NITROGEN AVAILABILITY, CROP GROWTH, AND WATER QUALITY IN A LIVING MULCH CORN PRODUCTION SYSTEM

by

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(Under the Direction of David Radcliffe)

ABSTRACT

The impact of living mulch corn production on nitrogen availability, crop growth, and water quality was investigated. Plant, soil, and water samples were taken over the course of two years in research plots and experimental watersheds. A living mulch system reduced corn growth and grain yield when compared to a cereal rye and crimson clover systems due to reduced N availability. A HYDRUS-1D model was used to estimate water and NO_3-N leaching below a 1-m depth from planting in April through February of the next year. In the first model period, the living mulch had lower NO_3-N loss than both the cereal rye and crimson clover systems, though little NO_3-N was lost during the second model period due to drought. On the experimental watersheds, the living mulch likely reduced runoff volume, and both the living mulch and cereal rye systems reduced sediment loss when compared to previous treatments.

INDEX WORDS: Living mulch, Nitrogen, Corn, Nitrate leaching, HYDRUS-1D, water quality, Cover crops.

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

1. Introduction

Due to its popularity as a livestock feed, its use in food products, and ability to be used as biofuel, corn is the most widely grown grain crop in the United States, with over 95 percent of total grain production devoted to corn. Corn growth covers over 90 million acres in the US, mainly in the Midwest region, with Iowa and Illinois alone accounting for nearly one-third of the annual US corn crop. Demand for corn and other grain crops have grown drastically in recent years largely due to energy policy mandates passed in 2005 and 2007 which call for increasing corn ethanol production. As demand has grown, production has begun in non-traditional areas in order to keep up with demand (Capehart, 2016). As the market continues to expand, producers must learn to balance inputs with yields in order to achieve efficiency and profitability.

A problem facing many producers is providing a balance between obtaining maximum yields and environmental stewardship. Many producers use large quantities of soluble inorganic fertilizers that can become a source of non-point source of pollution that causes eutrophication of aquatic ecosystems (Pionke et al., 2000) and pollution of groundwater systems (Di et al., 2002). It has been estimated that 50% of impaired lake area and 60% of impaired river reaches are the results of non-point sources from agricultural areas. Leaching of NO_3-N fertilizer through the soil can contaminate groundwater and have serious health effects if it reaches a drinking water supply (Carpenter et al., 1998). Leaching of NO_3-N through the soil has been reported as high as 100 kg NO_3-N ha^{-1} $year^{-1}$ in corn production (Hahne et al., 1977). To protect against sediment and nutrient loss in runoff and leaching of NO_3-N to groundwater, cover crops are used to reduce the effect of corn production on water quality. During winter fallow periods, cover crops uptake excess NO_3-N in the soil profile that is vulnerable to leaching, reducing the NO_3-N load to groundwater systems (Brandi-Dohrn et al., 1997; McCracken et al., 1994). In agricultural watersheds, cover crops prevent erosion by reducing detachment of soil particles during runoff events (Kaspar et al., 2001) and reduce runoff and nutrient load by improving the infiltration capacity of the soil (Steele et al., 2012, Keisling et al., 1994).

Nitrogen is generally the most limiting nutrient in corn production and requires 200 to 280 kg N ha^{-1} annually to achieve maximum economic yield (Lee et al., 2015; Raun et al., 1999). Legumes as cover crops have been used to fix atmospheric N for later use by a corn crop. For example, crimson clover has been shown to add between 80 and 170 kg N ha^{-1} to the soil when used as a winter annual (Young-Mathews, 2013). Similar to other cover crop systems, living mulches aim to reduce erosion and provide supplemental N for corn growth. However, living mulches are not killed prior to crop planting like annual cover crops, but are suppressed either with mechanical plows or herbicides and live alongside a cash crop (Echtenkamp et al., 1989; Zemenchik et al., 2000). A potential disadvantage of this system, however, is that both the living mulch and main crop may compete for resources, particularly soil N. Competition for water and soil N has been shown to reduce corn grain yield, but reductions may be overcome if moisture and N requirements are met during critical periods of corn growth (Affeldt et al., 2004, Kurtz et al., 1952; Zemenchik et al., 2000).

The overall objective of this research was to determine the impact of a living mulch on corn production and its effects on water quality. Therefore, specific goals were to:

1. Compare the soil N dynamics of a living mulch, cereal rye, and crimson clover cover crops on corn growth and yield.
2. Estimate and compare water and NO_3-N transport in the three systems using a HYDRUS-1D model
3. Compare water and sediment loss from corn production on a living mulch and a cereal rye watershed, as well as quantify nutrient and *E. coli* loss

2. Literature Review

2.1 Tillage

Corn, like many other crops, can be grown using different tillage practices. Three commonly used practices are traditional tillage, strip tillage, and no-till. Traditional tillage, sometimes referred to as mulch tillage, uses plows and discs to disturb the entire soil surface prior to planting. This method helps to aerate the soil and incorporates anything left on the soil surface, such as plant residue or manure, and functions as a method of weed control. In comparison, strip tillage disturbs up to 30 percent of the soil surface in areas, or "strips" that will later be planted. Strip tillage provides better erosion control than traditional tillage while still incorporating plant remains in areas that will be planted. Lastly, no-till involves planting directly into plant remains without disking or plowing any part of the soil surface, with the only disturbance being the seed planters. While this method provides the best erosion control of the three tillage methods, it also provides little weed control and does not incorporate plant remains into the soil system (USDA-NRCS, 2008).

In the United States, 35.5 percent of all agricultural land was part of a no-till operation in 2009, which was roughly 88 million acres. Despite having more acres planted in the US than any other crop, no-till corn made up only 23.5 percent of the entire corn crop. Therefore, the majority of agriculture in the United States still relies heavily on traditional tillage systems. In recent years, the amount of land devoted to no-till agriculture has steadily increased (Horowitz et al., 2010) due to the differences between the two systems, such as yields and effects on soil physical and chemical properties.

While many producers believe otherwise, there is no significant difference in yield between traditional tillage and no-till systems. A 45 year study by Cook et al. (2016) found that corn yields from traditional tillage and no-till systems were similar from both corn only and rotated corn and soybean production systems. Cook also suggested that the main factor influencing corn yield was not tillage, but proper fertilizer management. In fact, a no-till system can even have a yield advantage in certain instances. Dick et al. (1991) showed that corn yields from no-till systems were similar to those from traditional tillage systems, but no-till had higher yields in poorly drained areas due to the improvement of soil physical properties. In a similar study, Blevins et al. (1983) also found that the 10-year average corn yield from no-till systems was lower than traditional tillage systems at reduced N fertilization rates, but higher at moderate and high N fertilization rates.

Differences in tillage also have an effect on overall soil physical and chemical properties. Conventional tillage incorporates all additions to the system, such as plant residues, lime, and fertilizer and results in a surface soil layer that is significantly different from a reduced or no-till system (Blevins et al., 1983). Two of the main soil physical properties affected by tillage are bulk density and porosity. Soils under traditional tillage management typically have lower bulk density and higher porosity than soils under no-till management. This leads to variable infiltration rates in both tillage management systems. Infiltration in traditional tillage systems is largely unaffected by preferential flow, whereas no-till systems are mainly affected by water movement through macropores created by active soil fauna or roots of previous crops (Lipiec et al., 2006).

Distribution of soil nutrients is also largely affected by tillage practices. Concentrations of organic carbon (C) and nitrogen (N) have been shown to be higher in the 0-5 cm depth in no-till soils than in traditionally tilled soils. Organic C and N are higher in no-till soils mainly because plant residues are able to accumulate at the soil surface, unlike traditional tillage. Also, tillage increases the rate of organic C and N mineralization by improving aerobic conditions for soil microbes. Below the plow layer, however, organic C and N are typically higher in traditionally tilled soils as tillage allows plant residue to accumulate below the soil surface. The distributions of phosphorus (P) and potassium (K) are similar to organic C and N; both concentrations are higher in the 0-5 cm depth in no-tilled soils but are higher below 5 cm in traditionally tilled soils. This is due to fertilizer and residue incorporations associated with tillage in which a layer of accumulation is created below the plow layer (Ismail et al., 1994). Lastly, soil pH also varies by tillage practice. At the surface, the pH is typically lower in no-tilled soils while the pH is lower below 5 centimeters in the traditionally tilled soils. Both Blevins et al. (1983) and Ismail et al. (1994) found a strong correlation between N fertilization rate and soil pH. Nitrification occurring from N sources on the soil surface caused acidification in no-till systems near the soil surface while nitrification from incorporated N sources caused acidification lower in the soil profile in traditional tillage systems.

2.2 Cover crops

Along with different tillage practices, corn is typically grown under many different cover cropping systems. Cover crops provide many benefits and are used in accordance with the producer's needs. Grass cover crops, such as cereal rye, are used mainly to prevent erosion and hold in soil moisture. Leguminous cover crops, such as clover or vetch, can fix atmospheric N as well as control weeds and other pests. Cover crops are usually used on lands that have been previously harvested and need cover while the land is not currently under production and then killed before planting of the next crop (Snapp et al., 2005).

Cereal rye (*Secale cereal)* is a widely used annual grass cover crop and is used mainly by farmers to reduce soil erosion. Below the soil surface, root growth stabilizes and holds the soil in place while aboveground plant growth intercepts the impact of raindrops, reducing the potential for soil loss. Many farmers choose cereal rye as a cover crop due to its rapid growth, quickly providing up to 30% ground cover after one month of growth (Snapp et al., 2005). In addition to preventing erosion, cereal rye also helps to control weeds and scavenge N left in the soil from previous cash crop fertilizations. Weeds are controlled in cereal rye cover crops by creating a physical barrier to weeds at the soil surface through production of large amounts of biomass. In addition to physical barriers, cereal rye also suppresses weeds through the production of allelopathic chemicals. After harvest of a cash crop, a cereal rye cover crop typically assimilates between 25 and 55 kg N ha^{-1} from the soil. However, it has been shown that a cereal rye cover crop can retain as much as 112 kg N ha^{-1}. By taking up excess soil N, cereal rye helps to prevent N loss through leaching and erosion and can resupply that N to another cash crop after spring planting (Casey, 2012).

Annual legumes, such as crimson clover (*Trifolium incarnatum L.)* are also used as cover crops due to their ability to fix atmospheric N, as well as provide soil stability and cover to prevent erosion. Legumes are able to add N to soils through biological N fixation, in which *Rhizobium* bacteria that are symbiotically associated with the roots of legumes take in atmospheric N and make it available for plant use in the form of ammonium. A crimson clover cover crop can add 70 to 170 kg N ha^{-1} to the soil when used as a winter annual. In economic terms, corn production in a crimson clover cover crop system has been shown to be more profitable than typical N-fertilization programs, with savings near 10 percent. Much like cereal rye, crimson clover cover crops provide weed control by shading out competing weeds and erosion control by stabilizing the soil and intercepting the impact of falling raindrops, and are commonly used in the southeastern US for roadside erosion control and beautification (Young-Mathews, 2013).

2.3 Living mulches

Similar to other cover crop systems, living mulches aim to reduce erosion, provide N, control weeds, and maintain soil moisture for crop growth. However, living mulches are not killed prior to crop planting like annual cover crops, but are suppressed either with mechanical plows or herbicides and live alongside a cash crop. Two main advantages of a living mulch system are reduced soil erosion and increased weed control. If a legume is used as the cover in the living mulch system, another advantage is N additions from biological N fixation. For example, corn grown with white or ladino clover had 75% higher yields than corn grown without a living mulch due to N fixation (Echtenkamp et al., 1989). A potential disadvantage of this system is that both the living mulch and cash crop may compete for resources, particularly water if it is not in abundance.

Legumes are well suited for a living mulch system due to their ability to provide N to a cash crop as well as provide good soil cover and erosion control. Unlike grasses, leguminous living mulches do not compete with crops for N. Echtenkamp et al. (1989) found that crop yields were higher in all trials that used a legume as a living mulch rather than a grass. To achieve the same yields as the living mulch trials, the grass living mulches received supplemental N fertilizer. Positive effects of legumes as living mulches have been reported in many crops, including alfalfa, crownvetch, hairy vetch, subterranean clover, and white clover (Hartwig et al., 2002). Overall, legumes as a living mulch can reduce the need for supplemental N fertilizer. Also, it has been shown that legumes, when grown with cereal crops such as corn or wheat, do not compete with the main crop for soil P or potassium (Duiker et al., 2004). This is mainly due to the fact that P and potassium are immobile in the soil and legumes and cereals have differing root systems. While there is weak competition for nutrients, leguminous cover crops compete strongly for water and can reduce crop yields. However, if water limitations are removed then crop yields can be equaled or sometimes even increased (Duiker et al., 2004).

Living mulch systems have also been shown to help increase the availability of otherwise immobile nutrients, such as P, by facilitating arbuscular mycorrhizae formation on the main crop. An established living mulch during a fallow period can host arbuscular mycorrhizae and aid in formation on the main crop, improving P uptake. Deguchi et al. (2007) found that the P concentration of corn shoots and mycorrhizal colonization of corn roots were higher in living mulch grown corn than in traditionally tilled and fertilizer corn. Also, the yield of living mulch grown corn was similar to that of traditionally grown corn that was fertilized with 2000 kg ha^{-1} of P_2O_5. This suggests that a white clover living mulch facilitates arbuscular mycorrhizae formation and improving P uptake, thereby improves corn yields (Deguchi et al., 2007).

Living mulches have been shown to control pests, such as weeds, insects, and diseases. Much like annual cover crops, living mulches help to control weeds by creating a physical barrier on the soil surface and competing with weeds for light and other resources. A subterranean clover living mulch has been shown to suppress weeds through the production of allelopathic chemical, much like cereal rye. A white clover living mulch controls weed growth through shading and competition, and has been shown to provide weed control comparable to herbicide applications in no-till sweet corn systems (Hartwig et al., 2002). Living mulches have also been shown to control insect pests by improving the biodiversity of an agricultural area and providing a habitat for natural predators. Decreased pest density has been shown in many different vegetable crops grown in living mulches, particularly of whiteflies and aphids (Koota et al, 2013). Brandsaeter et al. (1998) found that white clover and subterranean clover living mulches used in cabbage production significantly reduced insect damage compared to traditionally grown cabbage, giving a greater number of marketable heads and greater economic returns.

An important aspect of a living mulch system is the mechanism of mulch suppression to eliminate mulch and plant competition. Typically, annual cover crops are terminated either with mechanical mowing or plowing or with herbicides and either left on the soil surface or incorporated using discing or plowing. Living mulch systems, however, rely on banded herbicide applications or strip-tillage. Hill (unpublished data; NSF) showed that plant population density was similar in living mulches that were suppressed with herbicides to conventionally tilled unmulched areas. Areas that were neither tilled nor treated with herbicides and had crops planted directly into the living mulch had significantly lower plant population density than conventionally tilled and banded herbicide treatments due to competition between the main crop and living mulch. Hill and Affeldt et al. (2004) both showed that the mulch can be continuously suppressed through continued banded herbicide applications during early crop growth in both herbicide resistant and non-herbicide resistant corn.

2.4 Nutrients and fertilizers

Like many other high-yielding grain crops, corn requires high levels of fertilization. Many corn fertilization programs in the Piedmont region of the southeastern United States call for 200 to 250 kg/ha of supplemental N. Insufficient available N is the primary limitation to corn growth and yield, and corn is typically fertilized with inorganic forms of N, such as ammonium and nitrate. Currently, cereal production accounts for 60% of N fertilization worldwide. Nitrogen use efficiency, the percent of N taken up by the plant versus the amount applied, is currently 42% in the developed world and 29% in the developing world. Nitrogen can be lost from agronomic systems in many different ways, including denitrification, leaching through the soil profile, runoff during irrigation and rainfall events, and even along with sediment (Raun et al., 1999).

Phosphorus is another major nutrient needed in large quantities by the corn plant, but is considered one of the least available and least mobile nutrients. In acidic soils typical of the Piedmont region of the southeastern US, P is highly adsorbed to iron and aluminum oxides, and is most available for plant uptake around pH 6.5. However, the vast majority of P remains bound to the soil, causing many producers to provide supplemental P fertilizer in the form of phosphate. Due to high soil adsorption, P use efficiency for supplemental P is quite low, typically around 10%. If supplemental P is not applied, however, yields have been shown to decrease between 5 and 15% in soils with sub-optimal P levels (Shenoy et al., 2005). Like N, P can be lost from agricultural systems, but is much less common due to the high adsorption rate with most soils.

2.5 Agriculture and Water Quality

Runoff and nutrient loss from agricultural sources is a main contributor to non-point source pollution in the United States. During storm events, runoff from agricultural watersheds can be discharged into streams, lakes, rivers, and other bodies of water. Discharge from these areas can include nutrients, sediment, bacteria, and pesticide, which can have significant effects on downstream ecosystems and bodies of water. High concentrations of nutrients, such as N and P, in many different forms, are the main contributors to eutrophication in many aquatic ecosystems. For example, agricultural areas in the contributing area of the Chesapeake Bay were implicated as a major contributing factor to the expansion of a large anoxic zone within the bay. Inputs of N and P were the main concern for environmental officials, and an agreement was reached with the US Environmental Protection Agency in 1987 to reduce both inputs by 40% before the year 2000 (Pionke et al., 2000). In fact, roughly 50% of impaired lake area and 60% of impaired river reaches are the result of non-point sources from agricultural areas (Carpenter et al., 1998).

Many different agricultural practices can lead to eutrophication of downstream areas. One main input is large amounts of inorganic fertilizer, usually in the form of ammonium (NH_4^+), nitrate (NO_3^-), and phosphate (PO_4^{3-}). It has been estimated that around 600×10^6 Mg of P were applied to agricultural areas between 1950 and 1960, with only 42% of that being removed as crop or forage, leaving most applied P susceptible to export by runoff and erosion. Due to the high adsorption rate of P to soil, eroded soil that is exported to water bodies can also contribute to high P levels and slowly desorb, making restoration efforts in affected areas difficult. While high N and P inputs can lead to eutrophied waters, whose symptoms include algal blooms and anoxic zones, there are direct health effects of contaminated waters. For example, high levels of nitrate, an inorganic source of N for plants, can cause methemoglobinemia in humans and other mammals. Methemoglobinemia can reduce the oxygen carrying capacity in mammals and is sometimes fatal to infants, which has caused the US Environmental Protection Agency to establish a maximum contaminant level for nitrate-N at 10 mg/L (Carpenter et al., 1998). While nutrient loss from agricultural areas is the main contributing factor to non-point source pollution, runoff volume, nutrient loads, sediment loss, and pesticide loss vary significantly in agricultural watersheds based on many different factors and management practices.

Different cropland management methods can have a dramatic effect on the quantity and quality of runoff from agricultural watersheds. Tillage can have a significant impact on the quantity of runoff due to differences in bulk density and infiltration rates. In order to reduce runoff, conservation tillage is a Best Management Practice that is commonly suggested by the Conservation Technology Information Center. Long term conservation tillage practices, including no-till, reduce runoff by allowing the development of macropores that are normally disrupted by tillage. A greater number of macropores allow for higher infiltration rates in no-till soils and allow for water to bypass upper soil layers when rainfall exceeds the infiltration capacity of the soil (Fawcett et al., 1994). In some instances, no-till practices result in dramatic decreases in water runoff. A four year study by Glenn et al. (1987) showed that no-till practices resulted in 27% less runoff than conservation tillage, also reducing the amount of pesticide that was lost in runoff. In other cases, no-till has been shown to eliminate runoff in certain situations

(Triplett et al., 1978). While many long-term studies show that runoff volume can be affected by tillage practice, Blevins et al. (1990) noted that many short-term studies show mixed results, and that runoff volumes have been shown to be lower in both conventional tillage and conservation tillage or even statistically not significant when compared.

Soil cover can also have a significant effect on the quantity of runoff produced in agricultural watersheds. Living and decaying plant tissue provide protection from the impact of falling raindrops, preventing crusting on the soil surface and allowing water to infiltrate more quickly than bare soils. Bare soils typically have greater runoff volumes than covered soils due to a crust that forms at the soil surface that slows infiltration. Steele et al. (2012) showed that a cereal rye winter annual cover crop had a positive effect on many soil physical properties. This included a reduction of soil bulk density, as well as an increase in water infiltration rate and saturated hydraulic conductivity. Keisling et al. (1998) showed similar results when studying rye, vetch, and lupine as cover crops in cotton: in all cases, bulk density was decreased while hydraulic conductivity and infiltration rates increased. This suggests that most cover crops reduce surface runoff and increase infiltration and percolation of rainfall during high volume events.

Sediment loss from agricultural areas has an adverse effect on water quality and is considered agriculture's main pollutant. Erosion can occur in most agricultural settings, even in relatively flat areas. Sediment transported into lakes, rivers, and other bodies of water can become suspended in water and cause temperatures to rise, often limiting the growth of fish and affecting aquatic ecosystem dynamics (Dabney et al., 2001). Chichester et al. (1992) showed that tillage practice significantly affected sediment loss in a paired watershed study. Mean annual sediment loss was 10 times higher in conventionally tilled watersheds compared to no-tilled watersheds, even when there were no significant differences in runoff amounts. Blevins et al. (1990) noted that most comparison studies show that no-till reduces sediment loss significantly when compared conventional tillage. The reduction ranged from 5 to 36 times less than the sediment loss in no-till.

One of the primary uses for growing cover crops is reducing erosion by providing soil cover outside of the normal cropping season. Cover crops help maintain and improve soil structure, which reduces sediment transport into lakes, rivers, and other water bodies by reducing detachment of soil particles during runoff events (Kaspar et al., 2001). Most reviewed articles show that cover crops reduce sediment loss when compared to bare soils. For example, in a corn and soybean rotation, downy brome and Canada bluegrass were shown to reduce sediment loss by 95 and 96%, respectively (Zhu et al., 1989), though reductions of this magnitude are atypical. Perennial living mulches also reduce sediment loss by providing year-round soil cover. Hall et al. (1984) reported a 50% reduction in water runoff and 97% reduction in eroded soil when corn was planted into a crown vetch living mulch. This also included a 99% reduction in atrazine, a preemergence herbicide used in corn, lost in sediment.

In agriculture, nutrient loss is a large concern as applied fertilizers are soluble and have great potential to be lost during runoff events. Eutrophied waters are unfit for recreation and expensive to clean. A main effect of eutrophication in both marine and freshwaters is explosive growth of algae. Algae and other nuisance weeds interfere with water that is used for fisheries, recreation, and drinking. In marine ecosystems, red tides from algal blooms can release toxins into ecosystems and disrupt marine life. These toxins can cause mortality in marine mammals as well as poison humans who eat shellfish affected by the toxins. In freshwater, excess nutrients can cause blooms of cyanobacteria which can also release toxins lethal to humans and livestock when ingested. Decomposition of algae in both marine and freshwater systems can also lead to oxygen shortages, sometimes enough to limit oxygen availability to aquatic life and cause fish kills, leading to an overall loss of aquatic biodiversity (Carpenter et al., 1998).

