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

Victoria A. Nuzzo and Kenneth S. Mierzwa

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Principal Investigators:

Victoria A. Nuzzo
1403 Sovocool Hill Road
Groton NY 13073
vnuzzo@earthlink.net

Kenneth S. Mierzwa
1647 N. Clybourn Avenue
Chicago IL 60614-5507
kmier@ripco.com

Project Administration:

Citizens for Conservation
1086 Camellia Place
Fox River Grove, IL 60021
Thomas W. Vanderpoel

Funding Sources and Cooperating Agencies:

United States Environmental Protection Agency
Great Lakes National Program Office (GLNPO)
77 W. Jackson Boulevard
Chicago, IL 60604-3590
GLNPO Project GL98114
GLNPO Project Officer: Kent Fuller

Lake County Forest Preserve District
2000 N. Milwaukee Avenue
Libertyville, IL 60048
Jim Anderson

Forest Preserve District of Will County
P.O. Box 1069
Joliet, IL 60434-1069
Dave Mauger

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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 herbaceous vegetation in June, and 2) the presence of water was an important determinant of amphibian abundance. Amphibian abundance was most closely related to the length of time that ponds retained water (ponds with later dry dates permit a greater percentage of larvae to achieve metamorphosis), and the number of ponds within 0.5 km. These results indicate that hydrology is the dominant force driving amphibian populations in upland forests in the Chicago Region, and forest structure is important only when hydrology is suitable. By implication, amphibian populations in sites with suitable hydrology (clustered ponds, one or more that retain water through mid-July) and unsuitable structure (low herbaceous cover in June) may benefit if the vegetational structure is managed. We propose two ways to test this premise. First, expand the current study to include multiple ponds within several sites, over several years. Second, actively manage half of the adjacent upland forests for increased groundlayer vegetation, and monitor the response of the amphibian community to these changes.

Specific site management strategies supported by the results of this study include removal of drain tiles and filling of ditches; restoration or creation of additional wetlands which hold water well into the summer but dry in at least some years; management for increased leaf litter in spring and increased herbaceous vegetation in summer; and understory thinning through removal of exotic or weedy shrubs and saplings, and judicious use of prescribed fire. Pending the outcome of current research on fire effects, we encourage either very early spring or late fall burns (when few amphibians are surface active), and either a conservative fire-return interval or use of multiple burn units around the best amphibian breeding wetlands.

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 at least six local amphibian species (Mierzwa 1998; Phillips et al. 1999). Many amphibians are non-migratory or short distance migrants (Phillips and Sexton 1989; Madison 1997) and have small home ranges (Kleeberger and Werner 1983). Habitat-restricted species, such as *Ambystoma maculatum* and *Rana sylvatica* are likely more impacted by habitat degradation than habitat generalist species, such as *Bufo americanus* and *Rana catesbeiana*.

This study was an investigation of the relationship between forest structure (and, by implication, natural quality) and amphibian diversity and abundance in the Chicago Region. Specific research questions were: 1) Is there a significant difference in abundance or diversity of amphibians in high vs low natural quality forests?; and 2) If so, what factors are associated with higher amphibian abundance or diversity? Adult pond-breeding salamanders spend the majority of the year in upland habitat, in underground refuges (Semlitsch 1998) with occasional intervals of surface movement and foraging (Madison and Farrand 1997). We therefore also investigated whether the ‘natural quality’ of the upland forests affected salamander survival, specifically, were adult salamander numbers higher in high quality forests than low quality forests adjacent to breeding ponds. We hypothesized that high quality natural sites would support more species, and higher abundance, of amphibians than low quality sites. Further, we hypothesized that high quality sites would be more likely to support “habitat restricted” species, and low quality sites would support “habitat generalist” species. This information is critical for the long-term preservation of amphibians within the Chicago region, where many forest species survive in relatively isolated populations within existing preserves.

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METHODS

Study sites were selected that: 1) consisted of a minimum of 40 ha of contiguous wooded habitat in public ownership; 2) contained one or more known or probable amphibian breeding ponds, defined as ephemeral ponds at least 10cm deep and 20m diameter in spring 1999. Plots were located at least 50m from an edge, defined as a road, trail, housing development or field, and at least 500m from any other study site (*Ambystoma maculatum* travels a mean of 125m between upland and breeding habitats, and 95% remain within 164m of the breeding pond; Semlitsch 1998); and 3) could be paired on the basis of assumed natural quality (INAI grade A or B vs INAI Grade C or D) with another study 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.

Grade C: Mid-successional or moderately to heavily disturbed communities; for example, a heavily grazed old-growth forest, or a young to mature second-growth forest.

Grade D: Early successional or severely disturbed communities; for example, a recently clearcut forest, or a mature second-growth but severely grazed forest.

