Kirkwood-Cohansey Project

Nitrogen Dynamics Study Final Report

Part II. Dynamics of Nitrogen under Field Conditions

Prepared in cooperation with the Pinelands Commission

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Introduction

Nitrogen cycling in wetlands is a central focus of research on wetland ecology, as the ability of wetlands to remove excess nitrogen through denitrification and through storage in soil organic matter is one of the most critical wetland functions (Simmons et al. 1992, Saunders and Kalff 2001, Butturini et al. 2003, Hansson et al. 2005, Kaye et al. 2006, Merrill and Benning 2006). The ability of wetlands to retain nitrogen reflects several components of both the nitrogen cycle and the properties of the wetland soil. Rates of net mineralization determine the potential supply of ammonium to ammonium oxidizing bacteria, and the activity of this group of bacteria is regulated in large part by the presence of oxidized conditions in the soil (Paul 2007). The presence of oxidized conditions, in turn, is a function of both the presence and duration of flooding, and the organic matter content of the soil. Prolonged flooding combined with high soil organic matter content will result in rapid consumption of oxygen and thus the establishment of anaerobic conditions that prevent nitrification from occurring (Reddy and DeLaune 2008). Conversely, when wetland soils are drained, or if they dry during precipitation-free periods of time, nitrification rates may be high because of the presence of high concentrations of ammonium.   The high nitrification rates that can occur under such conditions can potentially result in the production of net nitrate (i.e., nitrate in excess of denitrification losses). This nitrate is then available for either leaching to ground water and/or movement to surface waters.

Despite an extensive amount of research on wetland nitrogen cycling (Reddy and DeLaune 2008), comparative studies of nitrogen mineralization among different wetland types in the same region are relatively uncommon. Most such studies compare different types of peatland, such as bog and fen peats supporting different types of plant communities and having different amounts of minerotrophic ground water inputs (Updegraff et al. 1995, Bridgham et al. 1998, Chapin et al. 2003, Keller et al. 2004), particularly in northern (boreal and arctic) ecosystems (Jonasson and Shaver 1999, Bayley et al. 2005, Fellman and D'Amore 2007). Conversely, studies of riparian and mineral soil wetlands have compared mineral soils in different types of riparian wetlands or in different locations (Simmons et al. 1992, Bechtold and Naiman 2006, Wassen and Olde Venterink 2006), and have emphasized the role of soil textural differences, geomorphological influences on water flow, and atmospheric deposition as controlling factors differentiating sites. Fellman and D’Amore (2007) compared peatland soils (bog and forested peatland) with riparian mineral soils in Alaska, and found that N mineralization rates were faster in riparian soils than either peatland soil. Other studies that have compared organic and mineral wetland soils have concentrated on fluxes other than net mineralization (e.g., Hanson et al. 1994). In the New Jersey Pinelands, mineral soil and organic soil wetlands occur in close proximity to each other along shallow water table gradients (Ehrenfeld 1986, Forman 1998). In these adjacent wetlands, there is overlap of species composition (Ehrenfeld 1986, Zampella et al. 1992), but differences in water table position, and differences in hydric soil properties along the hydrologic gradient. Previous studies of nitrogen dynamics in these wetlands (Poovarodom et al. 1988, Sangemeswaran 1995) have suggested that the differences among wetlands on different soil types are not related to environmental factors.

We have monitored net mineralization and nitrification rates in three types of wetlands in the New Jersey Pinelands to determine the natural patterns of production of ammonium and net nitrate. The wetlands vary in hydrology (position of the water table), soil type (histosol vs. mineral hydric soils), soil moisture, and soil organic matter content, and also vary in plant community composition (Ehrenfeld 1986, Zampella et al. 1992). Previous studies of nitrogen cycling in these soils (Poovarodom et al. 1988, Zhu and Ehrenfeld 1999) have reported that nitrate and nitrification are extremely low under most circumstances, but can be elevated under disturbed conditions, such as in wetlands affected by stormwater inputs in urban areas (Zhu and Ehrenfeld 1999). The overall goal of the study was to conduct a field study to test a conceptual model relating water table dynamics to nitrogen mineralization rates (Fig. 1), in order to ascertain how variable moisture regimes, driven by differences in water table dynamics and the strength of the linkage between water tables and soil moisture, affect nitrogen mineralization and nitrification rates, and to evaluate the potential for the release of nitrogen from the wetlands due to groundwater withdrawals. We posit in this model that soil moisture, through its control of microbial activity, controls the accumulation of organic matter in the soil, with greater moisture levels resulting in slower decomposition and therefore more organic matter accumulation (Reddy and DeLaune 2008). Further, this model posits that soil moisture is ultimately controlled by water table depth. Organic matter supplies the organic N which can be mineralized, and possibly nitrified, depending on moisture as a control on oxygenation.

Specifically, we address the following questions: 1) how do net mineralization and net nitrification rates vary among different wetland types found in close proximity to each other? 2) what is the relative importance of water table, soil moisture, and soil organic matter in determining these rates? 3) is there net production of nitrate that moves to groundwater? These questions have been addressed by conducting *in-situ* measurements of N cycling rates throughout a year in two sets of adjacent wetland sites.

Methods

Sites

A catenary set of three wetland plots was established at each of two locations (referred to below as “sites”) in the McDonalds Branch watershed in Brendan T. Byrne State Forest (39^{o}53’05” N, 74^{o}30’20”), designated here M10 and M6 (Fig. 2). The plots were a subset of those used in related vegetation studies conducted as part of the Kirkwood-Cohansey Project (Laidig et al. 2010). At each site, three wetland types were distributed along a drainage and soil catena (Zampella et al. 1992). They included pine wetlands, which are pitch pine (*Pinus rigida*)-dominated communities at the upland-wetland boundary (here termed ‘PW’), pine-hardwood swamps, which contain mixtures of red maple (*Acer rubrum*), black gum (*Nyssa sylvatica*), sweet bay magnolia (*Magnolia virginiana*) and pitch pine (here termed ‘PH’), and cedar swamps, which are dominated by Atlantic white-cedar (*Chamaecyparis thyoides*) (here termed ‘CS’) in the lowest topographic positions bordering streams. The wetland communities are termed “types’ below, and the set of 6 sampled wetlands are referred to as ‘plots.’ Plant community composition for these wetland types has been described by Ehrenfeld and Gulick (1981), Ehrenfeld (1986), Ehrenfeld and Schneider (1991), Zampella et al. (1992), Forman (1998), Laidig and Zampella (1999), Laidig et al. (2010) and others. Figure 2 illustrates the locations of the two sets of wetland plots.

The pine wetlands and pine hardwoods communities are found on wetland mineral soil of the Atsion and Berryland series. The Atsion series (sandy, siliceous, mesic Aeric Alaquods) is poorly drained, and is characterized by a thick organic horizon and a dark gray sandy upper mineral horizon that is 20+ cm deep. The Berryland series (sandy, siliceous, mesic Typic Alaquods) is very similar but is classed as very poorly drained, and has a surface mineral horizon that is black to dark gray sand, and up to 25 cm thick (Soil Survey Staff Natural Resources Conservation Service). The white cedar swamps, in contrast, are found on histosols (mapped as Manahawkin muck) that are 1-2m deep (M6 site, 2 m depth; M10 site, 0.6m depth)

Data on climate (air and soil temperatures and daily precipitation) were obtained for the study period from the US Forest Service climate station at the Silas Little Experimental Forest, New Lisbon, NJ. Air and surface soil temperatures were highly correlated (r=0.88), and so air temperatures were used as an indicator of temperature conditions during each month of incubation.

Soil Sampling and Analysis

Nitrogen mineralization and nitrification rates were measured on a monthly basis for a period of 12 months (April 2005 to April 2006) in each plot (Fig. 2). On each sampling date, three samples were taken from just outside the plot boundary at each site except the M10C site; at this site, samples were taken around a similar $100m^2$ plot placed near the boundary of the cedar plot and the pine hardwoods community. Sampling was done on three of the plot boundaries, in order to ensure that any spatial variability in the conditions at each of the sites was fully represented in the study. Each month, the sampling location on each plot boundary was moved about 1 m, so that no soil area was sampled after disturbance from the previous month's sampling. This sampling scheme also ensured that the variability in soil conditions across any gradient in elevation around the plot area was captured in the sampling. Thus, there were three replicate samples from each plot per month.

We used a hammer corer with a 20 cm long plastic liner to take two adjacent soil cores of approximately 15 cm length from each sampling spot. Preliminary observations of soil properties around the perimeters of the PW and PH plots showed that the organic horizon thickness was quite variable (3 -12 cm at each plot). We have previously found that extractable N in Pinelands soils are strongly affected by the thickness of the organic horizon (Ehrenfeld et al. 1997) so that small differences in the amount of organic horizon material included in the core would result in large differences in extractable N and N process rates. Therefore, the depth of the organic horizon material was measured, and then this horizon was removed prior to sampling, so that the cores contained only mineral soil material. In addition, the sensitivity of nitrogen dynamics to variations in water table and/or soil moisture would be more readily observed in the sandy mineral material, without the presence of a variable amount of organic horizon material. Removal of the overlying organic horizon was facilitated by a sharp boundary between it and the underlying mineral 'A' horizon. All data reported below refer exclusively to the mineral soil material. In the two cedar swamps, any poorly-decomposed material (usually *Sphagnum* spp.

tissues) was removed before sampling the highly decomposed muck (saprist) soil material, and all samples were taken from hollows. Again, this procedure was used to ensure the minimal amount of extraneous variability among replicates from each site. In addition, because the majority of plant roots are located within the organic horizon (Ehrenfeld et al. 1993), it is likely that the net N mineralized within this horizon would be taken up by plants, and not leach to lower horizons or the water table.

Of the two adjacent cores taken at each sample point, one was returned to the soil (‘incubation core’), and one was sealed and returned to the laboratory for analysis (“fresh core”). The “incubation core” was sealed at the bottom end with a rubber stopper and was covered on the top with a piece of aluminum foil. It was replaced in the sample hole and covered with the organic layer material. This allowed the core to incubate at field temperatures; it was collected after a one-month period, following standard protocols (Robertson et al. 1999). The fresh core was transported back to the laboratory in an iced cooler. Soils from this core were extracted with 2 M KCl within 24 hr, and the extracts analyzed for inorganic nitrogen contents (NH_4^+ and NO_3^- $+NO_2^-$) on a Lachat® QuickChem® FIA+ 8000 autoanalzyer (Milwaukee, WI, USA). Net nitrogen mineralization rate and net nitrification rate in soil are calculated from differences of inorganic nitrogen content between the "incubation core" and the "fresh core" divided by the number of incubation days, expressed as "mg N m^{-3} soil day^{-1}". All nitrogen data were expressed on a volume basis because the very large difference in bulk densities of the mineral and organic soils made comparison on a mass basis misleading. Soil moisture is calculated based on dry weight, and adjusted to a volumetric basis using the measured bulk density at each plot. All soil variables reported below have a sample size of three.

The following measurements were made on all soil samples: organic horizon thickness (cm) was measured in the field prior to extraction of the cores, soil pH (1:1 in H_2O), soil electrical conductivity (EC, 1:1 in H_2O) as $\mu S\ cm^{-1}$, soil moisture as percent moisture, and soil organic matter measured as loss-on-ignition, expressed as kg m^{-3} on the basis of oven-dry soil. Bulk density was determined for each plot as the average of three samples, and used to convert all values calculated on a mass basis to a volumetric basis.

Water table levels were estimated by developing regression equations between partial-series well data collected at observation wells located in the center of each plot (data from A. Brown, Pinelands Commission) and continuously-recorded water level data collected at the US

Geological Survey's well located at M10C (USGS well MB1) over the period April 19, 2004 to June 4, 2006 (data from R. Nicholson, U. S. Geological Survey). All regressions had r^2 values >0.90. The hourly data from the continuously-recording well were averaged to produce a value representative of each 24 hour period; these data were corrected for topography within the wetland (data from A. Brown, Pinelands Commission), and then were regressed against the partial-series data from the well located within each vegetation plot to generate water level data for the sampling dates of this study.

Comparative study of cedar wetlands

In order to ascertain that the information on nitrogen concentrations and process rates measured at the 6 intensive study plots was indeed representative of wetlands throughout the Pinelands region, we undertook a one-month study of cedar wetlands in four of the study watersheds associated with the Kirkwood-Cohansey project. This study was undertaken in July to August 2005. The cedar wetlands within each of the watersheds, including McDonalds Branch (5 plots), Morses Mill Stream (5 plots), Skit Branch (4 plots) and Albertson Brook (5 plots), were a subset of swamp sites used in related vegetation studies (Laidig et al. 2010). The plots for the McDonalds Branch watershed included the M6 and M10 cedar wetlands used in the year-long study, and the study was timed to coincide with the regular monthly sampling at these two sites. The watersheds were chosen to represent undisturbed (McDonalds Branch and Skit Branch) and moderately disturbed (Albertson Brook and Morses Mill Stream) sites. Upland agriculture and developed lands were prominent features of the Albertson Brook and Morses Mill Stream basins, which were characterized by streams with circumneutral waters and elevated dissolved-solid concentrations; see Laidig et al. 2010 for more complete descriptions of these watersheds.. In each site, three replicate sets of samples were established adjacent to the vegetation survey plot, as was done for the regular year-long sampling, and were incubated for a single 4-week period. Soils were analyzed as described above.

