

## **Water Quality Impacts of the Invasive Species Autumn-olive**

by:

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## INTRODUCTION

While regulations and government initiatives targeting agricultural fertilizer applications can help lower the overall nitrogen loads in stream systems, watershed managers can fail to recognize the nitrogen produced by N-fixing species in agricultural watersheds. Ammonium ( $\text{NH}_4^+$ ) is produced by N-fixing plants such as soybeans (*Glycine max*) and autumn-olive (*Elaeagnus umbellata* Thunb.). These plants are able to fix N gas ( $\text{N}_2$ ) and convert it to  $\text{NH}_4^+$  due to a symbiotic relationship with bacteria and actinomycetes (Richardson et al., 2000). Soybeans are associated with the bacteria *Bradyrhizobium*, while autumn-olive is associated with actinomycetes *Frankia* (Kirichenko et al., 2004; Wang et al., 2005). Microbial-mediated nitrification processes in the soil can then convert the  $\text{NH}_4^+$  to nitrate ( $\text{NO}_3^-$ ) (Hart et al., 1994). While N-fixers can provide N to N-deprived areas, they can also add excess N, which has the potential to leach to soil water and subsequently into groundwater (Baer et al., 2006). Additionally, the presence of N-fixers can influence soil properties and processes such as pH and base cation leaching (Wang et al., 2005). As nitrification occurs,  $\text{H}^+$  is yielded during the chemical reaction, leading to acidification of the soil. The  $\text{H}^+$  ion has a stronger bond with cation exchange sites and can displace base cations, causing a loss of nutrients from the system (Matson et al., 1999). One major output of N is through leaching, typically in the form of  $\text{NO}_3^-$ , which is a highly soluble and mobile form of N (Meisinger and Delgado, 2002). N leaching in soil and water systems is getting more attention from both natural resource managers and policy makers.

While there is little information concerning the impacts of autumn-olive on water quality, studies have been conducted on other N-fixing species. One species, grey alder (*Alnus incana* (L.) Moench) can be effective as a natural fertilizer to forest sites due to its ability to fix N (Uri et al., 2003). A study by Montagnini et al. (1991) also showed higher nitrification rates below black locust (*Robinia pseudo-acacia* L.) stands compared to other hardwoods or pines.

Other researchers have discovered significant  $\text{NO}_3^-$  leaching under native woody N-fixers. Red alder (*Alnus rubra* Borg.) dominated ecosystems leached  $2.2 \text{ kg ha}^{-1} \text{ yr}^{-1}$  of  $\text{NO}_3^-$ -N compared to  $0.6 \text{ kg ha}^{-1} \text{ yr}^{-1}$  in Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco) ecosystems in the Pacific northwestern United States (Cole et al., 1978). Van Miegroet and Cole (1984) found up to  $50 \text{ kg ha}^{-1} \text{ yr}^{-1}$   $\text{NO}_3^-$ -N leached below 40 cm in red alder stands. Montagnini et al. (1991) found that soil water  $\text{NO}_3^-$ -N concentrations were 87 times greater at a 60 cm depth under a black locust dominated stand compared to a pine-mixed hardwood stand.

In addition to soil water  $\text{NO}_3^-$  leaching, stream water  $\text{NO}_3^-$  levels have been found to be higher under N-fixing species. Compton et al. (2003) conducted a study in the Oregon Coast Range and determined N fixed under upland red alder was the major N source to the local stream systems. Streams in North Carolina containing riparian black

locust were found to have three times the  $\text{NO}_3^-$  levels of streams with other forest cover types (Swank and Douglass, 1977). Native N-fixers can be difficult to manage. However, exotic, invasive N-fixers can create additional management challenges.

While the definition of an invasive species varies, generally it is defined as a species that has, or is likely to, spread into native systems by seeds or detachable propagules and becomes thoroughly established to the extent that the invasive species displace pre-existing native species (American Nurseryman, 1999). Next to habitat loss and fragmentation, invasive species are currently the second greatest threat to biodiversity (World Conservation Union, 2000). Controlling invasive plants can be difficult, as treatments are typically species specific and require persistent action by managers. Many field and academic professionals agree public education is the first step of any invasive species treatment program (American Nurseryman, 1999).

It has been estimated that approximately 23 billion dollars nationwide is lost annually to invasive plants and that 4600 acres of land are invaded daily (Harper-Lore, 2000). While invasive species appear to spread quickly, they go through a succession process like most plants. According to Williamson (1996), invasive plants first arrive and become established on the site, then spread, causing loss of native species, and then achieve an equilibrium level in the ecosystem. Lastly, the true ecosystem effects of the invasion, such as increased loss of native species and  $\text{NO}_3^-$  leaching become apparent (Williamson, 1996).

In addition to N leaching, invasive species have been found to increase N mineralization and nitrification rates (Ehrenfeld, 2003; Ehrenfeld et al., 2001). A review conducted by Ehrenfeld (2003) found that soil N mineralization rates increased under invasive species due to biomass and fine root differences. Invasive species usually have elevated above ground biomass compared to native species. The increased biomass adds more litter to a site, which typically decomposes more quickly under invasives (Ehrenfeld, 2003; Witkowski, 1991). The decomposed litter adds previously immobilized nutrients to the soil for plant and microbial uptake (Witkowski, 1991). One study showed that *Berberis*, an invasive, had higher N in its foliage compared with native *Vaccinium*, and faster decomposition rates under the invasive *Berberis* could be driving nitrification rates (Ehrenfeld et al., 2001). Soil pH, which affects the solubility of many nutrients, can also decrease under red alder, another N-fixing species. This occurs due to the nitrification process and the production of a  $\text{H}^+$  ion during conversion from  $\text{NH}_4^+$  to  $\text{NO}_3^-$  (Van Miegroet and Cole, 1984). N can become more soluble under higher pH levels and can be a major growth limiting factor to plants. A benefit of N-fixing species is the added N available to the system. Fayatree (*Myrica faya*), an N-fixing invasive, has been found to add four times more N than all other sources combined (Vitousek and Walker, 1989). Increased decomposition, higher soil pH and the possibility of N fixation combine to make more N available to organisms. This increase in N availability influences the biogeochemical cycles in the ecosystem, though not all changes are noticeable.

