Report for 2002ND15B: Northern Forest Wetlands: Characteristics and Influences on Invertebrate and Amphibian Community Structure

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Report Follows:

Northern Forest Wetlands: Characteristics and Influences on Invertebrate and Amphibian Community Structure

ND WRRI Graduate Research Fellowship Project
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Abstract

Because timber harvesting, especially clear cutting, dramatically modifies vegetation and local patterns of hydrology, it is likely that biotic communities, productivity, and physical features of wetlands adjacent to harvested areas are influenced by that activity. Best Management Practices (BMPs) have been developed as guidelines for timber harvesting for protecting water quality in wetlands. Unfortunately, these general BMP guidelines were developed without the benefit of ecological data derived from these wetland types and enlightened future refinements will be impossible without empirical data. Macroinvertebrates and amphibians are very sensitive to water quality characteristics and monitoring these populations is likely to provide key insights on impacts from forest harvesting, ecological recovery of impacted areas, and usefulness of BMPs. Lack of such information limits conducive efforts of wildlife and forest managers and may compel them to rely upon paradigms from studies of prairie potholes or other systems that may not be comparable to northern forest wetlands. The goal of the proposed project is to provide relevant data and synthesis improving basic understanding and allowing development of management guidelines (such as future BMPs) conducive to maintaining wildlife habitat and biodiversity in forested landscapes.

STATEMENT OF THE WATER PROBLEM

Agriculture has transformed grasslands across most northern prairie regions of the U.S. Similarly, silvacultural activities are responsible for the most widespread landscape-scale modification of northern forest regions since arrival of European settlers. Because timber harvesting, especially clear cutting, dramatically modifies vegetation and local patterns of hydrology, it is likely that biotic communities, productivity, and physical features of wetlands adjacent to harvested areas are influenced. As an important first step, Best Management Practices (BMPs) have been developed as guidelines for timber harvesting in Minnesota (MDNR Division of Forestry 1999). BMPs are only voluntary, but do recommend general practices for protecting water quality in wetlands. Unfortunately, these general BMP guidelines were developed without the benefit of ecological data derived from these wetland types and enlightened future refinements will be impossible without empirical data. Macroinvertebrates and amphibians are very sensitive to water quality characteristics (Lannoo 1998, Brown et al. 1997, Mitsch and Gosselink 1993) and monitoring these populations is likely to provide key insights on impacts from forest harvesting, ecological recovery of impacted areas, and usefulness of BMPs. Lack of such information limits conducive efforts of wildlife and forest managers and may compel them to rely upon paradigms from studies of prairie potholes or other systems that may not be comparable to northern forest wetlands. The goal is to provide relevant data and synthesis improving basic understanding and allowing development of management guidelines (such as future BMPs) conducive to maintaining wildlife habitat and biodiversity in forested landscapes.

BACKGROUND

Northern Minnesota's landscape includes over 1.2 million hectares of forested wetlands (Trettin et al. 1997) ranging from vernal pools to permanently-flooded wetlands. Seasonally-flooded wetlands (*sensu* Stewart and Kantrud 1971) within forested regions probably support a large share of the biodiversity in these landscapes. Unfortunately, these unique wetlands have been overlooked and little is known regarding fauna, dynamics, productivity, physical characteristics, or other major features of these wetlands. To make matters worse, a large (but unknown) share of these small wetlands (< 5 ac) are essentially unaccounted for because they are unidentified in NWI data layers or by conventional aerial photography (pers. comm., Richard Buech).

Estimates indicate that Minnesota has lost approximately 45% of the 2,279,473 hectares of forest wetlands thought to have existed before arrival of European settlers (Trettin et al. 1997) and remaining areas represent a common and remarkably understudied aquatic ecosystem (Higgins and Merrit 1999); obviously, this increases ecological significance of remaining sites. On a national scale, the recent National Wetland Trends study (USFWS) indicates that for the first time the U.S. has dropped to less than 50 million acres of forested wetlands in the lower 48 states (Gary Pierce 1997). Mature forests are economically valuable to forest harvesters, and landscape modifications resulting from silvicultural activity threatens adjacent wetlands and wetland-dependent communities (Trettin et al.1997). Such threats will intensify as demand for wood, wood fiber, and paper products increase (Trettin et al.1997). In Minnesota, large, mature aspen (*Populus*) is abundant, sometimes predominant, throughout several plant communities of the Mesic Hardwood System (Almendinger and Hanson 1998) and small, seasonally-flooded wetlands are often associated with these aspen stands. Such stands are placed on a harvest rotation schedule and routinely clear cut. Special urgency exists among wildlife and forest managers concerned about fate of wetlands in these landscapes with remaining mature aspen stands (stand age 60-70 yr) because near-future demand for fiber and various wood products appears to be great.