Because this effort was not part of the original work plan for the K-C project but represented additional labor and sampling costs not accounted for in the project as funded, we only focused on cedar swamps, as the most critical wetland resource and the focus of greatest conservation interest. While it would have been desirable to do similar additional sampling for the pine-wetlands and pine-hardwood types, resources did not permit such additional sampling.

Nitrate in shallow groundwater

In order to relate changes in nitrogen dynamics within the surface soils to potential movement of nitrate into the groundwater, we installed and sampled shallow piezometers at the six plots used for the monthly sampling (M10 and M6 sites). Piezometers (wells screened over a short length) were chosen in order to specifically sample the permanently saturated zone, rather than the entire soil profile. Thus, any nitrate present in the soil had to have moved downward into the water table in order to be detected.

The piezometers were constructed following methods suggested by R. Walker, U. S. Geological Survey. Samples were taken from a 10 cm stainless steel mesh screen installed at the end of a galvanized iron slotted pipe (2.5 cm diameter). The partial record series from the well within each vegetation plot was used to estimate the approximate depth of the water table during the summer months, and this depth was chosen as the depth at which the screened portion of the piezometer was placed. The piezometer well depths were as follows: M10PW 100cm; M10PH 60cm; M10CS 40 cm; M6PW 62-73 cm; M6PH 50-55 cm; M6CS 40 cm. The piezometers were installed along the edges of the vegetation plots, in the same positions used for the field mineralization sampling (Fig. 2). We note that we had considerable difficulty in installing the piezometers in the M6 pine hardwood and pine wetland sites, so that the three piezometers were not at exactly the same depths. An apparent clay lens made it very difficult to drive the well to the target depths (the wells would buckle rather than penetrate the soil). The screened tips of the piezometers were installed below the clay lenses.

The pore water was sampled by evacuating the wells and then taking a sample from water freshly entering the tube. Water samples were placed in coolers on ice in the field, kept at 4^oC, and analyzed in the laboratory within 48 hr of collection. Pore water sampling was implemented once every month from August 2005 to July 2006. Because the piezometers were installed during July 2005 and first sampled in August 2005,, they were sampled until July 2006 in order to have a full year of sampling. Inorganic nitrogen contents (NH_4^+ and NO_3^-+NO_2^-) were measured in pore water using the Lachat ® automated flow injection analyzer (QuikChem ® FIA+ 8000 Series).

Statistical analyses

The data were analyzed using several approaches. Because each site represents a single datum for inter-site comparisons, the sample size for analyses of differences among wetland types is n=2, and there is thus no power for statistical testing among types. Therefore, comparisons among types were made qualitatively, and apparent differences discussed. However, the within-site replicates were used to calculate means and standard errors of site properties, in order to report the variability within each site. The within-plot replicates were also used to analyze relationships among the response variables within each plot; relationships among plots used only the mean value for each plot (n=6) to avoid pseudoreplication issues.

Analyses of data within plots used analysis of variance when assumptions of normality and homoscedasticity were met, and Kruskal-Wallis non-parametric tests when the data and transforms of the data did not meet the assumptions. Post-hoc analyses used the Holm-Sidak post-hoc tests of means among groups (SigmaPlot ver. 11.0). Variables expected to show seasonal variation (soil moisture, extractable NH4, mineralization rate) were analyzed by repeated measures analysis of variance for each plot separately (n=3 per month, 12 month time periods) using SAS ver. 9.1 software, and the Huynh-Feldt Epsilon correction for probability estimates.

Results

Monthly sampling of plot soils

Inspection of the overall site means (Table 1) suggests that the two mineral soil wetland types had similar soil properties to each other, whereas the cedar swamps (organic soils) were quite different, with apparently much higher organic matter, moisture, and initial extractable NH_4-N. Apparently higher soil moistures were observed in the M10 plots than the M6 plots. Pine wetland plots were uniformly slightly less acidic than cedar swamps, but two pine hardwood stands appear to differ from each other. It is not possible to ascertain from this study whether differences between the two plots of a given type are significant.

Water levels in the six plots followed similar patterns, with levels in the PW and PH plots always below the ground surface and levels in the cedar swamps at or above the surface for much of the year except late summer through October (Fig. 3). The two cedar swamps appeared to differ from each other (Table 1) in that the M6 site rarely experienced surface water, whereas water levels were above the surface for much of the year in M10. The two PW plots were also apparently different from each other (Table 1), with water levels at the M6 plot consistently lower than those at the M10 plot. However the two PH plots were very similar to each other (Table 1). As in previous studies (Zampella et al. 2001), water levels among types within each site were highly correlated with each other (Table 2). Correlations between the replicate types, however, were lower for the two sets of PW and CS plots than for the two PH plots (Table 2). Water levels in the M6 PW plots were notably variable during the spring and summer, possibly because of the partially-permeable clay lens encountered during the installation of the piezometers at this location.

Soils of the two cedar swamps stayed markedly wetter than the soils of the PW and PH wetlands throughout the study. Indeed, the four PW and PH plots all appeared to have similar levels of soil moisture (Fig. 4). There were high levels of variation among the three replicates of each monthly measurement within all plots, as is evident from the error bars for each month's value in Fig. 4. In repeated measures analyses of variance of moisture contents within each plot, the soils from M6PW showed significant seasonal variation ($F_{11,22}=10.49$, $P<0.001$); August and September soils were significantly drier than the preceding or following months, and the mid-winter soils (January and February) were significant wetter than the soils in March and April (Fig. 4). A similar pattern was found in the M6PH site (significant variation among months, $F_{11,22}=5.66$, $P<0.0001$). Seasonal patterns of soil moisture in the other sites were obscured by high variability among replicates within each plot, and no significant temporal variation could be detected. Although local precipitation (measured at the Silas Little Research Station) showed approximately similar temporal patterns as soil moisture (Fig. 5), there were no significant regression relationships within plots between the measured soil moisture and the monthly precipitation.

Soil moisture varied with water tables in each of the mineral soil wetland plots (Table 3). Simple linear regression relationships between the estimated water levels on the date of sampling and soil moisture were stronger (higher r^2 values) in the M10 site than the M6 site, for which the water tables could explain only about half of the variation in soil moisture (Table 3). In the cedar swamps, relationships between water tables and soil moisture were notably weaker, with a weak relationship found for one plot (M6CS) but no relationship found for the other (M10CS; Table 3). Finally, a comparison of the graphs of the annual patterns of soil moisture in all plots (Fig. 4) with the graphs of observed water table elevations (Fig. 3) suggest that variation in soil moisture, especially in the cedar swamps, is damped compared to the magnitude of annual variation in water table elevation, and indeed the coefficients of variation of water table levels are about twice as high as the CVs of soil moisture for each plot (with the exception of M6PW) (Fig. 6).

Soil moisture provided a moderately useful explanatory variable for the organic matter content of the soil (Table 3). These relationships were stronger for the cedar swamp wetlands than the mineral soil wetlands, for which r^2 values were all less than 30%. Thus, our posited link between moisture as a controlling factor the amount of organic matter in the soils is only partially supported.

Extractable NH_4 was two to three times higher in the cedar swamps than the mineral soil wetlands throughout the study period, and levels appeared to be higher in the M6 than the M10 sites (Fig. 7) with maximum values observed during the winter. Significant variation over time within plots was observed in both PW plots (M10PW: $F_{11,22}=5.64$, $p=0.043$; M6PW: $F_{11,22}=5.24$, $p<0.001$) and in the M10PH plot ($F_{11,22}=9.34$, $p<0.001$), but in neither of the CS plots. Plots with significant seasonal variation showed trends of higher concentrations during the winter, presumably associated with a lack of plant uptake. Regressions of extractable NH_4 concentrations against soil moisture, conducted separately for the each plot (Table 4), showed that although most regressions were statistically significant, they had little explanatory power (all $r_{adj}^2 < 0.26$) and the slopes for all relationships were also very low. There were no significant relationships within plots between extractable NH_4 and soil organic matter. A comparison of plots using an overall mean value for each site (Fig. 8A; n=36 per site, n=6 for the regression) clearly shows that although there is a highly significant regression relationship with high predictive value, the relationship results simply from the strong contrast between the two CS sites (high soil moisture) and the four mineral-soil sites (lower soil moisture), which cluster closely together.

In almost all samples, extractable NO_3 was near or below the detection limit (0.01 mg N L^{-1}) of our method. Because samples at the detection limit are very subject to measurement error, these low values probably explain the few slightly higher levels recorded for individual samples (Fig. 9). Without more extensive sampling, using a more sensitive method, we cannot say definitively that no nitrate was present. However, even if the positive values displayed in Fig. 9 are reliable, the absolute concentrations are extremely low.

There was little variation in net N mineralization rate with time in any of the sites (Fig. 10; repeated measures analyses within plots only significant for the M10PW site ($F_{11,22}=7.43$, P<0.001 and the M6CS site ($F_{11,22} = 3.36$, p=0.007). The plots in Fig. 10 suggest that the M6 sites have a higher rate of mineralization than the M10 sites in the mineral soil sites, and also suggest that there is little difference between the two types of mineral soil types; however, due to pseudoreplication issues, this could not be tested statistically. The plots also suggest that the two cedar swamps had very similar rates of N mineralization. However, the cedar swamps clearly have much higher levels of N mineralization than do the mineral soil wetlands (on a comparable volumetric basis). In the M10CS plot, N mineralization was much higher in September, when soil moisture and water tables were low, than at other times; this was the only plot in which expected patterns of change in N mineralization with time of year was observed.

Net nitrification was rarely observed in any plot during the sampling year (Fig. 11). Occasional samples showed measureable rates, but when this occurred, it was not uniform among the three samples per plot, leading to very high standard errors of the mean values for each wetland. Curiously, the only departures from the lack of net nitrification occurred in the cedar swamps in September, during a period of drought and lowered water tables and soil moisture, but the two cedars swamps showed opposite patterns: in M10, some of the samples experienced relatively high net nitrification rates, but in M6, there was net consumption (immobilization) of nitrate during this measurement period (Fig. 11). As with the initial concentrations discussed above, these anomalous values are probably the result of all concentrations in the extracts being close to or at our limit of detection. Unlike the M10 CS plot, the soils in M6CS plot did not dry during the August-September period (Fig. 4), which may explain the lack of response of nitrification to the dry period. The occurrence of some nitrification in March 2006 in the M6CS site was not associated with dry soils. Rather, it may have reflected the mineralization of microbial biomass killed during winter freezing events. The variations in nitrification in the mineral soil wetlands (Fig. 11) are even less clearly related to variations in soil moisture, precipitation, or water tables, and differ for the plots in the two sites. The lack of nitrification corresponds well to the lack of observed nitrate in the extractable soil samples (Fig.9).

We explored the relationships of net N mineralization rate to environmental variables by examining relationships within each plot in order to determine whether environmental variation at the plot scale account for within-plot or within-type variation in mineralization (Table 5). The range of the data for each variable, across the year of sampling within plots was for most variables 8->10-fold (i.e., almost an order of magnitude difference between the lowest and the highest values within each dataset), suggesting that there was ample variation within each dataset to allow any relationships to emerge. Surprisingly, N mineralization rates within each plot were poorly correlated with environmental factors. Organic matter was not an important factor within any plot nor did it explain variation among sites (based on site means). Soil moisture variations were a significant factor for only the M6PW plot, and explained only a small amount (13%) of the variation in net N mineralization. Nor was the water table position a useful predictive variable, both for individual sites and for data pooled for all sites. The most important factor emerging from these analyses was the initial extractable NH_4 concentration, which was a significant factor for four of the six plots, and for all sites pooled (Table 5). Notably, the relationship between N mineralization and NH_4 concentrations was opposite in the cedar swamps and the mineral soil wetlands; mineralization rates increase with increasing amounts of extractable NH_4 in the cedar swamps, but decrease with increasing NH_4 in the mineral soil sites.

In the analysis of all sites pooled (n=6 from overall site means), both organic matter extractable NH_4 were excellent predictors (86% and 98% of variation explained, respectively), but as is evident in Fig. 8B and C, the strength of the statistical relationships result from the large difference between the four mineral-soil plots and the two organic-soil plots. These analyses thus suggested that variations in environmental conditions within plots provides only poor explanations for the variations in N mineralization rates, but that there are striking differences between the mineral-soil plots and the organic soil plots.

In order to explore patterns with respect to soil moisture and nitrogen mineralization across wetland types, the data were also analyzed by pooling data from all plots within each month (Table 6). Within a given month, and across the entire set of wetlands, there were significant relationships between mineralization and moisture, particularly during the growing season. Significance largely reflected the substantial differences between the CS plot values and the mineral-soil plot values. Soil moisture explained differences in N mineralization across the set of plots best during the summer (highest r^2 values). However, during September, when soil moisture values were at their lowest values, the regression relationship explained less of the variance (r^2=0.24), although the slope of the relationship was large, suggesting a large change in N mineralization rate per unit change of soil moisture across the wetlands. Conversely, rates in October were strongly related to soil moisture (r^2=0.69), but with a low slope indicating that there are relatively small changes in mineralization rate with differences in moisture during the late autumn. Notably, these regression relationships show that net N mineralization *increases* with increasing moisture content, suggesting that N mineralization rates may be more limited by dry conditions in the mineral soil sites than by saturation in the any of the sites. Net N mineralization rates across the pooled set of samples were not related to water table depths (although a regression analyses was significant, F=8.78, p=0.01, the r^2=0.04 (n=72) suggests statistical but not ecological significance).

