All elements Runoff Correlations with Rainfall-mm and Runoff-mm';

data loads;
  infile 'F:\McMullen\RunoffSAS\Correlations\AnnualROHalfLimitCorrelations.csv' firstobs=2 DLM=',' truncover LRECL = 600 DSD;
  input Year BL $ lys Block Vol ROmm pH redox EC NO3 NH4 PO4 DOC Al As Ca Cd Cr Cu Fe K Mg Mn Na Ni TP S Se Zn fwmNO3 fwmNH4 fwmPO4 fwmDOC fwmAl fwmAs fwmCa fwmCd fwmCr fwmCu fwmFe fwmK fwmMg fwmMn fwmNa fwmNi fwmP fwmS fwmSe fwmZn rain;
  * year = 2002 + year;
  label BL = 'Broiler Litter';
run;

proc sort data=loads; by year block lys;
run;
quit;

title3 'INITIAL DATA LISTING';
proc print data=loads noobs; by year block;
  id year block;
  var lys bl ROmm pH redox EC NO3 NH4 PO4 DOC Al As Ca Cd Cr Cu Fe K Mg Mn Na Ni TP S Se Zn fwmNO3 fwmNH4 fwmPO4 fwmDOC fwmAl fwmAs fwmCa fwmCd fwmCr fwmCu fwmFe fwmK fwmMg fwmMn fwmNa fwmNi fwmP fwmS fwmSe fwmZn rain;
run cancel;
quit;

title3 'Correlations';
title4 "---------- ---------- ";

ods graphics on;
proc corr data=loads plots=matrix(histogram);
  var Year ROmm pH redox EC NO3 NH4 PO4 DOC Al As Ca Cd Cr Cu Fe K Mg Mn Na Ni TP S Se Zn fwmNO3 fwmNH4 fwmPO4 fwmDOC fwmAl fwmAs fwmCa fwmCd fwmCr fwmCu fwmFe fwmK fwmMg fwmMn fwmNa fwmNi fwmP fwmS fwmSe fwmZn rain;
run ;
ods graphics off;
run;
quit;

155
Appendix 2-4. Example SAS program for analysis of variance for annual and 8-yr cumulative above ground dry matter production run as a macro and related data.

options mprint;

title 'Annual Above Ground Biomass and 8-yr Cumulative AGB -- Richard McMullen --';
title2 'ANOVA blocks fixed';

data AGB;
infile 'F:\McMullen\Biomass\SAS\AGByear1thru8.csv' firstobs=3 DLM=',' truncover LRECL = 600 DSD;
input Lys BL $ Yr1 Yr2 Yr3 Yr4 Yr5 Yr6 Yr7 Yr8 CumAGB;
label BL = 'Broiler Litter';
run;

proc sort data=AGB; by bl lys;
run;
quit;

%Macro xxx(var=);

title3 'INITIAL DATA LISTING';
proc print data=AGB noobs; by bl lys;
id bl lys;
var Yr1 Yr2 Yr3 Yr4 Yr5 Yr6 Yr7 Yr8 CumAGB;
run;
quit;

%MEnd xxx;

%xxx(var=Yr1);
%xxx(var=Yr2);
%xxx(var=Yr3);

%Macros xxx(var=);
%xxx(var=Yr4);
%xxx(var=Yr5);
%xxx(var=Yr6);
%xxx(var=Yr7);
%xxx(var=Yr8);
%xxx(var=CumAGB);
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Chapter Three

Long-term Runoff and Runoff Water Quality Trends from a Broiler Litter-amended Udult in the Ozark Highlands
Abstract

The United States produced 8.4 billion broiler chickens (*Gallus gallus*) in 2012. Similarly, the state of Arkansas produced 1.0 billion broilers and an estimated 1.1 million Mg of broiler litter (BL). Repeated annual land applications of BL to pastures to increase forage yields has increased concerns for potential surface water contamination from runoff. The objective of this study was to determine long-term linear trends in runoff and runoff water quality under natural precipitation for 8 years from a Captina silt-loam soil (fine-silty, siliceous, active, mesic Typic Fragiudult) under forage management amended annually with BL at three application rates [0 (control), 5.6 (low), and 11.2 (high) Mg BL ha\(^{-1}\)] after having a history of BL amendments. Runoff was collected after each runoff-producing event during the 8-yr period (i.e., May 2003 to May 2011). Runoff pH, oxidation-reduction potential (ORP), electrical conductivity (EC), and soluble plant nutrients (i.e., NO\(_3\)-N, NH\(_4\)-N, PO\(_4\)-P, Ca, K, Mg, Na, and P), trace metals (i.e., As, Cd, Cr, Cu, Fe, Mn, Ni, Se, and Zn), and dissolved organic carbon (DOC) were measured. Annual flow-weighted-mean (FWM) concentrations and annual loads were determined. Average annual runoff, FWM runoff concentrations of Ca, Cd, Cu, Na, and Se, and all nutrient and metal loads increased over time, but were unaffected by BL application rate. Average annual runoff pH, EC, FWM runoff concentrations of NO\(_3\)-N, NH\(_4\)-N, PO\(_4\)-P, Cr, K, Mg, Mn, Ni, P, Zn, and DOC did not vary over time and were also unaffected by BL application rate. Average annual ORP and FWM As decreased over time and were unaffected by BL. Continued annual additions of BL at the high application rate increased average annual FWM runoff Fe (0.23 mg L\(^{-1}\)) concentrations relative to the unamended control (0.13 mg L\(^{-1}\)) and low (0.08 mg L\(^{-1}\)) BL treatments, but did not vary over time. Eight-year cumulative runoff loads of C, N, P, K, Ca, Mg, Mn, Cr, Cu, Fe, Ni, and Zn represented less than 2% of that applied in litter treatments,
while cumulative runoff loads of NO$_3$-N, Na, and Cd were between 3 and 58 % of that applied in BL. Cumulative runoff Se loads exceeded 100% of that applied in BL. A 205.6 mm rainfall event during a 42-hr period was responsible for 96 % of the total annual runoff during 2010, which was highly influential for the year, as well as for the entire 8-yr study, and emphasized the importance of long-term observational studies. Results indicated that pasturelands with a history of BL application may continue to release BL-derived As and Se at concentrations harmful to health regardless of current management practice long after litter application has ceased.
**Introduction**

The United States produced 8.4 billion broiler chickens (*Gallus gallus*) in 2012 (USDA-NASS, 2013). The top five broiler-producing states (i.e., Georgia, Alabama, Arkansas, North Carolina, and Mississippi) jointly produced 4.9 billion broilers (USDA-NASS, 2013), many of which were produced in relatively small geographic regions within each state. Management of the large quantities of waste associated with broiler production in relatively small geographic regions has raised environmental concerns regarding water quality. Referred to as broiler litter (BL), the waste is a mixture of excreta, feathers, feed, and bedding material, such as rice (*Oryza sativa* L.) hulls, saw dust, or straw, and is generated at a rate of 1.1 to 1.5 Mg BL per 1000 birds (UADACES, 2002). In 2012, Arkansas produced 1.0 billion broilers and an estimated 1.1 to 1.5 million Mg of BL, the majority of which was concentrated in the Ozark Highlands region (Major Land Resource Area 116A; UADACES, 2002; USDA-NRCS, 2006; USDA-NASS, 2013) of northwest Arkansas.

Broiler litter is a source of plant nutrients and is routinely surface-applied to pasture soils near broiler production facilities to increase yields of tall fescue (*Lolium arundinaceum* Shreb.) and other forages (Hileman, 1973; Huneycutt et al., 1988; Brye et al., 2010). Chemical composition and production rates of BL are variable and are influenced by regional practices, company practices, type of storage, amount and type of bedding, feed and feed additives, type of flooring, and number of flocks raised between cleanouts (Kunkle et al., 1981; Patterson et al., 1998; Mitchell and Donald, 1999; Hatten et al., 2001; Chamblee et al., 2002; Applegate et al., 2003; Garbarino et al., 2003). In the past, BL also contained trace metals, such as arsenic (As), cadmium (Cd), copper (Cu), selenium (Se), and zinc (Zn; van der Watt et al., 1994; Garbarino et al., 2003; Franzluebbers et al., 2004; Pirani et al., 2006; Menjoulet et al., 2009). Trace metals
like As originated from dietary supplements, such as 3-nitro-4-hydroxylarsonic acid (roxarsone) and 4-aminophenylarsonic acid (p-arsanilic acid), that were used to enhance growth by controlling coccidiosis, a disease caused by coccidian, a single-celled parasite of the intestine (Garbarino et al., 2003; Garbarino et al., 2009).

Repeated applications, sometimes for decades, of BL to pastures based solely on forage nitrogen (N) requirements have resulted in elevated soil concentrations of some nutrients and metals (Sharpley et al., 1993; Sharpley et al., 1994; Kingery et al., 1994; Kpomblekou-A et al., 2002; Sharpley et al., 2004; Daigh et al., 2009) because the rates of nutrient and metal additions were greater than the rates of forage uptake and removal. Similarly, long-term BL applications have been reported to increase soil pH (Vadas and Sims, 1998; Sharpley et al., 2004; Daigh et al., 2009) and soil organic carbon (C; Sharpley et al., 1993; Sharpley et al., 2004) relative to unamended soil.

Runoff from BL-amended pastures has the potential to contaminate nearby surface waters with BL-derived nutrients, which may lead to eutrophication. Not only does eutrophication of surface water reduce the aesthetics of natural environments, but eutrophication also has an economic effect by limiting industrial, commercial, recreational, and municipal water uses. Additionally, aquatic life, including fisheries, and drinking water may be affected. In karst regions, similar to the Ozark Highlands, impaired surface water may also rapidly enter groundwater, which may be used for drinking purposes. In karst regions, the formation of caves and preferential water pathways through soluble carbonate bedrock allows for the free exchange of water between surface water and groundwater. Common surface features associated with the free movement of water include loosing streams, sinkholes, springs and cave discharge areas. For these reasons, anthropogenic activities, including BL amendments to pasturelands in karst
regions, may contaminate surface and groundwater resources (MacDonald et al., 1976; Fetter, 2001a, b; Peterson et al., 2002; Scott and Ward, 2002; Graening and Brown, 2003; Stueber and Criss, 2005; USDA-NRCS, 2006).

For these reasons, the environmental fate of BL-derived nutrients in runoff must be studied. Because phosphorus (P) is one of the primary limiting nutrients in most water systems (Stumm and Morgan, 1996; Carpenter et al. 1998; Kalff, 2002; Schindler et al., 2008; Smith and Schindler, 2009) and because P accumulates in BL-amended soil, previous runoff studies have tended to focus on P.

The effect of BL application on runoff nutrient concentrations and losses has been widely explored within laboratory (Adeli et al., 2006; Vadas et al., 2004; Kleinman et al., 2002) and field studies (DeLaune et al., 2004a, b; DeLaune, 2002; Pote et al., 1999; Sauer et al., 1999; Moore et al., 1998; Edwards et al., 1997; Nichols et al., 1994; Edwards and Daniel, 1993) using simulated rainfall. Simulated rainfall studies are important because researchers are able to manipulate dependant variables, like rainfall intensity, duration, and frequency, which are difficult to control within natural environments. However, simulated rainfall studies, many times, report rainfall rates that are infrequent (i.e., storm events with return rates greater than 5 years) in nature. In contrast, watershed studies (Sharpley et al., 2008; Sharpley et al., 1992) use natural precipitation and stress the importance of storm events on runoff, but few studies have monitored small plots during naturally occurring precipitation (Menjoulet et al., 2009; Sistani et al., 2006; Wood et al., 1999) over extended periods of time. The long-term (i.e., > 5 yr) use of field plots to collect runoff in response to natural precipitation events would be well suited to determine annual runoff and runoff nutrient loss patterns over time. However, monitoring runoff
losses from natural precipitation events is labor intensive and expensive and is seldom reported in the literature.

Important results of simulated-rainfall-induced runoff studies include increased BL application rate increasing runoff concentrations and losses of BL constituents (DeLaune et al., 2004a; Sauer et al., 1999; Moore et al., 1998; Edwards and Daniel, 1993). Edwards and Daniel (1993) reported linear relationships relating increased runoff concentrations and losses of total nitrogen (N), ammonium-N (NH$_4$-N), nitrate-N (NO$_3$-N), total P, dissolved-reactive P (DRP), total suspended solids, and chemical oxygen demand (COD) to increased BL application rate. DeLaune et al. (2004a) reported similar results for DRP and Moore et al. (1998) reported similar BL effects for trace metal (i.e., As, Cu, Fe, K, Na, and Zn) concentrations within runoff.

Edwards and Daniel (1993) also reported a dilution effect pertaining to rainfall intensity and runoff. Runoff concentrations of total N, total P, DRP, and COD decreased as rainfall intensity increased, but losses increased. The increased runoff thus diluted concentrations, but carried more mass (Edwards and Daniel, 1993).

Few studies have evaluated long-term (i.e., > 5yr) runoff water quality from BL-amended soil under natural precipitation (Edwards et al., 1996; Vervoort et al., 1998; Wood et al., 1999; Pierson et al., 2001; Sistani et al., 2006; Menjoulet et al., 2009). Wood et al. (1999) conducted a 2-yr study of BL application rate and commercial fertilizer effects on seasonal runoff quality in Alabama in response to naturally occurring precipitation in a corn (Zea mays L.)-winter rye (Secale cereal L.) rotation cropping system. Results showed that increased BL application rate increased flow-weighted-mean (FWM) runoff concentrations of NO$_3$-N, NH$_4$-N, total P, and DRP, but results differed among seasons (Wood et al., 1999). Additionally, seasonal runoff losses of K, Mg and Mn were reported to increase with increased BL application rate, but results
were not consistent for both years (Wood et al., 1999). Sistani et al. (2006) evaluated the effect of alum-treated BL on runoff water from a sandy-loam soil with a 2-3% slope in Alabama during a 2-yr period. Results showed that runoff concentrations of NH$_4$-N, total P, soluble-reactive P, and particulate P were reduced when BL was treated with alum prior to land-application compared to non-alum-treated BL (Sistani et al., 2006). Vervoort et al. (1998) evaluated the effect of fresh and composted BL at 10 and 20 Mg ha$^{-1}$ yr$^{-1}$ and a mixture of fresh and composted BL applied in split applications on field-scale N and P runoff losses during a 2-yr period in Georgia. Vervoort et al. (1998) reported the greatest runoff DRP to occur during the winter and a BL treatment effect was observed for mean runoff DRP concentrations. Pierson et al. (2001) evaluated the amount of time that runoff P and NH$_4$-N concentrations remained elevated when two consecutive years of BL applications were ceased and then followed by two more consecutive years of inorganic-N fertilizer. Results showed decreasing runoff P and NH$_4$-N concentration trends from four different Ultisols with 6 to 8% slopes after BL applications ceased (Pierson et al., 2001). In northwest AR, Edwards et al. (1996) evaluated the effect of BL and ammonium nitrate (NH$_4$NO$_3$) applications on runoff water quality from paired small fields with histories of organic amendments. Runoff NO$_3$-N, NH$_3$-N, total N, PO$_4$-P, and COD concentrations were greatest shortly after BL application and then decreased over time during the nearly 3-yr study (Edwards et al., 1996).

In a concurrent study to the one reported here, Menjoulet et al. (2009) summarized the four years of BL application rate effects on seasonal and annual plant nutrient and trace metal runoff from a tall-fescue-dominated pasture soil in northwest AR in response to natural precipitation. Results indicated a lower than expected 4-yr cumulative runoff of 6 mm, which was < 0.1% of the cumulative rainfall for the same time period (Menjoulet et al., 2009). Broiler
litter application rate effects were observed for some plant nutrients and trace metals contained in runoff. Menjoulet et al. (2009) indicated that non-regulated, non-point runoff P concentrations from BL-amended soils may attain concentration levels of the same order of magnitude as regulated point sources.

Since broiler litter amendments to pasture soil in regions with intense broiler production have occurred for many years, and will likely continue to occur, it is important to identify long-term (i.e., > 5 yr) annual runoff trends that may affect water quality. Therefore, the objective of this study was to determine long-term linear trends in runoff and runoff water quality under natural precipitation for 8 years from a Captina silt-loam soil (fine-silty, siliceous, active, mesic Typic Fragiudult) under forage management amended annually with BL at three application rates [0 (control), 5.6 (low), and 11.2 (high) Mg BL ha\(^{-1}\)] after having a history of BL amendments. It was hypothesized that continued annual additions of BL would increase mean annual runoff pH and electrical conductivity (EC), as well as annual FWM concentrations and loads of BL-derived dissolved organic carbon (DOC), \(\text{NO}_3\)-N, \(\text{PO}_4\)-P, As, Ca, Cd, Cu, Mg, Na, P, Fe, Ni, Se, and Zn relative to an unamended control over the 8-yr study period. Similarly, it was hypothesized that continued annual additions of BL would have no effect on annual runoff and annual FWM concentrations and loads of BL-derived \(\text{NH}_4\)-N, K, and Mn relative to an unamended control. Additionally, it was hypothesized that 8 years of continued BL application would increase 8-yr FWM concentrations and cumulative loads of DOC, \(\text{NO}_3\)-N, \(\text{PO}_4\)-P, As, Ca, Cd, Cu, Mg, Na, P, Fe, Ni, Se, and Zn relative to an unamended control.
Materials and Methods

Site Description

As previously described by Pirani et al. (2006) in a concurrent and related drainage study, research was initiated in 2002 at the Agricultural Research and Extension Center in Fayetteville, Arkansas (36°05’49.18”N 94°10’44.65”W; elevation: 394.7 m). Six plots, 6-m long by 1.5-m wide, were selected based on similar soil pH {6.2 [standard error (SE) = 0.5]} and Mehlich-3 extractable P [210 (SE = 24) mg kg\(^{-1}\)] in the top 5 cm. All plots had previously received organic amendments in simulated rainfall studies (Edwards and Daniel, 1994; Nichols et al., 1994) and were equipped with rustproof, metal edging around each plot’s perimeter to prevent runon and collect all runoff water from each 5% west-to-east sloping plot (Pirani et al., 2006). Aluminum gutters were positioned on the down-slope edge of each plot to collect runoff (Pirani et al., 2006). Collection gutters were sloped to direct plot runoff into subsurface collection bottles and were covered to prevent direct precipitation from entering the runoff collection system (Menjoulet et al., 2009). The research area was mapped as a Captina silt loam (fine-silty, siliceous, active, mesic Typic Fragiuodult; USDA-NRCS, 2013). The soil textural class of the top 10 cm was silt loam with 63% silt and 5.5% clay (Pirani et al., 2006). Initially, ground cover was predominately tall fescue (Pirani, 2005), but in recent years other species have become increasingly common {i.e., clover \((\text{Trifolium spp.})\), Johnson grass \((\text{Sorghum halepense (L.) Pers.})\), and Bermuda grass \((\text{Cynodon dactylon L.})\)}.

The 30-yr mean annual air temperature and precipitation in Fayetteville, AR are 13.9 °C and 123 cm, respectively. The average date of the first frost is Oct 17 and the average date of the last frost is April 15 (NOAA, 2013).
Experimental Design and Treatments

The six field plots were arranged in a randomized complete block design with two replications to evaluate BL application rate effects on runoff and runoff chemistry. The minimum replication in this study was due to the initial (Pirani et al., 2006) and concurrent drainage studies (Pirani et al., 2007; McMullen, 2014) that limited replication because of costs associated with installation and maintenance of automated equilibrium-tension lysimeters (AETL; Masarik et al., 2004) under each plot. Broiler litter treatments in this study included three application rates imposed annually as a single application. A control treatment received no annual BL or inorganic fertilizer. A low (5.6 Mg dry litter ha$^{-1}$) and high (11.2 Mg dry litter ha$^{-1}$) BL rate treatment were established based on the current University of Arkansas Cooperative Extension Service’s BL application recommendations at the beginning of the study in 2002 (Pirani et al., 2006). However, BL application recommendations in Arkansas have since changed and are now based on the Phosphorus Index (DeLaune et al., 2004a, b; DeLaune et al., 2006). In order to maintain treatment consistency over time, BL treatment application rates remained unchanged throughout the study.

Broiler Litter Application and Analyses

Broiler litter was evenly applied by hand to plots once annually starting 30 April 2003. Application occurred approximately the first week of May each year thereafter. The BL used in this study had been collected from a single chicken house after production of 6 to 8 flocks, had an age ranging from 12 to 18 months, and had bedding material composed of an equal mixture of sawdust and rice hulls (Pirani et al., 2006). Broiler litter moisture content was determined prior to application each year so dry-weight-equivalent amounts of BL could be applied to plots.
receiving litter. Each year at the time of application, three BL sub-samples were collected and later characterized using procedures for manure analysis (Peters, 2003). Litter pH and EC were determined potentiometrically using a 1:2 BL:water mixture. Litter NO$_3$-N and NH$_4$-N concentrations were determined using a Skalar San Plus automated wet chemistry analyzer (Skalar Analytical B.V., The Netherlands) after extraction with potassium chloride. Total C and N were determined by high-temperature combustion using a LECO CN-2000 analyzer (LECO Corp., St. Josseph, MI). Total Ca, Cu, Fe, K, Mg, Mn, Na, P, S, and Zn were determined by inductively coupled, argon plasma (ICP) mass spectrometry (ICP; CIROS CCD model, Spectro Analytical Instruments, MA) after nitric acid digestion and treatment with hydrogen peroxide. Similarly, ICP was used to determine total recoverable Al, As, Cd, Cr, Ni, and Se after digestion with nitric and hydrochloric acids, hydrogen peroxide, and heat (USEPA, 1996).

**Runoff Collection and Analyses**

Beginning immediately after the 2003 BL application, runoff was collected from each plot after every runoff-producing precipitation event for an 8-yr period. Total runoff volume was measured for each plot for each event. Because runoff volumes varied by event, a hierarchal procedure was followed based on sample volume and importance of measurements. The first 20-mL of sample from each plot was used to determine runoff pH, EC, and oxidation-reduction potential (ORP) immediately upon returning to the lab and then discarded. A conductivity cell (VWR™ symphony™, No. 11388-382) was used for EC measurements and a combination pH/ORP electrode (Orion Triode™, No. 91-79 ORP) was used for pH and ORP measurements. Any remaining runoff subsample (up to 350 mL) was then filtered twice. A 1.6-µm glass microfiber filter (Whatman GFA- 1820-110; Whatman International Ltd., Maidston, England)
was used for the first filtering and a 0.45-μm Metricel® membrane filter (GN-6; Pall Life Sciences Corporation, Ann Arbor, MI) and vacuum pump were used for the second filtering. Once filtered, six, 20-mL aliquots were bottled. Three aliquots were acidified with one drop of 36 % (w/w) HCl for approximately every 10 mL of filtrate. The three remaining aliquots were left unacidified. Samples were then stored at 4 °C until analyses. Any runoff in excess of 370 mL was discarded.

Acidified aliquots were used to determine total dissolved As, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Se and Zn concentrations by ICP and NH₄-N and PO₄-P concentrations were determined using a Skalar San Plus automated wet chemistry analyzer. Unacidified aliquots were used to determine DOC using a Shimadzu Total Organic Carbon Analyzer (Model TOC-CSH, Shimadzu Scientific Instruments, Columbia, MD) and NO₃-N concentrations using the Skalar San Plus automated wet chemistry analyzer.

**Plot Management**

Plots were regularly monitored and maintained since 2002. Above-ground dry matter (DM) was removed using a bagging push mower eight times in 2003 and 2004 and four times (i.e., first week of May, June, July, and September) annually thereafter to a height of 9 cm. Prior to each mowing, two randomly selected 0.25-m² subsamples were hand collected and combined from each plot. Samples were dried for approximately 1 week at 55 °C in a forced-air drier and weighed for DM determination. Annual DM production was summed for each plot. Runoff collection gutters were cleaned prior to most precipitation events. Precipitation was monitored by two on-site rain gauges, a simple funnel-reservoir system and a micrometeorological weather
station that monitored wind speed, air temperature, relative humidity, total solar radiation, photosynthetically active radiation, and rainfall via a tipping bucket every 30 min.

Calculations

Flow-weighted mean runoff concentrations (mg L$^{-1}$) and runoff loads (kg ha$^{-1}$) were determined annually and cumulatively for the entire 8-yr period for each dissolved runoff constituent measured. For the purpose of this study, a year was designated as starting the day BL was applied in late April or early May of one year and ending the day before BL was re-applied in the following calendar year. Flow-weighted mean concentrations were calculated by dividing the total elemental mass collected in runoff during the time period of interest for each plot by the total runoff volume for the same time period. Loads were calculated by dividing the total elemental mass in runoff for a given plot during the time period of interest by the plot area (9.0 m$^2$). Similarly, annual and 8-yr mean runoff pH, EC, and ORP were calculated for each plot. Additionally, annual runoff hardness was calculated for study year 2010 (APHA, 2005).

Statistical Analysis

Analysis of covariance (ANCOVA) was used to identify BL application rate (covariate) effects on the relationship between annual runoff; mean annual runoff pH, EC, and ORP; and annual FWM runoff concentrations and runoff loads of NO$_3$-N, NH$_4$-N, PO$_4$-P, DOC, As, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Se, and Zn (dependent variables) over the 8-yr period (time, independent variable) using the PROC MIXED procedure in SAS (version 9.2; SAS Institute Inc., Cary, NC) while treating blocks as a random variable. Initially, a full model was used to test for different slopes among BL treatments. If slopes were similar, a second, reduced
model was used to test for different y-intercepts among BL treatments. When appropriate, slopes and y-intercepts were estimated and then separated using contrast statements at $\alpha = 0.05$. In cases where BL and time had no effect, treatment means, standard error of the mean, and the overall grand mean were calculated for informational purposes. Blocking variance was expressed as a percentage of total variance by dividing the blocking estimate for a given runoff property by the sum of the blocking and error estimates for that runoff property and multiplying by 100. In addition, relationships between annual precipitation, annual runoff, and annual runoff dependent variables were assessed by Pearson’s correlation analysis using the PROC CORR procedure in SAS. Similarly, simple linear regression was used to describe the relationship between annual precipitation and annual runoff.

