# The Effect of Forest Structure on Amphibian Abundance and Diversity in the Chicago Region

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Citizens for Conservation
U.S. Environmental Protection Agency
Great Lakes National Program Office
Lake County Forest Preserve District
Forest Preserve District of Will County

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## SUMMARY

Amphibian populations are under increasing threat in the Chicago region due to habitat loss and habitat degradation. The impacts of habitat loss are self-evident and well documented. The impacts of habitat degradation are less clear. In the Chicago Region the majority of forests have been degraded (altered from their natural pre-settlement condition) by grazing, logging, and fire exclusion, and excessive deer herbivory. We investigated whether amphibian abundance and diversity was related to the condition of upland forests adjacent to breeding ponds. We monitored vegetation composition and amphibian abundance in April and June, 1999, in six high quality (Grade B) forests and six low quality (Grade C and D) forests adjacent to ponds in Lake County (eight sites) and Will County (four sites), Illinois.

A total of 205 amphibians of six species were recorded at all sites in drift fences (65 trap-nights at each site, total 780 trap-nights). The six high quality forests supported higher amphibian species richness and diversity than the six low quality forests, and nonsignificantly higher numbers of amphibians. Down wood was significantly more abundant in the higher quality forests, which had more and larger logs, especially well-decayed logs, than the lower quality forests. Overstory tree density was lower in the high quality forests, due to the lower abundance of trees in smaller size classes. Cover and species diversity of herbaceous vegetation was similar in both high and low quality forests. When forests were grouped on the basis of amphibian abundance (six 'better' habitat sites vs six 'poorer' habitat sites) percent cover of herbaceous vegetation in both April and June was significantly higher in sites with greater numbers of amphibians.

Multiple linear regression indicated that 1) amphibian abundance was higher in sites with higher cover of

## INTRODUCTION

Researchers interested in assessing interactions between amphibians and upland habitat have focused on areas characterized by distinct differences, comparing old growth forests to recently logged forest stands (Ash, 1997; Welsh and Lind 1991, 1995; Petranka et al. 1993; Pearman, 1997), undeveloped sites to developed sites (Delis et al. 1996, Dodd 1996, Means et al. 1996), disparate habitat types (Jones 1988), or sites subjected to different logging treatments (Renken, 1997). No studies have investigated the impact of gradual habitat degradation on amphibian abundance and species richness, nor the relationship between forest quality and amphibian abundance and diversity.

The majority of upland forests in the Chicago Region have been moderately to severely degraded by urban development (fragmentation), land use activities (fire suppression, grazing, logging), white-tailed deer herbivory, and invasion of non-indigenous species (Bowles et al. 1998). Few of the forests in the Chicago region retain high natural quality, yet upland forest provides critical habitat for site that a) had a similar sized pond with similar vegetation and canopy cover, b) had a similar upland forest community type, and c) was located in the same forested tract or in a nearby forested tract. Twelve sites were located that met these criteria. Eight were in Lake County, 30 miles north of Chicago, and four were in Will County, 30 miles south of Chicago (Figure 1 and Table 1). Within each county, sites were paired on the basis of pond size and vegetative structure, and apparent natural quality of the adjacent upland forest of similar size, soil, hydrology, and aspect. All forests were located within the Northeastern Morainal Division (Schwegman, 1973) to minimize biogeographic variation in the potential species assemblage.

"Natural quality" is a qualitative assessment of the perceived similarity of a natural community to the presettlement condition, based on visual evidence of past impacts. While used extensively throughout Illinois and other states, "natural quality" lacks a quantitative basis that would substantiate the qualitative assignments, and that would allow comparisons between sites with similar or dissimilar assigned grades. While most experienced natural area biologists agree on the assignment of sites to very high or very low natural quality, there is a large grey area for sites between these two extremes.