Survey of N Mineralization in Cedar Swamps

Table 7 lists the plots, the net N mineralization rate and the nitrification rate measured for each plot. No nitrification was observed in any plot. There was surprisingly little mineralization in any plot other than M10C. As the personnel and methodologies used for this study and for the rest of the N mineralization study were the same, we could not attribute this result to methodological problems. As with the year-long intensive study at M6 and M10, we found no relationship between soil moisture and N mineralization or nitrification across this population of sites (Fig. 12a); when the higher values of mineralization observed in the M6C and M10C sites were excluded from the analyses (Fig. 12b), a marginally significant negative relationship was observed. However, the very low r^2 value and the equally low value of the slope coefficient imply that there is little effect of moisture changes on net mineralization in cedar swamp soils over a large range of possible moisture contents.

Porewater Nitrogen

Figure 13 shows the results of a year of monitoring the shallow groundwater. Concentrations of both ammonium and nitrate-N were invariably very low, less than 1 mg/L. The slightly elevated mean concentrations in the M10-pine hardwood samples in September and November 2005 and the M10 cedar swamp sample in February2006 nevertheless have large standard errors, reflecting the fact that there was a high degree of variability among the three piezometer samples per plot. The slightly higher concentration of nitrate in M10 in September in all plots does not correspond well with the extractable nitrate concentrations measured during this time, and also have high standard errors, indicating high variability among replicate piezometers at each plot. Using the data from August 2005 to April 2006, and pooling all data over the different plots and months during which both porewater and soil nitrate concentrations were measured, the correlation coefficient for these data (nitrate in porewater and nitrate in soil) is 0.12, a non-significant result. Indeed, on the date on which the highest porewater concentrations were observed (February 2006, M6C), no nitrate was recorded in the soil, and no nitrification was observed. The very low concentrations of both nitrate and ammonium observed, and the high variability among samples, suggests that 1) there is little connection of these values to processes in the overlying soil, 2) positive observations of nitrate may reflect the fact that the observed values are all very close to the limit of detection. Therefore, under the field conditions observed, it is likely that any nitrate in shallow ground water does not originate from mineralization processes in the overlying soil. Because the weather remained normally wet, we did not have the opportunity to observe porewater concentrations during a period of more extended drought.

Discussion

This study addressed the differences in soil nitrogen mineralization among three types of forested wetland of the New Jersey Pinelands, and the correlated environmental factors that may control N mineralization rates within these sites. The results did not support our conceptual model that there is a tight relationship between the measured environmental factors and net N mineralization rates. Surprisingly, the nitrogen dynamics in all of the plots were poorly related to within-plot in either soil moisture or organic matter content. The main predictor of N mineralization rates is the amount of extractable NH_4 in the soils, rather than either the water table depth or soil moisture. Two of the wetland types, the pine wetlands and pine hardwoods, are located on hydric mineral soils, whereas the third type, Atlantic white-cedar swamps, is located on histosols, and the three types of wetlands occur in close proximity to each other along drainage catenas throughout this region. Our results demonstrate that the two types of wetland on mineral soils were quite similar to each other in soil properties and nitrogen dynamics, despite significant differences in water table depths, but quite different from the cedar swamps. Thus, the main differences were between sites on these different soil types, rather than between different positions on the water table gradient.

The patterns of relationship between soil moisture and water tables suggest that in addition to the absolute water table levels, several other factors affect soil moisture. First, the presence of partially confining clay lenses in the profile may disconnect regional water tables and surface soil moisture levels. The M6 PW and PH plots had notably poor relationships between water tables and soil moisture (Table 3), which may reflect the observed presence of lenses at these plots. Indeed, the partial disconnect between the surface soil and the water table may have contributed to the seasonal soil moisture variation documented for the M6 mineral soil plots, but not the M10 mineral soil plots. Second, the high water-holding capacity of organic soils in the CS type (Mitsch and Gosselink 2000, Rabenhorst and Swanson 2000), may help insulate these stands from the effects of seasonally lowered water tables. Soil moistures in the cedar swamps were saturated or near-saturated most of the time, with little seasonal variation, (Fig. 3), despite the drop in water tables in August through September (Fig. 2), whereas decreases in soil moisture during these months were evident in the mineral soil wetlands. Third, the cedar swamps are also notable for the dense shade cast by the tree and shrub canopy (Little 1958), and this shading may reduce evaporation from the soil surface and thus contribute to the ability of the soils to retain moisture from both precipitation and ground water discharge sources. These results suggest that short-term (1-2 month) decreases in water tables will have little effect on soil properties of the cedar swamps; effects of long-term (permanent) water table decreases or long-term decreases below the minimum depths recorded during this study (about 40 cm below the ground surface) cannot be estimated from these data.

The importance of periodic precipitation events, even during seasonal water table declines, may be an important factor in divorcing soil moisture levels from water table dynamics. Even small rain events during a dry period may be sufficient to keep the surface soil layers moist. In order to determine the frequency of completely rain-free periods in the region, a record of daily precipitation amounts was obtained from the Utah Climate Center (http://climate.usurf.usu.edu/products/data.php) for the station at Indian Mills (ID number 284229, approximately 20 km from the McDonald's Branch watershed) for the period May 3, 1901 – May 3, 2009 (24,641 days, 7,297 episodes of precipitation-free consecutive days). The distribution of episodes is shown in Fig. 14. This analysis shows that >99% of precipitation-free periods are less than 14 days long, and 92% are less than one week long. The median rain-free period over this precipitation record is 4 days. Interestingly, the few very long periods of complete drought (arbitrarily defined here as > 25 days with no precipitation) occurred during October to November (with one exception in 1995, when the drought occurred during August to September). The late fall is a time of very reduced evapotranspiration, implying that even during the rare extended droughts, water table drawdowns would not be exacerbated by transpirative demands for water. Thus, the precipitation record suggests that most of the time, surface soil moisture is influenced as much by precipitation as by the water table, and that precipitation may maintain soil moisture during seasonal water table drawdowns.

Nitrogen dynamics in these wetlands are characterized by the near-absence of nitrification under most conditions. All observations of nitrate concentrations above the detection limit (0.01 mg N L^{-1}) in extracts of soil samples and in pore water samples were isolated occurrences within the set of three replicate samples for any given date. While we cannot determine from our data whether these values represent measurement error or true, albeit rare, nitrate production in the sediments, the high variability and the infrequency of these events suggest that under the conditions of this study, nitrate production does not occur, even during summer drawdowns of the water table. In previous studies of N mineralization in these soils (Poovaradom et al. 1988a, 1988b, Sangemeswaran 1995), extractable nitrate and nitrification were also seldom observed, and similar results have been found in other bogs and fens (Fellman and D'Amore 2007). Indeed, in a laboratory study of the Atsion soil that tested a wide range of temperature and moisture conditions, nitrate was almost never observed (Poovaradom et al. 1988b). Sangemeswaran (1995) observed one pulse of nitrate production in two cedar swamps within the same region as the study sites in this study; this pulse was observed in early spring, following unusually deep and prolonged soil freezing, and thus may have been an unusual response to microbial mortality following the freeze-thaw event (Paul 2007); however, we note that the role of freeze-thaw events in producing pulses of mineral N remain uncertain (Henry 2007). However, Sangeswaram's samples were taken from the more aerobic hummock soils (Ehrenfeld 1995), in contrast to the hollows in which our samples were taken. Although there is some evidence that repeated freeze-thaw events promote N mineralization and nitrification, even in acidic forest soils (Joseph and Henry 2008), field studies in snow-removal experiments have not confirmed the production of leachable NO_3 in response to freezing (Hentschel et al. 2009).

Low pH is a well-known inhibitory factor for ammonia-oxidizing bacteria (Paul 2007), and thus the extremely low pH of all soils in this study (<4) is likely to be the main controlling factor preventing rapid response to fluctuations in soil moisture. Nitrification has been observed in histosols subject to prolonged, but not intermittent drainage (e.g., Hanlon et al. 1997, Prevost et al. 1999, Tarre and Green 2004). In these studies, however, the drained organic soils are also associated with somewhat higher pH values (>4) and higher mineral content than those studied here. Our previous laboratory studies of the effects of soil moisture on nitrification in soils of these study plots (Yu and Ehrenfeld 2009) similarly found that prolonged periods (months) of unsaturated conditions resulted in measurable nitrification, but that bi-weekly moisture fluctuations between saturated and unsaturated conditions did not permit nitrification. Thus, both our laboratory and field studies (see Part I report; Yu and Ehrenfeld 2009) suggest that only with prolonged periods (months) of unsaturated conditions will net nitrification be observed in these very acidic soils. Despite the drawdown in the water tables of the plots from August to October (Fig. 3), there was relatively little change in soil moisture, and not enough to reach the condition of prolonged soil aeration that the laboratory study showed is necessary for nitrification to begin.

Net mineralization rates and extractable inorganic N (almost entirely NH_4-N) were much higher in the organic soils than in the mineral soils (Table 1, Figs 6, 8, 9). However, within each plot, both extractable N concentrations and mineralization rates were unrelated to the organic matter content of the soil, despite nearly 10-fold variations among the replicate samples, and contrary to expectation. The one statistically significant relationship (for the M6CS soils), explained only 11% of the variation. N mineralization rates were also poorly related to soil moisture contents within wetlands (Table 4). Indeed, only one site (M6PW) had a significant relationship, and again, the relationship explained a small amount (13%) of the variation. In parallel with these results, the survey of other cedar swamps also showed no relationship between N mineralization and soil moisture. N mineralization rates in the survey wetlands were notably low, all < 1 mg N kg^{-1} day^{-1}), while rates in the two intensive-study wetlands (M6CS and M10CS) were higher (1-3 mg N kg^{-1} day^{-1}). It is likely that the lack of relationship in the survey wetlands may reflect both the small absolute values and the small range of variation in the rates. Agehara and Warncke (2005) found that nitrogen mineralized from organic materials did not vary with soil moisture in some types of organic materials. Poovaradom et al. (1988b) tested the role of soil moisture in laboratory incubations of organic horizons of the Atsion soil, and found that moisture had little effect on rates or on calculated mineralizable N until the soil was at very low moisture levels (-1.5 MPa), although in the Lakehurst sand, N mineralization did decrease with moisture at a level of -0.1 MPa. These authors also reported that this pattern was similar to that seen in some muck soils. Drury et al. (2003) showed that in mineral agricultural soils (various types of loams), there is a range of water contents over which N mineralization does not vary, but that it will decrease below some lower limit and above some upper limit. They found that over the range of soils and soil conditions (with and without amendments and compaction), net N production was insensitive to moisture level between 20% WHC and 80% WHC (with variation among soil types). While these values cannot be extrapolated directly to the sandy mineral soils and organic soils in this study, the concept that N mineralization is insensitive to variations in moisture content over a large range of moisture contents is probably applicable. Fellman and D'Amore (2007), in a comparative study of N mineralization in Alaskan bogs, fens, and riparian wetlands also found that the rates were not well correlated with soil properties, other than pH (which varied over a larger range in their sample of sites than in ours). These results are perhaps not surprising in that N mineralization can occur over a wide range of oxygen saturation and moisture content; indeed, production of NH4 under completely flooded conditions is a commonly used assay for potential N mineralization rate (Robertson et al. 1999). In conclusion, the results do not support our conceptual model that water tables, soil moisture, N mineralization and nitrification rates are closely related and that therefore water table dynamics can be used to predict either net N mineralization or the potential for leaching of nitrate during water table drawdowns.

Other factors may have also contributed to the lack of nitrification during the seasonal drawdown. First, the slow growth rates of ammonia oxidizing bacteria (Paul 2007), may prevent these microbes from responding to relatively short periods of oxic conditions, especially at low pH. Secondly, the maintenance of high soil moisture in the soils during the seasonal drawdown, especially in the cedar swamps, suggests that sufficient saturated microsites remained within the soil to remove any nitrate by denitrification processes, even under conditions of reduced moisture. The importance of microsites in promoting denitrification within partially dry soils is well recognized (Dhondt et al. 2003, Verhoeven et al. 2006, Liu et al. 2007, Pinay et al. 2007). Our data suggest that a drought and drawdown would have to be significantly longer than the event in 2005 covered by our sampling period to remove these microsites and allow nitrate to accumulate in the soil and leach to shallow groundwater.

The low absolute amounts of NH_4-N in the soil, particularly in the mineral soil wetlands, may have also contributed to the high variability and lack of significant patterns of explanation between N cycling variables and environmental variables. Sierra (1996) demonstrated that low initial concentrations of NH_4 can lead to high measurement errors in the *in situ* method of determining field N mineralization rates, because of real differences in NH_4 concentrations between the two adjacent cores used for the initial and incubated measurements (i.e., that the 'initial' concentration in the extracted core is not the same as the concentration in the adjacent core incubated in the ground). Although the two cores used for each monthly measurement in this study were taken within 5 cm of each other or less, we cannot discount the possibility that measurement error from this source was not important in masking relationships between N processes and soil properties.