Analysis of variance was used to identify BL application rate effects on annual and 8-yr cumulative above ground DM production; 8-yr cumulative runoff; 8-yr mean pH, EC, and ORP; 8-yr FWM runoff concentrations and cumulative runoff loads; and runoff mass losses expressed as a percentage of mass added as BL of NO$_3$-N, NH$_4$-N, PO$_4$-P, DOC, As, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Se, and Zn using the PROC MIXED procedure in SAS while treating blocks as a random variable. When appropriate, means were separated using a protected least significant difference (LSD) at $\alpha = 0.05$. In cases where BL had no effect, treatment means, the standard error of the mean, and the overall grand mean were calculated.

Results and Discussion

Pre-treatment Uniformity

Since field plots used in this study had received organic amendments prior to 2002, it was necessary to address pre-treatment plot uniformity. Three months prior to the initial BL
application in 2003, precipitation, runoff, drainage, and DM were monitored and soil samples were collected (Pirani et al., 2006). During this 3-mo period, 179 mm of precipitation fell, which was 98 mm below the 30-yr normal for the area during February, March, and April (Pirani et al., 2006). Mean 3-month cumulative runoff (Menjoulet et al., 2009) and drainage (Pirani et al., 2006) prior to the first BL application did not differ among pre-assigned BL treatments. In addition, mean runoff EC and FWM runoff concentrations of NO$_3$-N, NH$_4$-N, DOC, As, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Se, and Zn did not differ among pre-assigned BL treatments during the 3-mo period (Menjoulet et al., 2009). Similarly, mean soil leachate pH, EC, and ORP (Pirani et al., 2006) as well as FWM concentrations and loads of DOC (Pirani et al., 2006), NO$_3$-N, NH$_4$-N, Ca, K, Mg, Na, and P (Pirani et al., 2007) and FWM leachate concentrations of Mn, Ni, and Zn (Pirani et al., 2006) at the 90-cm depth did not differ among pre-assigned BL treatments. Additionally, 3-mo cumulative DM production did not differ among pre-assigned BL treatments (Pirani et al., 2006). Soil pH, EC, and organic matter concentration (Pirani et al., 2006); total recoverable soil Cd, Cu, and Zn; and Mehlich-3 extractable soil P, K, Ca, Mg, and Na (Pirani et al., 2007) did not differ among pre-assigned BL treatments for any 10-cm soil depth interval to a depth of 90 cm prior to the first BL application in 2003.

Based on the number of measured parameters that did not differ among pre-assigned BL treatments during the 3-mo period prior to the initial litter application in 2003, the plots in this study were assumed to be as uniform as reasonably could be expected (Brye and Pirani, 2006; Pirani et al., 2006; Pirani et al., 2007; Daigh et al., 2009; McDonald et al., 2009; Menjoulet et al., 2009; McMullen, 2014). Therefore, it was also assumed that any subsequent observed differences were due to the response to imposed BL treatments rather than to inherent differences among experimental plots (Pirani et al., 2006; Pirani et al., 2007; Menjoulet et al., 2009).
**Broiler Litter Composition**

The mean annual composition of the BL used throughout this study was 37.1% C, 4.4% N, 3.7% Ca, 3.5% K, and 2.2% P on a dry-weight basis and had 24% moisture by mass when applied (Table 3-1). Similar BL compositions have been previously reported in Pennsylvania (Kleinman et al., 2002), New York (Brock et al., 2007), Arkansas (Adams et al., 1994; Daigh et al., 2010; McMullen, 2014), and Nigeria (Agele et al., 2004). The mean annual BL C:N ratio was 8.4 averaged over the 8-yr period and suggested that BL decomposition by soil microorganisms would have been relatively quick with a likely net increase in soil N levels that would have promoted plant growth. The mean annual BL N:P ratio was 2.0 averaged over the 8-yr period and suggested that the BL used in this study supplied P in excess of plant growth requirements further suggesting that soil P concentrations would have also increased during this time period. In a concurrent and related study, Daigh et al. (2009) reported acid-recoverable, Mehlich-3-extractable, and water-soluble soil-P increased with the addition of BL over the first 5 years of this 8-yr study period. Mean annual inputs of plant nutrients and trace metals associated with BL treatments are summarized in Table 3-1.

**Precipitation**

During the 8-yr study, 410 individual precipitation events occurred with varying rates of intensity and duration. The most common form of precipitation was rainfall (95 %) followed by snow mixes (i.e., snow with rain, hail, sleet, and/or ice, 3 %), and snow (1 %). Averaged over all events, mean precipitation was 23.1 (SE = 1.3) mm per event. However, the histogram of precipitation events was skewed with a geometric mean (i.e., the \( n^{th} \) root of the product of the
data) of 12.0 mm per event. Classification of events based on precipitation amount revealed that 39 % of events were < 10 mm, 20 % were between 10 and 20 mm, 16 % were between 20 and 30 mm, and 9 % were between 30 and 40 mm.

Mean annual precipitation at the study site during the 8-yr period was 1178 mm (SE = 78), which was 4.4% below the 30-yr mean annual precipitation for Fayetteville, AR of 1232 mm (NOAA, 2013). Annual precipitation ranged from a low of 739 mm in 2005 to a high of 1508 mm in 2010 (Figure 3-1). Study years 2003, 2006, 2007, 2008, and 2009 were within ± 7% of the 30-yr mean, while study years 2004 and 2005 were 13 and 40% below the 30-yr mean, respectively. Study year 2010 exceeded the 30-yr mean by 22% and on April 26, 2011, 205.6 mm of rainfall occurred within a 42-hr period at the study site. For the 5-d period ending on the same day (April 26, 2011), NOAA (2014) reported 302 mm of rainfall for Fayetteville, AR, which resulted in extreme flooding in the area. During the current study, Fayetteville, AR set three record highs for monthly precipitation totals: March 2008 (study year 2007, 255 mm), October 2009 (study year 2009, 272 mm), and April 2011 (study year 2010, 388 mm). Similarly, record low total monthly precipitation also occurred in November 2007 (study year 2007, 9 mm) and August 2010 (study year 2010; 0.5 mm; NOAA, 2013).

**Above-ground Dry Matter**

Similar to previous studies (Huneycutt et al., 1988; Menjoulet et al., 2009; Brye et al., 2010; McMullen, 2014) and as would be expected, additions of BL increased DM relative to the unamended control (Table 3-2). Annual above-ground DM production ranged from a low of 4.9 Mg ha\(^{-1}\) in the unamended control in 2003 to a high of 21.6 Mg ha\(^{-1}\) in the high-litter treatment in 2010 (Table 3-2) when the greatest amount of annual precipitation occurred (Figure 3-1). Brye
et al. (2010) reported that, during the first 6 years of this study, DM production increased over time for both the low- and high-BL treatments, while DM in the unamended control did not change over time. McMullen (2014) reported that from 2008 to 2010, annual DM did not differ ($P > 0.05$) among BL treatments because plant speciation shifts increased DM variability within treatment conditions. In June 2006, Johnson grass started to encroach into the research area, but contributed little to DM. By July 2007, Johnson grass had become a prominent species contributing to DM in one plot and was observed in two other plots. By September 2008, Johnson grass was observed in five of six plots, and by November 2010 Johnson grass was observed in all plots.

Eight-year cumulative DM production in the high-BL treatment was greater ($P < 0.05$) in the unamended control, but did not differ ($P > 0.05$) from the low-BL treatment (Table 3-2). As would be expected, annual DM and annual precipitation were positively correlated ($r = 0.45$, $P < 0.01$) during the 8-yr study period, indicating that approximately 20% of the observed variability in annual DM could be attributed to changes in annual precipitation. Dry matter production and yield responses to irrigation and rainfall have been well-documented for forage grasses and crops (Jensen et al., 2001; Fay et al., 2003).

**Runoff and Runoff pH, ORP, and EC Trends over Time**

During the 8-yr study, individual runoff events varied greatly. Sometimes runoff was similar among all experimental units and sometimes runoff was quite different among experimental units for any given rainfall event. For example, steady, moderate rainfall over an extended period of time, or intense rainfall could produce relatively similar runoff among all experimental units, but smaller rainfall events might produce dissimilar runoff among
experimental units. Runoff was also closely related to antecedent soil moisture conditions prior to rainfall; with very dry or very wet soil conditions promoting runoff, especially when rainfall was intense. Although uncommon, intense initial rainfall and the subsequent initial runoff volume could enter the runoff collection system with sufficient force so as to disrupt the runoff collection funnel system in a manner that would cause runoff to not enter the subsurface collection bottle. It was at times like this, when runoff quantities were greater than normal, that runoff would collect within the polyvinyl chloride (PVC) pipe used to house the collection bottle below the soil surface and many times the bottle would float. In situations like this, runoff would be pumped from the PVC pipe and measured to determine runoff amount, but unless enough runoff entered the collection bottle, chemical analyses were not preformed on water samples from these instances due to insufficient collected water volumes.

During the 8-yr study, individual runoff events ranged from a low of zero during small precipitation events to a maximum of 205.6 mm following a 42-hr period of heavy rain in April of 2011 (i.e., study year 2010). The second largest runoff event was 3.1 mm, which occurred in response to 65 mm of rainfall. The third largest runoff event was 2.5 mm, which occurred in response to 85 mm of rainfall. Since runoff could vary greatly for a given rainfall event, descriptive statistics were calculated using experimental units (i.e., six observations per precipitation event). During the 8-yr study, 31.4 % of all precipitation events resulted in no runoff. Averaged over all experimental units and all precipitation events including events with no runoff, mean runoff was 0.17 (SE = 0.01) mm. Similar to individual precipitation events, the data were skewed with a geometric mean of zero.

To better describe actual runoff that occurred, all precipitation events that resulted in no runoff and the 205.6 mm runoff event were excluded from the data set. Averaged over all
experimental units and all runoff-producing precipitation events, mean runoff was 0.25 (SE = 0.01; geometric mean = 0.025) mm per runoff event. The resulting bimodal histogram of runoff depth was skewed with the first, skewed curve occurring at runoff amounts < 0.5 mm and the second, normally distributed curve occurring at runoff amounts > 1 mm. Classification of runoff events based on amount revealed that 75 % of runoff events were between 0 and 0.1 mm, 4 % were between 0.1 and 0.2 mm, 2 % were between 0.2 and 0.3 mm, and 1 % were between 0.3 and 0.4 mm runoff depth. The second, more normally distributed curve within the data set histogram accounted for 11 % of all runoff events and was centered between runoff depths of 1.2 and 2.1 mm with each 0.1 mm runoff depth interval not accounting for more than 2 % of all runoff events. Additionally, a plot of runoff amount as a function of precipitation amount suggested that a minimum of 10 mm of precipitation was required to initiate an appreciable amount of runoff. Furthermore, precipitation events with ≤ 55 mm of rain accounted for the majority runoff events with ≤ 0.2 mm of runoff. Furthermore, runoff events associated with the second, more normally distributed curve within the histogram were associated with rain events that had ≥ 80 mm of precipitation. The y-intercept estimate (0.037 mm) for the linear relationship between runoff depth and precipitation depth did not differ from zero (P = 0.06); although the slope (0.007 mm mm\(^{-1}\)) was significant (P < 0.01; R\(^2\) = 0.12). This relationship predicted that for every 1 mm of precipitation an estimated 0.007 mm of runoff would occur, which would be equivalent to collecting 63 ml of runoff in the current study.

Annual runoff ranged from a low of 0.2 mm in the unamended control between May 2005 and May 2006 (i.e., study year 2005) when annual precipitation was lowest to a maximum of 215 mm in both the control and high-litter treatments between May 2010 and May 2011 (i.e., study year 2010) when precipitation was greatest (Figures 3-1 and 3-2). Annual runoff, averaged
over BL treatments and expressed as a percentage of annual precipitation was 0.2, 0.2, 0.1, 0.1, 1.4, 1.7, 1.0, and 14.2 % for year 1 through year 8, respectively. Similarly, Menjoulet et al. (2009) reported that the 4-yr cumulative runoff was 0.1 %, expressed as a percentage of the 4-yr cumulative precipitation, during the first 4 years of the current study. In Alabama, in a corn-winter rye agroecosystem on a silty-clay soil with a 4 % slope, Wood et al. (1999) reported a seasonal maximum runoff of 50 % of precipitation during the winter and spring. In Georgia, Pierson et al. (2001) reported an annual runoff range of 11.9 to 21.2 % of annual precipitation from 0.75-ha grassland paddocks on four different Ultisols with 6 to 8 % slopes during a 4-yr period. In a concurrent drainage study, McMullen (2014) reported that annual drainage averaged 471 (SE = 51) mm yr⁻¹ or 40 % of annual precipitation during the same 8-yr period as the current study. Additionally, annual drainage, averaged over BL treatments and expressed as a percentage of annual precipitation, was 35.7, 71.3, 0.3, 75.9, 37.6, 31.1, 36.4, and 24.3 % for years 1 through year 8, respectively. These results indicated that 2.4 % of annual precipitation left the research site as runoff, 57.6 % left as evapotranspiration, and 40.0 % left as drainage below the 90-cm soil depth at this location. Additionally, these values may differ annually because of variances in annual precipitation. For example, in 2005, annual precipitation was low; consequently annual drainage was below normal. Similarly, in 2010, a large 42-hr rainfall event effectively increased annual runoff and likely decreased annual drainage.

Annual runoff increased over time \((P < 0.01; \text{Table 3-3})\) during the 8-yr period, but neither the slopes nor the \(y\)-intercepts for the linear relationships between runoff and time differed among BL treatments \((P > 0.05; \text{Table 3-3; Figure 3-2})\). Averaged over BL treatments, annual runoff increased 19 mm yr⁻¹ \(\text{(Table 3-4)}\) during the 8-yr period and was highly influenced by the high annual runoff observed during the last year of this 8-yr study. On April 24 and 25,
2011 (i.e., study year 2010), 205.6 mm of rainfall occurred at the study site, all of which was attributed to runoff and represented 96% of the total annual runoff in the eighth year of this study. During a 3-day period prior to this rainfall-runoff event, 142 mm of rainfall created antecedent soil moisture conditions approaching soil saturation that favored runoff. Similarly, NOAA (2014) reported 302 mm of rainfall during the 5-d period ending on April 26 with extreme flooding in Fayetteville, AR. As would be expected, annual runoff was positively correlated to annual precipitation ($r = 0.65; P < 0.01$; Table 3-5). Increased rainfall has been reported to increase runoff in simulated rainfall studies (Edwards and Daniel, 1993) and increased stream flow in watershed studies (Sharpley et al., 2008; Brion et al., 2010). Similar to the results of the current study, other researchers have also reported runoff to be unaffected by BL application rate (Edwards and Daniel, 1994; Pote et al., 2003; Menjoulet et al., 2009; Kibet et al., 2013; Lamba et al., 2013).

Mean annual runoff ORP decreased 6.3 mV yr$^{-1}$ (Table 3-4) during the 8-yr period, did not differ ($P > 0.05$) among BL application rates (Table 3-3; Figure 3-2), and was negatively correlated to both annual precipitation ($r = -0.61, P < 0.01$) and runoff ($r = -0.42, P < 0.01$, Table 3-5). The observed decrease in annual runoff ORP regardless of litter treatment may be related to the correlations of ORP with annual precipitation and runoff. Although precipitation is well aerated, runoff may become less aerated if allowed to sit because microbial activity will consume dissolved oxygen if nutrients are available for microbial growth. Although a strict protocol was followed regarding sample collection, during the spring season precipitation events could occur daily with the next day’s sample starting to be collected the moment collection bottles were changed. At these times runoff could potentially sit uncollected in the field for up to 24 hr. Similarly, Menjoulet et al. (2009) also reported that mean annual runoff ORP did not differ
among BL treatments and averaged 28.3 mV during the first four years of this study. Additionally, in a concurrent drainage study, McMullen (2014) reported no BL application rate effect on mean annual drainage ORP at the 90-cm soil depth, and annual drainage ORP also decreased during the same 8-yr period as the current study.

Annual runoff pH and EC did not differ among BL application rates ($P > 0.05$) or over time ($P > 0.05$; Table 3-3, Figure 3-2) and were summarized by averaging over all treatment conditions (i.e., grand means; Table 3-6). Mean annual runoff pH was 6.71 (SE = 0.04) and mean annual EC was 369 (SE = 17) µS cm$^{-1}$. Although it could be hypothesized that the annual additions of BL-derived base-forming cations and salts would increase runoff pH and EC over time, this was not observed in the current study. Since only 2.4% of annual rainfall left the research site as runoff, it may be assumed the remaining 97.6% of rainfall moved BL-derived cations and salts into the soil profile, thus not affecting runoff. Similarly, Nichols et al. (1994) reported no difference in runoff EC among BL or inorganic fertilizer, either incorporated or surface applied to soil. Menjoulet et al. (2009) also reported that mean annual and 4-yr mean runoff pH and EC did not differ among BL treatments. In contrast, Pote et al. (2003) reported runoff pH and EC to increase in BL-amended soil (5.6 Mg BL ha$^{-1}$) relative to an unamended control under both natural and simulated rainfall. Similar to the current study, McMullen (2014) reported no difference in mean annual drainage pH or EC among BL application rates at the 90-cm soil depth during a concurrent 8-yr study. In the current study, neither mean annual runoff pH nor EC were correlated ($P > 0.05$) to annual precipitation or runoff (Table 3-5). In 2005 and 2006, mean annual runoff pH was below the United States Environmental Protection Agency’s (USEPA) chronic continuous concentration for aquatic life (CCC) of 6.5 (USEPA, 2014). The CCC is an estimate of the greatest concentration of material or element in surface water to which
an aquatic ecosystem can be indefinitely exposed without adversely affecting that ecosystem’s community(ies).

**Runoff Concentration Trends over Time**

Annual FWM runoff Fe concentration differed \( P = 0.01 \) among BL treatments, but did not differ \( P > 0.05 \) over time (Table 3-3), indicating similar slopes that did not differ from zero for the linear relationship between annual FWM Fe concentration and time across BL treatments, but different y-intercepts for the same relationship across BL treatments (Figure 3-3). The y-intercepts were 0.13, 0.08, and 0.23 mg L\(^{-1}\) for the control, low, and high BL treatments, respectively. While the y-intercept for the unamended control did not differ from that for the low- \( P > 0.05 \) or the high-BL \( P > 0.05 \) treatments, the y-intercepts for the low- and high-BL treatments differed from one another \( P < 0.05 \). Annual additions of Fe and organic matter contained in BL may have interacted together to form Fe-chelated compounds that effectively increased the solubility of Fe and potentially increased Fe losses via runoff. Menjoulet et al. (2009) reported the same BL treatment effect when comparing the 4-yr FWM runoff Fe concentrations. Additionally, Moore et al. (1998) reported that runoff Fe concentrations increased linearly with increasing BL application rates, for both alum-treated and untreated BL, from a Captina silt loam soil with a 5 % slope under simulated rainfall in northwest AR. Similarly, Sauer et al. (1999) also reported greater runoff Fe concentrations with the addition of 6.7 Mg BL ha\(^{-1}\) (wet weight basis) during two simulated rain events relative to an unamended control. Edwards et al. (1997) reported that runoff Fe concentrations could be reduced with a 3-m wide vegetative filter strip using simulated rainfall. However, strip lengths > 3 m did not further reduce runoff Fe concentrations relative to the 3-m-length strip (Edwards et al., 1997).
the current study, mean annual FWM runoff Fe concentrations in the unamended control equaled the National Secondary Drinking Water Regulations’ standard of 0.3 mg L\(^{-1}\) (USEPA, 2009) in 2004, and the high-litter treatment exceeded the standard in 2004 and 2007 (Figure 3-3). Similarly, Menjoulet et al. (2009) reported that occasionally FWM runoff Fe concentrations during individual runoff events exceeded the same drinking water standard in the unamended control and high-BL treatments during the first four years of the current study.

Similar to mean annual runoff ORP, annual FWM runoff As concentrations decreased over time \((P < 0.01)\) during the 8-yr period (Figure 3-4), and neither the slope nor the y-intercept for the linear relationship between the annual FWM As concentration and time were affected by BL treatment \((P > 0.05;\) Table 3-3). Averaged across litter treatments, annual FWM runoff As concentrations decreased \((P < 0.01)\) 0.01 mg L\(^{-1}\) yr\(^{-1}\) during the 8-yr period (Table 3-4) and ranged from a low of < 0.01 mg L\(^{-1}\) in the unamended control treatment in study year 2006 to a maximum of 0.1 mg L\(^{-1}\) in the high-litter treatment in the second year of this study (i.e., study year 2004; Figure 3-4). Moore et al. (1998) reported runoff As concentrations to increase with increased BL application rate during the first simulated rainfall event immediately after BL application, but also reported that no differences among BL application rates occurred during the second rainfall event, 7 d later. Kibet et al. (2013) also reported increased runoff As concentrations in a BL-amended soil relative to an unamended control using simulated rainfall and monolith soil cores (61 x 61 x 61 cm). Similar to the current study, Menjoulet et al. (2009) reported that the 4-yr FWM runoff As concentrations did not differ among BL treatments. Similar to mean annual runoff ORP, annual FWM runoff As concentrations were negatively correlated with annual precipitation \((r = -0.34; P = 0.02;\) Table 3-5). Additionally, annual FWM
runoff As concentrations never exceeded the USEPA CCC of 0.15 mg L$^{-1}$ during the 8-yr study (USEPA, 2014).

In contrast, and assuming that runoff may enter groundwater in the karst development in the Ozark Highlands of northwest Arkansas, annual FWM runoff As concentrations exceeded the National Primary Drinking Water Regulations’ Maximum Contaminant Level (MCL) of 0.01 mg L$^{-1}$ (USEPA, 2009) in five of eight years during the study. As previously reported by Menjoulet et al. (2009) in a concurrent runoff study, annual FWM runoff As concentrations exceeded the MCL during the first three years of the study for litter-amended treatments and in years two and three for the unamended control (Figure 3-4). Additionally, the MCL for As was exceeded in 2009 for the unamended control and low-litter treatments and again in 2010 for all litter treatments. Kibet et al. (2013) also reported runoff As concentrations to exceed the MCL. Although direct runoff from BL-amended pastures is not used for drinking water and best management practices, such as buffer strips (Chaubey et al., 1995; Edwards et al., 1997), minimize direct runoff from entering streams, results presented here suggest that surface reservoirs used for drinking water could potentially be at risk for As contamination from BL-amended pastures. McMullen (2014) also reported decreasing annual FWM leachate As concentrations regardless of BL application rate during the same 8-yr period in a concurrent drainage study, with concentrations exceeding the MCL during the first three years. Additionally, Daigh et al. (2009) reported that acid-recoverable soil As differed among pre-assigned BL treatments and soil depth in a concurrent soil chemical properties study. Most importantly, these results suggest that pastures with histories of BL amendments that contained As may continue to release As into the environment long after BL applications have ceased, regardless of current BL application practices. Additionally, the long-term implications of As-
containing organic amendments to soil may require prolonged best management practices to limit As release.

In contrast to As, annual FWM runoff Ca, Cd, Cu, Na, and Se concentrations increased over time \((P < 0.01)\) during the 8-yr period (Figure 3-4), but neither the slopes nor the y-intercepts for the linear relationships between these annual FWM concentrations and time differed among BL treatments \((P > 0.05; \text{Table 3-3})\). Averaged over litter treatments, the y-intercept estimates for the linear relationships between annual FWM runoff Ca, Na, and Se concentrations and time differed from zero \((P < 0.05; \text{Table 3-4})\), while annual FWM Cd and Cu y-intercepts did not differ from zero \((P > 0.05, \text{Table 3-4})\). Annual FWM runoff Ca and Cd concentrations were positively \((P \leq 0.01)\) correlated with annual precipitation (Table 3-5), while annual FWM runoff Na and Se concentrations were positively \((P \leq 0.02)\) correlated with both annual precipitation and runoff (Table 3-5). In contrast, the annual FWM runoff Cu concentration was not correlated \((P > 0.05)\) with annual precipitation or runoff (Table 3-5).

Annual FWM runoff Ca and Na concentrations increased gradually during the 8-yr study and were probably related to natural soil weathering processes. However, annual FWM runoff Cd, Cu, and Se concentrations were low and even approached detection limits during the first four years of the study and then demonstrated a notable increase during the fifth year (Figure 3-4). Improved analytical instrumentation with lower detection limits for Se during the study may have contributed to the observed changes over time for annual FWM Se concentrations. Additionally, these increased concentrations coincided with Johnson grass intrusion into the research area. Plant root exudates are known to increase solubility of some plant nutrients and trace metals and may be responsible for the increased solubility and movement of Cd, Cu, and Se in runoff. In contrast to the current study, Moore et al. (1998) reported increased runoff Ca
concentrations in response to increasing BL application rates relative to an unamended control using simulated rainfall. Sauer et al. (1999) reported similar results for runoff Cu concentrations and Edwards et al. (1997) reported that vegetative filter strips up to 3-m in length reduced runoff Cu concentrations. In a concurrent drainage study, McMullen (2014) reported similar drainage trends in annual FWM leachate Ca, Cu, and Se concentrations, which did not differ among BL treatments and increased during the same 8-yr period as in the current study.

