CRITICAL TRENDS ASSESSMENT PROGRAM

2003-04 REPORT

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DEPARTMENT OF
NATURAL
RESOURCES

CRITICAL TRENDS ASSESSMENT PROGRAM
2003-04 REPORT

Illinois Department of Natural Resources

Office of Realty and Environmental Planning
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- Cache River
- Calumet Area
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- Driftless Area
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- Illinois Big Rivers
- Illinois Headwaters
- Illinois River Bluffs
- Kankakee River
- Kaskaskia River
- Kinkaid Area
- Kishwaukee River
- La Moine River
- Lower Des Plaines River
- Lower Sangamon River
- Lower Rock River
- Mackinaw River
- Mississippi Western Five
- Prairie Parklands
- Shawnee Area
- Sinkhole Plain
- Spoon River
- Sugar-Pecatonica Rivers
- Thorn Creek
- Upper Des Plaines River
- Upper Rock River
- Upper Sangamon River
- Vermilion River
- Vermilion River (Illinois River Basin)

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Introduction

The Critical Trends Assessment Program has completed its first five-year cycle (1997-2001) and started its second cycle (2002-2006) of monitoring Illinois ecosystems. EcoWatch volunteers have also for eight years (1996-2003) collected data that complements the professionally-collected data. Some clear patterns have been apparent from the beginning, such as the highly disturbed nature of most Illinois ecosystems and the invasion of exotic species. This report starts to take a deeper look at these patterns and begins to look for trends. Future reports will explore the CTAP ecosystem monitoring data in even more depth to provide information to help better manage the natural resources of Illinois.

Terrestrial Ecosystems

Five papers examine the data collected on *terrestrial* ecosystems – that is forests, grasslands, and wetlands. One paper finds the seeming paradox that more mature second growth forests have less plant diversity than young second growth forests. Another shows that one in six upland sites is suffering from maple takeover and endeavors to quantify the diagnosis of this endemic problem. Another demonstrates that while ForestWatch sites in the LaMoine River are more mature forests than those in the Vermilion Watershed Task Force Ecosystem Partnership, invasive shrubs plague both areas. A fourth paper shows that sites with larger forest patches and surrounded by large amounts of forested land in the overall landscape support more area-dependent and habitat-dependent bird species, while sites in parts of the state dominated by small and isolated forest patches surrounded by agriculture support fewer bird species. A final paper finds significant differences among various Illinois habitats in the number of insect and arthropod species.

An analysis of the botanical data collected by CTAP scientists in the 133 forest sites monitored from 1997-2001 revealed some surprises. While conservative species – those generally associated with higher ecosystem quality– were more often found in relatively species rich sites, they were also, surprisingly, more often found in less mature rather than more mature forests. Species richness actually declined with increasing stand maturity. This is consistent with models of forest succession showing that plant diversity is initially highest because of the presence of old-field weeds, but then decreases as the forests mature and shade increases. Basal area is clearly not an accurate indicator of the conservation value of forest floras, nor can it be used as a stand-alone measure of forest biological quality. Given enough time, diversity should increase again when forests mature into old-growth, but ground layers in human dominated landscapes are not likely to recover their full array of wildflowers within observable lifetimes.

One of the intended uses of the ForestWatch data has been to track the extent of maple takeover around the state. Diagnosing sites as suffering from maple takeover, however, is more of an art than a science. In general takeover sites are upland forest sites dominated by oak-hickory in the large tree classes, with maple largely missing from the large classes and dominant in the small classes. Based on these somewhat subjective criteria, of the 194 upland forest monitoring records, 32 or one in six indicate a problem with maple takeover. Examining the data from these sites allows one to quantify the criteria for identifying maple takeover: a ratio of sugar maple abundance to total sugar maple/oak/hickory abundance between 17% and 67%, a high importance value of oak-hickory, presence of 5-10 cm sugar maples, and absence of 40-60 cm sugar maples.

In 2003, the LaMoine River Ecosystem Partnership and Vermilion Watershed Task Force Ecosystem Partnership began incorporating ForestWatch and PrairieWatch into their partnership planning. They are trying to establish a baseline, determine management needs, and monitor the results of management decisions. ForestWatch data show that LaMoine River forest sites have fairly low tree abundance and higher than average basal area, indicating relatively mature forests with larger but fewer trees. On the other hand, the Vermilion forest sites show very high tree abundance but low basal area and shading level, indicating a young forest of relatively small trees. However, ForestWatch found that the forest sites in both partnerships suffer from invasive shrubs. The LaMoine is plagued most by multiflora rose and gooseberry, and likely honeysuckle as well, while the Vermilion is clearly infested by all three, with 90% of the recorded shrubs being invasive.

To detect patterns in bird distribution, several different geographic scales were examined including the three major regions (north, central, and south), 14 natural divisions, 21 IEPA basins, and 40 Ecosystem Partnerships. The southern region of Illinois had more habitat dependent and area sensitive species than the northern and central region of Illinois. Three natural divisions in southern Illinois showed the highest bird species richness, including the Shawnee Hills, Ozarks, and Lower Mississippi River Bottomlands. The most diverse basin is the Mississippi South and most diverse ecosystem partnership is the Shawnee, both encompassing parts of this same area. All of these geographic areas are characterized by large, continuous, forest tracts that generally are a part of the Shawnee National Forest. The Rock River Hill Country natural division and the Rock, Kankakee, Sangamon, and Fox basins scored lowest in all measures of avian species diversity. Among the Ecosystem Partnerships, the Upper Salt Creek of the Sangamon had the least habitat dependent and area dependent species. These areas with low bird diversity are characterized by small and isolated forest patches that are surrounded by agriculture. Overall, patch size has the greatest impact on total bird species richness, while habitat dependent and area sensitive species are influenced also by the amount of forested land in the surrounding landscape and such factors as human disturbance, forest structure, age of tracts, and tree species diversity.

CTAP monitors terrestrial arthropods with special emphasis on a group of sap-sucking herbivores, Auchenorrhynchous Homoptera or AH (which include leafhoppers, planthoppers, spittle bugs, and treehoppers), that are highly diverse and abundant in most terrestrial habitats. The collection of terrestrial arthropods by CTAP between 1997-2001 has provided invaluable data on differences between habitats. Wetlands showed the lowest diversity in several arthropod orders – Arachnida (spiders), Diptera (flies), Coleoptera (beetles), Hymenoptera (bees, wasps, ants), and Lepidoptera (butterflies, moths) – due most probably to the lower plant diversity. Many CTAP wetland sites are highly disturbed and very small. Wetlands had higher numbers of species for only a couple of orders, namely Orthoptera (roaches, crickets) and Heteroptera (aphids, waterbugs). AH species were most abundant in grasslands, followed by forests and then wetlands. Monitoring of the AH species also detected as many as 28 exotic species as well as set 191 county and a few state records.

Aquatic Ecosystems

Some similar patterns and trends have been revealed by three papers analyzing the *aquatic* ecosystems. Data collected by CTAP and RiverWatch citizen scientists show that the highest quality streams are those in southern Illinois, particularly in the Shawnee Hills and Ozark natural divisions. The lowest quality streams are in the Northeast Morainal natural division, which covers the most urbanized part of the state, and some of the natural divisions in agricultural areas of the state. Enough data have been collected to detect trends in some individual streams but no overall statewide trend has been revealed. Regionally, only southern Illinois where quality is highest do RiverWatch sites show a declining trend. A study of nearly 5,000 stonefly records including those collected from the 149 CTAP stream monitoring sites confirm that stoneflies have fared best in areas where large blocks of high quality habitat remain, such as the Shawnee Hills, the narrow bluffs of some large rivers and the ravines of the Wabash border, while the Grand Prairie which historically supported diverse communities of stoneflies has endured the greatest losses. Overall, the data show that state aquatic ecosystems are highly disturbed and more than one-fourth of stonefly species historically found in Illinois are now extinct or extirpated.

Now that CTAP scientists have completed the first five-year cycle of monitoring and volunteers have completed eight years of RiverWatch monitoring, we are in a position to not only make generalizations about stream quality around the state, but also to start analyzing trends. Reliable data are now available for 814 sites. Overall, the Ozark/Shawnee Hills natural divisions in southern Illinois have the highest stream quality, while the Northeastern Morainal in northern Illinois has the poorest. Most streams show no trend over the past eight years, although some display a statistically significant change. In most regions of the state there is no overall change in quality, since streams with a positive trend balanced streams with a negative trend. However, all of the streams that indicated a trend in the Ozark/Shawnee Hills natural divisions declined in quality. The possibility that stream quality in the most pristine area may be declining deserves further investigation.

CTAP scientists have studied the complete collection of nearly 5,000 stonefly records from the past 150 years including those collected by CTAP from its 149 stream monitoring sites. They found that Illinois has lost 18 stonefly species through extirpation and two species through extinction, mostly in the perlid and perlodid families, including two entire genera. Stoneflies have the highest known rate of extinction + extirpation of any aquatic animal in Illinois, at 26% of species compared to 21% for mussels and 6% for fishes. The 1940s and 1950s may have experienced the greatest decline in several stonefly genera. One genus, *Perlesta*, has become very abundant, even "weedy", while another that was formerly very common, *Acroneauria*, has declined dramatically. The number of stonefly species with slow life cycles, that is species in which nymphs survive from about 11 months to two or more years, were reduced disproportionately more than fast-cycle species, in which the nymphs are exposed to stream conditions for only a few winter or spring months. Stoneflies have fared best in areas where large blocks of high quality habitat remain, such as the Shawnee Hills, the narrow bluffs of some of the large rivers, and the ravines of the Wabash border. The Grand Prairie, historically supporting diverse communities of stoneflies, has endured the greatest losses in species due to disturbance from agriculture; similarly the Mississippi, Illinois, and Ohio rivers now only support a few of the more tolerant species due to widespread human disturbance. Stream quality trends seen by the CTAP stream monitoring program are consistent with the distribution of losses in stonefly fauna revealed by the INHS collection. CTAP found several high quality sites in the Shawnee Hills and Wabash Border natural divisions but many poor sites in the Grand Prairie.

The strong quality assurance component of RiverWatch has done much to ensure that high quality data is collected and that the program is well respected. However, the program is constantly evaluating the quality and looking for ways to improve the data and the training of the volunteers. One persistent problem has been low sample sizes at some sites. Nearly half of samples contain less than the target number of 100 organisms, and 10% of samples contain less than 25 organisms, not enough to accurately gauge a site's quality. An evaluation of a wide range of factors concluded that low sample sizes were not a function of volunteer ability or the amount of time they spent monitoring, but were more a function of habitat availability and macroinvertebrate population dynamics. Sites where snags or particularly undercut banks were the primary habitat sampled tended to have lower samples because fewer organisms live in those habitats compared to riffles or sediment. Also sites with certain endemic taxa tend to have large sample sizes because these taxa are naturally abundant where they occur.

Terrestrial Ecosystems

The Relationship between Forest Maturity, Forest Diversity, and Plant Distributions: Will Wildflowers Return to Disturbed Forests?

Greg Spyreas, Jeffrey W. Matthews, James Ellis,
Connie Carroll, and Brenda Molano-Flores

Introduction

Few biological measures are available that clearly and concisely compare habitats and their plant communities with one another. For example, measures of species diversity like species richness or species abundance, say nothing about which species are present or absent (species composition). It is often species composition that is of most concern for forest conservation in Illinois. One species composition measure that has been useful in studying habitats is termed "nestedness". Nestedness has been used to measure patterns of species presence or absence on different islands, to determine whether their distribution follows a pattern. If the species distribution on a group of islands is nested, the most nested species are only present on very species diverse islands and they are present nowhere else. Less nested species will be present on all islands (widespread, abundant, habitat generalists), or they will be distributed randomly (Figure 1) (Patterson 1987). If species distributions are not nested, it can be assumed that most or all of the species occur randomly, and they are not affected by island size or diversity. In Illinois, many patches of forest in a sea of human habitat that is inhospitable to forest plants are functionally "islands" of habitat (i.e. they are surrounded by row crops or urban land).

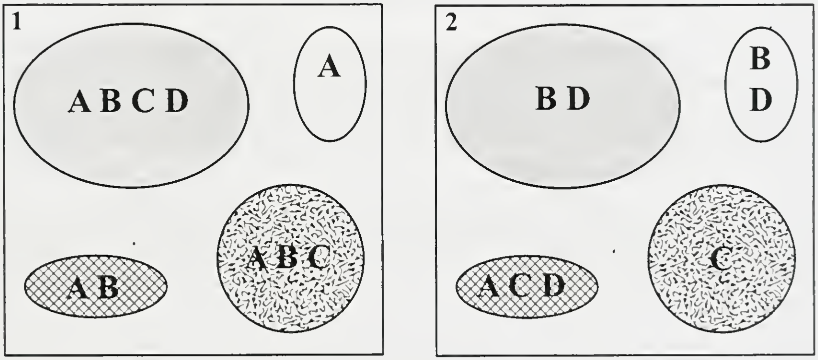

Figure 1. In Example 1, the species have nested distribution among the four islands with respect to island area and richness. Each set of species on an island is a perfect subset of the species of each larger island; as island size decreases certain species are lost. Species A represents a widespread generalist that is present in no particular pattern; it does not contribute to the nested pattern. Species B,C, and D are nested. Species D is the most area sensitive species, and may be a species of conservation concern because it requires large contiguous habitats. In Example 2, the species assemblage is not nested. Species are distributed randomly among islands.

The most commonly suggested causes for nested species distributions are: 1) Isolation--habitats are often isolated from one another, and 2) Area--the amount of habitat available for a species (Worthen 1996). According to these theoretical variables, in nested systems, the largest and least isolated habitat patches should contain the most species. However, the presence or absence of a species can be associated with many other habitat variables besides the size of a habitat, how isolated it is, or how species diverse it is. Therefore habitats should demonstrate nested species patterns in other ways. For example, some measurable human disturbances to forests have been used to explain nested patterns of bird presence (Fernandez-Juricic 2002), where the most disturbed habitats have lost several species that are present in less disturbed habitats.

This study explores nestedness and diversity of forest understory plants in Illinois in order to determine their relationship to forest disturbance. Because it has been shown that some species eliminated from clear-cut forests remain absent for centuries after forests have re-grown (Peterken and Game 1984), it is our expectation that some groups of sensitive forest plants (often spring and ant-dispersed wildflowers) will be disproportionally absent from heavily disturbed forests (McLachlan and Bazely 2001). This presence or absence of certain sensitive understory plants would be shown by nested patterns. Based on previous studies, we predict the following will be true in Illinois forests:

1. We expect greater ground layer species diversity in more mature forests (maturity is measured as the Basal Area of the canopy/overstory trees, which decreases with disturbance such as lumbering).

2. The ground layer species in mature forests will be nested subsets of less mature forests. We expect this because mature forests should be more likely to harbor sensitive late-successional species that will be more often absent from younger, more disturbed forests.

3. The ground layer species in forest sites of low floristic conservation value (low Mean C) will be nested subsets of sites of those of high conservation value. We expect this because certain species have been eliminated from sites of low conservation value, which have incurred more disturbance. These species will be present only at high conservation value sites.

4. We expect high conservation value forests to be more mature forests. This would occur if the historic disturbance to trees will also have affected the understory plants present, eliminating sensitive plants.

Methods

Site data were gathered as part of the Illinois Critical Trends Assessment Program (CTAP) (Molano-Flores 2002). We selected data from 106 randomly chosen CTAP sites (51 floodplain and 55 upland), which were sampled over five years (1998-2002) (Figure 2). We measured forest maturity as stand-level basal area m^2/ha (this was used as a surrogate for canopy disturbance to the stand). In Illinois, maximum basal area occurs between 100 and 150 years of stand age

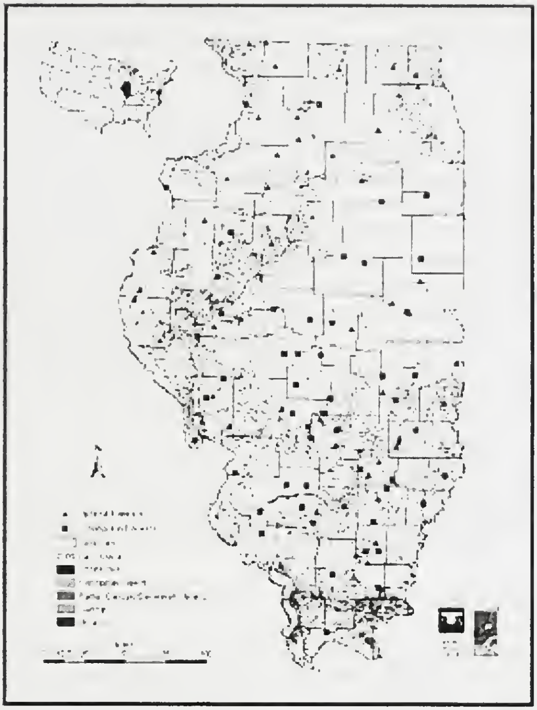

Figure 2. Distribution of 106 randomly selected forest sites used in the analysis. Approximately 14% of the state is in forest cover, compared to 8.5% in 1920, and 38% in 1820, before large-scale European settlement (Bretthauer and Edgington 2002).

(Fralish 1994). Due to the random nature of the study design we sampled no true primary or old-growth forests, because such 150-250 year old relatively undisturbed forests are extremely rare in the state (see data within Iverson *et al.* 1989).

Floral conservation value of each site was assessed using Floristic Quality Assessment (we use "floristic quality" synonymously with "floristic conservation value") (Swink and Wilhelm 1994, Taft *et al.* 1997). Conservative plants exhibit a high affinity for biologically undegraded, remnant habitats. Analysis of nestedness followed Matthews (2004). Relationships among site variables (species richness, stand-level basal area, and mean *C*) were investigated using Pearson correlation coefficients. Patterns of individual species distribution with respect to site species richness, basal area, and mean *C* were investigated using Wilcoxon two-sample rank tests (Mann-Whitney *U* tests). Sites were rank-ordered by decreasing species richness, and a Wilcoxon test was used to determine the degree to which an individual species' presences and absences among sites were ordered along this ranked sequence (Siegel 1956, Schoener and Schoener 1983). The sample statistic produced by the test is approximately normally distributed, and deviation from a random distribution of presences and absences among sites can be determined using the standard normal variate, z (Siegel 1956). Because the calculated value of mean C for a site is dependent upon the C-value of the individual species being tested, a bias is introduced. Highly conservative species increase the mean C of any site at which they occur, and thus are more likely to be found significantly ordered with respect to mean C. To eliminate this bias a species was excluded from the calculations of mean C for all sites before performing the Wilcoxon test for that species. Alpha-levels for this study were set at 0.05, unless otherwise stated. Spearman rank correlation coefficients were used to investigate the relationship between individual species C-values and species nestedness scores (Wilcoxon z-scores) in order to determine whether conservative species were more likely to be non-randomly associated with sites of high species richness, floristic conservation value, or basal area (see Matthews 2004). Analyses were performed using Statistix version 2.2 (Analytical Software 2000).

The natural logarithms of the tail probabilities of Wilcoxon tests for individual species were combined using Fisher's method (Sokal and Rohlf 1995) to obtain measures of nestedness of the overall species assemblages in floodplain and upland forests with respect to species richness, basal area, and mean C (Kadmon 1995, Hecnar and M'Closkey 1997, Matthews 2004).

Results

In both floodplain and upland forests, basal area was significantly negatively correlated with species richness (Figure 3), suggesting a decrease in species richness with forest maturity. Stand-level basal area was not significantly correlated with site floristic conservation value (mean C) in either floodplains ($r = 0.10, p = 0.49$) or uplands ($r = 0.04, p = 0.80$).

Conservative species were more likely than non-conservative species to be positively ordered with respect to site species richness, as indicated by a significant correlation between individual species, meaning that conservative species were more often found in relatively species rich sites (Table 1). Species assemblages in both floodplain forests and upland forests were significantly nested with respect to species richness ($\chi^2 = 669$, df = 230, $p < 0.0001$; and $\chi^2 = 710$, df = 264, $p < 0.0001$, respectively). This indicates that the understory flora of a site tended to be a nested subset of each site more diverse than itself.

Conservative species tended to occur in upland forests of lower basal area (Table 1). Combining the results for all individual species in each forest type revealed that the species assemblages in both floodplain forests and upland forests were significantly negatively nested with respect to basal area ($\chi^2 = 365$, df = 230, $p < 0.0001$; and $\chi^2 = 422$, df = 264, $p < 0.0001$, respectively). Therefore, floras of sites with high basal area tended to be proper subsets of floras of sites with low basal area. Conservative species are likely to co-occur, i.e. occur at sites that have greater conservation value (Figure 4). The overall species assemblage was significantly nested with respect to mean C in upland forests but not in floodplain forests ($\chi^2 = 394$, df = 264, $p < 0.0001$;

and $\chi^2 = 257$, df = 230, $p = 0.11$, respectively), indicating that for upland forests, floras of low conservation value sites are subsets of floras of higher conservation value sites.

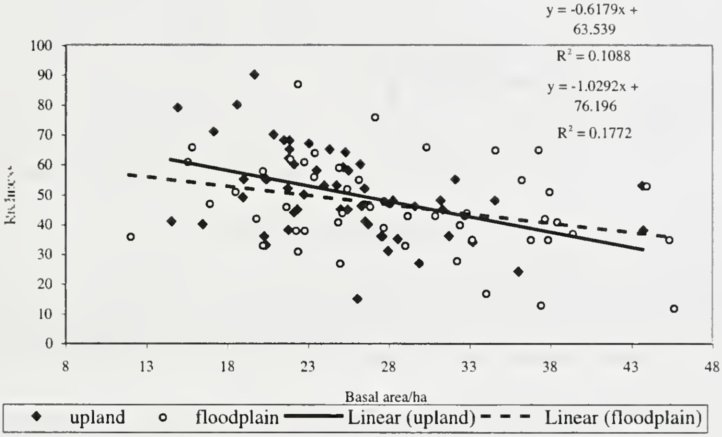

Figure 3. Relationship between understory plant species richness and stand-level tree basal area (m^2/ha) in Illinois floodplain forests (open circles, dashed line, $r = -0.34$, $p < 0.05$) and upland forests (closed circles, solid line, $r = -0.42$, $p < 0.01$).

Table 1. Spearman rank correlations (r_s) among species variables in floodplain forests (above diagonal) and upland forests (below diagonal). Variables include species coefficient of conservatism (C), and species z-scores with respect to basal area (m^2/ha), mean coefficient of conservatism, and species richness (Site basal area, Site conservation value, Z-meanC, and Site species richness)

	Species C	Site Basal Area	Site Conservation Value	Site Species Richness
Species C	...	-0.02	0.33***	0.20*
Site Basal Area	-0.17*	...	0.05	-0.40***
Mean C	0.42***	-0.13	...	0.20*
Site Species Richness	0.21*	-0.49***	0.28***	...

