

HIERARCHICAL HABITAT SELECTION BY THE ACADIAN FLYCATCHER:  
IMPLICATIONS FOR CONSERVATION OF RIPARIAN FORESTS

A Thesis

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## ABSTRACT

Riparian forests represent critical habitat for forest birds throughout midwestern United States, yet they continue to be fragmented and degraded due to agricultural practices and urban development. Knowledge of the ecology and management of riparian communities comes primarily from studies conducted at small spatial scales. However, recent work demonstrates that the surrounding matrix can affect diversity, abundance and reproductive success of breeding birds within a forest patch. In this thesis, I used the Acadian Flycatcher (*Empidonax virescens*), a species of concern in the Midwest, to examine multi-scale impacts of land uses on riparian forests. Habitat selection by Acadian Flycatchers was investigated at three distinct scales: territory (1 ha), stand (3-5 ha), and landscape (314 ha) in 36 riparian forests in central Ohio. I addressed potential mechanisms behind landscape-scale patterns by examining understory arthropod biomass, vegetation structure, and breeding productivity across a spectrum of rural and urban land uses.

Acadian Flycatchers were surveyed three times per year during June 2001-2002 in mature deciduous riparian forests along an urbanization gradient. Landscape metrics (e.g., % urban cover and riparian forest width) were quantified within 1 km of each study site (e.g., % urban cover range = 0-53%; forest width range = 56-565 m) while vegetation characteristics and understory arthropod biomass were measured within each

riparian forest and Acadian Flycatcher territory. In addition, I target-banded 11 male Acadian Flycatchers to map territories and monitor productivity. Habitat selection at the territory scale was evaluated by comparing features measured within the territory with average stand values. Two sets of *a priori* candidate models were developed representing plausible factors affecting abundance and productivity of Acadian Flycatchers at both the stand and landscape scale. Candidate models were ranked based on Akaike's information criterion (AIC<sub>c</sub>).

Results suggest that the criteria used by Acadian Flycatchers to select habitat change with spatial scale, suggesting that this species selects habitat in a hierarchical manner. Percent urban cover in the landscape best explained abundance and productivity of Acadian Flycatchers. Both abundance and productivity of Acadian Flycatchers were negatively associated with percent urban land cover while numbers of predators increased with urban development. Acadian Flycatchers were 3 times more likely to be detected in more rural ( $\leq 1\%$  urban development) riparian forests compared to more urban ( $\geq 10\%$  urban development) riparian forests. Interestingly, as urban cover in the landscape increased, Acadian Flycatchers selected wider riparian forests than those selected in rural landscapes. Vegetation structure and understory arthropod biomass had little explanatory power at the stand scale but were central in territory selection. Vegetation density was lower and understory arthropod biomass was greater within the territory than throughout the stand.

In short, I believe that increased numbers of predators coupled with reduced breeding productivity and site fidelity in urban riparian forests may be an underlying mechanism affecting Acadian Flycatcher landscape patterns. Therefore, my work

suggests that planning at the landscape scale is one of the most important factors for Acadian Flycatchers in riparian forests in central Ohio. Specifically, riparian forests in rural areas are preferred by Acadian Flycatchers and should be safeguarded. I suggest that midwestern riparian forests in rural landscapes be  $\geq 100$  m wide while riparian forests in urban landscapes should be  $\geq 150$  m wide. Next, manipulating vegetation characteristics within the stand can further enhance habitat quality for Acadian Flycatchers. For instance, removal of invading exotic shrubs may create a relatively open understory to accommodate multiple territories for Acadian Flycatchers and other avian species. This study is the first to examine multi-scale habitat selections of a Neotropical migrant across rural and urban landscapes. Further research is needed to (1) determine if manipulations of selected vegetation features affect habitat selection and productivity, (2) investigate if predator control increases selection of urban riparian forests, and (3) determine if conspecifics influence habitat selection of Acadian Flycatchers.

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## FIELDS OF STUDY

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## CHAPTER 1

### INTRODUCTION

Current trends in wildlife ecology have favored multi-species approaches and have moved away from single-species studies (Beatley 1994, Martin 1994, Chase et al. 2000, Clark and Harvey 2002). However, single-species studies can often better identify mechanisms responsible for habitat selection and population demographics (Chase et al. 2000). Also, they may facilitate development of more effective conservation strategies because management plans based on single-species are more likely to include adaptive management provisions, are revised more frequently, and are implemented more swiftly than multi-species plans (Clark and Harvey 2002, Lundquist et al. 2002). In this thesis, I used the Acadian Flycatcher (*Empidonax virescens*), a species of concern in the American Midwest (Thompson et al. 1993), to examine multi-scale impacts of land uses on riparian forests.

Acadian Flycatchers have a relatively high-priority ranking for management and monitoring in the Midwest, ranking 15<sup>th</sup> of 110 Neotropical migrants (Thompson et al. 1993, Whitehead and Taylor 2002). Population trends reported by Breeding Bird Survey data indicate that Acadian Flycatchers have experienced significant declines between

1966 and 2000 in Ohio and the Till Plains region (trend = -2.57 birds/route,  $P = 0.041$ , trend = -4.23 birds/route,  $P = 0.109$ , respectively; Sauer et al. 2001). Acadian Flycatchers are associated with mature, mesic deciduous forests in eastern North America. Breeding pairs exhibit a preference for the interiors of extensive woodlands, particularly those larger than 40 ha (Peterjohn 2001, Whitehead and Taylor 2002). In this respect, Acadian Flycatchers are thought to be an indicator species of riparian forest quality.

Riparian forests preferred by Acadian Flycatchers also are considered essential for many other wildlife species because they improve or maintain water quality, support plant and animal productivity, and function as movement corridors (Lowrance et al. 1984, Naiman et al. 1993, Machtans et al. 1995, Knutson et al. 1996, Kinley and Newhouse 1997, Basnyat et al. 2000). Additionally, because the majority of remaining midwestern forests lie along waterways, riparian forests represent critical habitat for forest-dependent birds (Laub 1979, Groom and Grubb 2002). Yet, continued fragmentation and degradation of riparian forests by agricultural practices and urban development pose a severe threat to many species (Robinson 1996).

The linear nature of riparian forests makes them particularly vulnerable to influences from the surrounding landscape. In fact, recent work demonstrates that the surrounding matrix can affect diversity, abundance and reproductive success of breeding birds within a forest patch (Croonquist and Brooks 1993, Andr  n 1994, Friesen et al. 1995, Bayne and Hobson 1997, Rodewald and Yahner 2001a, b, Rodewald 2002). However, our knowledge of the ecology and management of riparian communities comes primarily from local habitat characteristic studies that failed to address landscape-scale

influences (Keller et al. 1993, Darveau et al. 1995, Hodges and Krementz 1996, Meiklejohn and Hughes 1999). In general, residential (Friesen et al. 1995, Rottenborn 1999) and agricultural (Andr n 1994, Bayne and Hobson 1997) land uses are associated with decreased species richness and abundance when compared to other land uses (e.g., silviculture). These effects are most pronounced when forest patches are surrounded by large amounts of disturbance (e.g., nonforest habitat, Flather and Sauer 1996, Donovan et al. 1997). Neotropical migratory birds have shown greater landscape-sensitivity than either short-distance migrants or resident species (Keller et al. 1993, Flather and Sauer 1996), and in particular, the Acadian Flycatcher has demonstrated landscape-sensitivity across its breeding range (Whitcomb et al. 1981, Smith and Schaefer 1992).

Effective management of riparian forests in the Midwest requires multi-scale studies to address both local and landscape-scale influences. Because Acadian Flycatchers are area-sensitive Neotropical migrants (Temple 1986, Robbins et al. 1989, Keller et al. 1993), and have specialized habitat requirements (Johnston 1970, Bisson et al. 2000, Whitehead and Taylor 2002), they are an ideal focal species for the study of local and landscape-scale influences on riparian forests.

## OBJECTIVES AND HYPOTHESES

The goal of my research was to examine multi-scale impacts of land uses on habitat selection by Acadian Flycatchers, and their implications for conservation of riparian forests. Habitat selection by Acadian Flycatchers was investigated in 36 riparian forests in central Ohio at three distinct scales: landscape (314 ha), stand (3-5 ha) and territory (1 ha). I addressed potential mechanisms that might explain landscape-scale patterns by examining understory arthropod biomass, vegetation structure, and

productivity of Acadian Flycatchers in riparian forests surrounded by both rural and urban land uses. With this knowledge, I suggested effective conservation and management recommendations to improve suitability of riparian forest habitat for Acadian Flycatchers and other Neotropical migrants in the Midwest.

Based on previous literature and field observations, I predicted that abundance of Acadian Flycatchers would be positively associated with wider forests (>100m), and that riparian forests in rural landscapes would be preferred over riparian forests in urban landscapes. Additionally, I hypothesized that differences in productivity of Acadian Flycatchers in forests surrounded by different land uses would best explain habitat selection at the landscape scale, whereas resource-based selection (i.e., prey abundance and vegetation characteristics) would occur at smaller spatial scales (i.e., stand or territory scale). With a better understanding of the habitat selection process and consequences for Acadian Flycatchers, natural resource managers may better manage riparian zones for viable forest songbird populations.

## THESIS FORMAT

This thesis is organized as one introductory chapter and one chapter (manuscripts) formatted for submission to a scientific journal. In Chapter 2, I considered multiple spatial scales and factors guiding habitat selection by Acadian Flycatchers in riparian forests in central Ohio. In particular, I addressed potential mechanisms behind landscape-scale patterns by examining 3 factors: (1) understory arthropod biomass, (2) vegetation structure, and (3) productivity of Acadian Flycatchers. I then used Acadian Flycatchers as a model species to address conservation and management of riparian



forests at both local and landscape scales. This chapter is formatted for publication in the journal *Ecology*.

## BACKGROUND

### LIFE HISTORY OF ACADIAN FLYCATCHERS

Acadian Flycatchers breed in deciduous bottomland forests, swamps, riparian thickets and wooded ravines of eastern North America. Acadian Flycatchers prefer a wide ( $\geq 100$  m) mature forest with a closed canopy (Bell and Whitmore 2000, Bisson et al. 2000, Whitehead and Taylor 2002) and open understory for foraging (Johnston 1970, Ehrlich et al. 1988), and aerial nest defense (Wilson and Cooper 1998a). In many parts of their range, Acadian Flycatchers are rarely found in isolated woodlots less than 40 ha in size, and hence they are usually considered an area-sensitive species (Temple 1986, Robbins et al. 1989, Dawson et al. 1993, Keller et al. 1993, Peterjohn 2001).

Acadian Flycatcher nests are loosely constructed with bark, twigs, weed stems, grass and cobwebs, and are placed hammock-like between spreading twigs on a horizontal tree limb. Female Acadian Flycatchers select the nest site and build the nest. Average clutch size is 3 eggs that are creamy white with brown spots. Females incubate eggs for 13-15 days and both adults feed altricial nestlings (immobile, downless, eyes closed, fed) for 13-15 days and fledglings for up to 3 weeks (Ehrlich et al. 1988, Whitehead and Taylor 2002). Occasionally, Acadian Flycatchers will attempt a second brood and fledglings from the first nest are fed solely by the male when the female begins incubating a second clutch.

Acadian Flycatchers are commonly parasitized by Brown-headed Cowbirds (*Molothrus ater*). For example, average brood parasitism rate in Arkansas (Wilson and

Cooper 1998b) and Michigan (Walkinshaw 1966) was 21%. Acadian Flycatchers are particularly vulnerable to parasitism because they rarely raise any of their own young if the nest is successfully parasitized (Whitehead and Taylor 2002). Nonetheless, Wilson and Cooper (1998b) found predation to be the primary cause of nest failure (75%) for Acadian Flycatchers, resulting in a nesting success of 10-25%.