Annual FWM runoff Cd concentrations exceeded both the USEPA CCC of 0.25 µg L\(^{-1}\) and the Arkansas Pollution Control and Ecology Commission (APCEC) regulation CCC of 0.57 µg L\(^{-1}\) (i.e., calculated based on an estimated runoff water hardness of 44.9 mg equiv. CaCO\(_3\) L\(^{-1}\) for study year 2010, when hardness was greatest; APHA, 2005; APCEC, 2011) during six of the eight study years, and suggested that aquatic life could be adversely affected by dissolved Cd in runoff. If however runoff entered groundwater that might be used for drinking, the annual FWM runoff Cd concentrations never exceeded the MCL of 5 µg L\(^{-1}\) (USEPA, 2009) during the 8-yr study (Figure 3-4). Similarly, annual FWM runoff Cu concentrations exceeded the APCEC CCC of 0.006 µg L\(^{-1}\) (i.e., based on an estimated runoff hardness of 44.9 mg equiv. CaCO\(_3\) L\(^{-1}\); APCEC, 2011) in all years, but remained below the secondary drinking water standard of 1.0 mg L\(^{-1}\) (USEPA, 2009; Figure 3-4). However, annual FWM runoff Se concentrations exceeded the USEPA and APCEC joint CCC of 5 µg L\(^{-1}\) (USEPA, 2014; APCEC, 2011) and the primary drinking MCL of 0.05 mg L\(^{-1}\) (USEPA, 2009) during the last four years of the 8-yr study in all BL treatments (Figure 3-4).

Similar to runoff pH and EC, annual FWM runoff NO\(_3\)-N, NH\(_4\)-N, PO\(_4\)-P, DOC, Cr, K, Mg, Mn, Ni, P, and Zn concentrations did not differ among BL application rates \((P > 0.05)\) nor changed over time \((P > 0.05;\ Table 3-3;\ Figures 3-5 and 3-6)\) and were summarized by averaging
over all treatment conditions (Table 3-4). There are two main factors contributing to these results. First, BL treatment effects on runoff nutrient and metal concentrations are generally observed during the first runoff event following litter application. By the second runoff event BL treatment effects on runoff concentrations are normally not observed. Since annual FWM runoff concentrations were averaged over a year’s total runoff, any BL treatment effect that may have occurred during the first runoff event are effectively averaged with the remaining non-significant runoff events. The second factor relates to the availability of a nutrient or metal to be lost via runoff. Natural systems are efficient at retaining and cycling nutrients such as N thus reducing runoff losses. Additionally, P may be retained by soil as calcium phosphates, as hydrous oxides of Fe and Al, or by absorption to clay particles (Haseman et al., 1950). Similarly, soil solution pH may limit metal mobility of metals.

Similar runoff concentrations in response to natural precipitation have been reported by other researchers (Edwards et al., 1996; Vervoort et al., 1998; Wood et al., 1999; Pierson et al., 2001; Menjoulet et al., 2009). In contrast, Sistani (2006) reported runoff concentrations four times that observed in the current study for NO$_3$-N and P and 12 times that observed for NH$_4$-N in the current study. These result discrepancies between runoff studies in response to natural rainfall may have been related to the control treatment being amended with inorganic fertilizer in the Sistani (2006) study, while the control treatment in the current study was left completely unamended throughout the duration of the study. In the current study, annual FWM runoff PO$_4$-P, DOC, Cr, K, Mg, Mn, Ni, P, and Zn concentrations were negatively correlated ($P \leq 0.01$) with annual precipitation, but were not correlated ($P > 0.05$) with annual runoff (Table 3-5).

In contrast to the current study, simulated rainfall studies have reported that runoff from BL-amended pasture increased concentrations of total N, NH$_4$-N, NO$_3$-N, soluble-reactive P, K,
Mg, Mn (Sauer et al., 1999), and Zn (Sauer et al., 1999; Kibet et al., 2013) relative to an unamended control. Additionally, Moore et al. (1998) reported that runoff As, Cu, and Zn concentrations were highly correlated with soil organic C concentrations.

Similar to Cd, annual FWM runoff NO$_3$-N concentrations never exceeded the primary drinking water MCL of 10 mg L$^{-1}$ (USEPA, 2009) during the 8-yr study (Figure 3-5). Other researchers (Edwards et al., 1996; Vervoort et al., 1998; Sistani et al., 2006; Menjoulet et al., 2009) have also reported runoff NO$_3$-N concentrations to remain below this critical MCL for BL-amended pastures. However, Wood et al. (1999) reported that seasonal FWM runoff NO$_3$-N concentration was 10.5 mg L$^{-1}$ during the second season of corn production in Alabama with the addition of 18 Mg BL ha$^{-1}$, which differed from the low-litter (9 Mg BL ha$^{-1}$) and commercial fertilizer treatments, which were 3.9 and 4.1 mg L$^{-1}$, respectively. Surface water quality standards for aquatic life with regards to NH$_4$-N are based on water pH and temperature and the life stage and type of fish present. Since runoff temperature was not measured during the study, annual NH$_4$-N concentration limits could not be determined. However, air temperature exceeded 20 °C during the summer of all years and it may be assumed that air temperature and runoff temperature were positively correlated. During the summers of 2004 and 2006, runoff from either the low- or the high-litter treatments, respectively, may have exceeded the APCEC chronic criterion for NH$_4$-N in surface waters with fishes in early life stages of development if surface water temperatures exceeded 20 °C (APCEC, 2011; Figure 3-5). Annual FWM runoff Mn concentrations exceeded the secondary drinking water standard of 0.05 mg L$^{-1}$ (USEPA, 2009) during six of the eight years of the study (Figure 3-6). Annual FWM runoff Ni concentrations never exceeded either the EPA CCC of 52 µg L$^{-1}$ (USEPA, 2014) or the APCEC CCC of 79.8 µg L$^{-1}$ (i.e., with an estimated hardness of 44.9 mg equiv. CaCO$_3$ L$^{-1}$; APCEC, 2011) standards.
In Arkansas, point source discharges containing P have monthly average discharge permit limits for P, based on a facility’s designed daily flow. In this study, annual FWM runoff P concentrations exceeded the limit of 2.0 mg L\(^{-1}\) for facilities designed to discharge \(\geq 1\) million gallons per day (mgd) for all eight years of the study (Figure 3-6). Additionally, in 2005 runoff from the unamended control and high-litter treatments exceeded the limit of 5.0 mg L\(^{-1}\) (APCEC, 2011) used for smaller permitted facilities designed to discharge between 0.5 to < 1.0 mgd. These observations suggest that non-point runoff from pastures with a history of BL amendments may exceed P concentration discharge limits for regulated point sources regardless of the current BL application rate and may potentially lead to eutrophic conditions in receiving surface waters.

Annual FWM runoff Zn concentrations exceeded both the USEPA (2014) and APCEC (2011) aquatic life CCC of 0.12 and 0.05 mg L\(^{-1}\), respectively, during all eight years of this study, but remained below the secondary drinking water limit of 5.0 mg L\(^{-1}\) (USEPA, 2009; Figure 3-6). In a concurrent drainage study, McMullen (2014) reported that linear drainage trends in annual FWM leachate NO\(_3\)-N, PO\(_4\)-P, DOC, Cr, K, P, and Zn concentrations did not differ among BL treatments or change over time during the same 8-yr period as the current study.

In contrast to the current results, Wood et al. (1999) reported increased FWM runoff concentrations of NO\(_3\)-N and NH\(_4\)-N during the second season of corn production and NO\(_3\)-N during the second season of rye production to increase with increased BL rate. Wood et al. (1999) attributed the increased FWM runoff concentration of NO\(_3\)-N to be from increased NO\(_3\)-N contained within BL used during the second season, where the BL NO\(_3\)-N concentration increased from 12 to 363 mg NO\(_3\)-N kg\(^{-1}\) for year one to year two, respectively. Total-P and DRP runoff concentrations and losses were reported to increase with increasing BL application rate during the second season of corn production and were attributed to increased BL.
concentrations of P (Wood et al., 1999). Seasonal FWM runoff concentrations of Ca, K, Mg and Mn were reported to be greater in the 18 Mg BL ha\(^{-1}\) application rate treatment under corn production in either one or both corn seasons (Wood et al., 1999). Wood et al. (1999) concluded that FWM runoff concentrations of N, P, K, Mg, Mn, Cu and Zn were at levels that were adequate for algal growth in natural water bodies. Results from Wood et al. (1999) in conjunction with the current study’s results and similar reports from other pasture studies (Edwards et al., 1996; Vervoort et al., 1998; Pierson et al., 2001; Sistani et al., 2006; Menjoulet et al., 2009) highlight the importance of management practices on the potential environmental fate of BL-derived plant nutrients and trace metals. Litter application rates that may be appropriate for pastureland may not be appropriate for row-crop production systems. Additionally, trace metal uptake by crops that are later consumed may pose human health risks.

With the exception of annual FWM runoff Fe concentrations, BL application rate had no effect on annual FWM runoff nutrient and metal concentrations. Edwards and Daniel (1993) reported that runoff concentrations of BL-derived nutrients were greatest during the first simulated rainfall event, shortly after application, and then runoff concentrations decreased with successive simulated rainfalls thereafter. Additionally, Edward and Daniel (1993) reported that runoff losses increased as BL application rate increased. Similar results have been reported from simulated (Edwards and Daniel, 1994; Kleinman and Sharpley, 2003; Schroeder et al., 2004; Adeli et al., 2006) and natural precipitation (Edwards et al., 1996; 1997; Wood et al., 1999; Pierson et al., 2001). Because results of the current study were reported on an annual basis, BL treatment effects that may have occurred during the first rain event shortly after litter application were effectively averaged into all successive runoff events during the remainder of the year, when BL treatment effects were presumably minimal or nonexistent. Additional analyses using
the current study’s 8-yr data set could confirm the reports of Edwards and Daniel (1993) and others (Edwards et al., 1996; 1997; Wood et al., 1999; Pierson et al., 2001) by introducing time of rain event after BL application and runoff and precipitation amounts as covariates and allowing replication to occur across study years. An additional explanation for the lack of a litter rate effect on runoff nutrient concentrations in the current study relates to simulated runoff studies usually simulating runoff either immediately or within 24-hr of BL application. In natural runoff studies, the first runoff event may not occur for several weeks following BL application. In the current study, litter application was based on the current University of Arkansas Cooperative Extension Service’s BL application recommendations and was thus withheld from application if immediate rainfall was forecasted.

**Runoff Load Trends over Time**

Similar to runoff, all annual runoff plant nutrient and trace metal loads increased ($P \leq 0.01$) during the 8-yr period and did not differ ($P > 0.05$; Table 3-7) among BL treatments. The increased runoff load trends were directly related to increased annual runoff volumes during the study. In general, annual runoff nutrient and metal loads were lowest in the unamended control between May 2005 and May 2006 (i.e., study year 2005; Figures 3-7 through 3-9) when precipitation and runoff were low and greatest in either the unamended control or high-litter treatment between May 2010 and May 2011 (i.e., study year 2010) when precipitation and runoff were greatest. Similar to annual FWM runoff Na and Se concentrations, annual nutrient and metal loads were positively correlated ($P < 0.01$) with both annual precipitation and runoff, with annual runoff correlation coefficients being greater than corresponding annual precipitation coefficients for a given nutrient or metal (Table 3-8), suggesting that annual runoff might be a
better predictor of annual runoff loads than annual precipitation. In contrast, Kibet et al. (2013) reported that simulated-rainfall-induced runoff As and Zn loads were poorly correlated \( (r = 0.16) \) to runoff.

Pierson et al. (2001) reported that 60% of the total annual runoff DRP load occurred during three runoff events, within 12 days of BL application, during the first year of their study. Additionally, 61% of the total annual runoff DRP load occurred during two runoff events, within 3 days of application, during the following year (Pierson et al., 2001). Runoff associated with these events accounted for 33 and 28% of the annual runoff for the respective years (Pierson et al., 2001). In 2010 of the current study, 95.6% of the annual runoff occurred during a 42-hr rain event. This one event was highly influential for the annual nutrient and metal loads in 2010, as well as for the entire 8-yr study. If the influential runoff event was excluded from analyses, all 2010 annual runoff loads were lower than if the influential event had been included. Similarly, exclusion of the runoff event or the entire 2010 year from ANCOVA analyses reduced all slopes of all lines for the linear relationships between all annual runoff nutrient and metal loads and time.

Similarly, and to test if nonlinear analyses might better explain annual runoff load trends, an ANCOVA was performed using the \( \log_{10} \) transformed data of annual runoff and runoff loads. The upper and lower 95% confidence intervals (CI) for the y-intercepts and slopes were then calculated for BL treatments. Finally, the natural log of parameter estimates and CIs were then calculated and comparisons were made among BL treatments. Regardless of the analysis (i.e., exclusion of the influential runoff event, the entire influential year, or the transformed analyses while retaining influential data), all runoff load trend results were identical for annual runoff and all annual runoff nutrient and metal loads as previously reported in Tables 3-3 and 3-7. In some
cases and as would be expected, $P$-values were reduced using transformed data compared to non-transformed data. In all cases, runoff and runoff loads increased ($P < 0.01$) during the 7- or 8-yr period, and the slope of the linear or non-linear relationships between all annual runoff nutrient and metal loads and time did not differ ($P > 0.05$) among BL treatments. Menjoulet et al. (2009) reported that annual runoff loads of nutrients and metals appeared to be increasing during the first 4-yr period of the current study, but, in contrast to the current study, also reported BL treatment effects for annual runoff As, Fe, and Se loads during at least one of the first four years. Annual runoff loads were positively ($P < 0.01$; Table 3-8) correlated to annual runoff, which increased during the 8-yr study (Figure 3-2). It was this increasing trend in annual runoff, regardless of the influential 2010 year, that caused the increased runoff load trends in the current study. From 2003 to 2006, annual runoff ranged from 0.2 to 2.7 mm, and Menjoulet et al. (2009) reported that the 4-yr cumulative runoff averaged over BL treatments was 6.0 mm. From 2007 to 2009, annual runoff was greater than the first 4-yr period and ranged from 11.0 to 23.9 mm (Figure 3-2).

**Annual Blocking Variance**

During ANCOVA analyses, experimental blocks were treated as a random variable in order to attain insight regarding spatial variability at the research location. Variability among blocks accounted for < 1 % of the total observed variability in annual mean runoff (Table 3-9) and indicated that spatial variability in annual runoff was low at this study site. Similarly, variability among blocks accounted for < 2 % of the total variability in mean annual runoff pH, ORP, and annual FWM runoff concentrations and loads of most plant nutrients and trace metals (Tables 3-9 and 3-10). However, mean annual FWM runoff Fe and Mn concentration blocking
variances were 12.6 and 47.9 %, respectively, and indicated that the soil's natural spatial variability may warrant increased replication in future studies if inferences are to be made about annual FWM runoff Fe and Mn concentrations. The observed increases in block variability associated with annual FWM Fe and Mn runoff concentrations may also be related to previous runoff studies conducted on-site in which organic soil amendments were used prior to the initiation of the current study in 2003. Overall, blocking within the current study was successful with regards to number of blocks used. However, future research designs at this location may attain greater sensitivity to statistical analyses if a minimum of three blocks are used, especially if annual runoff Mn concentrations are to be studied. In a concurrent drainage study, McMullen (2014) reported variability in annual mean drainage at the 90-cm depth to be 38 %, suggesting that spatial variability in drainage is greater than that for runoff at this same location. Additionally, McMullen (2014) reported that variability among blocks was 53, 35, 13, and 62 % of the total variability in mean annual FWM leachate concentrations of PO$_4$-P, DOC, Fe, and P, respectively. Similar to the suggestions for annual runoff Mn concentrations during the current study, McMullen (2014) suggested that the soil's natural spatial variability may warrant an increase in the number of replications in future studies if inferences are to be made with regard to annual FWM leachate PO$_4$-P or P concentrations.

**Eight-year Mean Runoff Chemistry, FWM Concentrations, and Cumulative Loads**

After eight years of annual BL amendments, 8-yr cumulative runoff; 8-yr mean pH, ORP and EC; and 8-yr FWM leachate concentrations (Table 3-11) and runoff loads (Table 3-12) of nutrients and trace metals (i.e., NO$_3$-N, NH$_4$-N, PO$_4$-P, DOC, As, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Se, and Zn) did not differ ($P > 0.05$) among litter treatments. Additionally, and
with the exception of NO$_3$-N and total As and Cr, cumulative 8-yr runoff of total nutrients and trace metals expressed as a percentage of that applied in BL did not differ ($P > 0.05$) among the low- and high-litter treatments (Table 3-13). Cumulative 8-yr runoff losses of C, N, P, K, Ca, Mg, Mn, Zn, Cu, Fe, Ni, and Cr represented less than 2% of that applied in litter treatments, while cumulative 8-yr runoff losses of NO$_3$-N, NH$_4$-N, Na, and As represented between 2 and 14% of that applied in BL. Additionally, 8-yr cumulative runoff losses of Se and Cd exceeded 200 and 30% of that applied in BL, respectively. With the exception of NO$_3$-N, Se, and Cd, cumulative 8-yr runoff losses were low primarily because natural systems are efficient at retaining and cycling nutrients, soils have a natural buffering capacity for nutrients, and soil pH can limit mobility of metals. Cumulative NO$_3$-N losses appear elevated because the value is expressed as a percent of NO$_3$-N applied in litter. Since N compounds may undergo rapid transformations, a more appropriate measure of N loss would be total N, which was ≤ 2 %. The elevated cumulative Se and Cd losses may be related to: 1) increased losses associated with increased precipitation and runoff volumes during the later part of the study, 2) Johnson grass intrusion into the research area during the later part of the study or, 3) a combination the two.

Vervoort et al. (1998) also reported total N and P losses from 0.45-ha fields in Georgia to be < 1 % of that applied during a 2-yr period under natural precipitation. Similar to the results of the current study, Menjoulet et al. (2009) reported that cumulative 4-yr runoff losses of DOC, NO$_3$-N, PO$_4$-P, As, Ca, Cu, Mg, Na, P, Ni, Se, and Zn did not differ among BL treatments. In a concurrent drainage study, McMullen (2014) also reported that 8-yr drainage losses of NO$_3$-N, NH$_4$-N, PO$_4$-P, DOC, As, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Se, and Zn did not differ among BL treatments and that NH$_4$-N, C, N, P, and Cu losses were < 2 % of that applied in litter, while Cd and Se losses exceeded the amounts added in BL.
The elevated mass losses (i.e., > 200 % of that applied in the litter) for Se reported in the current study indicated that runoff loads of Se from BL-amended pasture soil with a history of organic amendments may eventually release Se into the environment via runoff, possibly because of increased precipitation and associated runoff (Table 3-8). Since BL treatment had no effect ($P > 0.05$) on annual and 8-yr cumulative nutrient and metal runoff loads (Tables 3-7 and 3-12), but Se leaching percents exceeded 200% of that applied, percent losses corrected for runoff losses from the unamended control were calculated for all plant nutrients and trace metals (Table 3-13) by subtraction of the control treatment’s 8-yr cumulative load from litter-treated 8-yr cumulative runoff loads within the same experimental block. Similar to 8-yr cumulative percent losses, cumulative 8-yr runoff of total nutrients and trace metals corrected for the unamended control losses and expressed as a percentage of that applied in BL did not differ ($P > 0.05$) among the low- and high-litter treatments for any plant nutrient or trace metal (Table 2-13). A negative corrected percent loss indicates that a nutrient or metal loss had originated from a source other than that applied in BL during the current study. Since corrected runoff Se losses were similar to zero, and since 8-yr Se losses exceeded 100 %, then Se contained in runoff originated from Se already in the soil prior to the current study. These results suggest that soil with a history of organic amendments that contained appreciable amounts of Se could potentially pose environmental risks relating to runoff water quality long after such amendments have ceased, regardless of any current or future litter applications. Similar conclusions were made by McMullen (2014) with regards to drainage losses of Cd and Se at the same location during the same time period.
Environmental Concerns

Based on the current 8-yr observational study and with the exception of annual FWM runoff Fe concentrations, long-term annual runoff patterns expressed in terms of annual FWM concentrations and loads of nutrients and metals did not differ among BL application rates in pasture land with a history of litter amendments. During the course of the current study, annual FWM runoff Ca, Cd, Cu, Na, and Se concentrations increased and Cd and Se annual concentrations exceeded the National Primary Drinking Water Regulations’ MCL of 5 µg L\(^{-1}\) and 0.05 mg L\(^{-1}\), respectively. Annual FWM runoff Cd concentrations exceeded the MCL during seven of the eight years and Se exceeded the MCL during four of the eight years. In contrast, annual FWM runoff As concentrations decreased during the study, but still exceeded the MCL of 0.01 mg L\(^{-1}\) during five of the eight years. Annual FWM runoff Fe concentrations differed among litter treatments and exceeded the secondary MCL of 0.3 mg L\(^{-1}\) during two of the eight years, while NO\(_3\)-N and Cu concentrations remained below the secondary MCL of 10 and 1.0 mg L\(^{-1}\), respectively, throughout the entire study. Although runoff from this study site may have had environmentally sensitive concentrations of metals that could potentially enter groundwater in karst regions, many locations where BL is applied to pastures use best management practices to reduce runoff contaminants and to minimize direct runoff from entering surface and groundwater.

Annual runoff, runoff loads, and precipitation were all positively correlated to each other and all increased during the 8-yr study period. The observed increasing linear trends in nutrient and metal losses over time were highly influenced by one rainfall event that occurred during a 42-hr period in the last year of the study, when most of the losses occurred. However, the observed increasing linear trends in nutrient and metal losses over time remained positive when
the influential data was excluded from analyses. Although management practices attempt to account for most rainfall scenarios, it may be necessary to alter practices to account for worst-case rainfall events. As climate change increases storm-size-return-rate variances, it may be necessary to be better prepared for worst-case scenarios as they may become more frequent in nature. Additionally, the occurrence of one influential rain event within an 8-yr period also highlights the importance of long-term observational studies and their ability to identify influential, but rare events that may contribute to the potential movement of contaminants within the environment.

**Summary and Conclusions**

Based on eight years of continuous monitoring of nutrient and metal losses from BL-amended pasture land with a history of organic amendments and under naturally occurring precipitation, annual FWM runoff Fe concentration was the only nutrient affected by litter application rate. Similarly, 8-yr cumulative runoff loads corrected for runoff losses from the unamended control provided evidence that Se from previous litter amendments prior to the current study were still increasing Se runoff losses even when BL amendments had ceased for more than eight years and furthermore emphasized the importance of long-term observational studies especially with regards to trace metals.

Correlation coefficients among annual precipitation, runoff, and runoff loads were all positive. Additionally, all annual runoff nutrient and metal loads increased over the 8-yr period and were highly influenced by one large rainfall event that occurred during the last year of the study. Mean annual runoff and annual FWM concentrations of Ca, Cd, Cu, Na, and Se increased during the 8-yr period. Mean annual runoff pH, EC, and FWM concentrations of NO₃-N, NH₄-
N, PO₄-P, DOC, Cr, K, Mg, Mn, Ni, P, and Zn were unchanged during the 8-yr period. Additionally, annual runoff FWM As concentration was the only nutrient or metal monitored that increased during the 8-yr study.

The lack of evidence to support BL-induced runoff effects in the current study can be attributed to four factors. First, FWM concentrations expressed on an annual basis could have diluted any BL-related effects that may have occurred shortly after litter application, but that did not persist throughout the remainder of the year. Second, runoff was collected in response to naturally occurring precipitation instead of simulated rainfall, which is commonly applied in quantities greater than normal precipitation. Third, pre-2002 organic soil amendments to the research plots effectively overwhelmed runoff chemistry changes that could have occurred in response to the current study’s imposed BL treatments. Fourth, limited replication within the study reduced statistical strength.
References


Table 3-1. Mean annual broiler litter (BL) composition and constituent added in low- (5.6 Mg ha\(^{-1}\)) and high- (11.2 Mg ha\(^{-1}\)) litter treatments over an 8-yr period to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas. Annual mean maxima and minima are provided as an indication of parameter range.

<table>
<thead>
<tr>
<th>Litter Property</th>
<th>Mean Annual Composition</th>
<th>Mean Maximum</th>
<th>Mean Minimum</th>
<th>Litter Rate kg ha(^{-1})</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moisture (g g(^{-1}))</td>
<td>0.24</td>
<td>0.27</td>
<td>0.21</td>
<td></td>
</tr>
<tr>
<td>pH</td>
<td>8.4</td>
<td>8.8</td>
<td>8.0</td>
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<tr>
<td>EC(^+) (dS m(^{-1}))</td>
<td>11.9</td>
<td>14.8</td>
<td>9.8</td>
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<tr>
<td>NO(_3)-N (mg kg(^{-1}))</td>
<td>207</td>
<td>513</td>
<td>38</td>
<td>1.1</td>
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<tr>
<td>NH(_4)-N (mg kg(^{-1}))</td>
<td>4640</td>
<td>7183</td>
<td>2877</td>
<td>26.0</td>
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</table>

Total Elements

<table>
<thead>
<tr>
<th>Element</th>
<th>Mean Annual Composition</th>
<th>Mean Maximum</th>
<th>Mean Minimum</th>
<th>Litter Rate kg ha(^{-1})</th>
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<tbody>
<tr>
<td>C (%)</td>
<td>37.1</td>
<td>39.5</td>
<td>33.9</td>
<td>2078</td>
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<tr>
<td>N (%)</td>
<td>4.4</td>
<td>5.3</td>
<td>4</td>
<td>246</td>
</tr>
<tr>
<td>P (%)</td>
<td>2.2</td>
<td>2.6</td>
<td>1.6</td>
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<tr>
<td>K (%)</td>
<td>3.5</td>
<td>4.4</td>
<td>2.9</td>
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<tr>
<td>Ca (%)</td>
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<td>4.4</td>
<td>2.9</td>
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<tr>
<td>Mg (%)</td>
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<td>0.8</td>
<td>0.6</td>
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<tr>
<td>S (%)</td>
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<td>1.6</td>
<td>0.6</td>
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</tr>
<tr>
<td>Na (mg kg(^{-1}))</td>
<td>9098</td>
<td>16094</td>
<td>3857</td>
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<tr>
<td>Al (mg kg(^{-1}))</td>
<td>347</td>
<td>558</td>
<td>243</td>
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<tr>
<td>Fe (mg kg(^{-1}))</td>
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<td>613</td>
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<td>Zn (mg kg(^{-1}))</td>
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<td>Cu (mg kg(^{-1}))</td>
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<tr>
<td>B (mg kg(^{-1}))</td>
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<td>60.9</td>
<td>46.5</td>
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<tr>
<td>Ni (mg kg(^{-1}))</td>
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<td>16.1</td>
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<tr>
<td>Cd (mg kg(^{-1}))</td>
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<td>0.60</td>
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</tr>
<tr>
<td>Cr (mg kg(^{-1}))</td>
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<td>15.6</td>
<td>3.1</td>
<td>0.04</td>
</tr>
<tr>
<td>As (mg kg(^{-1}))</td>
<td>26.8</td>
<td>39.9</td>
<td>19</td>
<td>0.15</td>
</tr>
<tr>
<td>Se (mg kg(^{-1}))</td>
<td>3.5</td>
<td>7.3</td>
<td>1.6</td>
<td>0.019</td>
</tr>
</tbody>
</table>

\(^+\)EC, electrical conductivity.
Table 3-2. Broiler litter application rate (control, 0 Mg ha$^{-1}$; low, 5.6 Mg ha$^{-1}$; and high, 11.2 Mg ha$^{-1}$) effects on mean annual above-ground dry matter and 8-yr cumulative production.