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

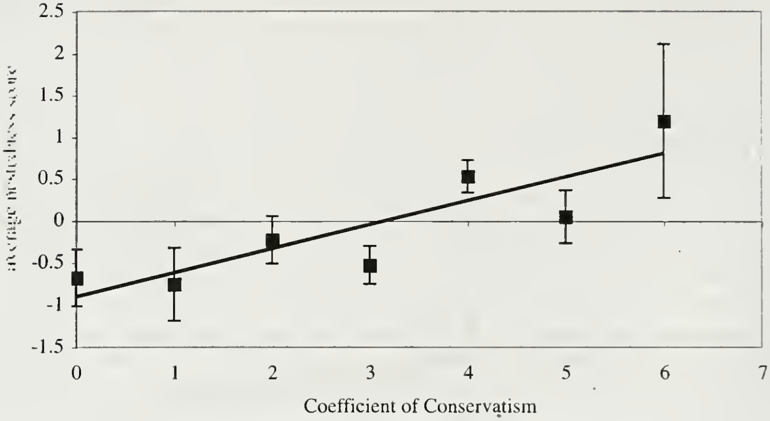

Figure 4. Mean (+/- one S.E.) individual species nestedness score versus species coefficient of conservatism (*C*) for understory plants in floodplain forests [A] and upland forests [B]. Positive values on the y-axis indicate that a species is more likely to occur in sites of high floristic quality, whereas negative values indicate that a species is more likely to occur in sites of low floristic quality. Data points are means for all species within a coefficient of conservatism category.

Discussion

Forest maturity and ground-layer plants

Following European settlement, most forests in eastern and central North America incurred heavy human use and disturbance, and in many cases they were completely cut over for timber and farmed. In both North America and Europe the amount of forested land has begun to increase over the past few decades (Bretthauer and Edgington 2002) (Figure 2). However, observations suggest that the species occupying these re-grown forests are different from those present in old-growth forests that were never cleared and were minimally disturbed. It has been shown that species diversity is greater in old-growth forests compared to mature second-growth forests that have a history of being clear-cut (Duffy and Meier 1992), and sensitive forest interior plants that are specialized to grow in mature forest (late-successional) account for these differences in diversity. These so called "ancient forest specialists" tend to be shade tolerant, slow growing, and have low seed production; they are often eliminated by massive disturbances and re-colonize very slowly (Meier *et al.* 1995). Effects on diversity are most pronounced in forests that have re-grown from cropland, as compared to forests that were clear-cut, or selectively logged forests (Hermy *et al.* 1999). Overall, this suggests that even though forests may be comprised of mature trees and canopy basal-area values matching those of pre-settlement forests, secondary tree re-growth does not insure understory species recovery (e.g. mature forests may re-grow their trees, but they may not have their original wildflowers).

Because older forests have been shown to have greater diversity, we expected that mature forests, without recent catastrophic canopy disturbances, would have a greater chance for re-colonization of their historic plants, causing them to be more diverse. However, our study has shown that species richness declined with increasing stand maturity, suggesting that colonization by ancient forest species is not occurring in Illinois (Figure 3). Our findings are consistent with theoretical models or forest succession showing that plant diversity is initially highest because of the presence of old-field weeds, which then decrease as the forests mature (Roberts and Gilliam 1995). Given enough time, diversity should increase again when forests mature and become old-growth. This increase in diversity in old-growth comes from plants whose seed comes from other forests. Whether increased colonization and diversity is a consequence of the amount of time and therefore number of chances for seed to get to the site, or if as some suggest, they reflect the development of specific "old-growth" habitat that must be present before establishment of ancient forest plants can occur, is unknown (see discussion below).

Our results did not support our expectation that floras of low basal area forests would be nested subsets of floras of mature forests; in fact, we found just the opposite. This suggests that species characteristic of young, open, forests are disappearing as basal area increases, and conversely, it indicates that a set of species preferentially associated with high basal area forests (ancient forest plants) is not present. Thus the plants of open, early successional forests are being shaded out as the forest matures, but they are not being replaced by late-successional plants.

The lack of old-growth plants in our mature second-growth forests may be explained by two possible mechanisms. First, these forests may be less suitable for establishment of these species. Environmental conditions present in second-growth forests may limit the suitability of habit for centuries after forests re-grow from cropland (Honnay *et al.* 2002). Complex factors associated with uneven aged tree structure, such as "gap-phase succession", tip-up mounds, micro-climatic, or other micro-site conditions may need to be in place in order for herbaceous recovery to occur. In addition, if the historic tree composition has been altered this can directly influence chemical and physical properties of litter and soil (shade, pH, nutrient levels, fungal and microbial communities), which may also play a role in altering understory plant communities (Whitney and Foster 1988).

Alternatively, dispersal limitation could be preventing colonization. Studies of second-growth forest directly adjacent to patches of old-growth seed sources show that re-colonization is amazingly slow for ancient forest herbs (Matlack 1994). In this context, re-colonization of re-growing forests by ancient forest species should not be expected to occur across the Illinois forest landscape, which contains almost no old-growth refugia, and is highly fragmented.

It has been suggested that colonization and recovery of herbs should be faster in floodplain versus upland forests (Honnay et al 2001). This is because floodplain forests are more contiguous and receive frequent seed deposition with flooding. However, we found no significant difference between the negative relationship between floodplain and upland forests (data not shown).

Biological conservation value of forest floras

As predicted, understory plant assemblages were nested with respect to site floral conservation value (though not significantly in floodplain forests). This means that certain species present on sites of high floral value are predictably absent from those of lower value. Highly conservative plants are sensitive to human disturbances species, and we can conclude that they have been eliminated from, or are unable to colonize, degraded sites. The pattern of non-conservative species that are predictably found on sites of low conservation value suggests the existence of a set of species that preferentially occur in degraded woodlands (e.g. in uplands these were *Taraxacum* [dandelion], *Smilax* [catbrier] and in floodplains they were *Urtica* [stinging nettle], *Ranunculus abortivus* [crowfoot]) (Figure 4). Overall these results suggest that mean *C* is a reliable metric of floristic conservation value in forests because conservative species tend to occur together at undegraded sites.

The prediction that site basal area and floristic conservation value would be positively correlated was not supported. This contradicts a previous study that showed a positive correlation between mean *C* and forest maturity in Ontario (Francis *et al.* 2000). In fact, in upland forests, a group of conservative plants was preferentially found in less mature forests (see C and Site basal; Table 1). Illinois contains many conservative species of open woodlands; these are often not tolerant of shade. Generally, this shows that the biological conservation value of many upland forests would benefit from being maintained in their historic condition as open-woodlands (i.e. with prescribed fire, selective logging of maple).

Whatever forest degradations are influencing the conservation value of Illinois forests, they are not the same as those that are influencing basal area, because basal area and conservation value measures showed disparate results in this study. We propose that intensity of historic and current livestock grazing is the most critical factor affecting conservation value of forests, as upwards of 90% of midwestern U.S. forests have undergone grazing of various intensities, and many of the forests sampled in this study were either actively grazed or had been grazed recently (Mabry 2002). Different grazing intensity among sites should lead to a nested distribution of ground layer species with respect to site mean *C* if conservative species are more sensitive to grazing than non-conservative. Previous studies have shown that grazed forests have similar species compositions to those of low conservation value forests in this study. Additionally, no differences have been found between paired grazed and ungrazed forests in mean DBH of overstory trees or tree canopy coverage, suggesting that the overstorey is minimally affected by grazing compared with the ground layer (Mabry 2002). Additionally, this suggests that unlike mean *C*, the basal area of a forest stand is not an accurate indicator of the conservation value of forest floras, nor can it be used as a stand-alone measure of forest biological quality (Norris and Farrar 2001).

Restoration and conservation of diverse forests

In future studies, specific attention should be paid to those factors that affect sensitive forest interior plants, as their absence is limiting the restoration of diverse, biologically complete forests. Indeed, the presence of certain plant groups, such as spring wildflowers and ant-dispersed wildflowers, is increasingly seen as the essential indicator of complete forest recovery. Along with this study, a large body of recent work suggests that mature forests with depauperate ground layers in human dominated landscapes will not recover their full array of wildflowers within observable lifetimes. Even for forests directly next to pristine old growth woods, re-colonization will likely be measured over centuries. On the other hand, since populations of these ancient forest interior species can be so long-lived in stable forests, conserving even small areas of high conservation value and diversity will be important for the future of forest conservation in Illinois.

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Maple Takeover: A Different Approach to Identifying an Ongoing Problem

Matt Buffington

Introduction

The problem of maple takeover in Illinois forests was highlighted in the original CTAP report in 1994 (IDENR 1994) and tracking its extent is one of the intended uses of the ForestWatch data. Under maple takeover, the abundance of sugar maples increases in forests where they were once just a small component. For the most part, the focus of this condition has been on upland oak-hickory forests. This is because: 1) sugar maple is an upland species capable of expanding its range into oak-hickory communities; and 2) oak-hickory forests are the dominant forest type all across the state, both presently and historically (Bretthauser and Edgington 2002; Ebinger 1997).

Maple takeover occurs when sugar maples are able to survive and grow in forests where they once were controlled by various disturbance factors, primarily fire in oak-hickory uplands (IDENR 1994, Fabry and Patterson 2000, Feist et al. 2004 [and citations within this article]). Fires that predominated in the prairie ecosystem typically found their way into forests and the degree to which a forest burned depended on its elevation, topography, aspect, and orientation to major waterways. Prevailing winds generally drove fires eastward across the landscape, and low, wet areas, north/east facing slopes and the east side of rivers and large streams burned less frequently due to their relative protection from fires. Unprotected areas that burned more frequently, such as south- and west-facing slopes, also tend to receive more light and are generally drier than protected areas. Oaks and hickories often dominate these drier forests while sugar maple, elm, and basswood are typically found in protected areas on north and east facing slopes (White and Madany 1978). On level areas, fires would burn through forests as long as conditions were right. In many uplands fire occurred with enough frequency or intensity to retard the growth of sugar maples, which do not survive fires well, and the canopy was kept open which provided enough light for oak and hickory seedlings to grow and mature quick enough to survive the occasional fires. Thus, the presence or absence of fire historically controlled the distribution of oak-hickory forests as well as communities dominated by maples.

In the modern Midwest landscape, fire is no longer a prominent factor and the forest communities are changing (Ebinger 1997). Sugar maple is not the only culprit in this change in forest composition. In some woodlands, white ash and slippery elm appear to be increasing as much or more than sugar maple (Fralish 1997). These three species are referred to as "mesic" upland tree species, mesic referring to a preference for soils of moderate moisture. However, this discussion will focus on sugar maple given its recognized ability to expand its range under an extended period of fire suppression.

In most cases, forest ecologists have relied on their own judgment to determine whether takeover was occurring in a forest, based on the relative abundance of sugar maple, oaks, and hickories in the different diameter classes. This is a case where art and science mix. There is no clear definition of when a forest is experiencing maple takeover, although many people recognize it when they see it. In forests that have a closed canopy and few (if any) oak and hickory seedlings and saplings but a good number of small sugar maple, maple takeover may be occurring. If the condition could be defined quantitatively one could objectively diagnose if a forest is undergoing maple takeover by relying on a set of data, such as ForestWatch monitoring data.

The following charts depict the diameter size class distribution of a hypothetical tree species under different situations. Figure 1 is an example of the size distribution of an invading species. All of the individuals are in the smaller size classes, indicating that most of the trees are relatively young and there are no dominant individuals present. Although in many cases a good number of these smaller individuals will die before becoming large trees, the large number of individuals in the smaller size classes means there are trees ready to fill any gaps that may occur.

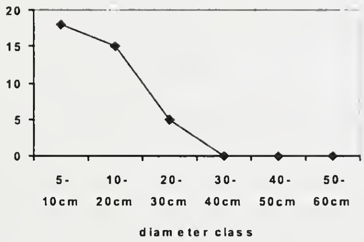

Figure 1

Figure 2 shows the diameter distribution of a tree species that is showing good regeneration. There are some large individuals present and more importantly, there are also a fair number of smaller individuals.

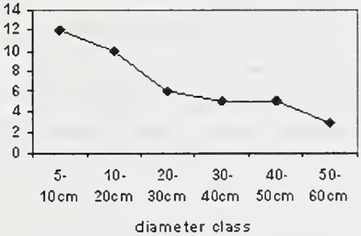

Figure 2

If an established species is not regenerating well the diameter class distribution may look like Figure 3. Most of the individuals are in the larger size classes with little or no recruitment in the smaller classes. Something like this typically happens with oaks and hickories in forests undergoing maple takeover.

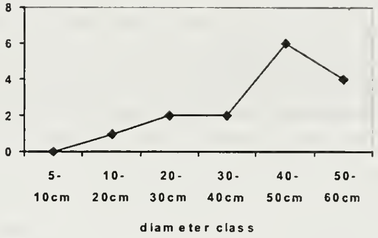

Figure 3

Maple takeover is one side of a two-sided problem. Maple takeover deals with the increasing abundance of sugar maple in the smaller size classes. If sugar maples greatly outnumber oaks and hickories in the smaller classes, it is in a position to fill any canopy gaps that occur, thereby replacing the other species. The other side of the problem is the lack of oaks and hickories in the smaller classes. This is not primarily the result of an increased amount of small maples. Rather, it deals with overall decreasing light levels on the forest floor as a result of an ever-closing canopy and sub-canopy, which inevitably occurs in forests as they age without disturbance. Much like takeover, decreasing light levels is the result of decreasing fire frequency and other changes to the landscape. Most oak and hickory species are shade intolerant, though a few hickories may be classified as moderately intolerant in Illinois. When the understory is heavily shaded, oak and hickory seedlings and saplings cannot survive (IDENR 1994). So if a canopy gap is created, there may not be any oaks or hickories in a position to fill it. Quite often, sugar maples are on hand to exploit the opportunity.

In summary then, sites suffering from maple takeover are upland forest sites dominated by oak-hickory in the large tree classes, with maple dominant in the small classes and mostly missing from the largest classes. Based on these somewhat subjective criteria, 32 ForestWatch monitoring sites were judged to show some level of maple takeover or about one in six of the upland sites. A more quantitative process for diagnosing maple takeover based on the monitoring data would be beneficial. With this in mind, the following discussion details the various steps taken to try to quantify how these 32 sites differ from all other upland sites.

Methods

Two criteria were used to select ForestWatch sites to include in the analysis. First, only uplands were considered; 194 of the 240 monitoring records are from upland sites. Second, only sites that had oak and/or hickory and sugar maple present were analyzed. If sugar maple is not at a site, it cannot be experiencing maple takeover. Ninety-three of the 194 upland site records met both criteria. Of these, 67 sites were oak-hickory forests, 25 were mesic forests (forests that had sugar maple, ash, basswood, or tulip naturally dominant), and another site was a sassafras-cherry-persimmon community. The data from the 93 sites were examined to determine the commonality among the 32 takeover sites.

A Pearson two-tailed correlation analysis was used to identify potential variables from the ForestWatch data that may help to quantify maple takeover. The initial set of variables included the following:

- Ratio of sugar maple abundance to total sugar maple + oak + hickory
- Ratio of sugar maple to total sugar maple + oak + hickory by size class
- Presence of sugar maple by size class
- Presence of oak + hickory by size class
- Sugar maple basal area
- Oak + hickory basal area
- Sugar maple importance value
- Oak + hickory importance value
- Ratio of sugar maple importance value to oak + hickory importance value
- Range of 17-65% in ratio of sugar maple to maple + oak + hickory abundance
- Range of 17-67% in ratio of sugar maple to maple + oak + hickory abundance[1]

The latter two were included in addition to those variables in the ForestWatch database, because an examination of the data showed that most takeover sites had a ratio of sugar maple abundance to total abundance of maple, oak, and hickory greater than 17% but less then about 65-68%. Two different ranges were used because of the uncertainty about where the borderline lay.

Variables with the highest correlations to maple takeover were then included in a stepwise regression to select the best quantitative model. The dependent variable was the presence of maple takeover coded as zero or one. The matrix of correlation coefficients was also used to examine relationships among independent variables to ensure that no two variables highly correlated with one another were retained in the model.

Results

The correlation analysis showed that maple takeover was significantly positively correlated to the presence and ratio of maples in the smallest size class (Class A 5-10 cm), oak basal area, oak importance value, presence of A class maples and E class oaks, and the ratio of maple to total abundance in the range of 17-65% and 17-67%. Maple takeover was also negatively correlated to the ratio of maples in the largest size classes (Classes E, F, and G, that is greater than 40 cm).

The variable that was most highly correlated with maple takeover was the ratio of maple to total abundance in the range of 17-67% (R=.781, p<.0001).

The stepwise regression resulted in several possible models, depending on exactly which independent variables were examined. In every case, one of the abundance ratios (either 17-65% or 17-67%) was selected, along with oak importance value or basal area, proportion of small maples, and absence of large maples. Below (Table 1) is one model that was successful in predicting all 32 of the sites that were determined to have maple takeover. However, it also results in seven false positives, predicting that seven sites have maple takeover that were not diagnosed with it. One of the alternative models has one less false positive but two false negatives, missing a couple of the sites that were diagnosed with maple takeover. Overall the model is about 80% successful in identifying sites with maple takeover.

Table 1. Model to predict maple takeover: R2 = .702, F = 51.9, p<.0001, df = 92

Variable	Coefficient	t statistic	Significance
	-.432	-4.433	<.0001
Range of 17-67% in ratio of sugar maple to total abundance	.593	9.616	<.0001
Oak importance value	.004	4.424	<.0001
Ratio of sugar maple in A class	.388	4.278	<.0001
Presence of F class sugar maple	-.175	-1.922	.05

Discussion

The regression model reflects in a quantitative way the subjective criteria used to diagnose sites with maple takeover. The model shows that when the total relative abundance of sugar maples is between the extremes of 17-67%, there is likely a problem. This makes intuitive sense. If there is only a small amount of sugar maple, the site is not likely to be undergoing takeover. Alternatively, if there are many sugar maples present, especially when compared to oaks and hickories, then the site probably is not an oak-hickory forest but rather a forest community in which sugar maple is naturally dominant.

Also, it is logical that only sites with a relatively high importance value for oak and hickory would be candidates for maple takeover, because maple takeover occurs in oak-hickory forests where oak and hickory are dominant features in the forest. All of the maple takeover sites had a combined oak-hickory importance value of at least 45; most have an importance value exceeding 80.

The relative abundance of sugar maple to maple/oak/hickory in the smallest diameter class (5-10 cm) also makes sense as the third variable in the model. The analysis shows that the higher this ratio, the more likely a site was to be suffering from maple takeover. All of the sites designated as takeover sites had at least 33% sugar maples and most had more than 50% sugar maples. This is consistent with the characteristic of maple takeover that sugar maples out compete oak and hickory seedlings and saplings in a site where fire is absent (see citation in Feist et al. 2004). However, there were some sites with a high proportion of maples that were not undergoing takeover; generally these were mesic forests naturally dominated by sugar maple.

Finally, the model shows that sites that had a relatively large number of sugar maples in the larger size classes, in particular Class F (50.1-60 cm), were generally not classified as undergoing takeover. The presence of large maples indicates the sites probably represent mesic forests with well established sugar maples, such as a sugar maple-basswood type forest community as opposed to oak-hickory (Bretthauer and Edgington 2002; Ebinger 1997). However, it is possible that sites that contain only a few larger maples may be in advanced stages of takeover. Only one of the sites designated as a maple takeover site contained F class sugar maples.

The model equation can be used to predict whether sites are suffering from maple takeover. Any sites scoring 0.50 or above would be considered as takeover candidates. Since sites that score from 0.50 to 0.67 are sometimes false positives, these sites should be examined more closely to determine if they are in fact sites with maple takeover. Any sites scoring above 0.67 can safely be assumed to have maple takeover.

The model can also be used in a less formal way to develop a set of rules of thumb. These rules of thumb are nearly as successful in predicting maple takeover, only missing one takeover site and resulting in one less false positive. In general, a site must:

1. Be an upland forest site
2. Contain both oak and/or hickory and sugar maple
3. Show an overall abundance of sugar maple to maple, oak, and hickory of one-sixth to two-thirds
4. Possess an oak-hickory importance value of 45 or greater
5. Display a relative abundance of sugar maples in A class (5-10 cm) of 33% or greater
6. Have no F class maples present (50.1-60 cm)

Conclusion

It is clear there is no simple, objective method to best identify a maple takeover site. However, a quantitative model offers good results in screening sites for maple takeover. The model includes as key variables a certain range in values for relative sugar maple abundance compared to maple/oak/hickory abundance, along with oak-hickory importance value, the proportion of small maples, and absence of large maples. None by itself categorizes maple takeover as well as the combination. The model can predict with about 80% accuracy sites that are suffering from maple takeover. It results in a few false positives. A set of rules of thumb based on this model, does as well in gauging whether takeover is a problem at a particular site. The CTAP program should further test the efficacy of the model and perhaps refine it using future sets of ForestWatch or CTAP data.

ForestWatch Data for Two Ecosystem Partnerships, Fall 2003

Matt Buffington

Introduction

In 2003, two Ecosystem Partnerships (EP) began incorporating ForestWatch and PrairieWatch monitoring into their partnership planning. The LaMoine River EP and Vermilion Watershed Task Force EP are trying to use the data to establish a baseline, get a sense of what is going on in their partnerships, and possibly determine what types of management is needed. There are several potential outcomes from this effort but the overall intention is to continue this monitoring to determine if there are changes within the partnership, positive or negative, as a result of management.

Using the baseline set of data, the partnership can compare its sites to other regional and statewide sites, interpret why the data are the way they are and glean some clues as to what some of the problems are within the partnership. However, these sites do not necessarily typify the partnership because of a limited number of sites monitored. Data interpretation and recommendations made in this report are based on the sites that were sampled.

Results and Discussion

The LaMoine River Ecosystem Partnership

There were five LaMoine River EP sites monitored in fall 2003. Two sites were in Brown County, one in McDonough, and another in Schuyler County. These sites fall in the central third of the state (Fig. 1) and comparisons can be made among the EP sites, other sites in the same third of the state, and all the sites across the state.

Figure 1. Three botanical regions of Illinois

Table 1 depicts some of the basic averages for trees from the partnership sites in comparison to the region and state. The five sites monitored by the LaMoine River EP had fairly low tree abundance but higher than average basal area. The average tree diameter corresponded with basal area as they are highly correlated with each other. Generally, a more mature forest contains larger but fewer trees (Fralish 1991, McCarthy et al. 1987), thus the inverse relationship between abundance and basal area (Ginrich 1967). This suggests the LaMoine River EP sites were on average older than sites elsewhere in the region. Tree taxa richness was very similar among the three groups. Tree richness typically increases with total abundance and decreases with increasing basal area and diameter so the situation with the LaMoine River EP partnership sites is somewhat unusual. Because younger forests can have a mix of early, mid, and late successional tree species, they may contain many taxa and many small trees, but their overall basal area may be low (Oliver 1981).

Table 1. Mean tree data for the LaMoine River EP compared to other regions, 2003.

	# of Sites	Tree Taxa Richness	Tree Abundance	Tree Basal Area (m2/ha)	Tree Diameter (cm)
LaMoine EP	5	11.6	88.0	26.6	23.4
Central Third	13	11.5	128.9	25.6	20.1
Statewide	52	11.3	111.0	24.5	21.3

As illustrated in Table 2, the sites monitored by the LaMoine River EP appear to have a problem with multiflora rose and Missouri gooseberry. These are two of the three invasive shrubs that are problems for the entire region. In general, autumn olive is not much of a problem in forests, as it tends to decrease as canopy shading increases; thus it is a good indicator of past disturbance. The total amount of invasive shrubs falls between the rest of the sites in the region and the statewide average. The lack of honeysuckle is good, although there are sure to be forests within the partnership where honeysuckle is a problem. The partnership should consider invasive shrub control to be a part of their restoration efforts. The percentage of invasive shrubs for the four partnership sites was quite high, despite a decent amount of "other" shrubs that are generally considered non-invasive. In all likelihood, all of the invasive shrubs are found in some forests in the region if not the partnership. However, multiflora rose and gooseberry, and likely honeysuckle, are the main problems.