To noticeably alter the ecosystem biogeochemistry three factors have to occur. First, N must be a limiting nutrient. In these areas, N-fixers have a competitive advantage. Secondly, the species must significantly alter N inputs. Exotic species, specifically N-fixers, have been found to increase soil  $\text{NO}_3^-$  and  $\text{NH}_4^+$  levels significantly (Ehrenfeld, 2003; Baer et al., 2006). Finally, fixed N must be available to other organisms, which has been found to occur (Vitousek and Walker, 1989). As the amount of N available to other organisms' increases, the potential for N to leach out of the system also increases. Since  $\text{NO}_3^-$  is a relatively mobile ion, managers need to be aware of potential water quality problems. While invasive species occur nationally, it may be more feasible to focus on state level efforts of monitoring and control.

In addition to species previously mentioned, autumn-olive is an example of an N-fixing invasive species that is creating problems on a large scale (USDA, 2007). Some problems include alterations of nutrient cycling, inhibition of natural succession, and replacement of native vegetation (U.S. Army Corps of Engineers, 2002). In addition to these issues, autumn-olive fits the category of high risk spread potential, which includes species that are insect pollinated and have fleshy fruits with small seeds (Richardson et al., 2000).

Autumn-olive is considered a woody shrub. It was originally planted in the eastern United States for its wildlife value (Edgin and Ebinger, 2001) and is currently becoming an increasing threat to native plant biodiversity in many states as it continues to spread. The Tennessee exotic plant council has placed it into its severe threat category (Tennessee Exotic Pest Plant Council, 2004). Other states, such as Virginia, have also listed it as a threat to native species biodiversity (VDCR, 1994). It was first planted in Illinois in the 1970's and became problematic a decade later as it became "highly invasive" (Edgin and Ebinger, 2001; Ebinger and Lehnen, 1981). One reason autumn-olive is spreading so quickly is the large amount of seeds the plant can produce annually. A single mature plant, on average, produces approximately three pounds of seed a year, which translates to roughly 66,000 seeds per plant annually (Munger, 2003). Autumn-olive is typically spread by wildlife, particularly birds, that consume the fruit and spread the seeds (Solecki, 1997; Catling et al., 1997). It characteristically invades and eventually overtakes abandoned pastures and fields.

Unlike other N-fixers, little research has addressed autumn-olive's potential effects on water quality. The water quality implications of autumn-olive have been investigated in southern Illinois. Baer et al. (2006) found significantly higher potential net N mineralization and net nitrification levels under autumn-olive plots compared to open field plots. They also discovered significantly higher soil water  $\text{NO}_3^-$ -N levels under autumn-olive plots compared to open field plots. Soil water  $\text{NO}_3^-$ -N concentrations were 20 times higher under autumn-olive compared with the open field plots (Baer et al., 2006). This soil water could contribute significant amounts of  $\text{NO}_3^-$  to groundwater and surface waters.

## **METHODOLOGY**

### **Study Sites**

The present study expands upon Church's (2004) study at three research sites [Tree Improvement Center (TIC), Pleasant Hill Road (PH), and Minetree Road (MT)] located in Jackson County, IL (Baer et al. 2006). Each site consists of an autumn-olive plot and an open field vegetation plot in riparian areas or areas unsuitable for agricultural development. The plots measure 8.5 m x 4.7 m with 0.6 m buffers around the perimeter of the plot to minimize outside plot interferences (Church, 2004).

### **Field Methods**

#### *Net N mineralization*

In summer 2004, soil net N mineralization, net nitrification, and net ammonification rates were measured in-situ with poly-vinyl chloride (PVC) pipe cores at the six plots. There were a total of seventy-two cores, twelve at each of the plots. Thirty-six cores, six at each of the plots, were sampled at the beginning of a four-week incubation period (pre-incubation) and the other thirty-six were left to incubate in-situ for four weeks (post-incubation). The cores were inserted into the mineral soil to a depth of ten centimeters. The pre and post-incubation soil samples were brought to the laboratory for gravimetric moisture content analysis (Gardner, 1986), and ammonium-N ( $\text{NH}_4^+\text{-N}$ ) and  $\text{NO}_3^-\text{-N}$  analysis of soil extracts.

#### *Soil water*

The six plots were divided into 18 subplots (1.2 m x 1.2 m). Six subplots were randomly selected in each plot to monitor soil water N with tension lysimeters, for a total of thirty-six tension lysimeters. The lysimeters were set at 60 centibars of tension with a hand pump. Soil water was sampled at least monthly from January 2003 through April 2005 and every two weeks from September 2005 through April 2007.

#### *Groundwater*

Twelve groundwater monitoring wells were installed in January 2006. Two wells were placed at each of the six plots for a total of twelve wells. Within each plot, the wells were placed nine meters apart, one up-slope from the plot and one within the plot. Once the wells were drilled, they were purged at least twice before sampling began. Groundwater was sampled every two weeks from April 2006 through April 2007.

### **Laboratory methods**

#### *Soil*

$\text{NH}_4^+\text{-N}$  was measured with the indophenol blue method at an absorbance of 660 nm and  $\text{NO}_3^-\text{-N}$  was measured using diazotization with sulfanilamide after a cadmium reduction at an absorbance of 540 nm (Keeney and Nelson, 1982).

### Water

All water samples were analyzed for pH, specific conductivity,  $\text{NH}_4^+$ -N, and  $\text{NO}_3^-$ -N in the Department of Forestry's Water Quality Laboratory. Samples were frozen if not analyzed within 48 hours. Groundwater was first filtered through a 0.45  $\mu\text{m}$  filter to remove any suspended solids.  $\text{NH}_4^+$ -N was tested on a spectrophotometer using the Nessler's reagent method (APHA, 2005).  $\text{NO}_3^-$  concentrations ( $\text{mg L}^{-1}$ ) were measured on an ion chromatograph.