Figure 1. Sample Plot Locations

Grade B forests as our “high quality” sites. We found only one pair of sites that met the selection criteria (Ryerson 5 and Lake-Cook). Consequently, we expanded the selection criteria to include dry-mesic, mesic, and wet-mesic forests, and also a range in ‘low’ natural quality (Grade C and Grade D). Thus, within pairs there was a distinct difference in natural quality, with one plot obviously more degraded than the other, but among all plots this distinction was less evident, and the plots formed a gradient of both natural quality and community type. Establishing study site criteria in the office helps focus the search for suitable sites, but locating sites that meet these criteria is often difficult, with the result that site selection criteria must often be expanded to allow a minimum number of replicate study sites (Petranka 1994).

Table 1. Sample Plot Coordinates

Plot Name	County	Latitude	Longitude	UTM E	UTM N
MacArthur	Lake	42 14 42	087 55 54	423084	4677406
Daniel Wright	Lake	42 12 50	087 55 22	423826	4673943
Elm North	Lake	42 13 00	087 54 52	424529	4674216
Elm South	Lake	42 12 55	087 54 44	424711	4673964
Ryerson North	Lake	42 10 50	087 54 27	424991	4670295
Ryerson South	Lake	42 10 20	087 54 17	425280	4669339
Ryerson 5	Lake	42 10 35	087 54 21	425187	4669773
Lake-Cook	Lake	42 09 06	087 54 05	425480	4667096
Plum West	Will	41 27 02	087 33 44	453131	4588934
Plum East	Will	41 27 10	087 33 27	453448	4589126
Thorn 19	Will	41 27 43	087 40 58	442977	4590262
Thorn 13	Will	41 27 29	087 40 52	443170	4589840

Within each site a single 0.25ha (50m x 50m) plot was located, with the plot center approximately 25m from the edge of the pond. We initially intended to establish 1ha plots, but found that most ponds were located less than 100m from an edge or disturbance. In the Chicago region virtually all large forested tracts that contain ephemeral/flatwoods ponds are publicly owned, and the majority have trail systems that traverse the entire tract, leaving few areas sufficiently isolated from trails and edges to meet the site selection criteria. Therefore, plot size was reduced.

We chose to conduct high intensity sampling (both amphibians and vegetation) in a relatively low number of plots (n=12), given the tradeoffs between number of replicates, plot size, and sampling effort (Hairston 1989 in Petranka 1994), and the difficulty in locating suitable study sites.

Data Collection, Amphibians and reptiles — Amphibians and reptiles were sampled with drift fences and time-constrained visual encounter surveys (Heyer et al., 1994; Sutherland, 1996), with drift fences installed at least one week prior to sampling activities. A single drift fence array was installed at the center of each sample plot, oriented parallel to and approximately 25m distant from the pond. Drift fences were constructed of aluminum flashing, 30m long and 50cm high, embedded several cm into the substrate. The array included two funnel traps constructed from cylinders of aluminum window screening and plastic funnels, one placed at each end of the drift fence, and two 5l buckets buried flush with the substrate surface at the center of the fence, one placed on each side 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 captures than during the earlier in-migration period. However, post-breeding animals are presumably moving more slowly and spending time foraging, and thus give a better representation of terrestrial habitat use.

All captured animals were identified to species and released away from the fence in the direction of original movement to minimize chances of recapture. Because most movement is directional, either toward or away from the pond (Dodd and Cade 1998), we assumed that placement of the animal on the opposite side of and several meters from the drift fence was sufficient to prevent the same animals from being recaptured. Results are reported as catch per trap night, with a trap night being the equivalent of a 24 hour period of sampling with each 30m long drift fence. When drift fences were not in use funnel traps were removed and buckets covered.

Time constrained visual encounter surveys were conducted at each site within 48 hours of rainfall by two trained observers on four visits between April 16 and June 1, 1999. Search area centered on the drift fence and covered the entire plot on each visit. Each round of sampling was conducted by the same individual(s) at all plots, to minimize bias. The observers turned logs and other cover objects, and observed animals under cover or active and in the open (Welsh and Lind, 1991; Churchwell and Mierzwa, 1998). Results are reported as catch per person hour.

Data Collection, Vegetation — Structure and composition of each forest was recorded within the 0.25ha (50m x 50m) plot centered on the drift fence array, using a systematic sampling design (Elzinga et al 1998). Five parallel 50m transects were established along a baseline parallel to and 25m distant from the drift fence, and more or less following the pond edge; thus, transects bisected the drift fence and extended from pond edge 50m into the forest. The first transect was randomly located within the first 10m interval along the baseline, and the remaining transects were systematically located at 10m intervals. Groundlayer data were recorded in 25 permanent $1m^2$ quadrats, five per transect; the first quadrat was randomly positioned within the first 10m of transect, and the remaining four quadrats were then systematically located at ten meter intervals. Groundlayer data consisted of presence and estimated cover (within 13 cover classes) of all vascular species <1m tall, and of exposed soil, wood, and leaf litter.