The Vermilion River Watershed Task Force Ecosystem Partnership

Despite the majority of the Partnership being in the central third of the state, four of the five sites were in the northern third of the state (four were in LaSalle County and one in Livingston County, Figure 1). Because of this, the partnership was compared to both the northern and central regions. As the data show, the EP shares characteristics of both regions, supporting the fact that the sites are on the border of the north and central regions.

Table 2. Amount of shrubs in the LaMoine River EP, 2003. Values for various shrub taxa and categories are in number of shrub stems per hectare.

	# of Sites	# of Invasive Species	Invasive Shrub Density (stems/ha)	Other Shrubs	Shrub Honey-suckles	Buck-thorn	European High-bush Cranberry	Autumn Olive	Multiflora Rose	Missouri Gooseberry	Japanese Honey-suckle	Other Vines	Invasive to Total Shrub Ratio
LaMoine EP	5	1.6	2860.0	1006.7	0	0	0	0	2353.3	506.7	0	153.3	80.7
Central Third	13	1.5	1528.2	666.7	430.8	0	0	107.7	697.4	292.3	0	205.1	61.3
Statewide	52	1.6	2739.3	558.3	955.8	557.1	1.5	28.2	681.4	515.4	29.5	298.7	65.3

The number of tree taxa, or richness, was very similar among the four geographic regions (Table 3). However, there were some larger differences with the number of trees recorded and their basal area. The five partnership sites had a large number of trees and lower basal area and corresponding lower average tree diameter. This relationship of numerous trees and lower diameter is very strong when looking at all of the ForestWatch data (i.e., sites with more trees tend to have smaller trees). It is common for younger forests to show this condition, where there are many younger, smaller trees and few, if any, large, dominant trees (Oliver 1981). Interestingly, abundance is not correlated to basal area. So total basal area is independent of how many total trees are present. Among the five partnership sites, three of them did not have a tree with a diameter over 40cm. This indicates these are relatively young forests.

Table 3. Mean tree data for the Vermilion Watershed Task Force EP compared to other regions, 2003.

	# of Sites	Tree Taxa Richness	Tree Abundance	Tree Basal Area (m2/ha)	Tree Diameter (cm)
Vermilion WTF EP	5	11.4	164.8	21.5	16.8
Northern Third	18	11.1	97.8	24.2	22.2
Central Third	17	11.5	114.2	26.0	21.3
Statewide	52	11.3	111.0	24.5	21.3

As with many parts of the state, the sites monitored in Vermilion Watershed Task Force Partnership have a large amount of invasive shrubs. In fact, the EP averages more shrub honeysuckle than any other region. Overall, the sites share characteristics of the northern and central thirds. Many of the northern counties have extensive problems with buckthorn and gooseberry while the central third has the biggest problem with multiflora rose. The partnership sites do not have problems with buckthorn but there are extensive amounts of honeysuckle, multiflora rose, and gooseberry.

Considering 90% of the recorded shrubs were invasive, control of these shrubs should be a priority. The amount of honeysuckle, multiflora rose, and Missouri gooseberry at these sites may be associated with the age of the sites – younger sites often have heavy undergrowth because of increased light levels. The average tree diameter (Table 3) suggested the sites were relatively young as does the below average level of shading on the forest floor (Table 5).

Table 4. Amount of shrubs in the Vermilion WTF EP, 2003. Values for various shrub taxa and taxa categories are in number of shrub stems per hectare.

	# of Sites	# of Invasive Species	Invasive Shrub Density (stems/ha)	Other Shrubs	Shrub Honeysuckles	Buckthorn	European High-bush Cranberry	Autumn Olive	Multiflora Rose	Missouri Gooseberry	Japanese Honeysuckle	Other Vines	Invasive to Total Shrub Ratio
Vermilion WTF EP	5	2.6	7329.1	626.7	4426.7	0	2.4	0	2353.3	546.7	0	513.3	90.3
Northern Third	18	2.4	4161.1	294.4	1248.2	1609.3	3.7	3.7	259.3	1037.0	5.6	429.6	81.9
Central Third	17	1.4	1458.5	768.6	298.0	0	0	82.4	1029.4	317.7	0	103.9	66.0
Statewide	52	1.6	2739.3	558.3	955.8	557.1	1.5	28.2	681.4	515.4	29.5	298.7	65.3

Table 5. Shading levels for the Vermilion Watershed Task Force EP and other areas.

	Average shading level
Vermilion WTF EP	73.9%
Northern Third	82.2%
Central Third	84.4%
Statewide	83.7%

Conclusion

The LaMoine River EP and Vermilion Watershed Task Force EP have taken the initial steps to conduct monitoring at several forests within their EP. Analysis of the first year's data has shown that the LaMoine River EP has older forests than the Vermilion Watershed Task Force EP. In addition, the data show that although these forests vary in age they all have invasive species problems. Efforts should be targeted to develop management activities that will decrease the abundance of these species. Once management practices (e.g., removal of shrubs, herbicide application) are put into place, additional monitoring should be conducted to determine the benefits of these practices (e.g., increase in species diversity).

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Regional, Landscape, and Patch Size Differences: Do They Matter for Forest Bird Species Across Illinois?

Brenda Molano-Flores, Steve Bailey, and Rhetta Jack

Introduction

Many bird monitoring programs provide information about the status of birds at a continental level, such as the U.S. Geological Survey's Breeding Bird Survey, or at a statewide level, such as the Critical Trends Assessment Program (CTAP) in Illinois. The combination of these data sets allows the determination of general patterns of bird species distribution and diversity. However, the scale of these programs can sometimes mask local patterns of species distribution and diversity. In Illinois, understanding local regional patterns of bird species distribution and diversity is important because it can aid land managers in the decision-making process for the conservation and management of forests in their particular region. In this report we present data on several geographic levels (regions [north, central, and south], Natural Divisions, EPA basins, and Ecosystem Partnerships) for species richness, habitat dependent species, and area sensitive species. It is our goal that state, federal, and local groups can utilize our findings in their future conservation and management efforts and strategies.

In addition, we want to establish if the forest landscape matrix in proximity to our CTAP forest sites can influence species richness, and habitat dependent (i.e., species only found in a particular habitat) and area sensitive species (i.e., species that show various levels of intolerance to habitat fragmentation). Rodewald (2003) has pointed out that the landscape matrix can alter the movement of species, provide alternative habitat to invasive species, and determine extent of isolation for species. Some evidence exists that the surrounding landscape can affect the presence or absence of bird species. In a Pennsylvania study (Rodewald and Yahner 2001a, b) mature forest within a landscape matrix disturbed by small amounts of pasture, cropland, and fallow fields had fewer forest habitat dependent species.

Finally, in this report we also address the question of the impact of Illinois forest patch size on species richness, and habitat dependent and area sensitive species. Forests across Illinois have changed considerably. In the early 1800's forests covered 40.8% (14,648,491 acres; IDNR 2003) of Illinois. Now only about 14% (4.9 million acres; IDNR 1996) remains in forest. Most forest tracts in Illinois are small, fragmented, and degraded. Fragmentation can have negative impacts on the organisms using these forests. In the case of birds, several studies have shown that the degree of fragmentation (i.e., small forest patches) can lead to lower species richness for habitat dependent and area sensitive species (Ambuel and Temple 1983, Hayden *et al.* 1985, Blake and Karr 1984, 1987, Freemark and Collins 1992, Herkert *et al.* 1993).

Methods

A total of 140 sites were visited from 1997-2001. Following the CTAP protocols (Niven *et al.* 2002) ten-minute point counts were carried out at each site. Depending upon the size of the forest patch between one to 15 point counts were conducted per site 150 meters apart. All bird species heard or seen were listed. Certain bird species were classified as habitat dependent and/or area sensitive species. For the purpose of data analyses both highly and moderately area sensitive species were combined.

Species richness, and habitat dependent and area sensitive species were analyzed by Regions (northern, central, and southern), Natural Divisions, EPA basins, and Ecosystem Partnerships using One-way ANOVA or a Kruskal-Wallis One-way ANOVA when the data were not normal followed by a Tukey or Dunn's test, respectively, for multiple comparisons. In addition, if the sample size for any region was one, it was not used for data analysis, nor were forest sites classified as Not Assigned Ecosystem Partnerships (N = 30, across the state). Means and standard errors are reported. Below we give an explanation for each regional comparison:

1. *Regions:* The state was divided into thirds (north, central, and south) because of north to south climatological and geological differences.

2. *Natural Divisions:* Fourteen Natural Divisions are recognized in Illinois; they identify regions of common ecological composition and setting (i.e., physiography and natural vegetation) (Schwegman *et al.*, 1973).

3. *EPA basins*: The Illinois Environmental Protection Agency has 21 basins or land around rivers or lakes from which water and streams run down into it. These basins are within seven major watersheds (i.e., Targeted Watershed Approach) that have been prioritized by EPA - Bureau of Water (http://www.epa.state.il.us/water/targeted-watershed/).

4. *Ecosystem Partnerships:* IDNR Ecosystems Division, Conservation 2000 Program sponsors Ecosystem Partnerships, which are coalitions of local stakeholders – private landowners, businesses, scientists, environmental organizations, recreational enthusiasts, and policy makers. They are united by a common interest in the natural resources of their area's watershed. Currently, there are 40 Ecosystem Partnerships covering approximately 80% of the state's land area (http://dnr.state.il.us/orep/c2000/ecosystem/partnerships/).

To address the question of the potential role that the forest landscape matrix and patch size may have on species richness and habitat dependent and area sensitive species, multiple regression analyses were used. To determine the forest landscape matrix we used county information from the 1996 Illinois Land Cover Map (IDNR 1996) for acres of forested land (i.e., forest, woodland, and conifers). For acres of timberland and Illinois Natural Areas Inventory (INAI) forested natural areas we used county information from Bretthauer and Edgington (2002). For the purposes of this paper, we define timberland as that given by Bretthauer and Edgington (2002) which is "forest land that is producing, or is capable of producing, more than 20 cubic feet per acre per year of industrial wood crops under natural conditions, that is not withdrawn from timber utilization, and that is not associated with urban or rural development. Currently inaccessible and inoperable areas are not included". This should not be confused with "total forested lands" which are forested regions in Illinois calculated from satellite imagery photography. Patch size (acres) was calculated based on the number of bird points that were conducted at each study site. Using the estimated hearing distance of 100 meters in forest sites, if a site had a single point area was calculated simply by multiplying πr^2, where r is the 100 meters. For forest sites with multiple points areas were determined by drawing boundaries around each point using the software Pathfinder Office ver. 2.9 (1999-2002), and omitting any overlap. Finally, to normalize the data, acres of forested land and patch size were log-transformed and acres of timberland and INAI forested natural areas were log (N+1)-transformed.

Results

Regions (north, central, and south): No significant differences were found between any of the regions for species richness (ANOVA; F = 1.65, df = 2, P =0.196, Figure 1). However, significant differences were found between the southern region of Illinois and the northern and central regions for habitat dependent (ANOVA; F = 6.362, df =2, P = 0.002; Figure 1) and area sensitive species (KW: H =19.452, df = 2, P <0.001), but not between the northern and central regions (Figure 1). The southern portion of Illinois had more habitat dependent and area sensitive species than the northern and central region of Illinois.

Natural Divisions: In the 12 Natural Divisions used for analysis (Figure 2), significant differences were found for species richness (ANOVA: F= 2.882, df = 11, P = 0.002). The Lower Mississippi River Bottomlands division had the greatest species richness with a mean of 37 followed by the Ozark and Shawnee Hills divisions (Figure 2). The Rock River Hill Country division had the lowest species richness with a mean of 21. In the case of habitat dependent species significant differences were found among Natural Divisions (ANOVA: F= 5.275, df = 11, P = <0.001). The Ozarks division had more species (mean = 25) compared to the other Natural Divisions (Figure 2). In the case of area sensitive species, significant differences were also found among Natural Divisions (ANOVA: F= 9.530, df = 11, P = <0.001, Figure 1). The two Natural Divisions with the greatest area sensitive species were the Ozarks (mean = 17) and Shawnee Hills (mean = 16) divisions (Figure 2).

EPA basins: Significant differences were found for species richness among the studied EPA basins (ANOVA: F= 2.167, df = 13, P = 0.015). The Miss. South basin (mean = 38) had the greatest species richness compared to the Kankakee, Sangamon, and Rock basins (Table 1). In addition, significant differences were found among EPA basins for habitat dependent species (ANOVA: F= 3.342, df = 13, P = <0.001). More habitat dependent species were found within the Miss. South basin (mean = 25.5) than the Kankakee, Sangamon, Fox, and Rock basins (Table 1). Also, differences were found in the number of area sensitive species (ANOVA: F= 5.883, df = 13, P = <0.001) across the basins. The Miss. South basin had a greatest number of area sensitive species than any other basin (mean = 17). However no significant differences were found between the Miss. South basin and Miss. North, Miss. South Central, and Ohio basins (Table 1).

Ecosystem Partnerships: No significant differences were found between any two Ecosystem Partnerships for species richness (KW: H = 29.913, df = 21, P = 0.094 (Table 2). However, it

Figures 1: Mean (± standard error) species richness, number of habitat dependent and area sensitive species for Northern (n=29), Central (n=67) and Southern (n=44) Regions.

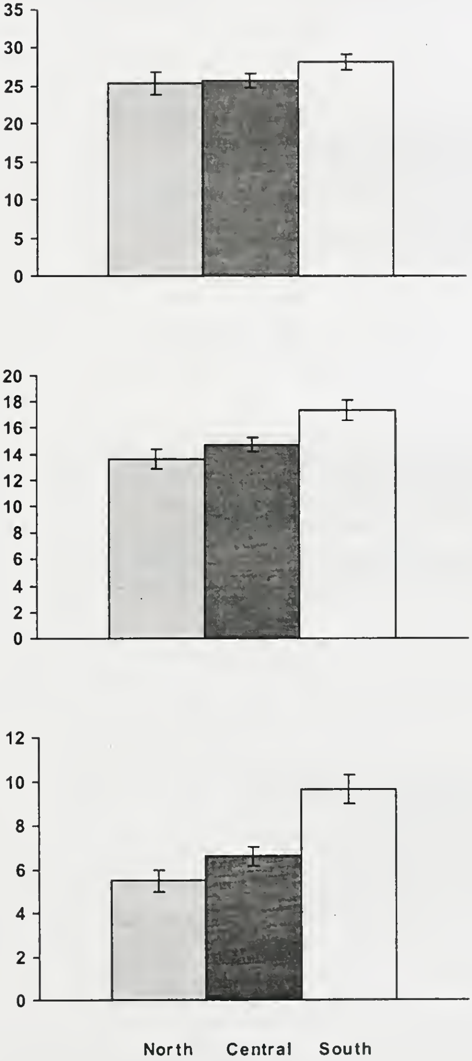
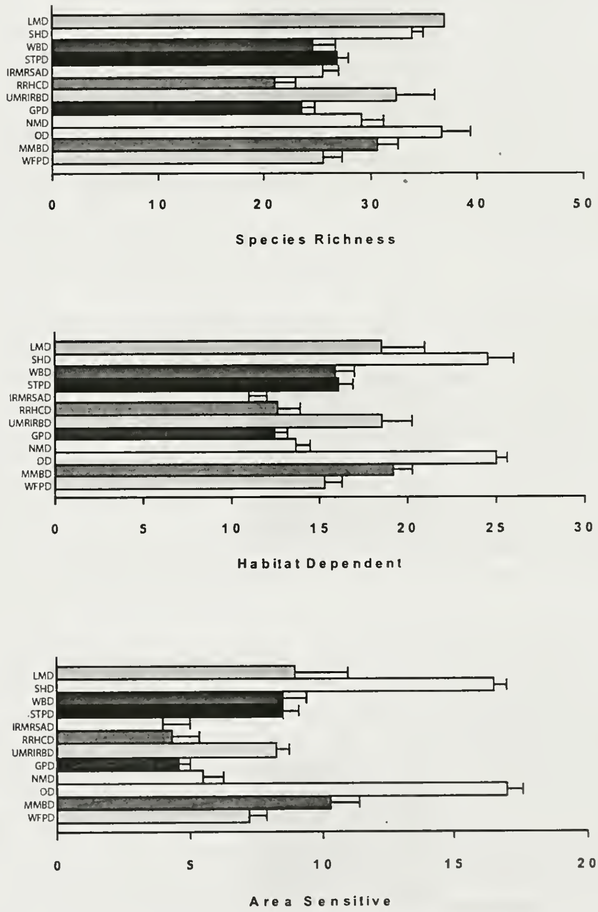


Figure 2: Mean (± standard error) species richness, number of habitat dependent and area sensitive species for Natural Divisions. WFPD = Western Forest-Prairie (n=22); MMBD = Middle Mississippi Border (n=6); OD = Ozarks (n=3); NMD = Northeastern Morainal (n=8); GPD = Grand Prairie (n=35); UMRIRBD = Upper Mississippi River and Illinois River Bottomlands (n=4); RRHCD = Rock River Hill Country (n=8); IRMRSAD = Illinois River and Mississippi River Sand Areas (n=2); STPD = Southern Till Plain (n=36); WBD = Wabash Border (n=10); SHD = Shawnee Hills (n=2); and LMD = Lower Mississippi (n=2).



*Not included in the figure: Wisconsin Driftless division (n=1): species richness=38, habitat dependent species=22, area sensitive species=12; Illinois Coastal Plain division (n=1): species richness=42, habitat dependent species=27, and area sensitive species =17.

Table 1. Mean (± standard error) species richness (SpR), number of habitat dependent (HD) and area sensitive species (AS) for Illinois EPA Basins.

EPA Basins	N	MeanSpR	SE	MeanHD	SE	MeanAD	SE
Miss North Central	9	25.2	2.5	15.8	1.3	7.0	0.9
Illinois	29	26.7	1.5	14.8	0.9	6.9	0.6
Miss South	4	38.0	2.4	25.5	0.6	17.0	0.4
Rock	13	24.2	1.9	13.4	1.0	4.5	0.6
Miss North	4	29.8	4.4	18.0	2.3	9.5	1.4
Sangamon	14	23.6	1.8	12.7	1.0	4.6	0.7
Wabash	27	25.5	1.1	15.4	1.0	7.4	0.7
Kaskaskia	15	26.6	1.8	15.9	1.2	8.7	1.0
Des Plaines/Lake Michigan	2	32.5	3.5	14.0	2.0	6.5	1.5
Big Muddy	6	27.8	3.1	16.8	1.5	9.2	1.6
Ohio	7	27.7	2.9	17.7	2.3	11.0	1.8
Kankakee	2	16.0	1.0	8.5	2.5	3.5	2.5
Miss South Central	3	37.0	0.0	20.3	2.3	10.0	1.5
Fox	5	24.6	5.0	12.0	1.8	4.6	1.1

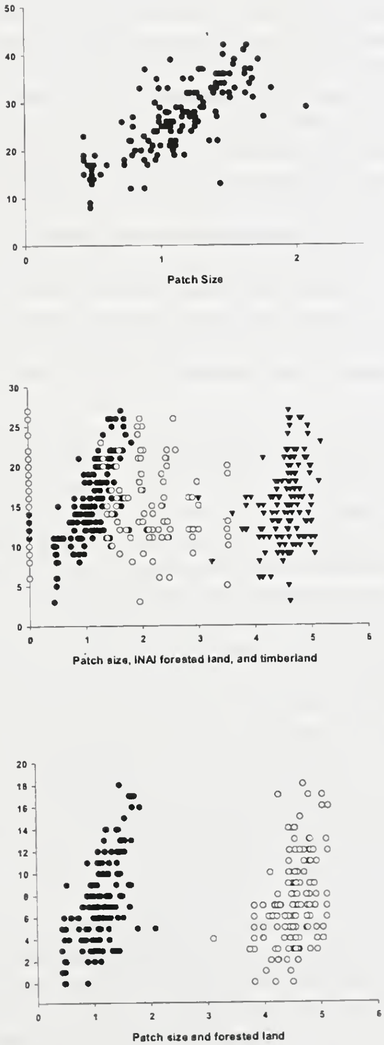
should be noted that the American Bottoms (mean = 37) and Shawnee Partnerships (mean = 34) had the greatest species richness and the Upper Salt Creek Partnership had the lowest (mean = 16). Significant differences were found for habitat dependent species among the Ecosystem Partnerships (ANOVA; F = 4.096, df = 21, $P < 0.001$). The Shawnee Partnership had the greatest number of habitat dependent species (mean = 24.5) compared to the Upper Salt Creek Partnership (mean = 7.33). In addition, significant differences among Ecosystem Partnerships were found for area sensitive species (ANOVA; F = 4.096, df = 21, P = 0.002) (Table 2). The Shawnee Partnership had the greatest number of area sensitive species (mean = 16.5) compared to nearly all Ecosystem Partnerships with the exception of the Driftless Area, Spoon River, Embarrass River, American Bottoms, Carlyle Lake, Big River, and Lower Kaskaskia (Table 2). In addition, fewer area sensitive species were found at the Upper Salt Creek compared to Carlyle Lake, Big River, and Lower Kaskaskia (Table 2).

Landscape Forest Matrix and Patch Size: For species richness the multiple regression was a good fit ($R^2_{adj} = 0.577$) and the overall relationship was significant ($F_{4,132} = 47.421$, $P < 0.001$). With other variables held constant species richness showed a strong positive relationship with patch size ($P < 0.001$; Figure 3), but not with the amount of forested land (forest, woodland, and conifers) (P = 0.314), timberland (P = 0.357) or INAI forested land (P = 0.077). In the case of habitat dependent species the multiple regression was also a good fit ($R^2_{adj} = 0.552$; Figure 3) and the overall relationship was significant ($F_{4,132} = 42.832$, $P < 0.001$). Habitat dependent species had a positive relationship with patch size ($P < 0.001$), timberland (P = 0.035) and INAI forested land (P = 0.016), but not total forested land (P = 0.297). Finally, the multiple regression was less of a good fit for area sensitive species ($R^2_{adj} = 0.470$; Figure 3), but the overall relationship was significant ($F_{4,132} = 31.122$, $P < 0.001$). Area sensitive species had a significant positive

Table 2: Mean (± standard error) species richness (SpR), number of habitat dependent (HD) and area sensitive species (AS) for Ecosystem Partnerships. Italicized partnerships were not used for statistical analyses.