Nutrient additions from non-point sources also have an effect on forested streams and waterways that are not typically in use by humans. Multiple studies (Benstead et al., 2009; Suberkropp et al., 2010) show that even relatively low concentrations of additional nutrients, particularly N and P, can affect stream ecosystems. In most forested headwater streams, leaf litter is the primary carbon (C) source for aquatic bacteria and fungi. Suberkropp et al. (2010) showed that leaf litter in streams enriched with nutrients decomposed much faster and had greater CO2 loss from microbial respiration than unenriched streams. This led to an overall decline in fungal biomass produced annually in enriched streams, resulting in lower food availability for macroinvertebrates and eventually altering the food web of the stream. In a similar study, Benstead et al. (2009) also saw increased rates of leaf litter decomposition. Additionally, nutrient enrichment of forested streams led to accelerated rates of organic matter transformation and export, potentially altering food-web dynamics and even long-term ecosystem stability.

Surface application of fertilizer is common in corn growth, as opposed to being incorporated into the soil, making it even more susceptible to loss and transport downstream. It is estimated that 0.01 ppm phosphate and 0.3 ppm nitrate are sufficient to support algal growth in lake water, concentrations that are often encountered in runoff from unfertilized non-agricultural watersheds, making fertilization rate and timing important factors in protecting water quality (Romkens et al., 1973). Some studies suggest that losses of dissolved N and dissolved P can be greater with conservation tillage than with conventional tillage due to the effect of increased N and P from crop residue and decreased soil loss (Johnson, 1979; McDowell and McGregor, 1980). Other studies suggest that Total N and Total P concentrations are higher in runoff from conventionally tilled corn watersheds due to ammonium and P bound to sediment which can desorb and affect water quality (Angle et al., 1984; Romkens et al., 1973).

In corn production, herbicide losses to surface and groundwater are a concern as they are another source of water contamination. Hall et al. (1972) found that the average loss of atrazine in corn plots per year in the form of surface runoff was 2.4% of applied herbicide. While no tillage practice has been shown to eliminate runoff or herbicide loss, bulk herbicide loss has been shown to be higher in conventionally tilled systems due to greater runoff volumes and greater sediment loss. However, herbicide concentrations can sometimes be higher in runoff from no-till systems due to reduced runoff and washoff of herbicide that was intercepted by crop residue (Felsot et al., 1990). Historical data suggests that conservation tillage systems can reduce herbicide runoff by 60% when compared to traditional tillage. To mitigate herbicide losses in cropping systems with and without covers, buffer strips have been used to increase infiltration, intercept suspended solids, and remove nutrients and pesticides from runoff. One study shows that grassed buffer strips can reduce runoff of 2,4-D by 70% and atrazine by up to 60% (Patty et al., 1997).

While runoff from agricultural areas high in N and P are typically to blame for contamination and eutrophication of surface waters, N in the form of nitrate is capable of contaminating groundwater by leaching through the soil profile. However, leaching occurs in many different land use systems, not only agricultural systems. Soil, climatic and management factors all play a role in nitrate leaching. In soils, the concentration of ammonium is typically low, so fertilizer applied in that form is quickly converted to nitrate. Since soils in temperate regions are negatively charged, nitrate is not retained by the soils and can move with soil water into groundwater if it is unused by plants or microbes. Groundwater contamination is highest in countries with developed agricultural systems as high amounts of N are applied as either synthetic fertilizer or organic waste in the form of manure. Significant nitrate leaching has also been recorded in many corn productions systems around North America. Annually, 11 to 107 kg N per ha leaching loss has been reported, of which most is lost when there is no vegetative growth for nitrate uptake. Also, most nitrate loss as leachate from corn production is from mineralization of soil N or remineralized N that had been previously immobilized. Tillage practices also contribute to nitrate leaching as aeration of the soil promotes mineralization and nitrification of soil N. Therefore, conservation tillage and cover crops to uptake excess soil N are necessary management strategies in soils that have high potential for nitrate leaching to groundwater (Di et al., 2002).

Watersheds with animal operations or that receive manure as a soil amendment are capable of contributing manure-borne pathogens to surface water in the form of runoff. *Salmonella* and *Escherichia coli* 0157:H7 are both manure-born pathogens that can be contributed from agricultural non-point sources and pose a risk to public health due to the low infective dose of each pathogen. Losses of pathogens from manure is mainly dependent on runoff during rainfall events and, as noted before, is heavily dependent on tillage and soil cover (Jenkins et al., 2015). Potential for surface water contamination from areas receiving manure or effluent can last for several months, depending on the survival of bacteria. Survival of *Escherichia coli* 0157:H7 has been shown to survive in soil for a few days up to 1 year in extreme cases, depending on soil factors, such as moisture and nutrient availability, as well as management practices, such as application timing and methods (Wang et al., 2014). While manure-based fertilizers are not typically used in conventional corn production, they are typically used as a fertilizer source in organic corn production as well as other organic agricultural systems. For this reason, the National Organic Program requires a 120-day interval between manure applications and crop harvest (Ingham et al., 2004). Grass filter strips are typically used to reduce bacterial loads in watersheds receiving manure applications. However, they have not been proven to be as effective in trapping bacteria as trapping sediment. A study by Coyne et al. (1995) showed that a 9-m grass filter strip that trapped 99% of soil from erosion only trapped 74% of fecal coliforms. While some might consider this sufficient, concentrations of fecal coliforms must meet the minimum standards of 200 colony forming units (cfu) per 100 mL, which is routinely exceeded by runoff from manure fertilized areas.

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CHAPTER 2

NITROGEN AVAILABILITY AND CROP GROWTH IN A LIVING MULCH CORN PRODUCTION SYSTEM[2]

[2] J.S. Andrews, Z.P. Sanders, N.S. Hill, U.K. Saha. To be submitted to *Agronomy Journal*.

Abstract

Successful living mulch (LM) systems are able to meet the N requirements of the corn crop without competing with the crop during critical periods of development. Corn production in a white clover LM was compared to both a cereal rye (CR) and crimson clover (CC) corn production system. The LM treatment received 56 kg N ha^{-1}, the CC treatment received 112 kg N ha^{-1}, and the CR treatment received 280 kg N ha^{-1}. The LM white clover released approximately 82 and 149 kg N ha^{-1} during the growing season in 2015 and 2016, respectively. However, the LM treatment had lower inorganic soil N during many parts of the 2015 and 2016 growing seasons due to competition from the clover and reduced N mineralization. Competitive water uptake by the LM clover likely reduced water available for N mineralization of LM clover residue. Lower N availability reduced corn growth in the LM treatment compared to the CC and CR treatments as corn height, light interception, and biomass were affected during both growing seasons. Total N uptake was lower in the LM treatment than the CC treatment, with the CR treatment not different from either. Grain yield was 10.4 Mg ha^{-1} in the LM treatment and 13.3 and 13.0 Mg ha^{-1} in the CC and CR treatments, respectively. Therefore, the success of the LM system is likely dependent on providing adequate moisture to reduce competition between the two plants and allow for efficient N mineralization of white clover residue.

1. Introduction

Nitrogen (N) is considered the most limiting nutrient to corn growth and yield and is usually supplied in inorganic forms (Raun et al., 1999). In the southern Piedmont, corn requires 200 to 280 kg mineral N ha^{-1} in order to achieve maximum economic yield (Raun et al., 1999; Lee et al., 2015). N deficiency during critical periods of growth can be detrimental to both vegetative and reproductive stages of the corn plant, and ultimately can reduce grain yields (Uhart & Andrade, 1995; Scharf et al., 2002; Lee et al., 2015).

Most soils in Georgia are considered dispersive and subject to erosion (Langdale et al., 1992). Cover crops are one way of preventing soil erosion and providing supplemental N for the main crop (Schomberg et al., 2006). Cereal rye is a popular cover crop used in the Southeastern US because of its ease of establishment and rapid growth (Snapp et al., 2005). Crimson clover is another popular cover crop which is used to fix atmospheric N for subsequent row-crop use (Young-Mathews, 2013). Living mulch systems attempt to capitalize on the perennial growth habit of the LM species to control erosion as well as the N fixing capability of a legume to supply nutrients to the subsequent crop (Echtenkamp et al., 1989; Zemenchik et al., 2000). However, there appears to be a cost to using the LM system because the perennial cover crop competes with the row crop for resources, particular water and N (Affeldt et al., 2004, Kurtz et al., 1952; Zemenchik et al., 2000). Additionally, conditions which promote mineralization of N within the LM system are poorly understood, and may limit availability and uptake of nutrients that are sequestered in the LM crop. Therefore, the objective of this study was to compare the soil N dynamics of a living mulch, cereal rye, and crimson clover cover crops on corn growth and yield.

2. Materials and Methods

2.1 Site description

The study was performed from October 2014 through February 2017 at the J. Phil Campbell Research and Education Center in Watkinsville, GA (33°52'09.5"N 83°26'59.8"W). The soil is classified as a Cecil sandy loam (fine, kaolinitic, thermic typic kanhapludults) which was later confirmed in August 2016 by a pedological description performed adjacent to the research plots. Weather data were collected by The Georgia Environmental Monitoring Network station located 150 m from the plot area (http://georgiaweather.net/?variable=SI&site=WATUSDA&title=Site%20Information).

2.2 Experimental design and field operations

The experimental design was a randomized complete block design with six replications using three cover crop treatments, cereal rye (*Secale cereal* L.) (CR), crimson clover (*Trifolium incarnatum* L.) (CC), and 'Durana' white clover (*Trifolium repens* L.) living mulch (LM). Prior to land preparation, soils were tested for pH, phosphorus, and potassium by the University of Georgia Soil Testing Laboratory. Lime, phosphorus, and potassium were applied so that pH was above 6.2 and available phosphorus and potassium by Mehlich 1 were at least 45 and 140 ppm, respectively. Land was prepared by disking the soil twice, which was followed by leveling and firming the ground with a cultipacker. Cover crops were seeded at rates of 100, 28, and 13 kg ha^{-1} of CR, CC, and LM on 17 October, 2014. Plots were cultipacked a second time to ensure good seed to soil contact. On 16 March and 8 April, 2015, the CC and CR plots were killed with a broadcast application of glyphosate (N-(phosphonomethyl)glycine) and dicamba (3,6-Dichloro-2-methoxybenzoic acid) at 1.12 and 1.20 kg a.i. ha^{-1}, respectively. The LM plots received a 20-cm banded herbicide application of the herbicides in rows centered on 90-cm on 8 April, 14 days prior to planting corn (*Zea mays*). All research plots were planted with corn (DeKalb DKC64-69, GENVT3P) on 21 April, 2015 using a John Deere 7300 MaxEmerge no-till planter at a population density of 90,000 plants ha^{-1}. All plots received a herbicide application of pendimethalin (3,4-Dimethyl-2,6-dinitro-*N*-pentan-3-yl-aniline) and atrazine (1-Chloro-3-ethylamino-5-isopropylamino-2,4,6-triazine) at rates of 1.20 kg a.i. ha^{-1} and 1.12 kg a.i. ha^{-1}, respectively, at the VE stage of development. The herbicides were applied as broadcast treatments for the CC and CR and a 20-cm banded application over the corn row in the LM plots. Plots were 6.1 by 7.3 m, allowing for eight rows of corn on 90-cm centers.

Cereal rye plots received 56 kg ha^{-1} supplemental N at planting and 224 kg N ha^{-1} at the V6 stage of development. Crimson and white clover plots received no N at planting but received 112 kg N ha^{-1} and 56 kg N ha^{-1} at the V6 stage of development, respectively. N fertilization for the CR treatment was based on a recommendation of 280 kg N ha^{-1} (Lee et al., 2015). N fertility for the CC and LM treatments was adjusted to lower rates based upon N credits from previous experiences (Sanders et al., in press). Campbell CS625 reflectometers (Campbell Scientific, Logan, UT) were placed at two different soil depths both within and between corn rows. The rods were 30 cm in length and installed at an angle of 30 degrees from the surface. One rod was inserted at the soil surface to measure water content from 0 - 15 cm and another at 15 cm to measure water content from 15 - 30 cm. All reflectometers were located within the center of the center rows of each plot. A soil moisture release curve based on the van Genuchten (1980) equation was created using the evaporation method (Arya, 2002) and a Decagon HYRPOP device (Pullman, WA) from several soil cores collected from the plot area. The soil moisture release curve indicated that field capacity (-0.03 MPa) was 0.24 cm^3 water cm^{-3} soil and wilting point (-1.5 MPa) was 0.11 cm^3 water cm^{-3} soil. Overhead irrigation was applied to maintain soil water between 40 and 90% plant available water based on soil water data from the reflectometers. Corn was harvested on 10 August, 2015.

Cover crops were reestablished in October 2015 in the same plot areas and at the same seeding rates as 2014. Herbicide applications were made to the CR and CC plots on 23 March and 4 April, 2016, and to the LM plots on 14 April as previously described. Plots were planted on 28 April with 90,000 plants ha^{-1} as previously described, and pendimethalin and atrazine applied at the VE stage of development as previously described. All plots were harvested on 8 August, 2016.

2.3 LM clover mass, N release, water content

Weekly changes in clover mass in the LM plots were determined using a pre-calibrated rising plate meter (RPM) (FarmWorks Precision Farming Systems, Feilding, NZ) as outlined by Sanders et al. (201x, in press). Ten to twelve RPM height measurements were made within the center two rows in each LM plot weekly to determine clover mass. Hand harvested clover samples were subjected to Kjeldahl digestion and N content determined at the UGA Agricultural and Environmental Services Laboratory in Athens, GA using a Timberline TL-2800 Ammonia Analyzer. Total N in the white clover was calculated weekly as the product of white clover mass and the total N content of the clover. Decline in clover mass from one week to the next was used to estimate N release from the clover.

Water content data were measured and recorded on 10-min intervals, stored on data loggers, and downloaded weekly. Data were averaged over 10-day periods from planting until 100 DAP.

2.4 Soil N

Eight soil cores were randomly sampled to a 15-cm depth weekly from each plot using a handheld soil probe that was 2 cm in diameter. Soil cores were taken from the center two rows of the plot, the cores combined, air dried, and stored at 4℃. Five grams of soil from each sample was extracted at 21℃ with 40 mL of 1M KCl for NO_3-N and NH_4-N analysis. Potentially mineralizable nitrogen (PMN) was estimated from three grams of soil from each sample with 20 mL of 2M KCl at 100℃ for 4 hours. Soil extracts were analyzed for ammonium and nitrate concentration at the UGA Agricultural and Environmental Services Laboratory in Athens, GA using a Timberline TL-2800 Ammonia Analyzer (Timberline, Boulder, CO). Potentially mineralizable soil N was calculated by subtracting the NH_4-N concentrations of the cold KCl extractions from the NH_4-N concentrations of the hot KCl extractions (Campbell et al., 1994). Soil bulk density to a 15-cm depth was determined by drying and weighing a known volume of soil and was later used to calculate NH_4-N, NO_3-N, soil inorganic N, and PMN on a per hectare basis.

2.5 Crop growth and yield

After emergence, plant height measurements were taken weekly from eight randomly selected corn plants in the center two rows of the plots. Light interception by the corn was monitored weekly in each plot using a LI-COR LI 191sb Line Quantum Sensor (Li-Cor, Lincoln, NE). Measurements were taken from above the plant canopy and at the soil surface immediately thereafter at two locations between the center two rows of the plots. Percent light interception was defined as:

$$\%\text{ light interception} = 1 - [(\text{light below corn canopy}/\text{light above corn canopy}) \times 100] \qquad (1)$$

Beginning 25 days after planting, five corn plants were harvested weekly by cutting the plant at the soil surface from the second row from the outside of each plot. Plants were dried at 65℃ and mass recorded. All corn plant samples were ground and analyzed for total N using near infrared reflectance (NIR). A subset of samples was analyzed for total N using Kjeldahl digestion (Baker & Thompson, 1992) and used to correct for bias of the NIR data (Schomberg et al., 2006). At the end of the growing season, corn ears from the center 3.0 m of the center two rows of each plot were hand harvested and dried at 65℃. Corn was hand-shelled, grain weighed, and weights adjusted to 15% moisture to calculate yield. Internal utilization efficiency (IUE) of N by the corn plant was defined as:

$$IUE = \text{yield (kg ha}^{-1}\text{)/N uptake (kg ha}^{-1}\text{)} \quad (2)$$

2.6 Statistical analysis

Data were analyzed using the PROC MIXED subroutine of SAS (SAS Institute, Cary, NC). Sampling dates (DAP) were used as a covariate to determine interactions with other treatment variables. Covariate analysis determined significant interactions with cover crop treatments for light interception, corn height, clover mass, NO_3-N, NH_4-N, and total inorganic N. Data were sorted by DAP and PROC GLM was used to determine differences among cover crop treatments. There were cover crop and year effects for total N uptake, total corn biomass, and grain yield, but there were no year × cover crop interactions. Therefore cover crop and year effects were analyzed with cover crop treatment and replications as fixed effects and year as a random effect. All means were separated using a Fisher's protected LSD at the $P = 0.05$ level of significance.

3. Results

3.1 Meteorological conditions

Average temperature and precipitation at the J. Phil Campbell Research and Education Center from March through August was 21.6℃ and 729 mm in 2015 and 22.1℃ and 435 mm in 2016 (Table 2.1). An additional 173 mm of irrigation water was applied in 2015 and 435 mm in 2016 as a result of a 2016 drought.

3.2 LM clover mass, N release, water content

Average clover mass in the LM treatment before the banded herbicide application was 2177 kg dry matter ha^{-1} in 2015 and 3292 kg dry matter ha^{-1} in 2016 (Figure 2.1A). However, by the end of the growing season clover mass was similar among years. Because clover N concentration was nearly constant within sampling dates, the estimated N release from reduced clover mass was greater in 2016 than 2015 Estimated N release in the LM treatment from herbicide application was 14.3 and 23.5 kg N ha^{-1} for 2015 and 2016, respectively (Figure 2.1B). Cumulative N release from LM clover at the end of the growing season was 82.0 in 2015 and 146 kg N ha^{-1} in 2016.

Volumetric water content between rows in the LM treatment was significantly lower than in the CC and CR treatments during parts of both the 2015 and 2016 growing seasons (data not shown). The between row soil moisture content in the living mulch plots up until corn canopy closure (40 DAP) was -0.14 MPa in 2015 and -0.35 MPa in 2016. After canopy closure, it was -0.19 MPa in both 2015 and 2016. Conversely, between row soil moisture content of the CR and CC plots up to canopy closure was -0.05 MPa in 2015 and -0.09 MPa in 2016. After canopy closure, it was -0.14 MPa in both 2015 and 2016.

3.3 Soil N

Soil inorganic N was significantly lower in the LM treatment than in the CC and CR treatments during much of the growing season in 2015 (Figure 2.2A). Average soil NH_4-N was not different among treatments in 2015, but the LM treatment had lower average soil NO_3-N than both the CC and CR treatments (Table 2.2). Average soil inorganic N was lower in the LM treatment than the CR treatment in 2015. Significant differences in soil inorganic N were observed in only 4 of 12 sampling dates in 2016, but the LM treatment had lower soil inorganic N when the differences were observed (Figure 2.2B). Average soil NH_4-N was not different among treatments in 2016, while the LM treatment had lower average NO_3-N than the CC treatment. Average soil inorganic N was lower in the LM treatment than both the CC and CR treatments in 2016 (Table 2.2). All cover crop treatments had significantly lower soil inorganic N in 2016 than 2015.

No differences between treatments were observed in soil PMN during the growing season in either 2015 or 2016 (Table 2.2). The soil PMN values were constant over the growing season in 2015, but were greater during midseason in 2016 (Table 2.3).

3.4 Crop growth and yield

Generally, corn height was similar among cover crop treatments in 2015 until 48 DAP, after which the LM treatment was shorter than the other cover crops (Figure 2.3A). In 2016, corn in the CC and CR treatments was taller than corn in the LM treatment 54 DAP and thereafter (Figure 2.3B).

Light interception was not significantly different between treatments throughout the 2015 growing season (Figure 2.3C). However, in 2016 the LM treatment had significantly less light interception than both the CC and CR treatments 40 DAP and thereafter (Figure 2.3D).

Aboveground corn biomass was not different during the 2015 growing season except for 29 DAP when the LM treatment was lower than the CC and CR treatment (Figure 2.4A). In 2016 the LM treatment had the lowest biomass from 26 DAP and thereafter, however, 64 DAP and afterwards the CR treatment was not different from the LM treatment (Figure 2.4B).

N uptake by corn was not different among cover crop treatments through 57 DAP in 2015 (Figure 2.4C). However, corn in the LM treatment had less N uptake than that in the CC treatment, but similar N uptake to the CR treatment 64 DAP and thereafter. Corn in the LM treatment had lower N uptake than either the CC or CR treatments for nearly all sampling dates in 2016 (Figure 2.4D). Total N uptake and corn biomass was greater in the CC treatment than the LM treatment, but the CR treatment was not different from either the CC or LM treatment (Table 2.4). Grain yields were greater in the CC and CR treatments than the LM treatment. Total N uptake, corn biomass and grain yield were all greater in 2015 than during the drought year of 2016. No differences in IUE of N were observed between treatments or between years.

4. Discussion

Small changes in soil volumetric water content dramatically affect the water content and N mineralization of crop residues (Quemada and Cabrera, 1997). The optimum soil moisture range for N mineralization is between -0.03 and -0.01 MPa (Myers et al., 1982, Stanford et al., 1973). These data suggest that white clover water use in the LM plots decreased the soil moisture content, which reduced the N mineralization rates of dead residue and decreased the conversion of released N in the residue to mineralized N in 2016. Reduced mineralization of LM-N, combined with the potential N release due to shading of the clover, suggests a significant amount of organic N was in the detrital layer of clover laying on the soil surface.

Lower inorganic soil N in the LM treatment is also likely to be affected by nutrient competition between the corn and clover (Affeldt et al., 2004; Kurtz et al., 1952; Zemenchik et al., 2000). Although legumes are capable of fixing their own N, they prefer mineral N over their physiological investment in the N fixation biochemical pathway (Svenning et al., 1996). Zemenchick et al. (2000) had similar soil N effects when growing corn in a kura clover living mulch.