Beal (1912) found the diet of Acadian Flycatchers included Hymenoptera (39.93%), Lepidoptera (18.87%), Coleoptera (13.76%), Diptera (8.15%), Hemiptera (6.03%), Arachnida (2.94%) and small amounts of plant material. Video recordings of adults feeding nestlings and fecal sac contents indicate the top 4 orders of insects fed to Acadian Flycatcher nestlings in northeastern Ohio are Diptera, Lepidoptera, Hymenoptera, and Arachnida (C. N. Willis, personal communication).

Acadian Flycatchers are classified as Type A territory holders (Hinde 1956). Type A territories are large breeding areas within which nesting, courtship, mating and most foraging usually occur (Nice 1941, Armstrong 1965). Males establish territories by moving about their territory and broadcasting their song (typically 'pee-tsup'). Average territory size ( $n=13$ ) for Acadian Flycatchers in Michigan studied by Mumford (1964) was 1.0 ha (range = 0.5-1.6 ha) while Walkinshaw (1966) reported an average of 1.2 ha ( $n=80$ ), but noted that territories became larger during drought years. Furthermore, Acadian Flycatchers exhibit strong site fidelity (e.g., 67% return rate documented by Walkinshaw 1966).

## HABITAT SELECTION

Habitat selection is a widely studied component of ecology that incorporates genetics, behavior, morphology, and evolution. In fact, a wide variety of taxa, including

insects (Morisita 1952, Rausher 1983, Istock and Weisburg 1987), fish (Bay et al. 2001, Munday 2001), mammals (Wecker 1963, 1964, Apps et al. 2001, McLoughlin et al. 2002), plants (Thery 2001), reptiles (Kiestler et al. 1975, Morrow et al. 2001, Compton et al. 2002), and birds (see Hildén 1965 and Cody 1985 for review) are known to select territories or sites with specific vegetation characteristics.

Habitat selection is probably best studied in birds. Habitat selection has been examined as a major driver of physiological, morphological and behavioral adaptations (Ulfstrand et al. 1981, Robinson and Holmes 1982, Thiollay and Clobert 1990), community structure and dynamics (Lack 1971, Cody 1974), speciation (Lack 1940, 1944, see Cody 1985), and species diversity (MacArthur and MacArthur 1961, Willson 1974). Habitat selection studies have repeatedly focused on the influence of local factors such as vegetation structure (Petit et al. 1988, Green et al. 1989, Jokimäki and Suhonen 1993, Germaine et al. 1998), prey abundance (Petit et al. 1988, Petit and Petit 1996), predation (Martin 1988a, b, Martin and Roper 1988, Lima 1993, Martin 1996), and site fidelity (Gavin and Bollinger 1988, Switzer 1997).

Most studies of habitat selection have traditionally focused on one particular spatial scale thought to be relevant to a focal species or group. However, studies at one scale may overlook important relationships by failing to explicitly consider multiple spatial scales. Recently, ecologists have recognized the importance of studying species or systems at multiple scales (Virkkala 1991, Rolstad et al. 2000, Illera 2001, Penteriani et al. 2001, McLoughlin et al. 2002). Habitat selection at multiple scales is often seen as a hierarchical process that has been classified into descending steps (Johnson 1980). For example, first-order selection refers to the selection of physical or geographical range of a

species. Within that range, second-order selection determines the home range of an individual, while third-order selection pertains to habitat components within the home range. Lastly, the actual procurement of resources (i.e., food or nest sites) from those available at a site is fourth-order selection (Johnson 1980). Thus, assessment of fitness consequences associated with features selected by Acadian Flycatchers is critical to understanding the adaptive significance of habitat selection in midwestern riparian forests (Jones 2001).

*Prey abundance.*—Prey abundance is frequently cited as a factor influencing territory selection in birds. Since higher prey biomass is often positively associated with reproductive success (Conner et al. 1986, Blancher and Robertson 1987, Martin 1987, Simons and Martin 1990, Rodenhouse and Holmes 1992, Burke and Nol. 1998, Zanette et al. 2000), territory placement may be critical to a pairs' productivity. Blancher and Robertson (1987) noted several other effects of high insect biomass including earlier clutch dates, larger clutch sizes, higher nestling growth rates, shorter time between foraging flights and shorter time between nestling feedings compared to birds with low food supply in Western Kingbirds (*Tyrannus verticalis*). Thus, studies that have found that birds preferentially chose territories with higher prey availability include Prothonotary Warblers (*Protonotaria citrea*, Petit and Petit 1996), Northern Goshawks (*Accipiter gentilis*, Kenward and Widén 1989), Canary Islands Stonechats (*Saxicola dacotiae*, Illera 2001), and Ovenbirds (*Seiurus aurocapillus*, Smith and Shugart 1987, Burke and Nol. 1998). In fact, Smith and Shugart (1987) found that Ovenbird territories had 1.6 times greater prey abundance than areas not occupied. Although prey abundance has seldom been used to predict the occurrence and abundance of forest birds at larger

spatial scales (i.e., stand), Brush and Stiles (1986) found prey abundance best predicted densities of insectivores at several scales. For instance, bird density was highly correlated with arthropod biomass and not with foliage volume at the stand scale. More ornithologists have detected this relationship since Brush and Stiles' study (Brush and Stiles 1986, Robinson 1996, Jokimäki et al. 1998, Rottenborn 1999).

Yet, area-related edge effects, like drier microhabitat conditions, may reduce arthropod abundance (Burke and Nol. 1998, Bolger et al. 2000), thereby, influencing both territory and site selection by birds. Burke and Nol (1998) demonstrated that large woodlots had 10 to 36 times greater leaf litter arthropod biomass than small woodlots, ultimately reducing Ovenbird pairing success in small woodlots. Zarette et al. (2000) found that invertebrate biomass in small fragments was about half of that found in large fragments. Consequently, female Eastern Yellow Robins (*Eopsaltria australis*) had a shorter breeding season, lighter eggs and smaller nestlings in smaller fragments. Similarly, biomass of arthropods has been shown to decline with urbanization due to increased fragmentation, pollution, climate changes or invasive plant introduction (Beissinger and Osborne 1982, Blair and Launer 1997, Deny and Schmidt 1998, Bolger et al. 2000, Bowman and Marzluff 2001).

*Vegetation structure.*—Vegetation structure throughout the habitat patch (MacArthur and MacArthur 1961, Beissinger and Osborne 1982, Green et al. 1989, Jokimäki and Suhonen 1993, Germaine et al. 1998) and at the nest (Stauffer and Best 1986, Martin and Roper 1988, Martin 1993, Larrison et al. 2001, Chase 2002) has been found to influence avian selection. Greater complexity in vegetation structure or floristic composition likely increases available singing perches, distribution and availability of

food, shelter from predators, and availability of suitable nesting locations (Wiens et al. 1987). Additionally, studies have shown that the type or amount of vegetation concealing a nest may directly affect predation risk (Martin and Roper 1988, Kelly 1993, Martin 1998, Clark and Shutler 1999, Chase 2002, Borgmann and Rodewald *in review*). For example, Martin and Roper (1988) found that Hermit Thrush (*Catharus guttatus*) selected nest sites with a larger number of potential nest sites nearby, reducing predation risk.

Few studies, however, have examined differences in vegetation structure and floristics along an urbanization gradient. Urban development may promote invasion of forests by exotic flora (Airola and Buchholz 1984, Hobbs 1988, McDonnell and Pickett 1990), possibly causing a shift from a diverse avian community to one dominated by a few invasive or exotic avian species. Therefore, structure of urban and rural forests may differ (Rudnický and McDonnell 1989, Zipperer et al. 2000). This shift in plant communities may negatively affect avian nesting success (Beissinger and Osborne 1982, Blair 1996, Schmidt and Whelan 1999, Borgmann and Rodewald *in review*) and alter associations and processes of forest communities in urban landscapes (McDonnell and Pickett 1990).

## FOREST WIDTH

It is well documented species abundance and diversity of Neotropical migrants are positively related with forest area (Forman et al. 1976, Ambuel and Temple 1983, Blake and Karr 1987, reviewed by Askins et al. 1990). In particular, Temple (1986) showed that more area-sensitive birds were found in forests with a size and shape that included a large core area of habitat as opposed to a narrow, linear band of habitat.

Because riparian forests are typically linear bands of habitat with reduced core area, they may experience exacerbated edge effects. Thus, the least complicated technique for improving core habitat in riparian forests is to increase the forest width. Not surprisingly, several studies have shown that avian species richness and abundance of area-sensitive species are positively associated with riparian forest width (Stauffer and Best 1986, Keller et al. 1993, Darveau et al. 1995, Hodges and Krementz 1996, Kinley and Newhouse 1997).

A range of riparian widths has been suggested by several organizations to provide suitable wildlife habitat. The Ohio Department of Natural Resources' (ODNR) Division of Natural Areas and Preserves recommends a forest  $\geq 40$  m to maintain a viable forest community on larger streams (ODNR 1998). Based on surveys in Delaware and Maryland, Keller et al. (1993) recommended that riparian forests be at least 100 m wide to provide some nesting habitat for area-sensitive birds. When studying this relationship, Kinley and Newhouse (1997) suggested an average width of 70 m for higher densities of all birds and especially those associated with riparian systems. In riparian forests of Georgia, Hodges and Krementz (1996) also found that species richness and abundance increased with increasing corridor width and suggested a 100 m forest strip for Neotropical migratory birds. Lastly, Peterjohn and Rice (1991) recommend that riparian forests be 120-150+ m wide before they are suitable for nesting Acadian Flycatchers in Ohio. Clearly, most work demonstrates that wider forests are preferable to narrow forests for Neotropical migratory songbirds.

## LANDSCAPE MATRIX

Although local habitat characteristics influence bird communities, management at this scale alone may provide inadequate results because landscape characteristics (e.g., forest cover and land use) can substantially influence the forest communities they surround (Askins 1995, Donovan et al. 1997, Saab 1999, Rodewald and Yahner 2001a, b). Landscape matrix, the dominant component in the landscape, is influential because it may alter dispersal and colonization rates, provide source habitat for exotic species and predators invading habitat fragments, and matrix vegetation type and structure can determine the severity of edge effects (Davies et al. 2001). As a result, habitat disturbance (e.g., reduced forest cover) in the landscape has been demonstrated to decrease species richness and abundance (Wilcove 1985, Triquet et al. 1990, Croonquist and Brooks 1993, Friesen et al. 1995, Mensing et al. 1998, Rottenborn 1999, Boulinier et al. 2001). In particular, Neotropical migrants were found to be more sensitive to landscape structure than either temperate migrants or resident species (Flather and Sauer 1996).

Disparate land uses such as urban development and agricultural practices may differentially affect forest-dependent bird communities (Blair 1996). Numerous studies have shown that diversity and abundance of Neotropical migrants consistently decreased as adjacent residential development increased, regardless of forest size (Wilcove 1985, Triquet et al. 1990, Croonquist and Brooks 1993, Friesen et al. 1995, Mensing et al. 1998, Rottenborn 1999, Cam et al. 2000, Boulinier et al. 2001). For example, a richer, more abundant Neotropical migrant avian community was found in 4-ha plots with no surrounding houses compared to 25-ha urban woodlots (Friesen et al. 1995). In addition,



numerous studies have found abundance of potential predators was positively related to housing density (Bowman and Marzluff 2001, Haskell et al. 2001, Marzluff 2001, except see Danielson et al. 1997 and Chalfoun et al. 2002). In Maryland and Tennessee, nest predation rates were especially intense in small woodlots near suburban neighborhoods compared to large rural woodlots (70.5% versus 47.5%; Wilcove 1985).

