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INFLUENCE OF EXURBAN NEIGHBORHOODS ON RIPARIAN VEGETATION AND STREAM SALAMANDERS IN THE SOUTHERN APPALACHIAN MOUNTAINS

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INFLUENCE OF EXURBAN NEIGHBORHOODS ON RIPARIAN VEGETATION
AND STREAM SALAMANDERS IN THE SOUTHERN APPALACHIAN
MOUNTAINS

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ABSTRACT

The southern Appalachian Mountains have experienced large population growth and a change in land use in the past 30 years. The majority of development has been low density, suburban land, known as exurban development. Lotic Systems and riparian areas are severely degraded by conversion from urban to rural land uses. The long-term effects of exurbanization on riparian vegetative communities and stream salamanders in the southern Appalachian Mountains are not well known. We sought to determine if vegetative community composition and structure change with time since neighborhood development or with the amount of impervious surface within the watershed. In order to determine the temporal influence of exurban housing on salamanders and riparian vegetation we sampled 27 streams and riparian areas in watersheds containing exurban neighborhoods ranging in age from four to forty-four years, along with eight forested streams, over the course of two field seasons. Watershed scale variables such as neighborhood age and impervious surface cover did not influence the aspects of riparian vegetation community we measured, and usually did not influence salamanders. Local habitat variables offered better predictions of vegetation community metrics as well as salamander occupancy and abundance. Exurban neighborhoods and their landowners may have the potential to manage for riparian vegetation and salamanders through the use of maintained stream buffer zones along the entire length of the stream and reduction in road salts and impervious surfaces.

DEDICATION

I would like to dedicate this thesis to my parents, for their never ending support in my six and a half years as a student. For their continual encouragement and inspiration in my childhood and for helping to lead me towards my profession as a wildlife biologist. I am forever thankful for my father getting me involved in the outdoors and the boy scouts; and for my mother encouraging the pursuit of scientific exploration.

ACKNOWLEDGMENTS

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CHAPTER ONE

THE INFLUENCE OF EXURBAN NEIGHBORHOOD AGE ON RIPARIAN

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INTRODUCTION

The United States is experiencing increases in population size and urbanization. Between 2000 and 2010 the population of the United States increased by 9.7%, and 83.7% of the population resides in urban areas (Mackun and Wilson 2011). Land use practices are shifting and forests are being converted into residential land (Wear and Bolstad 1998). The majority of development has been low density, suburban land, especially in the Southeast (Mcdonald et al. 2010), where population growth was 16.6% between 2000 and 2010 (Pollard and Jacobson 2011). Such development is often termed exurbanization because the proportion of impervious surface within the watershed and housing density are typically lower than thresholds associated with urban environments, yet higher than rural regions (Theobald 2004). Low density developments are projected to increase in future decades (Wear and Bolstad 1998, Theobald 2010), which may pose a serious threat to many species and ecosystems. By definition exurban developments are often near highly biodiverse areas (Gagne and Fahrid 2010).

Stream systems and their associated riparian zones are especially susceptible to changes following exurbanization. Impervious surfaces associated with buildings and roads alter the rates and movement of water flow such that less percolation of water occurs into the soil, leading to a reduction in base flow, but an increase in flow during

rain events (Paul and Meyer 2001). These hydrologic changes not only modify stream and riparian morphology, but can also reduce soil moisture, water table depth, organic matter, root density, and alter soil pH in adjacent riparian areas (Gift et al. 2010). Changes in soil moisture reduce a riparian area's ability to buffer the influences from the upland areas. For example, Gold et al. (2001) found that water nitrate-N removal rates were reduced in more hydric soils. Ultimately, both direct land use changes to riparian zones, altered hydrology, and modification of soil characteristics all influence riparian vegetation.

Riparian vegetation is integral in determining both composition and function of stream ecosystems (Warner and Hendrix 1984) and it serves as a buffer zone between upland areas and streams (Hill 1996, Lowrance 1998). Streamside vegetation helps maintain stream temperature, provides woody debris for habitat along and within the stream, and assists in the uptake of NO_3^- from shallow groundwater (Sweeney 1992, Tabacchi et al. 2002). Riparian vegetation also stabilizes banks and provides cover for many species of wildlife. Plants provide detritus material within the stream, creating both a food source and habitat for aquatic organisms (Warner and Hendrix 1984). Urbanization has greatly reduced vegetation at a global scale (McKinney 2002), and riparian forests are particularly sensitive to land use change (Malanson 1993). Urbanization directly alters vegetative community composition and structure through replacement of vegetation by urban infrastructure and fragmentation. Species diversity, tree basal area, and native plant density have been shown to decrease near urban areas (Porter et al. 2001, Moffatt et al. 2004). Loss of canopy cover can increase algal growth,

thereby changing low-order stream systems from allochtonous- to autochthonous-based systems (Doi et al. 2007, Hall et al. 2000, and Sobczak et al. 2002). A decrease in detrital inputs may yield lower macroinvertebrate biomass or altered macroinvertebrate community composition (Sobczak et al. 2002), which can have implications for higher trophic levels (Johnson and Wallace 2005). Furthermore, stream temperatures increase with canopy loss, which alters habitat suitability for many organisms (Bozinovic et al. 2011). Water quality and quantity are also altered by a loss in stream bank vegetation, which reduces sediment loading in streams (Osborne and Kovacic 2006).