We initially intended to sample mesic upland forests adjacent to ponds of similar size and structure, with the forests differing primarily in natural quality; very high (rich herbaceous understory, oldgrowth overstory) and very low (bare understory or an understory dominated by nonindigenous vegetation, and young or highly disturbed overstory). We failed to locate any Grade A mesic forests adjacent to suitable ponds, and used

**Illinois Natural Areas Inventory (INAI) natural quality grades. Summarized from White (1978).**

**Grade A**: Relatively stable or undisturbed communities; for example, old growth, ungrazed forest.

**Grade B**: Late successional or lightly disturbed communities; recently lightly disturbed, or moderately to heavily disturbed in the past but recovered significantly. For example, old-growth forest selectively logged or moderately grazed, and subsequently recovered.

p e h t 5 . e s o **Grade C**

Grade B forests as our “high quality” sites.  We found only one pair of sites that met the selection of the fence. Drift fences were checked at one to two day intervals over a three week period in spring (April 24 to May 18 1999) when early breeding amphibian species were leaving ponds and later breeding species were arriving, and a four week period in summer (June 22 to July 25 1999) when immature amphibians were leaving the ponds. Spring drift fence sampling was timed to coincide with the movement of early breeding species away from the ponds. This typically results in fewer points/quadrat. Canopy cover was measured at 0.3m above ground level using a concave densiometer. Vegetation thickness was measured by recording number of 30cm$^2$ (6cm x 5cm) squares obscured by vegetation (observed from 4m distant at a height of 1.5m above ground), on a board 0.30m x 2.0m, in four vertical layers; 0-.25m, >.25-.50m, >.50-1.0m and >1.0-2.0m above ground (100 squares/vertical meter, maximum 200 squares total). Diameter of all stumps and down logs $\geq$10cm in diameter were recorded to the nearest cm, and assigned to one of five 'decay classes' (Maser et al. 1979: 1= newly fallen tree with intact bark, branches and trees; 2=sagging slightly, with intact bark, some branches, and no twigs; 3=sagging near ground, with sloughing bark and no large branches; 4=completely on ground with little or no bark, and punky wood; 5= well decayed, with soft powdery wood and invasion of roots and seedlings). Because we were interested in measuring actual available habitat/shelter, we recorded only that portion of down logs that was actually on or within 3cm of the ground surface.

used to test for significant differences between the ‘good’ and ‘poor’ amphibian sites. Considering significantly and positively related to pond depth and cover of herbaceous vegetation in June (Figures 2a and 2b). Together, these two factors accounted for 64% of the variation in total salamander abundance, and 67% of the variation in April salamander abundance. In June, salamander abundance increased significantly as a function of pond drydate (Figure 2c).

Toads (*Bufo americanus*) were recorded at nine sites and were the dominant amphibians at three sites. Toads were more abundant in June (0.81/trap-night) than in April (0.54/trap-night). Toad abundance in June was positively related to pond drydate (Figure 3a), but toad abundance in April was unrelated to any of the tested variables. Total toad abundance (April and June combined) was significantly related only to horizontal vegetation thickness in April; toad abundance increased as vegetation thickness decreased (Figure 3b).

Frogs (*Pseudacris crucifer, Pseudacris triseriata*, *Rana pipiens*) were recorded at seven sites and were the dominant amphibian group at one site. Frogs were slightly more abundant in April (1.06/ trap night) than in June (0.81/trap night). Frog abundance in both April and June was consistently and significantly positively associated with herbaceous cover in June (Figure 4a). Frog abundance in April was also significantly and positively associated with reduced distance to the nearest pond (Figure 4b).

Total drift fence data reflected the interaction of the three amphibian groups. Pond depth and cover of leaf litter in April accounted for 68% of the variation in capture rate throughout the study period (Figures 5a and 5b). Leaf litter cover in April was also a primary influence on amphibian species richness (see below). In April, drift fence capture was significantly and positively related to both herbaceous cover in June and pond depth (Figures 6a and 6b). In June, total drift fence capture increased significantly with increased pond drydate (Figure 6c).

The number of amphibian species at any given site was very strongly related to just four variables; cover of leaf litter in April, and cover of herbaceous vegetation in June, and distance to the nearest pond and pond drydate. Together, these four factors explained 97% of the variation in species richness. Amphibian species diversity (H'), a measure of the relative number of species and evenness of species distributions among all sites (Brower et al. 1990), was strongly related to cover of leaf litter in April, and the distance to the nearest pond.