### Statistical Analyses

The general statistical design of the plot study was one factor (vegetation) with two levels (autumn-olive and open field). The study design was randomized complete block (RCB). Soil net N mineralization, net nitrification and net ammonification rates were analyzed with mixed model procedure to test treatment effects, using site as a random effect. Soil net N mineralization, net nitrification, and net ammonification rates were correlated with mean soil moisture content using a Spearman correlation test. The experimental design for the soil water and groundwater data was RCB. A mixed model procedure was used to analyze treatment effects, using site as a random effect. Two-way ANOVA for soil water and groundwater was used to test differences between the dormant and growing season. Linear regression was used on logged data to analyze changes over time for soil water  $\text{NO}_3^-$ -N under autumn-olive plots. Finally, spearman correlation was used to test the relationship between  $\text{NO}_3^-$ -N and each of the four base cations (n=143).

## RESULTS AND DISCUSSION

### Soil Nitrogen Cycling

Net N mineralization was significantly greater ( $p=0.003$ ) under autumn-olive plots ( $16.277 \pm 3.063 \text{ mg kg}^{-1}$ ) than open field plots ( $0.844 \pm 1.370 \text{ mg kg}^{-1}$ ) (Figure 1). Montagnini et al. (1986) measured N cycling rates in July at the Coweeta Hydrologic Laboratory under black locust and found that net N mineralization rates in the top 15 cm were more than two times higher under locust ( $30.94 \pm 1.37 \text{ mg kg}^{-1}$ ) than under either pine-mixed hardwood ( $12.00 \pm 3.32 \text{ mg kg}^{-1}$ ) or oak-hickory ( $2.14 \pm 1.37 \text{ mg kg}^{-1}$ ). Another study, conducted in Estonia, found net N mineralization rates in the upper 10 cm under grey alder between 200 to 300  $\text{mg kg}^{-1}$  in June and July (Uri et al., 2003). While mean values under the autumn-olive were not as high as other studies, values under autumn-olive plots were five to twelve times higher than the open field plots. Conversely; research comparing *Vaccinium* and *Berberis* (a non N-fixing exotic woody shrub) in northern New Jersey showed significantly higher net N mineralization rates under the native *Vaccinium* in June but the exotic *Berberis* was significantly higher in March (Ehrenfeld et al., 2001). *Berberis* N mineralization rates (10 to 20  $\text{mg kg}^{-1}$ ) in March were similar to the autumn-olive rates in this study.

Net nitrification was significantly higher ( $p=0.003$ ) under autumn-olive plots ( $17.606 \pm 2.541 \text{ mg kg}^{-1}$ ) than open field plots ( $2.022 \pm 0.816 \text{ mg kg}^{-1}$ ) (Figure 1). Montagnini et al. (1986) also showed significantly higher net nitrification rates in July in the upper 15 cm under black locust ( $34.26 \pm 1.36 \text{ mg kg}^{-1}$ ) compared to pine-mixed hardwood ( $15.74 \pm 4.11 \text{ mg kg}^{-1}$ ) and oak-hickory ( $0.79 \pm 0.47 \text{ mg kg}^{-1}$ ) stands. Sites under grey alder showed net nitrification rates between 250 to 300  $\text{mg kg}^{-1}$  in the upper 10 cm (Uri et al., 2003). Similar to net N mineralization, net nitrification means under autumn-olive plots were lower than either the black locust or grey alder studies. Non N-fixing invasive species have been found to have increased decomposition and higher litter N than native species (Ehrenfeld et al., 2001). While the N level of autumn-olive litter is not well studied, it could also be contributing to the relatively high observed nitrification rates.

Net ammonification rates were similar ( $p=0.849$ ) between autumn-olive plots ( $-1.329 \pm 0.679$ ) and open field plots ( $-1.178 \pm 0.828$ ) (Figure 1). The negative ammonification rates could be attributed to either immobilization of  $\text{NH}_4^+$  by biota or further conversion to  $\text{NO}_3^-$  (Havlin et al., 2005; Huss-Danell et al., 1982). Uri et al. (2003) showed similar mean net ammonification rates ( $\text{mg kg}^{-1}$ ) under grey alder, near zero or negative values, to the autumn-olive plots. *Berberis* also shows similar net ammonification rates to native *Vaccinium* (Ehrenfeld et al., 2001).

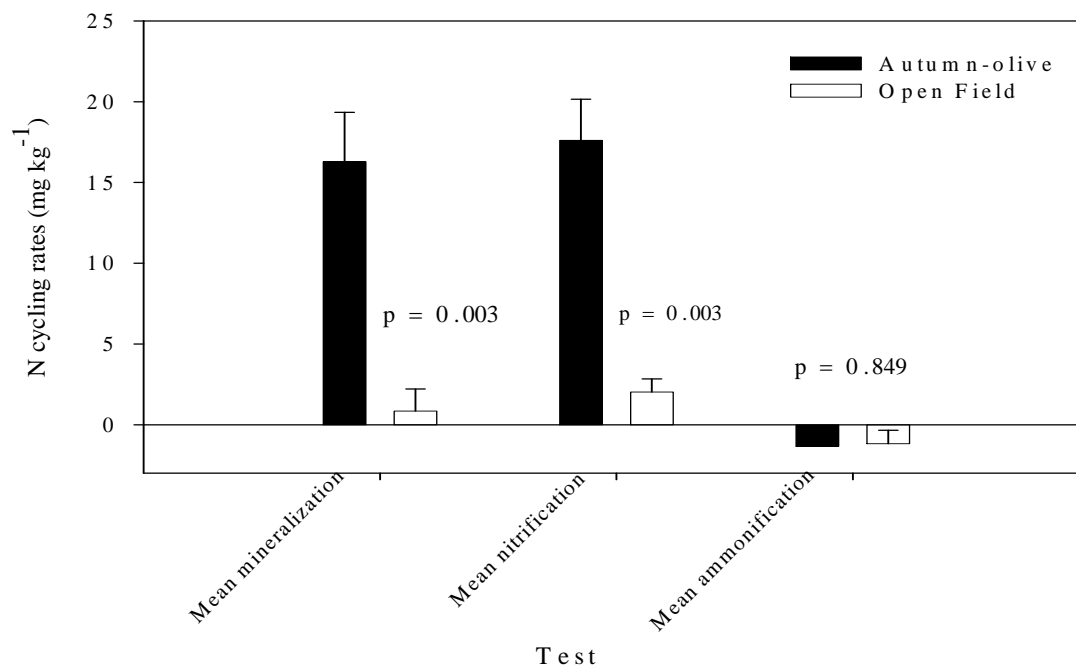


Figure 1: Mean net N Mineralization, net Nitrification and net Ammonification rates for autumn-olive plots (black bars) and open field plots (white bars) from June 16-July 16, 2004.