Ecosystem Partnerships	**N**	**MeanSpR**	**SE**	**MeanHD**	**SE**	**MeanAS**	**SE**
Upper Rock River	2	26.5	4.5	13.5	2.5	5.0	2.0
LaMoine River	5	22.2	5.3	13.0	2.5	6.0	1.4
Lower Rock River	8	26.9	2.6	15.0	1.5	5.8	1.2
Big Rivers	6	30.3	1.3	19.8	0.8	10.7	0.7
Driftless Area	3	27.7	5.5	16.7	2.7	8.7	1.7
Lower Sangamon	13	28.2	1.8	15.2	1.1	6.2	0.6
Headwaters	4	24.8	1.9	12.5	0.6	4.0	0.4
Embarras River	7	26.3	2.3	17.3	2.0	8.9	1.4
Upper Little Wabash	8	25.3	2.3	14.3	2.4	6.9	1.4
Kaskaskia River	5	22.6	1.9	12.8	1.3	6.0	1.3
Upper Kaskaskia	1	30.0	0.0	13.0	0.0	4.0	0.0
Upper Des Plaines	1	29.0	0.0	12.0	0.0	5.0	0.0
Chicago Wilderness	1	36.0	0.0	16.0	0.0	8.0	0.0
Upper Salt Creek of the Sangamon	3	15.0	1.5	7.3	0.7	0.7	0.7
Carlyle Lake	7	27.4	3.4	17.4	2.1	10.0	1.3
Spoon River	5	25.6	3.0	15.2	1.4	7.2	1.6
Prairie Parklands	2	20.0	11.0	9.5	4.5	5.0	1.0
Shawnee	2	34.0	1.0	24.5	1.5	16.5	0.5
Mississippi Western Five	3	25.0	2.1	15.0	1.5	6.0	0.6
Kankakee River	2	16.0	1.0	8.5	2.5	3.5	2.5
Kinkaid Area Watershed	1	39.0	0.0	18.0	0.0	12.0	0.0
American Bottoms	2	37.0	0.0	20.0	4.0	9.5	2.5
Fox River	5	24.6	5.0	12.0	1.8	4.6	1.1
Illinois River	3	29.0	2.9	13.3	1.3	6.0	1.7
Heart of the Sangamon	1	19.0	0.0	12.0	0.0	3.0	0.0
Mackinaw River	1	29.0	0.0	12.0	0.0	2.0	0.0
Rock River	1	25.0	0.0	14.0	0.0	3.0	0.0
Cache River	1	42.0	0.0	27.0	0.0	17.0	0.0
Sinkhole Plain	1	37.0	0.0	21.0	0.0	11.0	0.0
Lower Kaskaskia	2	32.0	0.0	19.5	1.5	13.0	1.0
Sugar-Pecatonica	3	19.0	4.7	11.7	3.2	4.0	1.5
Vermilion	1	33.0	0.0	18.0	0.0	9.0	0.0
Not Assigned	30	26.2	1.4	16.3	0.9	8.6	0.8

Figure 3: Multiple regressions for species richness, habitat dependent and area sensitive species using as regressors patch size, forested land (forest, woodland, and conifers), timberland and INAI forested land. Only significant relationships are shown.



relationship with patch size ($P < 0.001$) and amount of forested land within the county ($P = 0.016$), but not with timberland ($P = 0.084$) and INAI forested land ($P = 0.086$).

Discussion

Patterns of bird diversity across Illinois can be examined in many different ways, such as diversity within a region or diversity within an area stewarded by a particular group (i.e., Ecosystem Partnerships). Information regarding how forest bird species are doing in Illinois is important for land mangers since they are trying to determine how to protect and manage our forests with the ultimate goal of maintaining or increasing species diversity within these forests. The results of this report demonstrate that depending upon the scale of the comparison different patterns of diversity can be found. For example, in the case of northern, central, and southern Illinois we did not find differences for species richness. Given the scale of the regions (i.e., 1/3 of the state) and the fact that CTAP has numerous study sites in each of these regions, there is ample opportunity to find a rich and relatively equal number of bird species in each region. This is especially true considering the fact that each area of the state has several bird species that are uncommon to rare in the other two areas of the state. However, in the case of habitat dependent and area sensitive bird species the southern 1/3 of the state had greater numbers of these species due, most likely, to the larger tracts of forest found in this portion of the state (see explanation below).

In the case of Natural Divisions we found significant differences for species richness. Most were associated with CTAP forest sites in the Lower Mississippi River Bottomlands, Ozark, and Shawnee Hills divisions. These natural divisions have larger forest tracts, which usually can harbor greater avian species diversity. In addition, these larger forest tracks allow us to have more census points. Therefore an already diverse avian assemblage may show an even greater than average species richness because with more census points (likely 10-15) there is the increased likelihood that the census will pick up more of the species present in the area. This situation is different in an area like the Rock River Hill Country division, which is composed of sites that have a much more depauperate avifauna due to small size and isolation as the result of a more agricultural land matrix. In addition, the Rock River Hill Country division is in a part of the state where there are fewer possible forest bird species to detect, so it is not surprising that this division had the lowest species richness. However, although the Lower Mississippi River Bottomlands, Ozark and Shawnee Hills divisions had very similar species richness (Figure 2) the Lower Mississippi River Bottomlands division had slightly more (i.e., not statistically different). Several studies have shown that avian species diversity is higher in bottomland forest areas than upland forested areas (Stauffer and Best 1980, Robinson *et al.* 1997).

In addition to species richness differences among Natural Divisions we also found differences for habitat dependent and area sensitive species. The Ozark and Shawnee Hill divisions had the highest number of habitat dependent and/or area sensitive species. This should not be surprising since bird species richness in a forest may not necessarily be composed of just forest species (habitat dependent and area sensitive species) due to the surrounding landscape (i.e., grassland or pasture habitats and shrubland habitats). In addition, both the Ozark and Shawnee Hill divisions are in the southernmost position in the state, have large, continuous, forest tracts and therefore may have the highest number of potential species to record on a census. Again, several studies have shown the importance of habitat area on habitat dependent and area sensitive forest species (Robbins *et al.* 1989, Freemark and Collins1992, Andrade-Renata and Marini 2002).

In the case of Illinois EPA basins and Ecosystem Partnerships (EP) the same reasoning as above can explain the results for species richness and habitat dependent and area sensitive species. The Miss. South basin and the Shawnee EP, in general, have the highest number of species richness and the greatest number of habitat dependent and area sensitive species. This is mostly due to the greater amount of forested and unfragmented habitat in the southern portion of the state (e.g., Shawnee National Forest). For those EPs that have forest sites with greater species diversity or EPs that may want to improve their forests, several things can be done. In general we suggest that southern EPs concentrate more of their conservation efforts to maintain and/or increase the large tracks of unfragmented forest that they currently have. This will involve the potential purchase of adjacent forested land or restoration of degraded forests that in the end will have an optimum shape, representative habitat, and minimum isolation and human induced disturbances (Robbins *et al.* 1989). Even relatively small openings in a forest tract can have negative impacts on the reproductive success of forest bird species, so such areas should be eliminated or otherwise allowed to grow back to forest. Even so-called "wildlife openings", formerly thought to benefit certain forest species, have been shown to have negative effects on forest songbirds, and should be eliminated from forest patches of any size (see Robinson 1992a). In the case of other EPs that historically did not have much forested land and/or their landscapes have been altered, they can still put efforts into maintaining the existing plants, insects, mammals and a portion of the bird community found in their forests through management.

Regarding our question of the forest landscape matrix, our results in an indirect way suggest that the forest matrix surrounding a CTAP forest site can influence species richness, and habitat dependent and area sensitive species. We found that the amount of forested land (forest, woodland, and conifers), timberland or INAI forested land in the surrounding landscape does not seem to have any impact on bird species richness within CTAP forest sites. This is likely due to the fact that when a forested area becomes more and more fragmented (i.e. forest habitat decreases), the forest bird species that are lost in this transition are replaced by just as many or possibly more bird species that are either edge specialists or species common to other, more open habitats (e.g. Indigo Bunting, Song Sparrow, Common Yellowthroat, etc.). However, when it comes to habitat dependent or area sensitive species the amount and quality of the landscape can be an issue. For habitat dependent species the amount of timberland and INAI forested land, but not total forested land seems to influence the number of habitat dependent species at a site. But, in the case of area sensitive species it is the total forested land that seems to influence the number of these species at our sites. All of this is telling us that factors including forest anthropogenic degradation, forest structure, age of tract, and tree species diversity within forested acres adjacent to CTAP forest sites can have a big effect on these bird species at a landscape level. Although some evidence has been presented that shows the surrounding landscape can affect habitat dependent and area sensitive species, the results presented in this report should be taken with caution. Our landscape scale is at the county level and may be too broad to establish any conclusive patterns. Nonetheless, we believe that this is a useful initial step toward the landscape data analysis that can be done with the CTAP data.

Finally, although the forest landscape matrix can play a role in bird species diversity at our sites, it is patch size that has the greatest impact on species richness and habitat dependent and area sensitive species. Overall we found that as patch size increases, the higher the species richness and the greater the number of habitat dependent and area sensitive species can be found at a site (Figure 3). These results have confirmed what other studies, not only in Illinois but in other states and worldwide, have found, that larger tracts of forested land or habitat are needed to support habitat dependent and area sensitive bird species (Galli *et al.* 1976, Ambuel and Temple 1983, Blake and Karr 1984, Opdam *et al.* 1985, Freemark and Meriam 1986, Blake and Karr 1987, Robbins *et al.* 1989; Herkert *et al.* 1993).

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Importance of Monitoring Terrestrial Arthropod Biodiversity in Illinois Ecosystems, with Special Reference to Auchenorrhyncha

Adam Wallner and Chris Dietrich

Introduction

Because arthropods are the most diverse group of terrestrial organisms both in numbers of species, and in behavior and ecological traits, arthropod assemblages provide an invaluable source of data for use in monitoring and conserving biological diversity (Brown 1991, Kremen *et al.* 1993). Despite increased awareness of the importance of terrestrial arthropods (Samways 1994, Samson and Knopf 1994, Arenz and Joern 1996), most monitoring programs continue to rely on other, less diverse groups of organisms, or incorporate an extremely limited subset of the overall arthropod fauna. However, reliance on data from a few well-known taxa such as birds or butterflies assumes that variation in the diversity of these groups is strongly correlated with the diversity of unsampled groups; thus far, there is little evidence to support this assumption (Prendergast *et al.* 1993). Indeed, different groups of organisms respond quite differently to different kinds of environmental perturbations, either natural or anthropogenic. For example, disturbances such as fire may have drastically different effects on plants and insects (Daubenmire 1968, Cancelado and Yonke 1970).

Since arthropods are extremely sensitive to environmental change they are an excellent model for monitoring changes in the natural landscape, which will provide useful data on species abundance and distribution patterns, provide a list of endemic, rare, and economically important species, and observe effects of disturbance on natural communities. However, the potential number of species sampled is enormous (more than 17,000 species of arthropods are known to occur in Illinois). With nearly four times the number of vascular plants and vertebrate animal species combined (Post 1991), it is important to choose a taxon that is well studied, readily identifiable, and is affected by landscape disturbance, such as Auchenorrhynchous Homoptera or AH (i.e., leafhoppers, planthoppers, spittle bugs, and treehoppers). This particular group of sap-sucking herbivores is ideal for monitoring because they are highly diverse and abundant in most terrestrial habitats, are habitat and host specific, are highly sensitive to environmental change, and have been extensively studied in Illinois. Dwight M. DeLong, from the Illinois Natural History Survey, conducted an extensive survey of Illinois leafhopper taxonomy and distribution in the 1940s (DeLong 1948). Wilson and McPherson (1981), from Southern Illinois University at Carbondale, also conducted an extensive survey of Illinois planthoppers taxonomy and distribution in the early 1980s. These surveys and the life history characteristics of the AH species made them the principal insect group for Critical Trends Assessment Program (CTAP) to monitor, in addition to all the other terrestrial insect orders collected in grassland, wetland, and forest habitats across Illinois.

In this report we present the first five years of terrestrial insect data collected from 1997-2001 that will serve as the baseline for future monitoring of terrestrial arthropods across Illinois grassland, wetland, and forest habitats for CTAP. Our main objectives are to: 1) compare terrestrial arthropod species richness across habitats; 2) examine relationships in species richness among arthropod taxonomic groups, 3) evaluate if Auchenorrhynchous species are a predictor of other arthropod taxa as well as overall arthropod diversity. In addition in this report we will provide a list of some new state and county records of auchenorrhychous Homoptera species.

Methods

Sampling: From 1997 to 2001, a total of 388 terrestrial arthropod samples were collected: 128 from forests, 127 from grasslands, and 133 from wetlands. Quantitative sampling for terrestrial arthropods consisted of two 50 m linear transects at each site, using a standard sweep net (100 sweeps). Terrestrial arthropods were then transferred into PTOIEs (Photo Tactic Optimal Insect Extractors) for 30 minutes. Samples were later placed in plastic bags and stored in a freezer for later sorting. After processing, all samples were stored in vials of 70% ethanol.

Specimen Identification: All terrestrial arthropods were sorted and identified to order using the "morphospecies" approach. In this approach, specimens are sorted into groups (morphospecies) based on distinctive morphological characteristics, but these putative characteristics remained unnamed. Relatively little time is required to count the number of morphospecies in a typical sample. These morphospecies counts provide a convenient means for estimating and comparing species richness and diversity among sites. A disadvantage of the morphospecies approach is that without positive identification of species, it is difficult to compare sites based on their species composition. Although such comparisons could be accomplished by standardizing the definitions of each morphospecies across all sites, this approach is tedious and requires considerable expertise.

Finally, all Auchenorrhynchous were identified to species when possible following DeLong (1948), Wilson and McPherson (1981), Dietrich (1994), and Hamilton (2000). In addition, AH species were classified into two groups following Dietrich and Biyal (1997 and 1998, unpublished CTAP reports): Group 1 -common, widespread, and generalist in host and habitat preference; Group 2 -rare, restricted in distribution, and/or host- or habitat-specific (Table 1).

Data Analysis. Species richness was estimated for each site based on sample counts of species or morphospecies. A Kruskal-Wallis One Way Analysis of Variance on Ranks (KW) followed by a Dunn's test was used to determine differences between habitats for species richness among the terrestrial arthropod taxa. Linear regressions were used to determine the extent to which AH species richness predicted overall species richness and that of other arthropod groups. Because the terrestrial arthropod data was not normally distributed, data transformations (square root plus 0.375) were implemented for the regression analyses.

Results

Species richness across habitats: Different patterns of species richness were observed for each taxon group (Figure 1, all KW: H values > 7.03, all P values < 0.030). Auchenorrhynchous

Table 1: List of auchenorrhynchous Homoptera (AH) species collected randomly from forests, grasslands, and wetlands across Illinois. The 'H Group' indicates the level of conservatism (1 = generalist species, vagile, exotic; and 2 = host-plant and or habitat specific, native, poor flyer (see report for additional information), and 'Origin' indicates the location of AH species, according to literature and museum specimens.

Species Name	H Group	Origin
Acanalonia bivittata	1	Native, Nearctic
Acanalonia conica	1	Native, Nearctic
Acanalonia sp.	1	Native, Nearctic
Aceratagalia vulgaris	2	Native, Nearctic
Aceratagallia sp.	2	Native, Nearctic
Aceratagallia uhleri	2	Native, Nearctic
Acertagallia sanguinolenta	2	Native, Nearctic
Acutalis tartarea	1	Native, Nearctic
Agallia constricta	1	Native, Nearctic
Agallia sp.	1	Native, Nearctic
Agallopsis novella	1	Native, Nearctic
Agallopsis sp.	2	Native, Nearctic
Alebra albostriella	2	Native, Nearctic
Amblysellus curtisii	1	Native, Nearctic
Amplicephalus osborni	1	Exotic, Palaearctic (Europe)
Anormenis septentrionalis	1	Native, Nearctic
Anoscopus flavistriatus	1	Exotic, Palaearctic (Europe)
Anoscopus serratulae	1	Exotic, Palaearctic (Europe)
Apache degeerii	2	Native, Neartic
Aphrodes bicincta	1	Exotic, Palaearctic (Europe, Asia)
Aphrophora quadrinotata	1	Native, Nearctic
Aphrophora sp.	1	Native, Nearctic
Athysanus argentanus	1	Exotic, Palaearctic (Europe)
Atymna helena	2	Native, Nearctic
Atymna sp.	2	Native, Nearctic
Atymna sp.1	2	Native, Nearctic
Bakerella rotundifrons	2	Native, Nearctic
Balcultha abdominalis	1	Native, Nearctic
Balcultha impicta	1	Native, Nearctic
Balcultha impunctata	2	Exotic, Palaearctic (Europe)
Balcultha neglecta	1	Native, Nearctic
Balcultha sp.	1	Native, Nearctic
Bruchomorpha dorsata	2	Native, Nearctic
Bruchomorpha oculata	2	Native, Nearctic
Bruchomorpha pallidipes	2	Native, Nearctic
Bruchomorpha sp.	2	Native, Nearctic

Table 1. continued.

Species Name	H Group	Origin
Campylenchia latipes	2	Native, Nearctic
Catonia cinctifrons	2	Native, Nearctic
Cedusa sp.	2	Native, Nearctic
Cedusa sp.1	2	Native, Nearctic
Cedusa sp.2	2	Native, Nearctic
Cedusa sp.3	2	Native, Nearctic
Chloriona slossoni	2	Native, Nearctic
Chlorotettix spatulatus	2	Native, Nearctic
Chlorottetix balli	2	Native, Nearctic
Chlorottetix dentatus	2	Native, Nearctic
Chlorottetix fallax	1	Exotic, Palaearctic (Europe)
Chlorottetix galabanatus	1	Native, Nearctic
Chlorottetix limosus	2	Native, Nearctic
Chlorottetix lusorius	2	Native, Nearctic
Chlorottetix sp.	1	Native, Nearctic
Chlorottetix sp.1	2	Native, Nearctic
Chlorottetix sp.2	2	Native, Nearctic
Chlorottetix suturalis	2	Native, Nearctic
Chlorottetix tergatus	2	Native, Nearctic
Chlorottetix unicolor	2	Native, Nearctic
Chlorottetix viridius	2	Native, Nearctic
Cicadula melanogaster	2	Native, Nearctic
Cicadula sp.	2	Native, Nearctic
Cixius basalis	2	Native, Nearctic
Cixius sp.	2	Native, Nearctic
Cixius sp.1	2	Native, Nearctic
Cixius sp.2	2	Native, Nearctic
Cixius sp.3	2	Native, Nearctic
Clastoptera achatina	2	Native, Nearctic
Clastoptera obtusa	2	Native, Nearctic
Clastoptera proteus	2	Native, Nearctic
Clastoptera xanthocephala	2	Native, Nearctic
Colladonus clitellarius	2	Native, Nearctic
Crytolobus inermis	2	Native, Nearctic
Crytolobus maculifrontis	2	Native, Nearctic
Crytolobus sp.	2	Native, Nearctic
Crytolobus sp.1	2	Native, Nearctic
Daltonia estacada	2	Native, Nearctic
Delphacodes analis	2	Native, Nearctic
Delphacodes basivitta	2	Native, Nearctic
Delphacodes campestris	2	Native, Nearctic

Table 1. continued.

Species Name	H Group	Origin
Delphacodes hyalina	2	Native, Nearctic
Delphacodes lutulenta	2	Native, Nearctic
Delphacodes magna	2	Native, Nearctic
Delphacodes mcateei	2	Native, Nearctic
Delphacodes pacifica	2	Native, Nearctic
Delphacodes pellucida	2	Native, Nearctic
Delphacodes pitens	2	Native, Nearctic
Delphacodes propinqua	2	Native, Nearctic
Delphacodes puella	2	Native, Nearctic
Delphacodes sp.	1	Native, Nearctic
Delphacodes sp.1	2	Native, Nearctic
Delphacodes sp.2	2	Native, Nearctic
Delphacodes sp.3	2	Native, Nearctic
Delphacodes sp.4	2	Native, Nearctic
Delphacodes sp.5	2	Native, Nearctic
Delphacodes sp.6	2	Native, Nearctic
Delphacodes sp.7	2	Native, Nearctic
Delphacodes sp.8	2	Native, Nearctic
Deltacephalus balli	2	Native, Nearctic
Dikraneura angustata	1	Native, Nearctic
Dikraneura mali	1	Native, Nearctic
Dikraneura sp.	1	Native, Nearctic
Dikraneura sp.1	2	Native, Nearctic
Dikraneura sp.10	2	Native, Nearctic
Dikraneura sp.2	2	Native, Nearctic
Dikraneura sp.3	2	Native, Nearctic
Dikraneura sp.4	2	Native, Nearctic
Dikraneura sp.5	2	Native, Nearctic
Dikraneura sp.6	2	Native, Nearctic
Dikraneura sp.7	2	Native, Nearctic
Dikraneura sp.8	2	Native, Nearctic
Dikraneura sp.9	2	Native, Nearctic
Dikrella cruentata	2	Native, Nearctic
Dikrella sp.	1	Native, Nearctic
Dikrella sp.1	2	Native, Nearctic
Dikrella sp.2	2	Native, Nearctic
Dikrella sp.3	2	Native, Nearctic
Dikrella sp.4	2	Native, Nearctic
Doratura stylata	1	Exotic,.Palaearctic (Europe)
Draeculacephala angulifera	2	Native, Nearctic
Draeculacephala antica	1	Native, Nearctic

Table 1. continued.

Species Name	H Group	Origin
Draeculacephala constricta	1	Native, Nearctic
Draeculacephala inscripta	2	Native, Nearctic
Draeculacephala mollipes	2	Native, Nearctic
Draeculacephala noveboracensis	2	Native, Nearctic
Draeculacephala palodusa	2	Native, Nearctic
Draeculacephala robinsini	1	Native, Nearctic
Draeculacephala sp.	1	Native, Nearctic
Driotura gammaroides	2	Native, Nearctic
Driotura robusta	2	Native, Nearctic
Elymana acuma	2	Native, Nearctic
Elymana caduca	2	Native, Nearctic
Elymana inornata	2	Native, Nearctic
Empoasca fabae	1	Native, Nearctic
Empoasca recurvata	1	Native, Nearctic
Empoasca sp.	1	Native, Nearctic
Empoasca sp.1	1	Native, Nearctic
Empoasca sp.2	2	Native, Nearctic
Empoasca sp.3	1	Native, Nearctic
Empoasca sp.4	2	Native, Nearctic
Empoasca sp.5	2	Native, Nearctic
Empoasca sp.6	2	Native, Nearctic
Enchenopa binotata	2	Native, Nearctic
Endria inimica	1	Native, Nearctic
Entylia bactriana	1	Native, Nearctic
Entylia carinata	2	Native, Nearctic
Erythroneura sp.	1	Native, Nearctic
Erythroneura sp.1	2	Native, Nearctic
Erythroneura sp.10	2	Native, Nearctic
Erythroneura sp.2	2	Native, Nearctic
Erythroneura sp.4	2	Native, Nearctic
Erythroneura sp.5	2	Native, Nearctic
Erythroneura sp.6	2	Native, Nearctic
Erythroneura sp.7	2	Native, Nearctic
Erythroneura sp.8	2	Native, Nearctic
Erythroneura sp.9	2	Native, Nearctic
Erythroneura vitis	2	Native, Nearctic
Erythronuera sp.3	2	Native, Nearctic
Euides sp.	2	Native, Nearctic
Euides weedi	2	Native, Nearctic
Eupteryx flavoscuta	2	Native, Nearctic
Evacanthus nigramericanus	2	Native, Nearctic

Table 1. continued.