Shrub and tree data were recorded in 13 circular $100m^2$ (5.78m radius) quadrats centered on alternate groundlayer quadrats. Density was recorded by species for all woody plants >1m tall and ≤10cm dbh in three size classes; <1-2m tall; >2m tall and <5cm dbh; and 5-9.9cm dbh, and density and diameter at breast height were recorded by species for all trees (≥ 10 cm dbh). Groundlayer data were recorded in both April and June as we anticipated seasonal changes, while shrub and tree data were recorded only in April.

Abiotic features were recorded within the $1m^2$ quadrats. Litter depth, canopy cover, and vegetation "thickness" were recorded in April and June. Litter depth was measured to the nearest cm at four 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.

Data collection, wetland — Surface area of the pond was measured in the field and from aerial photographs. Depth was recorded at 5m intervals beginning at the pond edge and extending across the pond, along three transects parallel to vegetation transects and at right angles to the center of the drift array.

Data collection, landscape -- Features potentially affecting amphibian and reptile metapopulations were measured from one inch = 400 foot black and white aerial photographs, supplemented with coarser scale color infrared photos for most sites. The number of known or potential amphibian breeding wetlands within 0.5km was noted; this distance was chosen based on the greatest documented dispersal distance for juvenile blue-spotted salamanders in the Chicago Region (Mierzwa and Beltz, 1999). Also measured was the distance to the nearest known or probable breeding wetland, and the distance to the nearest forest edge.

Data Analysis: Amphibian data consisted of a single value/site, and therefore we used the mean value (average among quadrats at each site) for each environmental variable in all statistical tests. Because sites formed a gradient of natural quality and community type, we used stepwise multiple regression (using drift fence data) to determine if specific habitat features of the upland forests were associated with higher amphibian abundance and diversity. Variables for each regression were selected with the Best Subset Regression procedure. We used two-tailed t-tests to determine if 'high' quality sites collectively differed significantly from 'low' quality sites in amphibian abundance and diversity, and biotic and abiotic variables. We used paired t-tests to determine if plot pairs had similar between-plot differences in amphibian abundance and diversity, and biotic and abiotic variables.

Plots were ranked for amphibian habitat quality using total drift fence data (spring and summer combined). Two methods were used; the first ranked sites from high to low on the basis of total drift fence abundance, combining salamanders, toads, and frogs. The second method independently ranked sites from high to low for salamanders (two relatively specialized forest habitat species), toads (one habitat generalist), and frogs (three species, two usually associated with herbaceous vegetation and one with woodland habitat), and then summed the three ranks. Both methods produced similar rankings of sites. The six sites with the highest abundance of amphibians were classified as 'good' habitat and the remaining six sites as 'poor' habitat. Two-tailed t-tests were then used to test for significant differences between the ‘good’ and ‘poor’ amphibian sites. Considering toads (Bufonidae) and frogs (Hylidae and Ranidae) separately is a somewhat artificial split in a taxonomic sense; however it does take into consideration the presumed physiological and behavioral adaptations of toads for an existence in relatively xeric conditions.

All data were tested for homogeneity of variance, then tested for significant differences using parametric (t-tests) or non-parametric (Kruskal-Wallis) tests as appropriate. Statistical analysis was conducted with Statistix (Analytical Software 1996). Reptile data are presented in tables butwere not included in statistical analysis as the project focused on amphibian use of upland forests.
Detrended correspondence analysis was conducted on groundlayer and overstory data to determine if stands clustered on the basis of natural quality or amphibian abundance and diversity, and to assess which variables were most closely associated with a) natural quality and b) amphibian richness and density. Multivariate analysis was conducted with PC-ORD (McCune 1993).

RESULTS

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; Table 2). The highest abundance and diversity were recorded at Elm North, where 42 salamanders, nine toads, and 14 frogs were captured. The lowest abundance was recorded at Lake-Cook, where no amphibians were captured during the study period. Most amphibians were recorded during the spring capture period; 80% of salamanders, 41% of toads, and 92% of frogs.

Time constrained visual encounter surveys resulted in a total of 74 captures at all sites (2.5 hours/ site; Table 2). Ninety-six percent of captures were of one species, *Ambystoma laterale*. Single individuals were captured of *Pseudacris triseriata*, *Rana pipiens*, and *Thamnophis sirtalis*.