Urbanization causes a shift in vegetative communities and reduces native plant diversity while increasing the number of exotic and invasive species (Burton et al. 2005, Burton et al. 2008, King and Buckney 2001, McKinney 2001, McKinney 2002, Warren et al. 2015). A study by Loewenstein and Loewenstein (2005) found significantly more exotic plant species at urban sites along a rural-to-urban gradient in the Piedmont ecoregion of Georgia. Pennington et al. (2010) identified similar trends in Ohio, but also found an increase in early successional native plant species in urbanized riparian areas. Brush et al. (1980) showed that there were almost twice as many upland plant species in urban floodplains relative to non-urbanized floodplains, indicating higher soil moisture content in urbanized areas from irrigation.

Pennington et al. (2010) argued that previous studies on urbanization and stream response are too broad in scope and need to focus on local-scale variables like riparian vegetation. The authors showed that local vegetative community changed in response to urbanization, and that differences in these changes lead to different levels of influence of

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urbanization on streams. The authors make a final argument that future conservation efforts in the face of urbanization should focus on maintaining wide riparian forests and limiting impervious surface development within riparian areas. While other studies have offered the somewhat contrasting view that watershed-scale conservation is necessary to protect stream organisms (Willson and Dorcas 2003), the fact remains that Pennington et al. (2010) suggest an important hypothesis that may apply to some aspects of stream ecosystems. Increased riparian forest buffers may then lead to reduced exotic plant invasions and maintenance of hydrological function in riparian areas, even if they do not protect all stream species from declines.

Riparian vegetation response to exurban development may be similar to areas of timber management (a common land use in Southern Appalachia) because both land uses entail the removal of large quantities of forest cover. The life history traits of herbaceous understory plants vary, and as a result so does their recovery post timber harvest. Duffy and Meier (2003) compared herbaceous understory of old growth forests to secondary forests ranging in 45 to 87 years since clear-cutting. They found that neither cover nor species richness increased with age in these secondary forests. The authors argue that 87 years is insufficient time for understories to recover and that these species will never recover to primary forest states due to climatic differences today relative to when the old growth forests were established. Duffy and Meier (2003) also argue that a near complete recovery to pre-disturbance conditions will not occur until large trees have had time to grow, and then fall. This slow recovery with time may be dependent on amount and proximity of propagules, and it has been argued that recovery can actually occur over

time and that Duffy and Meier failed to fairly represent pre disturbance conditions in their chronosequence study (Johnson et al. 1993).

Buffers may offer one solution to increased recovery time, as they have proven to be an effective mitigation strategy for timber harvests (Clinton 2011, Gomi 2006). Clinton (2011) studied the influence of varying buffer width on stream water quality following upland forest management activities in headwater catchments and found that the greatest alteration to water quality occurred in the no buffer site, while sites with a buffer showed little or no change in water chemistry. They concluded that a buffer width of at least ten meters led to minimal changes in water quality following a timber harvest. While the initial effects of exurban development may be similar to timber harvest, the longer-term effects of these two land uses are likely to diverge, because impervious surface represents a continual stressor in exurban areas along with other aspects of exurban development like pesticide and fertilizer use.

While the response of riparian vegetation to timber harvest has been evaluated at various time steps following disturbance, no such knowledge exists for the same communities in the context of exurban housing developments (Pennington et al. 2010). It is unknown if riparian communities surrounded by exurban housing developments will undergo a process of recovery with time toward pre-disturbance conditions, or if they will remain altered in the long-term. To address this data gap we evaluated the influence of impervious surface, neighborhood age, and other landscape-scale variables on several measures of the riparian plant community. We also evaluated the ability of surrogate

measures for local habitat management to predict plant species richness, diversity, and the number of non-native species.

METHODS

Study Area

Our study sites were within the Southern Blue Ridge Ecoregion, which spans over 3804045 hectares and covers sections of Georgia, South Carolina, North Carolina, Tennessee, and Virginia. The mountains are currently between 450 and 2040 m in elevation. There are over 400 species of plants and animals endemic to this region, more endemics than any other North American ecoregion (The Nature Conservancy and Southern Appalachian Forest Coalition 2000). The southern Appalachians receive the highest level of rainfall in the United States east of the Cascades, and the climate of these mountains ranges from temperate to boreal.

Site Selection

To select focal streams we evaluated 2014 aerial images from the region to identify watersheds with exurban development. Within these watersheds, we used ARCGIS 10.1 (ESRI, Redlands, CA) to overlay a high resolution stream layer from the National Hydrography Dataset and a tax parcel data layer derived from Sevier County, TN and Macon County, NC. From these overlays we identified 80 potential locations. We attempted to contact the property owners along each stream by phone or in person. Once permission was obtained to access the property we traveled to each site in an attempt to standardize stream size and development to the extent possible. Following ground

validation we were left with 27 first- or second-order exurban streams across the two

counties in Western North Carolina and Eastern Tennessee. We selected eight additional

streams that contained no impervious surface. Four of these forested sites were located

within the Coweeta Hydrologic Laboratory property in Otto, North Carolina, and four

were within Walker Valley in the Great Smoky Mountain National Park, TN (Fig. 1.1).