Lower soil inorganic N in the LM treatment in 2016 likely led to reduced growth and development of the corn plant, which had notable effects on corn plant height, biomass, and canopy light interception. Presence of mineral N at the V3 to V6 stage of growth is critical to stimulate corn growth and yield (Uhart & Andrade, 1995; Scharf et al., 2002; Lee et al., 2015). However soil inorganic N was lower in the LM treatment compared to other treatment at this stage of development in 2016 (Figure 2.4B). While corn height and light interception was lower in the LM treatment than the CC and CR treatments in the 2016 growing season, shading was sufficient to elicit reductions in clover mass. White clover is a C_3 plant and high temperatures combined with reduced water availability may have resulted in environmental stresses that reduced biomass in 2016 (Weis & Berry, 1988).

Lower total N uptake and aboveground corn plant biomass in the LM treatment resulted in lower grain yield than the CC treatment. Lower yields in 2016 compared to 2015 were likely the result of insufficient N at critical periods when kernel development began (Lee et al., 2015;

Uhart & Andrade, 1995). Reduced inorganic soil N in the LM treatment may be due to less total N applied, competition from intercropped clover, and/or reduced mineralization of clover residue. Other studies found similar yield reductions in corn grain yield in living mulch systems (Affeldt et al., 2004; Echtenkamp & Moomaw, 1989; Scott et al., 1987; Zemenchik et al., 2000). The mechanism for yield reduction in a kura clover LM system was attributed to early season competition between the corn and the clover (Affeldt et al., 2004; Zemenchik et al., 2000). Results herein suggest that environmentally dependent N mineralization and availability to the corn plant are also likely sources of variation in corn development and yield within the LM system.

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Table 2.1: Monthly precipitation, irrigation, average temperature, and potential evapotranspiration from March through August of 2015 and 2016 at the J. Phil Campbell Research and Education Center in Watkinsville, GA.

	Year									
	2015					2016				
Month	Precip. (mm)	Irrig. (mm)	P.+I. (mm)	Avg. temp. (C)	PET (cm)	Precip. (mm)	Irrig. (mm)	P.+I. (mm)	Avg. temp. (C)	PET (cm)
March	86	---	86	13.1	5.8	53	---	53	15.1	8.4
April	208	---	208	17.9	8.4	58	19	77	17.0	12.0
May	47	78	125	21.6	13.8	32	95	127	20.8	14.3
June	59	76	135	25.4	14.5	117	152	269	26.1	17.2
July	131	19	150	26.7	14.7	33	114	147	27.2	16.9
August	198	---	198	25.1	13.6	142	---	142	26.4	13.0
Total	729	173	902	-	70.8	435	380	815	-	81.8

Table 2.2: Mean soil NH_4-N, NO_3-N, inorganic N, and potentially mineralizeable N (PMN) to a 15-cm depth during the 2015 and 2016 corn growing seasons in the crimson clover (CC), living mulch (LM), and cereal rye (CR) treatments.

	2015	2016	LSD
		NH_4-N (kg ha^{-1})	
CC	44.2	5.85	8.72
LM	53.8	5.22	10.2
CR	44.7	7.09	9.78
LSD	NS	NS	-
		NO_3-N (kg ha^{-1})	
CC	48.1	33.6	11.7
LM	23.0	23.4	NS
CR	55.1	31.9	12.8
LSD	12.4	9.1	-
		Inorganic N (kg ha^{-1})	
CC	92.3	39.5	13.0
LM	76.8	28.6	13.6
CR	99.8	39.0	14.8
LSD	16.9	10.4	-
		PMN (kg ha^{-1})	
CC	40.1	40.3	NS
LM	43.3	42.8	NS
CR	40.8	43.4	NS
LSD	NS	NS	-

Table 2.3: Mean soil potentially mineralizeable N (PMN) to a 15-cm depth of the three cover crop treatments (crimson clover, living mulch, and cereal rye) during each week of the growing season in 2015 and 2016. †Indicates a significant change from the previous measurement.

2015		2016	
DAP	Mean PMN	DAP	Mean PMN
14	40.1	0	32.7
21	40.6	7	34.8
29	40.9	15	39.2†
36	42.4	22	42.5
43	43.2	28	42.2
49	39.1	35	37.4†
55	37.6	44	60.0†
61	39.4	50	42.6
69	39.7	57	65.1†
76	34.8†	64	38.4†
82	35.9	72	35.5
		78	40.2†
LSD (0.05)		85	37.3

Table 2.4: The effect of cover crop treatment (crimson clover = CC, living mulch = LM, and cereal rye = CR) and year on total corn N uptake, final aboveground corn plant biomass, corn grain yield, and N internal utilization efficiency (IUE).

Treatment	N uptake	Corn biomass	Grain yield	IUE
	kg ha^{-1}	Mg ha^{-1}	Mg ha^{-1}	-
CC	225	20.7	13.3	67.7
LM	151	16.9	10.4	78.2
CR	183	18.0	13.0	74.7
LSD	48	3.2	1.0	NS
2015	215	21.3	13.0	70.9
2016	158	15.8	11.5	76.2
LSD	40	2.6	0.8	NS

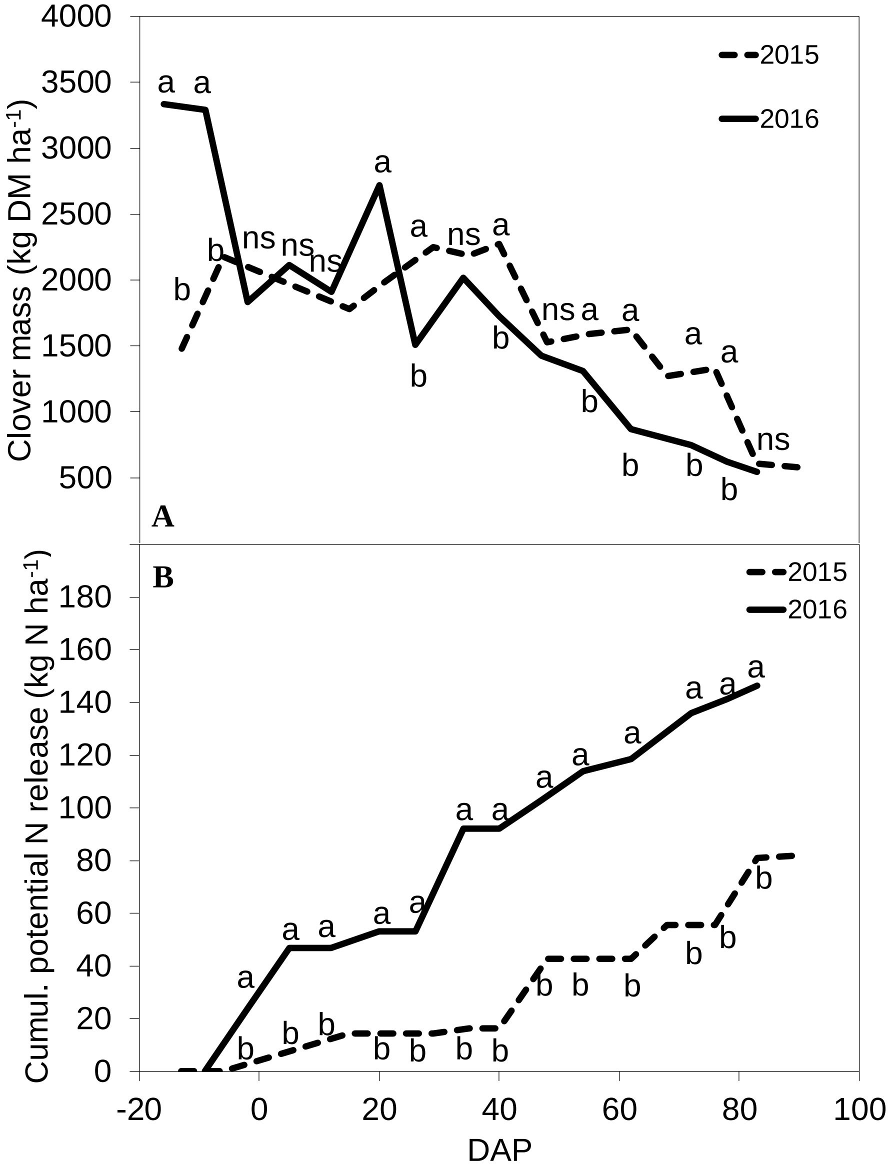

Figure 2.1: Clover mass and estimated cumulative potential N release in the living mulch treatment during the 2015 and 2016 corn growing seasons.

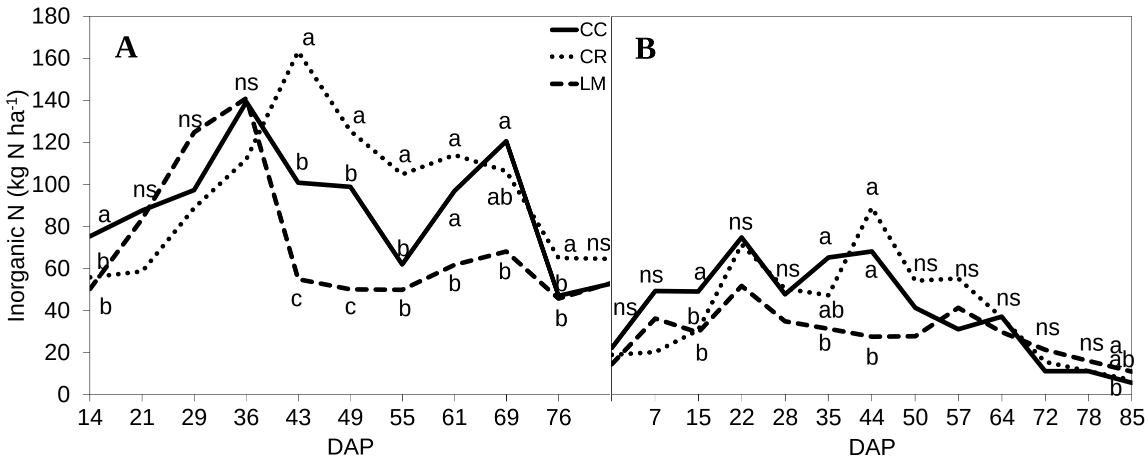

Figure 2.2: Inorganic soil N during the 2015 (2.4A) and 2016 (2.4B) corn growing seasons at the 0 to 15-cm depth in the crimson clover (CC), cereal rye (CR), and living mulch (LM) cover crop treatments.

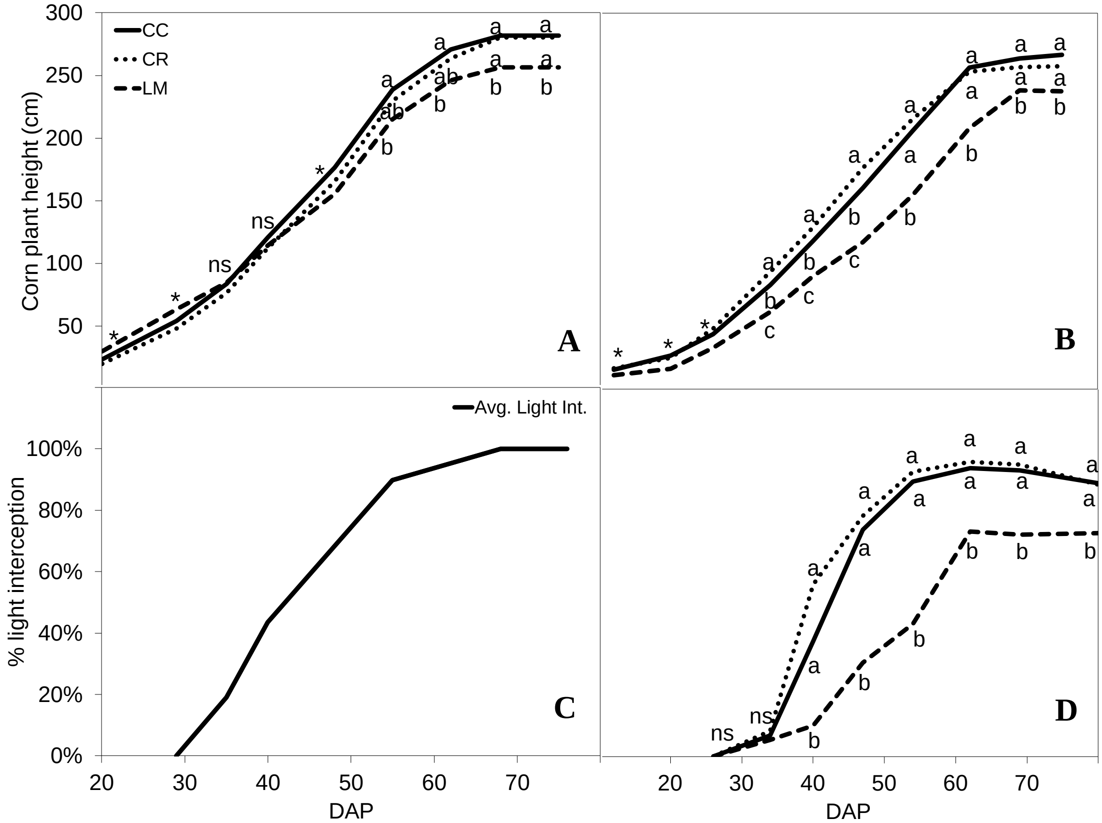

Figure 2.3: Corn height and light interception during the 2015 (left column) and 2016 (right column) growing seasons in the crimson clover (CC), cereal rye (CR), and living mulch (LM) cover crop treatments. An asterisk (*) is used to signify differences where space will not permit.

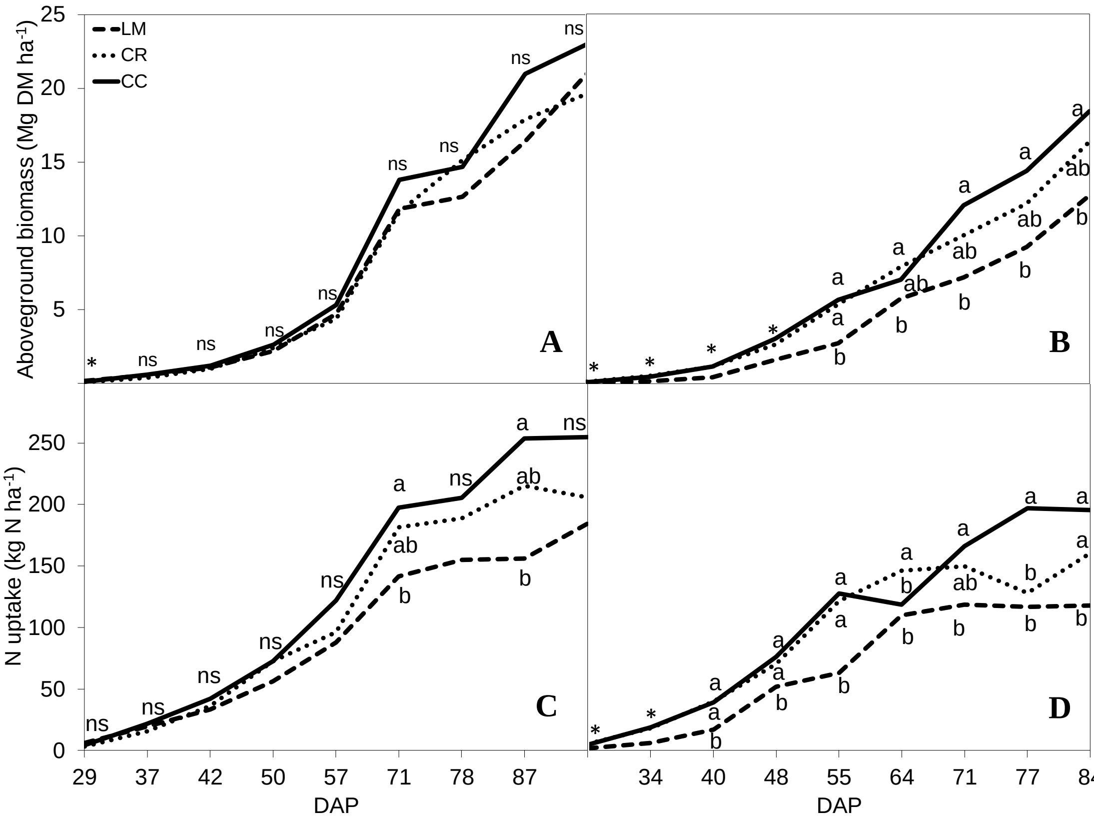

Figure 2.4: Aboveground corn plant biomass and corn N uptake during the 2015 (left column) and 2016 (right column) growing seasons in the crimson clover (CC), cereal rye (CR), and living mulch (LM) cover crop treatments. An asterisk (*) is used to signify differences where space will not permit.

CHAPTER 3

SIMULATION OF NITRATE LEACHING IN A LIVING MULCH, CEREAL RYE, AND CRIMSON CLOVER CORN PRODUCTION SYSTEM[3]

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ABSTRACT

A HYDRUS-1D model was used to simulate water and nitrate movement in three different corn production systems. Corn grown in the southern Piedmont requires 200 to 250 kg N ha^{-1} annually and can require up to 0.87 cm of water per day, making groundwater systems susceptible to nitrate pollution due to high nitrogen and excess irrigation. Three different cover cropping and fertilization treatments were simulated for 2015 and 2016: cereal ryegrass with 280 kg N ha^{-1} (CR), crimson clover with 112 kg N ha^{-1} (CC), and a white clover living mulch with 56 kg N ha^{-1} (LM). 2015 and 2016 were considered model calibration and validation periods, respectively. A water and nitrate flux model was created for each treatment and was evaluated using root mean square error (RMSE) and index of agreement (*d*). The model period simulated water and NO_3-N fate and transport from planting in April through February of the next year. NO_3-N leaching below 1 m in the 2015 - 2016 model period was 47.8, 35.4, and 75.4 kg NO_3-N ha^{-1} for the CC, LM, and CR treatments, respectively. NO_3-N leaching in the 2016 - 2017 was much lower than the previous model period due to reduced precipitation during the winter months. All treatments during that time lost less than one kg NO_3-N ha^{-1}. Differences in leaching amounts between treatments were due to differences in N application rate, timing, and N uptake by the corn plant.

1. Introduction

Intensive and irregular rainfall events and significant erosion from previous agricultural practices make the soils in the southern Piedmont relatively unproductive without careful management of irrigation and input of large quantities of supplemental fertilizer (Adams et al., 1970). Water and nitrogen are therefore important inputs for crop production, with higher yielding varieties of crops demanding greater quantities of water and nutrients over the course of the growing season. In this region, 200 to 250 kg N ha^{-1} is required for corn growth annually, with N generally applied as a soluble form (Raun et al., 1999). Cecil soils, which comprise roughly two-thirds of the land area of the southern Piedmont (Hendrickson et al., 1963), were found to be particularly well-drained by Bruce et al. (1983) due to their sandy upper horizons and lower clay horizons with large aggregates and strong structure. High amounts of soluble N fertilizer combined with well-drained soils make corn production areas of the southern Piedmont susceptible to nitrate (NO_3^-) leaching to groundwater.

As an anion, NO_3^- is not readily adsorbed by negatively charged soil particles and can move with soil water through the profile to groundwater in soils without positive charges. Leached NO_3-N reduces efficient use of nitrogen by producers as well as contaminates groundwater. Groundwater in the southern Piedmont is frequently used by industry, agriculture, and homeowners as drinking water (Fortuna, 2004). Drinking water contaminated with NO_3-N can have serious health effects, particularly methemoglobinemia, otherwise known as "blue baby syndrome." For this reason, the US Environmental Protection Agency has set a maximum contamination level of 10 mg L^{-1} NO_3-N to be safe for drinking water (Carpenter, 1998). Continued leaching of NO_3-N by agriculture thereby threatens groundwater systems and agricultural best management practices must be applied to protect a water source with multiple uses. NO_3-N leaching studies in corn show varying amounts of NO_3-N lost below the root zone. In a Cecil fine sandy loam, Hahne et al. (1977) reported N loss through leaching was as high as 50% of applied N in continuous corn cropping, which represented 100 kg NO_3-N ha^{-1}. Gold et al. (1990) showed that urea fertilized corn lost 79 kg NO_3-N ha^{-1} in Rhode Island using the CREAMS model. Results from Andraski et al. (2000) were representative of observed variability: NO_3-N leached in continuous corn ranged from 0 to 88 kg ha^{-1} based on N fertilizer rate, irrigation, and timing.

However, conservation practices have been shown to significantly reduce NO_3-N leaching losses in corn production. Using suction lysimeters, Angle et al. (1989) found that the average NO_3-N concentration in soil water at a 0.75-m depth was 15 mg L^{-1} in a no-till corn watershed and 30 mg L^{-1} in a conventionally tilled corn watershed. Yadav (1997) saw similar results; NO_3-N accumulation in no-till corn soil was 20 to 42% lower than conventionally tilled corn soil based on N application rate. In addition to tillage practices, cover crops have also been shown to significantly reduce NO_3-N leaching. During winter fallow periods, cover crops uptake excess NO_3-N in the soil profile that is vulnerable to leaching (Brandi-Dohrn et al., 1997; McCracken et al., 1994). Other conservation practices, such as split applications of fertilizer (Kanwar et al., 1988; Arora and Juo, 1982) can also reduce NO_3-N leaching, making conservation agriculture a better management practice for protecting vulnerable groundwater systems.

One common method used to monitor soil water quality is through the use of porous-cup suction lysimeters. These samplers are relatively noninvasive, can obtain leachate from a large number of treatments and replications, and a much less cost-prohibitive relative to more elaborate lysimeter systems. However, they are not capable of quantifying water drainage

(Andraski et al., 2000). Therefore, a modeling approach must be used with solute concentration data to evaluate NO_3-N movement. Several simulation models have been used to evaluate N leaching, such as NLEAP (Nitrogen Leaching and Economic Analysis Package), APSIM (Agricultural Production Systems Simulator), and HYDRUS. HYDRUS-1D was selected for simulating water and NO_3-N loss in different Piedmont corn productions systems in our study because it provides relative ease of use and is often used in similar studies.

The objectives of this study were to (1) calibrate and validate a HYDRUS-1D model that simulates water and NO_3-N transport in three different corn production systems that have different cover crops and N application rates and (2) use the model to compare water and NO_3-N transport in the three systems.

2. Materials and Methods

2.1 Site description

The study was performed from October 2014 through February 2017 on research plots at the J. Phil Campbell Research and Education Center in Watkinsville, GA. Data were generally collected during the corn growing season of 2015 and 2016, from April through August. The soil is classified as a Cecil sandy loam (fine, kaolinitic, thermic typic kanhapludults) which was later confirmed by a pedological description performed adjacent to the research plots. Weather data was collected by The Georgia Environmental Monitoring Network station located 150 meters from the plot area. Average annual rainfall and average annual temperature at the J. Phil Campbell Research and Education Center are 1,219 mm and 16℃, respectively.