As a group, the six 'good' amphibian sites had significantly more groundlayer vegetation in April and in June than the six 'poor' amphibian sites (18.2% and 64.8% vs 10.5% and 43.5% in April and June, respectively), and significantly more down wood in lower decay classes (670 $dm^3$ vs 118 $dm^3$, respectively; Table 3). No other significant differences were detected between the two groups of sites.

As a group, the six 'high' natural quality sites had significantly higher amphibian species richness and nonsignificantly higher H' diversity than the six 'low' quality sites (Table 3). Drift fence capture rates of all amphibians were two to three times higher in the six 'high' quality sites, but these differences were not significant. Both high and low quality sites had statistically similar cover of herbaceous vegetation, species richness, vegetation thickness, and canopy cover, in both April and June. The 'high' quality sites had significantly more (43.7 vs 14.5) and larger (1323 $dm^3$ vs 88 $dm^3$)

leaf cover supported four to five species. Potentially, leaf cover provided protection from predation and desiccation. Ash (1997) suggested that leaf litter provided an important foraging habitat for plethodontid salamanders in the Blue Ridge Mountains, and that changes in leaf litter characteristics could affect both moisture and food availability. A study by deMaynadier and Hunter (1998) determined that litter cover was an important habitat feature for amphibians in general. Sites located near (<200m) another potential breeding pond supported an average of 3.8 amphibian species, while sites located far (>400m) from a potential breeding pond averaged just 1.5 species. Both spotted and blue-spotted salamanders adults tend to remain near the breeding pond, but some individuals migrate between ponds (Semlitsch 1998). This migration allows both genetic and demographic exchange among established populations (Gill 1978, Berven and Grudzien 1990) and to colonization of new (or former) breeding sites (Laan and Verboom 1990). Several studies have documented an increased risk of amphibian extinction at isolated ponds (Sjogren-Gulve and Ray 1996; Sjogren-Gulve 1994). In general an assemblage of amphibians, or any other taxa, is more likely to persist over the long term when it is a component of a functioning metapopulation (Hanski 1997).

In this study, salamander abundance was strongly and positively associated with the number of nearby potential breeding ponds, as also found with other amphibians (Vos and Stumpel 1995). A single-year study cannot document source-sink relationships (Pulliam 1997), but we suggest that long-term viability of salamander populations requires presence of several breeding ponds within a site. In the Chicago Region, forested sites with breeding ponds are often isolated by streets and urban development, and salamanders can rarely if ever migrate between these sites (deMaynadier and Hunter 2000, Gibbs 1998). Consequently, migration between breeding ponds is frequently restricted to within-sites.

The length of time that the ephemeral ponds retained water was closely associated with abundance of salamanders and toads in June. This relationship reflected the presence of juveniles emerging from ponds that held water longer. The four sites that dried before June 28 had no recruitment; five sites that dried in the first week of July had low recruitment, and two of the three sites that retained water past July 10 had high recruitment. These results indicate that (in the Chicago Region, at least) some percentage of *Ambystoma maculatum* can develop from egg to juvenile in approximately 130 days. The actual percentage of larvae that emerge prior to mid July is likely low, as a minimum of 154 days is needed for just 10% of Pennsylvania *A. maculatum* larvae to achieve metamorphosis (Rowe and Dunson 1995).

At four ponds which dried on or before June 28 (Ryerson 5, Lake-Cook, Plum West, Plum East) only one juvenile amphibian, a *Pseudacris triseriata*, was captured. This species is typically the first to achieve metamorphosis in Chicago region ponds. Assuming that this dry date is typical, it is unlikely that juvenile recruitment of most amphibian species occurs at these four locations except perhaps in exceptionally wet years. Most adult amphibians inhabiting terrestrial habitat at these sites are almost certainly immigrants from nearby ponds. At Lake-Cook, which is isolated from other ponds by roads and residential development, the combination of an early drying pond and lack of available movement corridors has apparently resulted in complete amphibian extirpation. No amphibians were caught at that site in drift fences, time-constrained visual encounter surveys, or seining of the pond. No calling frogs were heard, and no egg masses were noted. Amphibians were known historically from the immediate vicinity (Field Museum of Natural History collection, and Richard A. Edgren Jr.; KSM personal communication, March 3, 2000).