Multiple linear regression indicated that just seven of the tested variables explained most of the differences in amphibian abundance (Table 3). Three of these variables reflected vegetation structure; percent cover of herbaceous vegetation in June, percent cover of leaf litter in April, and horizontal vegetation thickness in April, while four of the variables reflected the presence of water; average pond depth, pond drydate, number of ponds within 0.5km, and distance to the nearest pond. Pond drydate was positively and significantly correlated with both pond depth and pond number ($p<0.01$). The former correlation is expected, as deeper ponds tend to retain water longer, but the latter correlation is likely an anomaly dependent on two sites (Thorn 13 and Thorn 19) that retained water throughout the study period, and were also near a large number of other ponds. When these two sites were omitted, no correlation was detected between pond number and dry date ($p=0.62$), while the correlation between pond depth and drydate remained strong ($p<0.01$). Therefore, only pond drydate was used in multiple regressions, when both drydate and pond number were identified by the best subset regression procedure.

Salamanders, primarily *Ambystoma laterale* and some *Ambystoma maculatum*, were recorded at 11 of the 12 sites (no amphibians were recorded at the 12^{th} site) and were the dominant amphibian group at seven sites. Eighty percent of salamanders were captured in April. Salamander abundance was 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)

logs, greater volume of well-decayed logs (3888 dm^3 vs 890 dm^3), and deeper leaf litter in June, than the 'low' quality sites. Tree density was nonsignificantly greater in the 'low' quality sites (3569/ha) than in the 'high' quality sites (2365/ha) although basal area was similar. This indicates that the 'high' quality sites had fewer but larger trees while the 'low' quality sites had numerous smaller trees. The presence of large trees was one of the characteristics used to define 'high' natural quality. The 'low quality' sites had nonsignificantly more alien or nonindigenous trees (26% vs 7%), than the high quality sites. The 'low' quality sites also had nonsignificantly more shrubs (4234/ha vs 1785/ ha) and a higher percent of nonindigenous shrubs (5% vs 17%). The presence of nonindigenous trees and shrubs was one of the characteristics used to define 'high' and 'low' natural quality.
Paired t-tests indicated that within each pair of plots, the 'high' quality site had significantly more leaf litter cover and lower tree canopy cover in April, and deeper and more leaf litter cover and less exposed soil in June, and more down wood, than the 'low' quality site (Table 4). Nonindigenous shrubs were significantly denser in the 'low' quality site of each pair, as would be expected, as the abundance of nonindigenous shrubs was one of the characteristics used to define 'low' natural quality.

Decorana of groundlayer vegetation in both April and June separated the Will County sites from the Lake County sites along the first axis, and grouped plots within preserves along the second axis (Figure 7a). Decorana of overstory trees separated the 'high' quality sites from the 'low' quality sites along the first and second axes (Figure 7b). No correlations were detected between amphibian abundance or diversity and forest composition.

DISCUSSION

Amphibian abundance was strongly influenced by the presence of water. Sites with deeper ponds that dried later in the summer supported more amphibians (especially salamanders and toads) than shallow ponds that dried early in the summer. Sites that were located near other ponds also supported significantly more salamanders and toads. Interestingly, frog (but not salamander or toad) abundance was significantly related to the distance to the nearest pond; frog abundance in April increased as the distance to other ponds decreased. Neither the size of the pond (surface area in square meters) nor the distance to the forest edge was associated with any of the amphibian measures.

The importance of water to amphibian abundance was not unexpected, as all amphibians encountered in this study are pond-breeders. The relationship between upland forest vegetation and amphibians was surprisingly simplistic; sites with greater herbaceous cover in June supported more amphibians (especially salamanders and frogs) than sites with less herbaceous cover. While salamanders have long been associated with abundance of down wood (Welsh and Lind 1995, Dupuis et al, 1995), we found no relationship between salamanders and the number of logs, the area of log contact, the amount of down wood, or the abundance of well decayed wood.

The number and diversity of amphibian species were also closely correlated with June herbaceous cover and pond drydate, and with two additional features; cover of leaf litter in April, and distance to the nearest pond. Sites with $\leq$80% leaf cover supported zero to one species, while sites with >90%

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 of our hypothesis. The first part of the hypothesis was not supported: Although the six ‘high’ quality sites (INAI Grade B) supported more than twice the number of salamanders, frogs, and toads than the six ‘low’ quality sites (INAI Grade C and D), these differences were not significant (Table 4).

Rather than two distinct groups of ‘high’ and ‘low’ natural quality in the same community type, our 12 study sites formed a gradient in natural quality within three community types. We believe this is an artifact of the low number of sites (12) and the inherent variability between natural communities and along the quality gradient, resulting in considerable noise in the data set. This problem could be addressed in future by including a much larger number of sites.