Aspen clear cutting physically modifies landscapes; likely effects on adjacent wetlands and their associated communities are both direct and indirect and may be difficult to separate. It has been shown in other forest studies that removing trees from the landscape modifies the watertable (Roy et al. 2000). I expect that canopy removal elevates water temperatures, increasing surface evaporation rates and truncating natural wetland hydroperiods. It is widely believed that silvacultural activities increase soil compaction and enhance sediment translocation to adjacent wetlands. Timber harvesting reduces forest-floor litter and modifies structural complexity of understory vegetation adjacent to wetlands (de Maynadier and Hunter 1995). Clear-cut harvesting dramatically shifts forest age structure from a predominance of mature species to a landscape dominated by younger, often more commercially desirable trees (Mitsch and Gosselink 1993, Trettin et al. 1997). Large reductions or changes in characteristics of litter inputs to wetlands may modify energy transfer and ultimately reconfigure wetland food webs. Finally, road construction is often associated with timber harvest; new roads may constitute barriers for some amphibians, may allow nonnative plants to invade and suppress natural regeneration of wetlands that were once isolated, and are often permanent landscape features once harvesting ceases. Responses to cumulative impacts are difficult to predict, but some silvaculture activities like clear cuts are likely to have unexpected consequences for native communities and functions of northern forest wetlands (Trettin et al. 1997).

Native communities of aquatic invertebrates and amphibians integrate ecological conditions and reflect habitat quality and perhaps changes in wetlands and associated uplands. It is also likely that general wildlife values of seasonally-flooded wetlands in forested landscapes can be generally inferred from native invertebrate and amphibian communities. Aquatic organisms interact through direct and indirect trophic pathways and through a variety of non-trophic, habitat-dependent relationships (Murkin and Batt 1987, Murkin and Kadlec 1986). For example wetland invertebrates are vital food chain links supporting energy transfer through wetland food webs (Murkin and Kadlec 1986). Biotic communities, physical features, and functional characteristics of northern forest wetlands must be better understood if managers hope to predict results of wetland losses and modifications, or develop best management practices that preserve these wetland resources.

Evidence from related studies indicates that aquatic macroinvertebrates and larval amphibians will be useful indicators of at least some effects of clear cutting and perhaps altered forest-age structures adjacent to seasonally-flooded wetlands. Macroinvertebrates are important functional links in wetland food webs and certain macroinvertebrate groups such as amphipods, anostracans, conchostracans, and others require specific water conditions and hydroperiods (Wiggins et al. 1980). Also many aquatic invertebrate taxa are sensitive to sedimentation, seasonal water temperature patterns, and influx of organic matter (Thorpe and Covich 1991). Aquatic invertebrates have been widely used to assess ecological characteristics of lotic habitats (Resh and Jackson 1993) and a related approach recently has been proposed for prairie wetlands (Adamus 1996).

My study also focuses on amphibians; this is warranted due to their sensitivity at the egg and larval stages of their aquatic-obligate life cycles (Blaustein and Wake 1996) and because many amphibians are invertebrate predators, heavily dependent upon zooplankton and aquatic macroinvertebrates. Use of amphibians such as salamanders also seems justified because these organisms are terrestrial during portions of their lifecycle returning to the wetland solely to reproduce. Knutson et al (1999) report that the association between forest and amphibians is one of the most consistent landscape-scale habitat relationships reported in the literature. de Maynadier and Hunter (1995) review results of several studies indicating that salamanders are consistently reduced, at least locally, following clear cutting. Such amphibian declines appear related to reductions in forest-floor litter following timber harvest, but as Ash and Brush (1994) suggest, “we know they disappear from clear-cuts, but that is all we know”. Wisconsin and Maine has begun initiatives in ecological studies of the wetlands to aide in the refinements of future guidelines (Malcolm Hunter and Aram Calhoun, pers. comm). These studies will no doubt give managers in those regions critical information allowing them to better evaluate impacts from forestry.