Five ponds dried between July 2 and July 6 (Dan Wright, Elm South, Ryerson South, Ryerson North, and MacArthur). Low numbers of juvenile *Ambystoma laterale* were captured at the first three ponds. No juveniles were observed at the other two ponds, and none of the five ponds had captures of more than one species of juvenile amphibian. When juvenile *A. laterale* were captured, they made up a relatively high percentage of total captures for that species (30-40%) because the number of adult captures was also low. Unpredictable annual variation in juvenile survival at these ponds may limit the size of the adult population.

One pond (Elm North) dried on July 16. Juveniles of three species of amphibians were captured (*Ambystoma laterale*, *Bufo americanus*, *Pseudacris crucifer*) and a fourth (*Rana pipiens*) was observed in the dry pond basin but not captured. Juveniles were more abundant than at earlier drying ponds, but made up only 16.8% of total observations because adult amphibians were more common here than at any other site. The forest adjacent to this pond also had the highest amount of June herbaceous cover (81%).

Two ponds (Thorn 19 and Thorn 13) did not dry in 1999 and are believed to be permanent most years. At Thorn 19 juveniles made up 48.6% of the captures for three species (*Ambystoma laterale*, *Ambystoma maculatum*, and *Bufo americanus*). At Thorn 13 only toad juveniles were captured, and few amphibians of any age class were observed. These disparate results are difficult to interpret, because predator-prey relationships and competitive dynamics are likely very different in permanent ponds relative to the ephemeral ponds at most sites. Larval survival or growth rates could differ in the two ponds. Alternatively, the lower amount of herbaceous cover in June at Thorn 13 could result in higher predation on juveniles, increased desiccation, or an inability to move far enough from the pond to encounter drift fences.

In the Chicago Region, amphibians must contend with multiple impacts; habitat loss as well as habitat isolation due to roads and urban development, and historic and ongoing hydrological alteration. At the beginning of this study we assumed that all ponds were essentially 'undisturbed', based on visual assessment and general site history. We found during the course of this study that three of the 12 sites had anthropogenic alteration; the area surrounding the MacArthur pond had been drained many years prior, isolating the pond hydrologically from other ponds; the pond at Plum West was drained by a ravine that had been 'straightened' at a prior date and subsequently eroded back into the pond margin; and the pond at Lake-Cook on at least one occasion appeared to receive storm runoff from the right-of-way of a heavily trafficked four-lane highway, with the associated contaminants (we did not assess water quality in the ponds). We noted shallow ditches or tiles near other ponds, including two at Ryerson Woods, which did not directly drain the ponds but may have influenced runoff rates. It has also been suggested that reduced herbaceous vegetation contributes to more rapid runoff and a lowered water table (Swink and Wilhelm, 1994), although we did not document this relationship in this study.

Our study documented surface water conditions at 12 ponds in a year with a wet early spring and a dry late spring and summer. Longer-term conditions are more complex: We noted in February/

March 2000 that at least two of the 12 ponds (MacArthur and Lake-Cook) were still dry. Ponds in areas with high clay content soils, including Ryerson North and Ryerson 5, held snowmelt and had sufficient water on March 8, 2000 to support calling *Pseudacris triseriata* and *Pseudacris crucifer*. It would be useful to investigate the effect of pond hydroperiod on amphibians with a multi-year hydrology study addressing relative degree of groundwater and surface water influence on each pond, permeability of underlying soils, and influence of historic drainage.

Assessing amphibian abundance at different seasons (April and June) provides insight into temporal responses to habitat features. There was little correlation between the abundance of amphibians in April, based on drift fence data, and their abundance in June, and all three species groups (salamanders , toad, and frogs) were associated with different features in April and June. Because these animals occupy the sites on a year round basis, using data from a single season or a single species group may provide a one-sided assessment of the suitability of an upland site to support amphibians. Collecting data over multiple years would allow a better assessment of the long-term usefulness of any particular site.