Alternatively, Brown-headed Cowbirds and predators may increase in agricultural landscapes due to a constant food supply from adjacent farms. The Brown-headed Cowbird, an obligate brood parasite, is a serious problem for many eastern forest birds in fragmented landscapes (Donovan et al. 1997, Donovan et al. 2000, Petit and Petit 2000, Robinson et al. 2000, Thompson and Dijak 2000). For example, Brittingham and Temple (1983) documented that 67% of nests within 100 m of edges were parasitized by Brown-headed Cowbirds in Wisconsin. Predation rates also were higher near forest edges along farmlands and in landscapes with large proportions of agricultural land compared to forest interiors (Andr n 1992, 1995, Huhta et al. 1996, Bayne and Hobson 1997). Consequently, landscape variables have been found to be reliable predictors of abundance for many bird species in midwestern forests (Saab 1999, Howell et al. 2000) and should be considered when making bird population management decisions.

#### RATIONALE AND SIGNIFICANCE

Riparian forests represent a substantial amount of remaining forest throughout the Midwest (Laub 1979, Groom and Grubb 2002) and their conservation has received considerable attention. In Franklin County (Ohio) alone, forest area decreased by 42% from 1984 to 1999, making only 8.3% of the total area of Franklin County forested (Landsat thematic data). Yet, management recommendations for most riparian forests are

based on single scale studies that do not incorporate landscape-scale influences. In fact, appropriate management of riparian forests may differ based on the surrounding landscape. Thus, effective management plans can be developed by understanding how local and landscape features of riparian forests affect forest-dependent avian communities. This study system is ideal and applicable because riparian habitats extend the entire urban-rural gradient in central Ohio and other Midwest areas.

Examining multi-scale impacts of land uses on riparian forests is essential for several reasons (1) management at one scale may overlook important relationships, (2) managing for local features may improve productivity of songbirds, (3) as urban sprawl continues, evaluating the effects of urban development on riparian communities is essential for developing effective management plans, (4) understanding habitat selection by Acadian Flycatchers and its fitness consequences may be relevant when managing for other Neotropical migrants, and (5) managing for landscape-scale influences may potentially increase the amount of breeding habitat available for Neotropical migrants in the Midwest .

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## CHAPTER 2

### HABITAT SELECTION BY THE ACADIAN FLYCATCHER: A HIERARCHICAL APPROACH

*Abstract.* Selection of habitat can be viewed as a hierarchical process, from a geographic region down to an individual's home range or territory. This study used a hierarchical approach to examine habitat selection by Acadian Flycatchers (*Empidonax virescens*) across an urbanization gradient. Habitat selection was investigated at three distinct scales from territory (1 ha), and stand (3-5 ha) to landscape (314 ha) in 36 mature riparian forests in central Ohio. I addressed potential mechanisms behind landscape-scale patterns by examining understory arthropod biomass, vegetation structure, and breeding productivity. Riparian forests ranged from 55-565 m in width and from 0-53% urban development within 1 km. Acadian Flycatchers were surveyed three times per year during June 2001-2002. Vegetation characteristics and understory arthropod biomass were measured at multiple locations within each riparian forest stand and Acadian Flycatcher territory. I also monitored nest fate of 81 Acadian Flycatcher nests and season-long productivity of 11 banded Acadian Flycatchers. Territory selection was evaluated by comparing habitat features measured within the territory ( $\leq 1$  ha) with mean stand values. Two sets of *a priori* candidate models were developed to represent

plausible factors affecting abundance and productivity of Acadian Flycatchers at the stand (6 models) and landscape (4 models) scales. Models were ranked using an information theoretic approach and Akaike's Information Criterion (AIC). Results suggest that Acadian Flycatchers select habitat in a hierarchical manner. Percent urban cover in the landscape best explained variation in the abundance of Acadian Flycatchers. In fact, Acadian Flycatchers were over 3 times as likely to be detected in the most rural riparian forests ( $\leq 1\%$  urban) compared to more urban riparian forests ( $\geq 10\%$  urban). Likewise, productivity and nest success declined as urban development increased within the landscape. I suggest that increased nest predation, coupled with reduced productivity and site fidelity, in urban riparian forests may be an underlying mechanism of landscape-scale patterns in Acadian Flycatcher abundance. Although vegetation structure and understory arthropod biomass failed to explain habitat selection at the stand scale, they seemed central in territory selection. For example, density of understory vegetation ( $\leq 3.0$  m) was lower and understory arthropod biomass was greater within the territory than throughout the stand. Results provide evidence that selection criteria of Acadian Flycatchers change with spatial scale and, thus, demonstrate that examination of multiple spatial scales is critical to elucidating habitat selection. In particular, explicit consideration of the landscape matrix is critical to effectively manage riparian forests. In particular, establishment of low-development buffer zones around forests vulnerable to urban pressures may be warranted. Likewise, land acquisition efforts aimed at sensitive forest birds, like Acadian Flycatchers, may be best directed at rural landscapes.

## INTRODUCTION

Habitat selection is among the most well-studied aspects of ecology. Indeed, a wide variety of taxa, including insects (Morisita 1952, Rausher 1983, Istock and Weisburg 1987), fish (Bay et al. 2001, Munday 2001), mammals (Wecker 1963, 1964, Apps et al. 2001, McLoughlin et al. 2002), plants (Thery 2001), and reptiles (Kiestler 1975, Morrow et al. 2001, Compton et al. 2002) are known to select territories or sites with specific habitat characteristics. Undoubtedly though, birds are the most widely studied taxa in habitat selection studies, perhaps due to their ubiquity and mobility (see Cody 1985 for review).

Studies of avian habitat selection have progressed from local habitat comparisons among species (Grinnell 1904, 1917) to distribution (e.g., MacArthur 1972, Rotenberry and Wiens 1980) to community structure (e.g., Lack 1944, MacArthur 1958, Smith and MacMahon 1981, Wiens and Rotenberry 1981, Robinson and Holmes 1982). Examples of proximate factors that affect habitat selection include vegetation structure (e.g., MacArthur and MacArthur 1961, Willson 1974, Roth 1976, Beissinger and Osborn 1982, Rotenberry 1985, Petit et al. 1988, Green et al. 1989, Jokimäki and Suhonen 1993, Germaine et al. 1998) and resource availability (e.g., Verner and Willson 1966, Petit et al. 1988, Petit and Petit 1996). More recently, predation pressure has been seen as another mechanism driving avian habitat selection (Leonard and Picman 1987, Martin 1988, 1993, Martin and Roper 1988, Lima 1993, Sieving and Willson 1998, Rodewald and Yahner 2001a).

Traditionally, most studies of habitat selection have chosen a specific spatial scale, typically the local or microhabitat scale thought to be relevant to a focal species or

group. However, the selection process may start at a much larger scale (i.e., landscape) followed by selections at smaller scales (i.e., stand and territory). This hierarchical selection process has been characterized as descending steps from selection of geographic range down to procurement of resources (Johnson 1980). Recently, ecologists have investigated how abundance and connectivity of habitat at the landscape scale may affect this hierarchical selection process (Virkkala 1991, Rolstad et al. 2000, Illera 2001, Penteriani et al. 2001, MacFaden and Capen 2002, McLoughlin et al. 2002). Few, however, have examined the influence of the surrounding landscape matrix. Urban development, in particular, has been recognized as a primary threat to many natural habitats and species but has received relatively little attention in the literature, especially its impacts on riparian forest communities (Douglas 1992, Blair 1996, Niemelä 1999, McKinney 2002, Miller and Hobbs 2002). In these studies, however, constraints on study site selection prevented clear demonstration of how the landscape matrix influences the hierarchical processes. Ultimately, studies that fail to examine scale-dependence in habitat selection cues may have limited contributions to conservation and management efforts.

The aim of this study was to examine multi-scale impacts of land uses on habitat selection by Acadian Flycatchers (*Empidonax virescens*), and its implications for conservation of riparian forests. Acadian Flycatcher was selected as a focal species due to apparent landscape sensitivity (Whitcomb et al. 1981, Smith and Schaefer 1992) as well as high habitat specificity (Johnston 1970, Bisson et al. 2000, Whitehead and Taylor 2002). Acadian Flycatchers also have a relatively high-priority ranking for management and monitoring in the Midwest, ranking 15<sup>th</sup> of 110 Neotropical migrants (Thompson et



al. 1993; Whitehead & Taylor 2002). Population trends reported by Breeding Bird Survey data indicate Acadian Flycatchers have experienced significant declines between 1966 and 1998 in Ohio (trend = -1.70 birds/route,  $P = 0.064$ ; Sauer et al. 2001). Because habitat selection can occur at multiple spatial scales (Johnson 1980), I investigated habitat selection at the landscape (314 ha), stand (3-5 ha) and territory (1 ha) scale. I addressed the following potential mechanisms behind landscape-scale patterns by examining 3 factors: (1) understory arthropod biomass, (2) vegetation structure, and (3) breeding productivity. In particular, I hypothesized that habitat selection at the landscape scale was best explained by differences in productivity among landscapes, whereas resource-based selection (i.e., prey abundance and vegetation characteristics) occurred at smaller spatial scales. This study is the first to examine multi-scale habitat selection by a Neotropical migrant across an urbanization gradient.

## METHODS

### Study Area

Thirty-six riparian forest sites were selected within the Scioto River Watershed in central Ohio (Appendices A and B). Field sites were located on public and private lands in Franklin, Pickaway, and Delaware counties. Land cover in the three counties averaged 57.2% agriculture/open land, 29.6% urban, and 6.3% forest (USGS EROS Data Center 2000). Landsat thematic mapper imagery classified land covered by open vegetation (non-forested lacking wetlands or water) as agriculture/open land, and land covered by buildings and pavement as urban land (USGS EROS Data Center 2000). Using 1994-1995 USGS Digital Orthophoto Quadrangles (1:24,000), and detailed county maps I identified potential study sites in riparian forests that met the following criteria: (1)

mature deciduous forests (i.e., all strata present), (2)  $\geq 250$  m in length, (3)  $\geq 40$  m in width, (4) separated from each other by  $\geq 2$  km, (5)  $\geq 10$  m width of forest on the other side of the waterway and, (6) bordered waterways 20-40 m in width. These sites encompassed a range of forest widths from approximately 55-565 m on one side of the stream (or 85-585 m including both sides minus water width). Landscapes surrounding study sites spanned an urbanization gradient ranging from primarily agricultural matrices to urbanizing matrices with considerable coverage of paved surfaces and buildings.

Topography of sites was typical of bottomland forest with primarily flat, moist terrain interspersed with channels scoured by intermittent floods. Dominant canopy tree species included American sycamore (*Platanus occidentalis*), eastern cottonwood (*Populus deltoides*), silver maple (*Acer saccharinum*), sugar maple (*Acer saccharum*), boxelder (*Acer negundo*), common hackberry (*Celtis occidentalis*), and American elm (*Ulmus americana*). Understory vegetation consisted primarily of saplings of canopy trees and Ohio buckeye (*Aesculus glabra*), pawpaw (*Asimina triloba*), hawthorn (*Crataegus* spp.), amur honeysuckle (*Lonicera maackii*), multiflora rose (*Rosa multiflora*), northern spicebush (*Lindera benzoin*), and common elderberry (*Sambucus canadensis*). Invasion by *Lonicera* spp. and *Rosa multiflora* resulted in dense patches of shrubs in some sites. Common breeding birds found in the riparian forests were American Robin (*Turdus migratorius*), Northern Cardinal (*Cardinalis cardinalis*), Acadian Flycatcher, Wood Thrush (*Hylocichla mustelina*), Red-eyed Vireo (*Vireo olivaceus*), House Wren (*Troglodytes aedon*), Downy Woodpecker (*Picoides pubescens*), Yellow-throated Warbler (*Dendroica dominica*), Northern Parula (*Parula americana*), Eastern Wood-Pewee (*Contopus virens*), Great Crested Flycatcher (*Myiarchus crinitus*),

Tufted Titmouse (*Baeolophus bicolor*), and Carolina Chickadee (*Poecile carolinensis*). Potential nest predators commonly found within the riparian forests included Blue Jay (*Cyanocitta cristata*), American Crow (*Corvus brachyrhynchos*), common raccoon (*Procyon lotor*), eastern chipmunk (*Tamias striatus*), and eastern gray squirrel (*Sciurus carolinensis*).