The failure to demonstrate clear relationships leading from water table fluctuations to variations in N mineralization rates reflects the numerous factors affecting N mineralization rates, as well as the relatively poor connections demonstrated above between water tables and the assumed proximate controls on these rates (soil moisture, organic matter content). N mineralization rates are affected not only by the physical environment (moisture, pH), but also by the characteristics of the soil organic matter (C:N, lignin content, lignin:N, soluble N, and organic carbon fractions (Brierley et al. 2001, Bengtsson et al. 2003, Cookson et al. 2005, Reid et al. 2008), the characteristics of the microbial community (e.g., biomass, respiration rate, ATP content; Bengtsson et al. 2003), and the composition and chemical characteristics of the litter input to the forest floor (Jerabkova et al. 2006). Jerabkova et al. (2006) found that mixed hardwood-conifer stands had higher net N mineralization rates than did pure conifer stands, but we did not find in this field study that the mixed pine-hardwood wetlands differed from the pure pine wetlands. However, Yu and Ehrenfeld (2009), in laboratory incubation studies using soils from the same plots, did find that N mineralization rates in the pine-hardwood wetlands were notably higher than those of the pine wetlands. This contrast in results suggests that environmental factors kept under control in the laboratory incubations had strong effects in the field, sufficient to eliminate potential differences based on the types of litter input. Finally, the lack of a clear seasonal pattern in N mineralization probably reflects the fact that at 5 cm depth, equivalent to the depth of organic layer beneath which we obtained our soil samples, there is little variation in soil temperature, compared to surface soil or air temperatures (Henry 2007), and thus a seasonal signal would be not expected.

The results of the field studies are in line with the results of the fluctuating-moisture incubation experiments conducted in the laboratory (Part I). In those studies, increases in nitrification and nitrate concentrations during two-week periods of drought (30% WHC) were very small in absolute magnitude, and no differences in net nitrate or net mineral N production were observed. Although precipitation was very limited during August and September of the study period (Fig. 5), there were several small rainfall events, and the maximum period without any precipitation was 15 days. The patterns of soil moisture and nitrogen dynamics observed suggest that even small precipitation events are sufficient to prevent extreme reductions in moisture content or increases in nitrification.

Finally, the lack of clear relationships between water table and soil moisture variables and N dynamics within plots or vegetation types, but their presence in significant regressions at the landscape scale among types suggests that changes in N dynamics under possible future conditions of water table drawdowns would most likely result from landscape shifts in community types.   If the complex of soil properties (organic matter content, organic layer thickness, moisture relationships, pH, and water table relationships shift together with the above-ground plant community, then one might expect concomitant shifts in N mineralization rates as observed in this study. Under such a scenario, our results suggest that a shift from cedar swamps to pine hardwood or pine wetland would result in decreased rates of net N mineralization, but only if the community shift was accompanied by a loss of peat and the development of the plant community on a mineral soil material.   However, none of these potential community shifts implies an increase in net nitrification, at least under the hydrological conditions present during this study.

Conclusions

1. Net mineralization in the Pinelands wetlands is much higher in cedar swamps than in either pine wetlands or pine-hardwood wetlands, which are quite similar to each other in all measured properties. This suggests that nitrogen dynamics across the wetland gradient is largely regulated by soil type (mineral vs. organic) rather than differences in water table level.
2. Nitrate is very rarely produced, and when it is present, it is found at very low concentrations and is highly variable in occurrence on any sampling date. While some increases in nitrification and extractable nitrate were observed in some wetlands in association with a period of lowered water tables and reduced soil moisture, the high variability of these observations both within wetlands and between the replicate plots of each wetland type, and the problems associated with concentrations at or near the detection limit of our instrumentation do not support an extrapolation to the possibility of nitrate releases during longer drought events.

3. The absence of nitrate in the shallow ground water, again except for a very few observations that were not uniform within or across plots, further supports the inference that nitrate is not likely to be released during brief (time scale of weeks) droughts, under current conditions of water table dynamics. Although higher concentrations were observed in a few samples during a late-summer drought, the absolute values remained close to the limit of detection, and therefore may reflect measurement error as much as increased nitrate production during short-living periods of soil drying.
4. Analysis of a long-term precipitation record from the Pinelands region suggests that rain-free periods long enough to support a potential shift to nitrifying conditions are extremely infrequent under current climate conditions.
5. The poor relationship between water table location and soil moisture in the two cedar swamps under current conditions suggests that a combination of precipitation and the water-holding capacity of saprist peat have as large an effect on soil moisture in these wetlands as does the water table. Soil moisture in the mineral soil wetlands (PW and PH) are more tightly linked to water table dynamics.
6. The results are consistent with the fluctuating-moisture laboratory incubations. This suggests that increases in nitrate production sufficient to affect shallow groundwater and potentially surface water are only likely to result if soil moisture is decreased over long periods of time (at least months, as in the constant-moisture incubations of the laboratory studies).
7. The surprisingly poor relationships between soil properties and N mineralization rates within plots suggests that these rates reflect a complex set of relationships among microbial populations and activity, soil properties, and water tables, and cannot simply and directly be related to changes in water table dynamics.
8. Changes in nitrogen dynamics may accompany regional and prolonged water table drawdowns if they result in significant changes to plant community distributions, in parallel with shifts in associated soil properties.

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List of Tables and Figures

Tables

Figures

Figure 1. Conceptual model of controls among water tables, soil properties and N mineralization rates. Arrows indicate potential pathways of effect (e.g., soil moisture may affect N mineralization rate), without implication about the direction of those effects (e.g., increases in soil moisture could potentially either increase or decrease N mineralization rates.)

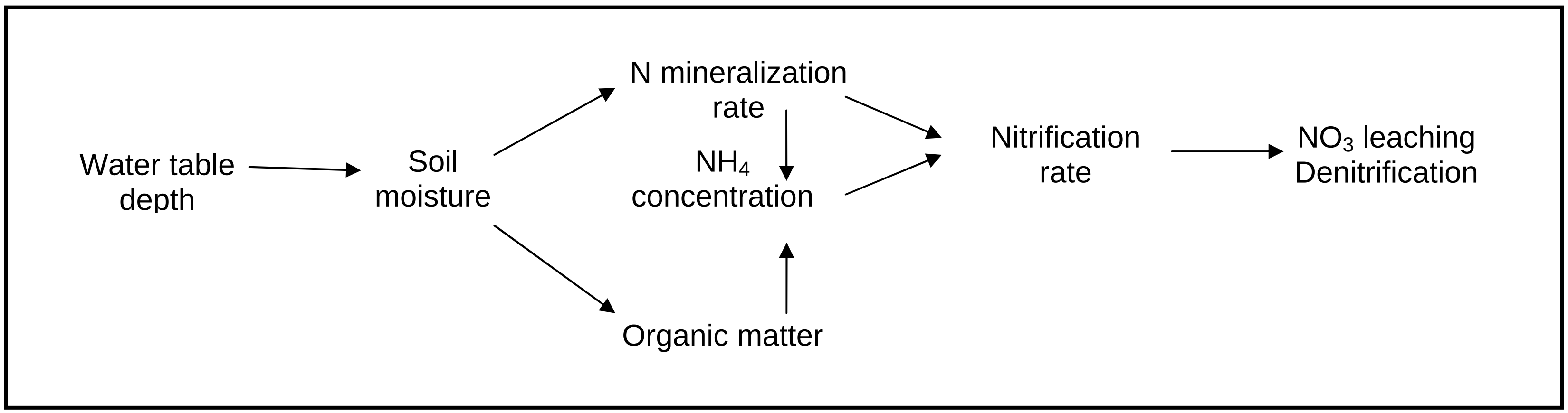


Figure 2 Location of the study plots in the Pinelands National Reserve (shaded area within the map of New Jersey), distribution of plots in the McDonald's Branch upper basin, and arrangement of sampling locations around each plot. Soils were sequentially sampled around the perimeter of the plot over the 12 month study period.

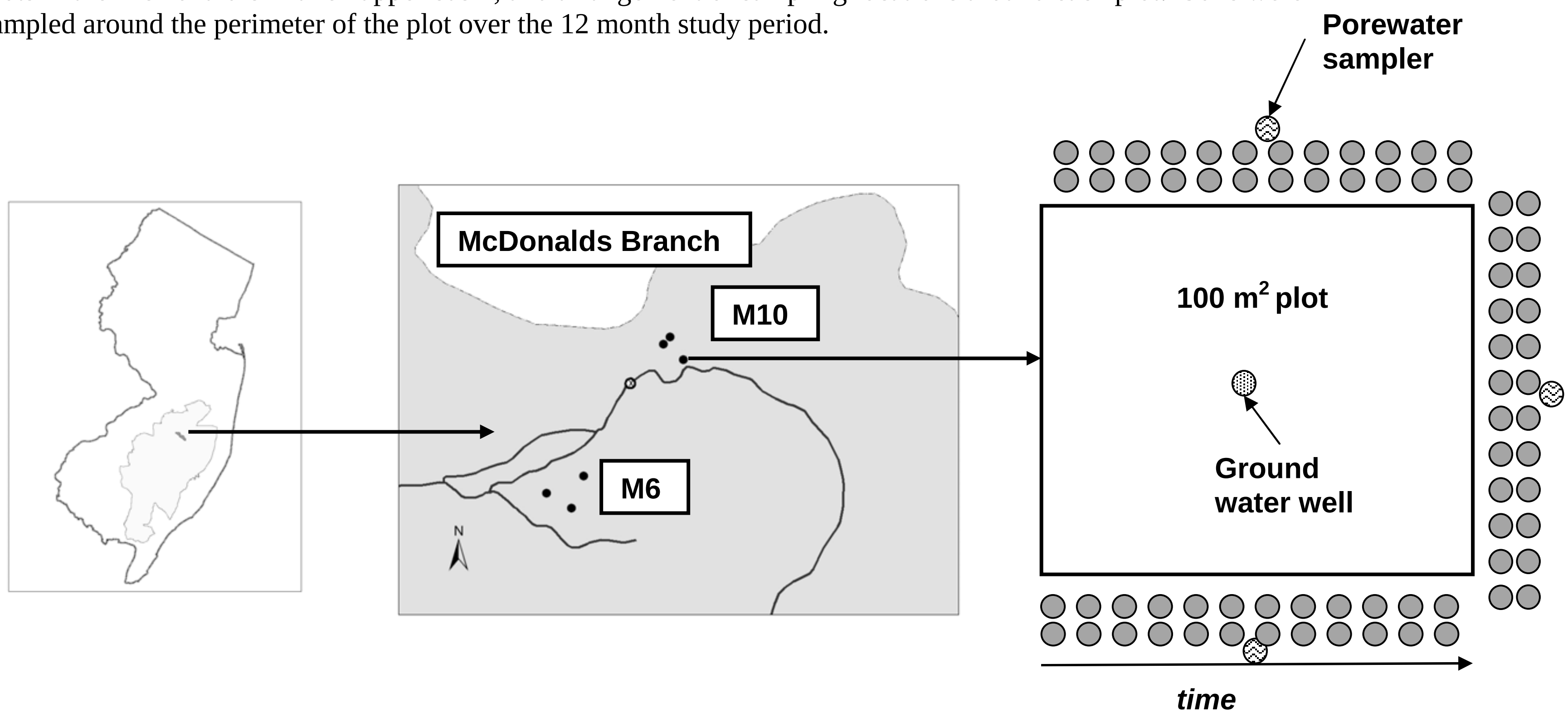


Fig. 3. Ground water levels measured in partial-series wells within each study plot. The dotted line indicates the ground surface level (bottom of hollows in the cedar swamps), and the y axis is in centimeters below the surface. Plotted values are the observed values recorded between 14 March 2005 and 1 May 2006.

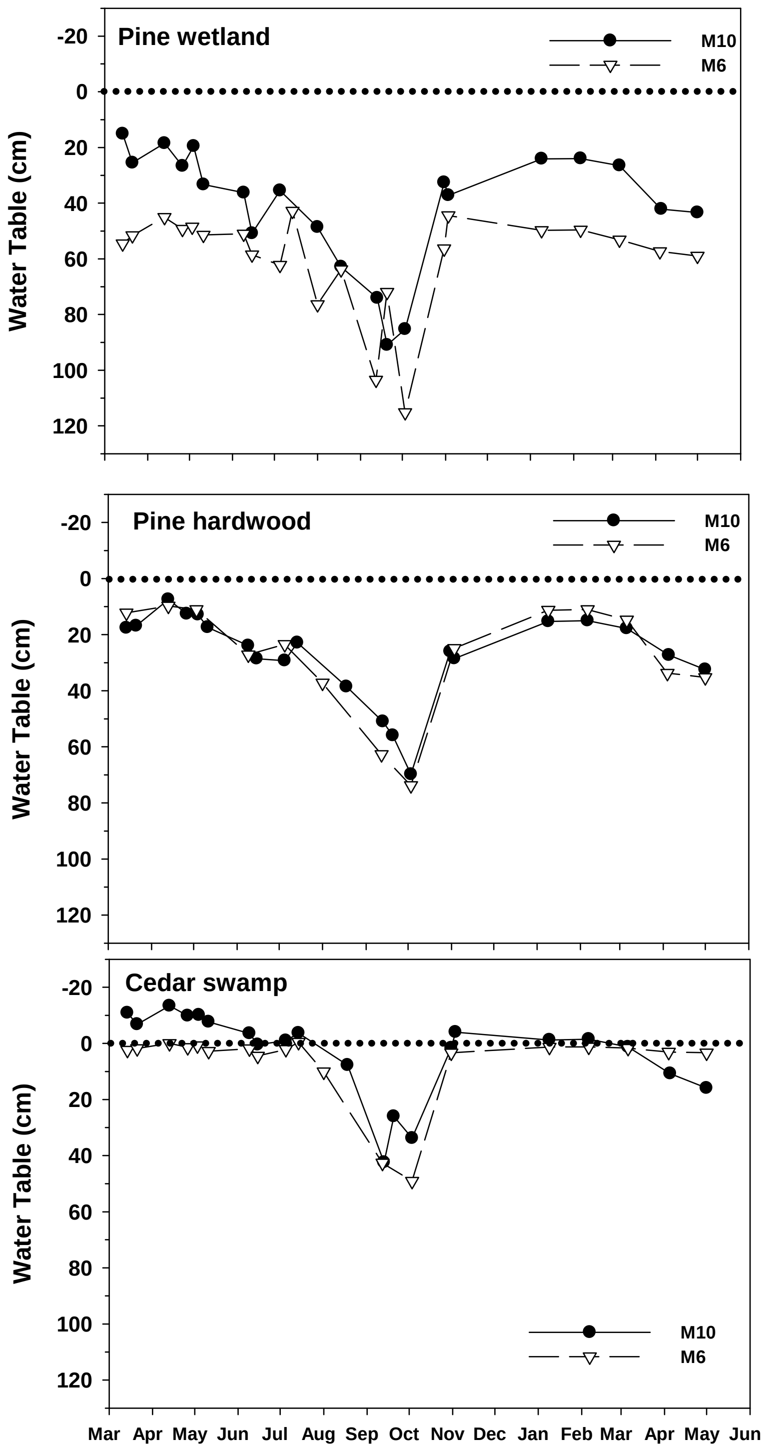


Figure 4. Soil moisture dynamics in field plots during the year of sampling (units of g H_2O 100 cm^{-3} soil; mean ± st. error, n=3). Note the difference in y-axis scale between the CS and the mineral soil wetland types.