<table>
<thead>
<tr>
<th>Time Period †</th>
<th>P-value</th>
<th>Mean Dry Matter (Mg ha$^{-1}$)</th>
<th>Control</th>
<th>Low</th>
<th>High</th>
</tr>
</thead>
<tbody>
<tr>
<td>2003‡</td>
<td>0.04</td>
<td>4.9a#</td>
<td>8.7ab</td>
<td>12.0b</td>
<td></td>
</tr>
<tr>
<td>2004‡</td>
<td>0.01</td>
<td>5.6a</td>
<td>9.2b</td>
<td>12.2c</td>
<td></td>
</tr>
<tr>
<td>2005§</td>
<td>0.04</td>
<td>5.3a</td>
<td>7.6a</td>
<td>11.4b</td>
<td></td>
</tr>
<tr>
<td>2006§</td>
<td>0.02</td>
<td>5.3a</td>
<td>11.0b</td>
<td>11.7b</td>
<td></td>
</tr>
<tr>
<td>2007¶</td>
<td>&lt; 0.01</td>
<td>5.6a</td>
<td>10.4b</td>
<td>15.3c</td>
<td></td>
</tr>
<tr>
<td>2008</td>
<td>0.19</td>
<td>9.5a</td>
<td>13.5a</td>
<td>16.1a</td>
<td></td>
</tr>
<tr>
<td>2009</td>
<td>0.11</td>
<td>10.4a</td>
<td>15.5a</td>
<td>20.8a</td>
<td></td>
</tr>
<tr>
<td>2010</td>
<td>0.18</td>
<td>12.0a</td>
<td>12.6a</td>
<td>21.6a</td>
<td></td>
</tr>
<tr>
<td>8-yr Cumulative</td>
<td>0.05</td>
<td>58.7a</td>
<td>88.4ab</td>
<td>121.0b</td>
<td></td>
</tr>
</tbody>
</table>

† Study years are designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. For example 2003 represents the time period from May 2003 to April 2004.

‡ Data for 2003 and 2004 were taken from Pirani (2005).
§ Data for 2005 and 2006 were taken from Menjoulet et al. (2009).
¶ Data for 2007 were taken from Daigh et al. (2009).
# Means in the same row followed by different letters are significantly different (P < 0.05).
Table 3-3. Analysis of covariance summary of the effects of broiler litter (BL) application rate, time (Year), and their interaction on the linear relationship between runoff properties and flow-weighted mean concentrations and time.

<table>
<thead>
<tr>
<th>Runoff Property</th>
<th>Source of Variance</th>
<th>BL§</th>
<th>Year¶</th>
<th>BL x Year#</th>
</tr>
</thead>
<tbody>
<tr>
<td>Runoff</td>
<td>P-value</td>
<td>0.99</td>
<td>0.99</td>
<td>1.00</td>
</tr>
<tr>
<td>pH</td>
<td>0.93</td>
<td>0.48</td>
<td>0.51</td>
<td></td>
</tr>
<tr>
<td>ORP</td>
<td>0.98</td>
<td>&lt; 0.01</td>
<td>0.66</td>
<td></td>
</tr>
<tr>
<td>EC†</td>
<td>0.53</td>
<td>0.07</td>
<td>0.38</td>
<td></td>
</tr>
</tbody>
</table>

Concentrations

<table>
<thead>
<tr>
<th></th>
<th>BL§</th>
<th>Year¶</th>
<th>BL x Year#</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO₃-N</td>
<td>0.33</td>
<td>0.83</td>
<td>0.46</td>
</tr>
<tr>
<td>NH₄-N</td>
<td>0.48</td>
<td>0.76</td>
<td>0.64</td>
</tr>
<tr>
<td>PO₄-P</td>
<td>0.20</td>
<td>0.86</td>
<td>0.77</td>
</tr>
<tr>
<td>DOC‡</td>
<td>0.98</td>
<td>0.19</td>
<td>0.98</td>
</tr>
<tr>
<td>As</td>
<td>0.54</td>
<td>&lt; 0.01</td>
<td>0.43</td>
</tr>
<tr>
<td>Ca</td>
<td>0.72</td>
<td>&lt; 0.01</td>
<td>0.22</td>
</tr>
<tr>
<td>Cd</td>
<td>0.99</td>
<td>&lt; 0.01</td>
<td>0.95</td>
</tr>
<tr>
<td>Cr</td>
<td>0.98</td>
<td>0.11</td>
<td>0.94</td>
</tr>
<tr>
<td>Cu</td>
<td>0.95</td>
<td>&lt; 0.01</td>
<td>0.79</td>
</tr>
<tr>
<td>Fe</td>
<td>0.01</td>
<td>0.51</td>
<td>0.81</td>
</tr>
<tr>
<td>K</td>
<td>0.93</td>
<td>0.29</td>
<td>0.98</td>
</tr>
<tr>
<td>Mg</td>
<td>0.76</td>
<td>0.71</td>
<td>0.96</td>
</tr>
<tr>
<td>Mn</td>
<td>0.53</td>
<td>0.10</td>
<td>0.99</td>
</tr>
<tr>
<td>Na</td>
<td>0.51</td>
<td>&lt; 0.01</td>
<td>0.86</td>
</tr>
<tr>
<td>Ni</td>
<td>0.63</td>
<td>0.41</td>
<td>0.81</td>
</tr>
<tr>
<td>P</td>
<td>0.32</td>
<td>0.88</td>
<td>0.68</td>
</tr>
<tr>
<td>Se</td>
<td>0.94</td>
<td>&lt; 0.01</td>
<td>0.89</td>
</tr>
<tr>
<td>Zn</td>
<td>0.90</td>
<td>0.34</td>
<td>0.84</td>
</tr>
</tbody>
</table>

† EC, electrical conductivity.
‡ DOC, dissolved organic carbon.
§ Test for different y-intercepts among BL treatments with common slope.
¶ Test if common slope is different than zero.
# Test for different slopes among BL treatments.
†† P ≤ 0.05 are indicated in bold.
Table 3-4. Summary of common intercept and slope estimates for the linear relationships between annual runoff properties and time. Soil was amended once annually with broiler litter (BL) at three application rates (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) for an 8-yr period.

<table>
<thead>
<tr>
<th>Runoff Property</th>
<th>Intercept</th>
<th>P-value(\ddagger)</th>
<th>Slope</th>
<th>R(^2)(\ddagger)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Runoff (mm)</td>
<td>-52.5</td>
<td>&lt; 0.01</td>
<td>19.2</td>
<td>0.41</td>
</tr>
<tr>
<td>ORP(\dagger) (mV)</td>
<td>40.2</td>
<td>&lt; 0.01</td>
<td>-6.3</td>
<td>0.34</td>
</tr>
</tbody>
</table>

Concentrations (mg L\(^{-1}\))

<table>
<thead>
<tr>
<th>Element</th>
<th>Intercept</th>
<th>P-value(\ddagger)</th>
<th>Slope</th>
<th>R(^2)(\ddagger)</th>
</tr>
</thead>
<tbody>
<tr>
<td>As</td>
<td>0.04</td>
<td>&lt; 0.01</td>
<td>-0.01</td>
<td>0.17</td>
</tr>
<tr>
<td>Ca</td>
<td>3.3</td>
<td>0.02</td>
<td>1.8</td>
<td>0.53</td>
</tr>
<tr>
<td>Cd</td>
<td>&lt; 0.01</td>
<td>0.84</td>
<td>&lt; 0.01</td>
<td>0.37</td>
</tr>
<tr>
<td>Cu</td>
<td>0.01</td>
<td>0.09</td>
<td>&lt; 0.01</td>
<td>0.19</td>
</tr>
<tr>
<td>Na</td>
<td>1.6</td>
<td>&lt; 0.01</td>
<td>0.05</td>
<td>0.34</td>
</tr>
<tr>
<td>Se</td>
<td>-0.1</td>
<td>&lt; 0.01</td>
<td>0.06</td>
<td>0.69</td>
</tr>
</tbody>
</table>

Loads (g ha\(^{-1}\))

<table>
<thead>
<tr>
<th>Element</th>
<th>Intercept</th>
<th>P-value(\ddagger)</th>
<th>Slope</th>
<th>R(^2)(\ddagger)</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO(_3)-N</td>
<td>-198</td>
<td>0.01</td>
<td>75</td>
<td>0.41</td>
</tr>
<tr>
<td>NH(_4)-N</td>
<td>-560</td>
<td>0.01</td>
<td>223</td>
<td>0.37</td>
</tr>
<tr>
<td>PO(_4)-P</td>
<td>-1052</td>
<td>0.03</td>
<td>392</td>
<td>0.27</td>
</tr>
<tr>
<td>DOC(\ddagger)</td>
<td>-4054</td>
<td>0.01</td>
<td>1635</td>
<td>0.42</td>
</tr>
<tr>
<td>As</td>
<td>-6.1</td>
<td>0.01</td>
<td>2.3</td>
<td>0.36</td>
</tr>
<tr>
<td>Ca</td>
<td>-7911</td>
<td>&lt; 0.01</td>
<td>2879</td>
<td>0.40</td>
</tr>
<tr>
<td>Cd</td>
<td>-0.85</td>
<td>&lt; 0.01</td>
<td>0.32</td>
<td>0.45</td>
</tr>
<tr>
<td>Cr</td>
<td>-1.2</td>
<td>0.01</td>
<td>0.5</td>
<td>0.40</td>
</tr>
<tr>
<td>Cu</td>
<td>-10.4</td>
<td>&lt; 0.01</td>
<td>4.1</td>
<td>0.47</td>
</tr>
<tr>
<td>Fe</td>
<td>-83.2</td>
<td>0.12</td>
<td>30.2</td>
<td>0.16</td>
</tr>
<tr>
<td>K</td>
<td>-2662</td>
<td>0.04</td>
<td>1326</td>
<td>0.38</td>
</tr>
<tr>
<td>Mg</td>
<td>-815</td>
<td>0.01</td>
<td>305</td>
<td>0.38</td>
</tr>
<tr>
<td>Mn</td>
<td>-15.9</td>
<td>0.15</td>
<td>6.2</td>
<td>0.15</td>
</tr>
<tr>
<td>Na</td>
<td>-3343</td>
<td>0.01</td>
<td>1177</td>
<td>0.36</td>
</tr>
<tr>
<td>Ni</td>
<td>-1.1</td>
<td>&lt; 0.01</td>
<td>0.4</td>
<td>0.45</td>
</tr>
<tr>
<td>P</td>
<td>-1187</td>
<td>0.04</td>
<td>440</td>
<td>0.26</td>
</tr>
<tr>
<td>Se</td>
<td>-154</td>
<td>&lt; 0.01</td>
<td>56</td>
<td>0.44</td>
</tr>
<tr>
<td>Zn</td>
<td>-51.9</td>
<td>0.01</td>
<td>19.4</td>
<td>0.37</td>
</tr>
</tbody>
</table>

\(\dagger\) ORP, oxidation-reduction potential.
\(\ddagger\) DOC, dissolved organic carbon.
\(\ddagger\) Test if common intercept is different than zero.
\(\ddagger\) Coefficient of determination (R\(^2\)) is provided as a measure of the strength of the relationship.
Table 3-5. Correlations coefficients ($r$) for both annual precipitation and annual runoff with annual runoff properties and flow-weighted mean runoff concentrations during an 8-yr period (n = 48).

| Runoff Property | Precipitation | | Runoff | | |
|-----------------|---------------|---------------------|---------------------|
|                 | $r$ | $P$-value$^\dagger$ | $r$ | $P$-value$^\dagger$ | |
| Runoff          | 0.65 | < 0.01$^\#$ | - | - | |
| pH              | 0.22 | 0.14 | 0.03 | 0.84 | |
| ORP$^\dagger$  | -0.61 | < 0.01 | -0.42 | < 0.01 | |
| EC$^\dagger$   | 0.26 | 0.08 | 0.17 | 0.24 | |

Concentrations

| Concentration | Precipitation | | Runoff | | |
|---------------|---------------|---------------------|---------------------|
| NO$_3$-N      | -0.26 | 0.07 | -0.12 | 0.42 | |
| NH$_4$-N      | -0.10 | 0.51 | -0.25 | 0.09 | |
| PO$_4$-P      | -0.50 | < 0.01 | -0.20 | 0.17 | |
| DOC$^\dagger$ | -0.57 | < 0.01 | -0.26 | 0.07 | |
| As            | -0.34 | 0.02 | -0.14 | 0.34 | |
| Ca            | 0.36 | 0.01 | 0.27 | 0.06 | |
| Cd            | 0.54 | < 0.01 | 0.18 | 0.23 | |
| Cr            | -0.50 | < 0.01 | -0.07 | 0.64 | |
| Cu            | 0.08 | 0.60 | -0.04 | 0.80 | |
| Fe            | -0.05 | 0.73 | 0.02 | 0.90 | |
| K             | -0.52 | < 0.01 | -0.31 | 0.03 | |
| Mg            | -0.41 | < 0.01 | -0.15 | 0.29 | |
| Mn            | -0.51 | < 0.01 | -0.16 | 0.26 | |
| Na            | 0.33 | 0.02 | 0.54 | < 0.01 | |
| Ni            | -0.36 | 0.01 | -0.13 | 0.37 | |
| P             | -0.47 | < 0.01 | -0.16 | 0.27 | |
| Se            | 0.60 | < 0.01 | 0.34 | 0.02 | |
| Zn            | -0.48 | < 0.01 | -0.25 | 0.09 | |

$^\dagger$ ORP, oxidation-reduction potential.

$^\dagger$ EC, electrical conductivity.

$^\dagger$ DOC, dissolved organic carbon.

$^\dagger$ Pearson test for correlation.

$^\# P \leq 0.05$ are indicated in bold.
Table 3-6. Summary of annual broiler litter (BL) treatment means and grand means across all BL treatments for runoff properties that were unaffected by BL, time, or their interaction (Table 3-3) during an 8-yr period as determined by analysis of covariance. Broiler litter was hand applied once annually to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas at three application rates (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively). Standard errors of treatment means and grand means are provided in parenthesis as estimates of variability.

<table>
<thead>
<tr>
<th>Runoff Property</th>
<th>Annual Broiler Litter Treatment Mean(\ddagger)</th>
<th>Grand Mean(\ddagger)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Control</td>
<td>Low</td>
</tr>
<tr>
<td>pH</td>
<td>6.73 (0.07)</td>
<td>6.70 (0.07)</td>
</tr>
<tr>
<td>EC(\dagger) (µS cm(^{-1}))</td>
<td>395 (34)</td>
<td>359 (17)</td>
</tr>
</tbody>
</table>

Concentrations (mg L\(^{-1}\))

<table>
<thead>
<tr>
<th></th>
<th>Control</th>
<th>Low</th>
<th>High</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO(_3)-N</td>
<td>0.38 (0.05)</td>
<td>0.64 (0.21)</td>
<td>0.92 (0.37)</td>
</tr>
<tr>
<td>NH(_4)-N</td>
<td>1.90 (0.25)</td>
<td>1.75 (0.39)</td>
<td>2.33 (0.39)</td>
</tr>
<tr>
<td>PO(_4)-P</td>
<td>2.16 (0.34)</td>
<td>3.01 (0.23)</td>
<td>2.82 (0.42)</td>
</tr>
<tr>
<td>DOC(\ddagger)</td>
<td>18.7 (4.3)</td>
<td>18.1 (1.8)</td>
<td>19.5 (6.1)</td>
</tr>
<tr>
<td>Cr</td>
<td>0.003 (&lt; 0.001)</td>
<td>0.003 (&lt; 0.001)</td>
<td>0.003 (&lt; 0.001)</td>
</tr>
<tr>
<td>K</td>
<td>22.7 (6.1)</td>
<td>21.5 (2.4)</td>
<td>24.4 (6.5)</td>
</tr>
<tr>
<td>Mg</td>
<td>1.9 (0.2)</td>
<td>2.0 (0.1)</td>
<td>2.2 (0.4)</td>
</tr>
<tr>
<td>Mn</td>
<td>0.06 (0.01)</td>
<td>0.05 (0.01)</td>
<td>0.08 (0.04)</td>
</tr>
<tr>
<td>Ni</td>
<td>0.003 (&lt; 0.001)</td>
<td>0.002 (&lt; 0.001)</td>
<td>0.003 (0.001)</td>
</tr>
<tr>
<td>P</td>
<td>2.4 (0.4)</td>
<td>3.2 (0.3)</td>
<td>3.0 (0.5)</td>
</tr>
<tr>
<td>Zn</td>
<td>0.14 (0.02)</td>
<td>0.14 (0.01)</td>
<td>0.15 (0.03)</td>
</tr>
</tbody>
</table>

\(^\dagger\) EC, electrical conductivity.
\(^\ddagger\) DOC, dissolved organic carbon.
\(\ddagger\) Treatment means, n = 16.
\(\ddagger\) Grand means, n = 48.
Table 3-7. Analysis of covariance summary of the effects of broiler litter (BL) application rate, time (Year), and their interaction on the linear relationship between runoff loads and time.

<table>
<thead>
<tr>
<th>Runoff Load</th>
<th>Source of Variance</th>
<th>BL†</th>
<th>Year§</th>
<th>BL x Year¶</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>P-value</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NO₃-N</td>
<td>0.96</td>
<td>&lt; 0.01#</td>
<td>0.83</td>
<td></td>
</tr>
<tr>
<td>NH₄-N</td>
<td>0.44</td>
<td>&lt; 0.01</td>
<td>0.21</td>
<td></td>
</tr>
<tr>
<td>PO₄-P</td>
<td>0.48</td>
<td>&lt; 0.01</td>
<td>0.23</td>
<td></td>
</tr>
<tr>
<td>DOC†</td>
<td>0.86</td>
<td>&lt; 0.01</td>
<td>0.77</td>
<td></td>
</tr>
<tr>
<td>As</td>
<td>0.99</td>
<td>&lt; 0.01</td>
<td>0.99</td>
<td></td>
</tr>
<tr>
<td>Ca</td>
<td>0.92</td>
<td>&lt; 0.01</td>
<td>0.75</td>
<td></td>
</tr>
<tr>
<td>Cd</td>
<td>0.98</td>
<td>&lt; 0.01</td>
<td>0.99</td>
<td></td>
</tr>
<tr>
<td>Cr</td>
<td>0.99</td>
<td>&lt; 0.01</td>
<td>1.00</td>
<td></td>
</tr>
<tr>
<td>Cu</td>
<td>0.85</td>
<td>&lt; 0.01</td>
<td>0.83</td>
<td></td>
</tr>
<tr>
<td>Fe</td>
<td>0.60</td>
<td>0.01</td>
<td>0.41</td>
<td></td>
</tr>
<tr>
<td>K</td>
<td>0.68</td>
<td>&lt; 0.01</td>
<td>0.42</td>
<td></td>
</tr>
<tr>
<td>Mg</td>
<td>0.88</td>
<td>&lt; 0.01</td>
<td>0.83</td>
<td></td>
</tr>
<tr>
<td>Mn</td>
<td>0.53</td>
<td>0.01</td>
<td>0.27</td>
<td></td>
</tr>
<tr>
<td>Na</td>
<td>0.94</td>
<td>&lt; 0.01</td>
<td>0.91</td>
<td></td>
</tr>
<tr>
<td>Ni</td>
<td>0.98</td>
<td>&lt; 0.01</td>
<td>1.00</td>
<td></td>
</tr>
<tr>
<td>P</td>
<td>0.49</td>
<td>&lt; 0.01</td>
<td>0.23</td>
<td></td>
</tr>
<tr>
<td>Se</td>
<td>0.99</td>
<td>&lt; 0.01</td>
<td>1.00</td>
<td></td>
</tr>
<tr>
<td>Zn</td>
<td>0.80</td>
<td>&lt; 0.01</td>
<td>0.66</td>
<td></td>
</tr>
</tbody>
</table>

† DOC, dissolved organic carbon.
‡ Test for different y-intercepts among BL treatments with common slope.
§ Test if common slope is different than zero.
¶ Test for different slopes among BL treatments.
# P ≤ 0.05 are indicated in bold.
Table 3-8. Correlation coefficients ($r$) for annual precipitation and runoff with annual runoff loads during an 8-yr period ($n = 48$).

<table>
<thead>
<tr>
<th>Runoff Load</th>
<th>Precipitation</th>
<th>Runoff</th>
<th>Runoff Load</th>
<th>Precipitation</th>
<th>Runoff</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>$r$</td>
<td>$P$-value$^\dagger$</td>
<td>$r$</td>
<td>$P$-value$^\dagger$</td>
<td></td>
</tr>
<tr>
<td>NO$_3$-N</td>
<td>0.62</td>
<td>$&lt; 0.01^\ddagger$</td>
<td>0.90</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>NH$_4$-N</td>
<td>0.62</td>
<td>$&lt; 0.01$</td>
<td>0.88</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>PO$_4$-P</td>
<td>0.52</td>
<td>$&lt; 0.01$</td>
<td>0.79</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>DOC$^\dagger$</td>
<td>0.66</td>
<td>$&lt; 0.01$</td>
<td>0.95</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>As</td>
<td>0.63</td>
<td>$&lt; 0.01$</td>
<td>0.99</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>Ca</td>
<td>0.64</td>
<td>$&lt; 0.01$</td>
<td>0.97</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>Cd</td>
<td>0.67</td>
<td>$&lt; 0.01$</td>
<td>0.99</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>Cr</td>
<td>0.65</td>
<td>$&lt; 0.01$</td>
<td>1.00</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>Cu</td>
<td>0.67</td>
<td>$&lt; 0.01$</td>
<td>0.96</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>Fe</td>
<td>0.42</td>
<td>$&lt; 0.01$</td>
<td>0.65</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>K</td>
<td>0.63</td>
<td>$&lt; 0.01$</td>
<td>0.77</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>Mg</td>
<td>0.62</td>
<td>$&lt; 0.01$</td>
<td>0.94</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>Mn</td>
<td>0.41</td>
<td>$&lt; 0.01$</td>
<td>0.62</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>Na</td>
<td>0.63</td>
<td>$&lt; 0.01$</td>
<td>0.98</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>Ni</td>
<td>0.68</td>
<td>$&lt; 0.01$</td>
<td>1.00</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>P</td>
<td>0.52</td>
<td>$&lt; 0.01$</td>
<td>0.79</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>Se</td>
<td>0.66</td>
<td>$&lt; 0.01$</td>
<td>1.00</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>Zn</td>
<td>0.62</td>
<td>$&lt; 0.01$</td>
<td>0.94</td>
<td>$&lt; 0.01$</td>
<td></td>
</tr>
</tbody>
</table>

$^\dagger$ DOC, dissolved organic carbon.
$^\ddagger$ Pearson test for correlation.
$^\ddagger$ $P \leq 0.05$ are indicated in bold.
Table 3-9. Blocking variance expressed as a percentage of total variance for annual mean runoff properties and flow-weighted mean concentrations. Percents were calculated by dividing the blocking estimate for a given runoff property by the sum of the blocking and error estimates for that runoff property and then multiplying by 100.

<table>
<thead>
<tr>
<th>Runoff Property</th>
<th>Percent of Total Variance</th>
</tr>
</thead>
<tbody>
<tr>
<td>Runoff</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>pH</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>ORP†</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>EC‡</td>
<td>10.9</td>
</tr>
</tbody>
</table>

Concentrations

<table>
<thead>
<tr>
<th>Compound</th>
<th>Percent of Total Variance</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO₃-N</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>NH₄-N</td>
<td>3.9</td>
</tr>
<tr>
<td>PO₄-P</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>DOC§</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>As</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Ca</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Cd</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Cr</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Cu</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Fe</td>
<td>12.6</td>
</tr>
<tr>
<td>K</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Mg</td>
<td>2.4</td>
</tr>
<tr>
<td>Mn</td>
<td>47.9</td>
</tr>
<tr>
<td>Na</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Ni</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>P</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Se</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Zn</td>
<td>0.5</td>
</tr>
</tbody>
</table>

† ORP, oxidation-reduction potential.
‡ EC, electrical conductivity.
§ DOC, dissolved organic carbon.
Table 3-10. Blocking variance expressed as a percentage of total variance for annual mean runoff loads. Percents were calculated by dividing the blocking estimate for a given runoff property by the sum of the blocking and error estimates for that runoff property and then multiplying by 100.