Sites varied substantially in vegetation structure and composition. This heterogeneity may have obscured any relationship between natural quality and amphibian abundance. Sites also varied in natural quality. While the two Grade D sites (Lake-Cook and Thorn13) supported the fewest amphibians, and two of the Grade B sites (Elm North and Thorn 19) supported the most amphibians, the remaining eight sites did not follow a consistent pattern. Three Grade C sites (Daniel Wright, Plum East, and Ryerson South) supported more amphibians than their paired Grade B sites. When only salamanders were considered, two of the Grade C sites (Daniel Wright and Ryerson South) still supported more animals than their paired Grade B sites. We conclude that the suite of characteristics used to determine natural quality are not necessarily the features that characterize suitable upland habitat for amphibians in general and salamanders in particular, although there may be substantial overlap.

Natural quality is an arbitrary and qualitative assessment of site degradation. Sites with a history of logging, grazing, fire exclusion, alien species invasion, etc., are deemed to have lower natural quality (less resemblance to presettlement conditions) than sites without these impacts. While this concept intuitively holds true, no studies have been conducted to document and substantiate this assumption. Basic parameters such as tree density, basal area, and groundlayer species richness, are insufficient measures of natural quality. Likely, a combination of factors, including abundance of species considered ‘conservative’ vs ‘disturbance-adapted’, density of trees in a range of size classes, age since disturbance, and degree of disturbance (including both direct anthropogenic disturbance, such as logging and grazing, and indirect anthropogenic disturbance such as excessive white tailed deer herbivory and localized lowering of the water table) will be necessary to verify the validity of the natural quality assessment.

This study was preliminary, looking only at 12 sites. Each preserve had several ponds to select from, and within forested tracts monitored ponds were selected more or less at random. Selecting different ponds would have produced different results; we suggest increasing the number of study sites, and monitoring multiple ponds within a forest to obtain insight into the actual relationships between ponds, upland forest, and amphibian abundance and diversity. In March 2000 (after this study was concluded) we established drift fences at an additional four ponds at MacArthur. Capture rates for a single night (March 8) ranged from none at the 1999 pond to 75 individuals at a pond just 400m north. The 1999 pond was dry on March 8 2000, and the adjacent upland forest had very little herbaceous vegetation. The new pond had shallow water, and part of the adjacent flatwoods forest 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 ponds within individual sites. 2) This variation is related to the length of time that an individual pond holds water and the number of nearby ponds, which in turn is affected by site-wide hydrologic conditions. Hydrology is likely the primary limiting factor for Chicago Region amphibians; 3) This variation is also related to the abundance of groundlayer vegetation in the adjacent forest. We suggest that forest vegetation structure is a limiting factor, but only IF a site is sufficiently large and with several ponds to allow between-pond migration, and IF some of those ponds hold water long enough to allow larvae to achieve metamorphosis.

It may be possible to test the relationship between upland forest structure and amphibian success. Elm North, with the highest amphibian abundance of the 12 sites, was the only actively managed site; understory saplings were removed and the site had been prescribed burned in prior years. Other researchers have assessed amphibian response to natural area management activities, including shrub removal and prescribed burning, and documented a positive response for at least some species (Mierzwa, 1997; Palis, 1994; Kirkland et al., 1996). We suggest that similar management be conducted on Elm South, and the abundance of amphibians and community structure be monitored over time in both sites. If the relative abundance of amphibians in Elm South increases with management, then the characteristics of the managed sites may be assumed to provide better upland forest habitat. Conducting this same study in two or more sites (we suggest MacArthur Woods and Thorn Creek) would provide replication and permit a broader application of results.

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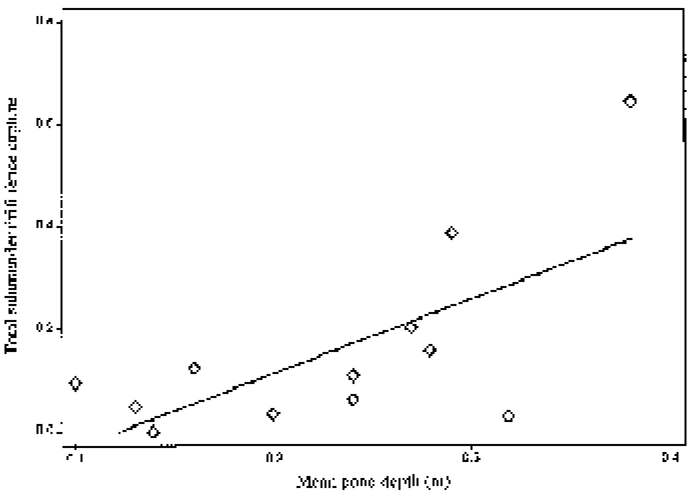

Figure 2a. Salamander abundance relative to pond depth.