All forested sites were presumed to have been logged, but the harvest was greater than 75

years ago.
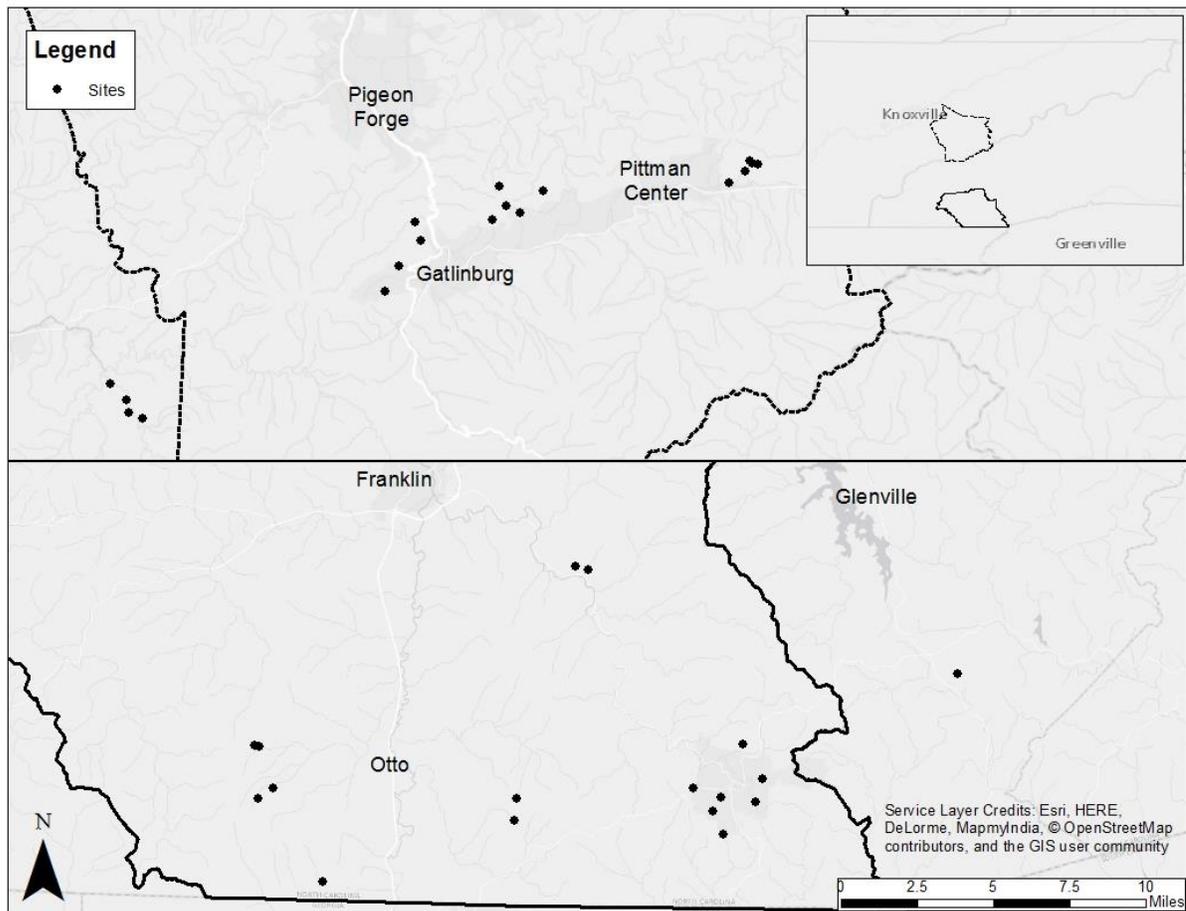
Figure 1.1. Sample locations were located in Sevier County, Tennessee (dashed) and
Macon County, North Carolina (solid).

To calculate the age of development in each exurban watershed, we used tax parcel information. We extracted the age of each individual structure within the study area, and averaged those ages across all buildings in the watershed. Exurban housing ranged in age from four to forty-two years (mean = 25.99 yrs) across the 27 watersheds with development. We also calculated impervious surface coverage for each watershed by obtaining 2014 leaf off aerial imagery (0.65 meter resolution) from the counties containing our study areas. This was done by hand-delineating polygons around all impervious surfaces and calculating the percent of the watershed they covered. We calculated distance to impervious surface using the "near" tool in ARCGIS 10.1, measuring the distance from stream sample plots to the nearest impervious surface polygon.

Field Methods

We established a 45-m transect along the length of each stream and established vegetation plots in the summer of 2014. Transects were selected by maximizing the amount of exurban development within the watershed, without allowing for other forms of land use, but was limited by where we could get access to private property. These transects were broken into three five meter sections, each ten meters apart. Along each of the five meter sections, we measured 10 m from the bank of the stream to establish a 50 m^2 plot (Fig. 1.2). The plot was measured on the right side of the first section, the left side of the second section, and the right side of the third section. Within each plot we identified all vegetation to species. If an identification could not be made on site, we took photos and identified them later. If a positive ID could not be made we categorized gave