<table>
<thead>
<tr>
<th>Runoff Load</th>
<th>Percent of Total Variance</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO$_3$-N</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>NH$_4$-N</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>PO$_4$-P</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>DOC$^+$</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>As</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Ca</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Cd</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Cr</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Cu</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Fe</td>
<td>1.5</td>
</tr>
<tr>
<td>K</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Mg</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Mn</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Na</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Ni</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>P</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Se</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Zn</td>
<td>&lt; 0.1</td>
</tr>
</tbody>
</table>

$^+$DOC, dissolved organic carbon.
Table 3-11. Analysis of variance summary of the effects of broiler litter (BL) application rate on 8-yr cumulative runoff; 8-yr mean runoff pH, EC, and ORP; and 8-yr flow-weighted mean runoff concentrations. Broiler litter was hand applied once annually to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas at three application rates (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) for an 8-yr period. Standard errors of treatment means and grand means are provided in parenthesis as estimates of variability.

<table>
<thead>
<tr>
<th>Runoff Property</th>
<th>P-value</th>
<th>Broiler Litter Treatment Mean(^\dagger)</th>
<th>Grand Mean(^#)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Control</td>
<td>Low</td>
</tr>
<tr>
<td>Runoff (mm)</td>
<td>0.84</td>
<td>263 (3)</td>
<td>268 (25)</td>
</tr>
<tr>
<td>pH</td>
<td>0.15</td>
<td>6.73 (0.02)</td>
<td>6.69 (0.03)</td>
</tr>
<tr>
<td>ORP(^\ddagger) (mV)</td>
<td>0.09</td>
<td>11.1 (1.8)</td>
<td>12.7 (1.5)</td>
</tr>
<tr>
<td>EC(^\ddagger) (µS cm(^{-1}))</td>
<td>0.65</td>
<td>395 (64)</td>
<td>359 (2)</td>
</tr>
</tbody>
</table>

Concentrations (mg L\(^{-1}\))

<table>
<thead>
<tr>
<th></th>
<th>NO(_3)-N</th>
<th>NH(_4)-N</th>
<th>PO(_4)-P</th>
<th>DOC(^\ddagger)</th>
<th>As</th>
<th>Ca</th>
<th>Cd</th>
<th>Cr</th>
<th>Cu</th>
<th>Fe</th>
<th>K</th>
<th>Mg</th>
<th>Mn</th>
<th>Na</th>
<th>Ni</th>
<th>P</th>
<th>Se</th>
<th>Zn</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0.90</td>
<td>0.07</td>
<td>0.37</td>
<td>0.46</td>
<td>0.86</td>
<td>0.36</td>
<td>0.90</td>
<td>0.90</td>
<td>0.57</td>
<td>0.71</td>
<td>0.53</td>
<td>0.75</td>
<td>0.68</td>
<td>0.83</td>
<td>0.10</td>
<td>0.39</td>
<td>0.94</td>
<td>0.70</td>
</tr>
<tr>
<td></td>
<td>0.42 (0.18)</td>
<td>1.5 (0.2)</td>
<td>1.0 (0.1)</td>
<td>8.5 (0.5)</td>
<td>0.01 (&lt; 0.01)</td>
<td>17.1 (1.2)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
<td>0.02 (&lt; 0.01)</td>
<td>0.09 (0.04)</td>
<td>8.1 (1.9)</td>
<td>1.4 (0.1)</td>
<td>0.04 (0.01)</td>
<td>5.3 (0.6)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
<td>1.1 (0.2)</td>
<td>0.28 (&lt; 0.01)</td>
<td>0.09 (0.03)</td>
</tr>
<tr>
<td></td>
<td>0.45 (0.02)</td>
<td>0.8 (0.1)</td>
<td>2.7 (0.6)</td>
<td>10.3 (0.9)</td>
<td>0.01 (&lt; 0.01)</td>
<td>15.0 (3.1)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
<td>0.02 (&lt; 0.01)</td>
<td>0.13 (0.06)</td>
<td>10.0 (0.5)</td>
<td>1.7 (0.1)</td>
<td>0.01 (&lt; 0.01)</td>
<td>5.8 (0.5)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
<td>3.0 (0.7)</td>
<td>0.28 (0.01)</td>
<td>0.10 (0.01)</td>
</tr>
<tr>
<td></td>
<td>0.37 (0.07)</td>
<td>1.6 (0.1)</td>
<td>2.6 (1.3)</td>
<td>10.5 (1.5)</td>
<td>0.01 (&lt; 0.01)</td>
<td>12.4 (0.4)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
<td>0.03 (&lt; 0.01)</td>
<td>0.26 (0.23)</td>
<td>11.3 (2.8)</td>
<td>1.8 (0.7)</td>
<td>0.06 (0.04)</td>
<td>6.3 (1.5)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
<td>2.9 (1.5)</td>
<td>0.28 (&lt; 0.01)</td>
<td>0.13 (0.03)</td>
</tr>
<tr>
<td></td>
<td>0.04 (0.05)</td>
<td>0.05 (0.02)</td>
<td>0.01 (&lt; 0.01)</td>
<td>9.8 (0.6)</td>
<td>0.01 (&lt; 0.01)</td>
<td>14.9 (1.2)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
<td>0.02 (&lt; 0.01)</td>
<td>0.16 (0.07)</td>
<td>9.8 (1.1)</td>
<td>1.7 (0.2)</td>
<td>0.04 (0.01)</td>
<td>5.8 (0.5)</td>
<td>&lt; 0.01 (&lt; 0.01)</td>
<td>2.3 (0.6)</td>
<td>0.28 (&lt; 0.01)</td>
<td>0.10 (0.01)</td>
</tr>
</tbody>
</table>

\(^\dagger\) ORP, oxidation-reduction potential.

\(^\ddagger\) EC, electrical conductivity.

\(^\ddagger\) DOC, dissolved organic carbon.

\(^\dagger\) Treatment means, n = 2.

\(^\#\) Grand means, n = 6.
Table 3-12. Analysis of variance summary of the effects of broiler litter (BL) application rate on 8-yr cumulative runoff loads. Broiler litter was hand applied once annually to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas at three application rates (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) for an 8-yr period. Standard errors of treatment means and grand means are provided in parenthesis as estimates of variability.

| Runoff Load | P-value | Broiler Litter Treatment Mean\(^\dagger\) | | | | | | | | | | | | | | | | | |
|-------------|---------|---------------------------------------------|---|---|---|---|---|
|             |         | Control                                     | Low | High | Grand Mean\(^\S\) | | | | | | | | | | | | |
| NO\(_3\)-N  | 0.93    | 1.1 (0.5)                                   | 1.2 (0.1) | 1.0 (0.1) | 1.1 (0.1) |
| NH\(_4\)-N  | 0.15    | 3.8 (0.3)                                   | 2.1 (0.4) | 4.6 (0.6) | 3.5 (0.5) |
| PO\(_4\)-P  | 0.33    | 2.7 (0.4)                                   | 7.4 (2.3) | 7.0 (3.2) | 5.7 (1.4) |
| DOC†        | 0.41    | 22.3 (1.0)                                  | 27.8 (5.0) | 29.1 (2.3) | 26.4 (2.0) |
| As          | 0.53    | 0.03 (< 0.01)                               | 0.03 (< 0.01) | 0.04 (< 0.01) | 0.03 (< 0.01) |
| Ca          | 0.71    | 45.1 (3.8)                                  | 40.8 (12.2) | 35.1 (3.4) | 40.3 (3.9) |
| Cd          | 0.87    | < 0.01 (< 0.01)                             | < 0.01 (< 0.01) | 0.01 (< 0.01) | < 0.01 (< 0.01) |
| Cr          | 0.85    | 0.01 (< 0.01)                               | 0.01 (< 0.01) | 0.01 (< 0.01) | 0.01 (< 0.01) |
| Cu          | 0.62    | 0.06 (< 0.01)                               | 0.07 (0.02) | 0.07 (< 0.01) | 0.06 (0.01) |
| Fe          | 0.70    | 0.23 (0.11)                                 | 0.35 (0.18) | 0.69 (0.60) | 0.42 (0.19) |
| K           | 0.21    | 21.4 (5.3)                                  | 26.5 (1.2) | 31.4 (5.9) | 26.4 (2.8) |
| Mg          | 0.54    | 3.8 (0.4)                                   | 4.6 (0.6) | 5.1 (1.7) | 4.5 (0.5) |
| Mn          | 0.66    | 0.10 (0.04)                                 | 0.04 (0.01) | 0.15 (0.11) | 0.09 (0.04) |
| Na          | 0.60    | 13.8 (1.3)                                  | 15.5 (0.1) | 17.6 (3.2) | 15.6 (1.1) |
| Ni          | 0.77    | 0.01 (< 0.01)                               | 0.01 (< 0.01) | 0.01 (< 0.01) | 0.01 (< 0.01) |
| P           | 0.35    | 2.9 (0.4)                                   | 8.2 (2.6) | 7.9 (3.8) | 6.3 (1.6) |
| Se          | 0.86    | 0.74 (< 0.01)                               | 0.76 (0.09) | 0.81 (0.06) | 0.77 (0.03) |
| Zn          | 0.58    | 0.23 (0.07)                                 | 0.27 (0.04) | 0.35 (0.06) | 0.28 (0.03) |

\(\dagger\) DOC, dissolved organic carbon.

\(\dagger\) Treatment means, n = 2.

\(\S\) Grand means, n = 6.
Table 3-13. Runoff mass balance for broiler litter-derived nutrients from a pasture soil amended with broiler litter (BL). Litter was hand applied once annually to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas at three application rates (0, 5.6 and 11.2 Mg BL ha\(^{-1}\); control, low and high, respectively) for an 8-yr period. Runoff was continuously monitored and collected after each runoff producing event. Values represent 8-yr cumulative totals.

<table>
<thead>
<tr>
<th>Nutrient</th>
<th>Mass Added†</th>
<th>Runoff Mass‡</th>
<th>Percent Loss§</th>
<th>Corrected Loss¶</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Low</td>
<td>High</td>
<td>Low</td>
<td>High</td>
</tr>
<tr>
<td>NO(_3)-N</td>
<td>8.8</td>
<td>18.5</td>
<td>1.2</td>
<td>1.0</td>
</tr>
<tr>
<td>NH(_4)-N</td>
<td>208</td>
<td>416</td>
<td>2.1</td>
<td>4.6</td>
</tr>
</tbody>
</table>

Total Elements

<table>
<thead>
<tr>
<th>Element</th>
<th>Mass Added†</th>
<th>Runoff Mass‡</th>
<th>Percent Loss§</th>
<th>Corrected Loss¶</th>
</tr>
</thead>
<tbody>
<tr>
<td>C</td>
<td>16624</td>
<td>33248</td>
<td>27.8</td>
<td>29.1</td>
</tr>
<tr>
<td>N</td>
<td>217</td>
<td>435</td>
<td>3.3</td>
<td>5.6</td>
</tr>
<tr>
<td>P</td>
<td>984</td>
<td>1968</td>
<td>8.2</td>
<td>7.9</td>
</tr>
<tr>
<td>K</td>
<td>1568</td>
<td>3136</td>
<td>26.5</td>
<td>31.4</td>
</tr>
<tr>
<td>Ca</td>
<td>1656</td>
<td>3312</td>
<td>40.8</td>
<td>35.1</td>
</tr>
<tr>
<td>Na</td>
<td>407</td>
<td>814</td>
<td>15.5</td>
<td>17.6</td>
</tr>
<tr>
<td>Mg</td>
<td>314</td>
<td>627</td>
<td>4.6</td>
<td>5.1</td>
</tr>
<tr>
<td>Mn</td>
<td>26</td>
<td>51</td>
<td>0.04</td>
<td>0.15</td>
</tr>
<tr>
<td>Zn</td>
<td>23</td>
<td>46</td>
<td>0.27</td>
<td>0.35</td>
</tr>
<tr>
<td>Cu</td>
<td>22</td>
<td>45</td>
<td>0.07</td>
<td>0.07</td>
</tr>
<tr>
<td>Fe</td>
<td>18</td>
<td>37</td>
<td>0.35</td>
<td>0.69</td>
</tr>
<tr>
<td>As</td>
<td>1.2</td>
<td>2.4</td>
<td>0.03</td>
<td>0.03</td>
</tr>
<tr>
<td>Ni</td>
<td>0.46</td>
<td>0.93</td>
<td>0.01</td>
<td>0.01</td>
</tr>
<tr>
<td>Cr</td>
<td>0.32</td>
<td>0.64</td>
<td>0.01</td>
<td>0.01</td>
</tr>
<tr>
<td>Se</td>
<td>0.15</td>
<td>0.30</td>
<td>0.76</td>
<td>0.81</td>
</tr>
<tr>
<td>Cd</td>
<td>0.01</td>
<td>0.02</td>
<td>&lt;0.01</td>
<td>0.01</td>
</tr>
</tbody>
</table>

† Cumulative mass of nutrient added from BL amendments.
‡ Cumulative mass of nutrient in runoff.
§ Percent loss equal to mass in runoff minus mass added, times 100.
¶ Corrected percent loss was corrected for runoff loss from unamended control before calculating percent loss. Negative values represent control treatment within same experimental block loss was greater than littered treatment.
Figure 3-1. Annual precipitation and runoff from broiler litter (BL) amended pasture soil during eight years. Broiler litter was hand applied once annually to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas at three application rates (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively). Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. The horizontal dashed line represents the 30-yr mean precipitation of 1232 mm (NOAA, 2013).
Figure 3-2. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha⁻¹; control, low, and high, respectively) effect over an 8-yr period on annual runoff, annual runoff pH [standard error (SE) = 0.04], annual electrical conductivity (EC; SE = 17), and annual oxidation-reduction potential (ORP). Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. The solid lines for runoff and ORP represent significant ($P < 0.05$) changes over time averaged across BL treatments.
Figure 3-3. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effect over an 8-yr period on annual flow-weighted-mean (FWM) runoff iron (Fe) concentrations. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. The solid, small-dashed, and long-dashed lines represent changes in annual FWM runoff Fe concentrations over time for the control, low, and high BL treatments, respectively.
Figure 3-4. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effect over an 8-yr period on annual flow-weighted-mean runoff arsenic (As), calcium (Ca), cadmium (Cd), copper (Cu), sodium (Na), and selenium (Se) concentrations. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. Solid lines represent changes over time averaged across all BL treatments.
Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effect over an 8-yr period on annual flow-weighted-mean runoff nitrate-nitrogen \([\text{NO}_3^{-}\text{N}]; \text{standard error (SE)} = 0.14\) \), ammonium-nitrogen \((\text{NH}_4^{+}\text{N}; \text{SE} = 0.20)\), phosphate-phosphorus \((\text{PO}_4^{3-}\text{P}; \text{SE} = 0.20)\), dissolved organic carbon \((\text{DOC}; \text{SE} = 2.5)\), chromium \((\text{Cr}; \text{SE} < 1)\), and potassium \((\text{K}; \text{SE} = 3.0)\) concentrations. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year.

Figure 3-5.
Figure 3-6. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effect over an 8-yr period on annual flow-weighted-mean runoff magnesium [Mg; standard error (SE) = 0.2], manganese (Mn; SE = 0.01), nickel (Ni; SE < 1), phosphorus (P; SE = 0.2), and zinc (Zn; SE = 0.01) concentrations. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year.
Figure 3-7. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effect over an 8-yr period on annual runoff nitrate-nitrogen (NO\(_3\)-N), ammonium-nitrogen (NH\(_4\)-N), phosphate-phosphorus (PO\(_4\)-P), dissolved organic carbon (DOC), arsenic (As), and calcium (Ca) loads. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. Solid lines represent changes over time averaged across all BL treatments.
Figure 3-8. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effect over an 8-yr period on annual runoff cadmium (Cd), chromium (Cr), copper (Cu), iron (Fe), potassium (K), and magnesium (Mg) loads. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. Solid lines represent changes over time averaged across all BL treatments.
Figure 3-9. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effect over an 8-yr period on annual runoff manganese (Mn), sodium (Na), nickel (Ni), phosphorus (P), selenium (Se), and zinc (Zn) loads. Study years were designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. Solid lines represent changes over time averaged across all BL treatments.
Appendix 3-1. Example SAS program for analysis of covariance run as a macro and related data.

options mprint;

title 'Annual Runoff LOADS and FWM -- Richard McMullen --';
title2 'All elements Runoff Models 1 and 2 Estimates pH, Redox, EC and DOC etc.';

data loads;
  infile 'F:\McMullen\Runoff\SAS\AnnualTrends\AnnualROHalfLimit.csv' firstobs=2 DLM=',';
  truncover LRECL = 600 DSD;
  input Year BL $ lys Block Vol ROmm pH redox EC NO3 NH4 PO4 DOC Al As Ca
  Cd Cr Cu Fe K Mg Mn Na Ni TP S Se Zn fwmNO3 fwmNH4 fwmPO4 fwmDOC fwmAl fwmAs fwmCa fwmCd fwmCr fwmCu
  fwmFe fwmK fwmMg fwmMn fwmNa fwmNi fwmP fwmS fwmSe fwmZn ;
  * year = 2002 + year;
  label BL = 'Broiler Litter';
run;

proc sort data=loads; by year block lys;
run;
quit;

%Macro xxx(var=);

title3 'INITIAL DATA LISTING';
proc print data=loads noobs; by year block;
  id year block;
  var lys bl ROmm pH redox EC NO3 NH4 PO4 DOC Al As Ca
  Cd Cr Cu Fe K Mg Mn Na Ni TP S Se Zn fwmNO3 fwmNH4 fwmPO4 fwmDOC fwmAl fwmAs fwmCa fwmCd fwmCr fwmCu
  fwmFe fwmK fwmMg fwmMn fwmNa fwmNi fwmP fwmS fwmSe fwmZn ;
run cancel;
quit;

%Macro xxx(var=);

title3 'Parameter ESTIMATES for 1st TEST- Diff Slopes, Diff Intercepts';
title4 "---------- &var ---------";
proc mixed data=loads method=type3 ;
  class block bl1 ;
  model &var = BL Year BL*Year / ddfm=kr solution; /* original model */
  random Block ;
run;
quit;
title3 'Parameter Estimates for 2nd TEST- if BL X Year is NS to test for common slope = ZERO';

title4 "---------- &var----------";

proc mixed data=loads method=type3;
  class block bl;
  model &var = BL Year / ddfm=kr residual outpm=phresid;    /* run when BL*year NS to test common slope = 0 */
  random Block;
run;

%MEnd xxx;

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Appendix 3-2. Example SAS program for analysis of variance run as a macro and related data.

options mprint;

title '8 YEAR Runoff LOADS and FWM -- Richard McMullen --';
title2 'All elements Runoff';

data loads;
  infile 'F:\McMullen\Runoff\SAS\8YearCumLoads\8yrCumulativeROHalfLimit.csv' firstobs=2
    DLM=',' truncover LRECL = 600 DSD;
  input BL $ lys Block ROmm pH redox EC NO3 NH4 PO4 DOC Al As Ca
       Cd Cr Cu Fe K Mg Mn Na Ni TP S Se Zn fwmNO3 fwmNH4
       fwmPO4 fwmDOC fwmAl fwmAs fwmCa fwmCd fwmCr fwmCu
       fwmFe fwmK fwmMg fwmMn fwmNa fwmNi fwmP fwmS fwmSe fwmZn ;
  label BL = 'Broiler Litter';
run;

proc sort data=loads; by block lys;
run;
quit;

title3 'INITIAL DATA LISTING';
proc print data=loads noobs; by block;
  id block;
  var BL lys ROmm pH redox EC NO3 NH4 PO4 DOC Al As Ca
       Cd Cr Cu Fe K Mg Mn Na Ni TP S Se Zn fwmNO3 fwmNH4
       fwmPO4 fwmDOC fwmAl fwmAs fwmCa fwmCd fwmCr fwmCu
       fwmFe fwmK fwmMg fwmMn fwmNa fwmNi fwmP fwmS fwmSe fwmZn ;
run;
quit;

%Macro xxx(var=);

title3 'ANALYSIS OF VARIANCE';
title4 "-------- &var --------";
proc mixed data=loads method=type3 ;
  class block b1;
  model &var = BL ;
  random block;
  lsmeans BL;
  contrast 'Control vs High' b1 1 -1 0 ;
  contrast 'Control vs Low' b1 0 1 -1 ;
  contrast 'Low vs High' b1 0 1 -1 ;
run;
%MEnd xxx;

%xxx(var=ROmm)
%xxx(var=pH)
%xxx(var=redox)
%xxx(var=EC)
%xxx(var=NO3)
%xxx(var=NH4)
%xxx(var=PO4)
%xxx(var=DOC)
%xxx(var=Al)
%xxx(var=As)
%xxx(var=Ca)
%xxx(var=Cd)
%xxx(var=Cr)
%xxx(var=Cu)
%xxx(var=Fe)
%xxx(var=K)
%xxx(var=Mg)
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%xxx(var=fwmCd)
%xxx(var=fwmCr)
%xxx(var=fwmCu)
%xxx(var=fwmFe)
%xxx(var=fwmK)
%xxx(var=fwmMg)
%xxx(var=fwmMn)
%xxx(var=fwmNa)
%xxx(var=fwmNi)
%xxx(var=fwmP)
%xxx(var=fwmS)
%xxx(var=fwmSe)
%xxx(var=fwmZn );
Note: The 8-yr data set is presented as transposed.

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<td>fwmP (mg/L)</td>
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<td>1.28</td>
<td>4.44</td>
<td>0.95</td>
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<td>fwmS (mg/L)</td>
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<td>1.33</td>
<td>3.58</td>
<td>1.41</td>
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<td>fwmSe (mg/L)</td>
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<td>0.287</td>
<td>0.283</td>
<td>0.280</td>
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<td>fwmZn (mg/L)</td>
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<td>0.115</td>
<td>0.156</td>
<td>0.063</td>
<td>0.106</td>
<td>0.095</td>
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Appendix 3-3. Example SAS program for Pearson’s correlation analysis.

options mprint;

title 'Annual Runoff LOADS and FWM -- Richard McMullen --';
title2 'All elements Runoff Correlations with Rainfall-mm and Runoff-mm';

data loads;
infile 'F:\McMullen\Runoff\SAS\Correlations\AnnualROHalfLimitCorrelations.csv' firstobs=2
DLM=', truncover LRECL = 600 DSD;
input Year BL $ lys Block Vol ROmm pH redox EC NO3 NH4 PO4 DOC Al As Ca
   Cd Cr Cu Fe K Mg Mn Na Ni TP S Se Zn fwmNO3 fwmNH4
   fwmPO4 fwmDOC fwmAl fwmAs fwmCa fwmCd fwmCr fwmCu
   fwmFe fwmK fwmMg fwmMn fwmNa fwmNi fwmP fwmS fwmSe fwmZn rain;
   * year = 2002 + year;
   label BL = 'Broiler Litter';
run;

proc sort data=loads; by year block lys;
run;
quit;

title3 'INITIAL DATA LISTING';
proc print data=loads noobs; by year block;
id year block;
var lys bl ROmm pH redox EC NO3 NH4 PO4 DOC Al As Ca
   Cd Cr Cu Fe K Mg Mn Na Ni TP S Se Zn fwmNO3 fwmNH4
   fwmPO4 fwmDOC fwmAl fwmAs fwmCa fwmCd fwmCr fwmCu
   fwmFe fwmK fwmMg fwmMn fwmNa fwmNi fwmP fwmS fwmSe fwmZn rain;
run cancel;
quit;

title3 'Correlations';
title4 "---------- ----------";

ods graphics on;
proc corr data=loads plots=matrix(histogram);
var Year ROmm pH redox EC NO3 NH4 PO4 DOC Al As Ca
   Cd Cr Cu Fe K Mg Mn Na Ni TP S Se Zn fwmNO3 fwmNH4
   fwmPO4 fwmDOC fwmAl fwmAs fwmCa fwmCd fwmCr fwmCu
   fwmFe fwmK fwmMg fwmMn fwmNa fwmNi fwmP fwmS fwmSe fwmZn rain;
run;
ods graphics off;
run;
quit;
Appendix 3-4. Example SAS program for analysis of variance for annual and 8-yr cumulative above ground dry matter production run as a macro and related data.

```sas
options mprint;

title 'Annual Above Ground Biomass and 8-yr Cumulative AGB -- Richard McMullen --';
title2 'ANOVA blocks fixed';

data AGB;
  infile 'F:\McMullen\Biomass\SAS\AGByear1thru8.csv' firstobs=3 DLM=',' truncover LRECL = 600 DSD;
  input Lys BL $ Yr1 Yr2 Yr3 Yr4 Yr5 Yr6 Yr7 Yr8 CumAGB;
  label BL = 'Broiler Litter';
run;

proc sort data=AGB; by bl lys;
run;
quit;

title3 'INITIAL DATA LISTING';
proc print data=AGB noobs; by bl lys;
  id bl lys;
  var Yr1 Yr2 Yr3 Yr4 Yr5 Yr6 Yr7 Yr8 CumAGB;
run;
quit;

%MMacro xxx(var=);

title3 'ANOVA';
title4 "---------- &var ----------";
proc mixed data=AGB method=type3 ;
  class bl;
  model &var = BL / ddfm=kr residual outpm=ph2resid;
  lsmeans bl;
  contrast 'CvsH' b1 1 -1 0;
  contrast 'CvsL' b1 1 0 -1;
  contrast 'HvsL' b1 0 1 -1;
run;
%MEnd xxx;

%xxx(var=Yr1 );
%xxx(var=Yr2 );
%xxx(var=Yr3 );
```

254
%xxx(var=Yr4   );
%xxx(var=Yr5   );
%xxx(var=Yr6   );
%xxx(var=Yr7   );
%xxx(var=Yr8   );
%xxx(var=CumAGB );
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Chapter Four

Soil Respiration as Affected by Broiler Litter Application

Rate from a Udult in the Ozark Highlands
Abstract

In 2012, the United States produced 8.4 billion broiler chickens (*Gallus gallus*) and an estimated 10.1 to 14.3 million Mg of broiler litter (BL). Arkansas’ production of 1 billion broilers in 2012 produced 1.2 to 1.7 million Mg of BL, the majority of which was concentrated in northwest Arkansas. Broiler litter is commonly land applied to pastures to enhance yields of tall fescue (*Lolium arundinaceum Shreb.*) and other forages. However, increased carbon dioxide (CO₂) release from soils associated with agricultural practices has generated concerns regarding the contribution of certain agricultural management practices to global warming. The objective of this study was to evaluate the effect of long-term (i.e., > 6 yr) BL application on soil respiration, temperature, and moisture to determine if soil temperature and moisture could be used to predict soil respiration in the Ozark Highlands region of northwest Arkansas. Soil respiration from a Captina silt loam (fine-silty, siliceous, active, mesic Typic Fragiudult) was measured routinely between May 2009 and May 2012 in response to annual BL application rates of 0, 5.6 and 11.2 Mg dry litter ha⁻¹ that began in 2003. Annual BL amendments increased \( P = 0.05 \) annual DM production during the three year study. Soil respiration varied \( P < 0.01 \) with BL rate, sample date, and year. Additions of BL stimulated respiration after application and rainfall events following dry-soil conditions also stimulated respiration in all years. Soil temperature at the 2-cm depth (\( T_{2\text{cm}} \)) varied \( P < 0.05 \) with BL rate and year and was attributed to shading associated with increased dry matter (DM) production. Soil temperature at the 10-cm depth (\( T_{10\text{cm}} \)), 0- to 6-cm soil volumetric water content (VWC), and annual CO₂-C emissions were unaffected \( P > 0.05 \) by BL application rate, but differed \( P < 0.01 \) among study years. Multiple regression indicated that soil respiration could be predicted using \( T_{2\text{cm}} \) and the product of \( T_{2\text{cm}} \) and VWC as predictors \( R^2 = 0.52; P < 0.01 \). Results indicate that organic amendments,
such as BL, can stimulate release of CO$_2$ from the soil to the atmosphere, potentially negatively affecting atmospheric greenhouse gas concentrations; thus there may be application rates above which the benefits of organic amendments may be diminished by potential adverse environmental effects. Furthermore, precipitation events following dry conditions may stimulate slow-phase decomposition of BL. Improved BL management strategies are needed to lessen the loss of CO$_2$ from BL-amended soils.
Introduction

In 2012, the United States produced 8.4 billion broiler chickens (Gallus gallus; USDA-NASS, 2013), most of which were concentrated in relatively small geographic regions. Broiler production at this scale produces large quantities of waste, which is referred to as broiler litter (BL). For example, Arkansas ranked third in the nation for production in 2012 with 1.0 billion broilers produced (USDA-NASS, 2013) and an estimated 1.2 to 1.7 million Mg of BL were generated (UADACES, 2002), the majority of which was concentrated in the Ozark Highlands region (Major Land Resource Area 116A; Scott and Ward, 2002; Brye et al., 2013) of northwest Arkansas. Broiler litter is a mixture of excreta, feathers, feed, and bedding material, such as rice (Oryza sativa L.) hulls, saw dust, or straw. Since, BL contains numerous plant nutrients, such as nitrogen (N), a common management practice in the Ozark Highlands is to land apply BL to pastures to increase yields of tall fescue (Lolium arundinaceum Shreb.) and other forages (Hileman, 1973; Huneycutt et al., 1988; Brye et al., 2010). Similar management practices have occurred in the Ozark Highlands and other regions where intense broiler production occurs, sometimes resulting in decades of repeated litter applications.