OBJECTIVES

The goals for this study are two fold. First, we will characterize physical features and aquatic communities of wetland sites located within aspen-dominated landscapes. This will include summaries of hydroperiods, primary productivity, botanical characterizations, and basic “limnological characteristics ,as well as species abundance, taxon richness, and community structure of aquatic invertebrates and amphibians. A second study phase will examine relationships among physical features, biotic communities, and forest age structure (as a result of clear cutting in adjacent uplands). The study will focus on potential "treatment effects" in the following: 1) Macroinvertebrate and amphibian abundance, community richness, and for invertebrates, proportions of functional groups (scrapers, collector-gathers, shredders, etc.). 2) Hydroperiods; depth and duration of annual flooding, primary productivity and other "limnological" characteristics.

LOCATION OF PROPOSED STUDY

This study focuses on seasonally-flooded palustrine emergent sites because these wetlands are known to be important for aquatic invertebrates and amphibians and often such sites are vulnerable to effects of activities in adjacent uplands. My study sites are located within aspen-dominated landscapes of the Paul Bunyan State Forest located in the Pine-Moraines and Outwash Plains subsection of the Ecological Classification System (ECS) (Almendinger and Hanson 1998) and Buena Vista State Forest found within the Chippewa Plains subsection of the ECS.

METHODS AND MATERIALS

In each forest I have identified and begun sampling 12 palustrine seasonal wetlands within the Buena Vista State Forest. To provide some measure of inference into other landscapes, I replicated these efforts in the Paul Bunyan State Forest. Both state forests are located in different subsections of the Minnesota ECS (Almendinger and Hanson 1998). Both forests are found within the Laurentian Mixed Forest Province of north-central Minnesota. Each forest contains three clusters of four wetlands, each including one control and 3 treatment groups. Control sites are those with no adjacent forest harvesting within the past 75 years. The "effect group" (treatment) will include one site per cluster which was harvested during the winter of 2000-2001. Two "effect/ recovery" age-classes (young-age class and mid-age class) include sites adjacent to harvest events of 10-34 and 35 -59 years (respectively) before present. Age-classes were determined using a GIS-procedure that revealed "natural breaks" of aspen forest-age structure. Replicate sites within each cluster are needed to allow for anticipated high variance in response variables. Previous wetland field studies indicate that 4-5 replicate sites are likely sufficient for detecting trends in invertebrate and amphibians (Hanson and Riggs 1995, Hanson and Zimmer 1996, and Cox et al. 1998).

Invertebrate sampling will be done three times per field season with surface-associated activity traps (Hanson et al. 2000) at random locations in each wetland. Samples are placed in ETOH, transported to the laboratory, and using stereomicroscopes will be identified and enumerated. Amphibians are collected with modified minnow traps, identified, and released on site. Exploration of other amphibian collection methods such as drift fences, seining, and pitfall traps, indicated that these techniques would be less efficient in these sites. Data from recent studies indicates the use of these traps are the single best method (pers.comm. Richard Buech). Amphibians will also be sampled by Visual Estimate Surveys (VES) (Heyer et al 1994), once in early-spring and again in mid-summer.

Aquatic macroinvertebrates and amphibians will be sampled throughout the open water period, once each during April, May, June, and July. April (after ice-out). This will allow estimates of breeding adult amphibians and over-wintering macroinvertebrate populations. May, sampling should coincide with peak populations of macroinvertebrates. June, samples should reflect the recruitment of larval amphibians and early-summer populations of macroinvertebrates including predatory species. July, samples will assess remnant amphibian and macroinvertebrate populations.

Concurrent (monthly) water-quality monitoring will be done using one-liter water samples collected from the center of each wetland. Water will be tested for chlorophyll *a*, pH, dissolved oxygen, total alkalinity, total phosphorus, total nitrogen and specific conductance. Samples will be shipped to MN. Department of Agriculture's laboratory (St. Paul) for testing. Turbidity will be measured at each wetland with a portable nephelometer. Other physical data will be collected twice per field season and includes, water temperature, upland soil temperature, and concurrent monitoring of the hydroperiod. These data allow assessment of deviations from the control group and will be useful for detecting trends in the response variables.

Given the complexity of ecological data and the lack of previous experimental studies on these communities both univariate and multivariate statistical procedures will be used. Initially, I will use multivariate techniques to explore the data sets for broad trends and possible relationships among wetland parameters (pH, turbidity, etc.) and amphibian and macroinvertebrate taxa. Treatment effects will be assessed via a parametric ANOVA within PROC MIXED in GLM (SAS 1992).

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RESULTS/DISCUSSION

Community structure in seasonal wetlands may be subject more to physical features than to biotic influences. Because forest wetlands are functionally linked to the adjacent upland forest, physical modifications to the uplands may alter annual development of community structure. Timber harvesting modifies vegetation and local patterns of hydrology, including hydroperiod; thus it is likely that biotic communities and physical features of adjacent wetlands are influenced.