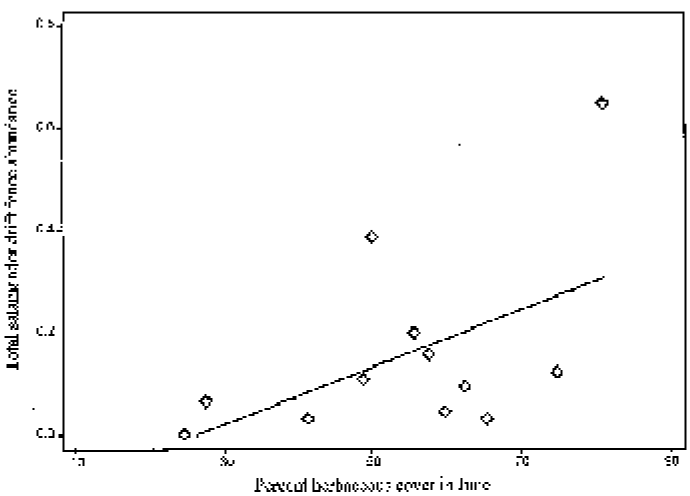

Figure 2b. Salamander abundance relative to herbaceous vegetation.

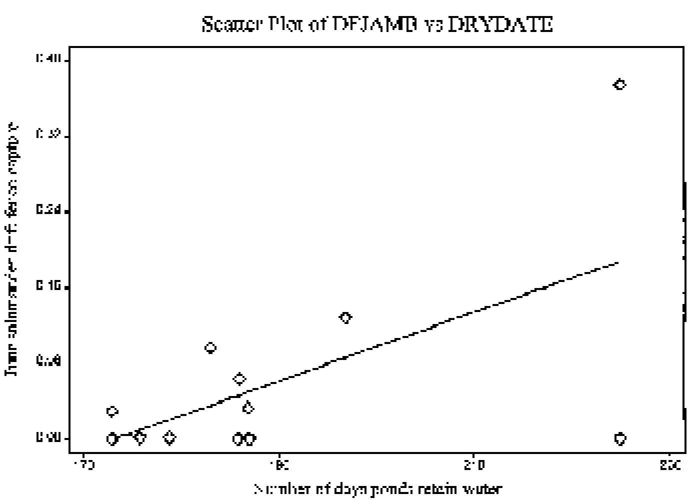

Figure 2c. Salamander abundance relative
to pond drydate.

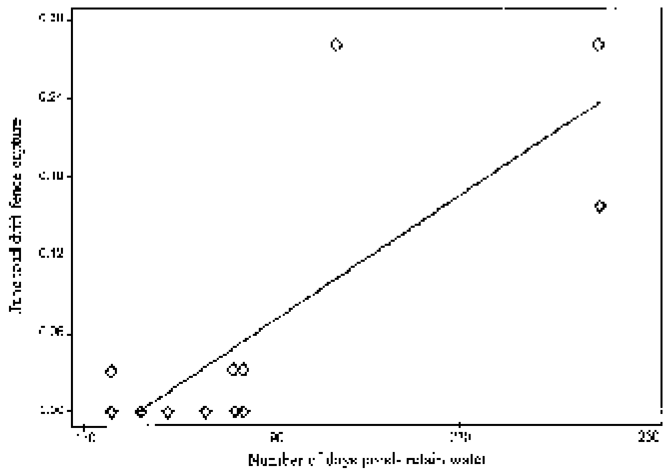

Figure 3a. Toad abundance relative to pond drydate.

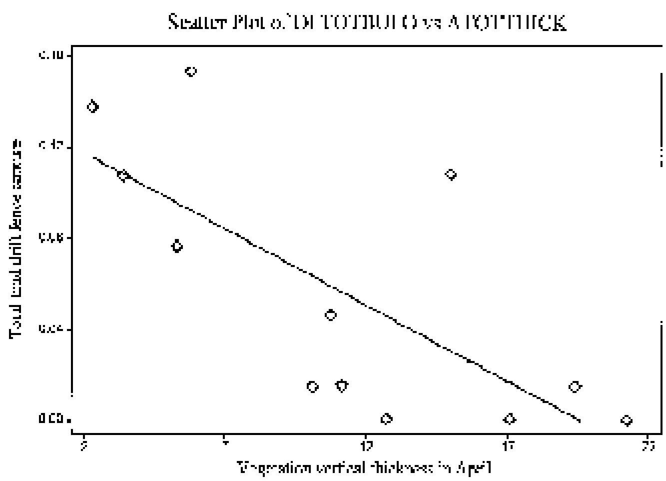

Figure 3b. Toad abundance relative to
vegetation vertical thickness in April.