N-fixing species are usually pioneer species that come into an area and provide N for later succession vegetation that utilizes the soil N pool. Autumn-olive, as an invasive species, has the potential to alter the future invasibility of the ecosystem (Richardson et al., 2000), which can facilitate increased spread of autumn-olive. The  $\text{NH}_4^+$  produced then has the potential to become nitrified and subsequently leach  $\text{NO}_3^-$  into soil water systems.

In-situ soil net N mineralization, net nitrification, and net ammonification rates have been found to be positively correlated with soil moisture content (Hill and Shackleton, 1989; Tietema et al., 1992). This study showed no significant relationship between moisture content and net N mineralization ( $p=0.623$ ), net nitrification ( $p=0.623$ ) and net ammonification ( $p=0.623$ ) rates.

### **Soil Water**

Mean  $\text{NO}_3^-$ -N concentrations in soil water under autumn-olive plots ( $1.841 \pm 0.243 \text{ mg L}^{-1}$ ) were significantly greater ( $p=0.002$ ) than open field plots ( $0.037 \pm 0.008 \text{ mg L}^{-1}$ ) (Figure 2). Other N-fixers have shown similar increases compared to non N-fixers. Black locust mean  $\text{NO}_3^-$ -N values ( $3.730 \pm 0.250 \text{ mg L}^{-1}$ ) were significantly greater than pine-mixed hardwoods ( $0.032 \pm .010 \text{ mg L}^{-1}$ ) and oak-hickory ( $0.008 \pm 0.007 \text{ mg L}^{-1}$ ) stands (Montagnini et al., 1991). A previous study also showed significantly higher  $\text{NO}_3^-$ -N concentrations under autumn-olive plots. Concentrations during this study were greater than the current study, showing mean  $\text{NO}_3^-$ -N levels under autumn-olive plots  $>10 \text{ mg L}^{-1}$  (Church, 2004). These heightened levels in soil water were attributed to increased net N mineralization and net nitrification rates in the soil profile. While the concentrations measured under autumn-olive were lower than those of other N-fixers, the levels were still above mean  $\text{NO}_3^-$ -N concentrations found under maple ( $0.3\text{-}1.0 \text{ mg L}^{-1}$ ) and pine ( $<0.06 \text{ mg L}^{-1}$ ) stands in Ontario (Hill and Shackleton, 1989).

Soil water mean  $\text{NH}_4^+$ -N concentrations under autumn-olive plots ( $0.075 \pm 0.004 \text{ mg L}^{-1}$ ) were similar ( $p=0.392$ ) to open field plots ( $0.006 \pm 0.004 \text{ mg L}^{-1}$ ) (Figure 2). These results agree with a previous study on autumn-olive, which showed  $\text{NH}_4^+$ -N concentrations under autumn-olive of  $0.110 \pm 0.010 \text{ mg L}^{-1}$  (Church, 2004; Baer et al., 2006). The similar values of soil water  $\text{NH}_4^+$ -N found under the vegetation plots could be attributed to immobilization by plants and microbes, further transformation to  $\text{NO}_3^-$ -N, or low soil mobility. These concentrations are comparable with natural forested systems as well. Under both maple and pine stands in Ontario, soil water  $\text{NH}_4^+$ -N concentrations were always  $<0.06 \text{ mg L}^{-1}$  (Hill and Shackleton, 1989).

$\text{NH}_4^+$  is the preferred form of N for uptake by many plants and microbes since it is already in a reduced form (Huss-Danell et al., 1982). Also, net nitrification rates are much higher than net ammonification rates (Figure 1), so the  $\text{N}_2$  fixed by the actinomycetes could be converted from  $\text{NH}_4^+$  to  $\text{NO}_3^-$  before it is sampled by the soil water lysimeters.

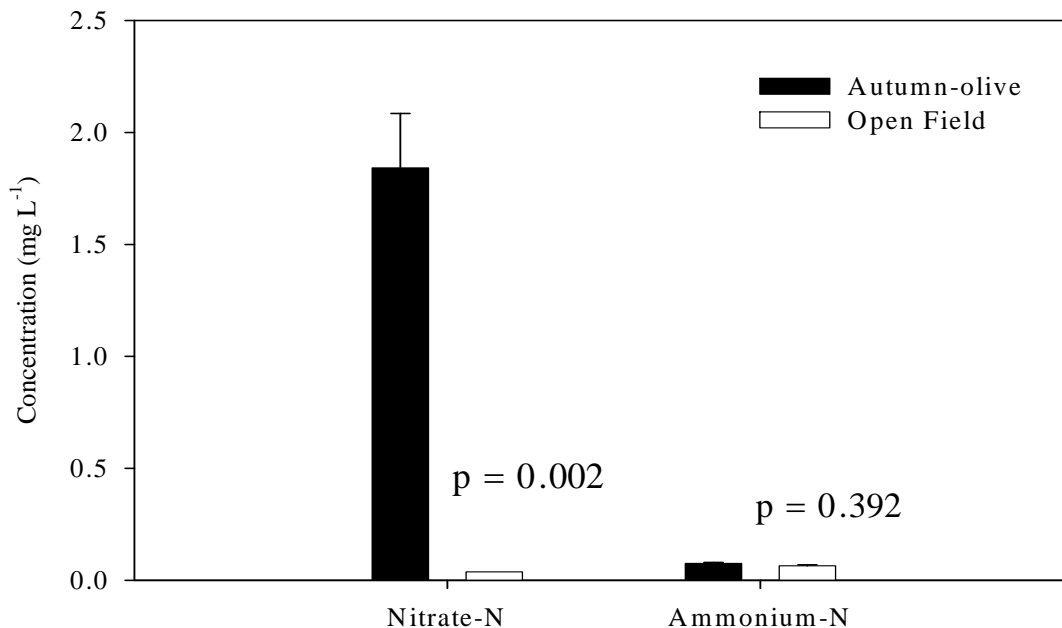


Figure 2: Mean soil water  $\text{NO}_3^-$ -N and  $\text{NH}_4^+$ -N concentrations from February 2004 to April 2007 in autumn-olive and open field plots.