2.2 Experimental design and field operations

The experimental design was a randomized complete block design with six replications of three cover crop treatments, cereal rye (*Secale cereal* L.) (CR), crimson clover (*Trifolium incarnatum*) (CC), and 'Durana' white clover (*Trifolium repens*) (LM). Prior to land preparation, soils were tested for pH, phosphorus, and potassium by the University of Georgia Soil Testing Laboratory. Lime, phosphorus, and potassium were applied so that pH was above 6.2 and available phosphorus and potassium were at least 90 and 280 kg ha^{-1}, respectively. Land was prepared by disking the soil twice, which was followed by leveling and firming the ground with a cultipacker. Cover crops were seeded at rates of 90, 28, and 11 kg ha^{-1} of CR, CC, and LM on 17 October, 2014. Plots were cultipacked a second time to ensure good seed to soil contact. On 16 March, 2015, the CC and CR plots were killed with a broadcast application of glyphosate (N-(phosphonomethyl)glycine) and dicamba (3,6-Dichloro-2-methoxybenzoic acid) at 1.12 and 1.20 kg a.i. ha^{-1}, respectively. The LM plots received a 20-cm banded herbicidal application to rows centered on 90-cm prior to planting corn (*Zea mays*). All research plots were planted with corn (DeKalb DKC64-69, GENVT3P) on 21 April, 2015 using a John Deere 7300 MaxEmerge no-till planter at a population density of 90,000 plants ha^{-1}. All plots received an herbicide application of pendimethalin (3,4-Dimethyl-2,6-dinitro-*N*-pentan-3-yl-aniline) and atrazine (1-Chloro-3-ethylamino-5-isopropylamino-2,4,6-triazine) at rates of 1.20 kg a.i. ha^{-1} and 1.12 kg a.i. ha^{-1}, respectively, at the VE stage of development. CC and CR plots received a broadcast application of herbicide while LM plots received a 20-cm banded application over the corn row. Plots were 6.1 by 7.3 meters, allowing for eight rows on 90-cm planting centers. Corn was harvested on 10 August, 2015 in all plots.

Cereal rye plots received 80 kg ha^{-1} supplemental N at planting and 200 kg N ha^{-1} at the V6 stage of development. Crimson and white clover plots received no N at planting but received 112 kg N ha^{-1} and 56 kg N ha^{-1} at the V6 stage of development, respectively,. Fertilization rates were based on a recommendation of 280 kg N ha^{-1} (Lee et al., 2015). In three replications, Campbell CS625 reflectometers (Campbell Scientific, Logan, UT) were placed at two different depths both within and between corn rows. The rods were 30 cm in length and installed at an angle of 30 degrees from the surface. One rod was inserted at the soil surface to measure water content from 0 - 15 cm and another at 15 cm to measure water content from 15 - 30 cm. All reflectometers were placed within the center row of each plot and located at the center point of the row. Water content data were measured and recorded on 10 minute intervals and stored on data loggers for weekly download. In the other three replications, two suction lysimeters were placed 1 meter apart in the center of each plot. A soil auger was used to remove soil to a depth of 75 cm, and a 1-m long suction lysimeter was installed with a slurry of Bt horizon soil to cover the lysimeter's ceramic cup. Above the Bt soil slurry, a slurry of A horizon soil and kaolinite was used to fill in the remainder of the augered hole, firmly hold the suction lysimeter in place, and prevent preferential flow down the side of the lyimeters. Large PVC pipes with caps were constructed and placed over the tops of the lysimeters to prevent damage to the lysimeters and prevent preferential flow. Overhead irrigation was applied to maintain soil water between 40 and 90% plant available water based on soil water data from the reflectometers.

Plots were reestablished in October 2015 in the same plot areas and at the same seeding rates as 2014. Herbicide applications were made to the CC and CR plots on 23 March, 2016, and to the LM plots on 14 April, to prepare the plots for planting. As in 2015, the LM plots received a 20-cm banded application of glyphosate and dicamba on 90-cm rows. Plots were planted on 28

April with 90,000 plants ha^{-1} as previously described. All plots were harvested on 8 August, 2016.

2.3 Field sampling and laboratory analysis

Water content for the 0 - 15 cm and 15 - 30 cm depths was recorded at ten minute increments and downloaded weekly throughout the year. Suction lysimeters, after being cleared of contents, had a vacuum applied at 7 kPa using a handheld vacuum pump. The lysimeters were kept under suction for approximately 18 - 36 hours before sample collection, and samples were generally taken once a week. To collect samples, a tube was inserted into a lysimeter and connected to a 1-L side-arm flask through a rubber stopper on the top opening of the flask. A handheld vacuum pump was used to create suction so that water would flow from the lysimeter into the side-arm flask. Generally, only one lysimeter per plot was able to collect a water sample, but 25 mL of water from each lysimeter were combined when both lysimeters produced samples. In addition to soil water samples, soil samples of each plot were taken at 0 – 15 cm depths each week using a soil sampling probe. Several soil samples were taken weekly from each plot from various distances within and between rows to get a combined sample representative of the entire plot area. Samples from each plot were combined in a plastic bucket and air dried for 48 hours and were stored at 4℃ until they could be extracted. Five grams soil from each sample was extracted with 40 mL of 1M KCl for NO_3-N and NH_4-N. Potentially mineralizable nitrogen (PMN) was extracted from three grams soil from each sample with 20 mL of 2M KCl and heated in a hot water bath at 100℃ for 4 hours. PMN was calculated by taking NH_4-N concentrations from the heated soil extracts and subtracting it from the NH_4-N concentration in the unheated extracts. After extraction, all soil samples were stored at -18℃. Soil water samples and soil extracts were analyzed for nitrate concentration at the UGA

Agricultural and Environmental Services Laboratory in Athens, GA using a Timberline TL-2800 Ammonia Analyzer (Boulder, CO).

Weekly changes in clover mass in the LM plots were determined using a rising plate meter. The plate meter was calibrated by creating a weekly regression equation that related clover height, measured using a FarmWorks Model F200 rising plate meter (RPM) (Feilding, NZ), to clover mass. For each weekly regression, six clover height measurements taken using the RPM and clover mass quantified from hand sampling clover from a 0.1-m^2 quadrat placed immediately below where the RPM measurement was taken. Clover samples were dried at 65℃ and mass recorded. Ten to twelve RPM height measurements were made within the center two rows in each LM plot every week to determine clover mass. The clover samples from the RPM calibration were subjected to Kjeldahl digestion for plant samples (Baker and Thompson, 1992) and analyzed for N content. Nitrogen mass in the white clover was calculated weekly as the product of white clover mass and the nitrogen content of the clover. Changes in N mass from one week to the next were used to estimate N release from the clover and can be seen in Figure 3.1.

2.4 Model selection and description

The HYDRUS-1D model was used to simulate water and nitrate movement in the three corn production systems from planting in April of 2015 and 2016 through February of the following year. The one-dimensional model was developed to simulate the vertical movement of soil water, heat, and solutes in variably saturated-unsaturated media (Simunek et al., 2016). Soil water movement for the experimental scenario is described in the model by Simunek et al. (2016) using a numerical solution to the Richards (1931) equation:

$$\frac{\partial\theta}{\partial t} = \frac{\partial}{\partial z}\left[K(h)\cdot\left(\frac{\partial h}{\partial z}+1\right)\right] - S \tag{1}$$

where h (cm) is the pressure head, z (cm) is the gravitational potential head, t (day) is time, and S (cm day^{-1}) is root water uptake rate. Many equations are available in HYDRUS-1D for describing both soil water retention and unsaturated hydraulic conductivity functions. The van Genuchten (1980) equation was used for soil water retention:

$$\theta(h) = \frac{\theta_s - \theta_r}{[1+(-\alpha h)^n]^m} + \theta_r \tag{2}$$

where α (cm^{-3}), m (dimensionless), and n (dimensionless) are fitted parameters, $\theta(h)$ is the volumetric water content (cm^3 cm^{-3}), θ_s is the saturated volumetric water content (cm^3 cm^{-3}), and θ_r is the residual volumetric water content (cm^3 cm^{-3}). The unsaturated hydraulic conductivity was also described by van Genuchten (1980):

$$K(h) = K_s\left|\frac{\theta(h)-\theta_r}{\theta_s-\theta_r}\right|^{0.5}\left(1-\left\{1-\left|\frac{\theta(h)-\theta_r}{\theta_s-\theta_r}\right|^{1/m}\right\}^m\right) \tag{3}$$

where K_s is the saturated hydraulic conductivity (cm day^{-1}), m is the fitted parameter from Eq. (2), and it is assumed that m=1-1/n.

Initial estimates of the water retention parameters for Eq. (2) – (3) were determined using the evaporation method (Arya, 2002) and a Decagon HYPROP device (Pullman, WA) with multiple soil samples taken from each soil layer in a nearby soil pit. Soil water retention parameters in the top two soil horizons were then optimized using the inverse modeling mode in HYDRUS across all models using collected soil moisture data. Final soil water retention parameters can be seen in Table 3.1.

Solute transport in HYDRUS is described using a numerical solution to the advection-dispersion equation (ADE) that has linear adsorption and chemical equilibrium, which is as follows:

$$\frac{\partial(\theta c+\rho_b K_d c)}{\partial t} = -\frac{\partial}{\partial t}\left(J_w c - \theta D_e \frac{\partial c}{\partial z}\right) - \theta\mu c - Sc \qquad (4)$$

where c is the dissolved concentration of the solute (g cm^{-3}), t is time (day), K_d is an adsorption coefficient (cm^3 mg^{-1}), D_e is the effective dispersion coefficient (cm^2 day^{-1}), z is the vertical dimension (cm), J_w is the vertical Darcy water flux (L day^{-1}), and μ is the first-order rate constant for solute transformation processes (day^{-1}).

2.5 Model domain and boundary conditions

The HYDRUS-1D model used a profile that was 100 cm deep and four materials represented the four horizon depths and soil water retention parameters outlined in Table 3.1. A total of 101 nodes were used in the model space with an equal density of one node per centimeter of model space. The upper boundary condition for water flow was an atmospheric boundary with surface runoff at the soil surface while free drainage was used as the lower boundary condition. Observation nodes were placed at 15 and 30 cm below the soil surface to represent the time domain reflectometers installed within the corn rows. Observed data was input in each observation node as the average volumetric water content (cm^3 cm^{-3}).

Solute transport had a concentration flux boundary condition at the top of the model space with no concentration gradient as the lower boundary condition. Observation nodes were placed at 8 and 75 cm below the soil surface to represent points where soil water NO_3-N concentration was sampled either by soil sampling or suction lysimeters. N fertilizer, applied as granular urea, was input as a time variable boundary condition and converted into a concentration based on the amount of N in granular fertilizer and amount of irrigation water applied. Again, nitrogen release from the clover in LM plots was added when clover mass declined from one measurement to the next.

2.6 Water uptake

The Feddes et al. (1978) model was used to describe root water uptake in HYDRUS-1D:

$$S(h) = \alpha(h) S_{max} \tag{5}$$

where $\alpha(h)$ is the coefficient of root water uptake and S_{max} is the potential water uptake rate (day^{-1}). In this model, root water uptake is zero at saturation (h_0) due to lack of oxygen and increases linearly as pressure heads decrease to h_1. Root uptake is optimum (no water stress) for pressure heads within the range of h_1 to h_2. From h_2 to h_3, root uptake decreases linearly and stops at pressure heads below h_3 due to insufficient soil water. The root water uptake parameters that provided the best fit to observed data and subsequently used in this model for all crops were h_0 = -15 cm, h_1 = -30 cm, h_2 = -400 cm, h_3 = -15,330 cm, and S_{max} = 0.5 cm day^{-1}. For reference, the parameters used for corn in Wesseling (1991) are h_0 = -15 cm, h_1 = -30 cm, h_2 = -600 cm, h_3 = -8,000 cm, and S_{max} = 0.5 cm day^{-1}.

2.7 Soil NO_3-N transformations and concentrations

Experimental results on Cecil soils show that both mineralization and nitrification can occur rapidly and potentially mineralizable N (PMN) and ammonium are generally in low concentrations (Dubey, 1968; Anderson, 1960; Cabrera, 1993), so PMN and NH_4-N in the soil profile were considered together as one bulk solute. Additionally, HYDRUS-1D only allows for the utilization of two solutes when observed data is input into the model. Other relevant HYDRUS-1D NO_3-N leaching publications, such as Wang et al. (2010) as well as Tafteh and Sepaskhah (2012), used a similar strategy involving transformations of PMN and NH_4-N into NO_3-N.

To simulate N transformations in the CC, LM, and CR models, an N chain model was used. This model included the nitrification of NH_4-N and PMN to NO_3-N. Similar to a HYDRUS-2D model by Bradshaw et al. (2013), the intermediate product of nitrification, NO_2-N, was ignored to simplify the chain model. Denitrification of NO_3-N to N_2/$N_2$0 gas was also included in the model. The first-order reaction rates for the change in NH_4-N+PMN and NO_3-N concentrations that result from the N chain models are:

$$\frac{\partial[NH_4-N+PMN]}{\partial t} = -k[NH_4N + PMN] \tag{6}$$

$$\frac{\partial[NO_3-N]}{\partial t} = -k[NH_4N + PMN] - \mu[NO_3N] \tag{7}$$

where k and μ are the nitrification and denitrification rate coefficients (day^{-1}), respectively, and t is time (T). It was assumed that these coefficients only applied to N in solution. N rate coefficients, along with adsorption (K_d), were fitted parameters that were optimized across all models in the inverse solution. Water content dependence of reaction rates were incorporated into the solute models. Water content dependence of reaction rates uses a modified of the Walker (1974) equation:

$$\omega(\theta) = \omega_r(\theta_{ref}) min\left[1, \left(\frac{\theta^B}{\theta_r}\right)\right] \tag{8}$$

where ω_r is the rate constant (day^{-1}) at the reference water content (θ_{ref}), $\omega(\theta)$ is the rate constant (day^{-1}) at the actual water content (θ), and B is a dimensionless solute-dependent parameter. The modified longitudinal dispersivity and molecular diffusion coefficient of NO_3-N in free water were used as 1.0 cm and 1.65 $cm^2\ d^{-1}$, respectively (Tafteh and Sepaskhah, 2012).

2.8 Model initial conditions and inverse data

The initial condition for soil pressure head at the uppermost observation node in each model was set to a pressure head corresponding to the water content that was measured at the beginning of the model period. The rest of the model space was set so that soil water movement was only due to gravitational flow. Inverse data for water content was input into HYDRUS-1D as the average measured volumetric water content of the three replications. Initial NO_3-N concentrations were input into the model based on both soil samples and suction lysimeter samples taken before planting, when the model period begins. Average soil water nitrate concentrations from soil sampling were determined by taking the average soil nitrate level in the soil from the three replications and converting it to a concentration using the water content and bulk density (ρ_b) (g cm^{-3}) of the sampled soil.

2.9 Model performance criteria

Observed data in the model were average soil water content ($cm^3\ cm^{-3}$) at 15 and 30-cm depths and average soil water nitrate concentration ($\mu g\ cm^{-3}$) at 8 and 75-cm depths. Two statistical procedures, root mean square error (RMSE) and index of agreement (d), were used to compare agreement between predicted and observed data.

(i) RMSE:

$$RMSE = \sqrt{\sum_{i=1}^{n} \frac{(Y_i - O_i)^2}{n}} \tag{9}$$

(ii) Index of Agreement (*d*):

$$d = 1 - \left\{ \frac{\sum_{i=1}^{n} (O_i - Y_i)^2}{\sum_{i=1}^{n} (|O_i - O| + |Y_i - O_e|)^2} \right\} \tag{10}$$

where *n* is the number of observations; Y_i is the predicted value that corresponds to observed value O_i; *O* is the mean of all observed values; O_e is the mean of all predicted values. The closer the RMSE is the 0, the model is more accurate. The value of *d* ranges from 0.0 to 1.0 and is more accurate the closer the value is to 1.0.

3. Results and Discussion

3.1 Meteorological conditions

Over the course of the study period, the average annual rainfall was 1,400 mm and the average annual temperature was 16℃. During the 2015 to 2016 model period, which was from 21 April, 2015, through 29 February, 2016, total rainfall was 1236 mm. Frequent rainfall occurred during November 2015 through January 2016, receiving 689 mm of precipitation over those months. In contrast, drought occurred during most of the 2016 to 2017 model period, with rainfall totaling 706 mm from 28 April, 2016, through 26 February, 2017. During the same November through January period in 2016 and 2017, there was only 311 mm of precipitation, leading to noticeable differences in water and NO_3-N loss through the soil profile. Figures 3.2 and 3.3 shows maximum temperature, minimum temperature, and precipitation and irrigation inputs during the two respective model periods.

3.2 Model parameters

Model inputs included soil water hydraulic properties, solute transformation, and solute transport parameters. Soil water retention parameters were optimized through auto-calibration in the top two soil horizons across all models using collected soil moisture data. Soil water retention parameters in the two remaining soil horizons were not changed from initial determination using the evaporation method. Final soil water parameters can be seen in Table 3.1. Optimized parameters are represented by the 0 to 18-cm and 18 to 36-cm depths in the table. Nitrogen transformation and transport parameters are shown in Table 3.2. The nitrification rate constant of 0.2 day^{-1} was found to provide an acceptable fit and was within ranges reported in the literature; 0.02 - 0.5 day^{-1} in Lotse et al. (1992), 0.226 - 0.432 day^{-1} in Iskandar and Selim (1981); 0.15 - 0.25 day^{-1} in Ling and El-Kadi (1998), and 0.2 day^{-1} in Hanson et al. (2006). Initial adsorption coefficients were based on values used in Bradshaw et al. (2013) due to similar soils and geographic location, but were optimized across models to provide a more acceptable fit. Water content dependence of nitrification and denitrification uses a modified version of the Walker (1974) equation and can be seen in Figure 3.4. Nitrification and denitrification rates were considered to be optimum at pressure heads of -500 cm and -20 cm, respectively.

3.3 Model calibration and validation

Water content data and NO_3-N concentrations in soil water from April 2015 through August 2015 were used to calibrate the models. The data from April 2016 through February 2017 was used to validate the models. During the calibration period, the RMSE of soil water contents ranged from 0.03 to 0.04 $cm^3\ cm^{-3}$ and *d* values ranged from 0.77 to 0.83. The RMSE of soil water NO_3-N concentrations ranged from 24 to 28 mg NO_3-N L^{-1} and *d* values ranged from 0.97 to 0.98. For validation models, the RMSE of soil water contents ranged from 0.03 to 0.04 cm^3 cm^{-3} and *d* values ranged from 0.85 to 0.89. The RMSE of soil water NO_3-N concentrations ranged from 30 to 37 mg NO_3-N L^{-1}, and *d* values ranged from 0.78 to 0.89.

Visual comparisons of observed to predicted data for the three models can be seen in Figures 3.5 - 3.7. The CC calibration model tended to under predict water content at the 15-cm depth but was more accurate at the 30-cm depth. Predicted water content was visually much more accurate at both depths in the CC validation model. In the LM calibration model, water content was generally under predicted at 15 cm and over predicted at 30 cm. The validation model tended to over predict water content at both depths, particularly in September through November 2016. The CR calibration model under predicted water content during most of the model period at the 15-cm depth and for a portion of the model period at the 30-cm depth. However, the CR validation model provided a good visual fit to data during the validation model period.

All calibration models somewhat over predicted NO_3-N concentration at the 0 to 15-cm depth, particularly at the beginning of the model period. The LM and CR calibration models had a tendency to under predict NO_3-N concentration during the middle part of the calibration period, but were closer to observed data during later measurements. At the same depth, the CC and CR validation models generally under predicted NO_3-N concentration at the beginning of the model period, but had a better visual fit during the second half of the model period. Both calibration and validation models in all treatments typically were near the mean of observed data at the 75-cm depth throughout both model periods, likely because NO_3-N concentrations at that depth did not typically deviate from the beginning to end of each monitoring period.

3.4 Soil water

Cumulative water drainage through the bottom of the profile a 1 m over the calibration model period for the CC, LM, and CR treatments was 62.7, 61.5, and 61.7 cm, respectively (Figure 3.8, Table 3.4). Little to no water moved below the bottom of the modeled soil profile (1 m) during the corn growing season (April through August, 2015), probably due to soil water depletion through evapotranspiration by the corn plant as well as less frequent precipitation during the summer months. From planting through harvest in August 2015, less than 5 cm of water had drained through the bottom of the soil profile for all treatments. The majority of water loss for all treatments occurred during the winter season from September 2105 through January 2016, which was roughly 55 cm for all treatments.

The validation model shows much less cumulative water loss through the bottom of the soil profile during the 2016 - 2017 model period (Figure 3.9). Cumulative water loss according to the validation model was 8.15, 6.32, and 6.40 cm for the CC, LM, and CR treatments, respectively (Table 3.4). Less than 1 cm of water was lost during the corn growing season for all treatments, probably due to high evapotranspiration and less precipitation during summer. The majority of water lost below 1 m during the model period occurred in late December 2016 through February 2017, similar to the calibration model period.

Large amounts of water movement observed in the 2015 - 2016 models are likely the result of an abnormally large amount of rainfall over the winter period and a well-drained soil that allows for relatively quick drainage. Cecil soils in this region are typically considered well-drained due to their sandy upper horizon and well-structured Bt subsurface horizons. Both Bruce et al. (1983) and Shoeneberger and Amoozegar (1990) have demonstrated relatively high saturated hydraulic conductivities in the Bt horizons of Cecil soils due to the presence of macropores and lack of shrink-swell clay minerals. Additionally, previous regular application of liming materials at the experimental site has been used to prevent soil acidification and promote aggregation, which generally leads to an improved soil structure (Haynes and Naidu, 1998). Frequent intense precipitation could very easily saturate the upper 1 m of the soil profile, and saturated flow would quickly provide water drainage below that zone. While large amounts of water loss were observed in the 2015 - 2016 model period, very little cumulative loss occurred during the 2016 - 2017 model period. Difference in water loss between the two periods is most likely due to differences in soil water storage at the end of the growing season as well as differences in precipitation over the two winter periods. Numerous studies report that precipitation is the driving factor for leaching in corn agrosystems, particularly unevenly distributed intensive rainfall events in which water is not available for uptake (Poch-Massegú et al., 2014; Jabloun et al., 2015; Masarik et al., 2014). If the soil did not reach saturation in the 2016 - 2017 as frequently as in the previous model period, it is unlikely that large drainage events would occur.