### Territory Scale

On a subset of 11 sites, one male Acadian Flycatcher was banded and his territory was mapped. Only one male per site was monitored to avoid pseudoreplication of data. This subset of sites averaged 9.45% ( $\pm 4.14\%$ ) urban cover (range = 0-46%) and 103.85 m ( $\pm 40.94$  m) in forest width (range = 104-565). Assistants and I then searched for and monitored nests of Acadian Flycatchers on these sites from late-May through mid-August 2001 and 2002. Each active nest ( $n = 23$ ) was checked every 3-5 days. For each banded male, I also directly measured breeding productivity, which was defined as number of fledglings produced per female (Rowley and Russel 1991, Murray 2000). If a territory had  $\geq 1$  female (in 3 cases), I averaged productivity across females.

Mean territory size of Acadian Flycatchers in the study system was  $1.0 \text{ ha} \pm 0.1$  SE (Appendix D) and all vegetation and prey characteristics were collected within known breeding territories. Vegetation characteristics were measured within a 0.04-ha plot centered on each Acadian Flycatcher nest using a modified Breeding Bird Survey Protocol (BBIRD, Martin et al. 1997). Characteristics measured included tree species in 3 diameter breast height (dbh) size class (8 - 23 cm, 23 - 38 cm,  $>38$  cm), number of standing dead trees ( $>15$  cm dbh), and number of fallen logs ( $\geq 7.5$  cm dbh). Two 20-m perpendicular transects, running North-South and East-West, were established in the

center of each plot. Percent canopy cover ( $> 5$  m), percent ground cover ( $< 0.5$  m), and understory vegetation density (0.5-3.0 m) were measured at 2-m intervals along these transects (James and Shugart 1970, Martin et al. 1997, Chase 2002).

Relative prey abundance for Acadian Flycatchers was assessed using the sweep net sampling method (Beaver and Baldwin 1975, Gray 1993). Because Acadian Flycatchers forage predominantly in the understory (Via 1979, Whitehead and Taylor 2002, Bakermans pers. obs.), and flycatchers often glean insects from the top layers of vegetation (Murphy 1986, Gray 1993, Huhta et al. 1999) samples obtained by sweep netting should accurately reflect potential prey abundance. In fact, numerous studies have used sweep netting to relate some aspect of flycatcher ecology (e.g., diet, foraging behavior, territory size, productivity) to arthropod abundance: Eastern Kingbird (*Tyrannus tyrannus*, Murphy 1986), Willow Flycatcher (*Empidonax traillii*, Prescott and Middleton 1988), Pied Flycatcher (*Ficedula hypoleuca*, Forsman et al. 1998, Huhta et al. 1999), Eastern Wood-Pewee and Great Crested Flycatcher (Gray 1993), and Western and Hammond's Flycatchers (*Empidonax difficilis* and *Empidonax hammondi*, Beaver and Baldwin 1975).

I estimated food abundance in 11 Acadian Flycatcher territories. Within each territory, 10 samples were collected using randomly selected compass coordinates and distances (0-100 m, depending on its size and shape) from the center. Prey abundance was then averaged among the 10 points for each territory (Conner et al. 1986, Smith and Shugart 1987). A standard sample unit consisted of two strokes of the net in a continuous back-and-forth motion with each stroke approximately 2 m in length (modified Hughes 1955). Samples were collected at each cardinal direction from 0 to 1.0, 1.1 to 2.0, 2.1 to

3.0 meters and from the air using a 35-cm diameter net (0.5-mm mesh, Prescott and Middleton 1988, Huhta et al. 1999). Samples were frozen and later weighed to determine relative understory arthropod biomass (Conner et al. 1986). Arthropods were sampled in late morning on days without rain and only when the vegetation was dry (Forsman et al. 1998, Huhta et al. 1999). Samples were collected the first two weeks of July during peak Acadian Flycatcher nestling and fledging stages for Ohio (C. N. Willis, Youngstown State Univ., pers. comm., Bakermans pers. obs.), and when food should be most limiting (Martin 1987).

### Stand Scale

Stand scale for this study was 3-5 ha. Vegetation characteristics were measured in four 0.04-ha circular plots centered on the 250-m long transect running parallel and 20 m from the river edge (modified James and Shugart 1970). Plots were placed at 50-m intervals from 50-200 m along the transect. Methods were identical to those described earlier for the territory scale and habitat characteristics were measured at all 36 sites.

Potential food abundance (i.e., understory arthropods) was estimated on a subset of 22 sites. This subset of sites averaged 9.3% ( $\pm 2.7\%$ ) urban cover (range = 0-46%) and 167.0 m ( $\pm 22.3$  m) in forest width (range = 55-565 m). Again, sweep netting was used because it targeted potential prey for Acadian Flycatchers and previous studies found sweep netting a useful method when comparing relative site-to-site abundance of invertebrates (Jokimäki et al. 1998). Arthropods were collected at 10-m intervals from 25–225 m along a transect bisecting the site (e.g., 20 sampling points per site; see Figure 2.1) in areas which should be equally accessible to Acadian Flycatchers. At each interval, a sampling point was chosen at a random distance (0 m to site half-width or a

maximum of 150 m) from the transect (half toward the water and half toward the edge). Understory arthropod biomass at each point was sampled in an identical manner as described under territory scale methods. Arthropod samples collected throughout a site were averaged to generate a mean relative understory arthropod biomass.

### Landscape Scale

I chose a 1-km radius area (3.14 km<sup>2</sup>) to represent a landscape scale because it far exceeded average territory size for Acadian Flycatchers, and thus, should be perceived as a landscape (Freemark et al. 1993, Hostetler 2001). Furthermore, the 1-km-radius spatial scale is a convenient scale for management activities because it tends to include less diversity in land ownership and use than larger scales, and is easily compared to other studies, which frequently use the same scale. Landscape matrix (% urban land cover within 1 km) and riparian forest width and their interaction were examined in landscape scale habitat selection. Forest width, rather than percent forest cover, was included in landscape scale analyses because it better characterized habitat availability for Acadian Flycatchers within the landscape. Area estimates were not possible because riparian forests were relatively continuous along waterways and did not exist as distinct habitat patches.

Riparian forest width was measured from digital orthophotos (Table 2.1). Forest width was averaged over four locations along the transect. These measurements were taken at 50-m intervals and were positioned perpendicular to the general flow of the river. Mean forest width was 163.9 m  $\pm$  15.0 SE (range = 55.8-565.4 m). Because riparian forest width (on the side of the waterway where surveys, arthropod and vegetation measurements occurred) was highly correlated with the total forest width (encompassing

both sides of the waterway;  $r = 0.72$ ,  $P < 0.001$ ), only the width of the surveyed side was used in analyses.

Percent urban and agricultural land cover within 1-km radius of each site were measured using thematic mapper imagery and ARC/VIEW geographic information software (USGS EROS Data Center 2000; Table 2.1). Because percent urban and agricultural land cover were negatively correlated ( $r = -0.63$ ,  $P < 0.001$ ) only urban land (range = 0-53%; mean =  $14.5\% \pm 2.9$  SE) was used in analyses. Neither width nor percent forest cover within 1 km were confounded with urban development in the landscape ( $r = -0.13$ ,  $P = 0.45$  and  $r = -0.27$ ,  $P = 0.12$ , respectively). Thus, riparian forests in urban and rural landscapes had comparable widths and surrounding forest cover.

Abundance of Acadian Flycatchers was determined at all sites by belt transect surveys (Emlen 1971, Bibby et al. 1992). A single observer slowly walked each 250-m-long transect in 20 minutes, recording all Acadian Flycatchers seen or heard within 40 m of the river. Sites were surveyed between 0.5 hours before sunrise and 4.0 hours after sunrise (0545-1000) on days with good weather (wind  $< 32$  km/hr and little or no precipitation). Riparian forests were surveyed once per week from June 4 - June 24, 2001 and 2002 (3 times per year).

#### Data Analysis

Abundance of Acadian Flycatchers was averaged over three visits in each year for each site. Because no annual differences were detected for numbers of Acadian Flycatchers ( $\chi^2 = 2.34$ ,  $P = 0.13$ ; PROC NPAR1WAY, SAS Institute 1990), abundance was averaged over the two years for each site. Vegetation measurements were averaged

over 4 plots per site. Vegetation variables that were excluded from analyses included one member of highly correlated pairs ( $r \geq 0.70$  and  $P \leq 0.05$ ) and variables that were relatively constant across sites (coefficient of variation  $\leq 10$ ). Remaining vegetation variables included large trees (dbh  $>38$  cm), small trees (dbh 8 - 23 cm), standing dead trees ( $>15$  cm, hereafter snags), and understory vegetation density (0.5 - 3.0 m, hereafter vegetation density).

Habitat selection at the territory scale was evaluated by comparing habitat features measured within the territory ( $\leq 1$  ha) with average stand values. An information-theoretic approach was used in modeling because it allowed for selection of the best model out of several while providing effect sizes rather than just significance (Larson et al. 2001). At the stand scale, a set of 6 *a priori* candidate models were developed representing biologically plausible factors affecting abundance of Acadian Flycatchers. These included four vegetation variables (large trees, small trees, snags, and vegetation density), understory arthropod biomass, and a null model. The null model (i.e., a constant) was used as a comparison to judge if the inclusion of no explanatory variables produced a better model than one containing selected variables. Additionally, at the landscape scale, a set of 4 *a priori* models was developed incorporating potential factors affecting (1) productivity, and (2) abundance of Acadian Flycatchers. The variables percent urban development within 1 km, forest width, and their interactions were used for all landscape analyses. I then selected the best model based on Akaike's Information Criterion (AIC) corrected for bias due to small sample size relative to number of parameters ( $AIC_c$ ). The best model has the lowest  $AIC_c$  value while subsequent models were assessed by delta  $AIC_c$  ( $\Delta_i$ , the difference between the model's



AIC<sub>c</sub> and that of the best model) and Akaike weights ( $\omega_i$ , weight of evidence for each model; Burnham and Anderson 1998). Models with a  $\Delta_i < 2$  were considered equally reasonable given the data (Burnham and Anderson 1998). I used a generalized linear model with specified distributions (i.e., negative binomial distribution for abundance and productivity of Acadian Flycatchers, Schabenberger and Pierce 2002) to calculate log-likelihood values (PROC GENMOD, SAS Institute 1990) that were subsequently converted to AIC<sub>c</sub> values.