### Base Cations

Soil water  $\text{NO}_3^-$ -N showed a positive correlation with calcium ( $p < 0.001$ ,  $r = 0.462$ ), magnesium ( $p < 0.001$ ,  $r = 0.385$ ) and sodium ( $p = 0.001$ ,  $r = 0.276$ ) in soil water (Table 1). There was no significant difference between autumn-olive and open field plots base cation concentrations. A study by Christ et al. (1999) tested the mobile anion concept on soil water biogeochemistry. The mobile anion concept indicates that in a soil with low anion adsorption ability anion concentration controls the concentration of base cations (Christ et al., 1999). They found that the concept held true at the soil plot scale, showing a correlation between anion and base cation concentrations (Christ et al., 1999). The current study showed a significant positive correlation between soil water  $\text{NO}_3^-$ -N concentrations and cations calcium, magnesium, and sodium (Table 1). Christ et al. (1999) also showed that at increasing depths (lysimeter depth 48 cm) fluctuations in  $\text{NO}_3^-$  were linked with greater base cation fluxes than with other anions. The average depth of the lysimeters in the current study was 54 cm (Church, 2004). Another indication that base cation leaching is likely occurring is the soil water pH range. A study by Schulze (1989) indicated that when soil pH ranged from 4.2 to 6.2 the soil exchanges cations for hydrogen ions. The average pH under the autumn-olive plots (5.43) and open field plots (5.59) both fell within this range, indicating a significant leaching potential.

Table 1: Relationship between soil water NO<sub>3</sub><sup>-</sup>-N and base cations (potassium, magnesium, calcium, and sodium) in twelve first order watersheds in southern Illinois.

	Calcium	Magnesium	Sodium	Potassium
r value (p value)	0.462 (<0.001)	0.385 (<0.001)	0.276 (0.001)	-0.019 (0.821)

*Seasonal Variation*

Mean soil water NO<sub>3</sub><sup>-</sup>-N concentrations in autumn-olive plots were similar (p=0.302) in the dormant season (1.689 ± 0.726 mg L<sup>-1</sup>) and growing season (1.476 ± 0.789 mg L<sup>-1</sup>) (Figure 3). Mean soil water NO<sub>3</sub><sup>-</sup>-N concentrations in open field plots were also similar (p=0.099) in the dormant season (0.027 ± 0.006 mg L<sup>-1</sup>) and growing season (0.087 ± 0.044 mg L<sup>-1</sup>) (Figure 3). In this study, dormant season was defined as November 1 to March 20, which coincides approximately with leaf fall to leaf out for autumn-olive. A previous study showed that dormant season soil water NO<sub>3</sub><sup>-</sup>-N levels were greater than growing season values under autumn-olive plots but not under open field plots (Baer et al., 2006). Data from the present study were collected over multiple years compared to the previous study, which only collected one year of data. Precipitation variability between the two studies may have also influenced NO<sub>3</sub><sup>-</sup>-N concentrations. Increased flushing is more likely to occur during the dormant season when more water is in the soil profile (Meisinger and Delgado, 2002); yet higher net N mineralization and net nitrification rates during the growing season could provide enough NO<sub>3</sub><sup>-</sup> for leaching to occur, even with reduced soil moisture levels.

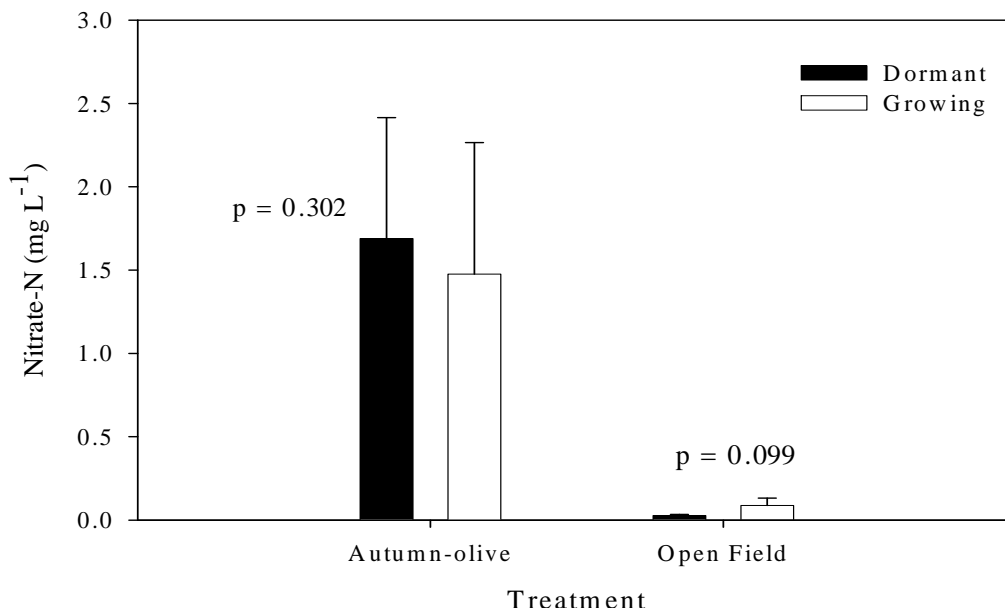


Figure 3: Mean soil water NO<sub>3</sub><sup>-</sup>-N concentrations from February 2004 to April 2007 for autumn-olive plots and open field plots for the dormant (black bars) and growing (white bars) season.