3.5 Crop N uptake and yield

In 2015, observed total mean uptake of N by corn for the CC, LM, and CR treatments was 255, 184, and 206 kg N ha^{-1}, respectively (Table 3.5). Analysis of variance (ANOVA) shows that there were no differences in N uptake between the treatments in 2015. In 2016, the CC and CR treatments were shown to have greater N uptake than the LM treatment with values of 196, 161, and 118 kg N ha^{-1}. Differences in observed N uptake are most likely due to the amount of supplemental N that was applied to each treatment. During the corn growing seasons of 2015 and 2016, roughly 133 kg N ha^{-1} and 202 kg N ha^{-1} was supplied by the clover and supplemental N fertilizer in the LM system, whereas 280 kg N ha^{-1} of granular N fertilizer was applied in the CR treatment. N additions from the clover could also be taken up by living clover in the LM treatment and unavailable to the corn, as has been noted in other living mulch systems (Affeldt et al., 2004; Kurtz et al., 1952; Zemenchik et al., 2000). N additions from the clover might also not mineralize in a timely manner if moisture conditions are not met, which would reduce N availability and corn N uptake (Stanford and Epstein, 1974; Paul et al., 2003).

In 2015, corn grain yield in the CC treatment in 2015 was 14.3 Mg ha^{-1}, which was greater than the LM treatment at 11.7 Mg ha^{-1} (Table 3.6). The CR treatment yielded 12.9 Mg ha^{-1}, which was not significantly different than either the CC or LM treatment. In 2016, the CC and CR treatments yielded 12.3 and 13.2 Mg grain ha^{-1}. Both were greater than the LM treatment, which yielded 9.20 Mg grain ha^{-1}. While the LM treatment had lower yields in both years compared to the CC treatment in 2015 and both treatments in 2016, it did not appear to differ much from the state average of 10.4 Mg ha^{-1} (National Agricultural Statistics Service, 2016). Reductions in grain yield in the LM treatment could be due to insufficient N at the beginning of kernel development, which has been shown to reduce corn grain yield. As mentioned before, insufficient N in the LM treatment could be due to competition from the clover or reduced mineralization of clover residue. Reduced N availability would also reduce vegetative plant growth, reducing plant growth potential and the ability to further accumulate N for kernel development (Uhart and Andrade, 1995).

3.6 NO_3-N leaching below the 1 m depth

Less than one kg NO_3-N ha^{-1} was leached below 1 m during the corn growing season in 2015 (April through August) for the three treatments according to the HYDRUS-1D calibration model (Figure 3.10). The first major loss of NO_3-N below 1 m occurred in November 2015, more than two months after corn harvest. The majority of NO_3-N loss below 1 m occurred in late December of 2015 through January of 2016 for all treatments, where differences in NO_3-N loss between treatments became more apparent. From 26 December, 2015, through 22 January, 2016, the CC treatment leached 19 kg NO_3-N ha^{-1}, the LM treatment 15 kg NO_3-N ha^{-1}, and the CR treatment 34 kg NO_3-N ha^{-1}. Cumulative NO_3-N loss below the 1-m depth for the CC, LM, and CR treatments was 47.8, 35.4, and 75.4 kg NO_3-N ha^{-1}, respectively (Table 3.4).

Less NO_3-N was leached below 1 m in the 2016 - 2017 validation model period than was in the calibration model period (Figure 3.11) for all treatments. Essentially no NO_3-N was lost in any treatment during the corn growing season of April through August despite high levels of N fertilization and intensive irrigation practices. In contrast with the calibration model period, less than one kg NO_3-N ha^{-1} was lost in all treatments in the validation model period, the majority of which occurred from late January through February 2016. Table 3.4 shows cumulative NO_3-N loss below 1 m for the CC, LM, and CR treatments, which had values of 0.43, 0.46, and 0.23 kg NO_3-N ha^{-1}.

While many studies have quantified the amount of N lost through NO_3-N leaching in corn, few to none have occurred in the southern Piedmont. Instead, more studies have focused on regions where greater corn production occurs. Leaching losses in this study are similar to values reported in other corn production systems, though a wide range of values have been reported based on differences in soil type, weather system, and crop management practices (Sogbedji et al., 2000). In the midwest, NO_3-N leaching in no-till corn has been shown to range from 41 to 57 kg N ha^{-1} (Brye et al., 2001; Kanwar et al., 1997; Randall and Iragavarapu, 1995). In the northeast, Gold et al. (1990) reported leaching losses of 41.8 and 79.3 kg NO_3-N ha^{-1} in corn that received 202 kg N ha^{-1} of urea while Jemison and Fox (1994) reported losses from 45.5 to 58.9 kg NO_3-N ha^{-1} in corn that received 200 kg N ha^{-1} of ammonium nitrate. In a similar soil, Hahne et al. (1977) reported an average loss of 100 kg NO_3-N ha^{-1} over a five year period in irrigated plots that received 280 kg N ha^{-1}.

In another living mulch experiment, Ochsner et al. (2010) found that a kura clover living mulch that received 90 kg N ha^{-1} or no fertilization reduced NO_3-N leaching by 31% and 74% relative to a N non-limited control. In two dormant seasons, the fertilized living mulch leached 14 and 36 kg NO_3-N ha^{-1}, comparable to the 35.4 kg NO_3-N in the calibration period of this experiment, while the unfertilized living mulch lost 8 and 7 kg NO_3-N in the two dormant periods. Yields in both the fertilized and unfertilized kura clover living mulches were slightly lower when compared to the N non-limited control, similar to the effect on yield in the LM treatment of this experiment.

Overall, NO_3-N losses through leaching in the calibration period of this study are similar to other reported values. NO_3-N leaching losses in the validation period are obviously much lower than other reported values as very little water leached below 1 m in this period, resulting in very little NO_3-N loss. In the 2015 - 2016 model period, NO_3-N leaching was lowest in the LM treatment, probably as a result of receiving less N than the CC and CR treatments. The LM treatment leached slightly more in the 2016 - 2017 model period, but loss below 1 m was below 1 kg NO_3-N ha^{-1} in all treatments due to reduced rainfall compared to the calibration model period.

4. Conclusion

A HYDRUS-1D model was used to simulate the flux of water and NO_3-N in three different corn production systems that were grown on a Cecil sandy loam. Treatments were a crimson clover cover crop that received 112 kg N ha^{-1}(CC), a white clover living mulch that received 56 kg N ha^{-1} (LM), and a cereal rye cover crop that received 280 kg N ha^{-1} (CR). In both the calibration and validation periods, RMSE values ranged from 0.03 to 0.04 cm cm^{-3} and *d* values ranged from 0.77 to 0.89 when compared to the average of observed water content data. Soil sampling and porous-cup samplers were used to monitor soil water NO_3-N concentration, and an N chain model was used to simulate the transformation of PMN and NH_4-N into NO_3-N and NO_3-N into N_2. The first-order nitrification rate constant used provided good fit between observed and predicted data and was within the range of values reported in other studies (Hanson et al., 2006; Iskandar and Selim, 1981; Ling and El-Kadi, 1998; Lotse et al., 1992). RMSE values ranged from 24 to 37 mg NO_3-N L^{-1} while *d* values ranged from 0.78 to 0.98 compared to average soil water NO_3-N concentration data.

During the 2015 - 2016 model period, all treatments leached over 60 cm of water below 1 m. Less than 10 cm of water was leached below 1 m in the 2016 - 2017 model period due to reduced rainfall. For NO_3-N, 47.8, 35.4, and 75.4 kg NO_3-N ha^{-1} was lost below 1 m in the CC, LM, and CR treatments, respectively. This amount is within the range of other studies examining NO_3-N leaching in corn (Brye et al., 2001; Gold et al., 1990; Hahne et al., 1977; Jemison and Fox, 1994; Kanwar et al., 1997; Randall and Iragayarapu, 1995). The amount of NO_3-N leaching was likely lower in the LM treatment due to reduced N application, even when considering N inputs from the clover during the growing season. NO_3-N loss was less than one kg NO_3-N ha^{-1} in all treatments in the 2016 - 2017 model period due to little water loss below 1

m. Corn grain yields were lower in the LM treatment in 2015 and 2016 due to competition from the clover and reductions in available N.

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Table 3.1: Soil properties (A) and hydraulic parameters (B) used in the HYDRUS-1D model.

A)

Soil layer (cm)	Texture	Particle Franction (%)			ρ_b (g cm^{-3})
		Sand	Silt	Clay	
0 - 18	Sandy loam	55	29	16	1.38
18 - 38	Clay loam	40	30	30	1.52
38 - 72	Clay	36	22	42	1.42
72 - 100	Clay	34	12	54	1.46

B)

Soil layer (cm)	θ_r (cm^3 cm^{-3})	θ_s (cm^3 cm^{-3})	α	n	l	K_s (cm d^{-1})
0 - 18	0.04	0.30	0.0062	1.28	0.5	61.08
18 - 38	0.03	0.27	0.0005	1.33	0.5	66.42
38 - 72	0.33	0.54	0.0535	1.40	0.5	89.52
72 - 100	0.11	0.46	0.0051	1.42	0.5	10.08

Table 3.2: Nitrogen transport and reaction parameters used in the HYDRUS-1D models.

Soil layer (cm)	K_d (cm^3 g^{-1}) NH$_4$-N + PMN	K_d (cm^3 g^{-1}) NO$_3$-N	Nitrification rate k (day^{-1})	Denitrification rate μ (day^{-1})
0-18	10.0	0.78	0.20	0.20
18-38	10.0	0.25	0.20	0.20
38-72	10.0	0.25	0.20	0.20
72-100	10.0	0.25	0.20	0.20

Table 3.3: Model performance statistics comparing predicted soil water content and soil NO_3-N concentration to observed data.

		Water Content		NO_3-N concentration	
Treatment	Period	RMSE (cm^3 cm^{-3})	*d*	RMSE (mg NO_3-N L^{-1})	*d*
CC	Calibration	0.03	0.83	28	0.97
	Validation	0.03	0.89	32	0.84
LM	Calibration	0.03	0.85	26	0.97
	Validation	0.04	0.84	37	0.78
CR	Calibration	0.04	0.77	24	0.98
	Validation	0.03	0.87	30	0.89

Table 3.4: Cumulative water and nitrate loss through the bottom of the model profile over both the 2015 - 2016 and 2016 - 2017 model periods.

Treatment	Period	Water leached (cm)	NO_3-N leached (kg ha^{-1})
CC	2015-2016	62.7	47.8
LM	2015-2016	61.5	35.4
CR	2015-2016	61.7	75.4
CC	2016-2017	8.15	0.43
LM	2016-2017	6.32	0.47
CR	2016-2017	6.40	0.23

Table 3.5: Comparison of total nitrogen uptake between the HYDRUS-1D models and the average of observed data.

Treatment	Year	Model Period	Predicted N uptake (kg N ha^{-1})	Observed mean N uptake (kg N ha^{-1})
CC	2015	2015-2016	267	255
LM	2015	2015-2016	191	184
CR	2015	2015-2016	214	206
			LSD	NS
CC	2016	2016-2017	195	196
LM	2016	2016-2017	109	118
CR	2016	2016-2017	171	161
			LSD	36.5

Table 3.6: Corn grain yields for the 2015 and 2016 growing seasons.

	Grain yield (Mg ha^{-1})	
Treatment	2015	2016
CC	14.3	12.3
LM	11.7	9.20
CR	12.9	13.2
LSD	2.19	1.19

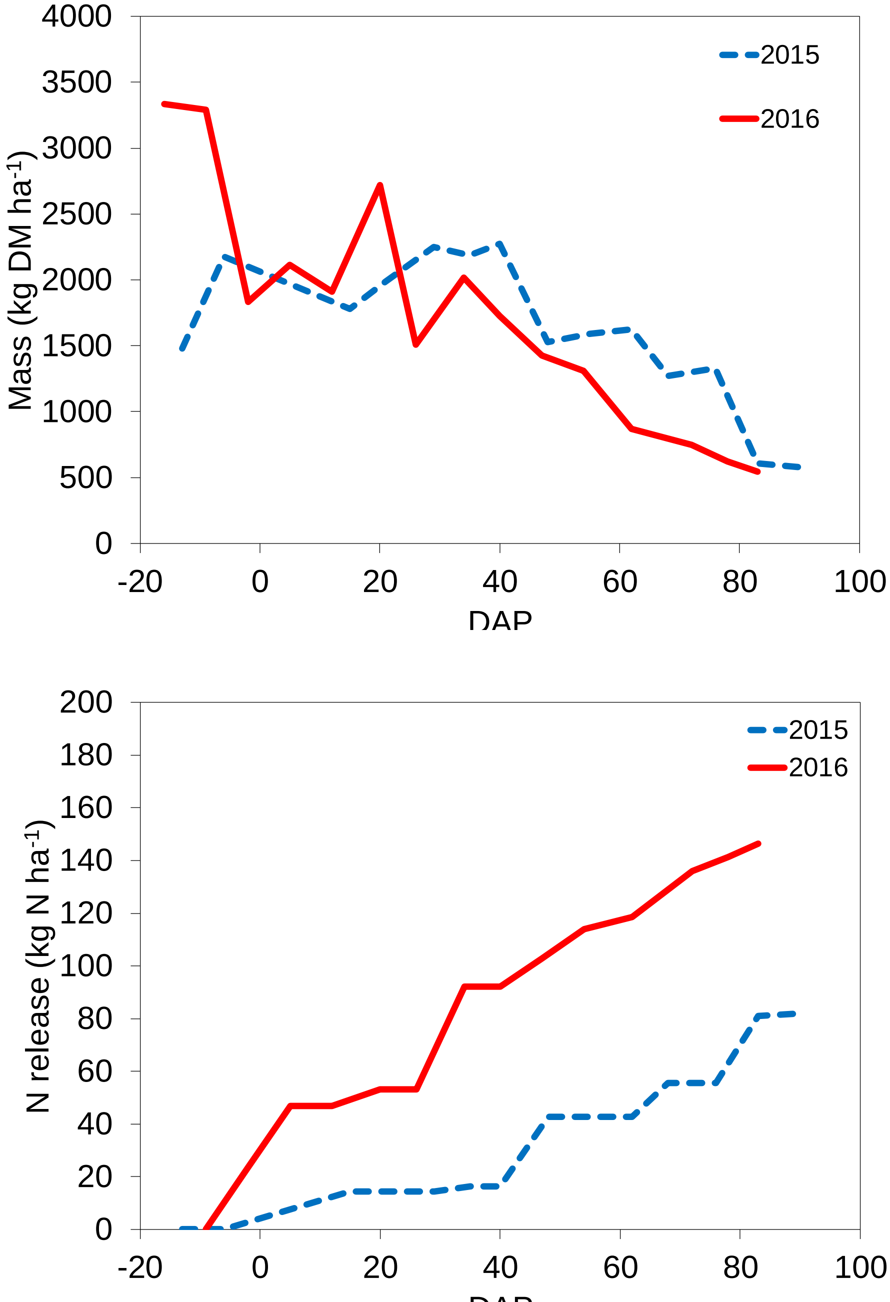

Figure 3.1: LM clover mass (top) and estimated N release (bottom) during the 2015 and 2016 growing seasons.

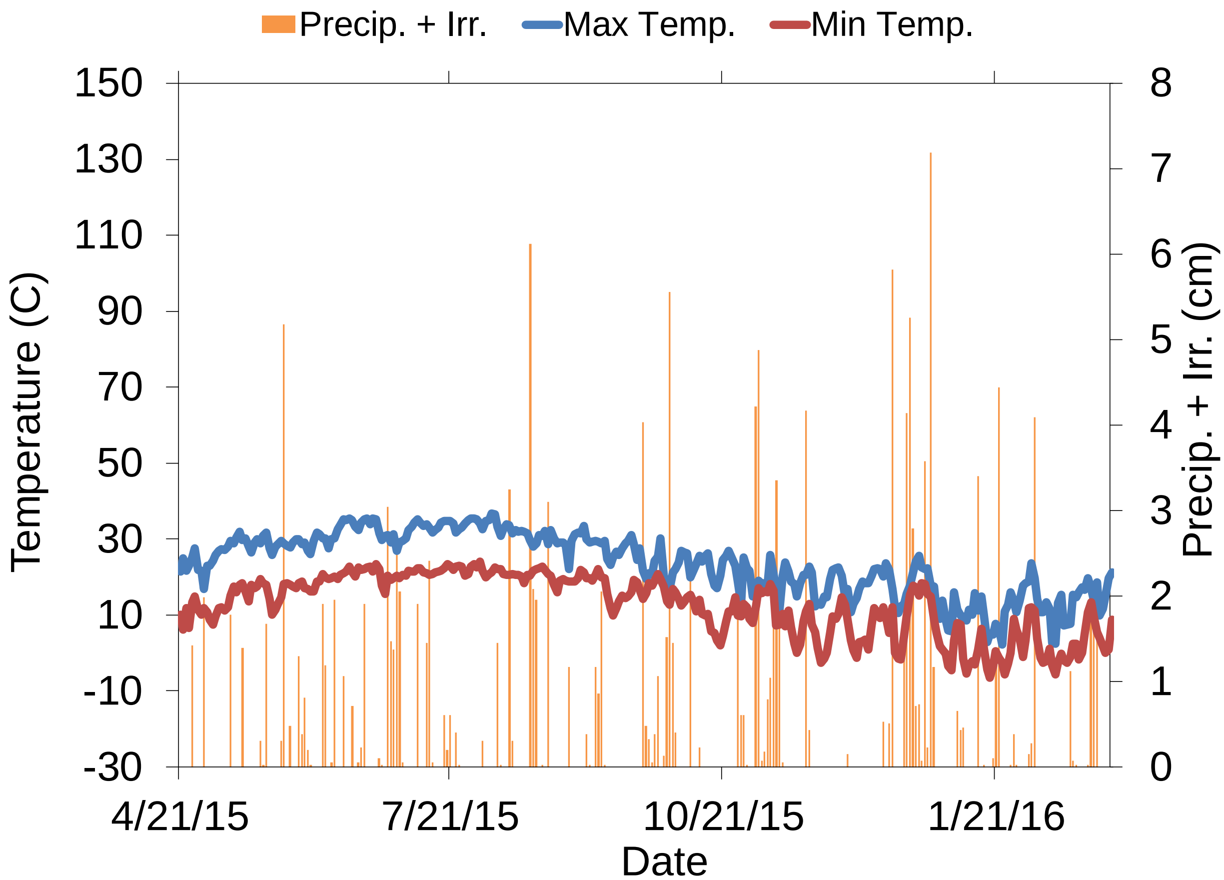

Figure 3.2: Maximum temperature, minimum temperature, and precipitation and irrigation inputs during the 2015-2016 model period.

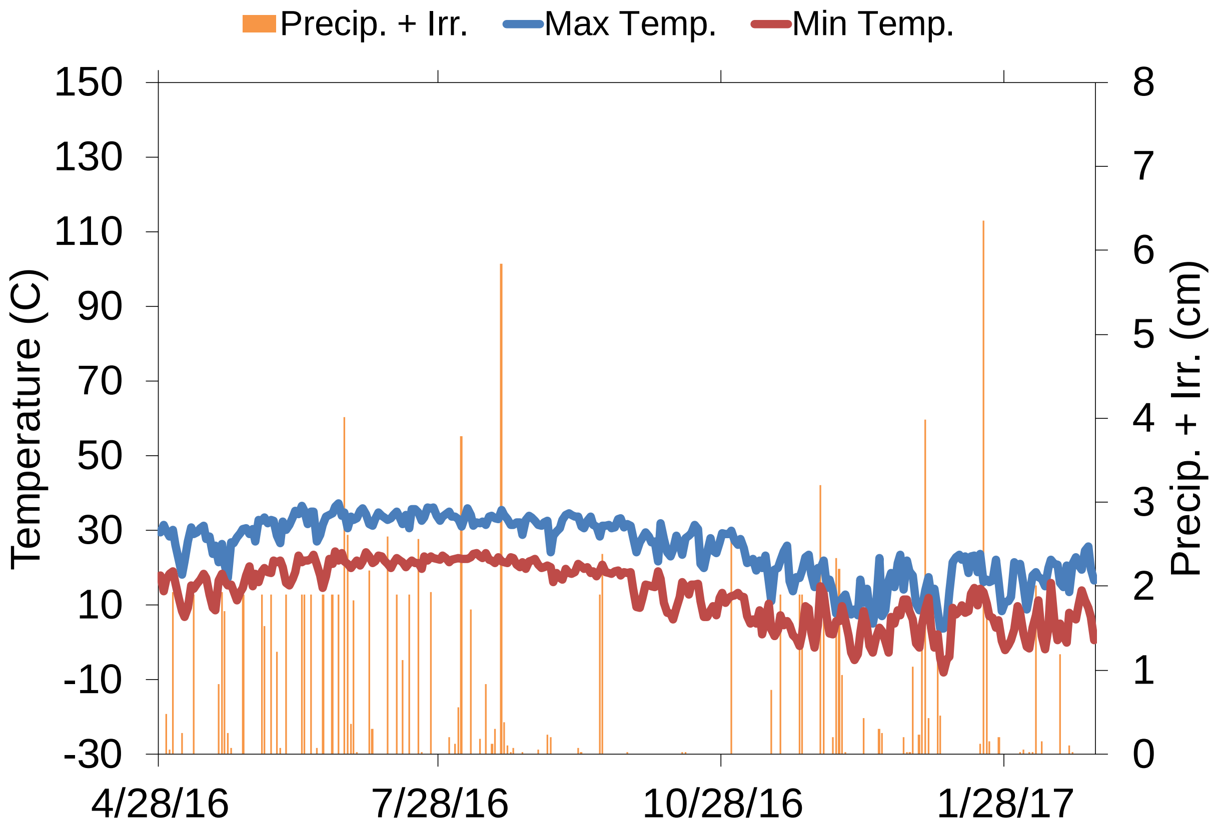

Figure 3.3: Maximum temperature, minimum temperature, and precipitation and irrigation inputs during the 2016-2017 model period.