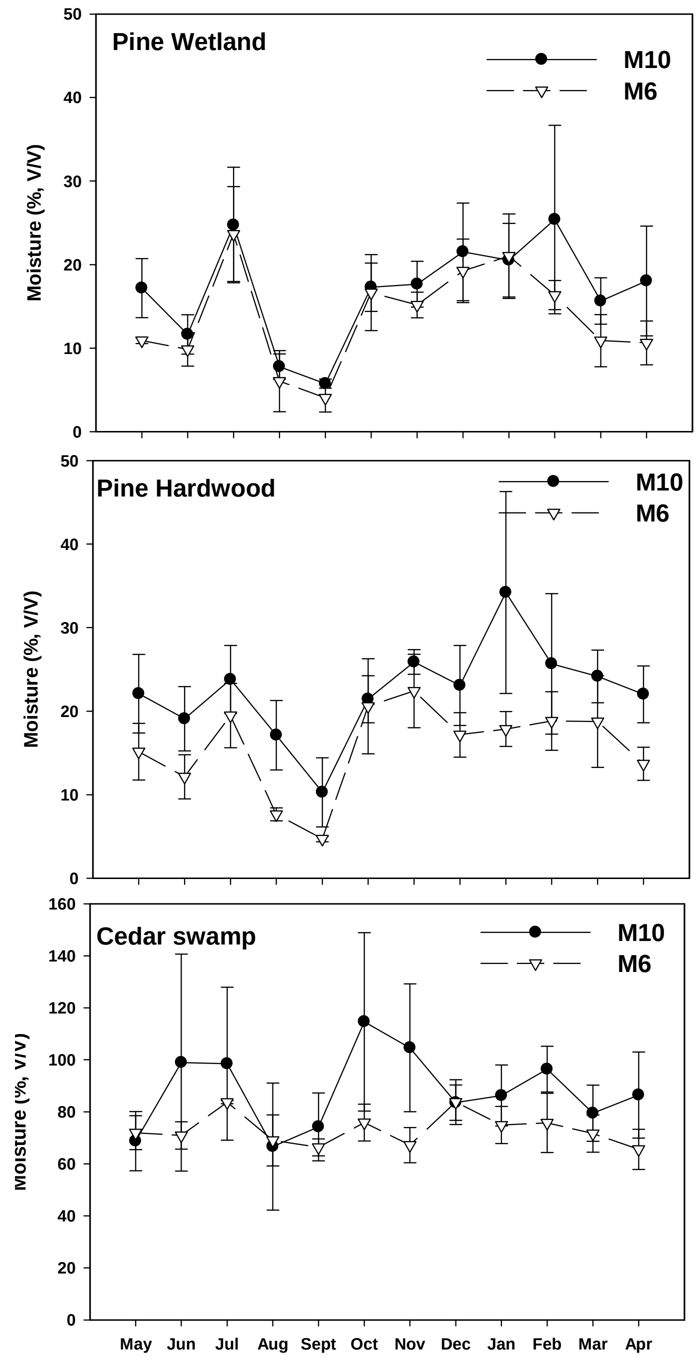


Fig. 5. Mean monthly air temperatures and daily precipitation during the study period. Daily values recorded at the Silas Little Research Station were averaged for each month (mean ± st. error plotted).

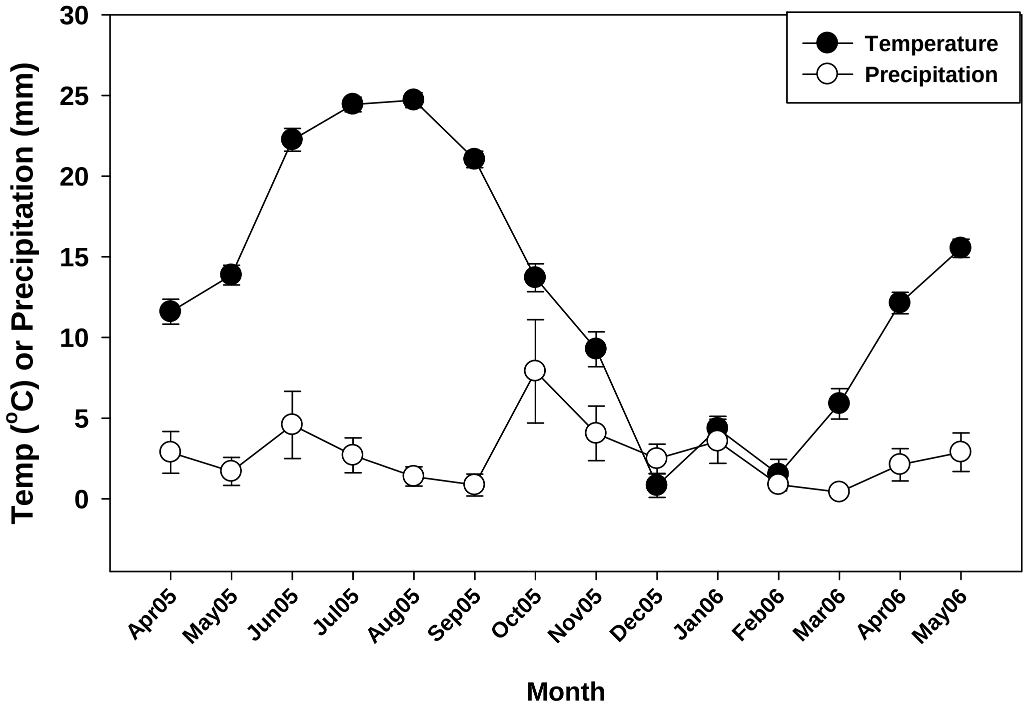


Fig. 6. Coefficients of variation of observed water table levels and soil moisture values, calculated over all measurements (n=39 per site for soil moisture; n=23 to

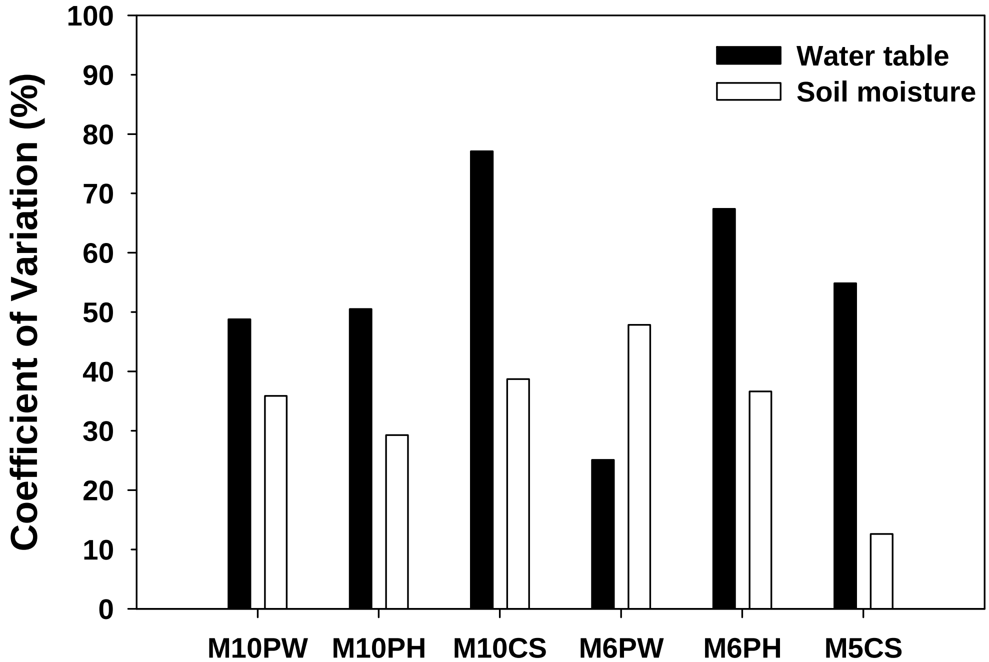


Fig. 7. Monthly concentrations of extractible NH_4 concentrations in the six study plots (mean ± st. error, n=3). Note the difference in scale between the cedar swamps and the mineral soil wetlands