We found no relationship between drift fence sampling and time constrained visual encounter surveys. Drift fences are effective for sampling nocturnal and fossorial species such as salamanders of the genus *Ambystoma* and many frogs, but are less effective with large active species able to climb over the fence. Visual encounter surveys will often encounter these more active species, although in our study 96% of the

captures were of *Ambystoma laterale*. Used in combination, these two methods can provide an accurate survey of the fauna at a given location (Heyer et al., 1994; Karns, 1987).

When we grouped sites on the basis of higher vs lower amphibian abundance, we found that sites with more amphibians had significantly more herbaceous vegetation in both April and June than sites with few amphibians. (18% vs 11% in April, and 65% vs 44% in June). We were unable to find significant differences in terms of down wood, overstory cover or composition, herbaceous species richness or other site characteristics that could explain the differences in observed amphibian abundance. While other studies have found strong correlations between upland habitat structure and salamander abundance, many of these studies assessed sites with substantial macroscale differences, such as logged vs unlogged (Renken 1997, Petranka et al. 1993), different community types (Beauregard and Leclair 1988), or moisture gradients. In this study, we investigated sites that were similar on a macroscale (all were upland forests adjacent to flatwoods ponds in the Chicago Region) but differed substantially on a microscale. Thus, it is not unexpected that our results differ from those of previous studies. Alternatively, it may imply that other unmeasured variables are important to salamander density, or that salamanders are surviving in vestigial habitats. Adult salamanders are long-lived, and gradual change in habitat may have delayed impacts on salamander density, in contrast to rapid change such as logging. Without longterm data to determine trends (increased or decreased density over time at each site) it is difficult to determine factors responsible for different amphibian abundances in these forests adjacent to flatwoods ponds.

We were interested in whether the quality of the upland forest community was related to amphibian abundance and diversity. We predicted that ‘high’ quality sites would support more individuals and more species than ‘low’ quality sites. Amphibian species richness and diversity were both significantly higher in ‘high’ quality forests than in ‘low’ quality forests, supporting the second part was densely vegetated. Based on this study, and the additional drift fence work in MacArthur, we conclude that 1) There is substantial variation in salamander and amphibian abundance among the

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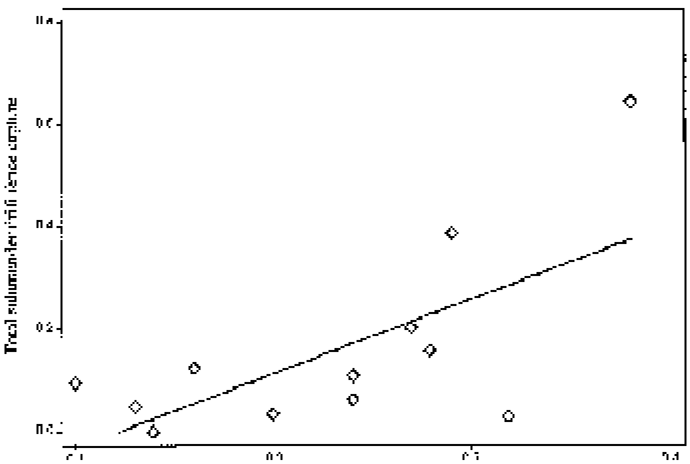