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the plant a number and used that if we found it at another site. This only occurred on a handful of occasions. We counted all trees within a plot; however we only recorded incidence data for herbaceous and shrubby vegetation. We measured percent canopy cover three times in the middle of the stream using a densiometer. We estimated percentage of ground covered by coarse woody debris, vegetation, and bare ground (defined as rock or soil not covered by vegetation) within each plot to the nearest 5%. We considered any fallen limb or tree larger than 10 cm in diameter to be coarse woody debris. We recorded basal area within an acre (40.47 dam^2) of each section using a 10BAF basal area prism.
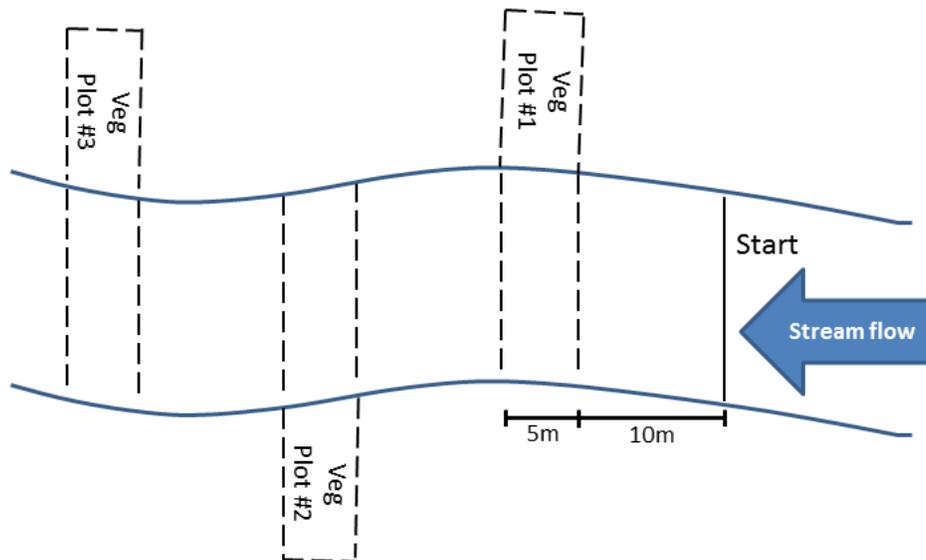
Figure 1.2. A graphic of the sample transect and vegetation plots for each field site.

Analysis

We evaluated both species richness and Shannon diversity between forested sites (no impervious surfaces in the watershed) and exurban sites (1-17% watershed impervious surface). Both metrics were calculated using the Chao 1 and 2 indices (for

abundance and incidence data, respectively) as calculated by the iNEXT package (Hsieh et al. 2014) in Program R (R Core Team 2013). Using the iNEXT package we created species accumulation curves for both the number of sites sampled and the number of individuals sampled for the forested and the exurban sites. Because of the differences in the number of sites, exurban species accumulation was rarefied to only eight sites, which represented the total number of forested sites. To assess differences in vegetative community composition as a function of neighborhood age and percent impervious surface, we used a canonical correspondence analysis (CCA). This type of ordination analysis is appropriate when the goal is to understand the structure of community data in the context of a specific set of environmental variables (McCune and Grace 2002). We examined all vegetation (using only incidence data) and tree species (using count data) in two separate CCA analyses. We conducted these analyses using only data from sites with development, since neighborhood age was not applicable to fully forested sites. We used the vegan package (Oksanen et al. 2015) in Program R (R Core Team 2013) to run the CCAs.

We used multivariate multiple regression to examine the influence of percent impervious surface, neighborhood age, and distance to impervious surface on a suite of uncorrelated (R<0.75) plant-related response variables. Specifically, we examined the influence of our selected predictors on basal area, canopy cover, vegetative cover, coarse woody debris (CWD), Shannon-Wiener Diversity (for trees), tree species richness, and total plant species richness. Because almost all of our sites have a relatively high proportion of forested area within the watershed, we also wanted to examine the

influence of habitat structure within the riparian zone on vegetation. We used basal area, canopy cover, and coarse woody debris to quantify differences in forest structure at the local scale. These variables indicate the level of disturbance to the riparian area around a site. Sites with higher basal area, canopy cover, and CWD would be considered less disturbed, or a longer period since the previous disturbance. We again used multivariate multiple regression to test for relationships between local site characteristics predictors and Shannon diversity of trees, tree species richness, total plant species richness, and exotic species.

We used logistic regression models to examine the influence of hypothesized predictor variables on the presence or absence of selected species. Eastern-Hemlock (*Tsuga candensis*) and yellow-poplar (*Liriodendron tulipifera*) were chosen because they each made up more than 10% of all trees identified. We also evaluated rhododendron (*Rhododendron maximum*) because it was the most common species and indicative of a climax riparian ecosystem in this region (Keever 1953). We evaluated the presence of ericaceous shrubs [rhododendron, mountain laurel (*Kalmia latifolia*), and mountain dog hobble (*Leucothoe fontanesiana*)] based on their importance in these ecosystems in terms of pH regulation and soil nutrients (Monk et al. 1985). We evaluated red maple (*Acer rubrum*) and species not native to the United States because both were present at more than fivefold as many exurban sites relative to forested sites.

RESULTS

Descriptive characteristics

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There were a total of 36 tree species found across all sites. Mean species richness and Shannon diversity were higher in forested sites relative to exurban sites; however, there was considerable overlap in the range of these measures between site categories (Table 1). Sample-based rarefaction (n = 8) revealed no significant difference (95% confidence intervals overlapped) between exurban and forested sites in the accumulation of tree species, based on abundance data (Fig. 1.3a), however, the shape of the accumulation curve suggest that richness in both categories of sites is not fully represented by our sample. Across all sites, we identified 179 species of herbaceous and shrubby plants. Individual-based species accumulation curves for herbaceous plants again showed no significant difference between forested and exurban sites (Fig. 1.3b). Exotic plants were found at 13 out of 27 exurban sites, and at none of the forested sites.

Table 1.1. Mean and range of Shannon index, species evenness, and species richness values for riparian vegetation community data collected from forested and exurban sites within the Blue Ridge Mountain region of North Carolina and Tennessee, U.S.A.