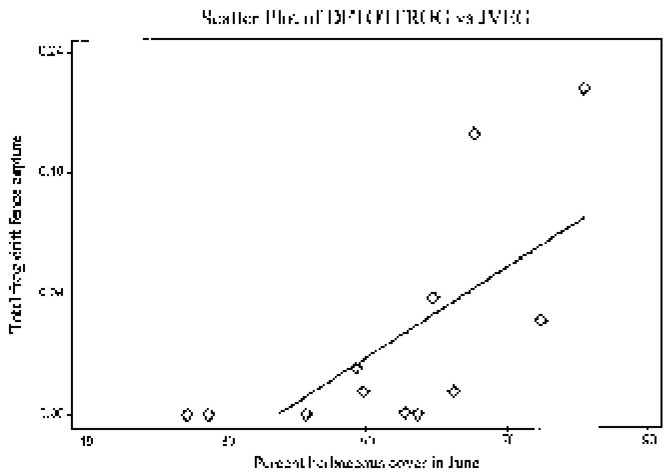

Figure 4a. Frog abundance relative to herbaceous cover in June.

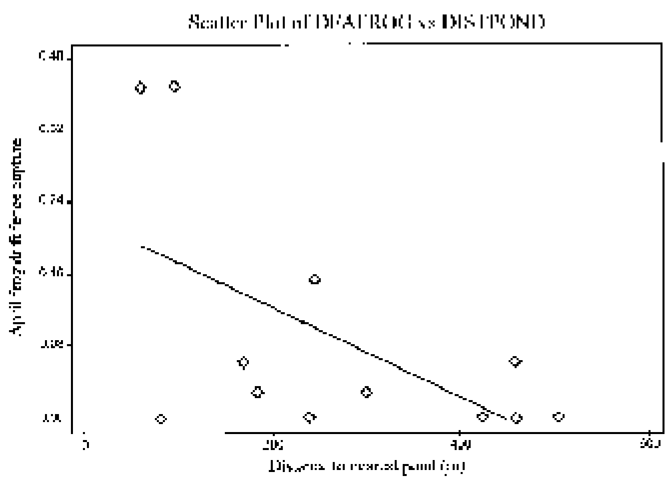

Figure 4b. Frog abundance relative to distance to nearest pond.

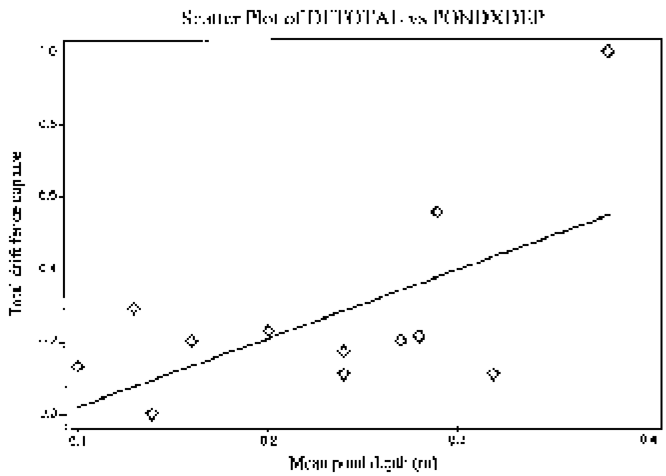

Figure 5a. Total amphibian abundance relative to mean pond depth.

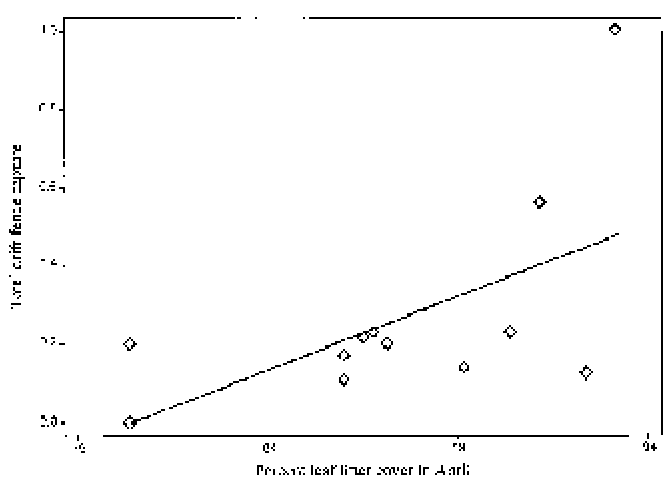

Figure 5b. Total amphibian abundance relative to leaf litter cover in April.