Mean  $\text{NH}_4^+$ -N concentrations in autumn-olive plots were similar ( $p=0.121$ ) in the dormant season ( $0.088 \pm 0.012 \text{ mg L}^{-1}$ ) and growing season ( $0.075 \pm 0.012 \text{ mg L}^{-1}$ ). Mean  $\text{NH}_4^+$ -N concentrations in open field plots were also similar ( $p=0.420$ ) in the dormant season ( $0.070 \pm 0.018 \text{ mg L}^{-1}$ ) and growing season ( $0.063 \pm 0.015 \text{ mg L}^{-1}$ ). As previously mentioned, net N mineralization rates are usually higher in the growing season, which leads to more  $\text{NH}_4^+$  production. This  $\text{NH}_4^+$  can either be immobilized by microbes and plants, converted to  $\text{NO}_3^-$ , bound to soil cation exchange sites, or fixed in 2:1 clay lattices (Havlin et al., 2005). These potential fates of  $\text{NH}_4^+$  typically results in little leaching year round.

### **Groundwater**

Mean groundwater  $\text{NO}_3^-$ -N concentrations were low (Figure 4). They were similar ( $p=0.398$ ) between autumn-olive plots ( $0.048 \pm 0.011 \text{ mg L}^{-1}$ ) and open field plots ( $0.007 \pm 0.003 \text{ mg L}^{-1}$ ) (Figure 4). With significantly higher  $\text{NO}_3^-$ -N concentrations in soil water, the  $\text{NO}_3^-$  would be expected to leach to the groundwater systems. Elevated levels have been found under agricultural fields but not under adjacent riparian forests. Concentrations exiting corn fields in Maryland have been shown to be as high as 6.76 and 7.40  $\text{mg L}^{-1}$ , but water exiting an adjacent riparian forest has shown values as low as 0.101  $\text{mg L}^{-1}$  (Peterjohn and Correll, 1984). Other forested systems have shown similar results as well. Hill and Shackleton (1989) found low  $\text{NO}_3^-$ -N concentrations under both maple (0.2-0.6  $\text{mg L}^{-1}$ ) and pine (0.05-0.15  $\text{mg L}^{-1}$ ) stands.

One likely reason for the low observed mean groundwater  $\text{NO}_3^-$  levels is denitrification. Denitrification is usually highest in the upper two meters, where labile carbon is plentiful (Trudell et al., 1986, Davidson and Swank, 1987), though it has been found to occur in deeper horizons and aquifers. A study by Francis et al. (1989) found a denitrifier population in deep subsurface areas and determined activity occurred if conditions were suitable. Factors that are most important to denitrification are a source of  $\text{NO}_3^-$ , a neutral or slightly basic pH, a lack of oxygen, and a carbon source. The  $\text{NO}_3^-$  is likely present as evidenced by soil water collections, though less was found in the soil profile in the last year of the study (while the groundwater was collected) than in previous years. The optimum pH range for denitrification is 7.0 to 8.2 (Wijler and Delwiche, 1954). Mean groundwater pH under autumn-olive plots ( $6.968 \pm 0.025$ ) was similar ( $p=0.377$ ) to open field plots ( $6.850 \pm 0.016$ ). Thus, pH was just below the optimum range, and fell within the adequate range for denitrification (Wijler and Delwiche, 1954). Another influence on denitrification is anaerobic conditions. Depth to the water table under autumn-olive plots ( $4.850 \pm 0.222 \text{ m}$ ) was similar ( $p=0.326$ ) to open field plots ( $3.225 \pm 0.080 \text{ m}$ ). Previous studies have shown denitrification occurring in areas with a water table less than four meters below the surface (Trudell et al., 1986). Mean water table levels in this study were around four meters though depths did vary slightly throughout the collections. The deepest water table depths were found in summer months, which was found in previous studies (Pinay et al., 1993). Pinay et al. (1993) found the lowest denitrification rates during the summer and attributed it to the lowered water table. The lowest concentrations of groundwater  $\text{NO}_3^-$ -N under the

autumn-olive were found in the summer months, so it is likely that denitrification rates are higher during the summer at these sites. As water table depth during this period was below two meters, uptake by vegetation was not a likely component of  $\text{NO}_3^-$ -N removal. A source of labile carbon for denitrifying bacteria is the final major factor influencing denitrification rates. Dissolved organic carbon (DOC) has been found to drop drastically from  $10 \text{ mg L}^{-1}$  to  $1.0 \text{ mg L}^{-1}$  between 1.25 meter and 2.5 meter water table depth, apparently supporting denitrifying activity in the shallower water table depths and the lack of labile carbon in deeper substrates could restrict denitrification (Starr and Gillham, 1993). DOC was not measured in this study, though it is assumed that there were adequate labile carbon levels to support denitrification.

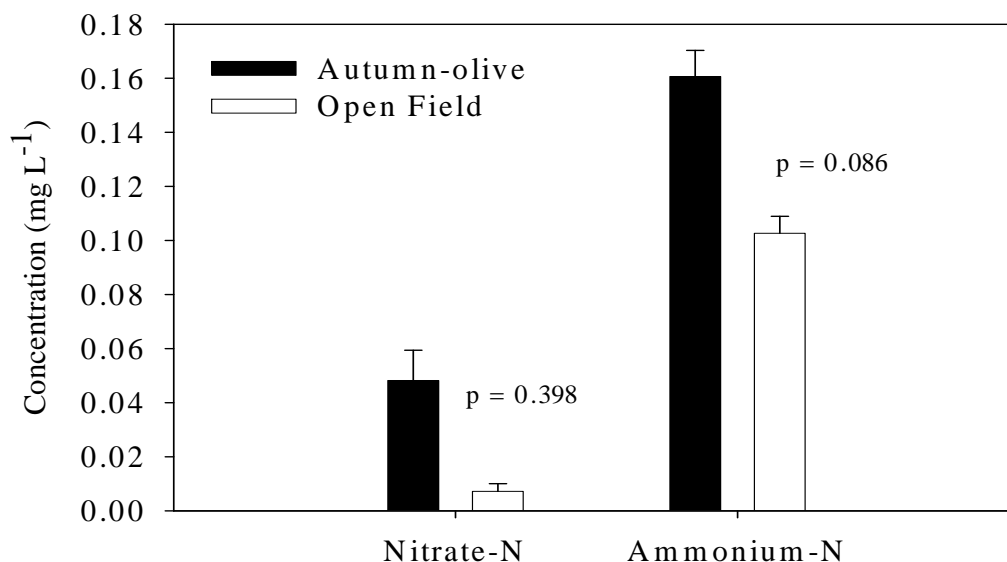


Figure 4: Mean groundwater  $\text{NO}_3^-$ -N concentrations from April 2006 to April 2007 for autumn-olive plots and open field plots.