Parameter	Forested		Exurban	
	Mean	Range	Mean	Range
Shannon Index	1.73	1.43-2.19	1.32	0-2.13
Species Richness	7.4	5-11	4.6	0-10
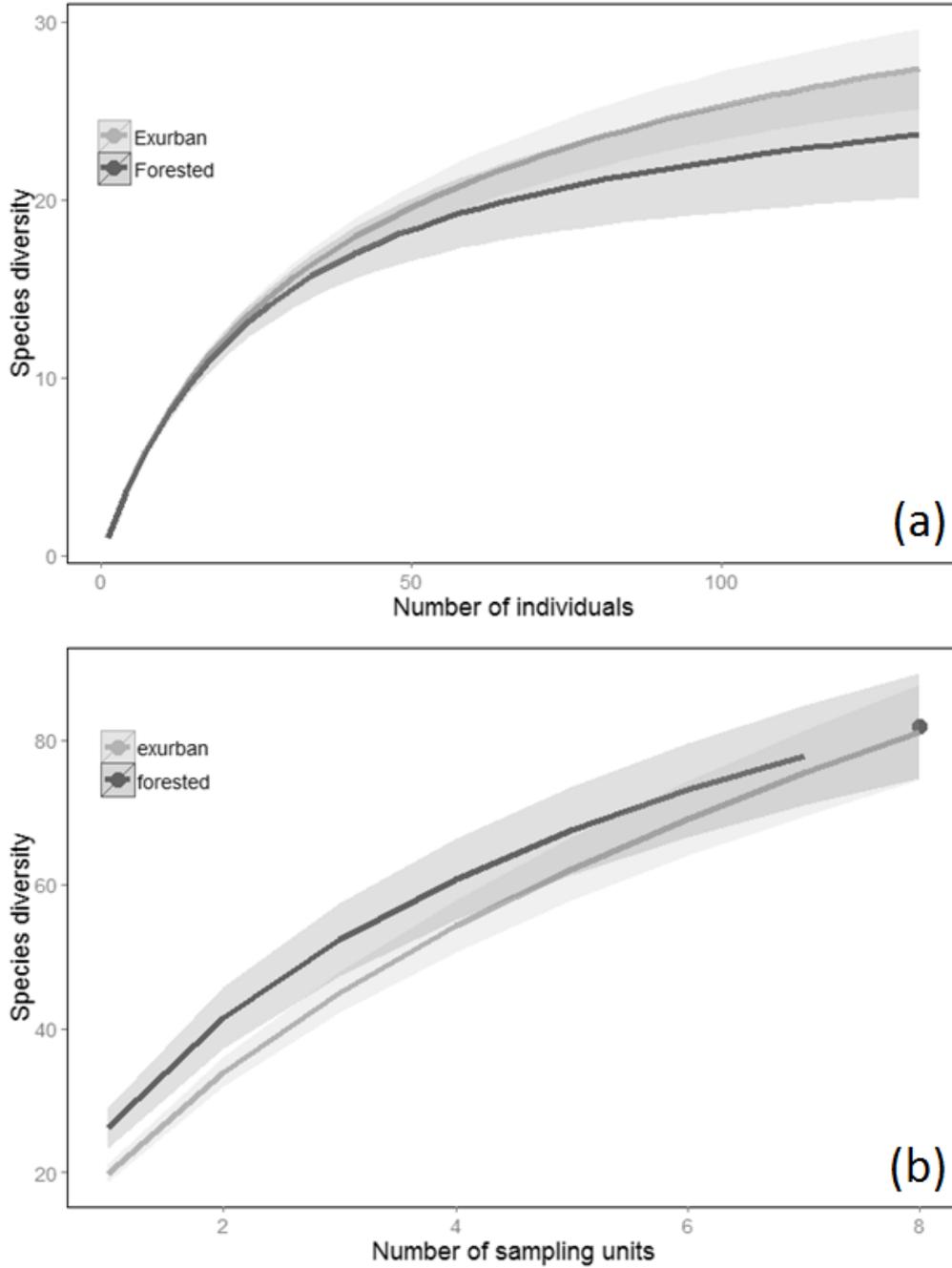
Figure 1.3. Species accumulation curves for abundance data on trees (a) and incidence data on plants (b) for exurban and forested sites for riparian vegetation community data within the Blue Ridge Mountain region of North Carolina and Tennessee, U.S.A.

The most common tree (i.e. found at the greatest number of sites) was the yellow-poplar, occurring at 22 sites. The most frequently counted tree across all sites was the eastern hemlock, occurring at 22 sites. It is of note that of the 67 hemlocks identified, 45 were dead, presumably from the hemlock woolly adelgid (*Adelges tsugae*). The most frequently counted living tree was the yellow-poplar. The most common mid or understory plant across all sites was rhododendron. A total of nine exotic species were found, all at exurban sites. The most common exotic species was Japanese stilt grass (*Microstegium vimineum*).