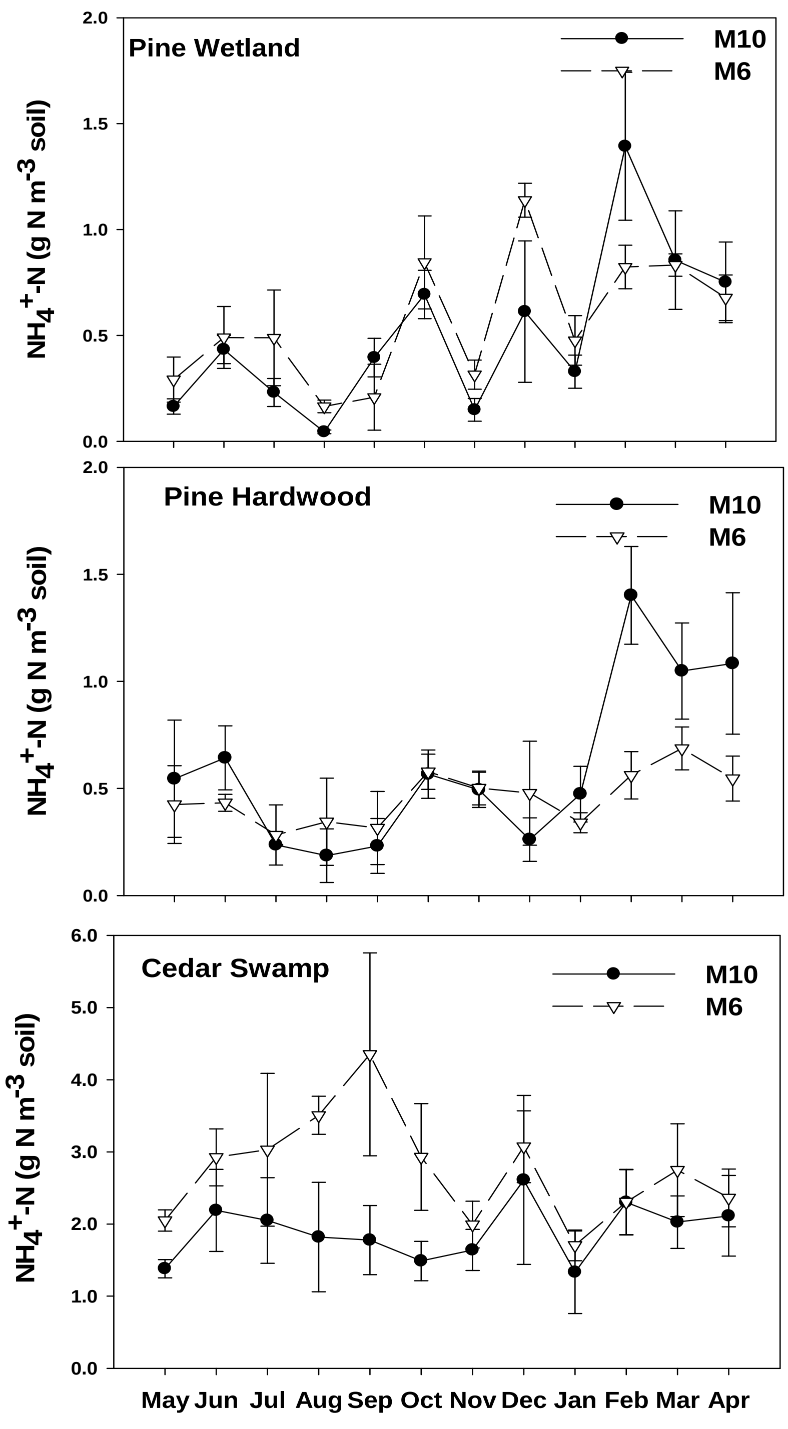
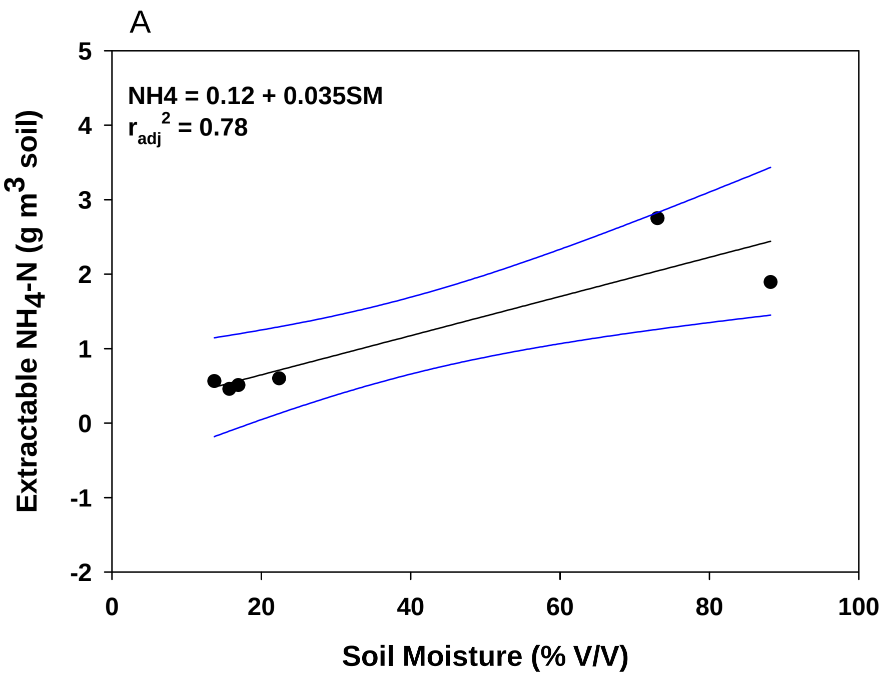
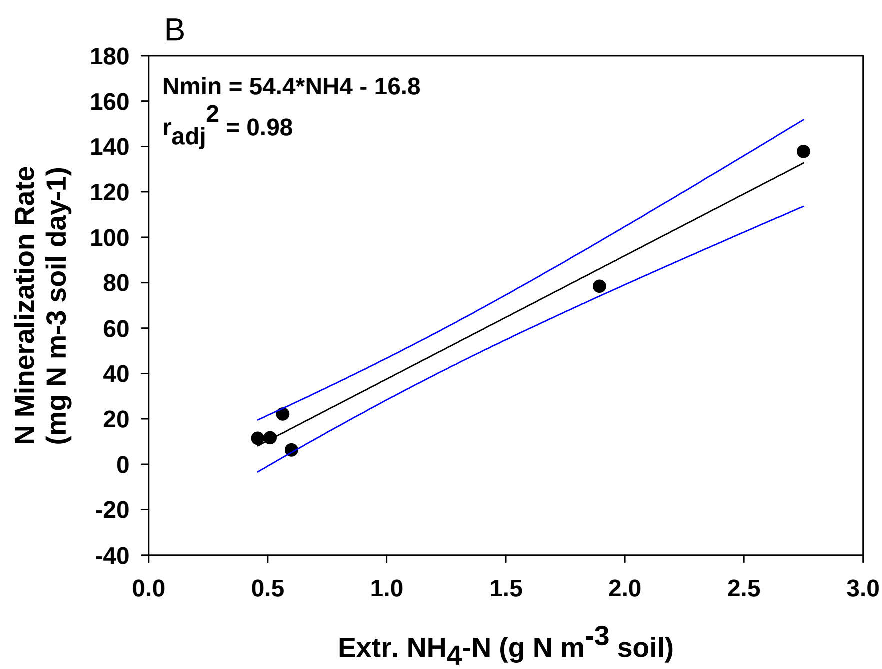
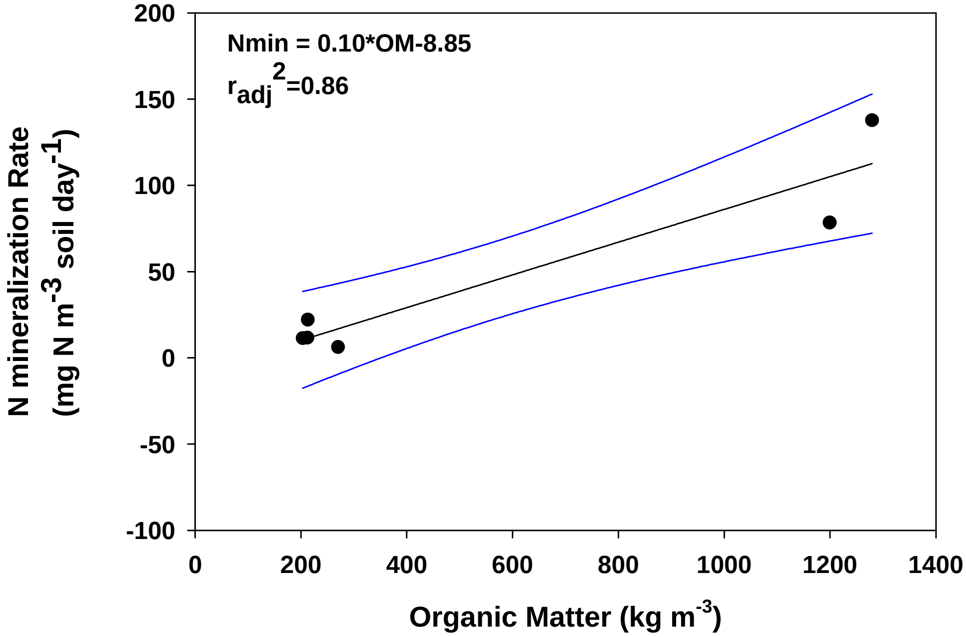


Fig 8. Relationships based on site means (n=6). A) Extractable NH_4 in relation to soil moisture over the wetland gradient. B) N mineralization rate in relation to NH_4, and C) N mineralization rate in relation to soil organic matter. Regressions performed on a single mean value per site (n=6; F=18.47, p=0.002). 95% confidence intervals are shown.





C



Nmin = 0.10*OM-8.85
r_{adj}^2=0.86
N mineralization Rate
(mg N m^{-3} soil day^{-1})
200
150
100
50
0
-50
-100
0
200
400
600
800
1000
1200
1400
Organic Matter (kg m^{-3})

Fig. 9. Extractable nitrate concentrations in the six study plots (mean ± st. error, n=3. Note the difference in scale between the cedar swamps and the mineral soil wetlands.

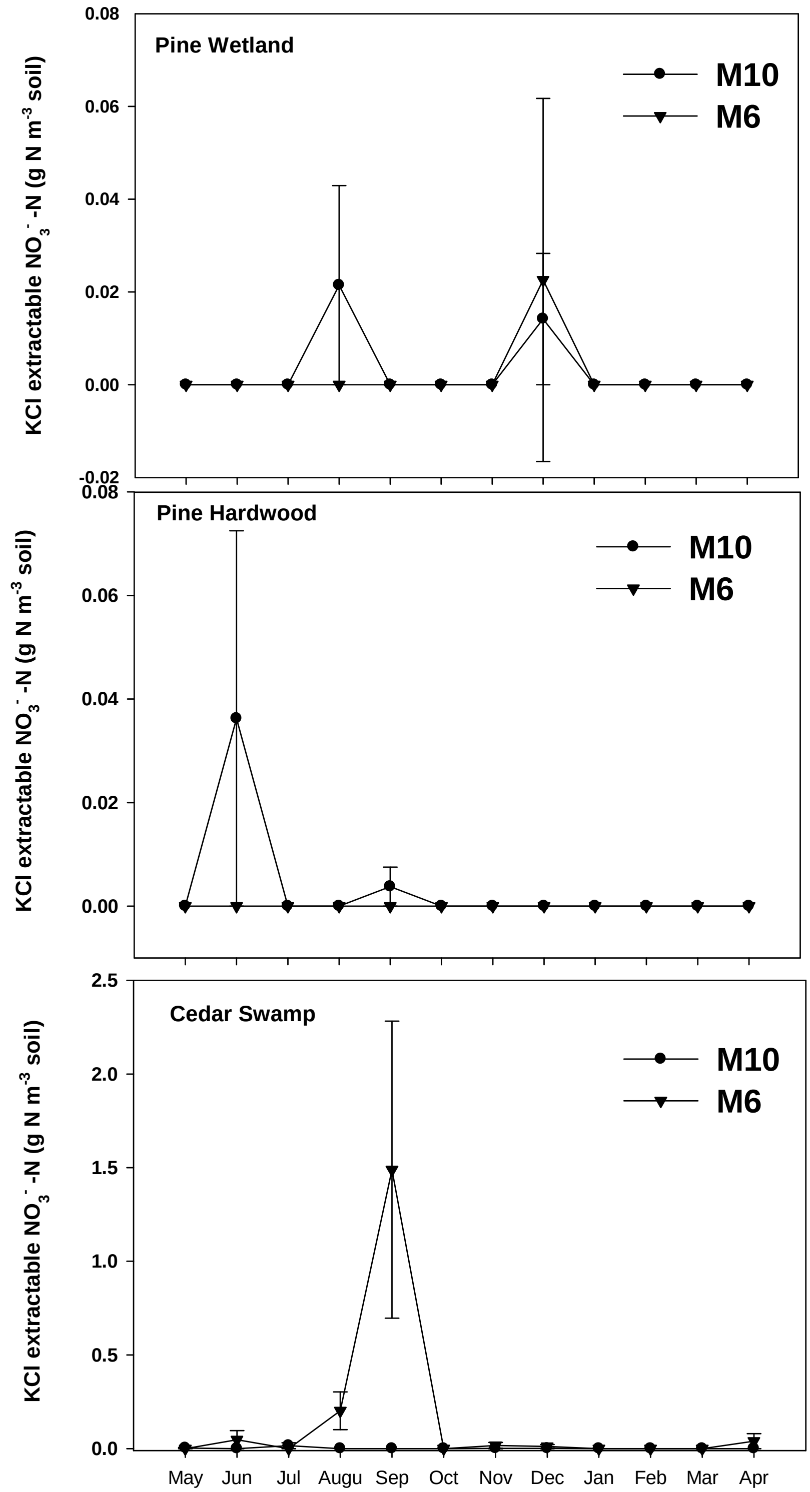


Fig. 10. Patterns of net N mineralization rates (mg N m^{-3} day^{-1}; mean ± st. error, n=3) in the study wetlands. Note the differences in y axis scale between the cedar swamps and the mineral soil wetlands.

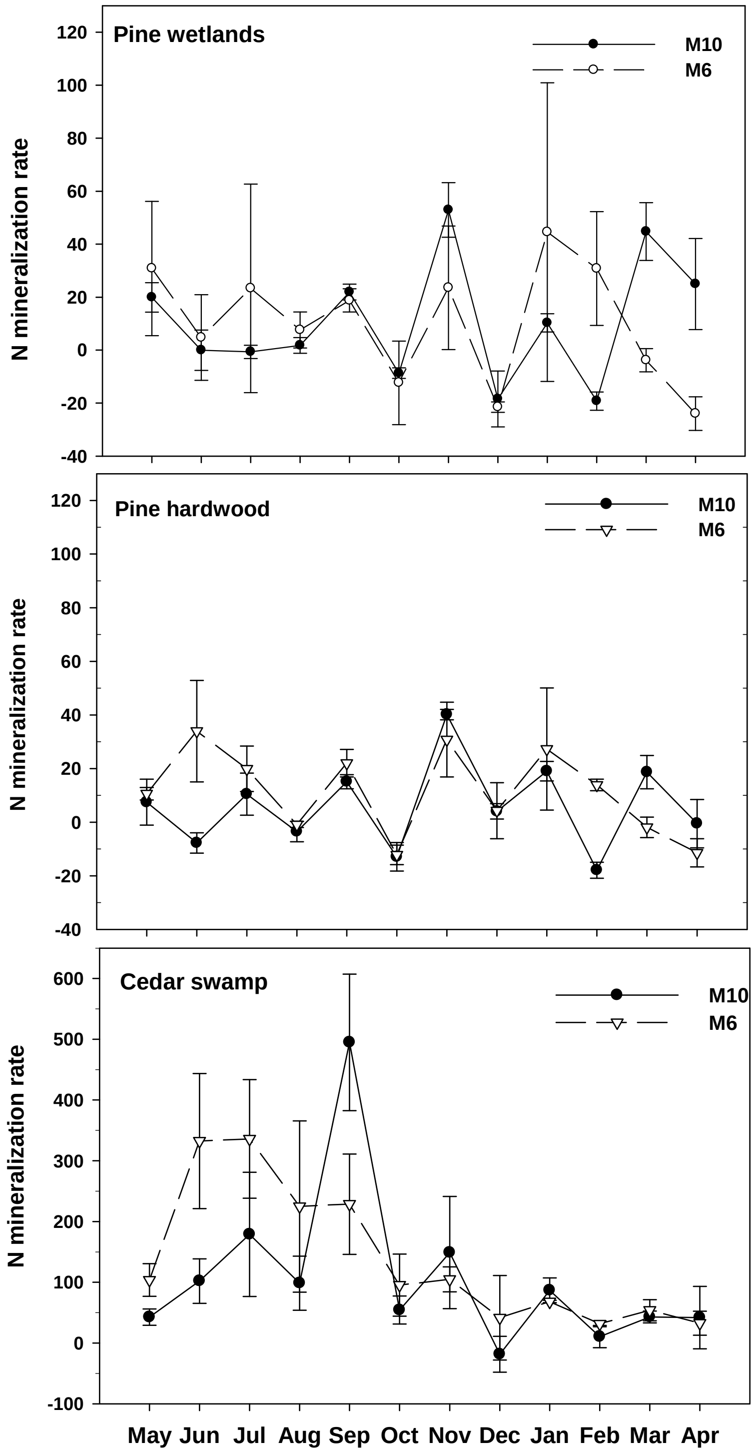


Fig. 11.. Patterns of net nitrification (mg NO_3-N m^{-3} day^{-1}; mean ± st. error, n=3) in the study wetlands. Note the differences in y axis scale between the cedar swamps and the mineral soil wetlands.