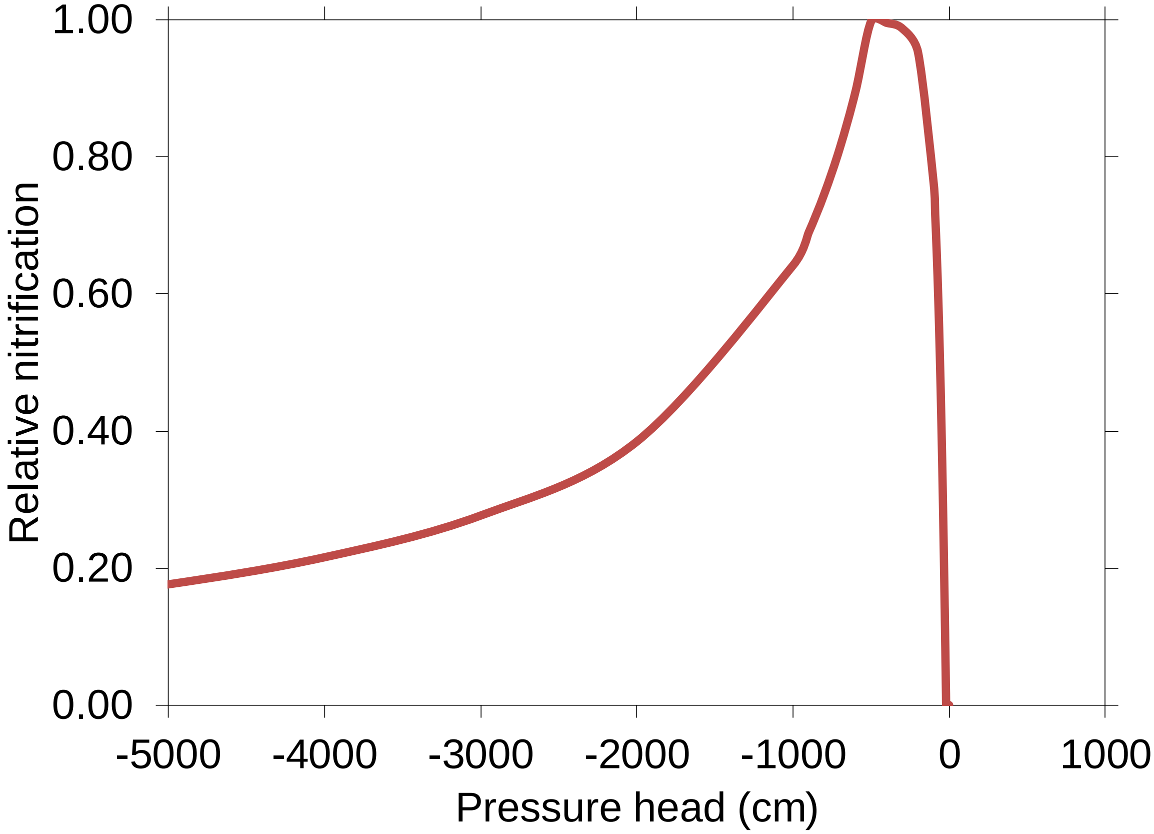

Figure 3.4: Nitrification and denitrification as a function of pressure head based on the Walker (1974) equation.

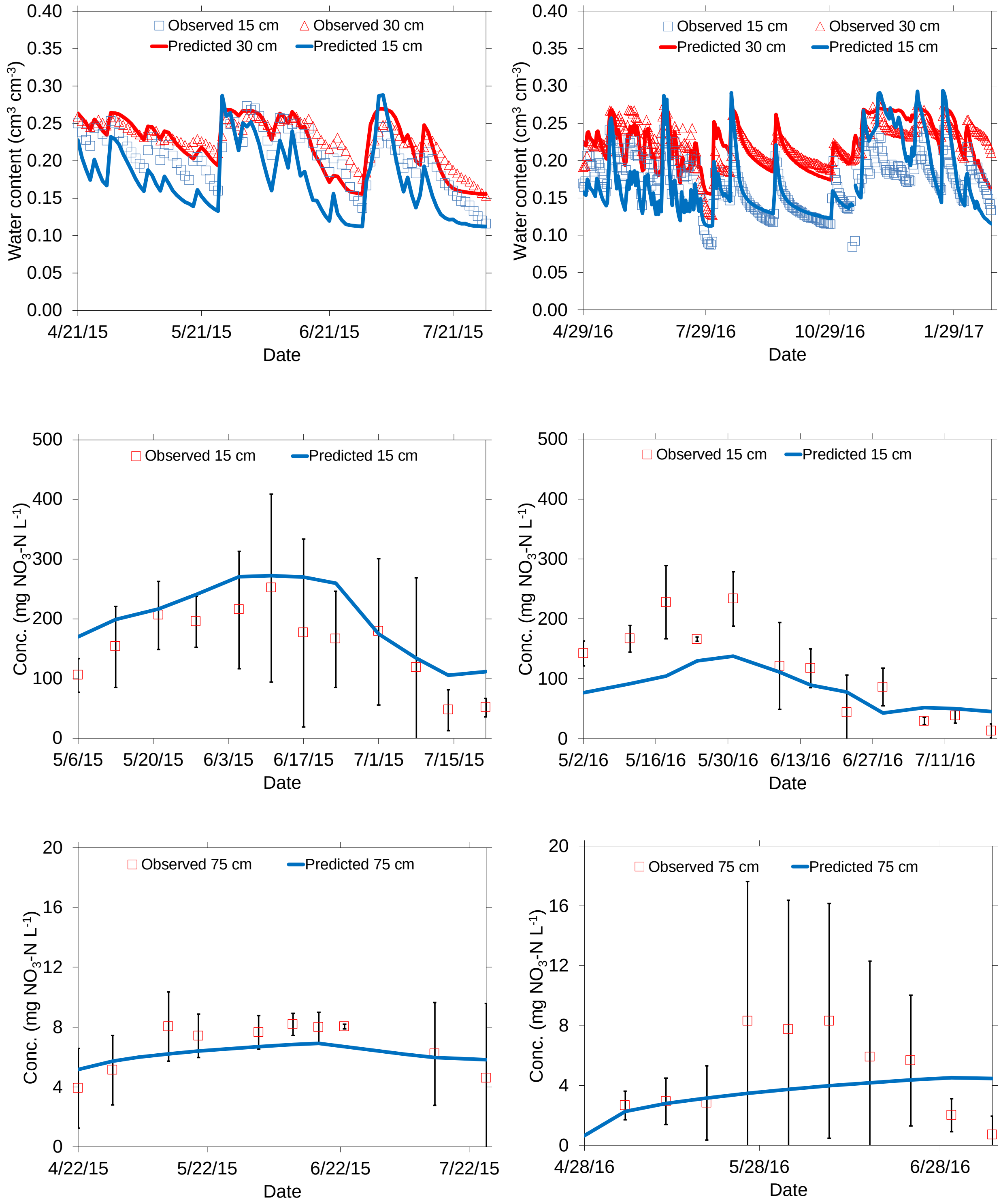

Figure 3.5: Observed and predicted data for the CC treatment for water content, soil water NO_3-N concentration at the 15-cm depth, and soil water NO_3-N concentration at the 75-cm depth. Calibration (2015 - 2016) data is in the left column, validation (2016 - 2017) data is in the right column. Error bars represent one standard deviation from the mean.

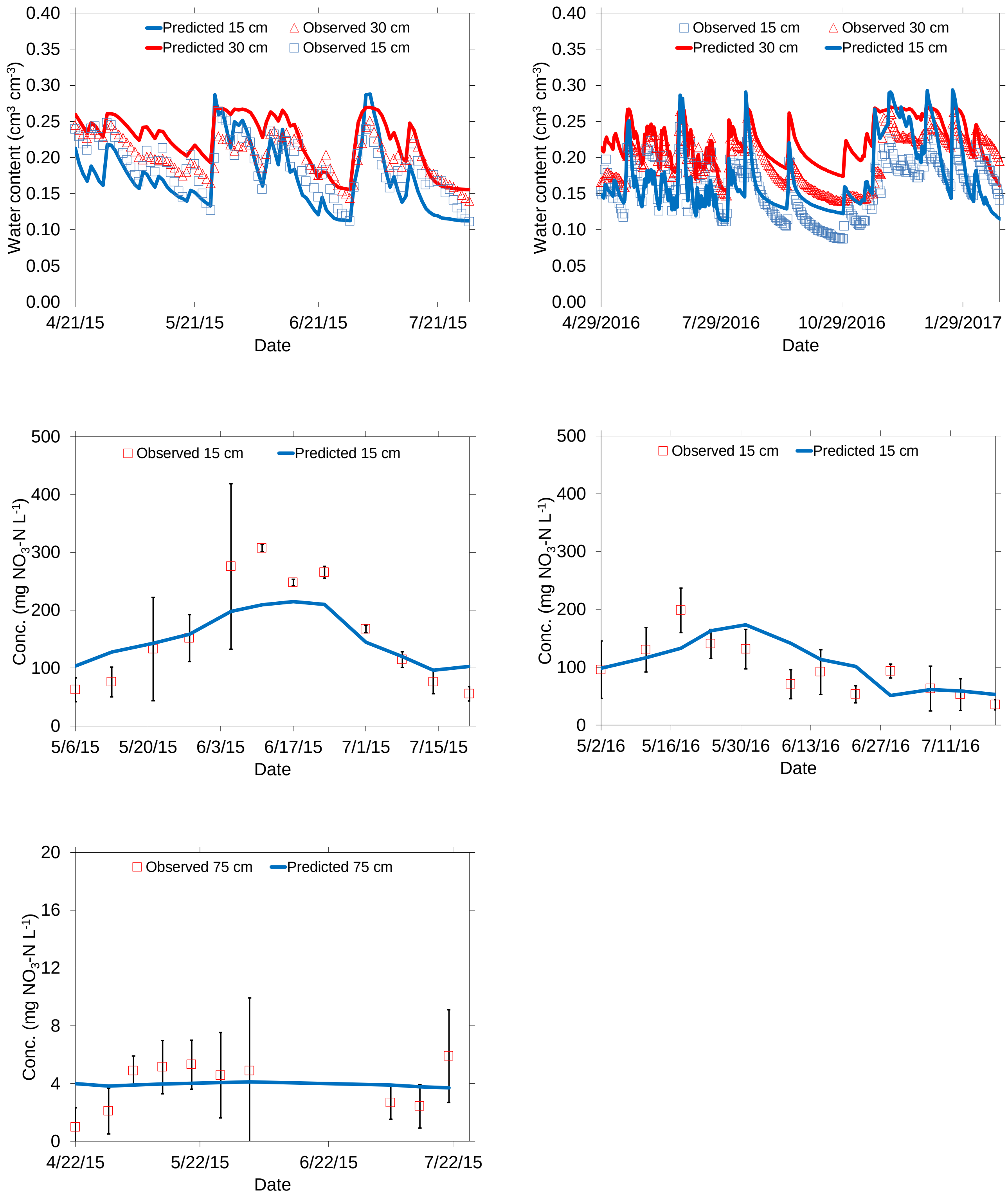

Figure 3.6: Observed and predicted data for the LM treatment for water content, soil water NO_3-N concentration at the 15-cm depth, and soil water NO_3-N concentration at the 75-cm depth. Calibration (2015 - 2016) data is in the left column, validation (2016 - 2017) data is in the right column. Error bars represent one standard deviation from the mean.

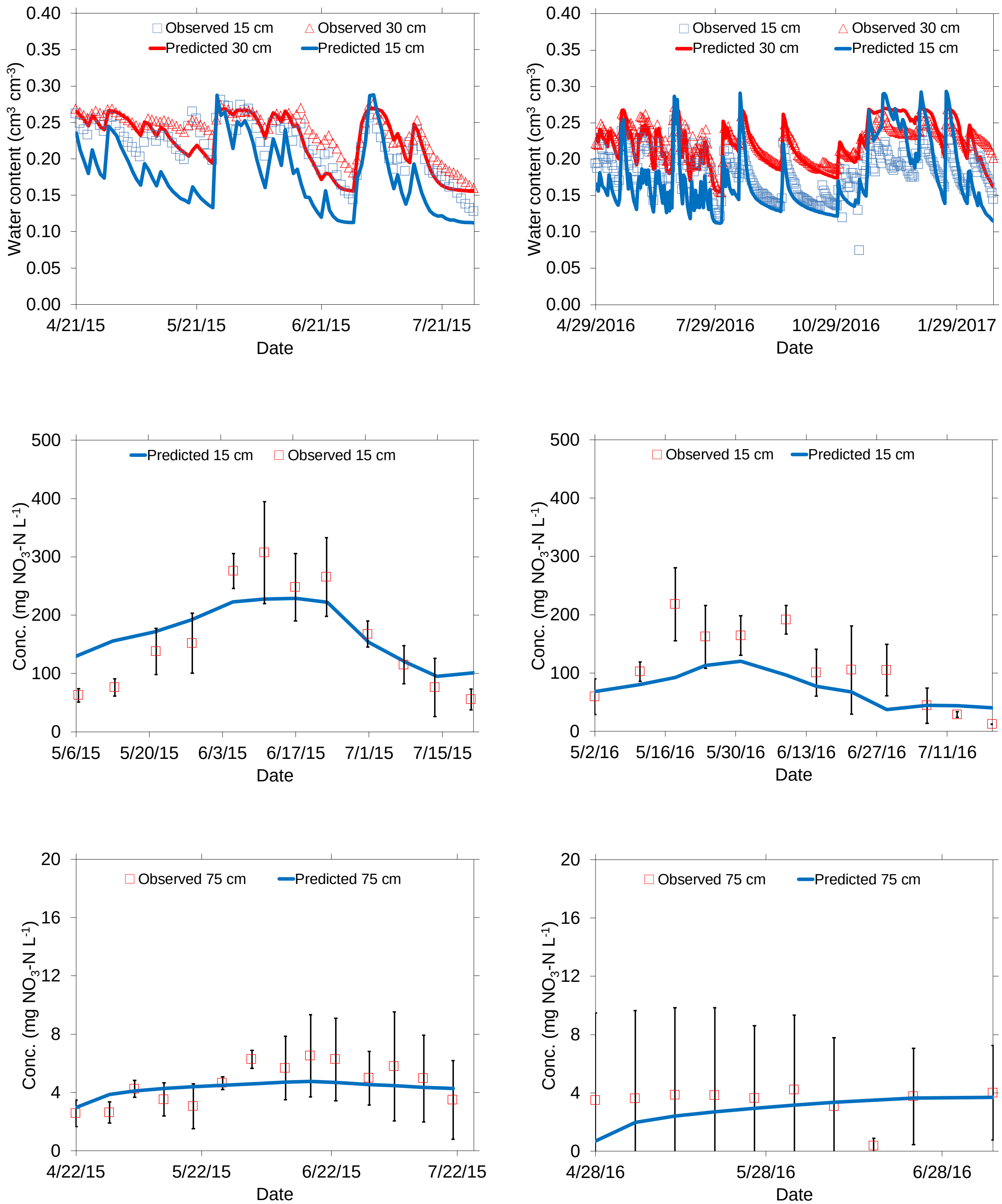

Figure 3.7: Observed and predicted data for the CR treatment for water content, soil water NO_3-N concentration at the 15-cm depth, and soil water NO_3-N concentration at the 75-cm depth. Calibration (2015 - 2016) data is in the left column, validation (2016 - 2017) data is in the right column. Error bars represent one standard deviation from the mean.

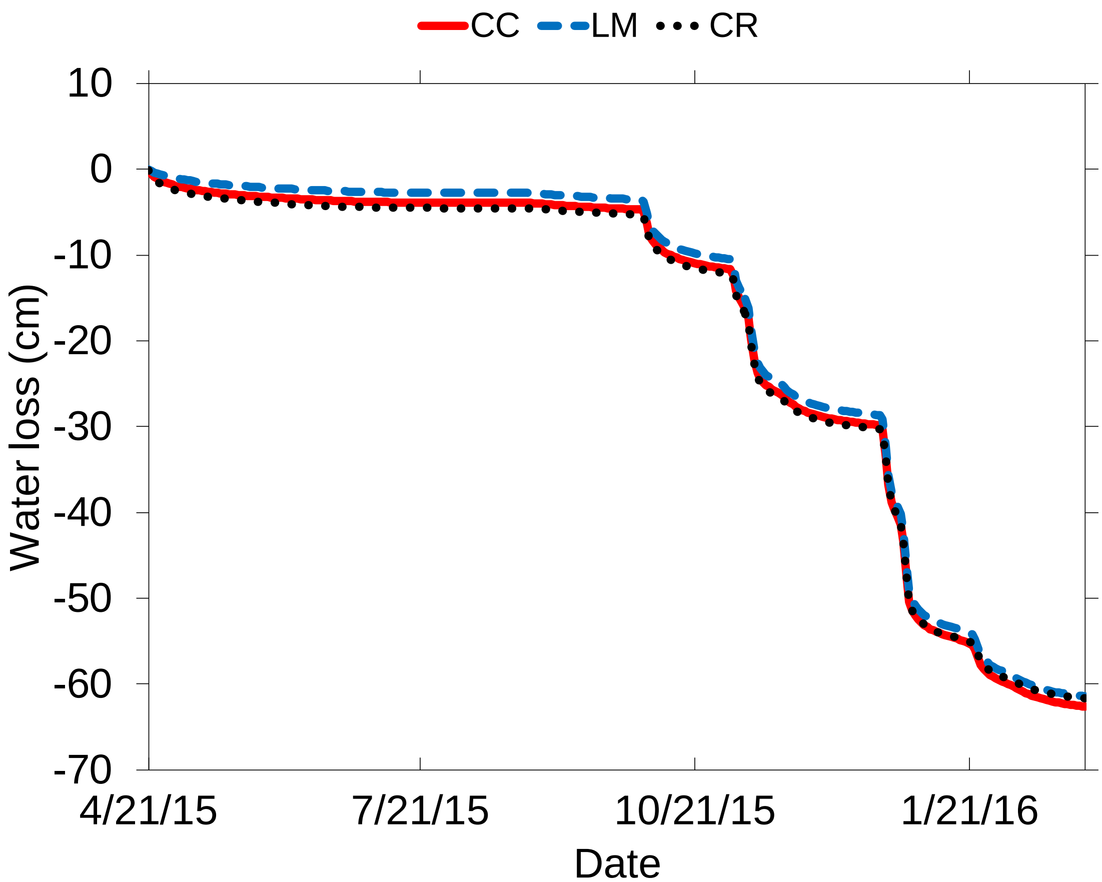

Figure 3.8: Cumulative water loss through the bottom of the soil profile (1 m) during the 2015 - 2016 model period.

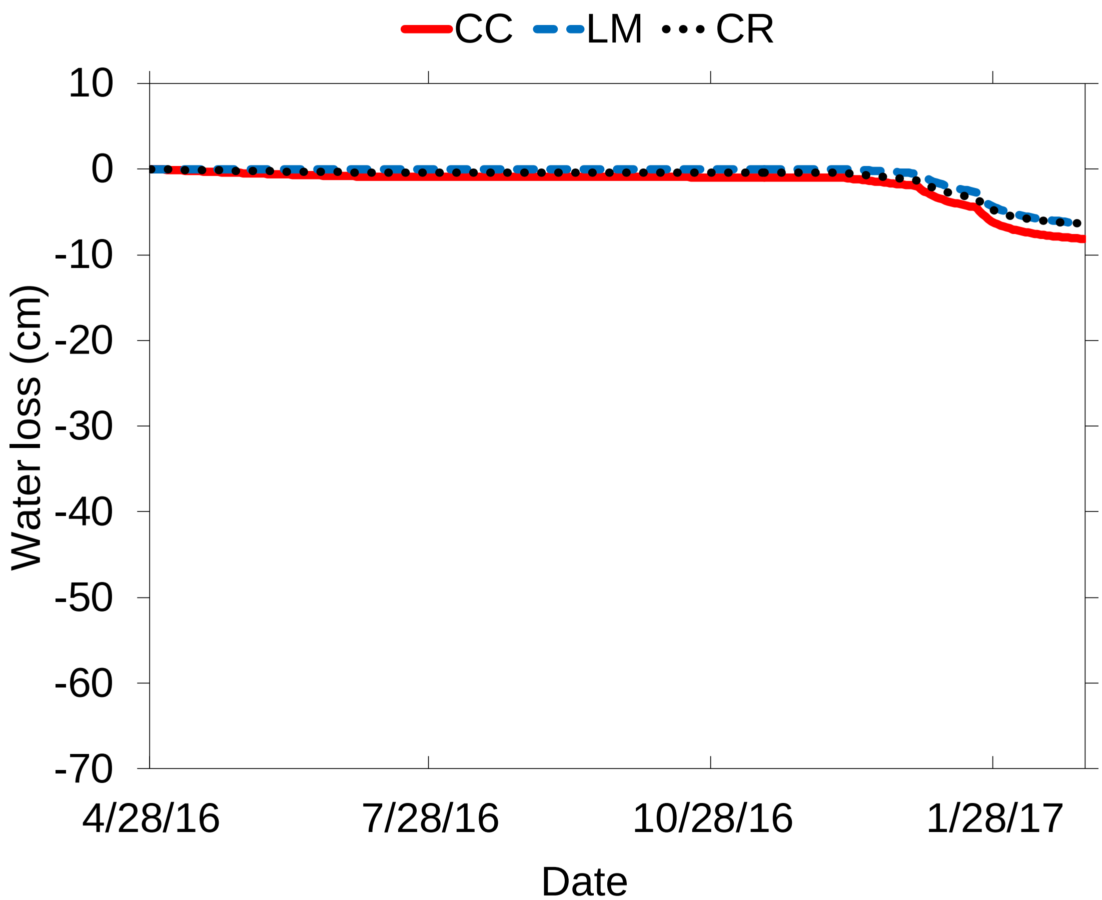

Figure 3.9: Cumulative water loss through the bottom of the soil profile (1 m) during the 2016 - 2017 model period.

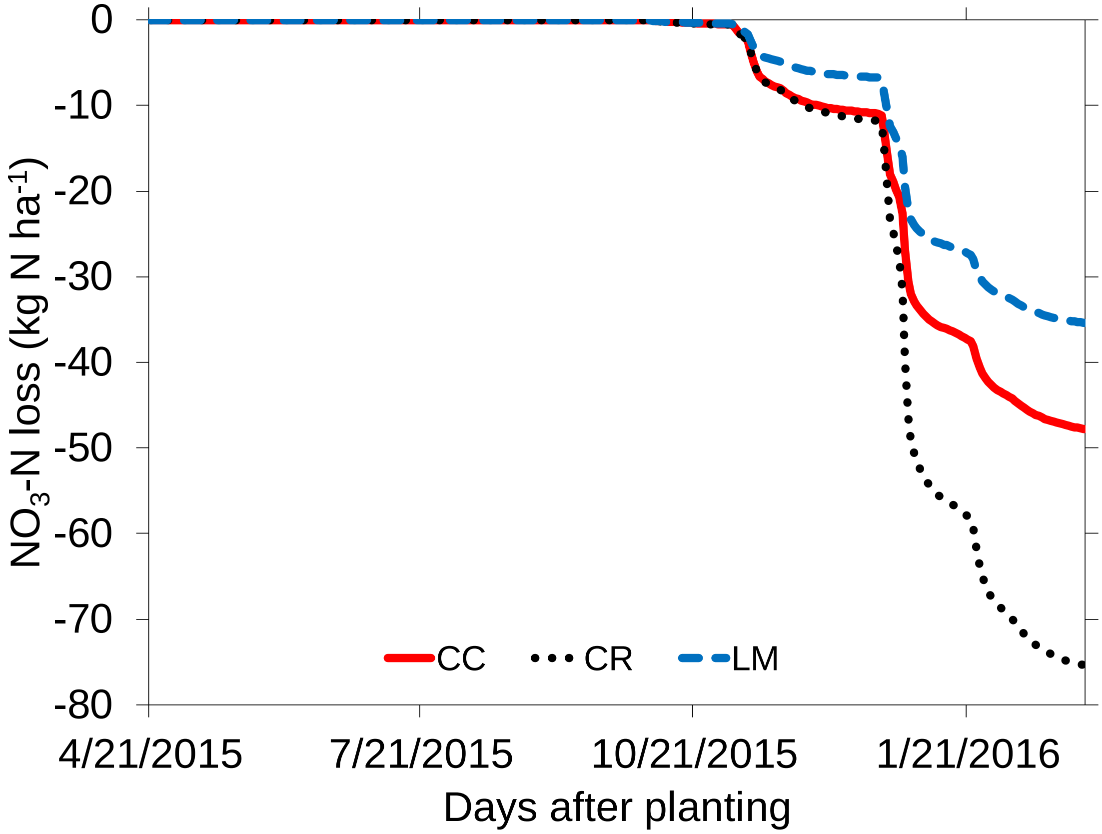

Figure 3.10: Cumulative NO_3-N leaching below the 1-m depth during the 2015-2016 model period.