Vegetation community response to urbanization

In a CCA using only tree abundance data, 4.2% and 3.8% of variance was explained by age and impervious surface, respectively (Fig. 1.4a). When incidence data were used for a CCA with all plants, 5% and 4% of the variance was explained by age and impervious surface respectively (Fig. 1.4b).
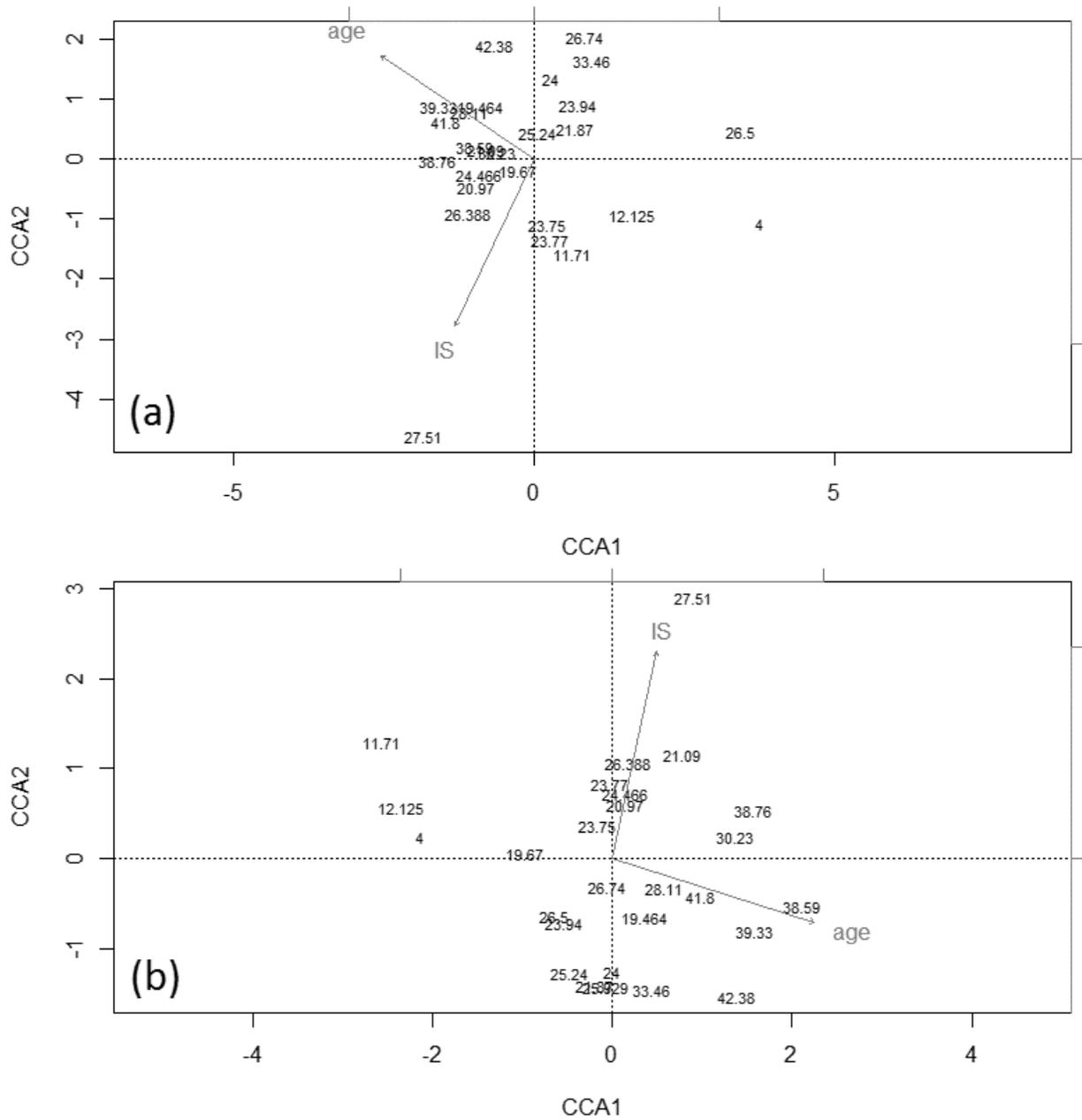
Figure 1.4. Tri-plots from canonical correspondence analyses using impervious surface (IS) and neighborhood age (age) as constraining variables on an ordination of riparian vegetation data from 27 streams in North Carolina and Tennessee, U.S.A. We examined the ability of these variables to explain variation in (a) count data from trees and (b) incidence data for all vegetation. Sites are represented on the plot as the mean value of neighborhood age for that site.

Multivariate multiple regression revealed no significant relationships between neighborhood age, impervious surface, or distance from impervious surface and any of our vegetation community response variables. Multivariate multiple regression using canopy cover and basal area as predictor variables showed that canopy cover was a significant predictor. Because canopy cover was the only significant local site predictor, linear regressions were used to evaluate bivariate relationships with response variables. Canopy cover negatively influenced herbaceous cover ($R^2 = 0.38$, $P < 0.001$) and the number of exotic species ($R^2 = 0.33$, $P < 0.001$), and positively influenced Shannon diversity of trees ($R^2 = 0.26$, $P < 0.001$).

Predictors of presence varied widely across target species. Presence of red maple and yellow-poplar were not significantly related to any of the selected predictors ($P > 0.05$ for all logistic regressions). Higher basal area increased the probability of Eastern hemlock ($P = 0.02$), rhododendron ($P = 0.03$), and ericaceous shrub ($P = 0.01$) presence, and decreased the likelihood that exotic species would be present ($P = 0.02$). Areas of higher canopy cover increased the likelihood of rhododendron [($P = 0.02$ (all sites) and $P = 0.03$ (exurban sites)] and ericaceous shrub presence ($P < 0.01$) and decreased the likelihood that exotics were present ($P = 0.04$ across all sites). For every five percent increase in canopy cover there was a four percent decrease in the likelihood of exotic species presence, and an increase in the percent likelihood of rhododendron presence by ~ 6% at all sites and 7% at exurban sites. Neighborhood age did increase likelihood of rhododendron presence ($P = 0.04$ across exurban sites) and the presence of ericaceous shrubs ($P < 0.01$). For every five year increase in neighborhood age there was a 6.1%

increase in the likelihood of rhododendron presence, which at least for this one species, is a result consistent with recovery toward more forested conditions.