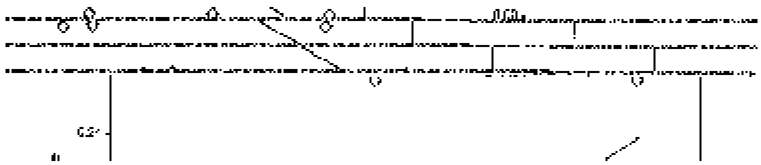


**Figure 3a**. Toad abundance relative to pond drydate.

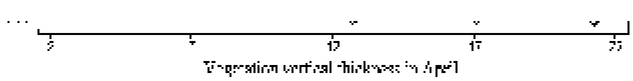
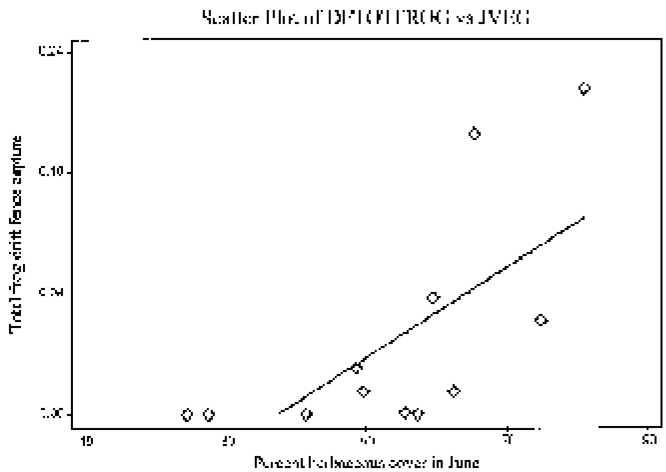



**Figure 3b**. Toad abundance relative to vegetation vertical thickness in April.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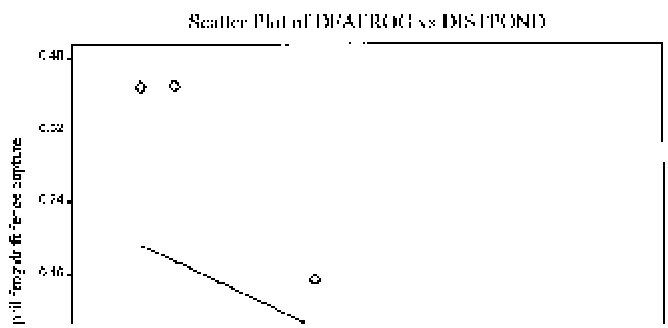
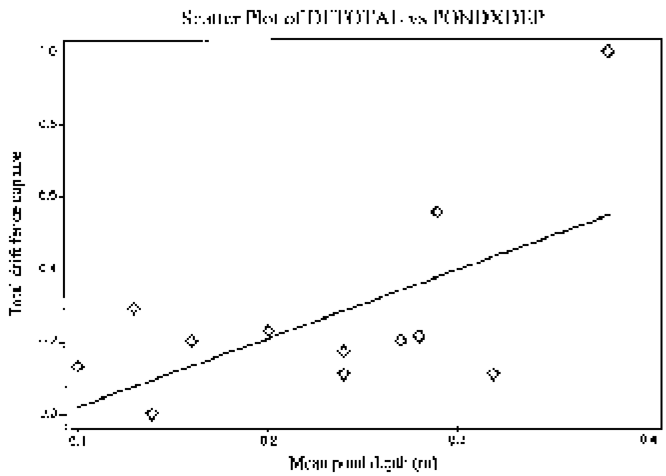
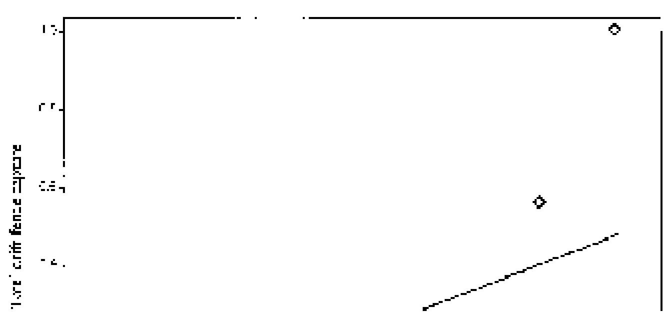
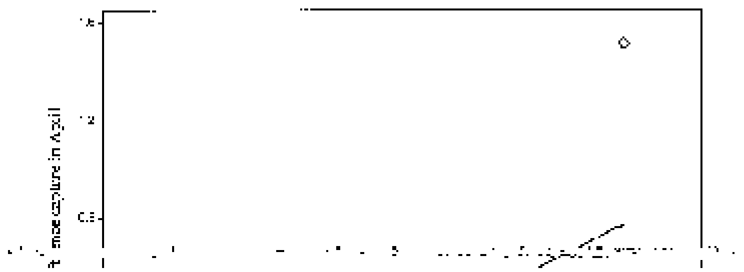
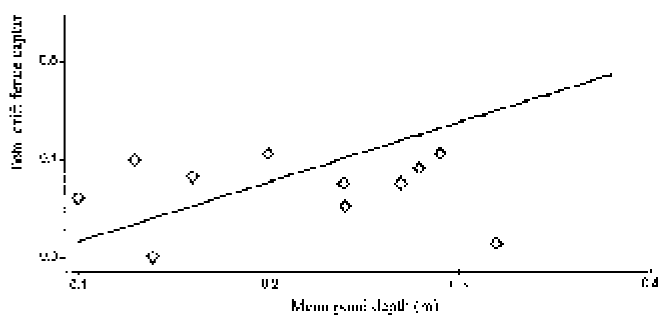
Percent herbaceous cover in June