I characterized physical features and biotic communities of northern forest wetland sites, and assessed relationships among wetland communities and forest age structure in adjacent uplands. I sampled macroinvertebrates and recorded physical measurements in 24 seasonally flooded wetlands, within aspen-dominated landscapes, throughout the growing season (figure 1). Sampling was conducted three times per field season, initiating upon ice-out on these wetlands. Data were described using a combination of statistical techniques including multivariate Redundancy analysis (RDA), Principal Components Analysis (PCA), and mixed-model ANOVA (PROC MIXED, Littel et al. 1996).

I used partial-RDA to test for effects from physical attributes with the invertebrate community as my response variable. I used both sample period (time) and forest location , as covariables in the model. These two variables, and their associated variances, were accounted for, and removed from the model. Partial-RDA indicated that invertebrate community structure was influenced by upland forest age-structure, forest location, amount of canopy, initial water depth, hydroperiod, total alkalinity, and the presence of predatory invertebrates (Figure 2). A variance partioning exercise was conducted on the environmental variables to determine what portion of the total variance from the partial-RDA was attributable to each variable (Figure 3). Figure 3 shows sampling period and forest location was found to have the highest proportion of the variance (hence were removed from the p-RDA), with variances for canopy, hydroperiod, forest-age, initial water depth, pH, an total alkalinity were also determined.

Additionally, I assessed fixed-effects of forest-age structure and location (state forests) using a repeated measures, mixed-model (Littell et al. 1996) analysis of variance (ANOVA) on the PCA site scores. I used the invertebrate data (represented here as site scores) as the response variable; all tests were conducted at the alpha = 0.10 level. All mixed-model procedures were performed using the MIXED Procedure in SAS (SAS 1992, Littell et al. 1996). Mixed-model ANOVA showed a significant ($p < 0.05$) spatial (location) and structural (age-class) interaction among wetland sites.

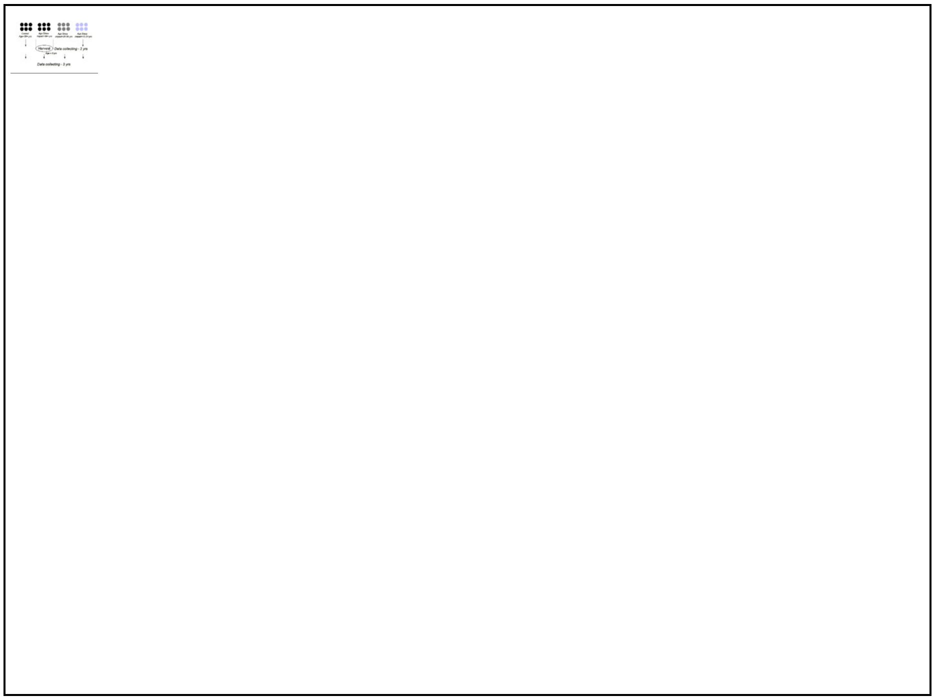

Figure 1. Study design diagram depicting the treatment groups and effect/recovery groups. Phase I includes data collected from the first two years. Phase II includes three years of data collection following clear-cut treatments to selected wetlands. Note: The four groups represent the chronology of the adjacent landscape relative to years since last forest harvest.