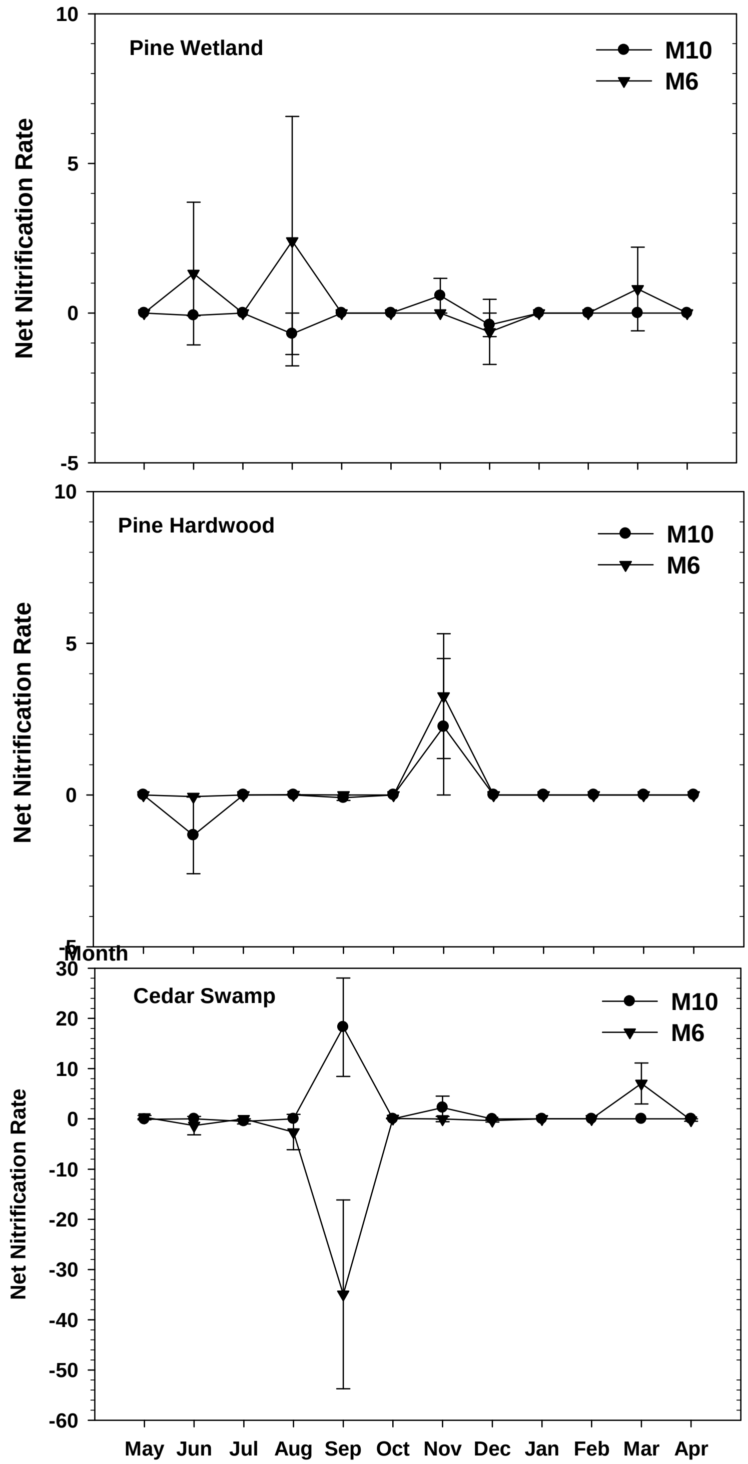
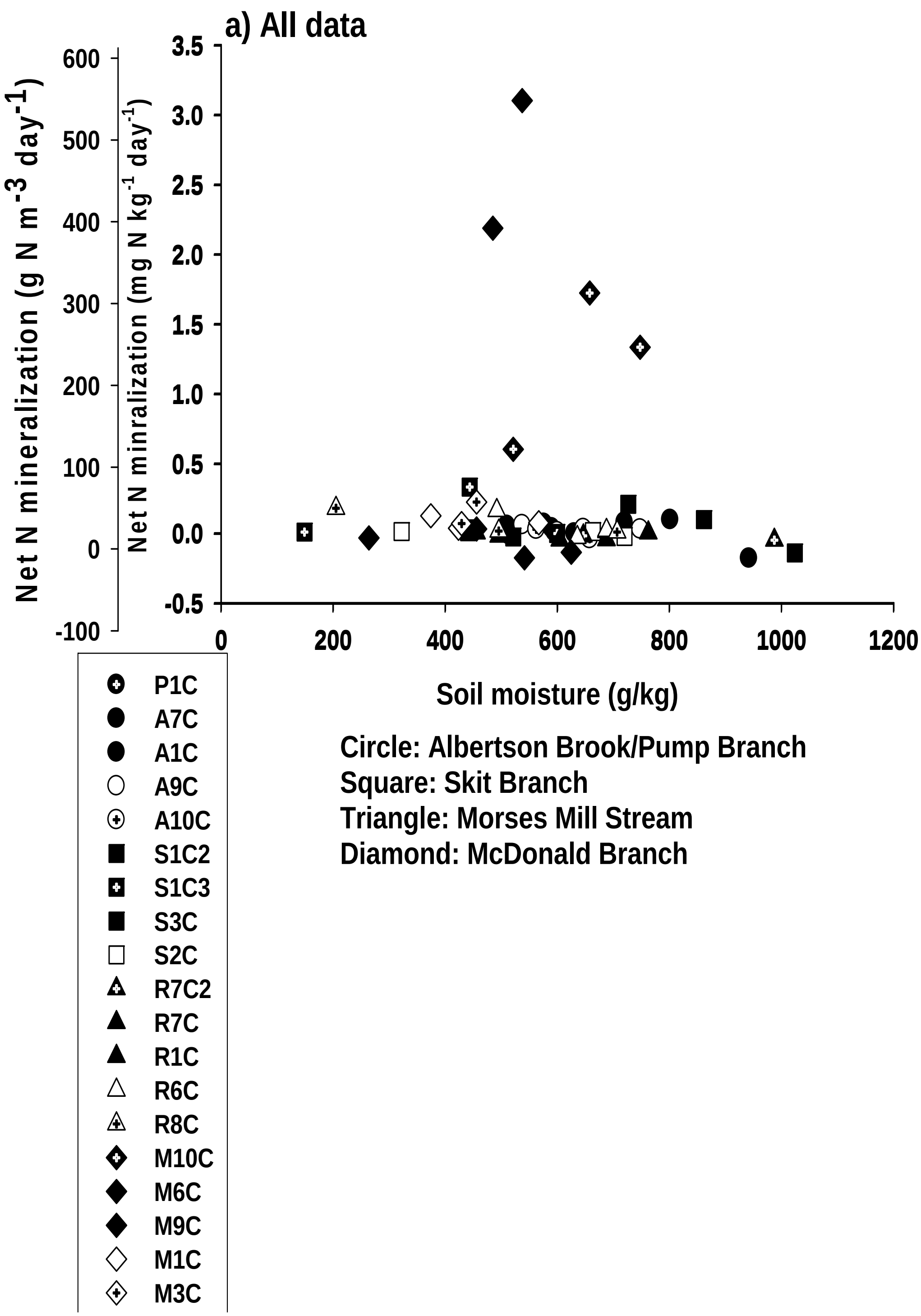
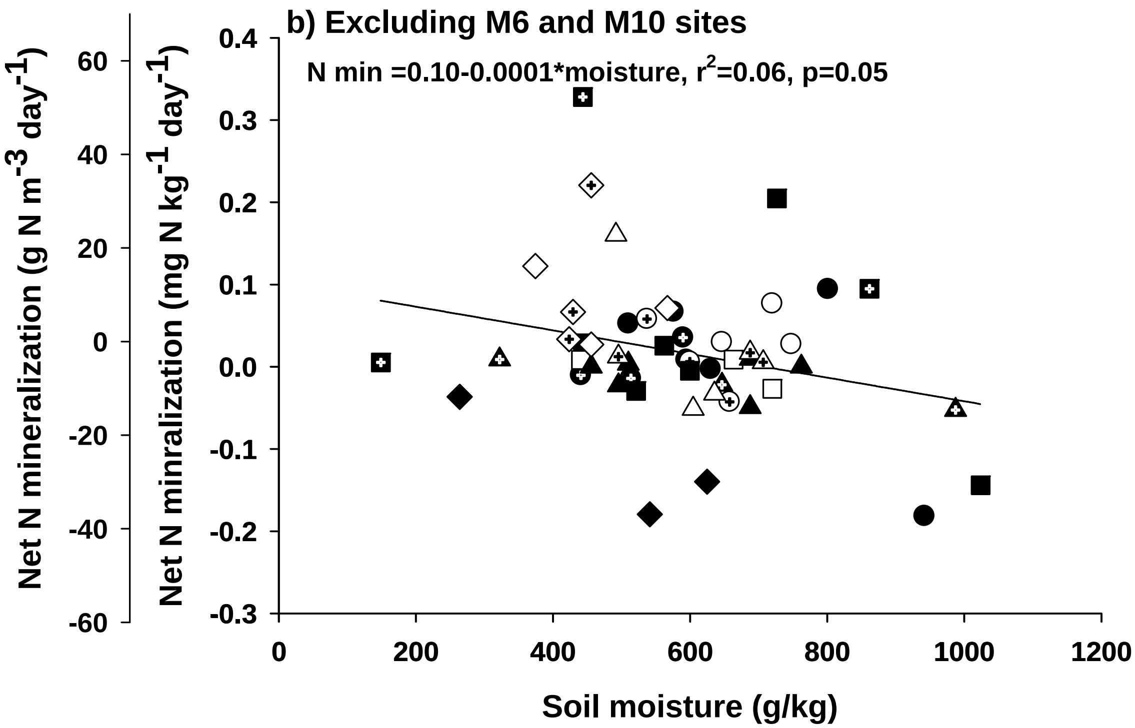


Fig. 12. Net N mineralization rates of cedar swamps sampled across four watersheds. All samples taken in July-August, 2005. ;.





b) Excluding M6 and M10 sites
N min =0.10-0.0001*moisture, r2=0.06, p=0.05
Net N mineralization (g N m-3 day-1)
Net N minralization (mg N kg-1 day-1)
60
40
20
0
-20
-40
-60
0.4
0.3
0.2
0.1
0.0
-0.1
-0.2
-0.3
0
200
400
600
800
1000
1200
Soil moisture (g/kg)

Fig. 13. Monthly patterns of nitrate and ammonium concentrations observed in ground water piezometer samples. Mean ± st. error, n=3 for each observation.

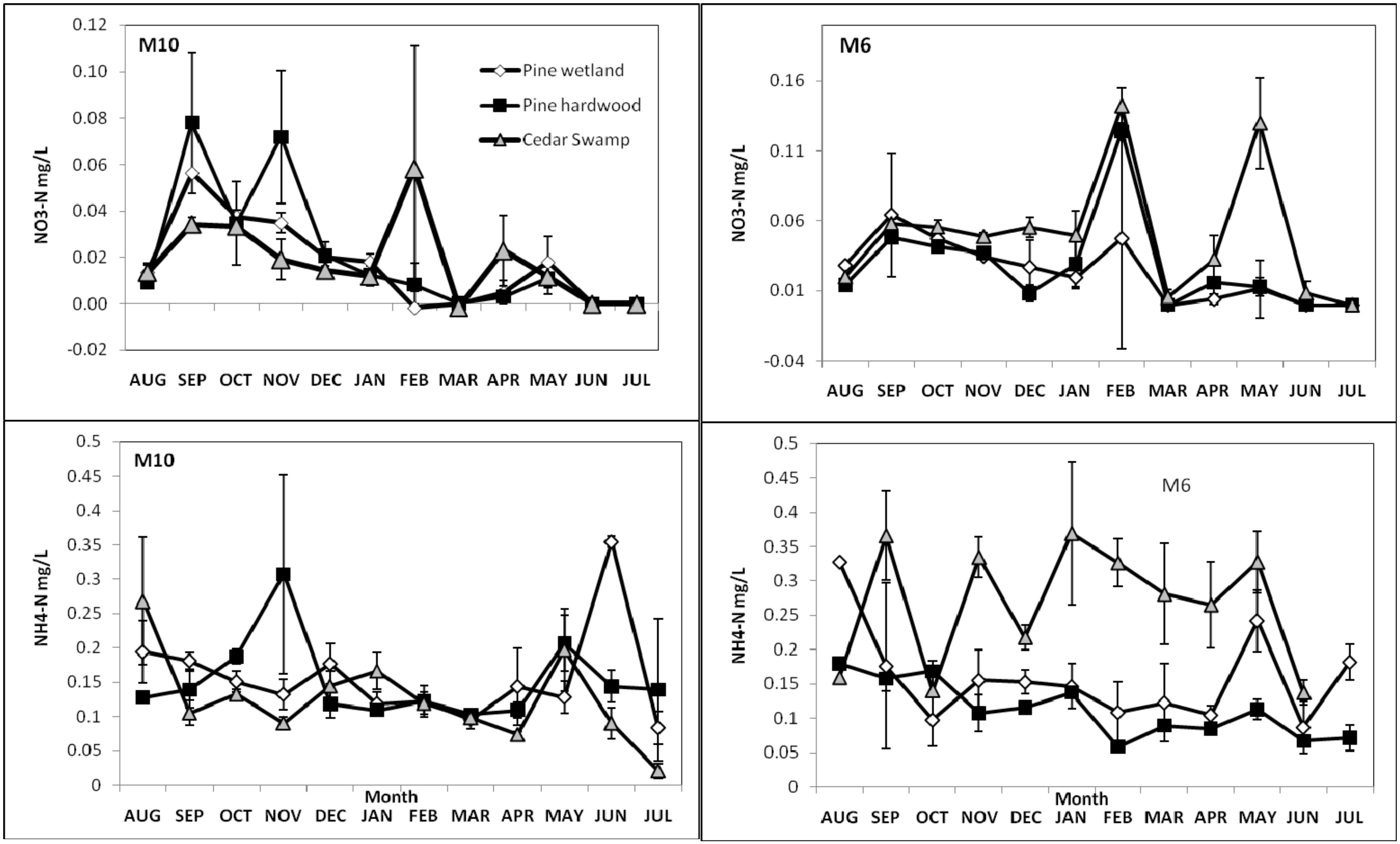


Fig. 14. Frequency distribution of episodes of precipitation-free days, May1901 – May 2009, from the Indian Mills weather station.