The low observed groundwater  $\text{NO}_3^-$ -N concentrations may also be due to hydrologic connectivity between soil water and groundwater, given the relatively deep water tables (> 3 m) at the study sites. Riparian soil characteristics such as soil texture and structure can affect hydrologic connectivity between soil water and groundwater (Bardgett et al., 2001). As the wells were dug, it was noted that there were thick clay horizons on top of glacial till. This soil texture likely lowered water percolation, creating slow recharge to groundwater systems. At PH there was also an entrenched stream adjacent to the plots. Entrenchment can reduce groundwater levels (Groffman et al., 2003) creating a limited hydrologic connection. The lack of hydrologic connection can prevent transport of materials such as nutrients from the soil water to groundwater systems and may help explain why most of the wells failed to show detectable levels of  $\text{NO}_3^-$  during most of the study. Low groundwater  $\text{NO}_3^-$ -N concentrations beneath the open field plots (Figure 4) may also have been due to the low  $\text{NO}_3^-$ -N concentrations found in soil water.

The shallowest well had an average water table depth of  $1.678 \pm 0.156$  m. It was one of the few wells that may have had a good hydrologic connection with near surface horizons. With the exception of two dates (January 27 and February 9), this well had the only measurable  $\text{NO}_3^-$ -N levels. Prior to the two collections (January 27 and February 9) where more than one well had detectable  $\text{NO}_3^-$ -N concentrations, it rained 26.9 mm on January 13 and 67.8 mm on January 15. This rain during the dormant season likely helped to flush  $\text{NO}_3^-$  into the wells (Wondzell and Swanson, 1996). Groundwater  $\text{NO}_3^-$ -N concentrations dropped in this well to  $0 \text{ mg L}^{-1}$  by August 2006 and did not increase until November 17, 2006, indicating substantial denitrification rates during the summer months. As water table depth during this period was below two meters, most of the reduction would likely be attributed to denitrification as opposed to uptake by vegetation.

Mean groundwater  $\text{NH}_4^+$ -N concentrations were similar ( $p=0.086$ ) between autumn-olive plots ( $0.161 \pm 0.010 \text{ mg L}^{-1}$ ) and open field plots ( $0.103 \pm 0.006 \text{ mg L}^{-1}$ ) (Figure 4), but were higher than  $\text{NO}_3^-$ -N concentrations, which was not expected given that  $\text{NH}_4^+$  is relatively immobile compared to  $\text{NO}_3^-$ .  $\text{NH}_4^+$ -N concentrations were elevated compared to other forested systems. Hill and Shackleton (1989) found  $\text{NH}_4^+$ -N concentrations less than  $0.06 \text{ mg L}^{-1}$  under both maple and pine stands.  $\text{NH}_4^+$  reaching the groundwater was likely mineralized or fixed  $\text{NH}_4^+$  that was not further nitrified.

#### *Seasonal variation*

Mean groundwater  $\text{NO}_3^-$ -N concentrations in autumn-olive plots were similar ( $p=0.097$ ) in the dormant season ( $0.079 \pm 0.064 \text{ mg L}^{-1}$ ) and growing season ( $0.030 \pm 0.030 \text{ mg L}^{-1}$ ) (Figure 5). Mean groundwater  $\text{NO}_3^-$ -N concentrations in open field plots were significantly greater ( $p=0.006$ ) in the dormant season ( $0.019 \pm 0.004 \text{ mg L}^{-1}$ ) than the growing season ( $0.000 \pm 0.000 \text{ mg L}^{-1}$ ) (Figure 5). The significance was probably due to the fact that no wells within the open field plots showed  $\text{NO}_3^-$  during the growing season. The increased  $\text{NO}_3^-$ -N concentrations in the dormant season are likely due to increased flushing of  $\text{NO}_3^-$  from the soil profile to the groundwater and lack of biotic immobilization (Meisinger and Delgado, 2002, Wondzell and Swanson, 1996). Hill and Shackleton (1989) showed the highest  $\text{NO}_3^-$ -N concentrations between November and April under maple and pine, due to increased leaching from October to May.

Mean groundwater  $\text{NH}_4^+$ -N concentrations in autumn-olive plots were similar ( $p=0.993$ ) in the dormant season ( $0.160 \pm 0.026 \text{ mg L}^{-1}$ ) and growing season ( $0.161 \pm 0.021 \text{ mg L}^{-1}$ ). Mean  $\text{NH}_4^+$ -N concentrations in open field plots were also similar ( $p=0.279$ ) in the dormant season ( $0.099 \pm 0.014 \text{ mg L}^{-1}$ ) and growing season ( $0.107 \pm 0.014 \text{ mg L}^{-1}$ ). As  $\text{NH}_4^+$  is the preferred form of uptake by plants and microbes (Huss-Danell et al., 1982), little  $\text{NH}_4^+$  is available to leach to groundwater.