## RESULTS

Acadian Flycatchers were detected in 18 of 36 riparian forests and only in forests  $\geq 85$  m wide (Table 2.1). Understory arthropod biomass ranged from 0.003-0.142 g (mean =  $0.088 \pm 0.015$  g SE,  $n=11$ ) within Acadian Flycatcher territories and 0.002-0.118 g (mean =  $0.059 \pm 0.008$  g SE,  $n=22$ ) throughout the forest stand (Table 2.2). Territories of Acadian Flycatchers had 1.5 times greater understory arthropod biomass than random locations throughout a site. Large trees, small trees, snags, and vegetation density averaged 1.2-2.5 times lower within the territory than throughout the stand (Table 2.2). Of the 4 vegetation variables examined at the territory scale, vegetation density was the only variable that differed by  $> 1$  SE from the stand average (Table 2.2). In fact, vegetation density was 2.5 times lower within the territory than throughout the stand. However, at the stand scale, the best model explaining the observed variation in Acadian Flycatcher abundance did not include any explanatory variables (Table 2.3). In fact, based on Akaike weights, the null model was ranked 3.2 times better than the next best model, strongly suggesting that specific habitat features were not selected at stand scales.

Percent urban cover best explained observed variation in abundance of Acadian Flycatchers (Table 2.4), and was 4.7 times as likely to be the best model based on Akaike weights. Abundance of Acadian Flycatchers was negatively related to urban development within the landscape [ $y = -0.17 + (-0.05 \times \text{percent urban development}) + (0.23 \times \text{dispersion parameter})$ ; Figure 2.2]. In fact, numbers of Acadian Flycatchers were over 3 times greater in the most rural landscapes ( $\leq 1\%$  urban development) than in the more urban landscapes ( $\geq 10\%$  urban development).

A total of 81 nests, including 23 nests in Acadian Flycatcher territories were monitored. Nests varied from 0-200 m (mean =  $74.99 \pm 5.13$  m SE) from the forest edge. Territories produced an average of  $1.36 \pm 0.49$  SE fledglings (range 0-4, Table 2.1), and three pairs successfully double-brooded. However, 58% of all nests failed due to nest predation. Similar to Acadian Flycatcher abundance, productivity was negatively related to urban development in the landscape [ $y = 0.87 + (-0.06 \times \text{percent urban development}) + (0.71 \times \text{dispersion parameter})$ ; Figure 2.3]. Although the model containing percent urban cover best explained Acadian Flycatcher productivity, forest width was equally plausible given the data (i.e.,  $\Delta_i < 2$ , Table 2.5). For example, edge effects were detected for Acadian Flycatcher nests where successful nests were further from the forest edge (mean =  $90.12 \text{ m} \pm 9.19$  SE,  $n = 34$ ) than unsuccessful nests (mean =  $64.04 \text{ m} \pm 5.38$  SE,  $n = 47$ ). These two models were ranked 5 times better than the remaining models, suggesting that both urban development within 1 km and riparian forest width affected Acadian Flycatcher productivity.

## DISCUSSION

Results suggest that criteria used by Acadian Flycatchers to select habitat vary with spatial scale and may change in a hierarchical manner. This finding is consistent with other hierarchical studies of habitat selection demonstrating that single-scale studies may not accurately identify factors to which a species responds in a landscape (Wiens et al. 1987, Orians and Wittenberger 1991, Hostetler and Holling 2000, Rolstad et al. 2000, Hostetler 2001, MacFaden and Capen 2002, McLoughlin et al. 2002). For example, Wiens et al. (1987) found that Sage Thrasher (*Oreoscoptes montanus*) density was positively related to the following vegetation features at different scales: vegetation height and coverage of shrubs (biogeographic scale), vegetation vertical structure (regional scale, Pacific Northwest), gray rabbitbrush (*Crysothamnus nauseosus*; local scale, 9 ha), and lower shrub coverage (territory scale). Although others have demonstrated that landscape-scale features influence habitat selection, my results differ from previous work in that I examined proximate factors that may be responsible for landscape associations.

At the smallest spatial scale (territory), Acadian Flycatchers seemed to select territories that contained 1.5 times greater biomass of understory arthropods than random locations in the stand. Other studies have found that prey availability may influence territory selection; Prothonotary Warblers (*Protonotaria citrea*, Petit and Petit 1996), Northern Goshawks (*Accipiter gentilis*, Kenward and Widén 1989), Canary Islands Stonechats (*Saxicola dacotiae*, Illera 2001), and Ovenbirds (*Seiurus aurocapillus*, Smith and Shugart 1987, Burke and Nol. 1998) preferentially chose territories with higher prey availability. For example, Smith and Shugart (1987) found that Ovenbird territories had

1.6 times greater prey abundance than areas not occupied. Greater prey biomass may be positively associated with reproductive success (Conner et al. 1986, Blancher and Robertson 1987, Martin 1987, Simons and Martin 1990, Rodenhouse and Holmes 1992, Burke and Nol. 1998, Zanette et al. 2000); thus, territory placement may be critical to a pairs' productivity. Indeed, in my study, Acadian Flycatcher territories with successful nests in 2001 had greater understory arthropod biomass than territories with unsuccessful nests (mean =  $0.13\text{g} \pm 0.01\text{ SE}$ ,  $n = 5$  versus mean =  $0.08\text{g} \pm 0.02\text{ SE}$ ,  $n = 4$ , Bakermans and Rodewald, unpublished data).

Interestingly, the apparent preference for territories with high understory arthropod biomass did not persist at larger spatial scales. In fact, urban landscapes had greater understory arthropod biomass than rural sites [ $y = 0.05 + (0.01 \times \text{percent urban development})$ ; Figure 2.4]. This result was inconsistent with Brush and Stiles (1986), who found arthropod abundance predicted densities of insectivores at the site level. Furthermore, Orians and Wittenberger (1991) found that Yellow-headed Blackbirds (*Xanthocephalus xanthocephalus*) selected marshes, not territories, based on prey availability. In my system, however, understory arthropod biomass seemed important only at small (i.e., territory) scales.

Vegetation structure may have been another cue for Acadian Flycatcher territory selection in my study system. Vegetation density was 2.5 times lower within the nest patch than throughout the stand, suggesting that Acadian Flycatchers may avoid areas with dense understory vegetation, possibly associated with invading exotic shrubs. These findings are consistent with Wilson (1997), Hazler (1999), and Bell and Whitmore (2000) who found that Acadian Flycatcher nests were often placed in more open spaces (i.e., less

shrub cover) than random locations. Vegetation density within the nest patch may be an important cue for Acadian Flycatchers because they prefer an open understory for foraging, nest maintenance and aerial defense (Johnston 1970, Via 1979, Maurer and Whitmore 1981, Ehrlich et al. 1988, Wilson and Cooper 1998, Whitehead and Taylor 2002).

An interesting contrast between my study and others is that vegetation characteristics failed to account for variation in Acadian Flycatcher numbers across study sites. Vegetation characteristics have long been recognized as a driving mechanism of avian diversity and abundance across sites (i.e., vertical foliage diversity hypothesis, MacArthur and MacArthur 1961). Although there was variation in vegetation and arthropod features across sites (i.e., coefficients of variation ranged from 37.5-65.0), none of these variables explained the variation in abundance of Acadian Flycatchers across sites. Instead, the null model, with no explanatory variables, was 3.2 times more likely to be the best model, based on Akaike weights. The most likely explanation is that rather than specific vegetation or arthropod features being important at the stand, these were more important at a smaller spatial scale (i.e., territory), as shown earlier. In addition, vegetation variables were not associated with urban development in the landscape ( $r < 0.70$ ), and thus, do not explain habitat selection at the landscape scale.

At the landscape scale, the amount of urban development within 1 km was negatively related to numbers of Acadian Flycatchers. This is consistent with studies showing that composition of the surrounding landscape matrix can affect diversity (Croonquist and Brooks 1993, Cam et al. 2000, Boulinier et al. 2001) and abundance (Friesen et al. 1995, Rottenborn 1999, Howell et al. 2000, Rodewald and Yahner 2001b)

of breeding birds within a forest stand. Influence of urban development was apparently strong, as the most rural riparian forests ( $\leq 1\%$  urban development) had over 3 times more Acadian Flycatchers than more urban riparian forests ( $\geq 10\%$  urban development). Interestingly, nest initiation date was positively related to percent urban cover in the landscape [ $y = 146.43 + (0.40 * \% \text{ urban development})$ ; Figure 2.5], suggesting that Acadian Flycatchers selected urban landscapes after preferred rural habitats were fully occupied.

Few studies of riparian birds have recognized the importance of landscape matrix characteristics (but see Rottenborn 1999, Saab 1999, Miller et al. 2001b). Instead, avian species richness and abundance of area-sensitive species are usually related to width of riparian forests (Stauffer and Best 1986, Keller et al. 1993, Darveau et al. 1995, Hodges and Krementz 1996, Kinley and Newhouse 1997). However, these recommended widths of riparian forests may prove inadequate if forests are surrounded by urban or suburban development (Miller et al. 2001a). In this study, I found that suitable forest width was dependent on the surrounding landscape matrix. As urban development in the landscape increased, Acadian Flycatchers selected wider riparian forests ( $\geq 105$  m) than those selected in rural landscapes ( $\geq 85$  m).

I investigated productivity of Acadian Flycatchers as a possible mechanism responsible for the observed landscape patterns. Few studies of avian ecology have measured actual productivity (except see Holmes et al. 1992; Roth & Johnson 1993; Underwood & Roth 2002) due to its time-intensive requirements. Instead, most studies use nest success as a measure of productivity, but this has been found to be a poor indicator of productivity (Underwood & Roth 2002). As with abundance of Acadian

Flycatchers, the amount of urban development within 1 km of riparian forests best explained, and was negatively related to, productivity of Acadian Flycatchers (Table 2.6). Not surprisingly, forest width also affected productivity of Acadian Flycatchers (Table 2.6). With the exception of one outlier, forest width was positively related to productivity of Acadian Flycatchers. For this outlier, which was in the widest forest tract, forest width did not adequately reflect how far the nest was placed from the forest edge because the nest was placed 20 m from a residential edge. Forest width may be an important factor influencing productivity because edge effects were detected for Acadian Flycatchers. In fact, for the 11 pairs of Acadian Flycatchers that had their territories mapped, nests that fledged young were 1.7 times farther from the forest edge than unsuccessful nests. This trend was also detected in all Acadian Flycatcher nests ( $n = 81$ ) monitored throughout the study system, where successful nests averaged  $90.12 \pm 9.19$  m ( $n = 34$ ) compared to unsuccessful nests that averaged  $64.04 \pm 5.38$  m ( $n = 47$ ) from the forest edge. Again, urban development within the landscape affected suitable forest width where successful breeding of Acadian Flycatchers did not occur until 130 m in the more urban forests compared to 106 m in the most rural forests.

Associations at the landscape scale were best explained by variation in Acadian Flycatcher productivity and nest predation, rather than arthropod or vegetation features. Breeding productivity was negatively related to urban development. In fact, no Acadian Flycatcher young were fledged in sites with  $> 12\%$  urban cover in the landscape. Similarly, nest success was negatively related to urban development ( $r = -0.50$ ; Figure 2.6). These findings were consistent with a larger study at our sites where nesting success of common understory nesters was 2.6 times lower in the most urban than rural

landscapes (Rodewald, unpublished data). Difference in productivity and nest success may be related to the number of potential nest predators, which were positively related to urban development. In fact, numbers of predators were over 2 times greater in urbanizing ( $\geq 10\%$  urban development) landscapes than more rural landscapes ( $\leq 1\%$  urban development; Appendix C). Difference in nest predation or productivity ultimately could affect landscape-scale distribution of Acadian Flycatchers by influencing individual decisions about habitat selection. Individuals may use their own reproductive success (or that of their neighbors) to assess the quality of habitat (Greenwood and Harvey 1982, Gavin and Bollinger 1988, Shields et al. 1988, Switzwer 1997, Doligez et al. 2002). In fact, in this study, Acadian Flycatchers that experienced nest predation were 4 times less likely to return to the site the following year (Appendix D). Nest predation, therefore, may be limiting populations of Acadian Flycatchers in urban landscapes due to reduced site fidelity following failed nesting attempts (Gavin and Bollinger 1988, Robinson and Wilcove 1994, Switzwer 1997, Schmidt 2001, Doligez et al. 2002).