DISCUSSION

When forested and exurban sites were considered categorically, there were no differences between the estimated richness or diversity values for trees or understory vegetation. The mean accumulation curve generated from exurban areas is slightly higher than that from forested streams, which is most likely due to the lower number of individuals sampled in any one forested plot. Trees from forested plots tended to be larger and thus more distantly spaced (on average basal area was 50% higher in forested riparian plots). Further evaluation of exurban sites focused on the continuous variation within this category that as a function of neighborhood age and impervious surface. CCA results suggest that neither age nor the amount of impervious surface structure vegetation communities. Linear regression analyses further revealed all watershed-scale variables to be poor predictors of the selected measures of community response. Other studies have noted a negative relationship between impervious surface and tree species richness (Burton et al. 2008, Moffatt et al. 2004, and Porter et al. 2001). While our data suggested a trend consistent with these findings, none of the results were statistically significant.

Although watershed-scale variables showed no statistically significant effect, canopy cover, a measure of local site conditions, did exhibit significant influences on the riparian vegetative community. Reduced canopy cover predictably led to an increase in the amount of ground cover by understory vegetation, however, exotic species were often

prominent members of the understory community. For example, an exurban site in East TN with little canopy cover had riparian plots that were 100% covered by herbaceous vegetation, in the form of kudzu (*Pueraria lobata*). This observation is likely due to increased light availability (Parendes and Jones 2001, Setterfield et al. 2005, Vidra and Shear 2008, Warren et al. 2015). Increased canopy cover is characteristic of more mature forests; however, our results suggest this variable does not necessarily increase as neighborhoods age.

Our results show that canopy cover was positively correlated with Shannon diversity of trees. This result was not simply a function of increased tree abundance, because basal area did not correlate with Shannon diversity. Higher levels of diversity promote increased primary productivity of plant communities (Nijis and Roy 2003, Wilsey and Potvin 2000). Increasing riparian ecosystem primary production, and consequently the detrital inputs, is very important for southern Appalachian headwater stream ecosystems; they have detritus based food webs (Hall et al. 2000). Reduced canopy cover reduces detritus input (Wallace et al. 1997) and decreases food chain length (Jenkins et al. 1992). Changes in detrital food base are linked to forest cover along the entire length of the stream, not the immediately present forest cover at the site, and can occur from minimal forest cover loss (England and Rosemond 2004).

Predictors of individual species presence or absence varied considerably among species. Exotic plants were only found in exurban neighborhoods, but the probability of finding exotics within exurban neighborhoods was not linked to any of our watershed-scale variables. Hemlock and rhododendron were associated with increasing basal area,

and likely represent species indicative of intact riparian zones. The likelihood of herbaceous ericaceous shrubs also increased with increasing forested riparian area. These species are an integral component of southern Appalachian stream ecosystems and have influences on ecosystems characteristics like soil pH, leaf litter depth, and nutrient retention (Monk et al. 1985). Exotic invasion has been linked directly to local vegetative structure in terms of the amount of canopy cover (Vidra and Shear 2008). Stream buffers to development provide increased basal area and canopy cover, which may encourage the persistence of native and climax community species during and after development of an exurban neighborhood. While higher basal area was associated with a higher probability of hemlock presence, it is unclear which drives which. Hemlock is a shade tolerant tree, and therefore persistence likely increases in areas with high basal area, but a large number of hemlocks obviously contribute to high basal area. In reality a positive feedback loop likely exists between intact forests and hemlock recruitment (Kobe et al. 1995), at least before the introduction of Hemlock Wooly Adelgid (*Adelges tsugae*) that decimated Eastern Hemlock populations which has implications for soil moisture and detritus quality (USDA Forest Service 2005).

Conclusion

Our work suggests that riparian vegetative composition in watersheds containing exurban developments is not driven by the amount of impervious surface (at ranges from 1 – 17%) or the age of the exurban development. Instead, local site variables such as canopy cover and basal area provided the best predictors of exotic species. These local-scale measures can be influenced by riparian management practices. In Macon and

Jackson County, North Carolina there are ordinances requiring 30 foot buffer zones along streams, but from our observations exurban developments allow impervious surface closer to the stream. During construction of a neighborhood, basal area and canopy cover can be reduced from clear-cutting, or land owners may clear vegetation after acquiring the property, then continue to clear through mowing or trimming over time. Furthermore, once the property is privately owned there is little enforcement of buffer regulations. On multiple occasions we met land owners with concerns about the neighbors removing trees along the stream bank, or observed it ourselves. The absence of a correlation between basal area and CWD could also mean that landowners are removing snags and fallen trees. About 65 percent of land in the southern Appalachians is privately owned. This means that cooperation with private land owners is integral to maintaining the biodiversity and function of these ecosystems. Future studies that assess the minimum forest buffer width required to maintain vegetative communities similar to forested sites would provide land owners and neighborhoods more specific target objectives for sustainable management of riparian habitats.

CHAPTER TWO

THE INFLUENCE OF EXURBAN NEIGHBORHOOD AGE ON STREAM

SALAMANDERS IN THE SOUTHERN APPALACHIAN MOUNTAINS

INTRODUCTION

As the global population increases, more people are moving into urban areas and

the size of these areas is increasing (Mackun and Wilson 2011). The population of the

United States increased by 9.7% from 2000 to 2010, with 83.7% of the population

currently residing in urban areas (Mackun and Wilson 2011). Due to land use change,

previously forested or agricultural area, especially forested hillsides, are being converted

to residential land (Wear and Bolstad 1998). The majority of this development has been

low density, suburban land, particularly in the Southeast (Mcdonald et al. 2010), where

population growth was 16.6% between 2000 and 2010 (Pollard and Jacobson 2011). This

form of residential development is termed exurbanization because the proportion of

impervious surface within the watershed and housing density are typically lower than

thresholds associated with urban environments, yet higher than rural regions (Theobald

2004). Exurbanization is projected to increase in the Southern Appalachian Mountains in

future decades (Wear and Bolstad 1998, Theobald 2010), which may jeopardize many

species and ecosystems. By definition exurban developments are usually near highly

biodiverse areas (Gagne and Fahrig 2010).