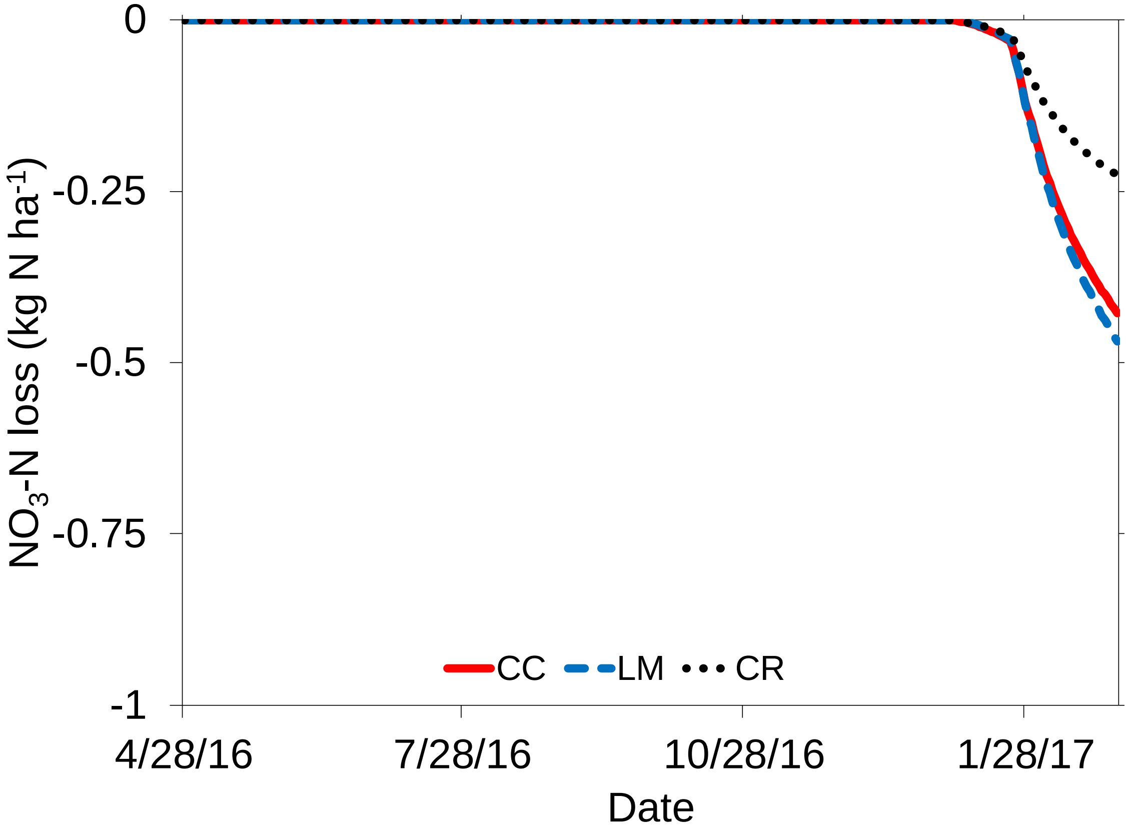

Figure 3.11: Cumulative NO_3-N leaching below the 1-m depth during the 2016-2017 model period.

CHAPTER 4

RUNOFF, SEDIMENT, NUTRIENT, AND E. COLI LOSSES FROM A LIVING MULCH AND A CEREAL RYE CORN PRODUCTION WATERSHED[4]

[4] J.S. Andrews, Z.S. Sanders, D.E. Radcliffe, N.S. Hill, M.L. Cabrera, U.K. Saha. To be submitted to *Journal of Plant Management*.

ABSTRACT

Corn production on agricultural watersheds is a major source of N, P, and sediment to susceptible water bodies. Cover crops are generally used in corn production to reduce the environmental effects of corn production and provide supplemental N for corn growth. The effects of a living mulch and no-till cereal rye corn production system on surface water quality was examined and compared to previous practices on experimental watersheds in Watkinsville, GA. From April 2015 through April 2017, runoff was monitored, collected, and analyzed for N, P, and sediment loss. Cattle were used to remove corn stover and *E. coli* in runoff was monitored after corn harvest in 2015 and 2016. By comparing data from this experiment to data from Schomberg et al. (2015), we were able to show that runoff was reduced by a living mulch cover crop. Both the living mulch and cereal rye systems were shown to reduce sediment load compared to conventional tillage cropping systems examined by Smith et al. (1978) due to cover cropping and reduced tillage practices. N and P loss was low on both watersheds over the course of the study period as well, and was within the range reported in similar studies and areas. *E. coli* in runoff was generally over the monthly mean established for recreational surface waters by the Georgia Environmental Protection Division, though no single sample limit for surface runoff has been established.

1. Introduction

Currently, corn is grown on more than 35 million hectares in the United States, and represents over 95 percent of total grain production nationwide. Demand for corn has grown in recent years due to energy policy mandates passed in 2005 and 2007 (Capehart, 2016). Like many other high-yielding grain crops, corn needs high levels of supplemental fertilizer, with insufficient available nitrogen being the primary limiting factor to corn growth and yield. In the southern Piedmont, most corn fertilization programs call for 200 to 250 kg ha^{-1} of supplemental nitrogen (N), generally in the form of ammonium or nitrate (Raun et al., 1999). While supplemental N fertilization can greatly increase grain yield, corn production areas are susceptible to non-point source losses of nutrient and sediment. It has been estimated that roughly 50% of impaired lake areas and 60% of impaired river reaches are the result of non-point source pollution from agriculture (Carpenter et al., 1998). Large quantities of dissolved nutrients in runoff, particularly N and phosphorus, can lead to eutrophication of waters downstream. Algal blooms routinely occur in eutrophied waters, and can cause anoxic zones which disrupt aquatic ecosystems. All water bodies can be affected; non-point source contributions from agricultural activities in the Chesapeake Bay watershed were determined to be the main contributor to the expansion of a large anoxic zone within the bay itself (Pionke et al., 2000).

The quantity and quality of surface runoff is dependent on different agricultural management practices, particularly the use of cover crops. Cover crops are typically used to reduce erosion by reducing detachment of soil particles during runoff events (Kaspar et al., 2001). In corn production systems, an annual Canada bluegrass cover and a crown vetch living mulch were shown to reduce erosion by 96 and 97%, respectively (Zhu et al., 1989; Hall et al., 1984). Cereal ryegrass has become a popular choice for corn producers to limit erosion as it establishes quickly, and has been shown to provide 30% ground cover after only one month of growth (Snapp et al., 2005). Cover cropping has also been shown to improve the physical properties of soil; reductions in bulk density and increases in saturated hydraulic conductivity and infiltration rate have been reported (Steele et al., 2012; Keisling et al., 1994). In both cases, improved soil physical properties led to reduced runoff due to increased infiltration and percolation of rainfall during high volume storm events. In addition to reducing erosion, legumes are frequently used as cover crops to biologically fix atmospheric N to provide to the following crop. A white clover cover crop can fix over 200 kg ha^{-1} of N when used as a winter annual (Erkovan et al., 2008), and using an annual clover cover crop has shown to be more profitable than typical N-fertilization programs for corn, with savings near 10 percent (Young-Mathews, 2013). A white clover living mulch aims to reduce erosion and provide supplemental N, but is not terminated prior to planting like traditional winter annual cover crops. Instead, it is grown as a perennial alongside a cash crop, eliminating the need for reestablishment during non-cropped periods. While the ability of a leguminous living mulch to provide supplemental N to a main crop is well documented (Echtenkamp et al., 1989; Hartwig et al., 2002; Duiker et al., 2004), the effect on runoff, erosion, and nutrient loss has not been quantified.

The objectives of this study were to (1) compare runoff volume and (2) sediment loss from adjacent corn production watersheds with a ‘Durana’ white clover living mulch and no-till cereal rye cover crop treatments, as well as (3) quantify nutrient and *E. coli* loss.

2. Materials and Methods

2.1 Study site

The study was performed from April 2015 through April 2017 on the P3 and P4 experimental watersheds (Figure 4.1) of the J. Phil Campbell Research and Education Center in Watkinsville, GA. The P3 and P4 watersheds are adjacent to each other and both have Cecil sandy loam (fine, kaolinitic, thermic typic kanhapludults) soils. Average annual rainfall and average annual temperature at the J. Phil Campbell Research and Education Center is 1,219 mm and 16℃, respectively. The P3 watershed is 1.26 hectares in area, while the P4 watershed is 1.40 hectares. P3 and P4 have approximately uniform slopes of 3 percent, and are both graded inwards towards central waterways with slopes of one to two percent (Smith et al., 1978). The waterways, while depicted in Figure 4.1 as grassed, were planted with the watersheds' respective cover crops and corn to be continuous with the rest of the watershed. At the outlet of each watershed, a Teledyne ISCO 4210 Ultrasonic Flow Meter (Lincoln, NE) was installed on 1.22-m H-flumes to monitor runoff volume and rate. Runoff samples were collected using a Teledyne ISCO 6712 Avalanche Portable Refrigerated Sampler. Samples were refrigerated in the field at 4℃ and frozen at -18℃ until preparation and analysis could be performed to determine N, phosphorus (P), and total suspended solids (TSS) concentrations.

The P3 and P4 watersheds have been used to monitor the effect of BMPs and other agricultural practices on runoff and water quality since the 1970's. Prior to 2012, research had been conducted on the watersheds by the Agricultural Research Service, a branch of the USDA, until it was turned over to the University of Georgia as part of the Consolidated and Further Continuing Appropriations Act of 2012. Notable research conducted on these sites include Mills et al. (1992), Smith et al. (1978), Jenkins et al. (2006), Endale et al. (2002), Franzluebbers et al. (2007), and Schomberg et al. (2014). Data collected on these watersheds in previous studies was used to compare the effect of the BMPs from the current study in a paired watershed (runoff) and before/after watershed (sediment) statistical design (Spooner et al., 1985).

2.2 Schedule of field operations

Prior to soil cover establishment in fall of 2014, both the P3 and P4 watersheds were tilled to a depth of 15 cm. P3 was seeded with 11.2 kg ha^{-1} 'Durana' white clover (*Trifolium repens* var. *Durana*) for the living mulch system (LM) on 20 October, 2014. P4 was planted with 28 kg ha^{-1} cereal rye (*Secale cereal* L.) (CR) on 1 November, 2014. In 2015, 20-cm bands of glyphosate (N-(phosphonomethyl)glycine) and dicamba (3,6-Dichloro-2-methoxybenzoic acid) at rates of 1.12 kg a.i. ha^{-1} and 1.20 kg a.i. ha^{-1}, respectively, were applied using a hooded sprayer on 90-cm rows on 1 April on the P3 watershed. Glyphosate (1.12 kg a.i. ha^{-1}) was broadcast applied on the P4 watershed on 21 March, 2015. Corn (*Zea mays*) (DeKalb DKC64-69, GENVT3P) was planted on 14 April, 2015, using a John Deere 7300 MaxEmerge no-till planter at 90,000 plants ha^{-1} on both the P3 and P4 watersheds. Pendimethalin (3,4-Dimethyl-2,6-dinitro-*N*-pentan-3-yl-aniline) and atrazine (1-Chloro-3-ethylamino-5-isopropylamino-2,4,6-triazine) were both applied one week later on both watersheds at 1.20 kg a.i. ha^{-1} and 1.12 kg a.i. ha^{-1}, respectively. The P4 watershed received a broadcast application of herbicide while the P3 watershed received a banded application. During both the 2015 and 2016 growing seasons, both watersheds were irrigated equally to maintain between 40 and 90% available water content (0.12 to 0.27 cm^3 cm^{-3}) as determine by Campbell Scientific CS625 Reflectometers (Logan, UT) placed in nearby research plots. Liquid urea N fertilizer was applied only on the P4 watershed. Sixty kg N ha^{-1} was applied at planting and 220 kg N ha^{-1} was applied at the V5 growth stage.

Both watersheds were later harvested on 14 September, 2015. Immediately following harvest, each watershed was sub-divided into 4 equally sized paddocks and 6 pregnant heifers (444 kg watershed^{-1}) were used to graze and remove corn stover and residue. The paddocks were grazed sequentially for seven days each, giving each watershed 28 total days of grazing. Runoff from both watersheds was monitored for *E. coli* from September 2015 through February 2016 and concentrations in runoff were measured using a commercial IDEXX Colilert kit (Atlanta, GA). *E. coli* in runoff was quantified using most probable number (MPN) methodology. The P4 watershed was replanted with cereal ryegrass at the same rate as 2014 on 20 October, 2015.

In 2016, the P3 watershed was broadcast treated with 1.20 kg a.i. ha^{-1} clethodim (*(E)-2-[1[((3-chloro-2-propenyl)-oxy)imino]propyl]-5-[2-(ethylthio)propyl]- 3-hydroxy-2-cyclohexen-1-one) on 17 March to control annual ryegrass. On 23 March, 2016, glyphosate was applied on the P4 watershed at the same rate as the previous year to terminate the cereal ryegrass cover. Glyphosate and dicamba were applied at the same rate and banding width as the previous year as well on the P3 watershed on 4 April and 20 April, 2016. Both watersheds were planted with corn on 28 April, 2016, at the same rate as 2015. Pendimethalin and atrazine were both applied one week later on both watersheds at 1.20 kg a.i. ha^{-1} and 1.12 kg a.i. ha^{-1}, respectively. The P4 watershed received a broadcast application of herbicide while the P3 watershed received a 20-cm banded application. Once again, N fertilizer was only applied to the P4 watershed. Sixty kg N ha^{-1} was applied at planting and 220 kg N ha^{-1} was applied at the V5 growth stage. Corn was harvested on both watersheds on 23 October 2016 and grazing began on 28 October. Total heifer weight on the P3 and P4 watersheds were 432 kg and 429 kg, respectively. Paddock grazing was the same in 2016; each paddock was grazed sequentially for 7 days each, giving a total of 28 days grazing. Runoff from both watersheds was monitored for *E. coli* from initial grazing in 2016 through February 2017 using the same methods as the previous year.

2.3 Sample collection and preparation

Samples collected from the P3 and P4 watersheds were analyzed for total N, dissolved organic N, soluble ammonium and nitrate, total P, dissolved orthophosphate, total dissolved P, and TSS. Runoff samples analyzed for total N concentration were prepared using an unfiltered Kjeldahl digestion (Bowman & Delfino, 1982). Samples analyzed for particulate N, dissolved organic N, and total dissolved P concentration were prepared through Kjeldahl digestion of runoff samples that had been filtered through a 0.45-µm filter. Soluble nitrate, soluble ammonium, and dissolved orthophosphate samples were prepared by filtering runoff through a 0.45-µm filter and leaving them undigested. All nutrient concentrations in prepared samples were measured at the UGA Agricultural and Environmental Services Laboratory in Athens, GA. N concentrations were determined using a Timberline TL-2800 Ammonia Analyzer (Boulder, CO) and the following equations:

[1] *TN (mg/L) =* NH_4*-N (mg/L) from unfiltered digestion +* NO_3*-N (mg/L) from filtered samples (mg/L)*

[2] *Particulate N (mg/L) =* NH_4*-N (mg/L) from unfiltered digestion -* NH_4*-N (mg/L) from filtered digestion*

[3] *Dissolved organic N (mg/L) =* NH_4*-N (mg/L) from filtered digestion -* NH_4*-N (mg/L) from filtered samples*

P concentrations for total P, dissolved orthophosphate, and total dissolved P were determined colorimetrically (King, 1932). TSS concentration was determined using EPA Method 160.2 (USEPA, 1999).

2.4 Previous treatments

In Schomberg et al. (2014), runoff data was collected from fall of 2005 through fall of 2009 in a no-till cereal rye and cotton system. Cereal rye was planted with a no-till grain drill at 125 kg ha^{-1} in the fall of each year while cotton (*Gossypium arboreum* L.) was planted every May on both the P3 and P4 watersheds. Application of herbicide, N fertilizer, lime, and harvesting always occurred on the P3 and P4 watersheds on the same day.

Unlike in Schomberg et al., the P3 and P4 watersheds did not receive the same treatment or management practices during the study by Smith et al. (1978). Operations occurred on both watersheds from June 1972 through November 1975. The P3 watershed was tilled to a 20-cm depth before initial planting with a moldboard plow and chiseled to the same depth every year to eliminate hardpans or crusted layers. From June 1972 through October 1974, Soybean (*Glycine max* L.) was planted in June of each year at 431,000 plants ha^{-1} and cereal rye was planted in late September/early October of every year at 4.98×10^5 seeds ha^{-1} (SB). Barley (*Hordeum vulgare* L.) was planted on the P3 watershed from October 1974 through May 1975 at 3.23×10^6 seed ha^{-1} (BA). On the P4 watershed, corn and cereal rye were planted from September 1972 through November 1975. Rye was planted every September/October at 4.98×10^5 seeds ha^{-1} and corn was planted every April/May at 53,600 plants ha^{-1} (CO). Like the P3 watershed, the P4 watershed was tilled initially to a 20-cm depth and chiseled to the same depth every year.

2.5 Statistical analysis

Analysis of covariance (ANCOVA) was used to compare runoff and sediment loss from the current treatment to previous treatments on the P3 and P4 watersheds. Runoff was compared to data from Schomberg et al. (2014) when the treatments on both watersheds were no-till with a cereal rye cover using a paired watershed design (Spooner et al., 1985). Since treatments were different on the P3 and P4 watersheds from Schomberg et al., ANCOVA was used to indicate that a change in the runoff relationship between P3 and P4 had occurred and the reasoning for any change is discussed. ANCOVA was also used to compare sediment losses from the current treatments to treatments used in Smith et al. (1978) on the same watershed as part of a before/after watershed design that uses runoff as the explanatory variable (Spooner et al., 1985). Data transformations were used for both runoff and sediment data to produce a normal distribution of model residuals using the Shapiro-Wilks (W) statistic, quartile-quantile plots, and residual plots (Kéry and Hatfield, 2003; Rutherford, 2001). Analysis of variance was used to check for model significance (Clausen et al., 1996). Statistical analysis of the data was performed using JMP version 13 software (SAS Institute, Cary, NC).

3. Results and discussion

3.1 Runoff

Based on ANOVA, the regression of P3 and P4 runoff was significant during the current treatments ($p < 0.0001$) and previous treatment in Schomberg et al. (2014) ($p < 0.0001$) (Figure 4.2). Analysis of covariance showed that the y-intercept of the 2005 - 2010 regression line was significantly higher than the y-intercept of the 2015 - 2017 regression line (Table 4.1). No difference was observed in the slopes of the two regression lines. This indicates that under the same treatments in the first study, the P3 watershed innately produced more runoff, probably due to small differences in soil, despite being the same soil series. In our study, the difference in runoff between P3 and P4 was less. Since the LM watershed was the response variable in the regression that had a significantly lower y-intercept, it is assumed that a runoff reduction was due to implementation of this treatment but cannot be confirmed with the current methodology. A t-test showed that percent runoff from the P3 watershed was reduced from the previous treatment to the current treatment ($p = 0.0291$) while no difference was seen on the P4 watershed ($p = 0.3832$), furthering the argument that the LM treatment caused a reduction in runoff on its respective watershed.

Other studies have shown that a living mulch can soil water content and runoff. For the effect on soil water content, Sanders et al. (unpublished), Liedgens et al. (2004), and Ochsner et al. (2010) all noted that use of a living mulch reduced soil water content during the corn growing season due to uptake by the intercropped mulch. A living mulch has also been shown to compete with the corn crop for water and other resources (Affeldt et al., 2004; Kurtz et al., 1952). Reducing antecedent soil water content increases the amount of available soil water storage and could reduce runoff during storm events.

Literature discussing the effect of a living mulch on runoff volumes is somewhat sparse, but a few studies have been reported. In a rainfall simulator study, a corn and red clover living mulch reduced mean runoff relative to a corn monoculture (Wall et al., 1991) with reductions ranging from 21 to 100%. Hall et al. (1984) saw similar results: runoff was reduced in two living mulch corn production systems relative to a corn monoculture. A living mulch of birdsfoot trefoil (*Lotus corniculatus*) reduced runoff by 93 to 96% while a crownvetch (*Coroniila varia* L.) living mulch reduced runoff by 93 to 94%.

3.2 Sediment loss

The relationship between sediment loss and runoff during the current and previous studies is shown in Figure 4.3 for the P3 watershed and in Figure 4.4 for the P4 watershed. Based on ANOVA, lines of regression in the P3 watershed were significant for the LM treatment ($p < 0.0001$) and for the SB ($p < 0.0001$) and CB ($p < 0.0001$) treatment from Smith et al. (1978). ANCOVA showed that the y-intercept of the LM regression was significantly lower than the y-intercepts of both the SB and CB regression lines, indicating a reduction of sediment loss per volume of runoff (Table 4.1). Overall reduction in sediment loss by the LM treatment compared to the SB and BA treatments was determined to be 99% based on comparisons of y-intercepts (Grabow et al., 1998).

Lines of regression were significant for both the CR ($p < 0.0001$) and CO treatments ($p < 0.0001$) in the P4 watershed. The y-intercept for the CR regression line was significantly lower than the y-intercept of the PC regression line. The slope of the CO regression line, however, was significantly higher than the slope of the CR regression line, indicating a proportionally greater increase in sediment loss as runoff volumes increase. Overall sediment reduction by the CR treatment over the CO treatment was determined to be 97%.