## CONSERVATION IMPLICATIONS

### The importance of spatial scale

Habitat selection studies are an integral part of conservation and management because they identify features that may have behavioral and fitness consequences for a particular species or group (Jones 2001). Frequently, practices such as habitat manipulation or reserve acquisition are based on characteristics identified in habitat selection studies. Yet, studies occurring at only one spatial scale may overlook important relationships and therefore, generate ineffective or misleading recommendations. This study suggests that Acadian Flycatchers select habitat in a hierarchical process.



Moreover, my results demonstrate that cues that may guide habitat selection change according to spatial scale. Land managers and biologists should consider a hierarchical approach when designing or improving reserves that addresses land use planning at landscape scales coupled with habitat management. Although management at the local scale may be most convenient, a central challenge facing ecologists and managers will be synthesizing the complex factors that influence animal populations and communities across scales into clear, usable management approaches.

#### Conservation of riparian forests

Riparian zones are among the most productive and ecologically valuable systems (Naiman et al. 1993), and they provide habitat for both aquatic and terrestrial flora and fauna. Yet, loss of floodplain forest has been particularly severe in the American Midwest (Pashley and Barrow 1993, Knutson et al. 1996). Although conservation of riparian forests is important for a variety of ecological functions, such as water quality, bank stabilization, and flood control, they are increasingly acquired and managed for their wildlife value. In this region, riparian forests are frequently constricted, seldom extending more than 100 m from the stream channel. At the same time, these forested stream corridors can extend for miles in an unbroken band of potential habitat, and thereby can profoundly influence local and regional bird communities by functioning as breeding habitat as well as dispersal and migration corridors (Knutson et al. 1996). Indeed, studies demonstrate that midwestern riparian habitats can support high bird diversity (Best and Stauffer 1980, Small and Hunter 1989, Mossman 1991). Additionally, midwestern riparian forests can support numerous Neotropical migratory birds of high management concern (mean score > 3.0 based on Thompson et al. 1993)

including Cerulean Warbler (*Dendroica cerulea*), Prothonotary Warbler, Acadian Flycatcher, Yellow-billed Cuckoo (*Coccyzus americanus*), and Great Crested Flycatcher (Knutson et al. 1996).

The long history of intensive agriculture and now rapid urban development in midwestern landscapes has made riparian forests among the few remaining habitats that can accommodate forest-dependent wildlife (Laub 1979, Groom and Grubb 2002). Continued urban expansion in the Midwest remains a threat to riparian forests because the areal extent of urban areas is increasing more rapidly than human population growth (Marzluff 2001). Science-based decision-making remains problematic because most studies have considered only local factors that can impact riparian forests. In particular, most studies have focused on forest width, showing that avian species richness and abundance of area-sensitive species are positively associated with increasing riparian forest width (Stauffer and Best 1986, Keller et al. 1993, Darveau et al. 1995, Hodges and Krementz 1996, Kinley and Newhouse 1997).

However, my results suggest that urban development within the landscape matrix surrounding riparian forests reduced their suitability for Acadian Flycatchers and possibly other breeding forest birds. These findings have important implications for biologists and land managers in both rural and urbanizing landscapes. First, emphasis should be placed on restoring and maintaining wide riparian forests in any landscape. In my central Ohio system, based on surveys and productivity data, a minimum width of 100 m in rural landscapes and 150 m in urban landscapes seems to be necessary to allow successful reproduction of Acadian Flycatchers. Second, if one goal of habitat protection is to provide habitat for forest birds like Acadian Flycatchers, then land acquisition and

protection efforts should be focused in rural landscapes. Third, because adjacent urban development appears to negatively affect Acadian Flycatchers, low-development buffer zones should be established surrounding riparian forests expected to receive development pressures. Fourth, enhancement of urban riparian forests (e.g., widening forests; controlling predators) may augment the amount of suitable habitat available to Acadian Flycatchers and other Neotropical migrants in the Midwest. Ultimately, consideration of landscape matrix effects on wildlife communities will be an essential component of protection and restoration of riparian forests.

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Site	Mean forest width (m)	Urban (%)	Number of Acadian Flycatchers (SE)	Number of Acadian Flycatchers fledged
3 Creeks Metro Park	132.7	7.8	1.5 (0.2)	4
Bexley Park	132.7	41.6	0.2 (0.2)	*
Big Walnut Park	115.4	45.8	0.3 (0.2)	0
Camp Mary Orton	565.4	15.8	0.0 (0.0)	0
Casto Park	201.9	47.2	0.0 (0.0)	*
Chapman Road	86.5	0.3	0.2 (0.2)	*
Cherrybottom Park	165.4	31.2	0.0 (0.0)	*
Darby Public Hunting	194.2	1.0	1.7 (0.5)	1
Elk Run Park	167.3	12.6	0.0 (0.0)	*
Galena	276.9	1.2	0.7 (0.2)	2
Gardner Road	125.0	1.2	0.8 (0.4)	0
Girl Scout Camp	200.0	1.4	0.7 (0.3)	*
Heisel Park	144.2	29.0	0.2 (0.2)	*
Highbank Metro Park	234.6	0.9	0.0 (0.0)	*
Innis Park	69.2	7.6	0.0 (0.0)	*
Kilbourne	105.8	0.4	1.3 (0.3)	1
Klondike Road	88.5	0.8	0.0 (0.0)	*
Lockbourne Park	255.8	2.7	0.0 (0.0)	*
Lou Berliner Park	155.8	53.0	0.5 (0.2)	*
N. Olentangy Parkland	101.9	29.5	0.0 (0.0)	*
North Galena	134.6	0.1	3.7 (0.4)	3
Old 3 C Road	125.0	4.6	1.3 (0.7)	*
OSU Wetland	86.5	49.1	0.0 (0.0)	*
Prairie Oaks Metro Park	148.1	1.0	1.2 (0.4)	0
Prindle	157.7	0.1	0.3 (0.2)	*
Redbank Road	279.4	0.8	0.0 (0.0)	*
Rocky Creek	150.0	2.7	0.0 (0.0)	*
Rush Run Park	150.0	21.0	0.0 (0.0)	*
Smith Farm Metro Park	144.2	14.9	0.0 (0.0)	*
South Galena	163.5	0.1	0.0 (0.0)	*
Sunbury	128.9	11.9	2.3 (0.4)	4
The Nature Conservancy	292.3	1.2	2.3 (0.4)	*
Westfall Park	55.8	0.5	0.0 (0.0)	*
Whetstone Park	153.9	30.6	0.0 (0.0)	*
Whitehall Park	105.8	35.4	0.0 (0.0)	*
Woodside Green Park	103.9	17.8	0.2 (0.2)	0

Table 2.1. Mean forest width and percent urban development within 1 km of 36 riparian forests. Mean number ( $\pm$  SE) of Acadian Flycatchers detected during surveys, averaged over 6 visits (3 per year), 2001 and 2002. Number of Acadian Flycatchers fledged within mapped territories at 11 riparian forests in central Ohio, 2001 and 2002. If a territory had  $\geq 1$  female (in 3 cases), I averaged productivity across females. \*Denotes no value for that site.

Variable	Territory Scale Mean (SE)	Stand Scale Mean (SE)
Large trees <sup>1</sup>	2.81 (0.85)	3.32 (1.30)
Small trees <sup>2</sup>	11.19 (1.79)	13.05 (1.95)
Snags <sup>3</sup>	1.49 (0.31)	1.84 (0.46)
Vegetation density <sup>4</sup>	1.02 (0.22)	2.57 (0.39)
Arthropod biomass <sup>5</sup>	0.09 (0.02)	0.06 (0.01)

<sup>1</sup> Large trees, number of trees with diameter breast height (dbh) > 38.0 cm.

<sup>2</sup> Small trees, number of small trees with dbh 8.0 – 23.0 cm.

<sup>3</sup> Snags, number of snags with dbh > 12.0 cm.

<sup>4</sup> Vegetation density, number of vegetation hits 0.5 - 3.0 m.

<sup>5</sup> Arthropod biomass, understory arthropod biomass (g).

Table 2.2. Mean and SE values for vegetation and arthropod biomass variables measured at the territory and stand scale within riparian forests in central Ohio, 2001 and 2002. Note that both vegetation density and arthropod biomass values at the territory scale differed > 1 SE from values at the stand scale.

Model <sup>a</sup>	K <sup>b</sup>	AIC <sub>c</sub> <sup>c</sup>	$\Delta_i$ <sup>d</sup>	$\omega_i$ <sup>e</sup>
Null model	2	44.84	0.00	0.47
Large trees	3	47.18	2.35	0.15
Snags	3	47.28	2.44	0.14
Small trees	3	47.45	2.61	0.13
Vegetation density	3	47.56	2.72	0.12
Arthropod biomass	3	47.63	2.79	0.12

<sup>a</sup> Potential factors affecting relative abundance of Acadian Flycatchers at the stand scale: Null model, includes no explanatory variables; Large trees, number of trees with diameter breast height (dbh) > 38.0 cm; Snag, number of snags with dbh > 12.0 cm; Small trees, number of trees with dbh 8.0 – 23.0 cm; Vegetation density, number of vegetation hits 0.5-3.0 m; Arthropod biomass, understory arthropod biomass (g).

<sup>b</sup> Number of parameters in model, includes the intercept, and dispersion parameter.

<sup>c</sup> Corrected Akaike's information criterion adjusted for small sample size relative to number of parameters.

<sup>d</sup> Delta AIC<sub>c</sub> indicating difference in AIC<sub>c</sub> value from that of the best model.

<sup>e</sup> Akaike weight indicating relative support for the model.

Table 2.3. Comparison of candidate models at the stand scale (3-5 ha) describing numbers of Acadian Flycatchers in riparian forests in central Ohio, 2001-2002. Models are ranked according to AIC<sub>c</sub> and delta AIC<sub>c</sub> ( $\Delta_i$ ) values where best-supported models have smaller  $\Delta_i$  values and larger Akaike weights ( $\omega_i$ ).

Model <sup>a</sup>	K <sup>b</sup>	AIC <sub>c</sub> <sup>c</sup>	$\Delta_i$ <sup>d</sup>	$\omega_i$ <sup>e</sup>
Urban	3	63.15	0.00	0.76
Urban Width	4	66.22	3.07	0.16
Width	3	68.84	5.69	0.04
Urban Width	5	69.46	6.31	0.03

<sup>a</sup> Factors affecting numbers of Acadian Flycatchers at the landscape scale: Urban is the percent urban within 1 km of each site; Width is mean riparian forest width (m) at each site. Vertical bar represents a full model that includes main effects and interactions between variables.

<sup>b</sup> Number of parameters in model, includes the intercept and dispersion parameter.

<sup>c</sup> Corrected Akaike's information criterion adjusted for small sample size relative to number of parameters.

<sup>d</sup> Delta AIC<sub>c</sub> indicating difference in AIC<sub>c</sub> value from that of the best model.

<sup>e</sup> Akaike weight indicating relative support for the model.

Table 2.4. Comparison of candidate models at the landscape scale (314 ha) describing numbers of Acadian Flycatchers in 36 riparian forests in central Ohio, 2001-2002.