Time of sampling, forest-age structure, pH, specific conductance, soil temperature, and especially hydroperiod and amount of forest canopy-closure are important influences of invertebrate community structure. Thus alterations to these physical components would likely induce a subsequent changes in the invertebrate communities inhabiting wetlands adjacent to forested areas. This characterization of the macroinvertebrate communities and their analyses in light of the physical properties of my study sites completes Phase I of this study. Phase II is currently underway with data for the fourth year still being sorted and analyzed. Data from this portion of the study tends to support that clear-cutting activities increase hydroperiod in those wetlands adjacent to the clear-cut. Trends also suggest that water depth in the spring increases in association with the clear-cut areas. This result is surprising given that the past two winters in northern Minnesota experienced below average snowfall amounts. Taken together, increased depth and hydroperiod, may result in a change in the legal definition of which classification these wetlands fall under.

However, if those physical changes are indeed taking place, it may be a short term effect. Water depth and hydroperiod may return to there normal levels and periodicity as the forest regenerates.

Macroinvertebrate communities in the clear-cut wetlands (post treatment) also appear to be more heavily influenced by predatory invertebrates than those communities found in pre-treatment conditions. Biological controls of the communities are more typical in lakes and more wetlands of longer duration (Wiggin et al. 1980, Wellborn et al. 1996). Increasing hydroperiods in temporary wetlands may remove ephemeral species from specific wetlands. Ephemeral organisms, such as Anostracans, have no defense from predators. Invertebrates, especially those that are predaceous that normally invade these wetlands in the spring may become established and displace or dominate the wetlands community. Changing food-chain-base structure may also displace organisms that rely upon these wetlands as a food source.