Species Name	H Group	Origin
Exitanius exitiosus	1	Exotic, Palaearctic (Europe)
Extrusanus extrusus	2	Native, Nearctic
Flexamia atlantica	2	Native, Nearctic
Flexamia reflexa	2	Native, Nearctic
Flexamia sp.	2	Native, Nearctic
Flexamia sp.1	2	Native, Nearctic
Forcipata loca	1	Native, Nearctic
Graminella aureovittata	2	Native, Nearctic
Graminella fitchi	1	Native, Nearctic
Graminella nigrifrons	1	Native, Nearctic
Graminella sp.	1	Native, Nearctic
Graphacephala versuta	1	Native, Nearctic
Graphocephala coccinea	1	Native, Nearctic
Graphocephala hieroglyphica	2	Native, Nearctic
Graphocephala sp.	1	Native, Nearctic
Gypona contona	2	Native, Nearctic
Gyponana brevita	2	Native, Nearctic
Gyponana conferta	2	Native, Nearctic
Gyponana expanda	2	Native, Nearctic
Gyponana melanota	2	Native, Nearctic
Gyponana ortha	2	Native, Nearctic
Gyponana panda	2	Native, Nearctic
Gyponana sp.	2	Native, Nearctic
Gyponana sp.1	2	Native, Nearctic
Hecalus kansiensis	2	Exotic, Nearctic (Western United States)
Hecalus major	2	Native, Nearctic
Hecalus sp.	2	Native, Nearctic
Helochara communis	2	Native, Nearctic
Homalodisca sp.	1	Exotic, Nearctic (West of Rocky Mountains)
Homalodisca triquetra	1	Exotic, Nearctic (West of Rocky Mountains)
Idiocerus distinctus	2	Exotic, Nearctic
Idiocerus nervatus	2	Native, Nearctic
Idiocerus raphus	2	Native, Nearctic
Idiocerus snowi	2	Native, Nearctic
Idiocerus sp.	2	Native, Nearctic
Idiocerus sp.1	2	Native, Nearctic
Idiocerus suturalis	2	Native, Nearctic
Idiocerus taxodium	2	Native, Nearctic

Table 1. continued.

Species Name	H Group	Origin
Idiodonus kennicotti	2	Native, Nearctic
Japananus hyalinus	1	Exotic, Oriental (Japan)
Jikradia olitoria	1	Native, Nearctic
Kansendria kansana	1	Exotic, Nearctic (Kansas, Oklahoma, Texas)
Keonalla dolabrata	2	Native, Nearctic
Laevicephalus slyvestris	2	Native, Nearctic
Latalus missellus	2	Native, Nearctic
Latalus personatus	2	Native, Nearctic
Latalus sayi	1	Native, Nearctic
Latalus sp.	1	Native, Nearctic
Lebradea flavovirens	1	Exotic, Palaearctic (Scandinavia)
Lepyronia gibbosa	2	Native, Nearctic
Lepyronia quadrangularis	1	Exotic, Nearctic (Canada)
Lepyronia sp.	1	Native, Nearctic
Liburniella ornata	1	Native, Nearctic
Limotettix cuneatus	2	Native, Nearctic
Limotettix striolis	2	Native, Nearctic
Macropsis fumipennis	2	Native, Nearctic
Macropsis insignis	2	Native, Nearctic
Macropsis sp.	2	Native, Nearctic
Macropsis sp.1	2	Native, Nearctic
Macropsis sp.2	2	Native, Nearctic
Macrosteles 4-lineatus	1	Native, Nearctic
Macrosteles lepida	2	Native, Nearctic
Macrosteles sp.	1	Native, Nearctic
Macrosteles variata	2	Native, Nearctic
Magicicada sp.	1	Native, Nearctic
Magicicada tredecassini	2	Native, Nearctic
Magicicada tredecim	2	Native, Nearctic
Mensoma cincta	2	Native, Nearctic
Metcalfa pruinosa	1	Native, Nearctic
Microcentrus perditus	1	Exotic, Nearctic (Missouri)
Micrutalis calva	1	Native, Nearctic
Myndus sp.	2	Native, Nearctic
Myndus sp.1	2	Native, Nearctic
Neocoelidia tumidifrons	2	Native, Nearctic
Neohecalus magnificus	2	Native, Nearctic
Neokolla gothica	2	Native, Nearctic
Norvellina seminuda	2	Native, Nearctic
Norvillina sp.	2	Native, Nearctic

Table 1. continued.

Species Name	H Group	Origin
Oncometopia orbona	1	Native, Nearctic
Oncometopia sp.	1	Native, Nearctic
Oncopsis sp.	2	Native, Nearctic
Ormiendus venusta	1	Native, Nearctic
Osbornellus auronitens	1	Native, Nearctic
Osbornellus consors	2	Native, Nearctic
Osbornellus sp.	1	Native, Nearctic
Otiocerus sp.	2	Native, Nearctic
Otiocerus sp.1	2	Native, Nearctic
Palus sp.	2	Native, Nearctic
Paraphlepsius incisus	2	Native, Nearctic
Paraphlepsius irroratus	1	Native, Nearctic
Paraphlepsius luxurious	2	Native, Nearctic
Paraphlepsius rossi	2	Native, Nearctic (East Coast and Illinois)
Paraphlepsius sp.	1	Native, Nearctic
Paraulazices irrorata	1	Native, Nearctic
Pentagramma variegata	2	Native, Nearctic
Penthimia americana	2	Native, Nearctic
Philaenarcys bileneata	2	Native, Nearctic
Philaenus sp.	1	Exotic, Nearctic (Canada)
Philaenus spumarius	1	Exotic, Nearctic (Canada)
Philaronia abjecta	1	Exotic, Nearctic (Canada)
Phylloscelis atra	2	Native, Nearctic
Phylloscelis pallescens	2	Native, Nearctic
Pintalia dorsovitlata	2	Native, Nearctic
Pissinotus brunneus	2	Native, Nearctic
Pissonotus delicatus	2	Native, Nearctic
Pissonotus dorsalus	2	Native, Nearctic
Pissonotus flabellatus	2	Native, Nearctic
Pissonotus nigra	2	Native, Nearctic
Pissonotus sp.	2	Native, Nearctic
Pissonotus sp.1	2	Native, Nearctic
Pissonotus sp.2	2	Native, Nearctic
Pissonotus sp.3	2	Native, Nearctic
Planicephalus flavicostatus	2	Native, Nearctic
Plesiommata tripunctata	2	Native, Nearctic
Polyamia apicata	2	Native, Nearctic
Polyamia caperata	2	Native, Nearctic
Polyamia compacta	2	Native, Nearctic
Polyamia sp.	2	Native, Nearctic

Table 1. continued.

Species Name	H Group	Origin
Polyamia weedi	2	Native, Nearctic
Ponana scarlitina	2	Native, Nearctic
Ponana sp.	2	Native, Nearctic
Prairiana sp.	2	Native, Nearctic
Prokelisia crocea	2	Native, Nearctic
Prosopia bicincta	1	Exotic, Nearctic (Canada, Eastern United States)
Prosopia sp.	1	Exotic, Nearctic (Canada, Eastern United States)
Psammotettix lividellus	1	Exotic, Nearctic (not found in Illinois from 1948)
Publilia concava	1	Native, Nearctic
Publilia reticulata	2	Native, Nearctic
Sanctanus sanctus	2	Native, Nearctic
Scaphoideus cinerosus	2	Native, Nearctic
Scaphoideus crassus	2	Native, Nearctic
Scaphoideus elongatus	2	Native, Nearctic
Scaphoideus forceps	2	Native, Nearctic
Scaphoideus minor	2	Native, Nearctic
Scaphoideus opalinus	2	Native, Nearctic
Scaphoideus sp.	2	Native, Nearctic
Scaphoideus sp.1	2	Native, Nearctic
Scaphoideus sp.2	2	Native, Nearctic
Scaphoideus sp.3	2	Native, Nearctic
Scaphoideus sp.4	2	Native, Nearctic
Scaphoideus sp.5	2	Native, Nearctic
Scaphoideus sp.6	2	Native, Nearctic
Scaphoideus tergatus	2	Native, Nearctic
Scaphoideus titanus	2	Native, Nearctic
Scaphoideus transius	2	Native, Nearctic
Scaphoideus veterator	2	Native, Nearctic
Scaphytopius abbreviatus	2	Native, Nearctic
Scaphytopius acutus	1	Native, Nearctic and Palaearctic
Scaphytopius cinereus	1	Native, Nearctic
Scaphytopius frontalis	2	Native, Nearctic
Scaphytopius rubellus	1	Exotic, Nearctic (East Coast of the United States)
Scaphytopius sp.	1	Native, Nearctic
Scaphytopius sp.1	1	Native, Nearctic
Scolops angustatus	2	Native, Nearctic
Scolops pungens	2	Native, Nearctic

Table 1. continued.

Species Name	H Group	Origin
Scolops sp.	1	Native, Nearctic
Scolops suclipes	2	Native, Nearctic
Sorhoanus pascuellus	1	Exotic, Palaearctic (Europe)
Spissistilus cornutus	2	Native, Nearctic
Spissistilus sp.	2	Native, Nearctic
Spissitilus borealis	2	Native, Nearctic
Stenocranus delicatus	2	Native, Nearctic
Stenocranus sp.	1	Native, Nearctic
Stenocranus sp.1	2	Native, Nearctic
Stictocephala bisonia	1	Native, Nearctic
Stictocephala brevitylus	2	Native, Nearctic
Stictocephala lutea	2	Native, Nearctic
Stictocephala sp.	1	Native, Nearctic
Stictocephala taurina	2	Native, Nearctic
Stirellus bicolor	1	Native, Nearctic
Stirellus obtusus	2	Native, Nearctic
Stobaera tricarinata	2	Native, Nearctic
Syndoche impunctata	2	Native, Nearctic
Syntames uhleri	2	Native, Nearctic
Telamona unicolor	2	Native, Nearctic
Texanus sp.	2	Native, Nearctic
Thamnotettix simplex	1	Exotic, Palaearctic (Europe)
Thionia simplex	2	Native, Nearctic
Tinobregnus viridescens	2	Native, Nearctic
Tylozygus bifidus	1	Native, Nearctic
Typhlocyba sp.	1	Native, Nearctic
Xestocephalus brunneus	2	Native, Nearctic
Xestocephalus piceus	2	Exotic, Nearctic (Ohio)
Xestocephalus pulicarius	2	Native, Nearctic
Xestocephalus sp.	2	Native, Nearctic

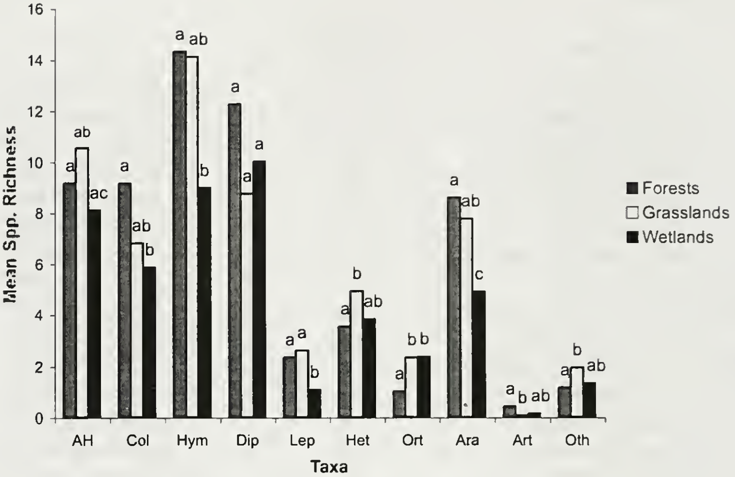

Fig. 1. Mean arthropod species richness across forest, wetlands, and grasslands, from 1997 to 2001.

species richness was higher in grasslands than wetlands and forests, although only significant differences were found between grasslands and wetlands. Coleoptera species richness was significantly higher in forests. Hymenoptera, Lepidoptera, and Arachnida species richness were significantly lower in wetlands than other habitats. Diptera species richness was not statistically different among any of the habitats.

Relationships among arthropod groups: A stronger relationship was observed between Coleoptera and total non-AH terrestrial arthropod species richness (Figure 2a) than AH species richness and total non-AH terrestrial arthropod species richness (Figure 2b). Hyperdiverse orders, such as Coleoptera showed a significant relationship to Hymenoptera (Figure 3). Heteroptera (i.e., seed, plant, and stink bugs) species richness had the highest significant relationship to AH species richness than any other terrestrial arthropod group (Figure 4).

AH State and County Records: A total of 344 AH species were identified. Out of these 344 species 95 species belong to group 1 (24 exotic species and 71 native species) and 249 belong to group 2 (4 exotic species and 245 native species) (Table 1). In addition, 191 out of 344 (56 percent) AH species collected represent new county records (Table 1). Some of the new county records include *Penthimia americana*, an indicator of oak savanna (Figure 5a), *Apache degeerii* (Figure 5b), and *Evacanthus nigramericana* (Figure 5c), which are indicators of highly undisturbed forest sites. Some new county and state records include: *Athysanus argentarius* (Figure 5d), an introduced European species that is known to vector economically important diseases to agriculture crops, which was found in wetlands and grasslands; and *Aphrodes bicinta*

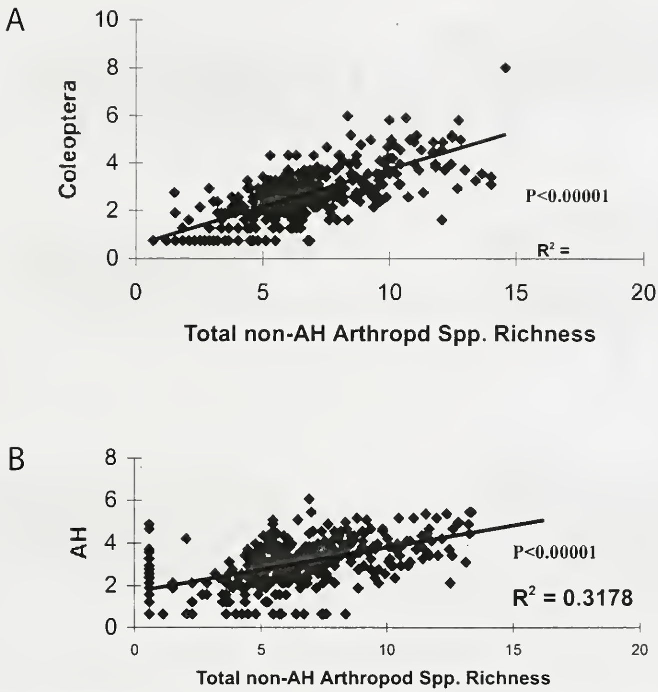

Fig. 2. Relationship (R2) of Coleoptera total species richness to total arthropod species richness (excluding Coleoptera) (a); and AH total species richness correlated to total arthropod species richness (excluding AH) (b).

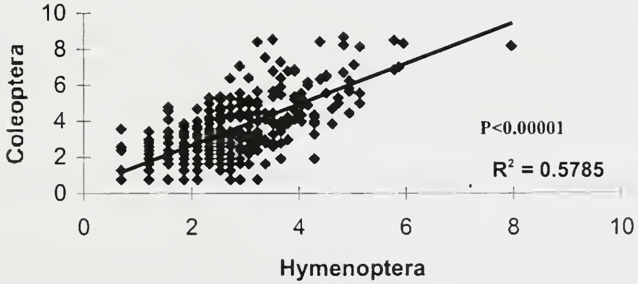

Fig. 3. Relationship of Coleoptera total species richness to Hymenoptera total species richness.

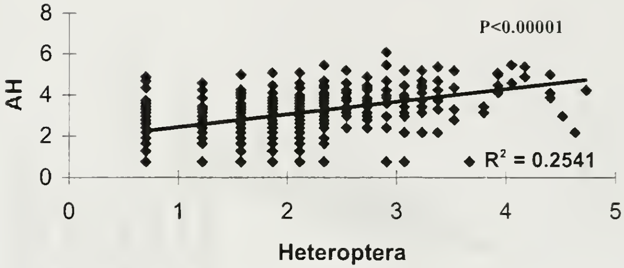

Fig. 4. Heteroptera total species richness relationship (R2) with AH total species richness.

(Figure 5e), also introduced from Europe and is often found with other exotic species, such as *Athysanus argentarus*, and *Lebradea flavovirens* (Figure 5f), which is introduced from Finland. *Lebradea flavovirens* is endangered in Finland and primarily feeds on *Calamagrostis* spp., which commonly occurs in wet prairies and wetland habitats.

Discussion

Species richness across habitats: Differences among habitats for each taxon group were found. In the case of Auchenorrhyncha (AH) species they were more abundant in grasslands, followed by forests, then wetlands, although statisitically significant differences were found only between grasslands and wetlands (Figure 1). Several explanations such as collection period, vegetation stratum sampled, and sampling technique can be provided for these results. Since different habitats were sampled at different times (forests in late spring/early summer, wetlands in early summer, grasslands in mid summer), the higher species richness in grasslands may simply reflect seasonal difference in AH richness among habitats. In addition, collection of AH during mid to late summer is preferred since in temperate regions, AH require several months to reach reproductive maturity, which occurs in mid-to-late June (Nickel 2003).

The vegetation stratum that is sampled can also explain the AH differences between habitats. Only the herbaceous stratum is sampled at CTAP sites. In forests, a lot of the insect diversity is within and just above the forest canopy (see citations in Su and Woods 2001). Due to sampling protocols we may miss additional AH species in forests. In addition, the successional stage of the sampled plant community in the herbaceous stratum may affect AH species richness. Hollier *et al.* (1994) and Stinson and Brown (1983) found that the successional stage and architecture of plant communities are significantly correlated with AH biodiversity. As the plant community

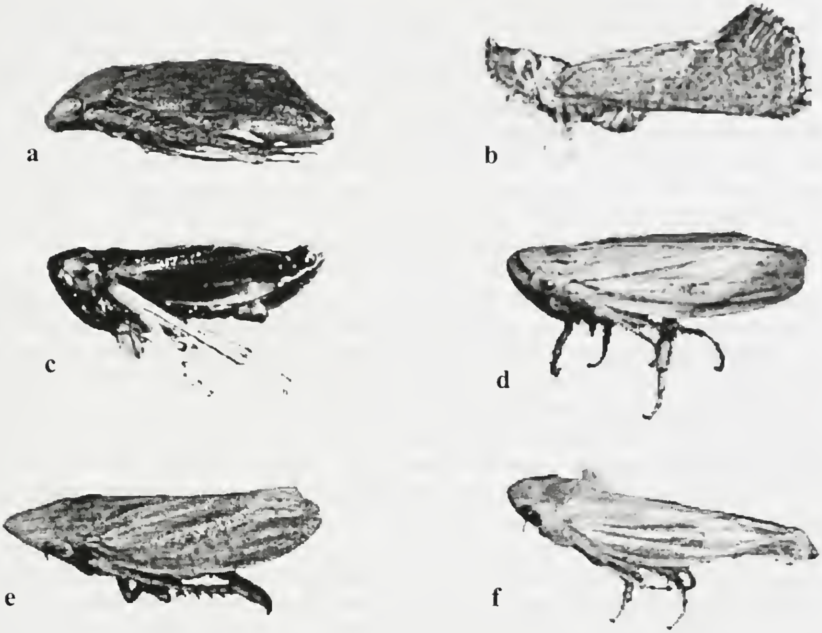

Fig. 5. Images of Penthimia americana (a); Apache degeerii (b); and Evacanthus nigramericana (c); Athysanus argentarius (d); Aphrodes bicincta (e); and Lebradea flavovirens (f).

becomes more diverse in species and the plant architecuture becomes more complex over time, it provides more resource availability for AH colonization. Based on our 1997-2000 CTAP vegetation data (Molano-Flores *et al.* 2002), forests, not grasslands, should have more AH species, since the forest plant communities are more species diverse. However as previously pointed out, AH are collected too early in the forests. In addition, we may be missing AH species due to the sampling technique we employ. Because we only conduct sweeps at our sites, some AH species that occur in the canopy will be missed (see a further explanation under *Terrestrial Arthropod Relationships*).

Other terrestrial arthropods showed different trends in species richness across habitats (Figure 1). Arachnida, Diptera, Coleoptera, Hymenoptera, and Lepidoptera species richness were lower in wetlands than forests and grasslands. The low diversity observed in these orders in the wetland sites may be a result of lower plant diversity. Many of the CTAP wetland sites have high levels of anthropogenic disturbance and are very small, which may have resulted in high extirpation rates of these highly speciose orders. These hyperdiverse groups, which comprise a plethora of guilds such as scavengers, detritivores, predators, parasitoids, and herbivores, may prefer forests because of the complex vertical stratitication that provides suitable habitat to support a variety of niches. However, phytophagous insects such as Orthoptera and Heteroptera had higher numbers of species in grasslands and wetlands than forests. Sap-sucking and leaf-chewing insects may favor grasses and forbs more than trees and shrubs for several reasons: grassland habitats may have a higher carbon to nitrogen ratio, the vertical stratification of grassland habitats may be more preferable for the location of mates, and grassland plant communities may not hinder the insects ability to disperse to new locations as much as forest habitats.

Terrestrial Arthropod Relationships: The species richness of both AH and hyperdiverse groups of terrestrial arthropods, in particular Coleoptera, showed signficiant positive relationships to total non-AH terrestrial arthropod species richness, however AH displayed a lower positive significant relationship to total non-AH arthropod species richness (Figure 2b) than hyperdivese groups of terrestrial arthropods (Figure 2a). This data suggests that hyperdiverse groups of terrestrial arthropods may be less sensitive to habitat disturbance than AH. Many of the sites sampled by CTAP are of poor to moderate quality, thus more vagile, highly competitive species, for example the spittlebug *Philaneus spumarius*, may be replacing other highly conservative species such as *Flexamia spp*. (personal observation). This process is accelerated when natural habitats such as forests and grasslands are fragmented, near an edge, and/or near a matrix of agriculture. Summerville and Crist (2004) studied moth species richness and abundance in fragmented deciduous forest fragments and observed that as forest size decreases or becomes closer to an agricultural landscape, species richness decreases and species composition changes. AH may be more susceptible to nearby agriculture fields and may be more dependent on larger habitat size than Coleoptera.

Another explanation of why Coleoptera species richness displayed a stronger relationship to total non-AH terrestrial arthropods than AH species richness is the sampling technique. Because AH and other terrestrial arthropods were sampled by using a sweep net, additional sampling methods, such as vacuum sampling, should be implemented to determine if these trends (Figures 2a, b) are a naturally occurring phenomenon or sampling artifact. Wilson *et al.* (1993) used a modified leafblower to sample planthopper species and other terrestrial arthropods from aquatic vegetation, as well as grasslands. Their results showed that the leafblower vacuum was more efficient in collecting adults and early instars than other sampling techniques such as the D-vac, sweep, and dip nets. In addition, Nickel (2003) observed that vacuum sampling is the most efficient method of quantitative sampling for Auchenorrhyncha species. Thus, additional sampling at CTAP sites, using a modified leafblower vacuum to sample AH and other terrestrial arthropd may be necessary to efficiently collect all possible AH and other terrestrial arthropod species, and to statistically analyze their differences in species richness and abundance across space and time.

Finally, when comparing non-AH terrestrial arthropods to AH, only Heteroptera displayed the strongest significant positive relationship to AH (Figure 3). This is most likey due to the fact that AH is a subgroup of Heteroptera and they share similar life histories. Most Heteroptera feed on plant sap, have similar numbers of generations, and reach reproductive maturity at the same time as AH. However, among all the non-AH terrestrial arthropod groups, Coleoptera had the strongest significant positive relationship to Hymenoptera (Figure 4). Several reasons that may explain this relationship are: these groups may have similar functional guilds (see Basset *et al.* 2004); have similar distribution patterns of species richness and abundance across space and time; and have similar patterns in abundance and species richness after anthropogenic disturbance. The CTAP data suggests that hyperdiverse groups may be better predictors of other hyperdiverse groups.