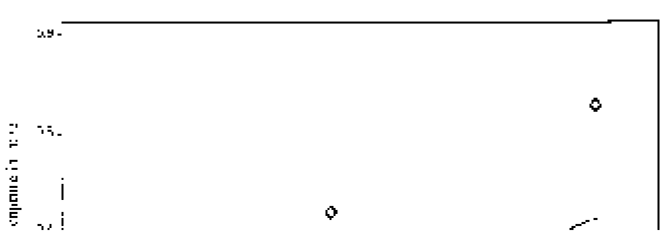

Scatter Plot

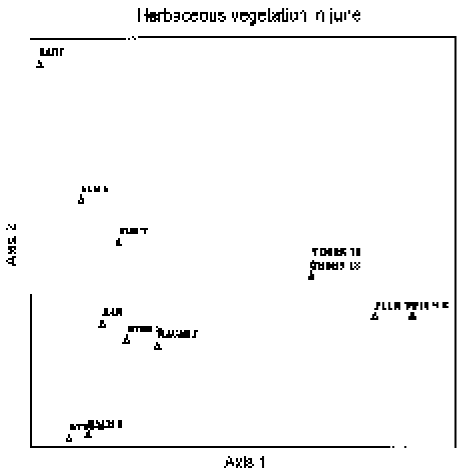

Scatter Plot of DFTOTAL vs PONDXDEP
Mean pond depth (m)

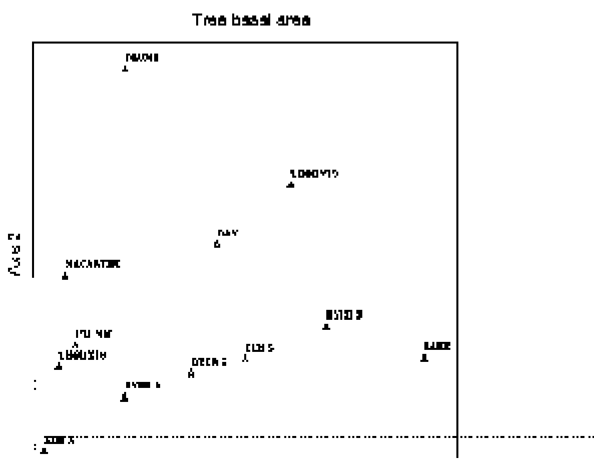





0.1
0.2
0.4






. Decorana of overstoryý Tees.

Figure 7a Decorana of groundlayer vegeta-

**Appendix A.** Common and Scientific Names of Amphibians and Reptiles Observed During This Study. Nomenclature follows Collins, J. T., Standard common and current scientific names of amphibians and reptiles. Society for the Study of Amphibians and Reptiles Herpetological Circular

**Amphibians**

| | |
|---|---|
| Blue-spotted salamander | *Ambystoma laterale* |
| Spotted salamander | *Ambystoma maculatum* |
| American toad | *Bufo americanus* |
| Gray treefog | *Hyla versicolor* |
| Spring peeper | *Pseudacris crucifer* |
| Western chorus frog | *Pseudacris triseriata* |
| Green frog | *Rana clamitans* |
| Northern leopard frog | *Rana pipiens* |

**Reptiles**

| | |
|---|---|
| Brown snake | *Storeria dekayi* |
| Common garter snake | *Thamnophis radix* |