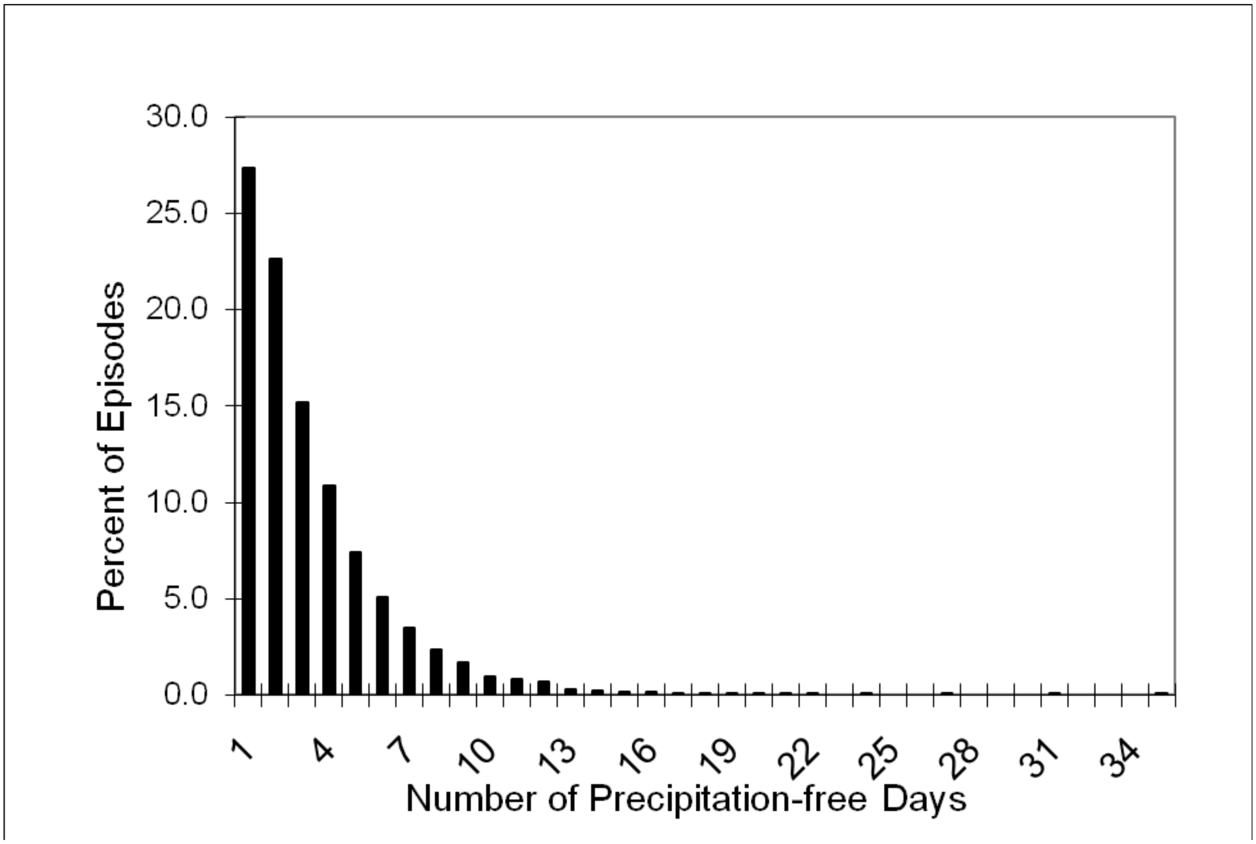


Table 1. Soil properties of the six wetland plots. Means ± standard errors calculated for all measured values (n=39 representing values for three samples per month, April 2005 to April 2006 inclusive). NH_4 – N values are from the KCl extracts of each initial soil sample. No nitrate was observed in any of the initial soil samples.

Plot	pH	EC (μS cm^{-1})	Moisture % (g 100 cm^{-3})	Organic Matter (kg m^{-3})	Extr. NH_4-N (g N m^{-3})	Median Water Table Depth (range) (cm)[a]
M10 PW	3.83 ± 0.02	49.62 ± 3.38	17.05 ± 1.47	203.8 ± 27.19	0.47 ± 0.07	34.9 (15.1 – 91.0)
M10 PH	3.67 ± 0.03	63.97 ± 3.88	22.77 ± 1.65	288.3 ± 35.68	0.55 ± 0.08	25.0 (7.5 – 69.8)
M10 CS	3.65 ± 0.02	75.13 ± 2.00	86.5 ± 5.48	1,178.5 ± 65.05	1.85 ± 0.14	-1.37 (-17.0 – 42.4)
M6 PW	3.82 ± 0.03	48.42 ± 3.02	13.82 ± 1.00	212.2 ± 18.31	0.54 ± 0.05	55.3 (42.6 – 115.2)
M6 PH	3.82 ± 0.02	51.45 ± 2.30	15.93 ± 1.12	198.01 ± 17.32	0.44 ± 0.04	23.5 (9.8 -73.8)
M6 CS	3.62 ± 0.02	85.79 ± 2.35	73.78 ± 2.00	1,281.6 ± 37.36	2.71 ± 0.18	3.1 (-0.4 – 49.1)

[a]Medians derived from observed water table depths in monitoring wells within each plot. N = 23 – 36 observations per well. Data courtesy of Dr. A. Brown (The Pinelands Commission)

Table 2. Pearson correlation coefficients for comparisons of observed ground water table levels. All correlation coefficients are significant at P<0.001.

Plot comparison	Type comparison	Correlation Coefficient
Within M10	PW v. PH	0.92
	PW v. CS	0.83
	PH v. CS	0.90
Within M6	PW v. PH	0.82
	PW v. CS	0.95
	PH v. CS	0.81
Between Sites	PW	0.75
	PH	0.90
	CS	0.77

Table 3. Regression relationships between ground water table levels (WT, cm below the surface of hollows) and volumetric soil moisture (SM), and between soil moisture and organic matter content (OM) of the soil within each plot. Each analysis uses the mean of n=3 measurements of soil properties in each plot for each month versus the estimated water table level on the date of sampling, so that n=13 for each regression analysis

SITE	VEGETATION	EQUATION	R^2_{adj}	F (P)
M10	Pine wetland	SM = -0.270WT + 27.78	0.61	15.97 (0.0025)
	Pine hardwood	SM = -0.420WT + 34.13	0.81	40.87 (<0.001)
	Cedar swamp	ns	ns	ns
M6	Pine wetland	SM= = -0.253WT + 20.77	0.52	11.68 (0.0068)
	Pine hardwood	SM = -0.367WT + 23.44	0.55	11.82 (0.0064)
	Cedar swamp	SM = -0.389WT + 77.15	0.35	7.45 (0.016)
M10	Pine wetland	OM = 9.00SM + 59.67	0.22	10.68 (0.002)
	Pine hardwood	OM = 9.00SM + 68.61	0.29	15.16 (<0.001)
	Cedar swamp	OM = 9.04SM + 412.7	0.62	78.02 (<0.001)
M6	Pine wetland	OM = 9.65SM + 80.78	0.25	12.80 (0.001)
	Pine hardwood	OM = 8.06SM + 76.95	0.25	12.63 (0.001)
	Cedar swamp	OM = 12.97SM + 332.4	0.43	27.67 (<0.001)

Table 4. Relationships between initial extractable NH_4-N (“NH4”) and soil moisture (“SM”). Linear regressions conducted on all replicates (n=36 per analysis).

SITE	WETLAND TYPE	EQUATION	R^2_{adj}	P
M10	Pine wetland	NH4 = 0.16 + 0.02SM	0.16	0.008
	Pine hardwood	NH4 = 0.11 + 0.02SM	0.19	0.005
	Cedar swamp	NH4 = 1.01 + 0.01SM	0.12	0.02
M6	Pine wetland	NH4 = 0.27 + 0.02SM	0.13	0.02
	Pine hardwood	NH4 = 0.19 + 0.02SM	0.26	0.001
	Cedar swamp	ns	-	-

Table 5. Relationships of the net N mineralization rate to potentially explanatory environmental variables. Linear regression analyses were used to explore relationships between quantitative variables, and one-way analyses of variance used to compare values among months. Statistical descriptors (F, P and r2) are given in parentheses below each equation. Sample sizes are n=36 for within-plot analyses, and n=6 for all-plot analyses (based on a single mean value for each plot).

SITE	VEGETATION	Organic Matter	Initial NH_4	Soil Moisture	Water Table
M10	Pine wetland	ns	-17.84*NH4 + 18.67 (F=4.23, p=0.05, r2=0.11)	ns	ns
	Pine hardwood	ns	ns	ns	ns
	Cedar swamp	ns	ns	ns	ns
M6	Pine wetland	ns	-42.97*NH4 +34.95 (F=21.71, p<0.001, r^2= 0.40)	4.83*SM - 44.15 (F=6.14, p=0.02, r2=0.13)	ns
	Pine hardwood	ns	-32.7*NH4 + 42.2 (f=5.06, p=0.03, r2=0.13	ns	ns
	Cedar swamp	ns	64.60*NH4-314.31 (F= 4.23, p=0.05, r2=0.11)	ns	ns
All plots		0.10*OM - 8.85 (F=31.701, P=0.005, r2=0.86)	54.37*NH4 – 16.81 (F=213.11, p<0.001, r2=0.98)	1.37*SM -7.78 (F=11.18, p=0.03, r2=0.67)	ns

Table 6. Regression relationships relating net N mineralization rate ('y') to soil moisture ('x'), analyzed for all wetland sites pooled for each month (pine wetland, pine hardwood, cedar swamp within M6 and M10 watersheds); n=18 for each regression (three replicates of each of 6 stands).

Month	**Equation**	R^2
April 05	y = 1.029x + 0.504	0.48
May 05	y = 1.900x + 7.066	0.34
June 05	y = 2.918x - 14.78	0.41
July 05	y = 2.841x - 27.81	0.57
Aug 05	y = 4.936x - 2.483	0.63
Sept 05	y = 4.907x - 32.39	0.24
Oct 05	y = 1.650x - 2.708	0.69
Nov 05	ns	
Dec 05	y = 0.693x + 13.21	0.30
Jan 06	ns	
Feb 06	y = 0.548x + 5.597	0.36
Mar 06	ns	

Table 7. Cedar swamp N mineralization survey results. All values are means ± st. errors of three replicates. Values are reported as mg N kg^{-1} soil day^{-1}, as (1) they are all organic soils, with similar bulk densities, and (2) we did not measure the bulk density of each soil.

Watershed	Site Code	Net N mineralization rate mg N/kg soil/day	Net Nitrification rate mg N/kg soil/day
Albertson- Pump Branch	P1C	0.00 ± 0.2	0.00
Albertson- Pump Branch	A7C	0.04±0.02	0.00
Albertson- Pump Branch	A1C	-0.03 ± 0.08	0.00
Albertson- Pump Branch	A9C	0.04 ± 0.02	0.00
Albertson- Pump Branch	A10C	0.01 ± 0.03	0.00
Skit Branch	S1C2	0.07 ± 0.07	0.00
Skit Branch	S1C3	0.14 ± 0.10	0.00
Skit Branch	S3C	-0.04 ± 0.05	0.00
Skit Branch	S2C	0.00 ± 0.01	0.00
Morses Mill	R7C2	-0.02 ± 0.02	0.00
Morses Mill	R7C	-0.01 ± 0.01	0.00
Morses Mill	R1C	-0.01 ±0.0.02	0.00
Morses Mill	R6C	0.03 ± 0.07	0.00
Morses Mill	R8C	0.01 ± 0.00	0.00
McDonalds Branch	M10C	0.83 ± 0.46	0.00
McDonalds Branch	M6C	2.20 ± 0.51	0.00
McDonalds Branch	M9C	-0.12 ±0.04	0.00
McDonalds Branch	M1C	0.07 ± 0.04	0.00
McDonalds Branch	M3C	0.11 ± 0.03	0.00