Anthropogenic activities, including land application of BL (Jones et al., 2005), can increase CO₂ release to the atmosphere, thereby potentially increasing global warming (Forster et al., 2007) and promoting climate change (Trenberth et al., 2007). Soil respiration, the combined product of CO₂ release from soil microorganisms and plant roots, is influenced by agricultural management practices (Risch and Frank, 2010; Roberson et al., 2008; Brye et al., 2006b; Al-Kaisi and Yin, 2005; Yamulki and Jarvis, 2002; Wagai et al., 1998; Linn and Doran, 1984) and is temporally (Ding et al., 2010; Grahmmer et al., 1990; Pingintha et al., 2010; Risch and Frank, 2010; Ruehr et al., 2010; Brown et al., 2009; Brye and Riley, 2009; Jones et al., 2006; Davidson
et al., 1998) and spatially (Aiken et al., 1991) variable. Soil moisture (Brown et al., 2009; Linn and Doran, 1984; Skopp et al., 1990) and temperature (Reth et al., 2009; Brye et al., 2006b; Fierer et al., 2006; Fang and Moncrieff, 2001) have also been shown to influence soil respiration. Additions of mineral fertilizers (Ding et al., 2010) and animal manures (Jones et al., 2005), including turkey (*Meleagris gallopavo*; Pengthamankeerati et al., 2005) and BL (Roberson et al., 2008; Jones et al., 2006; Jones et al., 2005; Adams et al., 1997) have been reported to increase CO$_2$ emissions from soil. In general, additions of animal wastes result in increased carbon sequestration (i.e., carbon storage) within soil (Roberson et al., 2008; Jones et al., 2006; Jones et al., 2005) and if managed correctly could potentially off-set adverse effects associated with BL-induced increases in CO$_2$ release from the soil to the atmosphere.

Additions of BL have been reported to increase soil respiration (Roberson et al., 2008; Jones et al., 2006; Jones et al., 2005; Adams et al., 1997). Adams et al. (1997) explored the use of BL slurry to increase CO$_2$ concentrations within closed-crop canopies as a way to increase crop photosynthesis. In Scotland, soil respiration from a grassland was monitored in response to soil surface amendments of three organic manures (i.e., BL, cattle slurry, and sewage sludge pellets) and two inorganic fertilizers (i.e., urea and a compound containing ammonium nitrate) on a sandy clay loam soil (Jones et al., 2006; Jones et al., 2005). Results from these studies suggest that BL amendment effects on soil respiration may be variable from year to year and that peak respiration rates may occur initially on an hourly scale but may be followed by later peaks that are on a weekly scale. Brye et al. (2006a) evaluated the effects of BL type and application rate on soil respiration from two silt-loam soils used for rice production in eastern Arkansas and reported no difference in respiration among BL types. Brye and Riley (2009) evaluated the effects of prairie restoration age on near-surface soil properties, including respiration, within the
Ozark Highlands in Benton County, Arkansas. Important results demonstrated a temporal effect on soil respiration, although respiration did not vary by location (i.e., prairie restoration age or native prairie).

Researchers have attempted to model soil respiration (Pingintha et al., 2010; Brye et al., 2006b; Šimůnek and Suarez, 1993), but few, if any, have attempted to account for changes in soil respiration in response to BL amendments. Soil moisture and temperature are the two most commonly used environmental factors used to predict soil respiration. Soil moisture has been reported to be positively correlated (Pingintha et al., 2010; Brown et al., 2009), negatively correlated (Brye et al., 2006b; Jones et al., 2006), and uncorrelated (Ding et al., 2010; Brye et al., 2006a; Al-Kaisi and Yin, 2005) to measured soil respiration rates. Brye and Riley (2009) demonstrated a quadratic relationship between water-filled-pore-space (WFPS) and soil respiration in a series of prairie restorations in the Ozark Highlands with respiration being greatest at approximately 50% WFPS. Jones et al. (2006) reported two linear regressions relating soil respiration and soil moisture. Soil temperature has been reported to be positively correlated to soil respiration (Ding et al., 2010; Ruehr et al., 2010; Brown et al., 2009; Brye et al., 2006a; Jones et al., 2006; Fang and Moncrieff, 2001) and variable by season (Ruehr et al., 2010; Brown et al., 2009) and time of day (Davidson et al., 1998; Grahammer et al., 1990; Phillips et al., 2010; Risch and Frannk, 2010; Ruehr et al., 2010). Currently it is unknown if predictive models should account for BL application rates when estimating soil respiration.

Broiler litter application to pasture soil has occurred for extended periods of time and will likely continue to occur in regions where broiler production occurs. Although, organic amendments to soil have been shown to increase CO₂ release to the atmosphere, which increases radiative forcing, few, if any, studies have examined the long-term (i.e., > 6 yr) effect of BL
amendments on soil respiration in non-cultivated soil. Furthermore, prediction of soil respiration based on measurable environmental factors is needed for larger-scale models that are used to predict global climate changes. Therefore, the objectives of this study were to 1) evaluate the effect of BL application rate on soil respiration, temperature, and moisture following 6, 7, and 8 years of annual BL amendments and, 2) determine if soil temperature and moisture parameters are significant predictors of soil respiration from a silt-loam soil in the Ozark Highlands.

It was hypothesized that annual amendments of BL would increase soil respiration, annual CO$_2$-C emissions, and 3-yr cumulative CO$_2$-C emissions relative to an unamended control. Soil temperature and moisture were hypothesized to be unaffected by BL application rate. The logic pertaining to the soil moisture hypothesis assumed that additions of BL would increase soil water holding capacity, but that increased water demand associated with increased DM production in littered treatments would offset any observable soil moisture differences. It was also hypothesized that soil temperature would be positively correlated to respiration and that soil moisture would have a quadratic relationship with respiration. It was also hypothesized that the best predictive model for soil respiration would include parameter coefficients for both soil temperature and moisture and that the moisture term would be quadratic.

**Materials and Methods**

**Site Description**

Research was initiated in 2002 at the Agricultural Research and Extension Center in Fayetteville, Arkansas (36°05’49.18”N 94°10’44.65”W; elevation: 394.7 m; Pirani et al., 2006) in an area mapped as a Captina silt loam (fine-silty, siliceous, active, mesic Typic Fragiudult; USDA-NRCS, 2013). Each of six field plots, 6-m long by 1.5-m wide, had a 5% west-to-east
slope, had a history of land-applied BL prior to 2002, and were initially chosen based on similar soil pH {6.2 [standard error (SE) = 0.5]} and high Mehlich-3 extractable P [210 (SE = 24) mg kg\(^{-1}\)] in the top 5 cm (Pirani et al., 2006). Soil particle-size distribution to a depth of 85 cm was determined with the soil textural class in the 0- to 10-cm depth interval confirmed to be silt loam with 63 % silt and 5.5 % clay (Pirani, 2005). Pirani et al. (2006) also reported increasing clay content with increasing soil depth to 85 cm and a significant textural class change from silt loam to clay loam in the 65- to 85-cm depth interval. Plots had previously been used in simulated runoff studies and were equipped with steel edging to prevent surface water runon as well as to channel runoff from within the plots to collection gutters positioned on the down-slope end of each plot. Initially, tall fescue was the predominate ground cover (Pirani, 2005), but during the current study other species were more common {i.e., clover (\textit{Trifolium spp.}), Johnson grass [\textit{Sorghum halepense} (L.) Pers.] and Bermuda grass (\textit{Cynodon dactylon} L.); McMullen, 2014a}.

The 30-yr mean annual air temperature and precipitation in Fayetteville, AR are 13.9 °C and 123 cm, respectively (NOAA, 2013). The average date of the first frost is October 17 and the average date of the last frost is April 15 (NOAA, 2013).

Since field plots used in the current study had previously received organic amendments prior to 2002, concurrent and related studies (Brye and Pirani, 2006; Pirani et al., 2006; Pirani et al., 2007; Daigh et al., 2009; McDonald et al., 2009; Menjoulet et al., 2009; McMullen, 2014a, b) had previously addressed pre-treatment plot uniformity. Three months prior to the initial BL application in 2003, precipitation, runoff, drainage, and dry matter (DM) were monitored and soil samples were collected (Pirani et al., 2006). During this 3-mo period, precipitation was 98 mm below the 30-yr normal for the area during the months of February, March, and April (Pirani et al., 2006). The mean 3-mo cumulative runoff (Menjoulet et al., 2009), drainage at a depth of
90 cm (Pirani et al., 2006), and total DM production (Pirani et al., 2006) did not differ among pre-assigned BL treatments prior to the first BL application. Additionally, mean runoff electrical conductively (EC) and flow-weighted-mean (FWM) runoff concentrations of NO$_3$-N, NH$_4$-N, dissolved organic carbon (DOC), As, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Se, and Zn did not differ among pre-assigned BL treatments during the 3-mo period (Menjoulet et al., 2009). Similarly, mean leachate pH, EC, and oxidation-reduction potential (ORP; Pirani et al., 2006) as well as FWM leachate concentrations and loads of DOC (Pirani et al., 2006), NO$_3$-N, NH$_4$-N, Ca, K, Mg, Na, and P (Pirani et al., 2007) and FWM leachate concentrations of Mn, Ni, and Zn (Pirani et al., 2006) at the 90-cm depth did not differ among pre-assigned BL treatments. Soil pH, EC, and organic matter, total recoverable soil Cd, Cu, and Zn, and Mehlich-3 extractable soil P, K, Ca, Mg, and Na concentration also did not differ among pre-assigned BL treatments for any 10-cm soil depth interval to a depth of 90 cm (Pirani et al., 2006; Pirani et al., 2007) prior to the first litter application in 2003.

Considering the demonstrated pre-treatment similarities, plots in this study were assumed to be as uniform as reasonably could be expected based on the number of measured parameters that did not differ among pre-assigned BL treatments during the 3-mo period prior to the initial litter application in 2003 (Brye and Pirani, 2006; Pirani et al., 2006; Pirani et al., 2007; Daigh et al., 2009; McDonald et al., 2009; Menjoulet et al., 2009; McMullen, 2014a, b). Furthermore, it was also assumed that any subsequent observed differences were due to the response of imposed BL treatments after 2003 rather than to inherent differences among experimental field plots (Pirani et al., 2006; Pirani et al., 2007; Menjoulet et al., 2009).
Experimental Design

A randomized complete block design with two replications was used to evaluate BL application rate effects on soil respiration. Each of the six field plots received one of two BL application rates that were imposed annually as a single application. A low (5.6 Mg dry litter ha$^{-1}$) and high (11.2 Mg dry litter ha$^{-1}$) BL rate treatment were established based on the current University of Arkansas Cooperative Extension Service’s BL application recommendations when the study began in 2002 (Pirani et al., 2006). Since then, Arkansas’ BL application recommendations have changed and are now based on the Phosphorus Index (DeLaune et al., 2004a, b; DeLaune et al., 2006). Despite the change in recommendations, BL treatment application rates remained unchanged throughout the study in order to maintain treatment consistency over time and to allow comparisons between related concurrent studies. A control treatment received no annual BL or inorganic fertilizer.

Broiler Litter Analyses and Application

Starting in April 2003, BL was manually applied once annually to plots. Subsequent annual applications occurred approximately the first week of May each year. Litter application dates for the three years in which respiration measurements were conducted were May 7, 8, and 10 in 2009, 2010, and 2011, respectively. For the purpose of this study, a study year was designated as starting the day BL was applied in early May of one year and ending the day before BL was re-applied in the following calendar year. The BL used throughout this study had been collected from a single chicken house after production of 6 to 8 flocks, had an age ranging from 12 to 18 months, and had bedding material composed of an equal mixture of sawdust and rice (Oryza sativa L.) hulls (Pirani et al., 2006). Prior to application, BL moisture was
determined so dry-weight-equivalent amounts of BL could be calculated for each plot receiving BL. At the time of application, three BL sub-samples were collected and characterized using procedures for manure analysis (Peters, 2003). Litter pH and electrical conductivity were determined potentiometrically using a 1:2 BL-mass-to-water-volume mixture. Litter NO$_3$-N and NH$_4$-N concentrations were determined using a Skalar San Plus automated wet chemistry analyzer (Skalar Analytical B.V., The Netherlands) after extraction with 2 M potassium chloride. Total C and N were determined by high-temperature combustion using a LECO CN-2000 analyzer (LECO Corp., St. Joseph, MI). Total Ca, Cu, Fe, K, Mg, Mn, Na, P, S, and Zn were determined by inductively coupled, argon plasma mass spectrometry (ICP; CIROS CCD model, Spectro Analytical Instruments, MA) after nitric acid digestion and treatment with hydrogen peroxide. Similarly, ICP was used to determine total recoverable Al, As, Cd, Cr, Ni, and Se after being digested with nitric and hydrochloric acid, hydrogen peroxide, and heat (USEPA, 1996).

**Soil Characterization**

Since field plots used in the current study had received annual BL amendments for six consecutive years prior to the current study and considering Daigh et al. (2009) had identified 5-yr chemical changes associated with litter amendments in the same plots, it was necessary to quantify soil chemical properties prior to respiration measurements. The day before BL application in 2009, four soil samples were collected to a depth of 10 cm and combined to form a single composite sample from each plot. Sample-site voids were then filled with similar soil collected from the same depth outside the plot to eliminate preferential flow pathways within the plots. Soil was then oven dried at 70 °C for 48 hr, crushed, and passed through a 2-mm sieve. Soil pH and EC were determined potentiometrically using a 1:2 soil-to-water mixture. Soil
Mehlich-3 extractable nutrients (i.e., P, K, Ca, Mg, S, Na, Fe, Mn, Zn, and Cu) were determined using a 1:10 soil-to-extractant ratio (Tucker, 1992) and analyzed by ICP. Soil organic matter (SOM) concentration was determined by weight-loss-on-ignition at 360 °C for 2 hr (Schulte and Hopkins, 1996).

**Soil Respiration, Temperature, and Moisture Measurements**

Beginning in May 2009, soil respiration was measured weekly between April and mid-November and periodically between mid-November and the beginning of March using a portable infrared gas analyzer (LI-6400 Portable Photosynthesis System fitted with a LI-6400-09 Soil CO₂ Efflux Chamber; LI-COR Biosciences, Lincoln, NE) during a 3-yr period. Similar to procedures described in Brye et al. (2006a) and Brye and Riley (2009), three sections of thin-walled polyvinyl chloride (PVC) pipe (interior diameter = 10.1 cm; height = 5.0 cm) were inserted into the soil in each plot to a depth of 2.5 cm. The soil collars were used to support the analyzer’s soil chamber during respiration measurements. Collars were moved within plots 24 hrs prior to measurements every two weeks throughout the study. Collar positions transected each plot along the slope with a spacing of 0.5 to 1.5 m between collars during study year 2009 and a spacing of 0.5 m thereafter. Transect position was randomly determined each time collars were moved with collar position within a plot being identical in all plots for a given sample date. All green vegetation was removed by cutting with scissors 20 to 30 min prior to measurements. The vented, closed-chamber system used a pump and soda lime CO₂ trap to lower chamber CO₂ concentrations below ambient atmospheric concentrations (AAC). Measurements were made as the chamber CO₂ concentration increased from 10 ppm below AAC to 10 ppm above AAC in
response to soil respiration. An in-line desiccant removed water vapor that could interfere with measurements (McDermitt et al., 1993).

Soil temperature was measured at the 2- (T$_{2\text{cm}}$) and 10-cm (T$_{10\text{cm}}$) depth using a pencil-type thermometer adjacent to each soil collar during respiration measurements. Additionally, soil volumetric water content (VWC) was measured in the 0- to 6-cm depth interval using a Theta Probe (Theta Probe ML2x and HH2 Moisture Meter; Delta-T Devices, Cambridge, England).

**Plot Management**

Plots were regularly monitored and maintained. Above-ground biomass was cut to a height of 9 cm using a bagging push mower four times (i.e., first week of May, June, July, and September) annually. Prior to mowing each year, two randomly selected, 0.25-m$^2$ biomass subsamples were hand-collected and combined to form one sample from each plot. Samples were dried at 55 °C for 5 d in a forced-air drier and weighed for dry matter (DM) determination. Above-ground DM production was summed annually.

Precipitation was also monitored by two on-site rain gauges. A simple funnel-reservoir system collected precipitation and a micrometeorological weather station monitored wind speed, air temperature, relative humidity, total solar radiation, photosynthetically active radiation, and rainfall via a tipping bucket every 30 min.

**Calculations**

For statistical analyses, triplicate soil respiration, T$_{2\text{cm}}$, T$_{10\text{cm}}$, and VWC measurements within each plot were averaged to yield a composite measurement for each of the measured
variables for each plot on each sample date. Additionally, estimated cumulative carbon (CO$_2$-C) emissions per plot were calculated by integration of the area under the curve created by plotting soil respiration over time. Similarly, annual and 3-yr cumulative CO$_2$-C emissions were calculated for each plot.

**Statistical Analysis**

Analysis of variance (ANOVA) was used to evaluate BL application rate effects on initial soil characteristics, annual DM production, annual CO$_2$-C emissions, and 3-yr cumulative CO$_2$-C emissions using the PROC MIXED procedure in SAS (version 9.2, SAS Institute Inc., Cary, NC). When appropriate, study year was included in the analyses to evaluate study year effects on dependent variables. Since sample dates differed across study years, an ANOVA based on a split-split plot design with sample date within study year was used to evaluate BL application rate, sample date within year, and study year effects on soil respiration, T$_{2cm}$, T$_{10cm}$, and VWC with blocks treated as a random variable. When appropriate, means were separated using a protected least significant difference (LSD) at $\alpha = 0.05$. Pearson’s linear correlation analysis using the PROC CORR procedure in SAS and regression analysis using JMP (version 9.0.0, SAS Institute, Inc.) were conducted to explore the relationships between soil respiration and T$_{2cm}$, T$_{10cm}$, VWC, their quadratic terms, and related interactions.

**Results and Discussion**

**Initial Soil Characteristics**

The day prior to BL application in 2009, soil pH, EC, and Mehlich-3-extracable Ca, Mg, S, Na, Fe, Mn, and Zn concentrations did not differ ($P > 0.05$) among BL treatments in the top
Averaged over litter treatment, mean soil pH was 6.88 (SE = 0.04) and EC was 0.12 (SE = 0.01) dS m⁻¹. Similarly, averaged over litter treatments, mean soil Mehlich-3-extractable Ca, Mg, S, Na, Fe, Mn, and Zn concentrations were 1387 (SE = 88), 168 (SE = 22), 14.0 (SE = 0.7), 12.9 (SE = 1.4), 206 (SE = 7), 183 (SE = 7), and 19.8 (SE = 3.1) mg kg⁻¹, respectively. However, as somewhat expected, SOM and Mehlich-3-extractable P, K, and Cu concentrations differed (P < 0.05) among BL treatments in the top 10 cm. Mean SOM concentrations were 3.1, 4.3, and 3.8 % in the unamended control, low- and high-BL treatments, respectively, with the control differing from the low-BL treatment, while the high-BL treatment was similar to both the control and low-BL treatments. Daigh et al. (2009) reported similar SOM results in a concurrent 5-yr study in the top 10 cm with the low- and high-litter treatments being similar and increasing 29 % while the unamended control SOM did not change during the 5 yrs. Daigh et al. (2009) also reported that the SOM was similar among BL treatments and did not change during the 5-yr period at soil depths greater than 10 cm. Six consecutive years of annual BL amendments increased mean soil Mehlich-3-extractable P, K, and Cu concentrations in the low- and high-BL treatments relative to the unamended control. Mean soil-P concentrations were 156, 310, and 411 mg kg⁻¹ in the control, low-, and high-BL treatments, respectively. Mean soil-K concentrations were 116, 206, and 235 mg kg⁻¹ in the control, low-, and high-BL treatments, respectively. Mean soil-Cu concentrations were 4.1, 10.4, and 12.3 mg kg⁻¹ in the control, low-, and high-BL treatments, respectively.

Additionally, soil respiration, temperature, and VWC were measured the day prior to BL application in 2009. Similar to other soil properties, soil respiration, T₂cm, T₁₀cm, and VWC did not differ (P > 0.05) among BL treatments. Averaged over BL treatments, mean soil respiration was 11.5 (SE = 1.6) µmol CO₂ m⁻² s⁻¹, mean T₂cm and T₁₀cm were 22.2 (SE = 0.5) and 17.9 (SE =
0.1 °C, respectively, and mean VWC was 0.39 (SE = 0.02) cm$^3$ cm$^-3$. These results suggest that six consecutive years of continuous management and at 364 DPBL soil respiration, temperature, and VWC were similar among BL treatment conditions.

**Broiler Litter Composition**

The mean annual composition of the BL used throughout the 9-yr study was 35.7% C, 4.3% N, 3.7% Ca, 3.5% K, and 2.2% P on a dry-weight basis and had 25% moisture by mass when applied (Table 4-1). Similar BL compositions have been reported by other researchers (Kleinman et al., 2005; Brock et al., 2007; Adams et al., 1997; Agele et al., 2004). The mean annual BL C:N ratio was 8.2 averaged over the 9-yr period and suggested that BL decomposition by soil microorganisms would have been relatively quick with a likely net increase in soil N levels that would have promoted plant growth. During the three years when respiration was monitored, BL moisture, EC, NO$_3$-N, NH$_4$-N, and total P, Ca, Mg, S, Na, Mn, Zn, Cu, Ni, Cd, and Cr concentrations did not differ ($P > 0.05$) among study years (Table 4-1). However, annual BL composition differed ($P < 0.05$) by study year for litter pH and total C, N, K, Al, Fe, B, As, and Se (Table 4-1). Although BL composition differences over time were not unexpected, differences in BL total N and C contents may influence respiration. The C:N ratios for study years 2009, 2010, and 2011 were 8.4, 8.1, and 9.4, respectively, with the 2011 C:N ratio differing ($P < 0.01$) from 2009 and 2010, which were similar to each other.