Model <sup>a</sup>	K <sup>b</sup>	AIC <sub>c</sub> <sup>c</sup>	$\Delta_i$ <sup>d</sup>	$\omega_i$ <sup>e</sup>
Urban	3	24.09	0.00	0.46
Width	3	24.53	0.44	0.37
Urban Width	4	26.45	2.36	0.14
Urban Width	5	29.87	5.34	0.03

<sup>a</sup> Factors affecting productivity of Acadian Flycatchers at the landscape scale: Urban is the percent urban within 1 km of each site; Width is mean riparian forest width (m) at each site. Vertical bar represents a full model that includes main effects and interactions between variables.

<sup>b</sup> Number of parameters in model, includes the intercept and dispersion parameter.

<sup>c</sup> Corrected Akaike's information criterion adjusted for small sample size relative to number of parameters..

<sup>d</sup> Delta AIC<sub>c</sub> indicating difference in AIC<sub>c</sub> value from that of the best model.

<sup>e</sup> Akaike weight indicating relative support for the model.

Table 2.5. Comparison of candidate models at the landscape scale (314 ha) for productivity of Acadian Flycatchers in 11 riparian forests in central Ohio, 2001 and 2002.



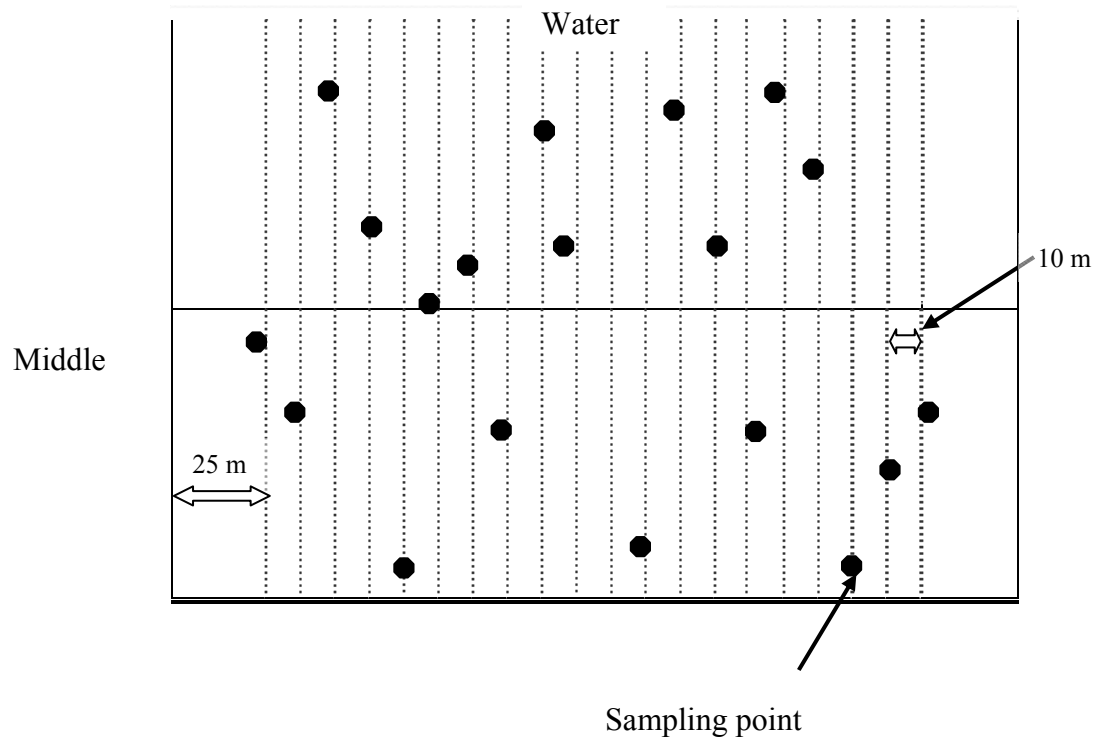
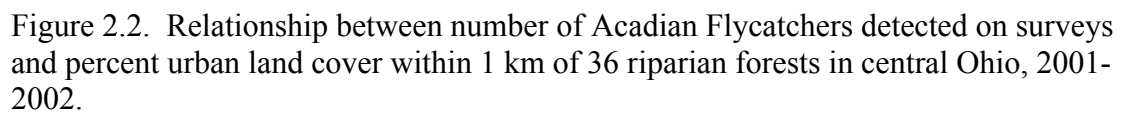


Figure 2.1. Arthropod sampling scheme used within 22 riparian forest stands in central Ohio, 2001 and 2002. Each site (250-m long by 40-m wide) was divided in half with a transect running down the middle. Starting 25 m from either end of the transect, understory arthropods were collected at 10 meter intervals (i.e., 20 sampling points per site). At each interval, a point was chosen perpendicular to the transect at a random distance from the transect.



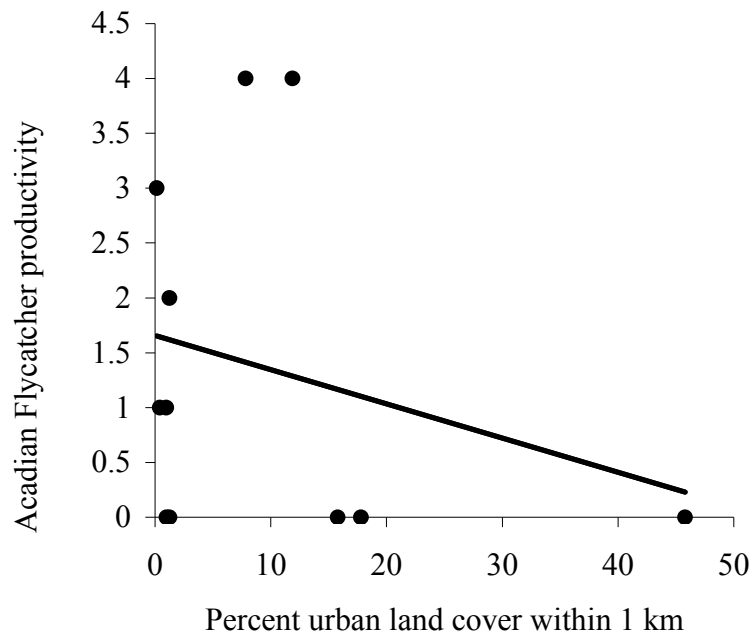


Figure 2.3. Relationship between Acadian Flycatcher productivity and percent urban land cover within 1 km of 11 riparian forests in central Ohio, 2001 and 2002. Productivity was measured as number of fledglings produced per female.

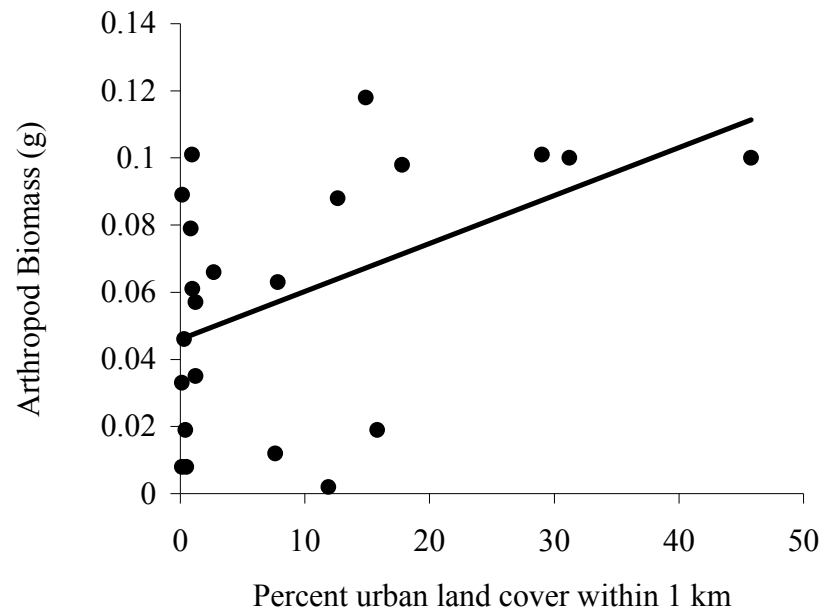


Figure 2.4. Relationship between arthropod biomass and percent urban land cover in the landscape across 22 riparian forests in central Ohio, 2001 and 2002.

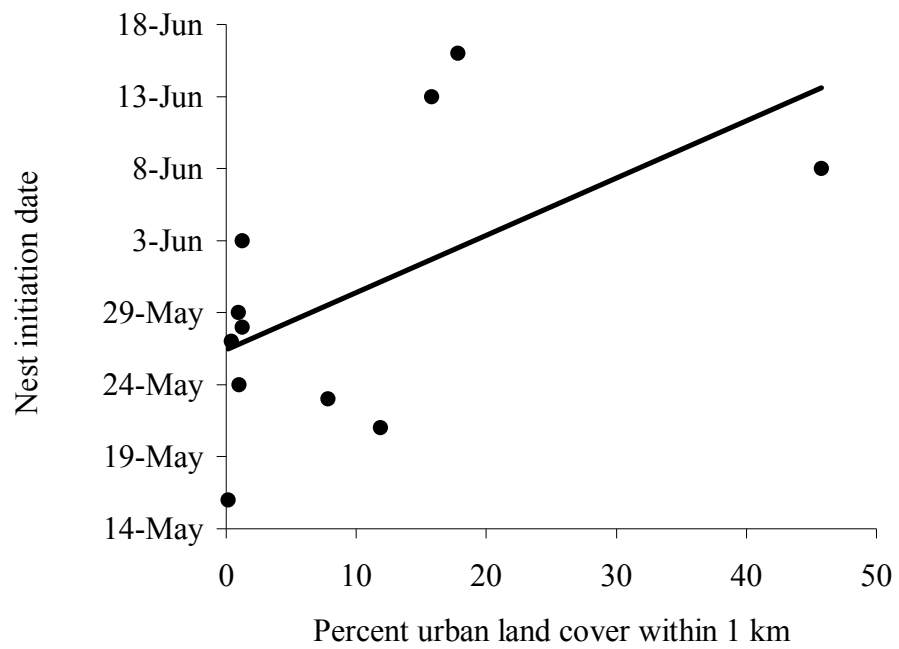


Figure 2.5. Relationship between nest initiation date and percent urban land cover within 1 km for 11 riparian forests in central Ohio, 2001 and 2002.