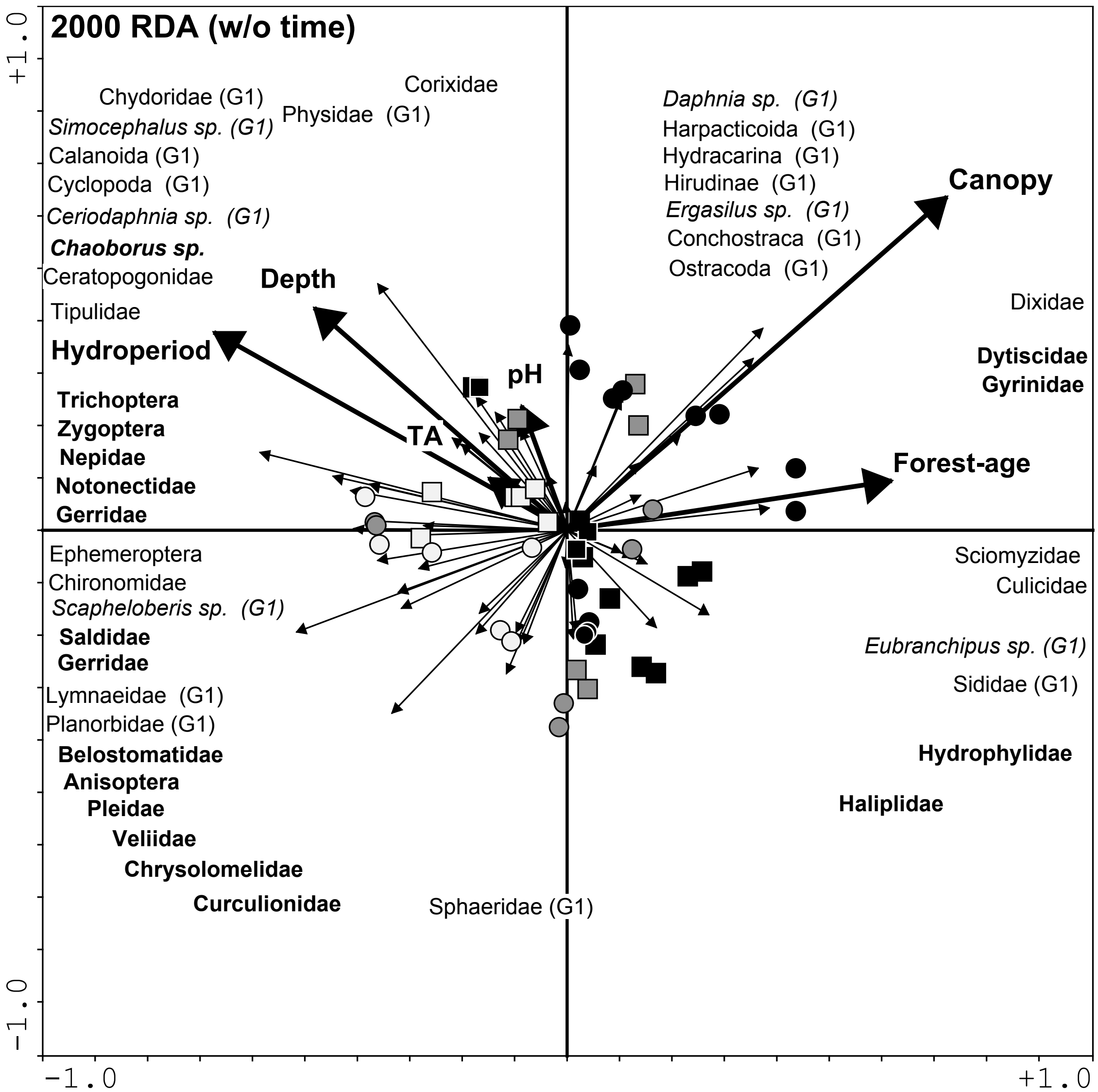

Figure 2. Partial-RDA ordination (triplot) of invertebrate communities during field season 2000 with sample period (time) and forest location treated as covariables. Partial-RDA triplot shows site locations, and species and environmental vectors. The figure shows relationships of invertebrate taxon (thin arrows) with significant environmental variables (bold arrows). Increasing length of environmental vectors indicates increasing variance accounted for and, thus, increasing influence. Circles = BVSF, Squares =PBSF, Black symbols = old-age stands, Gray = mid-age stands, and light gray = young-age stands. Predatory taxa are denoted in bold type.

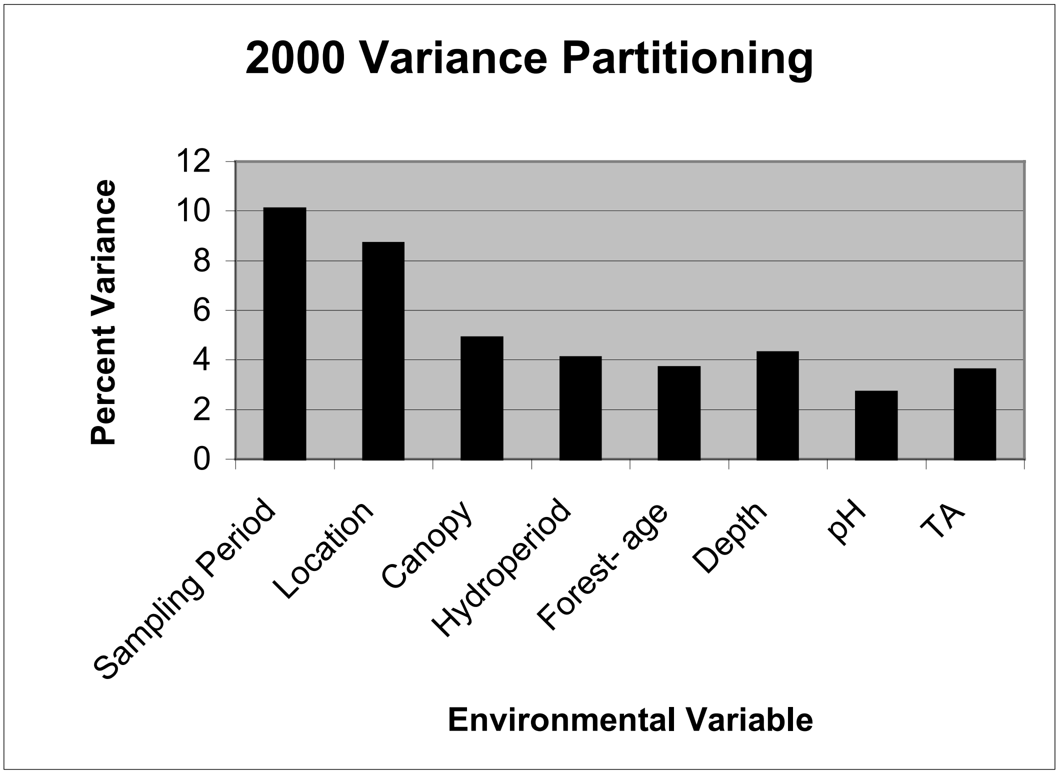

Figure 3. Results of variance partitioning following RDA of invertebrate data from field season 2000 (with sample period and forest location treated as a covariable). Variances given are reflective from total variance. Combined interaction effects between all combinations of variables were minimal at 3.9%.

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