**Dry Matter Production**

Annual above-ground DM production ranged from a low of 7.4 Mg ha$^-1$ in the unamended control in 2011 to a high of 21.6 Mg ha$^-1$ in the high-litter treatment in 2010 when
above average annual precipitation occurred. During the 3-yr study, annual precipitation at the study site averaged 1260 (SE = 138) mm, which was 2.3% above the 30-yr mean annual precipitation for Fayetteville, AR (1232 mm; NOAA, 2013). Annual precipitation was 1033, 1508, and 1239 mm during 2009, 2010, and 2011, respectively. Similar to previous studies (Huneycutt et al., 1988; Brye et al., 2010) and as would be expected, additions of BL increased DM relative to the unamended control ($P = 0.05$). Averaged over the 3-yr study, annual DM production was 9.9, 12.3, and 18.4 Mg ha$^{-1}$ for the unamended control, low- and high-BL treatments, respectively, with DM being greater in the high-BL treatment relative to the control, and the low-BL treatment being similar to both the control and high. During the first six years following the initial BL application and before the current study, Brye et al. (2010) reported that DM production increased over time for both the low- and high-BL treatments, while DM in the unamended control did not change over time. In a concurrent study spanning 8 years, McMullen (2014a) reported no difference in annual DM production from 2008 to 2010, presumably due to shifts in plant species that yielded greater biomass. McMullen (2014a) also reported that annual DM and annual precipitation were positively correlated ($r = 0.45$, $P < 0.01$) at this location during the 8-yr period ending May 10, 2011.

**Soil Respiration Rates**

Of the 100 samples dates during the 3-yr study, respiration ranged from a low of 0.7 µmol CO$_2$ m$^{-2}$ s$^{-1}$ on 124 DPBL in fall 2011 in the low-BL treatment to a high of 30.7 µmol CO$_2$ m$^{-2}$ s$^{-1}$ on 5 DPBL in summer 2010 in the high-BL treatment (Figure 4-1). Similar to Ussiri and Lal (2009), soil respiration rates had seasonal patterns and were greater in the summer (i.e., May to July) and lower in the winter (i.e., November to January) seasons. Soil respiration rates
peaked immediately after BL application in littered treatments in all three study years. Similarly, soil respiration rates also peaked following rain events that increased soil VWC (Figure 4-2) following dry periods and was most pronounced in littered treatments. Averaged over BL treatment and sample date, mean annual respiration was 8.0, 7.8, and 6.4 µmol CO$_2$ m$^{-2}$ s$^{-1}$ in 2009, 2010, and 2011, respectively, with 2011 respiration being lower ($P < 0.01$) than 2009 and 2010, which were similar to each other. Similar to soil respiration ranges, daily measurements and annual differences in respiration have been reported from silt-loam soils in eastern Arkansas (Motschenbacher, 2012; Smith, 2013). Smith (2013) reported a soil respiration range of 0.17 to 40.7 µmol CO$_2$ m$^{-2}$ s$^{-1}$ during two growing seasons in a wheat- (Triticum aestivum L.) soybean [Glycine max (L.) Merr.], double-crop production system. Motschenbacher (2012) reported the largest soil surface CO$_2$ flux of 9 µmol CO$_2$ m$^{-2}$ s$^{-1}$ in a rice- (Oryza sativa L.) soybean (Glycine max L.) rotation, under no-tillage conditions, during a soybean growing season.

Soil respiration differed among BL application rates across sample dates within years ($P < 0.01$) and differed among years ($P < 0.01$; Table 4-2; Figure 4-1). Comparing BL application rate effects on individual sample dates, BL treatment effects were observed on 34, 18, and 29 % of all sample dates within a given year in 2009, 2010, and 2011, respectively (Figure 4-1). Broiler litter treatment effects were observed on the first two sample dates following BL application in all three years and persisted until 49, 26, and 15 DPBL in 2009, 2010, and 2011, respectively. Additionally, 18 of the 27 dates when BL treatment effects were observed throughout the study occurred during the summer within 84 DPBL. The remaining treatment effects were observed again in fall 2009 and fall and spring 2011 (Figure 4-1). Twenty-five of the 27 dates when BL treatment effects were observed had a similar respiration pattern with annual amendments of BL increasing respiration relative to the unamended control. This pattern
differed during the summers of 2009 and 2010 on 70 and 46 DPBL, respectively, when soil respiration from the low-litter treatment was lower than that from the unamended control and high-litter treatments. Since \( T_{2\text{cm}} \), \( T_{10\text{cm}} \), and VWC never differed \((P > 0.05)\) among BL treatments on any given sample date during the 3-yr study (Table 4-2; Figure 4-2), it was assumed that these environmental factors were consistent among litter treatments on any given sample date and thus, differences in soil respiration within a sample date must be due to microbial or plant root respiration associated with BL application rate. Since measurements occurred during the day it may also be assumed that root respiration would be minimal with the exception of overcast days when increased root biomass associated with increased BL application rate might increase root respiration in the littered treatments relative to the control. Therefore, the observed differences in soil respiration among litter treatments within a given sample date must be related to BL-related-C oxidation by soil microorganisms.

Similar to soil respiration, soil temperature at the 2-cm depth differed among sample dates within study years \((P < 0.01; \text{Table 4-2; Figure 4-2})\) and differed among BL application rate within study years \((P < 0.05; \text{Table 4-2; Figure 4-3})\). During the 3-yr study, \( T_{2\text{cm}} \) ranged from a low of 4.3 °C on 210 DPBL during winter 2009 to a high of 36.5 °C on 80 DPBL during the end of summer 2011 (Figure 4-2). Averaged over sample date, mean annual \( T_{2\text{cm}} \) was lower in all BL treatments in 2009 than in 2010 or 2011 (Figure 4-3). In 2009 and 2010, \( T_{2\text{cm}} \) was lower in the high-litter treatment relative to the unamended control, possibly because increased DM production may have increased soil shading, thus reducing near-surface soil temperatures. In 2011, \( T_{2\text{cm}} \) was similar among all BL treatments. Although mean annual respiration in 2009 and 2010 (8.0 and 7.8 μmol CO\(_2\) m\(^{-2}\) s\(^{-1}\), respectively) were similar \((P > 0.05)\), mean annual \( T_{2\text{cm}} \) differed among the two years \((P < 0.01; \text{Figure 4-3})\) and as mean annual \( T_{2\text{cm}} \) increased from
2010 to 2011, mean annual soil respiration decreased \((P < 0.01)\) to 6.4 \(\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}\), suggesting that as a warming climate increases mean annual near-surface soil temperature to approximately 22 \(\degree\text{C}\), significant reductions in annual soil respiration can be expected.

In contrast to \(T_{2\text{cm}}\), \(T_{10\text{cm}}\) was unaffected by BL application rate \((P > 0.05)\), but similar to \(T_{2\text{cm}}\), \(T_{10\text{cm}}\) differed among sample dates within study years \((P < 0.01; \text{Table 4-2}; \text{Figure 4-2})\). During the 3-yr study, \(T_{10\text{cm}}\) ranged from a low of 3.8 \(\degree\text{C}\) on 250 DPBL during winter 2011 to a high of 33.8 \(\degree\text{C}\) on 96 DPBL during fall 2011. Similar to \(T_{2\text{cm}}\) and averaged over sample date, mean annual \(T_{10\text{cm}}\) differed each year \((P < 0.01)\) and averaged 17.8, 19.2, and 19.4 \(\degree\text{C}\) for 2009, 2010, and 2011, respectively. As expected, soil temperature at both the 2- and 10-cm depths were greatest during the late summer or early fall, then decreased during the fall, were lowest during the winter when soil respiration was also lowest, and finally increased during the spring. As expected, soil temperatures at the two depths were highly correlated \((r = 0.97; P < 0.01)\) and followed similar sinusoidal patterns with \(T_{10\text{cm}}\) lagging behind \(T_{2\text{cm}}\) (Figure 4-2).

Similar to reports by other researchers (Ding et al., 2010; Ruehr et al., 2010; Brown et al., 2009; Brye et al., 2006a; Jones et al., 2006; Fang and Moncrieff, 2001) soil respiration was positively \((P < 0.01)\) correlated to both \(T_{2\text{cm}}\) \((r = 0.49; \text{Table 4-3})\) and \(T_{10\text{cm}}\) \((r = 0.52)\). Although the correlation reported in the current study includes a 3-yr period, other researchers have reported this relationship to vary by season (Ruehr et al., 2010; Brown et al., 2009) and time of day (Ruehr et al., 2010). Diurnal hysteresis effects have also been observed (Pingintha et al., 2010; Ruehr et al., 2010) and differ by time of year (Ruehr et al., 2010) and soil depth (Pingintha et al., 2010). In a study of prairie restoration age affects on near-surface soil properties in northwest Arkansas, Brye and Riley (2009) reported a temporal effect on soil surface CO\(_2\) flux and a positive correlation between CO\(_2\) flux and soil temperature at both 2- and 10-cm depths.
Similarly, positive correlations between respiration and soil temperature have been reported in cultivated and non-cultivated row-crop ecosystems in eastern Arkansas (Motschenbacher, 2012; Smith, 2013). Additionally, nonlinear relationships have also been used to explain soil respiration using soil temperature variations (Ruehr et al., 2010; Brown et al., 2009; Jones et al., 2006; Fang and Moncrieff, 2001; Lloyd and Taylor, 1994).

Similar to T$_{10cm}$, soil VWC in the top 6 cm was unaffected by BL application rate ($P > 0.05$), but differed among sample dates within study years ($P < 0.01$; Table 4-2; Figure 4-2). During the 3-yr study, VWC ranged from a low of 0.03 m$^3$ m$^{-3}$ during late summer and early fall 2010 and 2011 to a high of 0.41 m$^3$ m$^{-3}$ on 15 DPBL during summer 2011 (Figure 4-2), which also corresponded to the third highest BL-induced respiration rate observed during the 3-yr study (Figure 4-1). Averaged over BL treatment, mean annual VWC was 0.27, 0.20 and 0.19 m$^3$ m$^{-3}$ for study years 2009, 2010, and 2011, respectively, and differed ($P < 0.01$) among all years, with the driest year of 2011 also being the year with the least precipitation, highest mean annual T$_{2cm}$ and T$_{10cm}$, and lowest mean annual soil respiration. Additionally and as expected, VWC varied by sample date within study year ($P < 0.01$), but did not differ among litter treatments either annually ($P > 0.05$) or on any given sample date ($P > 0.05$), suggesting that additions of BL did not influence soil moisture holding capacity.

Similar to other researchers (Ding et al., 2010; Brye et al., 2006a; Al-Kaisi and Yin, 2005), VWC was not correlated ($P > 0.05$) to soil respiration. Additionally, respiration was not correlated ($P > 0.05$) to the quadratic VWC term as hypothesized (Table 4-3). In contrast, other researchers have reported both positive (Pingintha et al., 2010; Brown et al., 2009) and negative (Brye et al., 2006b; Jones et al., 2006) correlations between soil moisture and soil respiration. Brye and Riley (2009) reported a quadratic relationship between WFPS and soil respiration with
respiration being greatest at 50% WFPS, while Jones et al. (2006) reported two linear regressions relating soil respiration and soil moisture. Jones et al. (2006) first used a positively correlated ($R^2 = 0.30$) regression line when VWC ranged between 0.1 and 0.3 m$^3$ m$^{-3}$, then used a negatively correlated ($R^2 = 0.12$) regression line when VWC exceeded 0.3 m$^3$ m$^{-3}$ (Jones et al., 2006). Similarly, Linn and Doran (1984) reported microbial activity to be water-limited when WFPS was less than 60 % and to be oxygen-limited when WFPS exceeded 60 %. In general, soil microbial activities, including respiration, have an optimal soil moisture range. At low soil moisture, nutrient diffusion to soil microbes may limit respiration. At very low soil moisture, internal osmotic regulation within soil microbes may limit respiration and at extreme levels, cell lysis may occur, thus decreasing soil respiration. On the other hand, at high soil moisture, diffusion of O$_2$ to soil microbes may limit respiration and at extreme moisture, microbes may be forced to utilize other terminal electron acceptors due to the lack of oxygen or die, thus reducing soil respiration once again.

As expected, VWC increased after precipitation events (Figure 4-2). Soil respiration also tended to increase after precipitation, especially among littered treatments. Precipitation-induced peak respiration was observed during the current study in late summer through early fall 2009, in late summer and mid- and late-fall 2010, and again in early and late fall 2011 (Figures 4-1 and 4-2). Similar precipitation-induced peak respiration has been reported by other researchers using laboratory incubation studies (Fierer and Schimel, 2003; Lee et al., 2004; Xu et al., 2004) and in situ soil measurements (Chen et al., 2008; Davidson et al., 1998; Motschenbacher, 2012; Roberson et al., 2008; Smith, 2013; Xu et al., 2004). Additionally, ecosystem respiration has been reported to increase after precipitation events using eddy covariance measurements (Lee et al., 2004; Xu et al., 2004).
Chen et al. (2008) reported increased soil respiration with as little as 5 mm of simulated precipitation one day after treatment in both grazed and ungrazed steppes in a semiarid region of China. The magnitude of the precipitation-induced peak respiration rate increased with increased precipitation in the ungrazed treatment with the greatest peak rate occurring with 100 mm of precipitation (Chen et al., 2008). Although treatments receiving 5 mm of precipitation increased respiration one day after treatment, precipitation amounts of 50 and 100 mm still had greater soil respiration 11 d-post-precipitation relative to the control, which received no precipitation (Chen et al., 2008). Chen et al. (2008) concluded that rain events > 10 mm increased soil respiration rates and that grazing had no affect on precipitation-induced peak respiration. Similar results were observed in an incubation study, where Xu et al. (2004) reported that as soil gravimetric water content decreased from 0.3 to 0.1 g g\textsuperscript{-1}, incubation half-life (i.e., the number of days for a 50 \% reduction in respiration) increased from 19 to 61 d, respectively. Xu et al. (2004) interpreted these results as evidence that soil microbes were consuming a relatively labile C pool after rain events. In an incubation study using \textsuperscript{14}C-labeled glucose, Fierer and Schimel (2003) suggested that the moisture-induced CO\textsubscript{2} emission pulse was from the mineralization of microbial biomass-C; although, no evidence of microbial cell lysis was observed upon rewetting of dry soil.

Of the 27 sample dates on which BL treatment effects on soil respiration were observed, eight sample dates coincided with precipitation-induced peak respiration. If the first two or three sample dates after litter application in each year when litter effects were observed were due to the rapid and intermediate phases of BL decomposition as described by Gale and Gilmore (1986), then that leaves 19 sample dates where litter effects were observed, of which 8 were associated with precipitation-induced peak respiration. These results suggest that the slow phase
of decomposition (Gale and Gilmore, 1986) of BL may be stimulated by precipitation events. Furthermore, and as suggested by Fierer and Schimel (2003), the increase in soil respiration after precipitation observed in the current study may be related to the release of physically protected SOM upon re-wetting. As previously mentioned, six consecutive years of annual BL applications prior to respiration measurements in the current study resulted in increased SOM in the low-litter treatment relative to the unamended control. It is likely that the source of C released as precipitation-induced peak respiration originates from this BL-induced increase in SOM.

Previous researchers have reported increased soil respiration after BL application to a temperate grassland (Jones et al., 2006), increased CO$_2$ efflux in a conventional tillage system after BL application at a rate of 100 kg N ha$^{-1}$ (Roberson et al., 2008), increased CO$_2$ flux at 15 DPBL in eastern Arkansas from a silt loam (Brye et al., 2006a), and increased CO$_2$ efflux after incorporation of turkey (*Meleagris gallopavo*) litter to a depth of 15 cm and left fallow (Pengthamkeerati et al., 2005). These results in conjunction with the results of the current study suggest that amendments of BL increase soil respiration for at least two weeks after application and maybe longer depending on the year. In a laboratory mineralization study of BL (i.e., litter-pine shavings mixture, similar to that used in the current study) conducted at 25 °C, Gale and Gilmore (1986) reported all but 1.5 % of the C added in BL was evolved as CO$_2$ by the end of the rapid phase of decomposition, which occurred between 5.9 to 7 DPBL application. Additionally, Gale and Gilmore (1986) reported that 1.4 % of the added C remained after 14 DPBL when the end of the intermediate phase of decomposition occurred. Other researchers (Jones et al., 2005; Adams et al., 1997) have also reported increased soil respiration with additions of BL.
Adams et al. (1997) attempted to increase photosynthesis within closed crop canopies by increasing canopy CO₂ concentrations with amendments of BL slurry. Five slurry treatments and a control receiving water were imposed on artificially packed soil columns. Adams et al. (1997) reported a 3- to 8.5-hr lag time between when BL slurry was added and when elevated respiration commenced. Repeated applications of slurry increased the time before respiration commenced and was presumably related to soil crusting (Adams et al., 1997). Litter slurry inoculated with fresh BL slurry and aged for seven days had the shortest time period before respiration commenced, the greatest respiration rate, and the greatest total amount of cumulative CO₂ released when compared to all other treatments.

During a 3-yr study in Scotland, CO₂ production from a grassland was monitored in response to three organic manures (BL, cattle slurry and sewage sludge pellets) and two inorganic fertilizers (urea and a compound containing ammonium nitrate) applied to the soil surface (Jones et al., 2006; Jones et al., 2005). Soil respiration responded similarly among inorganic fertilizers and the unamended control (Jones et al., 2006; Jones et al., 2005). During the first two years, cattle slurry and BL had greater cumulative soil respiration relative to the control (Jones et al., 2005), but were similar to the control in the third year (Jones et al., 2006). Sewage sludge pellets increased cumulative soil respiration only during the second year (Jones et al., 2005). During the first two years, the greatest respiration rates were from BL and sludge pellets which occurred during the summer, within a month of treatment application (Jones et al., 2005). These results, in conjunction with the results of the current study, suggest that BL amendment effects on respiration may be variable from year to year.

Brye et al. (2006a) evaluated the effects of fresh and pelletized BL applied at five different application rates on soil respiration from two silt-loam soils used for rice production in
eastern Arkansas. Brye et al. (2006a) reported no difference in respiration among BL types, but observed differences among locations. Additionally, Brye et al. (2006a) reported that BL application rate had no effect on soil respiration, except for on the first sample date 15 d after litter application when respiration increased as BL application rate increased. Although Brye et al. (2006a) suggested that the BL application rate effect could have been related to tillage, their results were similar to those reported in the current study, with treatment effects occurring shortly after BL application and then becoming less prominent over time within the same 1-yr period.

**Cumulative Carbon Emissions**

Within 50 days after litter application each year, marked increases in CO$_2$-C emissions were observed (Figure 4-4). Similar increases occurred again during mid- to late-spring as warmer temperatures stimulated soil respiration. In contrast to Jones et al. (2005, 2006), estimated annual CO$_2$-C emissions were unaffected ($P > 0.05$) by BL application rate or the interaction between BL application rate, but differed ($P < 0.01$) among study years. Averaged over BL treatments, estimated annual CO$_2$-C emissions differed each year and averaged 24.9, 21.7, and 20.2 Mg CO$_2$-C ha$^{-1}$ for 2009, 2010, and 2011, respectively. As would be expected, 2009, the year with the greatest estimated annual CO$_2$-C emissions, was also the year when mean annual respiration averaged over BL treatment and sample date was the greatest. Though somewhat unexpected, 2009 was also the year with the coolest soil temperatures and largest soil VWC. In general, wetter, cooler soils are associated with reduced respiration and increased soil organic matter, but it appears that soil VWC and temperature combinations throughout 2009 were more optimal for microbial activity than that in the other two years.
In contrast to the current study, annual CO$_2$-C emissions from a temperate grassland have been reported to increase with BL amendments relative to an unamended control on a sandy clay loam in Scotland (Jones et al., 2005, 2006). Jones et al. (2006) reported annual CO$_2$-C emissions from an unamended control to be 7.0, 9.2, and 5.6 Mg CO$_2$-C ha$^{-1}$ while emissions from the BL-amended treatment was 11.5, 13.8, and 7.0 Mg CO$_2$-C ha$^{-1}$ for 2002, 2003, and 2004, respectively. The annual CO$_2$-C emissions reported in the current study were twice that reported by Jones et al. (2006). However, Brown et al. (2009) reported 19.5 Mg CO$_2$-C ha$^{-1}$ for the growing season from a grazed pasture in New Zealand and Mielnick and Dugas (2000) reported 14 Mg CO$_2$-C ha$^{-1}$ for the entire year from a tallgrass prairie. Annual CO$_2$-C emissions have been reported from 4.0 to 12.5 Mg CO$_2$-C ha$^{-1}$ for forest ecosystems (Phillips et al., 2010; Ruehr et al., 2010) and from 0.3 to 9.2 Mg CO$_2$-C ha$^{-1}$ for agricultural ecosystems (Ding et al., 2007; Ding et al., 2010; Meijide et al., 2009; Motschenbacher, 2012; Smith, 2013; Ussiri and Lal, 2009).

Estimated 3-yr cumulative CO$_2$-C emissions were unaffected ($P = 0.07$) by BL application rate. However, a numeric increasing trend in 3-yr cumulative CO$_2$-C emissions was observed with increased BL application rate. Three year cumulative CO$_2$-C emissions were 57.5 (SE = 1.9), 67.0 (SE = 1.2), and 75.6 (SE = 2.3) Mg CO$_2$-C ha$^{-1}$ for the unamended control, low- and high-litter treatments, respectively, which were similar to each other. Averaged over BL treatments, 3-yr CO$_2$-C emission was 66.7 (SE = 3.4) Mg CO$_2$-C ha$^{-1}$.

Treating BL as a continuous variable and applying simple regression techniques to annual cumulative CO$_2$-C emissions and not treating blocks as a random variable resulted in a significant ($P < 0.01$, $R^2 = 0.52$) relationship with a y-intercept (19.2 Mg CO$_2$-C ha$^{-1}$) that differed from zero ($P < 0.01$). The y-intercept of the relationship between BL application rate
and annual CO₂-C emission represents the mean annual C lost from the unamended control during the 3-yr study and is a reasonable estimate of C loss from a similar silt-loam under similar management and climatic conditions. The slope of the relationship suggested that for every Mg of BL (dry weight basis) ha⁻¹ added to soil annually would increase annual CO₂-C emissions by 0.537 (SE = 0.128) Mg ha⁻¹. Since the mean 3-yr C content of the BL used in this study was only 34.8 % (Table 4-1), the addition of BL appears to have stimulated soil respiration in excess of the amount of C added as BL. This excess amount of C loss may be from plant root respiration associated with BL-induced increases in below-ground root biomass.

During 2009 and 2010, the 2-yr cumulative BL-derived-C added to the soil was 3.9 and 7.8 Mg ha⁻¹ in the low- and high-BL treatments, respectively (Table 4-4). Although environmentally important to surface and groundwater quality, C losses as runoff and drainage below the 90-cm soil depth were minimal relative to C stored in harvested DM or as CO₂ released to the atmosphere. Although considered a loss from the pasture ecosystem, harvested DM may be considered a C storage mechanism relative to the atmosphere. By assuming 43.3 % C concentration for harvested DM (Sanaullah et al., 2014; Niknahad-Gharmakher et al., 2012; Bélanger et al., 1994), 9.7, 12.1, and 22.5 Mg of DM-C ha⁻¹ was stored in the unamended control, low- and high-BL treatments, respectively, during the 2-yr period (Table 4-4). Annual amendments of BL increased DM-C storage by 1.3 and 2.3 times the control values in the low- and high-litter treatments, respectively. Jones et al. (2006) reported a 2.6 times increase over the unamended control when BL was applied at a rate of 300 kg N ha⁻¹ yr⁻¹. For comparison, the current study applied BL at a rate of 241 and 482 kg N ha⁻¹ in the low- and high-litter treatments, respectively, in both 2009 and 2010. Similar to DM-C storage, increased BL application rate also increased soil respiration. By subtracting DM-C storage and BL-C soil input from CO₂-C
emission, the net release of C to the atmosphere was 30.4, 30.9, and 22.2 Mg C ha\(^{-1}\) in the unamended control, low- and high-BL treatments, respectively, during the 2-yr period. These results suggest that BL-induced DM production may reduce net C release to the atmosphere. Additionally and as previously reported in the current study, six consecutive years of BL application in the low-litter treatment increased SOM relative to the unamended control prior to 2009. These results indicate that organic amendments, such as BL, can stimulate the release of CO\(_2\) from soil to the atmosphere, but that the potentially negative effect to the atmosphere greenhouse gas concentration may be offset by increased DM production. Furthermore, since SOM at the highest litter rate was similar to the control and because soil surface application of BL can adversely affect the environment it is possible that there may be BL application rates above which the benefits of organic amendments may be diminished by potential adverse environmental effects. Additionally, alternative BL management strategies may lessen CO\(_2\) loss from the soil. Incorporation of BL similar to Pote et al. (2003) to a depth of 8 cm using a knife might reduce CO\(_2\) release at the soil surface. Pote et al. (2003) reported that incorporation of BL reduced NH\(_4\)-N loss during the first simulated and first natural runoff events following litter application relative to surface applied BL (5.6 Mg BL ha\(^{-1}\)). Pote et al. (2003) attributed the reduction in NH\(_4\)-N to favorable soil conditions that promoted nitrification.

**Respiration Dependence on Temperature and Moisture**

Although BL application rate and sample date affected respiration, they were eliminated from the predictive model development process with the rationale that the predictive equation should be based solely on environmental factors so as to facilitate universal application. Respiration was positively \((P < 0.01)\) correlated with \(T\_2\text{cm}\) \((r = 0.49; \text{Table 4-3})\), \(T\_1\text{cm}\) \((r = 0.52;\)
data not shown), the T$_{2cm}$ quadratic term ($r = 0.42$), and the term representing an interaction between T$_{2cm}$ and VWC ($r = 0.60$). In contrast, respiration was uncorrelated ($P > 0.05$) with VWC or its quadratic term. Similar correlations have been reported by other researchers for soil temperature (Ding et al., 2010; Ruehr et al., 2010; Brown et al., 2009; Brye et al., 2006a; Jones et al., 2006; Fang and Moncrieff, 2001) and VWC (Ding et al., 2010; Brye et al., 2006a; Al-Kaisi and Yin, 2005). Since T$_{2cm}$ and T$_{10cm}$ were highly correlated with one another ($r = 0.97; P < 0.01$), T$_{10cm}$ was excluded as a model parameter in order to avoid the occurrence of multicollinearity (also called collinearity or intercorrelation) within the finished model.