AH State and County Records: A somewhat higher proportion of the known Illinois AH fauna were documented (344 from over 900) and most of these species belong to group 2 (i.e., rare, restricted in distribution, and/or host- or habitat-specific; 72 percent). The great number of species that were documented most likely was the result of the sampling method employed, which is particularly effective for this group of insects and to the numbers of sites that have been visited by CTAP. In addition, this sampling effort over a five-year period (1997-2001) has resulted in an increase in the number of new county records for AH. This increase in the number of new state and county records obtained by CTAP demonstrates the need for this type of statewide monitoring program to update current records and detect changes in biodiversity across Illinois which may include the detection of introduced and economically important arthropods. At this point, we have identified 28 exotic species from our CTAP sites.

The collection of terrestrial insects by CTAP between 1997-2001 has provided invaluable data on differences between habitats for terrestrial insects and new state and county records for Auchenorrhynchous Homoptera. These data in combination with the plant and bird data collected by CTAP will allow us to have a better understanding of the overall conditions of our forests, wetlands, and grasslands.

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Aquatic Ecosystems

Illinois Streams: Conditions and Trends

Karen M. Miller

Introduction

Through the combined efforts of scientists at the Illinois Natural History Survey and volunteers with the Illinois RiverWatch Network, the Critical Trends Assessment Program now has detailed information on 814 stream sites statewide. RiverWatch has been collecting stream data for eight years, and the NHS completed its first five-year cycle of data collection in 2001. It is now possible to make generalizations about the condition of streams in different parts of the state by combining the scientists and volunteer data. Also, with up to eight years of data for many sites, it is possible to begin detecting trends.

The monitoring protocols for the professionals and the volunteers were designed to be complementary, and three indicators—the MBI/HBI, EPT, and habitat score—in both data collections are generally comparable. For example, CTAP scientists calculate the Hilsenoff Biotic Index (HBI), a weighted average of the organic pollution tolerance of the aquatic insects they collect. RiverWatch volunteers calculate the Macroinvertebrate Biotic Index (MBI), a similar average but for a more limited number of macroinvertebrates taxa (37). Both scientists and volunteers count the number of **E**phemeroptera (mayflies), **P**lecoptera (stoneflies), and **T**richoptera (caddisflies) taxa collected, although the scientists identify the organisms at a more refined taxonomic level. Similarly, both scientists and volunteers calculate a habitat score, although the CTAP scientists consider a wider range of factors.

Condition by Natural Division

Indicator data for each of the 814 monitoring sites were rated[1] using the same scale to combine the professional and volunteer data. Because RiverWatch sites could have been sampled from a range of eight years to one year, data from the most recent sampling year were used in the analysis.[2] The combined data were then used to compare stream conditions in natural divisions (see Figures 1 - 3).[3]

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[1] Poor = sites ranking in the lower 50% of all sites
Fair = sites ranking between 50% and 75%
Good = sites ranking in the top 25%.

[2] Of the 665 RiverWatch sites used in the analysis, 183 had one year of data; 119, two years; 110, three years; 88, four years; 68, five years, 47, six years; 32, seven years; and 18, eight years of data.

[3] The 14 natural divisions were consolidated into six categories based on similarities in contiguous natural divisions. The Western Forest-Prairie also includes the Middle Mississippi Border Division, the Southern Till Plain includes the Wabash, the Ozark/Shawnee Hills includes the Coastal Plain Division, and the Grand Prairie includes the Upper Mississippi and Illinois River Bottomlands. Only four sites were monitored in the Lower Mississippi Bottomlands and the Sand Area categories and they were excluded from the analysis. Because both areas are small, they may be combined with adjacent natural divisions in the future.

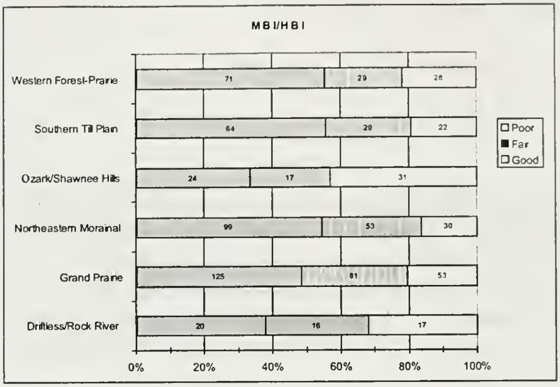


Figure 1. MBI/HBI Site Scores by Natural Division. The numbers on the bars indicate the number of sites.

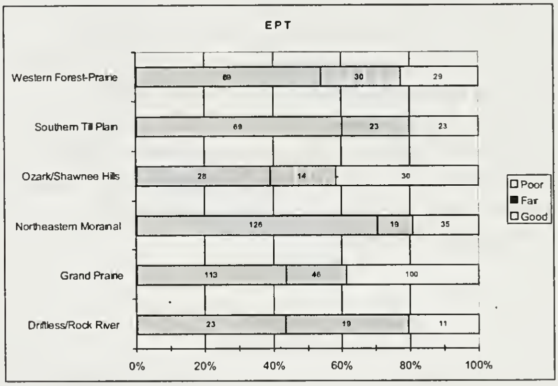


Figure 2. EPT Site Scores by Natural Division. The numbers on the bars indicate the number of sites.

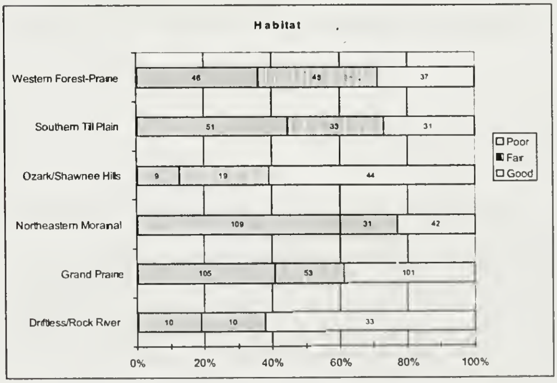


Figure 3. Habitat Site Scores by Natural Division. The numbers on the bars indicate the number of sites.

Overall, the Ozark/Shawnee Hills natural divisions have the highest quality sites. They have the highest percentage of sites rated *Good,* and the lowest percentage rated *Poor*, for both the MBI/HBI and the EPT. Sites in the Driftless/Rock River also scored relatively high for MBI/HBI and for Habitat, but did not do as well in EPT. The Northeastern Morainal fared worst among the natural divisions, with more than half of its sites rated *Poor* for all three indicators. The Western Forest-Prairie and Southern Till Plain also scored poorly on the biological indicators. Figures 4-6 show the statewide distribution of site scores for each of the three indicators.

Trends at RiverWatch Sites

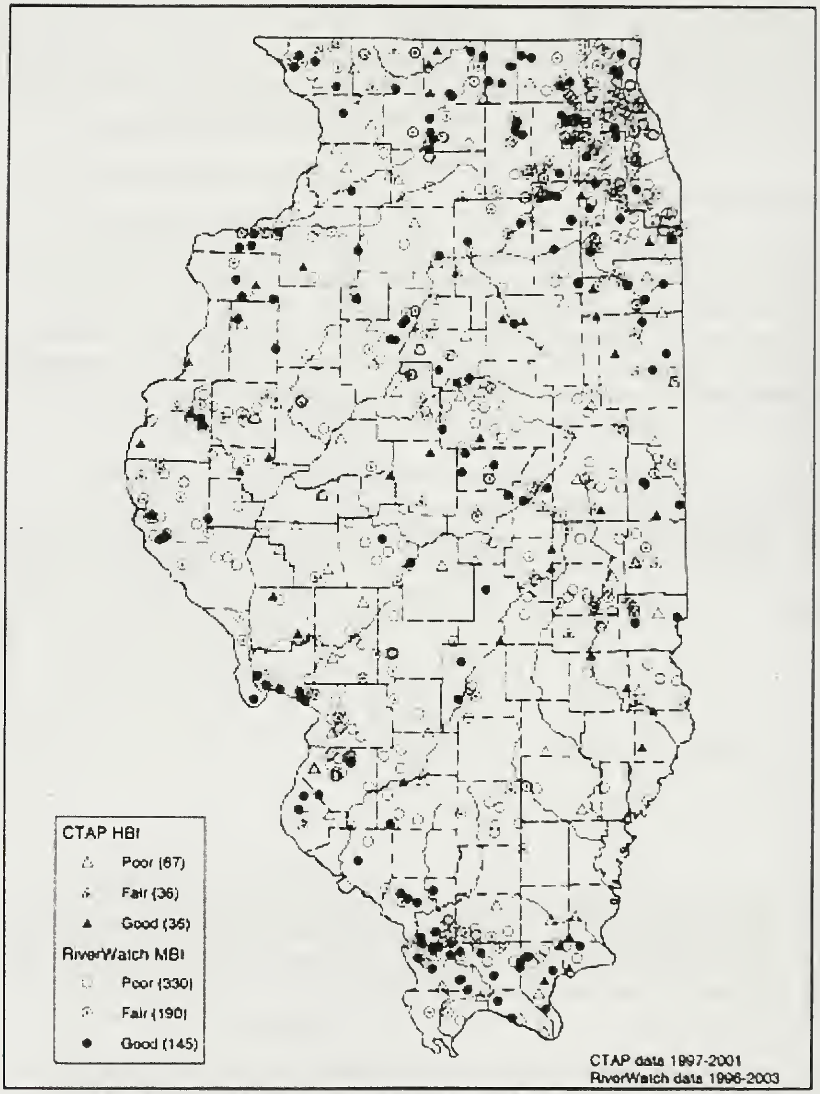
One hundred sixty-five RiverWatch sites have at least five years of data; most show no change in quality. However, using a Pearson two-tailed correlation coefficient (at a confidence level of 90%), 54 indicate a potential change in biological quality. The majority of these show declining quality—32 compared to 22 sites that show improved quality. Figure 7 illustrates the overall positive or negative trends for the 54 sites, and Figures 8 – 10 illustrate the geographic distribution of the sites according to the indicator showing the trend. Table 1 provides these trends broken out by the number of years sites were monitored.

Table 1. Indicator Trends

# of Years Monitored	# of Sites	# and % Showing a Trend	Trends ⇑ or ⇓ BioScore ⇑		EPT ⇑		MBI ⇑	
			BioScore ⇑	BioScore ⇓	EPT ⇑	EPT ⇓	MBI ⇑	MBI ⇓
8	18	6 (33%)	1	0	0	0	3	2
7	32	9 (28%)	3	3	2	3	2	2
6	47	15 (32%)	3	4	3	2	2	4
5	68	24 (35%)	3	8	5	12	2	2
Totals	165	54 (33%)	10	15	10	17	9	10

Only two sites, with seven years of data each, show significant trends in EPT, MBI and the overall biological score: a site on Crystal Creek in the Upper Fox watershed has positive trends while a site on the East Branch of Hurricane Creek in the Embarras watershed has negative trends. Nineteen other sites show trends in two indicators, most (11) in EPT and the overall Biological Score.

Table 2 and Figure 11 provide overall trends by natural division. The most striking is the Ozark/Shawnee Hills natural division, which has the best stream quality in Illinois but appears to be on a downward trend. This is an alarming situation, if true, and requires further investigation. Figures 12 through 17 provide trends for selected individual sites.



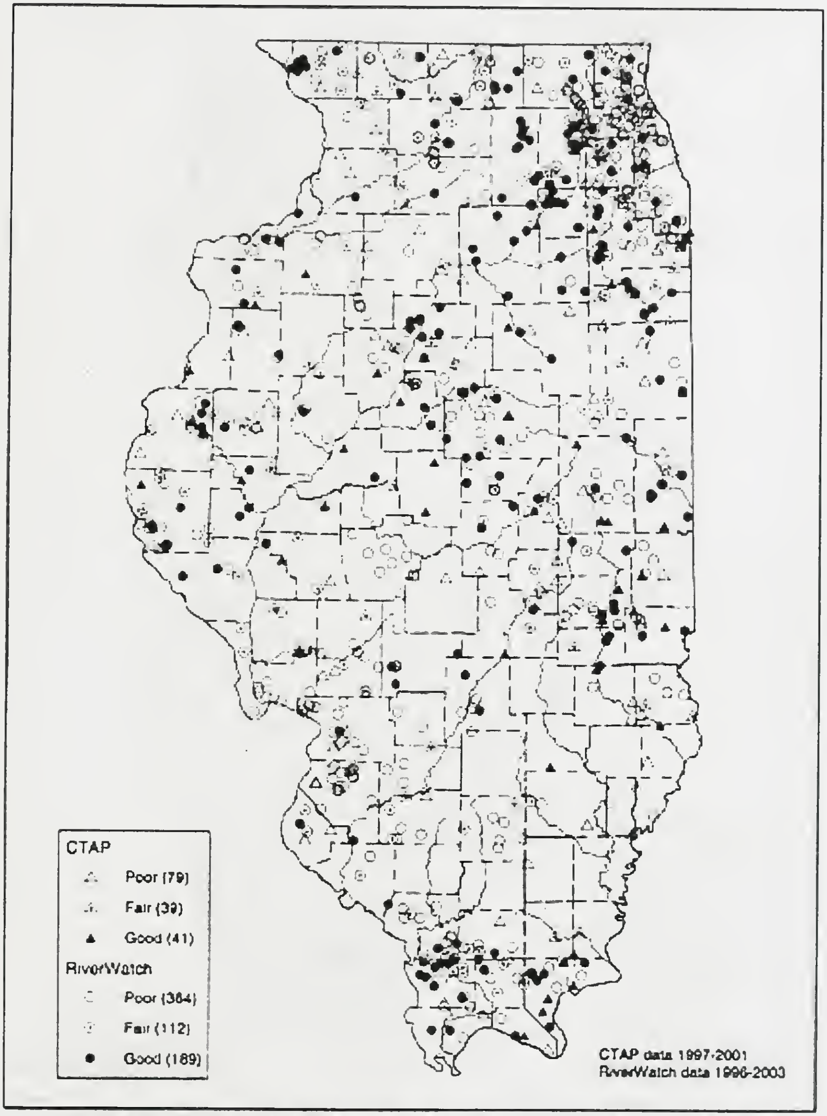
MBI & HBI Scores

CTAP data 1997-2001
RiverWatch data 1996-2003

Poor = sites ranking in the lower 50%, Fair = sites ranking between 50%, and Good = sites ranking in the top 25%

Figure 4

EPT Scores

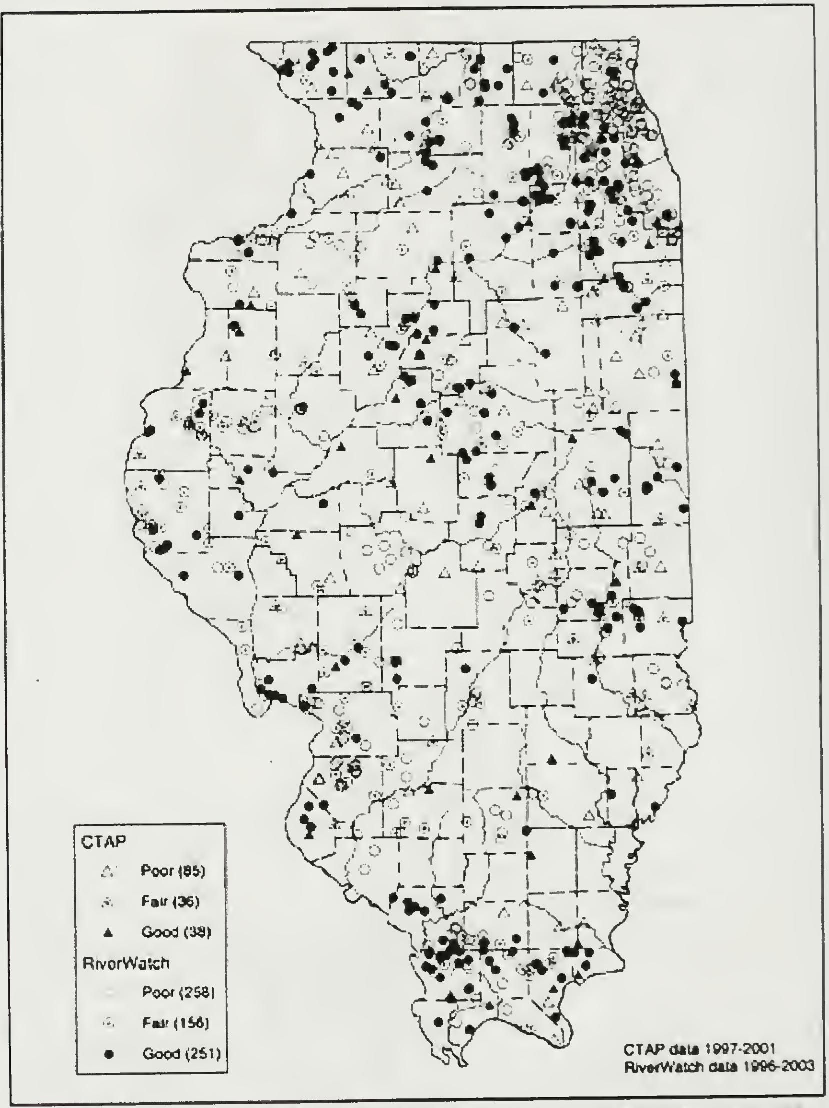


CTAP data 1997-2001
RiverWatch data 1996-2003

Poor = sites ranking in the lower 50%, Fair = sites ranking between 50%, and Good = sites ranking in the top 25%

Figure 5

Habitat Scores



Poor = sites ranking in the lower 60%, Fair = sites ranking between 50%, and Good = sites ranking in the top 25%

Figure 6

RiverWatch Sites Showing Statistically Significant Trends in Biological Indicators

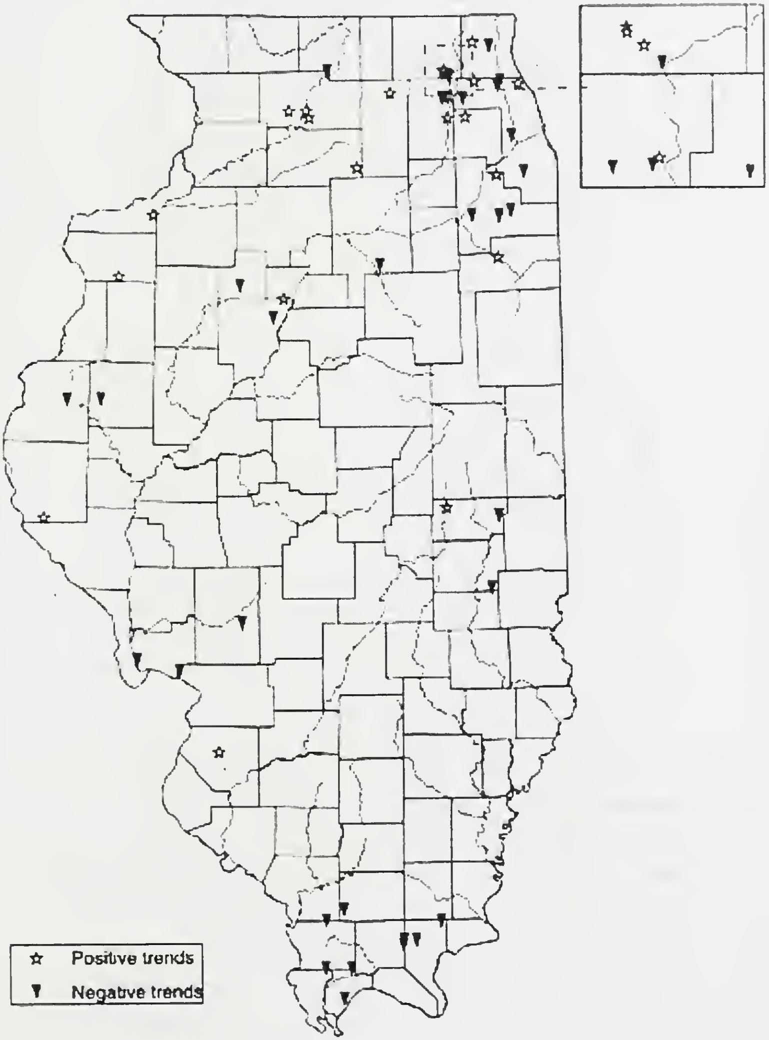


Figure 7

RiverWatch Sites Showing a Statistically Significant Trend in MBI Scores

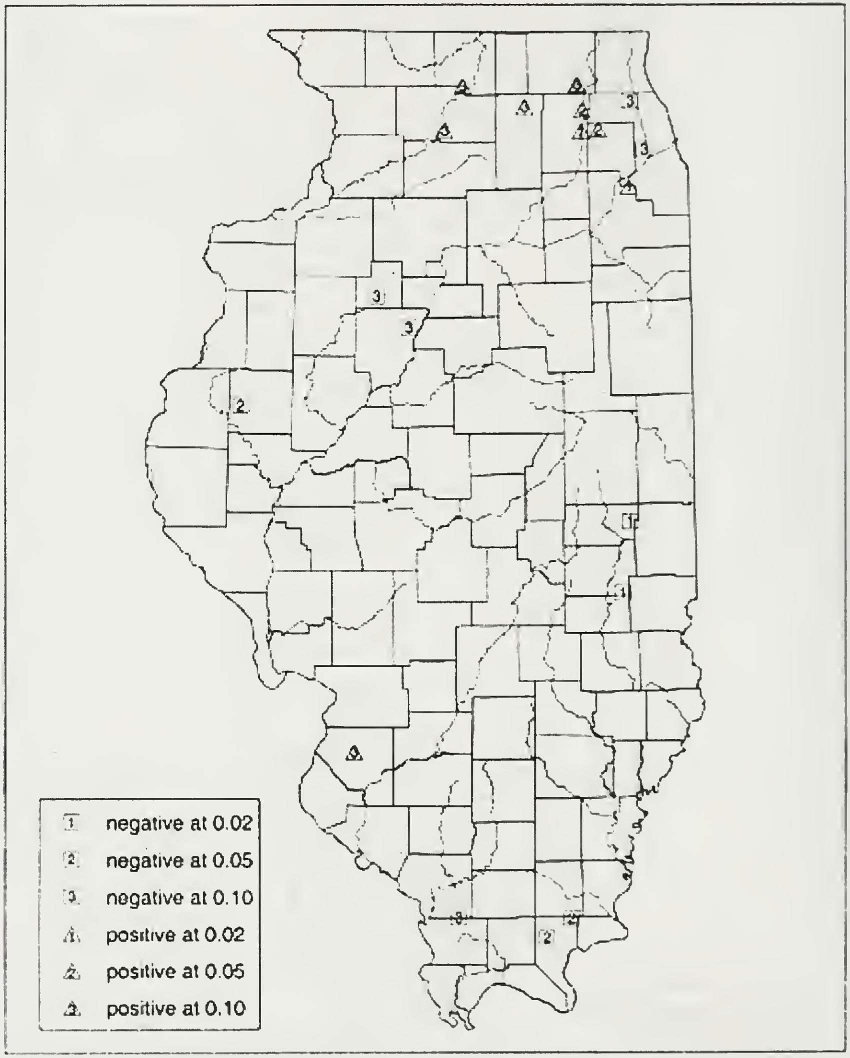


Figure 8

RiverWatch Sites Showing a Statistically Significant Trend in EPT Scores

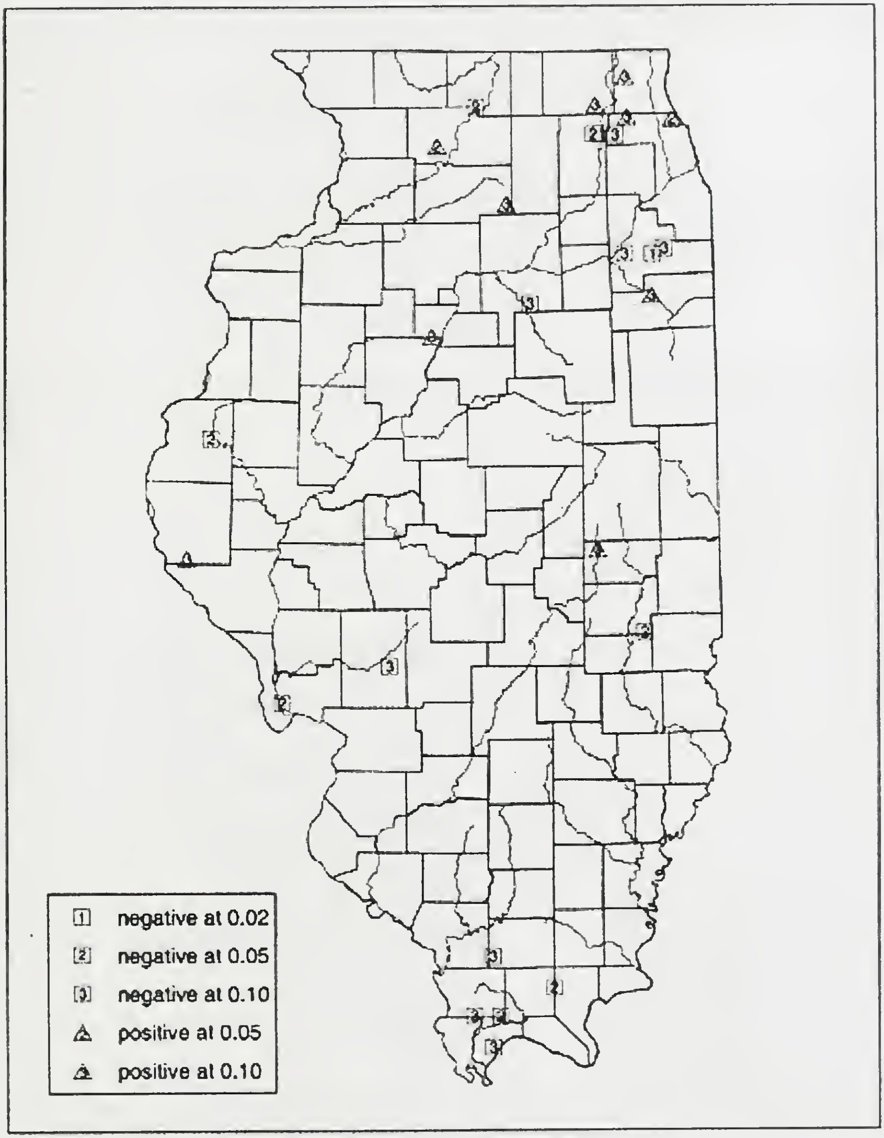


Figure 9

RiverWatch Sites Showing a Statistically Significant Trend in the Overall Biological Score