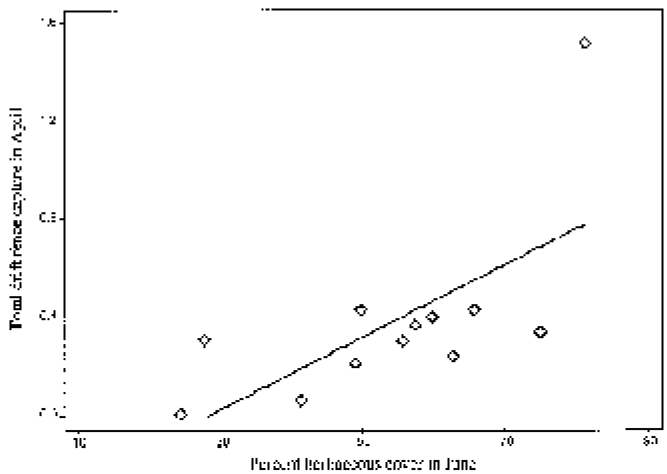

Figure 6a. Total drift fence captures in April relative to herbaceous cover in June.

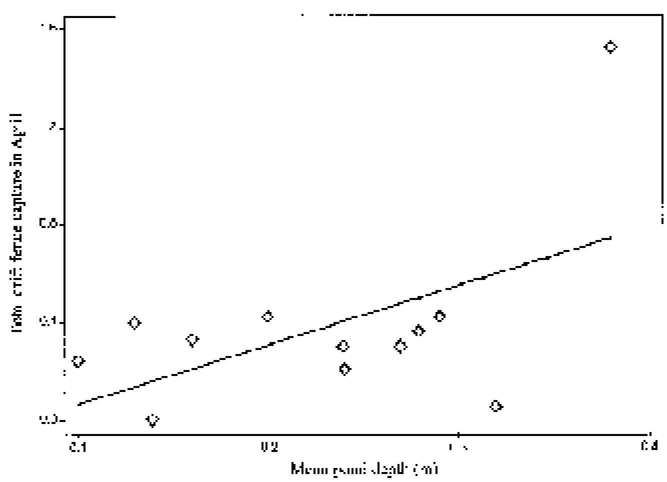

Figure 6b. Total drift fence captures in April
relative to mean pond depth.

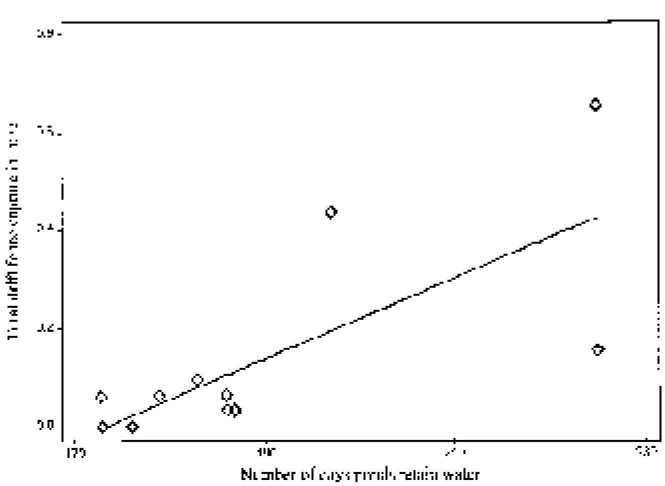

Figure 6c. Total drift fence captures in June
relative to pond drydate.

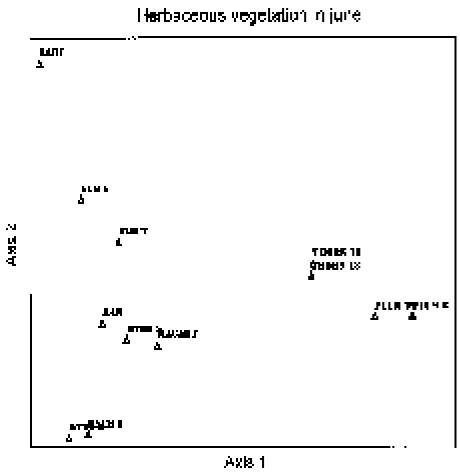

Figure 7a. Decorana of groundlayer vegetation.

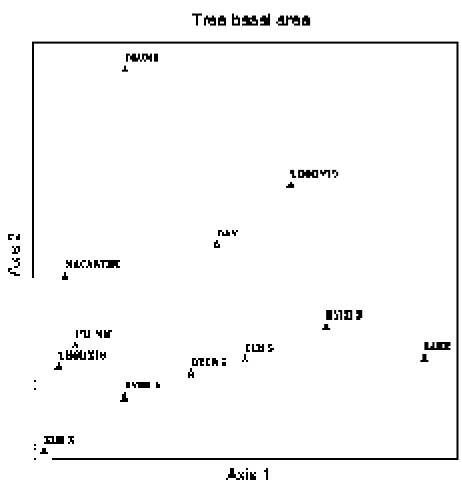

Figure 7b. Decorana of overstory trees.

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*