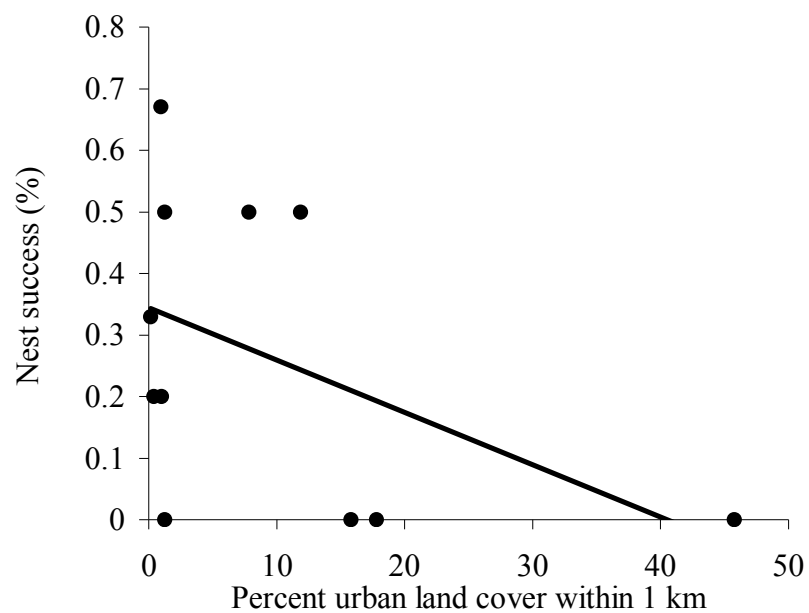
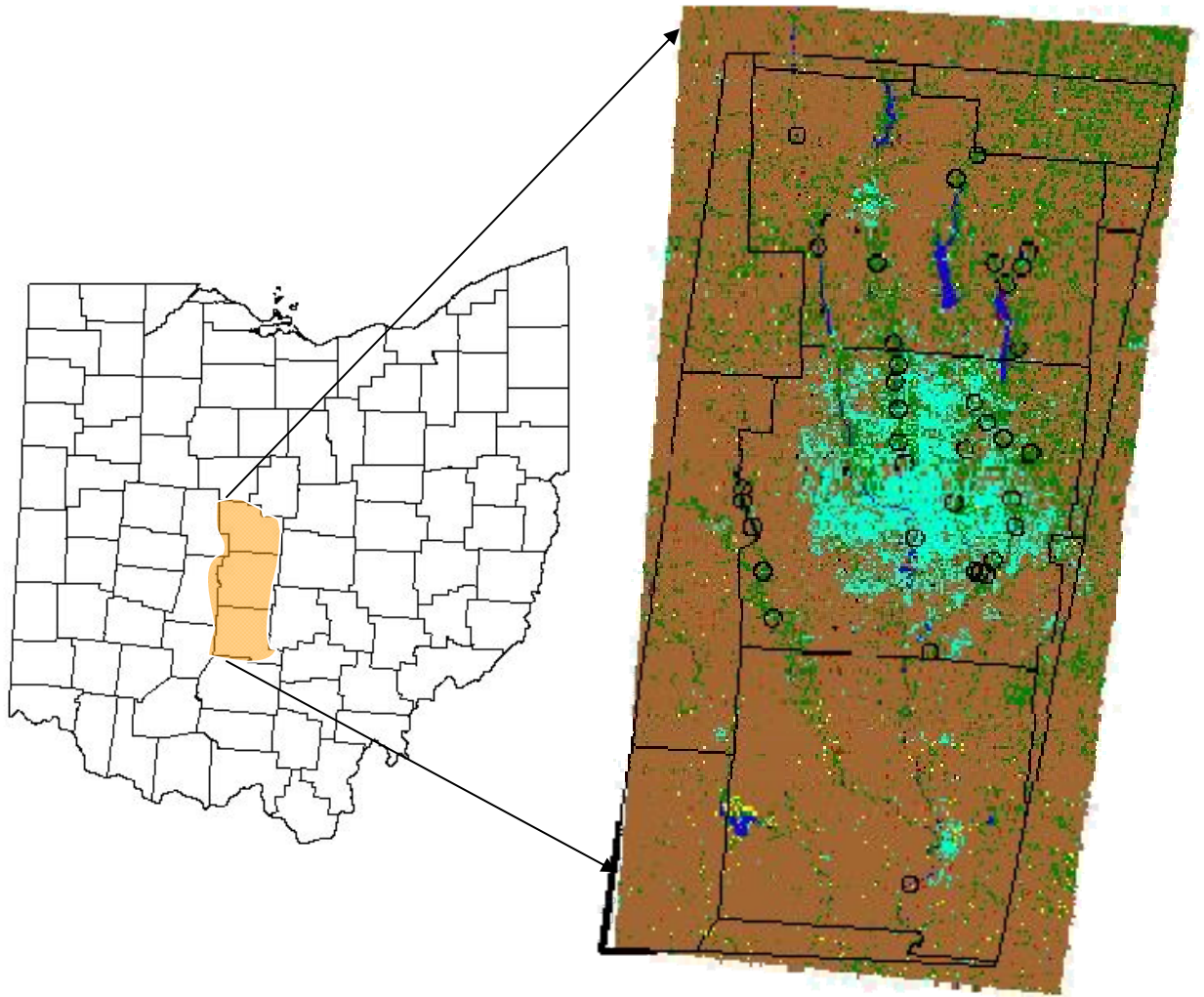


Figure 2.6. Relationship between Acadian Flycatcher nest success (%) and percent urban land cover in the landscape for 11 riparian forest sites in central Ohio, 2001 and 2002.

APPENDIX A. Location of riparian forest study sites in Delaware, Franklin, and Pickaway counties, Ohio, USA.



APPENDIX B. Locations and associated waterways for 36 riparian forest sites surveyed in central Ohio, USA, 2001-2002.

Site	River/Creek	Latitude	Longitude
3 Creeks Metro Park	Blacklick Creek	39N 52' 55"	82W 54' 32"
Bexley Park	Alum Creek	39N 58' 17"	82W 56' 55"
Big Walnut Park	Big Walnut Creek	39N 56' 52"	82W 51' 33"
Camp Mary Orton	Olentangy River	40N 07' 16"	83W 01' 58"
Casto Park	Alum Creek	40N 05' 00"	82W 55' 26"
Chapman Road	Olentangy River	40N 13' 57"	83W 03' 51"
Cherrybottom Park	Big Walnut Creek	40N 03' 44"	82W 54' 16"
Darby Public Hunting	Big Darby Creek	39N 50' 39"	83W 12' 08"
Elk Run Park	Big Walnut Creek	39N 53' 48"	82W 53' 59"
Galena	Big Walnut Creek	40N 12' 51"	82W 52' 50"
Gardner Road	Big Darby Creek	39N 53' 39"	83W 13' 04"
Girl Scout Camp	Big Darby Creek	39N 58' 10"	83W 14' 54"
Heisel Park	Big Walnut Creek	39N 54' 40"	82W 53' 27"
Highbank Metro Park	Olentangy River	40N 08' 47"	83W 02' 18"
Innis Park	Alum Creek	40N 01' 58"	82W 56' 11"
Kilbourne	Alum Creek	40N 19' 40"	82W 57' 31"
Klondike Road	Scioto River	40N 14' 56"	83W 08' 57"
Lockbourne Park	Big Walnut Creek	39N 48' 31"	82W 58' 37"
Lou Berliner Park	Scioto River	39N 56' 03"	83W 00' 14"
N. Olentangy Parkland	Olentangy River	40N 06' 11"	83W 02' 12"
North Galena	Alum Creek	40N 21' 14"	82W 55' 36"
Old 3 C Road	Big Walnut Creek	40N 15' 02"	80W 50' 54"
OSU Wetland	Olentangy River	40N 01' 01"	83W 01' 13"
Prairie Oaks Metro Park	Big Darby Creek	39N 59' 03"	83W 14' 56"
Prindle	Scioto River	40N 22' 15"	83W 11' 10"
Redbank Road	Hoover Reservoir	40N 08' 32"	82W 51' 34"
Rocky Creek	Rocky Creek	40N 01' 50"	82W 50' 33"
Rush Run Park	Olentangy River	40N 04' 28"	83W 01' 53"
Smith Farm Metro Park	Alum Creek	39N 53' 59"	82W 55' 07"
South Galena	Little Walnut Creek	40N 14' 08"	82W 53' 43"
Sunbury	Big Walnut Creek	40N 13' 56"	82W 51' 26"
The Nature Conservancy Reserve	Big Darby Creek	39N 56' 27"	83W 14' 01"
Westfall Park	Scioto River	39N 33' 22"	82W 59' 51"
Whetstone Park	Olentangy River	40N 02' 18"	83W 01' 49"
Whitehall Park	Big Walnut Creek	39N 58' 47"	82W 51' 52"
Woodside Green Park	Big Walnut Creek	40N 02' 41"	82W 52' 49"



APPENDIX C. Mean forest width and percent urban development within 1 km of 36 riparian forest sites in central Ohio. Mean number ( $\pm$  SE) of potential nest predators detected during surveys. Numbers (#/ha) were averaged over 6 visits (3 per year), 2001 and 2002.

Site	Mean Forest Width (m)	Urban (%)	Number of Predators (SE)
3 Creeks Metro Park	132.7	7.8	1.8 (0.7)
Bexley Park	132.6	41.6	4.3 (0.9)
Big Walnut Park	115.4	45.8	2.8 (0.8)
Camp Mary Orton	565.4	15.8	2.8 (0.9)
Casto Park	201.9	47.2	4.3 (0.8)
Chapman Road	86.5	0.3	3.8 (1.1)
Cherrybottom Park	165.4	31.2	2.0 (0.9)
Darby Public Hunting	194.2	1.0	1.7 (0.5)
Elk Run Park	167.3	12.6	3.8 (0.5)
Galena	276.9	1.2	2.5 (0.9)
Gardner Road	125.0	1.2	1.3 (0.5)
Girl Scout Camp	200.0	1.4	1.8 (0.7)
Heisel Park	144.2	29.0	5.0 (1.8)
Highbank Metro Park	234.6	0.9	2.3 (1.1)
Innis Park	69.2	7.6	3.3 (0.4)
Kilbourne	105.8	0.4	1.2 (0.7)
Klondike Road	88.5	0.8	2.0 (0.8)
Lockbourne Park	255.8	2.7	2.2 (0.8)
Lou Berliner Park	155.8	53.0	4.2 (1.2)
N. Olentangy Parkland	101.9	29.5	9.7 (1.6)
North Galena	134.6	0.1	1.3 (0.7)
Old 3 C Road	125.0	4.6	4.0 (0.6)
OSU Wetland	86.5	49.1	11.5 (1.9)
Prairie Oaks Metro Park	148.1	1.0	2.7 (0.8)
Prindle	157.7	0.1	1.7 (0.8)
Redbank Road	279.4	0.8	2.8 (1.4)
Rocky Creek	150.0	2.7	1.7 (1.0)
Rush Run Park	150.0	21.0	4.5 (1.3)
Smith Farm Metro Park	144.2	14.9	2.5 (1.0)
South Galena	163.5	0.1	1.5 (0.7)
Sunbury	128.9	11.9	2.0 (0.7)
The Nature Conservancy Reserve	292.3	1.2	4.0 (1.5)
Westfall Park	55.8	0.5	4.7 (1.4)
Whetstone Park	153.9	30.6	5.3 (1.2)
Whitehall Park	105.8	35.4	2.8 (0.9)
Woodside Green Park	103.9	17.8	4.5 (1.1)

APPENDIX D. In 2001, nest fate for twelve banded Acadian Flycatcher males was monitored at 9 riparian forest sites in central Ohio. These sites were revisited in the 2002 breeding season to determine if the same males returned. Note that males with successful nests in 2001 had greater site fidelity than males with unsuccessful nests (i.e., 67% return rate versus 16% return rate). Number of females per territory for banded and territory mapped males. Territory size of mapped male Acadian Flycatchers in 2001 and 2002. \*Denotes no value for that site.

Male Band Number	Site	Nest success in 2001	Return in 2002	Number of females per territory	Territory size (ha)
1830-14604	3 Creeks Metro Park	Y	Y	1	0.9
1830-14607	Big Walnut Park	N	N	0	*
1830-14609	Big Walnut Park	Y	Y	1	1.0
1830-14606	Camp Mary Orton	N	N	1	1.2
1830-14601	Darby Public Hunting	Y	Y	3	1.8
1830-14596	Galena	Y	N	2	0.7
1830-14608	Galena	N	N	0	*
1830-14602	Gardner Road	N	N	1	0.6
1830-14615	Kilbourne	*	*	1	0.8
1830-14597	North Galena	Y	Y	2	*
1830-14605	North Galena	Y	N	1	0.7
1830-14603	Prairie Oaks Metro Park	N	Y	2	1.4
1830-14613	Sunbury	*	*	1	0.8
1830-14611	Woodside Green Park	N	N	1	1.4

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