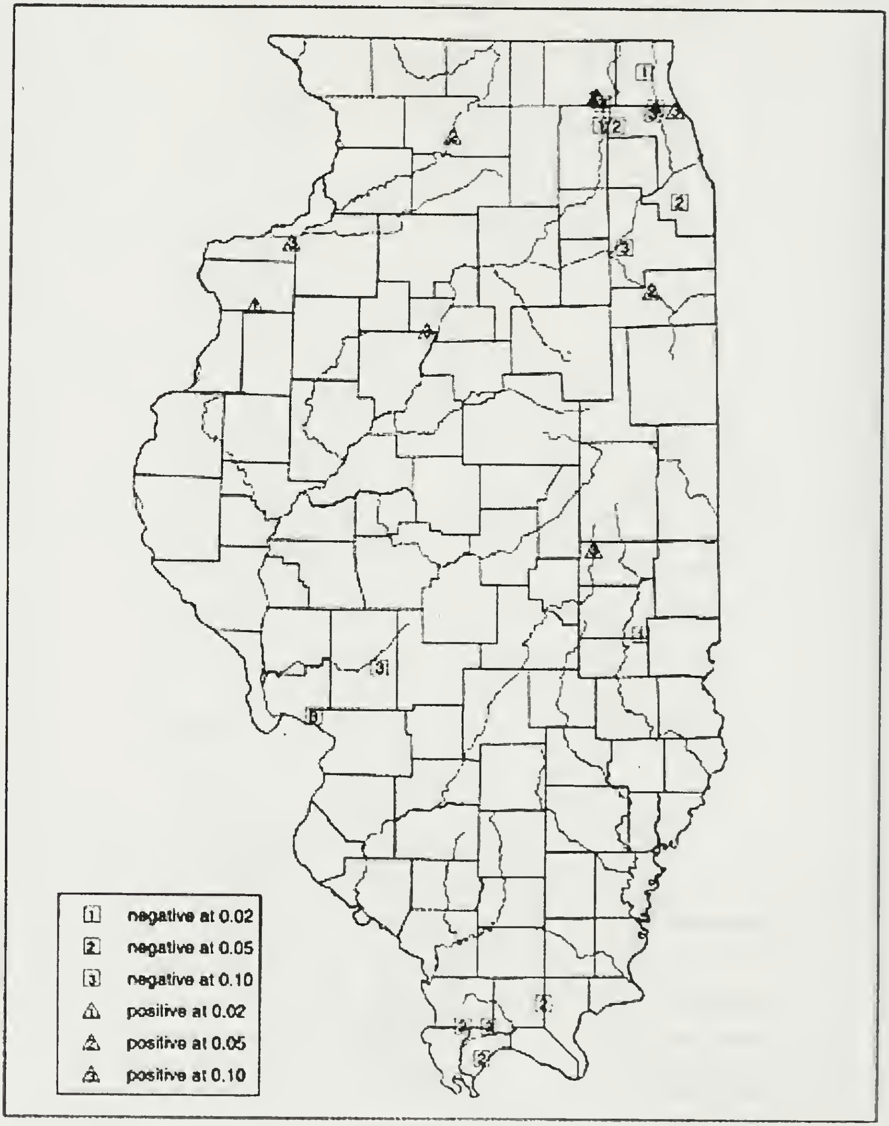


Figure 10

Table 2. Overall Trends by Natural Division

Natural Division[4]	# sites with 5-8 years of data	# and % of sites showing a trend	# improving	# declining
Driftless/Rock River	10	4 (40%)	3	1
Grand Prairie	30	10 (33%)	4	6
Northeastern Morainal	55	21 (38%)	11	10
Ozark/Shawnee Hills	23	9 (39%)	0	9
Southern Till Plain	14	2 (14%)	1	1
Western Forest-Prairie	33	8 (24%)	3	5
Total	165	54 (33%)	22	32

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[4] The Sand Area and Lower Mississippi Bottomlands have few sites that are monitored, and none which had five to eight years of data.

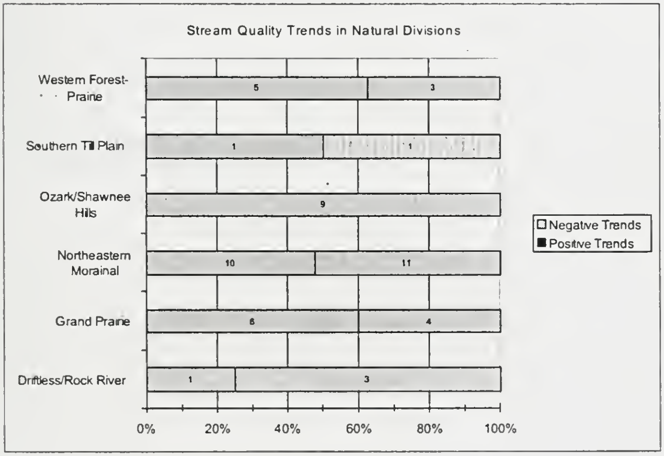

Figure 11. Biological trends broken out by natural division. The numbers on the bars indicate the number of sites.

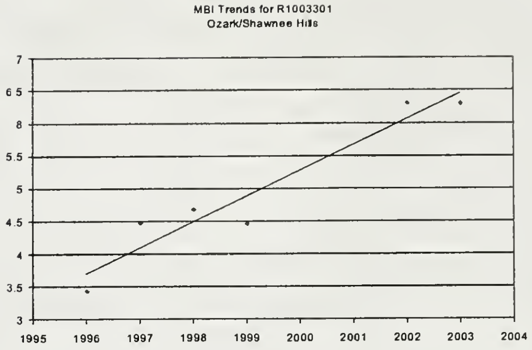

Figure 12. MBI trend for a site in the Ozark/Shawnee Hills (significant at 0.05).

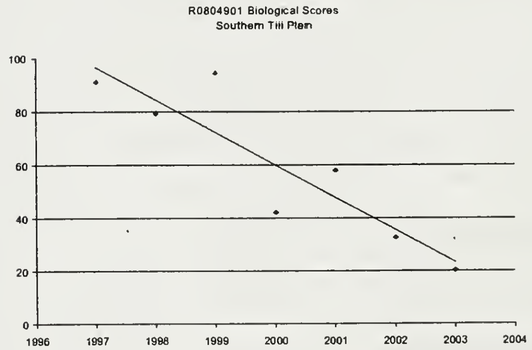

Figure 13. Biological score trend for a site in the Southern Till Plain (significant at 0.01).

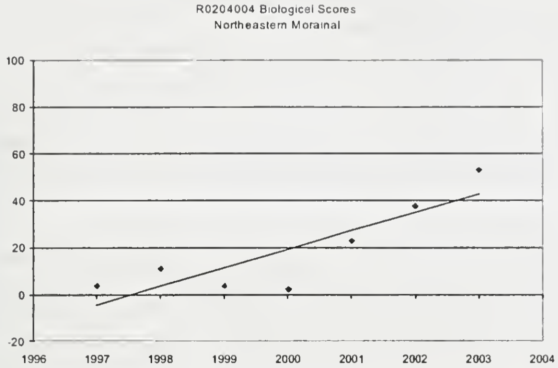

Figure 14. Biological score trend for a site in the Northeastern Morainal (significant at 0.05).

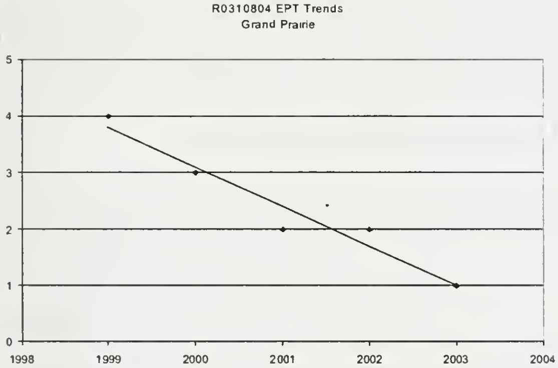

Figure 15. EPT trend for a site in the Grand Prairie (significant at 0.01).

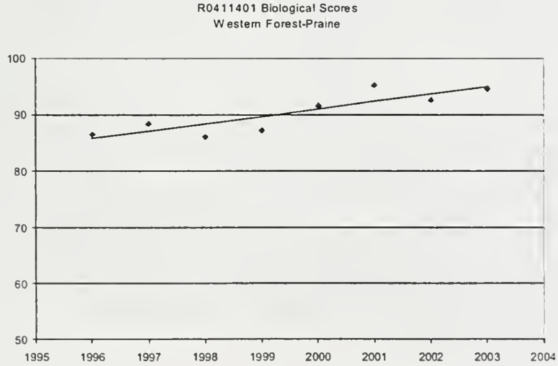

Figure 16. Biological score trend for a site in the Western Forest-Prairie (significant at 0.01).

One Hundred Fifty Years of Change in Illinois Streams, Stoneflies as a Case History

R. Edward DeWalt

Introduction

The prairies of Illinois were not always high and dry with tidy fields and arrow-straight ditches. The landform during presettlement times was covered in native tallgrass prairie, a community promoted by fire. Many areas were once poorly drained with extensive wet prairies and marshes. Along larger streams, firebreaks occurred, permitting a wooded riparian zone to form. This landform is a direct result of being recently glaciated, which scraped and filled the landscape as recently as 10,000 to 15,000 years ago.

Farmers improved drainage by straightened existing streams, shortening their length, and by adding streams, through ditching, where none had previously existed. They also tiled their fields, lowering the water table. Farmers also organized into cooperative, local drainage districts, and with state law behind them levied taxes on landowners to pay for drainage improvement. The advent of powerful machinery sped ditching and tiling of the land, so that by the middle of the 20th century about 27% of stream miles were channelized (Mattingly *et al.* 1993).

While these practices have allowed for a vast agricultural economy, they have wrought some negative consequences. Because of field tiling, stream channels now fill rapidly after rains and carry this flow downstream causing erosion, flooding, and scouring of the streambed. The lowering of the water table contributes to low flows and algae-choked channels by late summer. The removal of trees from larger streams and the reduced groundwater flow in summer, while not changing the average stream temperature, has caused greater fluctuations (higher highs and lower lows) (Wiley *et al.* 1990). It is surmised that a combination of these hydraulic, hydrologic, and temperature effects is partly responsible for the extinctions, local extirpations, and range reductions of aquatic fauna as diverse as mussels, fishes, and aquatic insects that have occurred throughout Illinois (Burr 1991, Cummings and Mayer 1992, DeWalt *et al.* 2002, Favret and DeWalt 2002).

The Critical Trends Assessment Program (see http://ctap.inhs.uiuc.edu for details of this IDNR program) has been sampling randomly chosen stream segments in Illinois streams since 1997. This design ensures that streams are sampled in proportion to the size and quality in which they occur statewide. The major objective of this sampling is to determine the overall condition of streams in space and time. CTAP sampling has found that many streams are heavily impacted in Illinois and that channelization of streams is a major factor in this degradation (DeWalt 2002b). A side benefit is that populations of three orders of sensitive aquatic insects are monitored. These orders are the Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies). Together, they are referred to as EPT taxa.

EPT taxa richness is an efficient index of condition (Lenat and Penrose 1996) and the study of EPT has a long history at the Illinois Natural History Survey (INHS). INHS scientists have published standard references for the identification of regional stonefly (Frison 1929 and 1935), caddisfly (Ross 1944), and mayfly (Burks 1953) species. More importantly, these scientists deposited all the specimens related to their publications in the INHS insect collection (now approaching seven million specimens). These specimens can be re-evaluated as the taxonomy of the group is revised—names of insects change all the time.

Recently, identification and location information for all EPT specimens in the collection were entered into a database, using nearly $200,000 of National Science Foundation and INHS matching funds. Illinois now has one of the largest web-based databases for insect specimens in North America with approximately 600,000 EPT and other insect and related specimens on-line (http://ctap.inhs.uiuc.edu/insect/search_inhs.asp). Although rates vary among groups of insects, approximately 25% of specimens originated from Illinois. Specimens date back to 1860 and, for some well studied groups, can provide a basis for comparison with more contemporary specimens deposited by CTAP and other projects. This report discusses the changes in a subset of the EPT fauna, the stoneflies (Plecoptera), which have been investigated in recent years by the author and his colleagues (DeWalt 2002 (a and b), Webb 2002, Favret and DeWalt 2002, Harris and Webb 1995, DeWalt *et al.* 1998, 1999, 2001, 2002). Mayfly and caddisfly comparisons cannot be made at this time since the historical fauna have not been adequately updated. The specific questions asked using this data set were: 1) Have the number of Illinois stonefly species changed over time and how does the change compare to other aquatic taxa? 2) Are changes specific to certain groups of stoneflies? 3) Is there a time frame corresponding to the greatest change? 4) Are changes more pronounced in certain regions? and 5) Do changes correlate to life cycle strategies of species affected?

Methods

CTAP stream assessment methodology is detailed at http://ctap.inhs.uiuc.edu and in DeWalt (2002c); however, a brief description is warranted. Sampling of streams began in 1997, with up to 30 sites visited each year through 2001, resulting in 149 sites. Beginning in 2002, sites were re-sampled. All specimens resulting from CTAP sampling have been entered into the INHS insect collection database. This database was mined for all Illinois stonefly records. All unique locations were geo-referenced, categorized by pre- and post-1950, and determined into which Illinois Natural Division (Fig. 1) they fell. Additionally, the majority of species were categorized by known or reasonably inferred life cycle type (Stewart and Stark 2002). Three general types occur in Illinois: univoltine-fast, univoltine-slow, and semivoltine-slow. “Fast” cycles involve an egg or nymphal (immature) diapause, limiting nymphal exposure to stream conditions to winter or spring months. In slow cycles, eggs hatch after a short developmental period, exposing the nymphs for about 11 months (univoltine) or two or more years (semivoltine). The numbers of species were graphically compared for time frame, natural division, and life cycle type.

Fig. 1. Natural Divisions of Illinois.

Results

Approximately 5,000 unique records (meaning a vial with one or more specimens or a pin with a single specimen) were mined from the database. Pre-1950 records totaled 2,573, while 2,422 records are dated Post-1950. The locations of these records are documented in Fig. 2, which also demonstrates that many more point locations were necessary to achieve a comparable number of stonefly records, suggesting that stoneflies are much harder to find Post-1950.

• *Have the number of Illinois stonefly species changed over time and how does the change compare to other aquatic taxa?* Illinois has reported 77 stonefly species since records began in 1860 (Frison 1935; Harris and Webb 1995, DeWalt 2002a, DeWalt *et al.* 2002, DeWalt *et al.* 2001, DeWalt *et al.* 1998). Several of these taxa have been reported only recently. This is due to improvements in the taxonomy of certain groups (species in the genus *Perlesta* and *Neoperla*), discovery of species new to science (DeWalt *et al.* 1998 and DeWalt *et al.* 2002), and discovery of regional taxa in peripheral counties. Forty-two species belonged to just two families, the Perlidae and the Perlodidae (Fig 2.). These families contain relatively large, predatory species, with the full range of life cycle types.

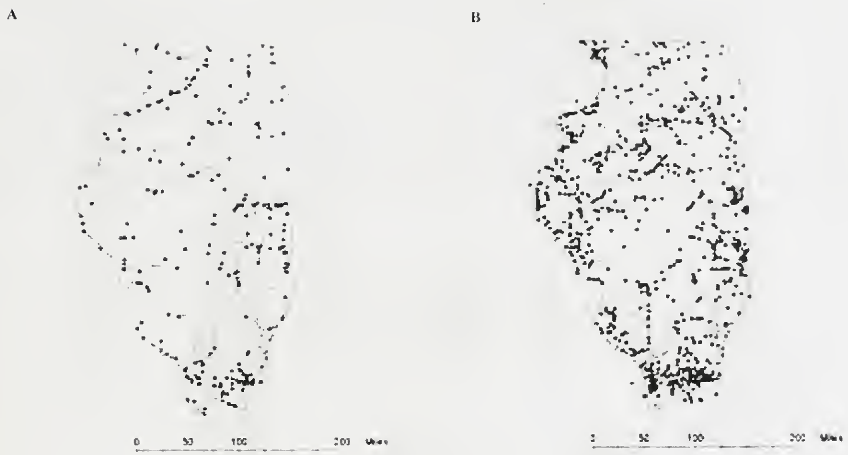

Fig. 2. Unique stonefly collecting locations from 1860 through present.
A. Pre-1950 locations, B. Post-1950 locations.

It is clear that Illinois has lost stonefly species through extirpation (18 species) and extinction (2 species). Extinctions include *Alloperla roberti* Surdick 1981 (Chloroperlidae), the Illinois Sallfly, and *Isoperla conspicua* Frison 1935 (Perlodidae), the Rare Stripetail. Extirpations are largely concentrated in the perlids and perlodids, contributing 15 of the 18 species (Fig. 3).

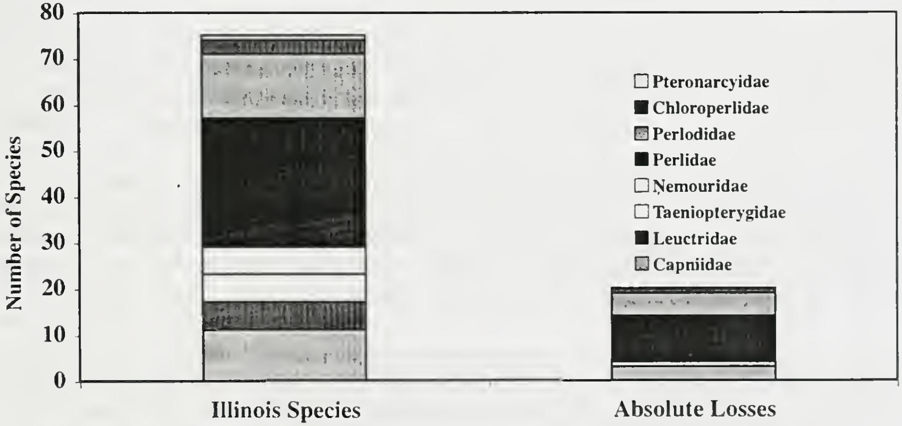

Fig. 3. Number of stonefly species known to occur and extirpations + extinctions in Illinois by family.

How does this compare with other aquatic groups? Unionid mussels, the large "clams" which historically formed huge shoals in the streams and large rivers of Illinois, also currently number 77 species (Cummings and Mayer 1992). They have long been viewed as one of the most imperiled freshwater groups in North America (Stein *et al.* 2000). Fishes are much more diverse in Illinois with 187 species (Smith 2002). Burr (1991) discusses imperilment of fishes in Illinois and Stein *et al.* (2000) summarize imperilment continent-wide. Figure 4 demonstrates that stonefly imperilment is on the same order as for mussels, and much greater than that for fishes. While stonefly imperilment is not a totally new topic, the magnitude of the problem has not been adequately demonstrated until now.

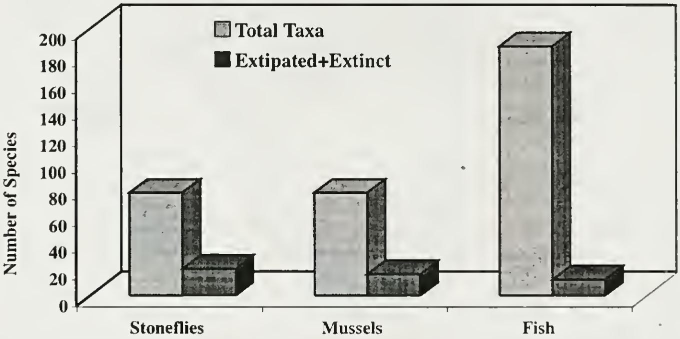

Fig. 4. Comparison of extirpations and extinctions across stoneflies, mussels, and fishes in Illinois.

- *Are changes specific to certain groups of stoneflies?* As noted previously, perlid and perlodid species were extirpated in the greatest numbers, with the perlids having lost 10 of the 28 species (36%) that ever occurred in Illinois. A closer examination of these losses demonstrates that certain genera within the perlids were affected differently (Fig. 5). It appears that Illinois has lost two genera completely, the *Attaneuria* and *Paragnetina*, while a third, *Agnetina*, has not been collected since 1976. The genus *Acroneuria* was once widespread and diverse throughout the state, with six species, but now three species can routinely be collected from a few locations in the state, with the distinct possibility that two species have been extirpated. *Neoperla*, historically known to have seven species, has lost four species, with the remaining three being relegated to southern Illinois and the periphery of the state in large rivers. Conversely, the genera *Perlesta* and *Perlinella* are still widespread, with the former being the most abundant, and often the only, stonefly collected during summer months.