Living mulches have been shown to reduce sediment load over conventional agricultural practices. The crownvetch and birdsfoot trefoil living mulch in Hall et al. (1984) reduced sediment loss by roughly 100% compared to a conventionally tilled corn system. Wall et al. (1991) also saw large reductions in sediment loss. In a rainfall simulator study, sediment loss reductions ranged from 46 to 78% in corn and intercropped clover over a corn monoculture system. While sediment losses in living mulches may not be well-documented, the use of cover crops for reducing erosion is well known, and is considered a best management practice for reducing erosion (Dabney et al., 2001). For example, Langdale et al. (1991) reported that cover crops reduced erosion by up to 62% on southeastern Ultisols.

However, all reductions in sediment loss cannot be entirely attributed to the LM treatment as tillage practice also changed from the comparison periods, as evidenced by the change in sediment loss on the P4 watershed. Therefore, reductions in sediment loss on the LM and CR watersheds should be viewed as due to conservation practices: cover cropping and no-tilllage. The LM and CR treatments provided 99 and 97% reductions in sediment loss compared to previous treatments, similar to near-100% reductions seen in other studies (Blevins et al., 1990; Chichester and Richardson, 1992; Clausen et al., 1996). Reductions in sediment loss is of particular interest to those in the southern Piedmont; greater than 40% of the southeastern Piedmont region is considered to be eroded, much of which is attributed to intensive tillage practices during the Cotton Era (Langdale et al., 1992). While sediment loss from the LM and CR treatment cannot be tested statistically, the total loss over the study period was 12.90 and 49.30 kg ha^{-1} for LM and CR, respectively. Lower sediment loss in the LM watershed could be due to an increased leaf area index that better intercepts raindrops, a root system that provides more soil stability, or even provides soil cover during susceptible periods due to its perennial growth habit.

3.3 Nutrient and E. coli loss

Losses of N and P species did not differ by more than 0.5 kg ha^{-1} between the LM and CR watersheds over the monitoring period, and less than 3 kg ha^{-1} of total N or total P was lost by either watershed (Figure 4.5, Table 4.2). Due to changes in watershed treatments and lack of available data, nutrient loss between the treatments could not be tested statistically. However, it likely that use these cropping systems reduced nutrient loss in runoff, particularly total N, compared to conventional practices. Jordan et al. (1997) estimated that N loss on Piedmont watersheds in the Chesapeake Bay area was between 29 and 42 kg N ha^{-1} $year^{-1}$, leading to excessive algal growth within the bay itself. The LM and CR watersheds, however, lost less than 3 kg N ha^{-1} over the study period, and less than one kg N ha^{-1} was in an inorganic, soluble form. N losses from these watersheds were more comparable to Langdale et al. (1979), which took place on the P2 and P4 watersheds. On these watersheds, an average annual loss of 9.5 kg N ha^{-1} in runoff from corn watersheds was observed, with annual NO_3-N and NH_4-N losses 1.19 and 2.24 kg ha^{-1}, respectively.

P losses from the LM and CR watersheds are also comparable to data from the southern Piedmont. On the P2 and P4 watersheds, Langdale et al. (1985) reported 0.1 to 4.0 kg P ha^{-1} $year^{-1}$ losses over a 9 year period, depending on tillage practices. Ranges of P loss from other areas in no-till management have also been reported: 0.2 to 1.3 kg P ha^{-1} $year^{-1}$ in the Southern Plains (Sharpley, 1995), 7.7 to 25 kg P ha^{-1} $year^{-1}$ in the Chesapeake Bay (Angle et al., 1984), and 1.24 to 1.65 kg P ha^{-1} $year^{-1}$ in the Great Lakes watershed (Gaynor and Findlay, 1995). Like Langdale et al. (1985), these studies report that total P loss is mainly a function of tillage practice, and total P loads can be reduced if conservation tillage practices are utilized.

For recreational water in Georgia, the limit for *E. coli* is 126 CFU 0.1 L^{-1} as the monthly geometric mean of at least four samples taken one week apart. 90% of all samples taken also must be below 410 CFU 0.1 L^{-1}. However, no single sample limit has been established in the state, as the use of *E. coli* as indicator organisms for pathogenic fecal bacteria is still currently being tested (GA EDP, 2016). In 2015 through 2016, after the first grazing period of the study, *E. coli* exceeded the 90% threshold concentration as runoff for roughly four months (Figure 4.6). The threshold was violated less frequently after the 2016 grazing period due to drought conditions and a reduced number of runoff events from the previous monitoring period. Manure-borne pathogens, such as *Salmonella* and *E. coli* 0157:H7 both have low infective doses of 100 and 10 cells, respectively, so management practices to reduce bacterial loads in runoff have been tested (Jenkins et al., 2015). For example, Coyne et al. (1995) found that a grass filter strip was able to trap 74% of fecal coliforms in runoff. Overall, however, the availability of pathogens for transport in runoff during storm evens is mainly influenced by the die-off rate of the bacteria, which is dependent on many factors, such as moisture, soil type, pH, temperature, and nutrient availability (Jamieson et al., 2002). Therefore it is difficult to compare *E. coli* losses to other studies and may rely on a paired approach to compare bacterial losses in the LM and CR systems.

4. Conclusion

A living mulch and no-till cereal rye corn production system was established on two historically studied watersheds in the southern Piedmont. Using a paired watershed design, our study showed that the LM system reduced runoff in our study when compared to data from Schomberg et al. (2014). Both the LM and CR treatments reduced sediment loss on their respective watersheds over treatments used in Smith et al. (1978) due to use of two important conservation practices: cover cropping and reduced tillage. Loss of N and P from the LM and CR watershed were low and were within ranges reported in the literature. *E. coli* load in runoff was measured, but is hard to compare to other data due to the many factors affecting bacterial survival. To better analyze the effect of a living mulch on runoff, sediment, nutrient, and *E. coli* loss, a calibration period could occur as part of a paired watershed study, considering that paired data collected during this study period would be considered part of a treatment period.

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Table 4.1: Results of statistical analysis comparing the LM and CR treatments to previous treatments in Schomberg et al. (2014) and Smith et al. (1978).

Constituent	Design	Statistical Significance		Associated Figure
		Slope	Intercept	
Runoff	Paired watershed	NS	0.0033	4.2
Sediment P3	Before/after - SB	NS	0.0001	4.6
Sediment P3	Before/after - BA	NS	0.0001	4.6
Sediment P4	Before after - CO	0.0425	0.0001	4.8

Table 4.2: Cumulative mass of nutrient and TSS loss on the LM and CR watersheds over the study period.

Mass lost in runoff (kg ha^{-1})	LM	CR
TN	2.670	2.380
TP	2.040	1.800
Particulate N	0.949	1.060
Total dissolved P	1.310	1.670
Dissolved organic N	1.240	0.742
Dissolved orthophosphate	0.711	0.521
NO_3-N	0.238	0.313
NH_4-N	0.281	0.279
TSS	12.90	49.30

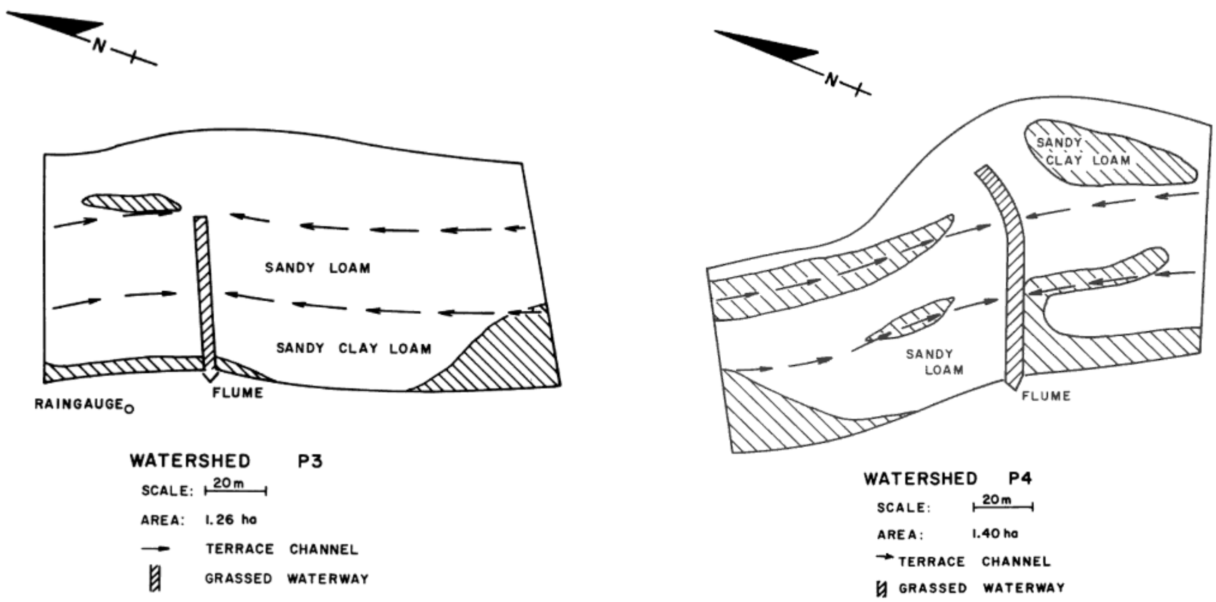

Figure 4.1: Depictions of the P3 and P4 watersheds from Smith et al., (1978).

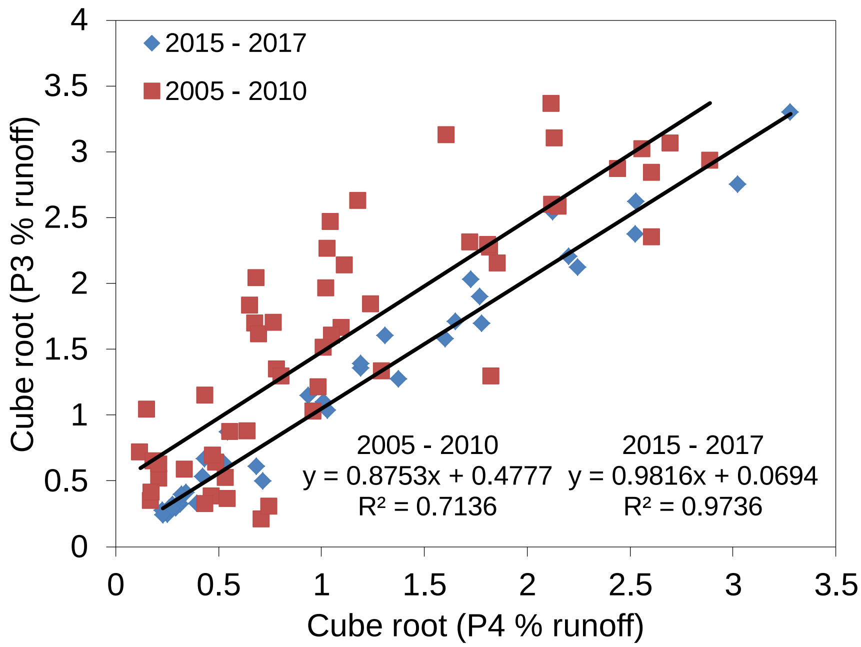

Figure 4.2: Regression of runoff values from the P3 and P4 watershed during the 2015 - 2017 and 2005 - 2010 treatment periods. In the 2015 - 2017 period, the P3 treatment was living mulch and the P4 treatment was cereal rye. In the 2005 - 2010 period, the treatment for both P3 and P4 was cotton and cereal rye.

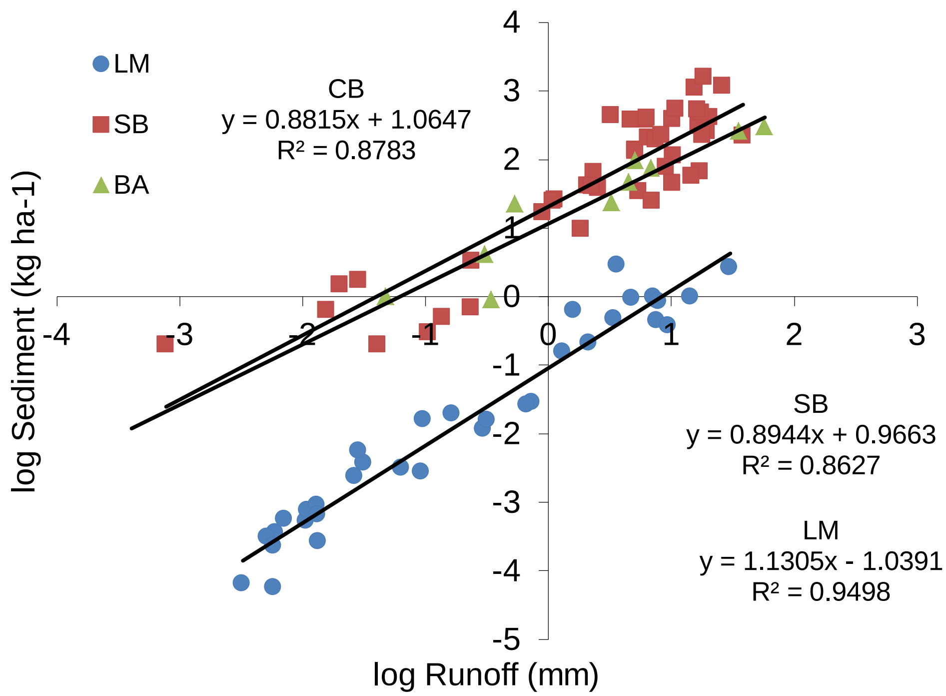

Figure 4.3: Regression of sediment losses on the P3 watershed from the SB, BA, and LM treatments.

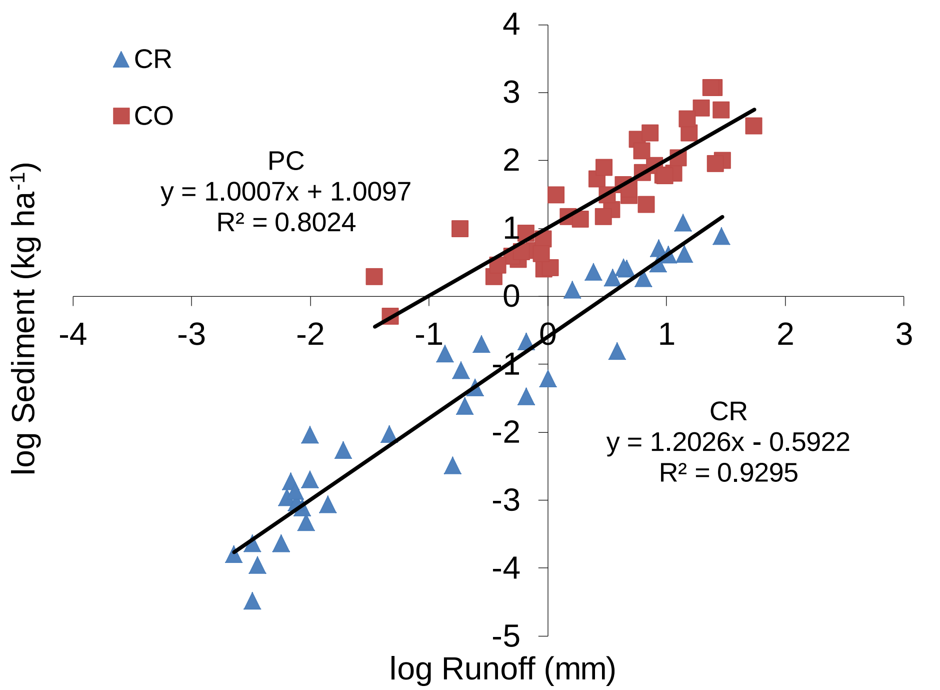

Figure 4.4: Regression of sediment losses on the P4 watershed from the CR and CO treatments.

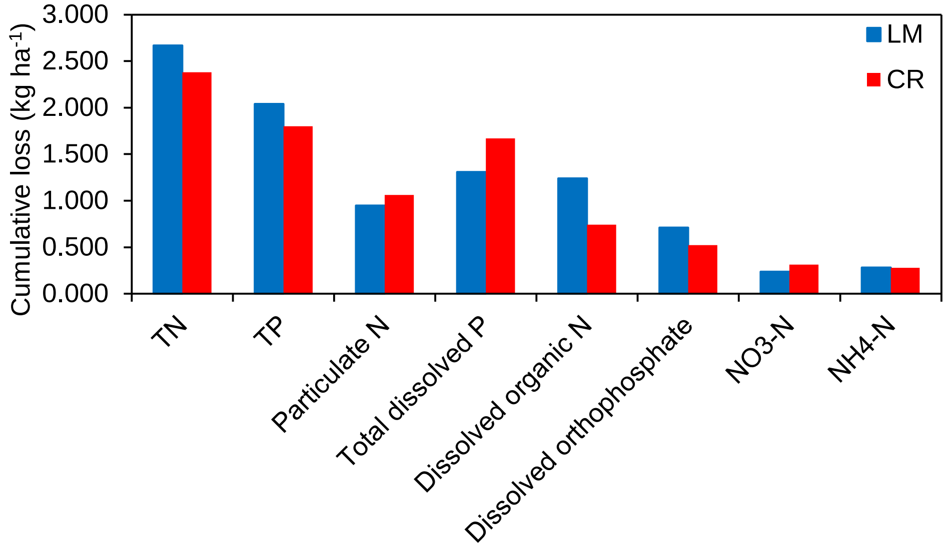

Figure 4.5: Cumulative loss of N and P species from the LM and CR watersheds.

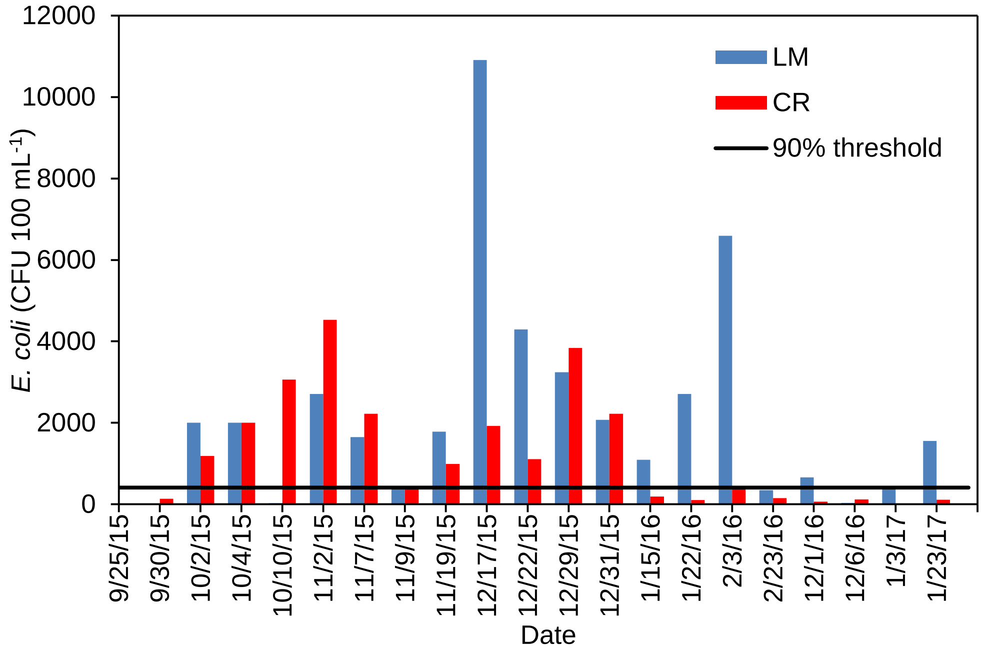

Figure 4.6: *E. coli* loss from the LM and CR watersheds over the study period.

CHAPTER 5

SUMMARY AND CONCLUSIONS

The overall object of this research was to determine the impact of a living mulch on corn production as well as its effects on water quality. The first goal was to compare the soil N dynamics of a living mulch, cereal rye, and crimson clover cover crops on corn growth and yield. Inorganic soil N was lower in the LM treatment than the CR treatment in 2015 and lower than both treatments in 2016. Insufficient inorganic soil in in the LM treatment caused reductions in corn height, N uptake, and aboveground biomass compared to the CC and CR treatments over the two growing seasons. It was hypothesized that inorganic soil N was lower in the LM treatment due to competitive water and N uptake from the intercropped clover and reduced mineralization of clover residue. Reduced mineralization of clover residue was likely the result of lower soil and residue moisture necessary for N transformation. Grain yield in the LM treatment was lower than both the CC and CR treatments, while corn biomass and total N uptake was lower than the CC treatment. Results suggested that environmentally dependent N mineralization and availability to the corn plant, along with competitive water and N uptake, were the most likely sources of variation in corn development and yield within the LM system.

The second goal was to estimate and compare water and NO3-N transport in the CC, LM, and CR systems using a HYDRUS-1D model. NO3-N loss below the 1-m depth during the 2015 - 2016 model period were 47.8 kg NO_3-N ha^{-1} for the CC treatment, 35.4 kg NO_3-N ha^{-1} for the

LM treatment, and 75.4 kg NO_3-N ha^{-1} for the CR treatment. Large NO_3-N losses occurred in all treatments during this model period due to large amounts of rainfall that occurred between November 2015 and February 2016, while very little loss was observed during the growing season. Less than one kg NO_3-N ha^{-1} was lost below the 1-m depth during the 2016 - 2017 model period due to drought that lasted the majority of that time. Results suggest that NO^3-N leaching in southern Piedmont corn production is likely to occur in the winter when frequent precipitation and low evapotranspiration occur. NO_3-N losses in all treatments during the 2015 - 2016 model period were similar to values reported in literature, despite the large amounts of precipitation that occurred during that time period.

The third goal was to compare water and sediment loss from corn production on a living mulch and a cereal rye watershed, as well as quantify nutrient and *E. coli* loss. It was argued that the LM treatment reduced runoff volume when compared to previous data collected on the experimental watershed, but could not be confirmed due to experimental design. The LM treatment reduced sediment loss by 99% when compared to soybean and barley treatments that had occurred previously on the same watershed. Similarly, the CR treatment reduced sediment by 97% when compared to a similar corn production system from the same watershed. Cumulative total N and total P losses from each watershed during the study period was less than 3 kg ha-1. After grazing, *E. coli* levels typically exceeded the mean and threshold limit for surfaces water in Georgia, though there are currently no limits established for agricultural runoff.

As demand for corn continues to increase, management practices that maintain yields and reduce the effect of corn production on water quality will become increasingly important. Conservations practices, such as cover cropping and reduced tillage, have been shown to dramatically reduce the effect of corn production on the environment. Like these conservation practices, living mulch systems have the potential to reduce NO_3-N leaching, reduce sediment and nutrient loads in runoff, and produce grain yields similar to those of traditional corn production systems.