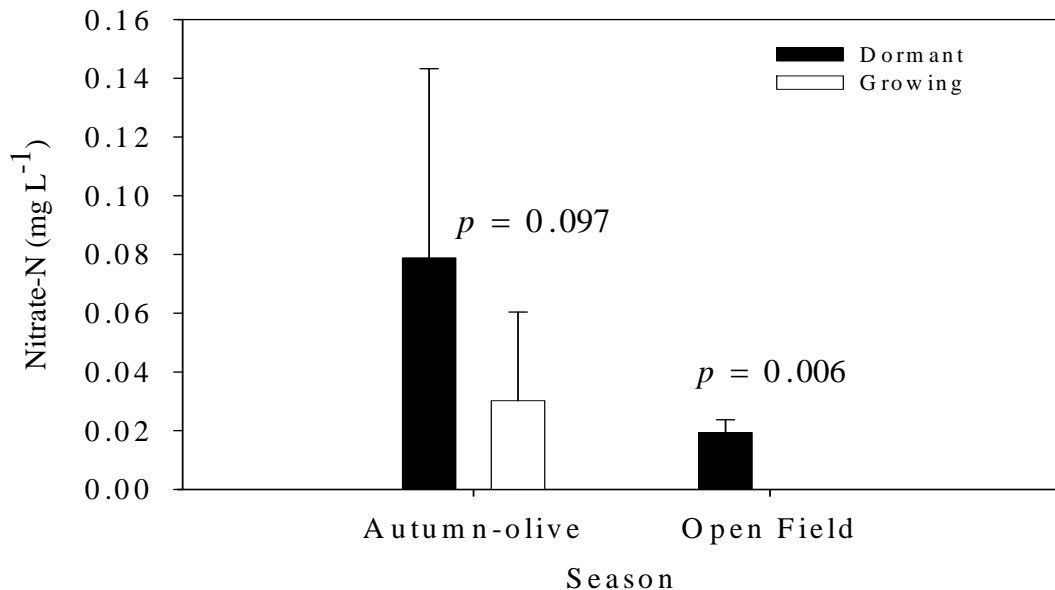


Figure 5: Mean groundwater NO<sub>3</sub><sup>-</sup>-N concentrations from April 2006 to April 2007 for autumn-olive plots and open field plots during the dormant (black bars) and growing (white bars) season.

## CONCLUSIONS

The objectives of the study were to measure the transport of nitrogen (N) through the ecosystem from production to export and to determine if more N leaches beneath autumn-olive plots compared to open field plots.

Net N mineralization and net nitrification rates were significantly higher under autumn-olive plots compared to open field plots. These results coincide with other N-fixers and their influence on nutrient cycling. As a N-fixer, the NH<sub>4</sub><sup>+</sup> produced by actinomycetes is likely quickly converted to NO<sub>3</sub><sup>-</sup>, a relatively water soluble N form that is prone to leach into water systems.

Mean soil water NO<sub>3</sub><sup>-</sup>-N concentration was significantly greater under autumn-olive plots than open field plots. The elevated concentrations are most likely due to the high net N mineralization and net nitrification rates found in the soil profile. NO<sub>3</sub><sup>-</sup>-N concentrations showed a significant decreasing trend. This decrease in NO<sub>3</sub><sup>-</sup>-N could be attributed to the increased age of the autumn-olive. Soil water NO<sub>3</sub><sup>-</sup>-N levels did not differ seasonally, indicating that although NH<sub>4</sub><sup>+</sup> is not prone to be nitrified in the dormant season; enough NO<sub>3</sub><sup>-</sup> was present in the soil to impact soil water systems year round. Mean soil water NH<sub>4</sub><sup>+</sup>-N concentrations were similar between the autumn-olive plots and

the open field plots. A previous study on autumn-olive also found significant  $\text{NO}_3^-$ -N concentrations, but showed negligible  $\text{NH}_4^+$ -N in soil water collections (Church, 2004).

Mean groundwater  $\text{NO}_3^-$ -N concentrations were relatively low, likely due to denitrification. The one well that seemed to have the best hydrologic connection with soil water showed  $\text{NO}_3^-$  consistently, except in the summer months, indicating a higher rate of denitrification during the summer. Although mean groundwater  $\text{NH}_4^+$ -N concentrations were not significant, they were higher than  $\text{NO}_3^-$ -N concentrations; this is surprising since  $\text{NH}_4^+$  is relatively immobile. The mean  $\text{NH}_4^+$ -N concentrations were also elevated compared to other forest systems (Hill and Shackleton, 1989).

## **MANAGEMENT IMPLICATIONS**

Autumn-olive was promoted for planting in the 1960's by the Soil Conservation Service primarily for its wildlife and erosion control benefits (Edgin and Ebinger, 2001). Autumn-olive holds its fruit later in the winter, making it available to wildlife such as birds, turkeys, and raccoons, among others. Its dense rooting network helps provide slope and bank stabilization. For this reason some of the planting was conducted in riparian areas. Finally, as a N-fixer it helps bring N into sites that are N-deprived, increasing site productivity.

The aforementioned beneficial characteristics of autumn-olive, particularly the wildlife food source and N fixation, may result in long-term negative ecosystem consequences via the adage of "*too much of a good thing*". For instance, with its prolific seed production and its ability to grow in a wide variety of sites it can spread rapidly and overtake an area (Munger, 2003; Edgin and Ebinger, 2001). Due to higher litter quality (i.e. lower C:N ratio) (Gorham et al., 1979), increased net N mineralization and net nitrification rates in soil, coupled with potentially faster decomposition rates (Ashton et al., 2005) could create ecosystems that are unfavorable to successional growth, thus leading to more N available for leaching. These negative vegetation and water quality impacts are likely outweighing any benefits and thus, control is needed.

Control of invasive species is typically accomplished by one of three methods: mechanical, chemical, and/or biological. When autumn-olive is young it can be pulled out by hand, but larger shrubs need to be cut or bulldozed. While this temporarily removes autumn-olive from a site, regeneration occurs rapidly (Solecki, 1997). Chemical applications on cut stumps tend to be the best available control mechanism (Solecki, 1997). The use of herbicides could lead to increased contamination of groundwater and stream water systems, thus chemical control should be used carefully. To our knowledge, a host-specific biological control for autumn-olive has not been discovered.

Currently, management of autumn-olive is more focused at State rather than Federal levels (Munger, 2003). Tennessee has listed it as a severe threat to native

vegetation (Tennessee Exotic Pest Plant Council, 2004) and Virginia has noted it as a troublesome species (VADCR, 1994). As it has spread throughout the eastern United States, control on a large scale may not be feasible initially. Areas that have small populations or contain ecologically sensitive areas should be targeted first for control (Munger, 2003). Another key aspect of control is increasing public awareness of the potential problems associated with autumn-olive infestation. With heightened public awareness, autumn-olive removal may be conducted voluntarily in an area before infestations become too wide spread.

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