- *Is there a time frame corresponding to the greatest change?* - This is a tough question to answer adequately because stoneflies were not collected equally throughout the period in question (Fig. 6a). However, it can be partially answered by examining relative changes in the frequency of collections of perlid genera throughout the 20th Century. Favret and DeWalt (2002) did this and demonstrated that the proportions of collections of several genera declined during the 1940s and

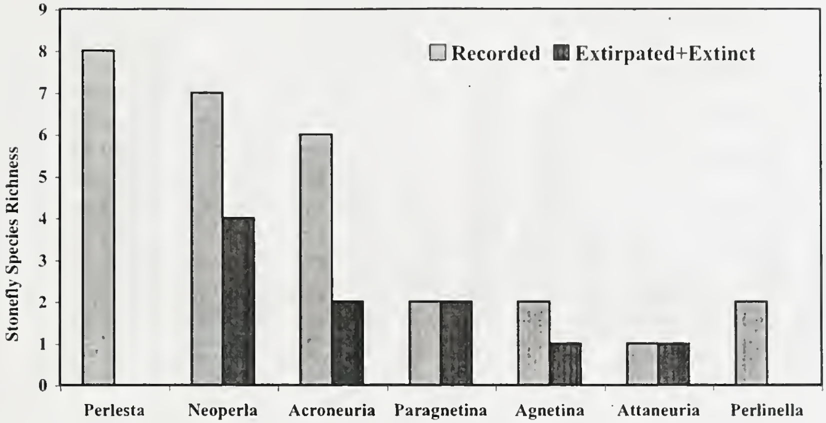

Fig. 5. Comparison of recorded versus extirpated + extinct perlid species by genus in Illinois.

1950s (Fig. 6b). All genera were a component of the perlid assemblage early in the century, but *Acroneuria* were by far the most commonly collected. As decades progressed, several genera were no longer routinely collected, and one, *Perlesta* became very abundant, even "weedy". The latter is now the most commonly collected stonefly genus in terms of the number of sites visited and in abundance at those sites during summer. A significant, negative correlation exists between the percentage of records of *Perlesta* and *Acroneuria*, indicative of a wholesale change in perlid communities in Illinois streams.

•*Are changes more pronounced in certain regions?*-Historically, the most diverse natural divisions in Illinois were the Grand Prairie, Shawnee Hills, and the Wabash Border, all with 30 or more species (Fig. 7). Other diverse divisions included the Upper Mississippi and Illinois River Bottomland and the Coastal Plain, both large river habitats. Losses varied greatly across natural divisions with three having one or zero losses, eight having lost more than one species, and three having added species.

The Shawnee Hills, Wabash Border, and Middle Mississippi Border have maintained a diverse fauna. The Shawnee Hills, due to its topography, has retained nearly its full complement of species. The Wabash Border has maintained its species richness by the recent addition of small, unnoticed species, at the expense of several large perlids. The Middle Mississippi Border contains bluff streams influenced by groundwater. These streams, while not extremely diverse, maintain a nearly natural fauna.

The Grand Prairie, historically a hyperdiverse area, has endured the greatest losses. Collections data confirm that the heart of the Grand Prairie supported sensitive, large perlids. Unfortunately, these no longer inhabit the Grand Prairie. Large rivers of the Upper Mississippi and Illinois River Bottomland and the Coastal Plain have been similarly affected. The Mississippi, Illinois,

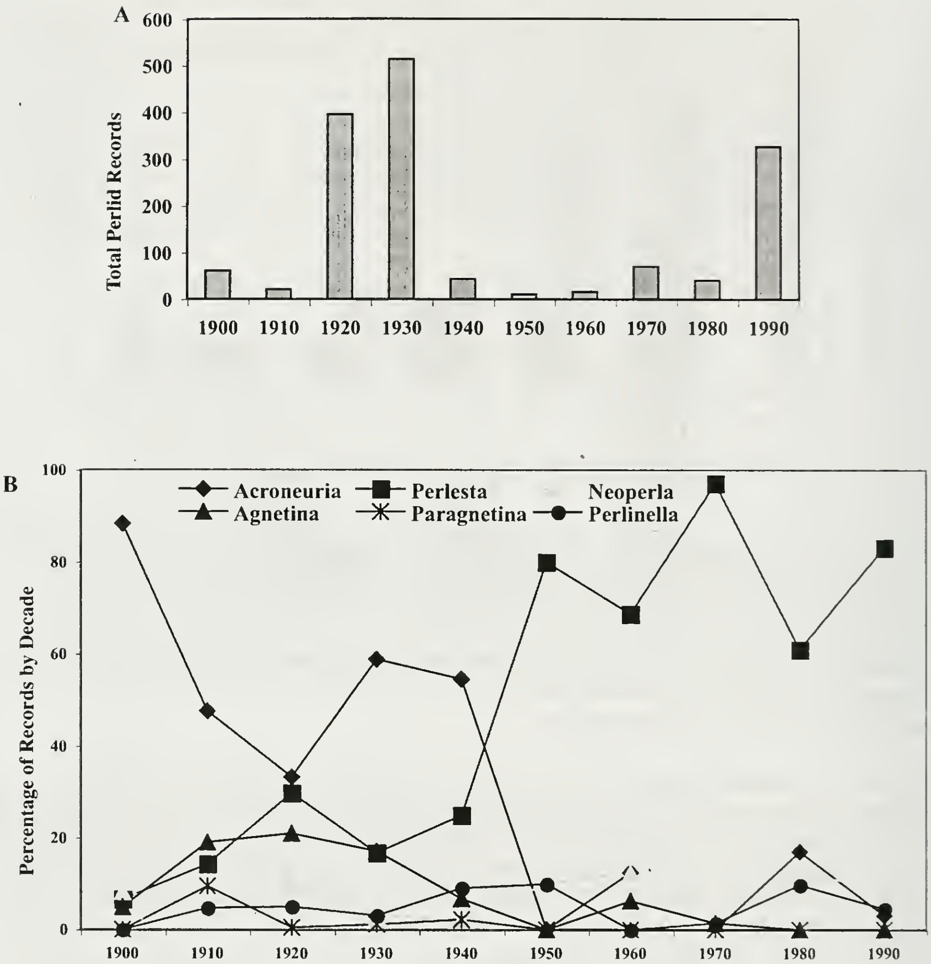

Fig. 6. A. Total perlid stonefly records by decade in the 20th Century; B. Percentage of perlid records in a decade corresponding to one of six genera.

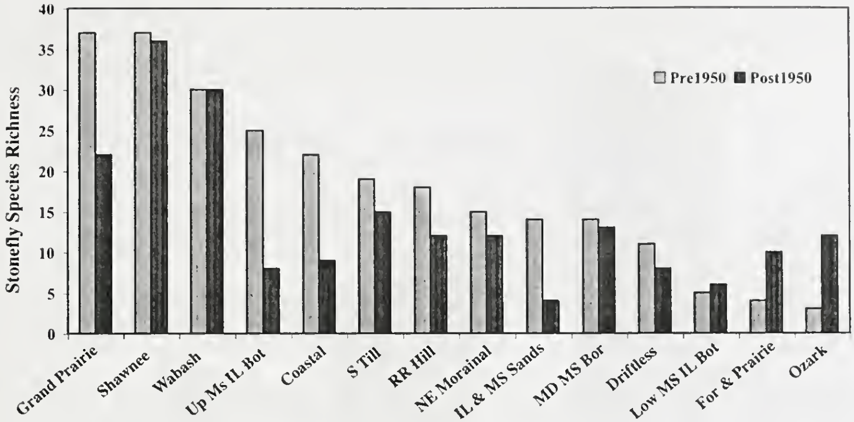

Fig. 7. Stonefly species richness by Illinois Natural Division and by Pre- and Post-1950.

and Ohio rivers supported great numbers of large perlid and perlodid stonefly species, but only a few of the more tolerant species now remain.

Those divisions that have added species—the Lower Mississippi and Illinois Bottomland, the Western Forest and Prairie, and the Ozark Hills—had low Pre-1950 species richness for two reasons. First, the confluence of the Illinois and Mississippi rivers has been degraded by domestic waste (the Chicago Sanitary and Ship Canal was opened in 1900) and by dredging and levee construction relating to navigation and flood control for a very long time. Secondly, these areas are great distances from Champaign, where stonefly researchers have been stationed, leading to less collecting effort. The increase in species richness in these areas after 1950 represents an improvement in conditions on the large rivers and more effort overall.

Do changes correlate to life cycle strategies of species affected?-Historically, the univoltine-fast cycle has been a mainstay for Illinois stoneflies, with 55.7% of the known species (Fig. 8). Slow cycles made up the remaining 44.4%. After 150 years of change, the percentage of fast cycles has increased to 66%, while the slow cycles have decreased to 34%. Most alarming of all is the loss of long-lived, semivoltine species that contributed 13% of species prior to 1950, but only 6% of species afterward.

Discussion

The distribution of losses in stonefly fauna have important parallels with regional trends in stream quality seen by the CTAP stream monitoring program (DeWalt 2002b). The natural divisions with the highest remaining stonefly richness are the Shawnee Hills and the Wabash Border, and this is where several high quality sites were found by CTAP. Likewise, the Grand Prairie experienced losses of many stoneflies and had many CTAP sites that scored poorly.

Fig. 8. Comparison of the distribution of stonefly life cycle type, Pre- and Post-1950.

Roth *et al.* (1996) demonstrated that local scale habitat quality was not as predictive of the health of the biological community as were broad, landscape-level scales. Large tracks of contiguous habitat are needed to protect stream biotic integrity. Wang *et al.* (1997) found support for this in Wisconsin, where agricultural land use approaching 50% and/or urbanized land cover at >10-15% amounted to the sure, if albeit slow, death of natural stream communities. The only places in Illinois where large blocks of high quality habitat exists is in the Shawnee Hills, the narrow bluffs of some of the large rivers, and the ravines of the Wabash Border.

Stein *et al.* (2002) ranked stoneflies as being one of the three most imperiled freshwater groups in the United States. This insect order was listed as having 43% of its more than 600 species being classified as "Vulnerable", "Imperiled", "Critically Imperiled", or "Presumed or Possibly Extinct". In Illinois, stoneflies have the highest known rate of extinction + extirpation of any aquatic animal. At 26%, it is higher than that for either mussels (20.8%, Cummings 1991) or fishes (6.4%, Burr 1991).

It would not have been possible to accumulate this information and analyze losses were it not for the foresight of the INHS in setting as their mission the documentation of the state's biotic resources. At a natural history institution, documentation means deposition of specimens as voucher, or proof, of occurrence and identification of a specimen in question. The use of natural history collections is the only reliable way to assess changes in the flora and fauna of a region (Suarez and Tsutsui 2004). Vouchered specimens can be re-evaluated using state-of-the-art taxonomy. DeWalt *et al.* (2001) and DeWalt *et al.* (2002) re-evaluated specimens from INHS collections and found that two genera of perlid stoneflies, thought to contain only three species in Illinois, actually contained 15, and that four had been extirpated. It was only through the vouchering of specimens throughout the 20th century that this could be determined.

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Occurrence and Causes of Low Abundance for RiverWatch Samples 1996-2003

Alice Brandon

Introduction

RiverWatch (RW) is the volunteer stream-monitoring component of the Illinois EcoWatch Network. Annually, over 500 volunteers survey more than 250 stream sites by collecting benthic macroinvertebrates to gauge stream health. The RW program adopted a multi-habitat, composite sampling approach to collect fairly large but manageable biological samples (approximately 100 organisms) (IDNR 2004). A similar sampling scheme demonstrated this approach to produce representative estimates of dominant species occurring in a given stream reach (Diamond *et al.* 1996). Technical advisors also think volunteers can easily collect this number of organisms in most streams (DeWalt 1999). This sampling approach relies upon volunteers to collect 100 organisms each time they monitor.

In practice, RW volunteers frequently collect less than the desired 100 organisms. For example, in 2003 48% of submitted samples contained less than 100 organisms. This is of concern because smaller samples may fail to collect all the dominant species present and, in turn, inaccurately gauge a site's quality. This report examines possible causes and trends for low sample abundance. By better understanding the reasons for low sample abundance the program hopes to reduce its occurrence.

Methods

The RW data from 1996 to 2003 were analyzed using stepwise regression and spearman rank correlation to better understand low abundance. In the stepwise analysis we used sample abundance as the dependent variable and examined various independent variables to measure their effect on abundance. See Table 1 for a complete list of factors and their hypothesized effect.

Spearman rank correlations were also run to examine relationships among all the independent variables. This was a necessary procedure because correlations that are high (above 50%) may cause regression estimates to change depending upon what independent variables were used (Cody and Smith 1997). Analysis revealed year and visit number to be significantly and highly correlated ($R = 64\%$). However, dropping either of them from the model had no significant effect on the Stepwise regression model results. Therefore, both variables were kept in the initial regression. None of the other variables were highly corrected with one another. Discrete independent variables (such as habitat) significantly related to abundance were examined further using Tukey post-hoc comparison tests to compare differences among means ($p < 0.05$).

Table 1. Independent variables analyzed to measure their possible relationship to sample abundance.

Variable:	**Hypothesized impact on abundance?**
Stormy Weather: disturbing / washing organisms downstream.	Decreases
Monitoring Time: spending less than 2 hours monitoring.	Decreases
Sampling Habitats: sampling different habitats for macroinvertebrates: riffle, leaf pack, snag area, undercut bank, and sediment.	Variable
Year Monitored: comparing monitoring data different years (1996-2003).	Increases over time.
Visit Number: number of times an individual site was monitored.	Increases over time
Region: separating sites into the five IDNR management regions from Northern to Southern Illinois (see Figure 2).	Variable
Stream Discharge: comparing different discharge levels at streams.	Variable
Macroinvertebrate Taxa with Facultative/Tolerant Stress Tolerance: organisms that increase their abundance under organic pollution/moderate disturbance to their environment (Voshell 2002). Taxa examined: sowbug, scud, hydropsychid caddisfly, midge, riffle beetle, and black fly.	If present, increases
High Taxa Dominance: sites where 3 or fewer taxa represent more than 80% of the total abundance.	Increases

Results

The strongest predictor of high sample abundance is the presence of the facultative/stress tolerant macroinvertebrates (FST) (sowbug, scud, hydropsychid caddisfly, riffle beetle, midge and black fly). This variable accounts for 67% of the variance in abundance levels at the sites ($p < 0.0001$). The next best predictors of abundance are region where site is located (see Figure 1), habitat type sampled, and taxa dominance, which accounts for an additional 7% of the variance. Therefore, the best four-way predictor explaining abundance is FST macroinvertebrates, region, habitat sampled and taxa dominance ($R^2 = 0.74$, $p < 0.0001$).

Samples containing none of the FST taxa have significantly lower abundances compared to samples with these taxa ($p < 0.05$, see Figure 2a). Additionally, the more FST taxa collected the higher the sample abundance. For example, mean abundance for samples with one FST taxon present is 56 organisms while mean abundance for sites collecting 6 of these taxa increases to 123 organisms ($p <0.05$).

Region 5 has significantly lower abundances when compared to all other regions of the state ($p < 0.05$, see Figure 2b). Mean abundance for region 5 is 36% lower than abundances collected in Region 2 (64.9 versus 100.8, respectively). All other regions also have significantly different abundances from each other with the exception of Regions 3 and 4, which have very similar abundances that are not statistically different from one another (87.3 versus 90.1, respectively).

Sites where volunteers sample riffles (with an additional habitat) have significantly higher abundances and a higher number of FST macroinvertebrates compared to samples where leaf packs, snags and undercut banks are sampled with an additional habitat ($p < 0.05$, see Figure 2c).

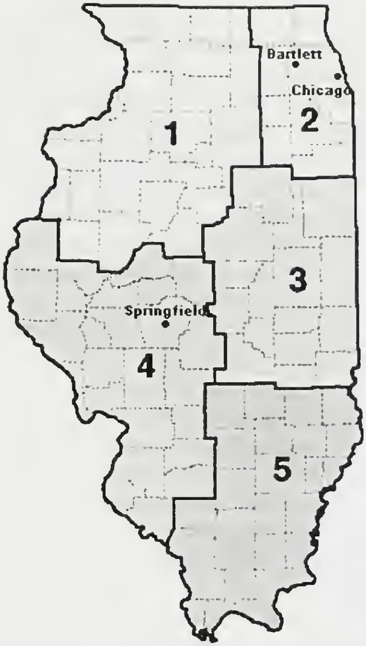

Figure 1. Map of Illinois split into the Illinois Department of Natural Resources 5 management regions.

The highest abundances are collected when volunteers sampled a combination of either riffle/leaf pack or riffle/snag. The lowest abundances occur when volunteer sample either snag/undercut bank or leaf pack/sediment. Sediment habitat samples have abundances that are only statistically different from undercut bank samples but not the other habitats (see Figure 2c).

Sites experiencing a high percentage of taxa dominance also collect more macroinvertebrates and have increased mean abundances ($p < 0.0001$). The other variables including year the sample is collected, number of times the site is monitored, and time spent monitoring explain less than 1% of the variance combined. Therefore, these variables were dropped from the model and further analysis.

Discussion

Our results suggest that collecting adequate abundances is mostly related to the population dynamics of different macroinvertebrate taxa rather than to volunteer ability. The strongest predictor for high sample abundance is the presence of facultative/stress tolerant (FST) taxa.

Additionally, as the cumulative number of FST taxa present increased so did abundance (see Figure 2a). In contrast, samples with high EPT taxa richness (with the exception of hydropsychid caddisfly) are negatively correlated with high abundance ($R = -0.50$, $p < 0.001$). These findings are explained by the life history strategies for different RW indicator taxa. Biologists note that organisms are spatially distributed in random, clumped or regular patterns (Sheldon 1984). Many EPT taxa naturally occur in low densities even under ideal habitat conditions (Voshell 2002). Highly abundant macroinvertebrate taxa collected by RW volunteers are often described as FST. These macroinvertebrates occur in a range of environments from high to low quality but are very abundant in streams experiencing moderate pollution or disturbance from organic wastes (Voshell 2002). This describes the condition of the majority of streams monitored by volunteers. Extremely high quality reference streams are not monitored often because they now exist in very few places while highly degraded streams are avoided due to safety concerns.

The findings also support region, type of habitat sampled and taxa dominance (though less important than FST) as predictors of abundance levels. Regions 1 and 2 have abundances at adequate levels while sites in the other regions, especially region 5, have low sample abundances (see Figure 1b). These results support anecdotal evidence from program staff. For example, during volunteer review sessions in northern Illinois (region 2) volunteers have the opposite problem. They often collect too many organisms to comfortably sort in a few hours even though the streams tend to be low in quality (A. Brandon, personal observation).

Statewide results also show that sampling riffles (and another additional habitat) is the most commonly sampled habitat (sampled 75% of the time). Riffles also support the highest abundances of macroinvertebrates with the exception of sediment, which is very rarely sampled (<1% of the time). Another important difference in abundance from habitat to habitat was the low abundances for snag areas and especially undercut bank samples (with an additional habitat). The average abundance for undercut bank samples is 58 organisms. However, if the habitat data

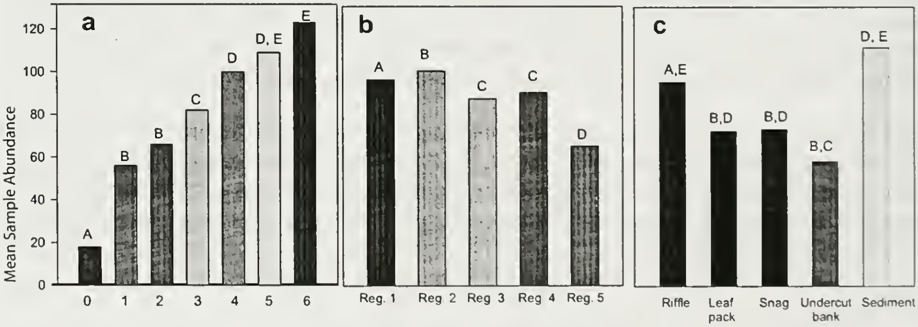

Figure2. Mean abundance of samples separated by a) number of facultative macroinvertebrate taxa, b) region, and c) type of habitat sampled. Capital letters indicate significant differences at the 0.05 level using Tukey post-hoc tests.

is separated into regions additional trends appear. The most obvious differences are between region 5 and region 2 data. Volunteers at only 12% of sites reporting low abundance in region 2 sample from riffles (with another habitat) but this percentage increases to 68% for the same habitat in region 5. Why is this occurring, especially since region 5 consistently scores very well on habitat quality ratings compared to other areas of the state (unpublished data)? Researchers often cite habitat quality as a primary determinant of aquatic community abundance. Habitat data explains why certain organisms are present or absent from a site (Barbour *et al.* 1999). A possibility is that sites in region 5 are monitored too late in the season since it warms up in southern Illinois much earlier compared to northern Illinois. Many aquatic macroinvertebrates will finish their life cycle and emerge as adults by mid May in southern Illinois. Volunteers are most likely to monitor in June (75% of all sites). Further examination indicated virtually identical mean abundances from May or June monitoring dates at the statewide level (91 versus 90, respectively). However, for region 5 mean abundance for June sampling is lower than for sites monitored in May (63 versus 75, respectively). We also suggest that some of the variables (such as stream discharge) that were non-significant may help explain low abundance for this region. This is because there is much less data for region 5 compared to region 2 (334 samples versus 704, respectively). Unequal sample sizes can cause problems in analysis by pulling means in their own direction. This means that because low abundance is not a problem in regions 1 and 2 (which represents the majority of the data) the regression may not be detecting actual differences at regional levels. While not statistically significant, region 5 did have much lower stream discharge and width means compared to the state averages (8 versus 33 ft^3/sec and 15 ft versus 20 ft, respectively). An on-going concern for RW staff is the possibility that some sites established early in the program for region 5 are intermittent streams that do not fit the monitoring criteria. These streams may not contain enough water to support adequate abundances during monitoring.

The presence or absence of FST taxa appears to be the key to understanding low abundance. Our results do not support volunteer ability or training problems as contributing significantly to low abundances. Instead, FST abundance mediated by an interaction between regional trends and the conditions occurring at individual streams sites including habitat availability, taxa dominance, and possibly physical stream characteristics such as discharge are causing low sample abundance.

Recommendations

Fortunately, low abundance appears unrelated to training quality or volunteer skill. This is good because it lends more credibility to volunteer data and volunteers' ability to follow the procedures. Since low abundance is not a statewide problem nor caused by a single factor, no one solution will work for every site. However, below are some specific suggestions for decreasing the occurrence of low abundance. First, efforts should be concentrated in Southern Illinois, especially region 5. Since the number of sites being actively monitored in region 5 is fairly small, it should be feasible to identify sites that need investigation and contact those volunteers for additional information about their site. Some of these sites likely need to be taken off the active list for monitoring. Second, RW should encourage volunteers in Southern Illinois (especially those streams with low discharge) to sample their sites in May. Staff may even want to consider changing the season of monitoring for the southern part of the state if this proves to solve the problem.

Third, we know that volunteers who sample from snag and undercut banks have lower abundances. It may be useful to revisit the issue of tweaking the sampling protocols for these habitats. Perhaps something as simple as increasing the number of sweeps of the net under undercut banks will improve abundances for this habitat type.

Lastly, we can solicit feedback from this report from volunteers. Since volunteers are intimately familiar with their stream they may have useful insights as to the causes for low abundance. Additionally, as volunteers become more aware of low abundance issues from reading this report or via reviews, training, etc. this should encourage volunteers to monitor earlier.

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