When combined across BL treatments, sample dates, and study years (i.e., the all-treatments model), T$_{2cm}$ and the interaction between T$_{2cm}$ and VWC explained 52% of the variation in observed respiration ($P < 0.01$; Table 4-5). The predictive expression,

$$R_s = -3.45 + (0.28 \times T_{2cm}) + (1.25 \times T_{2cm} \times VWC)$$

related soil 2-cm temperature (T$_{2cm}$, °C) and 0- to 6-cm volumetric water content (VWC, m$^3$ m$^{-3}$) to soil respiration ($R_s$, µmol CO$_2$ m$^{-2}$ s$^{-1}$). Although the overall model and regression coefficient estimates were significant ($P < 0.01$), a residual plot by predicted values suggested that the error variance increased as predicted values of the dependent variable increased. Similarly, a residual plot by sample date suggested that the error term was dependent on time, with error variance increasing shortly after BL application in study years 2009 and 2010 and then decreasing and averaging around zero until the next litter application in the following year. The observed trends in residuals suggest that a more accurate model for predicting soil respiration may exist.

Combined across BL treatments, sample dates, and years, the all-treatments model had 62 % of the sum of squares associated with the interaction between T$_{2cm}$ and VWC, suggesting that the interaction between soil temperature and moisture was a better predictor of respiration than
soil temperature or moisture alone. In contrast, other researchers (Brye et al. 2006b; Motschenbacher, 2012; Smith, 2013) have concluded that soil temperature explained more of the variation in respiration than did soil moisture. Additionally, Brye and Riley (2009) used soil temperature at the 2-cm depth and the linear and quadratic soil moisture terms to explain 19% of the observed variability in soil surface CO₂ flux in a temperate tallgrass prairie.

When the resulting equation from the all-treatments model was applied separately to data sets for each BL treatment, the resulting models were highly significant (P < 0.01) and all coefficient estimates differed from zero (P < 0.01) for all BL treatments (Table 4-5). Broiler litter treatment model coefficient estimates were within the 95% confidence intervals (CI) for the all-treatments model intercept and T_{2cm} coefficient estimates (Table 4-5), suggesting that one-time-annual additions of BL to pasture soil with a history of BL applications may not influence the intercept and 2-cm soil temperature predictive model estimates. In contrast, all BL treatment model coefficient estimates for the interaction between T_{2cm} and VWC fell outside the 95% CI of the all-treatments model coefficient (Table 4-5). The unamended control model estimate for the interaction between T_{2cm} and VWC was less than the all-treatments model estimate, while that for the low- and high-litter treatment models were greater than the all-treatments model estimate, suggesting that annual additions of BL may increase predictive model estimates relating to the 2-cm soil temperature and 0- to 6-cm soil moisture content interaction.

Since the y-intercept and T_{2cm} coefficient estimates among BL treatments were similar to the all-treatments model y-intercept and T_{2cm} coefficient estimate, and even though the all BL treatment model coefficient estimates for the interaction between T_{2cm} and VWC may have differed from the all-treatments model T_{2cm} x VWC coefficient estimate, the all-treatments model, as well as the BL treatment models, were all significant and all had a moderately good
coefficients of determination (Table 4-5). Therefore, it is reasonable to assume that soil respiration from a pasture soil with similar characteristics to the one reported here (i.e., silt-loam, siliceous, active, mesic, mixed tall fescue) could adequately be predicted with a regression model using 2-cm soil temperature and the product term of soil temperature and 0- to 6-cm VWC regardless of litter rate.

Summary and Conclusions

After six to eight consecutive years of annual BL amendments to soil with a history of organic amendments, this study demonstrated that BL application rate effects on soil respiration varied by sample date and among years and was significantly correlated with environmental factors. Litter application increased respiration relative to the unamended control immediately after application in all three study years. Of the 100 sample dates during the study, BL treatment effects were observed on 27 dates, of which 18 dates occurred within 84 DPBL application. Litter application also increased respiration relative to the unamended control when rainfall events occurred after dry periods, and suggests that the slow phase of BL decomposition may be stimulated by rainfall.

Both $T_{2cm}$ and $T_{10cm}$ varied by season and differed among years. Averaged over sample dates, $T_{2cm}$ was lower in the high-BL treatment relative to unamended control in 2009 and 2010, and was attributed to increased shading associated with BL-induced increases in DM production in those years. Soil temperature at the 10-cm depth was unaffected by BL application rate.

Estimated annual CO$_2$-C emissions were unaffected by BL application rate, but differed among years. Similarly, 3-yr cumulative CO$_2$-C emissions were unaffected by BL; however, an increasing trend in 3-yr cumulative CO$_2$-C emissions was observed as BL application rate increased. The linear relationship relating BL application rate to annual emissions predicted that
an unamended pasture soil with a history of organic amendments will release 19.2 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$ and that for every 1 Mg of BL ha$^{-1}$ applied annually will increase this baseline emission estimate by 0.5 Mg CO$_2$-C ha$^{-1}$.

Regression also indicated that soil respiration could be predicted using near-surface soil temperature (T$_{2cm}$) and the product of T$_{2cm}$ and VWC as predictors with moderate strength ($R^2 = 0.52$) in the relationship. Residual-plot patterns suggested that a more accurate model for predicting soil respiration may exist. When the model was applied to BL treatment data sets separately, coefficient estimates for the y-intercept and T$_{2cm}$ were within the 95 % CI of the all-treatments model. However, the coefficient estimate for the product of T$_{2cm}$ and VWC fell outside the 95 % CI of that for the all-treatments model suggesting that long-term applications of BL may influence the interaction between near-surface soil temperature and VWC as it relates to respiration.

Results indicate that organic amendments, such as BL, can stimulate CO$_2$ release from soil to the atmosphere under similar agricultural and climatic conditions in the Ozark Highlands region of northwest Arkansas, potentially negatively affecting atmospheric greenhouse gas concentrations and increasing radiative forcing. It is possible that increased CO$_2$ uptake associated with BL-induced increases in DM production may offset BL-related increases in CO$_2$ to the atmosphere that were observed in the current study. Furthermore, rainfall events following dry conditions may stimulate slow-phase decomposition of BL. Improved BL management strategies are needed to lessen the loss of CO$_2$ from BL-amended soils.
References


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Table 4-1. Mean annual broiler litter (BL) composition and element/compound applied over a 9-yr period. Litter was hand applied once annually to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas at three application rates (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively). Annual means for the last three years of the study are provided and coincide with years when soil respiration was measured.

<table>
<thead>
<tr>
<th>Litter Property</th>
<th>Mean Annual Composition</th>
<th>2009</th>
<th>2010</th>
<th>2011</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moisture (kg kg(^{-1}))</td>
<td>0.25</td>
<td>0.30(^a)</td>
<td>0.29(^a)</td>
<td>0.29(^a)</td>
</tr>
<tr>
<td>pH</td>
<td>8.2</td>
<td>8.67(^a)</td>
<td>8.68</td>
<td>6.68(^b)</td>
</tr>
<tr>
<td>EC (dS m(^{-1}))</td>
<td>12.2</td>
<td>14.4(^a)</td>
<td>14.8</td>
<td>14.6(^a)</td>
</tr>
<tr>
<td>NO(_3)-N (mg kg(^{-1}))</td>
<td>243</td>
<td>416(^a)</td>
<td>513</td>
<td>528</td>
</tr>
<tr>
<td>NH(_4)-N (mg kg(^{-1}))</td>
<td>4623</td>
<td>7183(^a)</td>
<td>7137</td>
<td>4484</td>
</tr>
</tbody>
</table>

Total Elements
- C (%) | 35.7 | 35.9\(^a\) | 33.9\(^b\) | 34.6ab |
- N (%) | 4.3 | 4.3\(^a\) | 4.2 | 3.7b |
- P (%) | 2.2 | 2.3\(^a\) | 2.3 | 2.5a |
- K (%) | 3.5 | 3.2\(^b\) | 3.4ab | 3.6a |
- Ca (%) | 3.7 | 3.7\(^a\) | 3.7 | 3.9a |
- Mg (%) | 0.7 | 0.7\(^a\) | 0.7 | 0.7a |
- S (%) | 1.1 | 1.4\(^a\) | 1.6 | 1.2a |
- Na (mg kg\(^{-1}\)) | 9554 | \(\dagger\) | 16094\(^a\) | 13198 |
- Al (mg kg\(^{-1}\)) | 343 | 363\(^a\) | 291b | 309b |
- Fe (mg kg\(^{-1}\)) | 448 | 381b | 503b | 731a |
- Mn (mg kg\(^{-1}\)) | 577 | 661\(^a\) | 666a | 650a |
- Zn (mg kg\(^{-1}\)) | 515 | 525a | 580a | 551a |
- Cu (mg kg\(^{-1}\)) | 511 | 585a | 604a | 631a |
- B (mg kg\(^{-1}\)) | 52.7 | 49.0b | 49.3b | 53.4a |
- Ni (mg kg\(^{-1}\)) | 10.7 | 13.2a | 15.8a | 13.2a |
- Cd (mg kg\(^{-1}\)) | 0.18 | 0.09\(^a\) | 0.15a | 0.13a |
- Cr (mg kg\(^{-1}\)) | 7.8 | 5.5a | 7.0a | 8.3a |
- As (mg kg\(^{-1}\)) | 26.6 | 27.3b | 39.9a | 24.7b |
- Se (mg kg\(^{-1}\)) | 3.3 | 1.8b | 5.0a | 1.9b |

\(\dagger\) EC, electrical conductivity.
\(\dagger\) Study years are designated as starting the day BL was applied in late April or early May of the listed year and ending the day before BL was re-applied in the following calendar year. For example 2009 represents the time period from May 2009 to April 2010.

\(\$\) Means in the same row for study years 2009, 2010, and 2011 followed by different letters are significantly different (\(P < 0.05\)).

\(\dagger\) 2009 Na results were unavailable due to instrument malfunction.
Table 4-2. Analysis of variance summary for a split-split plot design exploring the effects of broiler litter (BL) application rate, study year (Year), sample date within study year, and their interactions on soil respiration, soil temperature at 2- and 10-cm depths ($T_{2\text{cm}}$ and $T_{10\text{cm}}$, respectively), and soil volumetric water content (VWC) in the top 6 cm in a silt-loam soil in the Ozark Highlands region of northwest Arkansas.

<table>
<thead>
<tr>
<th>Source of Variance</th>
<th>Respiration</th>
<th>$T_{2\text{cm}}$</th>
<th>$T_{10\text{cm}}$</th>
<th>VWC</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>BL</td>
<td>0.08</td>
<td>0.60</td>
<td>0.13</td>
<td>0.31</td>
<td></td>
</tr>
<tr>
<td>Year</td>
<td>&lt; 0.01</td>
<td>&lt; 0.01</td>
<td>&lt; 0.01</td>
<td>&lt; 0.01</td>
<td></td>
</tr>
<tr>
<td>BL x Year</td>
<td>0.57</td>
<td>0.03</td>
<td>0.59</td>
<td>0.31</td>
<td></td>
</tr>
<tr>
<td>Day(Year)</td>
<td>&lt; 0.01</td>
<td>&lt; 0.01</td>
<td>&lt; 0.01</td>
<td>&lt; 0.01</td>
<td></td>
</tr>
<tr>
<td>BL x Day(Year)</td>
<td>&lt; 0.01</td>
<td>0.79</td>
<td>0.72</td>
<td>0.87</td>
<td></td>
</tr>
</tbody>
</table>
Table 4-3. Summary of correlations for soil respiration with 2-cm soil temperature (T$_{2\text{cm}}$), 0- to 6-cm volumetric water content (VWC), and their interaction and quadratic terms.

<table>
<thead>
<tr>
<th>Statistical Parameter</th>
<th>T$_{2\text{cm}}$</th>
<th>VWC</th>
<th>T$_{2\text{cm}}^2$</th>
<th>VWC$^2$</th>
<th>T$_{2\text{cm}}$ x VWC</th>
</tr>
</thead>
<tbody>
<tr>
<td>$r^\dagger$</td>
<td>0.49</td>
<td>0.07</td>
<td>0.42</td>
<td>0.02</td>
<td>0.60</td>
</tr>
<tr>
<td>$P$-value</td>
<td>$&lt; 0.01^\S$</td>
<td>0.09</td>
<td>$&lt; 0.01^\S$</td>
<td>0.69</td>
<td>$&lt; 0.01^\S$</td>
</tr>
<tr>
<td>$n^\ddagger$</td>
<td>606</td>
<td>594</td>
<td>606</td>
<td>594</td>
<td>594</td>
</tr>
</tbody>
</table>

$^\dagger$ $r$, correlation coefficient.
$^\ddagger$ $n$, number of observations.
$^\S$ $P \leq 0.05$ are indicated in bold.
Table 4-4. Summary of 2-yr cumulative carbon input and losses of a managed pasture receiving broiler litter (BL) applications during 2009 and 2010. Litter was hand applied once annually to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas at three application rates (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively). Values represent kg C ha\(^{-1}\) that were applied as BL or lost through runoff, drainage below the 90-cm soil depth, harvested as hay, or respired from the soil over the 2-yr period.

<table>
<thead>
<tr>
<th>Description</th>
<th>Broiler Litter Rate</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Control</td>
</tr>
<tr>
<td>Input</td>
<td>kg ha(^{-1})</td>
</tr>
<tr>
<td>Broiler litter</td>
<td>0</td>
</tr>
<tr>
<td>Losses</td>
<td></td>
</tr>
<tr>
<td>Runoff(^{†})</td>
<td>15</td>
</tr>
<tr>
<td>Drainage(^{‡})</td>
<td>24</td>
</tr>
<tr>
<td>Dry matter removal</td>
<td>9,696</td>
</tr>
<tr>
<td>Soil respiration</td>
<td>40,140</td>
</tr>
</tbody>
</table>

\(^{†}\) Data taken from McMullen (2014b).
\(^{‡}\) Data taken from McMullen (2014a).
Table 4-5. Summary of multiple regression coefficients for 2-cm soil temperature ($T_{2\text{cm}}$) and the interaction between soil temperature and 0- to 6-cm volumetric water content ($T_{2\text{cm}} \times \text{VWC}$) terms for data combined across all broiler litter (BL) treatments and separately for each BL treatment condition. Litter was hand applied once annually to a silt-loam soil at the Agricultural Research and Extension Center in Fayetteville, Arkansas at three application rates (0, 5.6, and 11.2 Mg BL ha$^{-1}$; control, low, and high, respectively) for three years. All coefficient estimates were significantly different ($P < 0.0001$) than zero for all models. Upper (u) and lower (l) 95% confidence intervals (CI) are provided to allow comparisons between models and coefficient of determination ($R^2$) is provided as a measure of the strength of the relationships.

<table>
<thead>
<tr>
<th>Model</th>
<th>Intercept</th>
<th>$T_{2\text{cm}}$</th>
<th>$T_{2\text{cm}} \times \text{VWC}$</th>
<th>$R^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Estimate</td>
<td>CI&lt;sub&gt;l&lt;/sub&gt;</td>
<td>CI&lt;sub&gt;u&lt;/sub&gt;</td>
<td>Estimate</td>
</tr>
<tr>
<td>All-treatments&lt;sup&gt;†&lt;/sup&gt;</td>
<td>-3.45</td>
<td>-4.39</td>
<td>-2.52</td>
<td>0.28</td>
</tr>
<tr>
<td>Control&lt;sup&gt;‡&lt;/sup&gt;</td>
<td>-2.54</td>
<td>-3.52</td>
<td>-1.55</td>
<td>0.27</td>
</tr>
<tr>
<td>Low&lt;sup&gt;‡&lt;/sup&gt;</td>
<td>-4.16</td>
<td>-5.55</td>
<td>-2.77</td>
<td>0.26</td>
</tr>
<tr>
<td>High&lt;sup&gt;‡&lt;/sup&gt;</td>
<td>-4.32</td>
<td>-6.28</td>
<td>-2.35</td>
<td>0.30</td>
</tr>
</tbody>
</table>

<sup>†</sup> $n = 594$.
<sup>‡</sup> $n = 198$.
<sup>§</sup> Estimates in bold lay outside the 95% confidence intervals of the all-treatments model.
Figure 4-1. Broiler litter (BL) application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) effects on soil respiration over a 3-yr period. Study years were designated as starting the day BL was applied in early May of the listed year and ending the day before BL was re-applied in the following calendar year. Summer (Su), fall (F), winter (W), and spring (Sp) seasons are separated by vertical lines. Asterisks denote sample dates when BL treatment effects occurred. Protected least significant difference (LSD) to compare soil respiration rates between any treatment combination is 3.3 µmol CO\(_2\) m\(^{-2}\) s\(^{-1}\).
Figure 4-2. Study year effects on soil 2- and 10-cm temperatures ($T_{2cm}$ and $T_{10cm}$, respectively) and 0- to 6-cm volumetric water content (VWC) during a 3-yr period. Study years were designated as starting the day BL was applied in early May of the listed year and ending the day before BL was re-applied in the following calendar year. Summer (Su), fall (F), winter (W), and spring (Sp) seasons are separated by vertical lines. Protected least significant differences (LSD) are provided for comparisons.
Figure 4-3. Broiler litter application rate (0, 5.6, and 11.2 Mg BL ha\(^{-1}\); control, low, and high, respectively) and study year effect on soil 2-cm temperature (T\(_{2\text{cm}}\)). Study years were designated as starting the day BL was applied in early May of the listed year and ending the day before BL was re-applied in the following calendar year. Different letters atop bars indicate a significant difference (P < 0.01) among all treatment combinations.
Figure 4-4. Cumulative CO₂-C emissions during a 3-yr period. Study years were designated as starting the day broiler litter (BL) was applied at one of three rates (0, 5.6, and 11.2 Mg BL ha⁻¹; control, low, and high, respectively) in early May of the listed year and ending the day before BL was re-applied in the following calendar year. Summer (Su), fall (F), winter (W), and spring (Sp) seasons are separated by vertical lines.
Appendix 4-1. Example SAS program for analysis of variance based on a split-split plot design with sample date within study year and related data.

```sas
options mprint;

title 'Soil CO2 Flux -- Richard McMullen --';
title2 '3 Year Data Set';

data Gases;
  infile 'G:\McMullen\CO2 Flux\SAS\3yrFlux.csv' firstobs=2 DLM=',' truncover LRECL = 600 DSD;
  input DayspostBL DPostBLNeg Studyyear Year Date $ DOY Block BL $ BLcontin Plot Flux C2avg T10 VWC T2 BD WFPS;
  label BL = 'Broiler Litter';
run;

proc sort data=Gases; by year date block;
run;
quit;

title3 'ANOVA all years with BL*sample date within year';
title4 "---------- soil VWC cm3 cm^-3 ----------";
ods rtf file='G:\McMullen\CO2 Flux\SAS\rtffiles\3yrTempVWC.rtf' bodytitle style=journal;
proc mixed data=Gases method=type3 ;
class block studyyear DayspostBL bl ;
model VWC = BL studyyear BL*studyyear dayspostBL(studyyear) BL*dayspostBL(studyyear) / ddfm=kr ;
random Block block*BL Block*BL*studyyear ;
  lsmeans studyyear ;
  lsmeans dayspostBL(studyyear);
run;
ods rtf close;
quit;
```
<table>
<thead>
<tr>
<th>Days</th>
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Appendix 4-2. Example SAS program for Pearson’s correlation analysis.

```
options mprint;

title 'Soil CO2 Flux -- Richard McMullen --';
title2 '3 Year Data Set';

data Gases;
  infile 'G:\McMullen\CO2 Flux\SAS\3yrFlux.csv' firstobs=2 DLM=',' truncover LRECL = 600 DSD;
  input DayspostBL DPostBLNeg Studyyear Year Date $ DOY Block BL $ BLcontin Plot Flux C2avg T10 VWC T2 BD WFPS;
  label BL = 'Broiler Litter';
run;

proc sort data=Gases; by year date block;
run;
quit;

title3 'INITIAL DATA LISTING';
proc print data=Gases noobs; by year date block;
  id year block;
  var DayspostBL DPostBLNeg Studyyear Year DOY Block BLcontin Plot Flux C2avg T10 VWC T2 BD WFPS;
run cancel;
quit;

title3 'Correlations';
title4 "---------- All ----------";

ods graphics on;
proc corr data=Gases plots=matrix(histogram);
  var DayspostBL DPostBLNeg Studyyear Year DOY Block BLcontin Plot Flux C2avg T10 VWC T2 BD WFPS;
run;
ods graphics off;
run;
quit;
```
Appendix 4-3. Example SAS program for analysis of variance and related data.

options mprint;

title '3 yr Cumulative CO2-C Losses -- Richard McMullen --';
title2 'Losses kgCO2-C per ha';

data Losses;
  infile 'G:\McMullen\CO2 Flux\Anual Mass Loss\SAS\3yrCumMassLoss.csv' firstobs=2
    DLM=',' truncover LRECL = 600 DSD;
  input Block BL $ Blcont Plot Loss;
  label BL = 'Broiler Litter';
  /* Units of Loss are kgCO2-C per ha */
run;

proc sort data=Losses; by BL block ;
run;
quit;

title3 'INITIAL DATA LISTING';
proc print data=Losses noobs; by BL block ;
  id BL;
  var Block Loss ;
run ;
quit;

title3 'INITIAL DATA LISTING';
proc print data=Losses noobs; by BL block ;
  id BL;
  var Block Loss ;
run ;
quit;

title3 'ANOVA 3yr cum mass lost';
title4 "---------- ---------- ";
* ods rtf file='G:\McMullen\CO2 Flux\SAS\rtffiles\3yrFlux.rtf' bodytitle style=journal;
proc mixed data=Losses method=type3 ;
  class block BL ;
  model Loss = BL / ddfm=kr ;
  random Block ;
  lsmeans BL;
run ;
* ods rtf close;
quit;
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Summary and Conclusions

Based on eight years of continuous monitoring of nutrient and metal losses in runoff and drainage waters from BL-amended pastureland with a history of organic amendments and under naturally occurring precipitation, annual flow-weighted-mean (FWM) runoff Fe and annual FWM leachate Na and Fe concentrations were the only water quality parameters affected by litter application rate. However, results indicated that pasturelands with a history of litter application may continue to release BL-derived metals, such as As and Se, at concentrations harmful to health regardless of the current management practice long after litter application had ceased.

Correlation coefficients among annual precipitation, runoff, and runoff loads were all positive. Additionally, all annual runoff nutrient and metal loads increased over the 8-yr period and were highly influenced by one large rainfall event that occurred during the last year of the study. Mean annual runoff and annual FWM concentrations of Ca, Cd, Cu, Na, and Se increased during the 8-yr period. Mean annual runoff pH, EC, and FWM concentrations of NO₃-N, NH₄-N, PO₄-P, DOC, Cr, K, Mg, Mn, Ni, P, and Zn were unchanged during the 8-yr period. Additionally, annual runoff FWM As concentration was the only nutrient or metal monitored that increased during the 8-yr study.

Correlation coefficients between annual drainage and annual FWM leachate concentrations were mostly negative and, in conjunction with the positive correlations between annual drainage and annual leachate loads, suggested a leachate dilution effect similar to those observed in previous runoff studies. Mean annual drainage and leachate pH and EC were unchanged during the 8-yr period. Mean annual FWM concentrations and loads of NH₄-N, As, Mn, and Ni decreased and mean annual FWM concentrations and loads of Cu and Se increased.
during the eight years. In contrast, mean annual FWM concentrations and loads of NO$_3$-N, PO$_4$-P, Cd, Cr, K, P, Zn, and DOC remained constant over the same time period.

This study also demonstrated that BL application rate affected soil respiration, which varied by sample date and among years and was significantly correlated with environmental factors. Litter application increased respiration relative to the unamended control immediately after application in all three study years. Of the 100 sample dates during the study, BL treatment effects were observed on 27 dates, of which 18 dates occurred within 84 days-post BL application. Litter application also increased respiration relative to the unamended control when rainfall events occurred after dry periods, and suggested that the slow phase of BL decomposition may be stimulated by rainfall. The relationship relating BL application rate to annual CO$_2$-C emissions predicted that an unamended pasture soil with a history of organic amendments will release 19.2 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$ and that for every 1 Mg of BL ha$^{-1}$ applied annually will increase this baseline emission estimate by 0.5 Mg CO$_2$-C ha$^{-1}$.

Regression also indicated that soil respiration could be predicted using near-surface soil temperature (T$_{2cm}$) and the product of T$_{2cm}$ and soil volumetric water content (VWC) as predictors with moderate strength ($R^2 = 0.52$) in the relationship. When the model was applied to BL treatment data sets separately, coefficient estimates for the y-intercept and T$_{2cm}$ were within the 95 % confidence interval (CI) of the all-treatments model. However, the coefficient estimate for the product of T$_{2cm}$ and VWC fell outside the 95 % CI of that for the all-treatments model suggesting that long-term applications of BL may influence the interaction between near-surface soil temperature and VWC as it relates to respiration.

Runoff and drainage water quality trends and harmful concentration limits for some BL-derived nutrients and metals required time periods greater than three years to identify or exceed,
respectively. These results emphasized the importance of long-term observational studies especially with regards to trace metals. Additionally, annual amendments of BL stimulated CO₂ release from the soil and could potentially increase atmospheric CO₂ concentrations which would promote radiative forcing in the atmosphere.

Future research should focus on seasonal patterns as well as individual runoff and leaching events. Important insights regarding BL application rate and time effects could be identified if time (i.e., expressed as the amount of time between BL application and time of precipitation), precipitation amounts, and runoff or drainage amounts were treated as covariates, while replication was allowed to occur across study years. The data set generated during this study would allow these analyses. Additionally, the BL-induced CO₂ release from the soil to the atmosphere observed in the current study could potentially be offset by increased CO₂ uptake associated with BL-induced increases in dry matter production. A mass balance for C would address this hypothesis. Similarly, a mass balance of all nutrients and metals should be performed after forage dry matter analyses are complete.