Relative to forested streams, urban streams exhibit a less stable hydrograph,

elevated nutrients and contaminant concentrations, altered morphology, reduced

biodiversity, and reduced nutrient uptake that collectively create a diagnosis of the "urban

stream syndrome" (Walsh et a. 2005). Even at relatively low levels of development (<5% impervious surface) within a watershed, streams can exhibit significant differences in width, depth, and particle size of sediment (Price and Leigh 2006). It has also been found that long term, low intensity urbanization can yield levels of degradation similar to acute, major urbanization (Weaver and Garman 1994).

Exurbanization may have effects on streams and stream biota that are different from other forms of land use. Urbanization often results in infilling of developed landscapes (Wu et al. 2010) and is usually due to the private interests of relatively few land owners (McDonald et al. 2010). The Southern Appalachian Mountains represent an area of the United States where exurbanization is increasing rapidly and the long-term effects are poorly understood (Kirk et al. 2012). Furthermore, salamander response to urbanization is not well documented in the Southern Appalachians, the region that boasts the highest level of diversity of salamanders in the world (Barrett and Price 2014). We know that even low-density development can reduce abundance and diversity of other wildlife over time. One study found that as a low-density development aged the diversity and abundance of five frog species declined (Gagne and Fahrig 2010). In the same study only the gray tree frog was found to increase in abundance after about 40 years (Gagne and Fahrig 2010). A study of bird diversity in suburban areas found that as developments aged, the diversity of birds decreased as well; the newest housing developments typically had the highest bird diversity (Loss et al. 2001). As (ex)urban developments age, the influences they exert on stream systems may decrease or increase depending upon the specific mechanisms influencing biota. For example, a potential stressor such as

sedimentation is likely to decrease as neighborhoods age; however, losses to riparian vegetation or changes in stream chemistry may be maintained at a high level or increase over time.

There has been a wealth of research focused on biotic response to timber operations (deMaynadier and Hunter 1995, Petranka et al. 1994, Homyack and Haas 2009). This information may offer insight into the effects of exurbanization on stream communities in the Appalachians, because timber and housing both entail removal of large quantities of vegetation and considerable erosion. In the case of timber harvests, disturbances typically return to a traditional pulse regime approximately 20 years post-harvest (Ash 2003). Changes in salamander assemblages have been documented in timber operations, and the recovery time varied between 20 and 60 years depending on the conditions or the species studied (Petranka et al. 1994, Demaynadier and Hunter 1995, Crawford and Semlitsch 2008, Homyack and Haas 2009). Declines of salamanders in timber operations are linked to decreased soil moisture, which most likely results from decreased canopy cover followed by increased temperature and lower leaf litter depth (Crawford and Semlitsch 2008). Changes in stream salamander assemblage following forest harvest can be mitigated by forested buffers (Demaynadier and Hunter 1995, Peterman and Semlitsch 2009); however, buffers in urban areas have not been shown to limit the influence of riparian forest loss on stream salamanders (Wilson and Dorcas 2003). Thus, it remains uncertain whether the model of stream salamander recovery observed following timber harvests can be applied to species inhabiting exurban watersheds.

Salamanders are an excellent candidate for monitoring stream disturbances in the Southeast. They are of high importance to Appalachian streams because they are the top predator in low-order, fishless streams (Johnson and Wallace 2011) and drive processes like nutrient uptake and cycling (Milanovich 2010). Salamanders also delay decomposition of detritus by controlling macroinvertebrate populations (Davic and Welsh 2004, Keitzer and Goforth 2013). Stream salamanders can be useful indicator species because populations are relatively stable and have high abundance (Peterman et al. 2007), but respond quickly to environmental change (Hairston 1986, Price et al. 2011, 2012). They also exhibit site fidelity (Grant et al. 2010), and they represent a connection between terrestrial and aquatic ecosystems (Regester 2006). Additionally, salamanders undergo long-term exposure to the negative influences of urbanization at various life stages both on land and in the water, and as a result salamander densities and species richness are known to be reduced by urbanization (Orser and Shure 1972, Willson and Dorcas 2003, Barrett and Guyer 2008, Price et al. 2011, 2012). Previous studies indicate salamander abundance decreases with the amount of impervious surface in the watershed (Orser and Shure 1972, Willson and Dorcas 2003), and that this results from increased flooding in urban streams (Barrett et al. 2010a). Most knowledge on salamander response to development has been derived from the Piedmont ecoregion of the U.S. (Barrett and Price 2014), and very few studies have focused on the Appalachian Mountains, which represent the global center of biodiversity for this taxon (but see Surasinghe and Baldwin 2015, Cecala et al. 2013).

The recovery trajectories of headwater streams and resident salamander populations following exurbanization are unknown. Can such sites return to conditions similar to undisturbed sites over the long-term? Which key environmental variables are most altered during neighborhood development, and do such alterations relate to the size or age of the development? Following rapid growth and expansion of suburban housing developments in the Appalachian Mountains (McDonald et al. 2010), there is a need to answer these questions to better inform habitat- and species-specific conservation plans in a biodiversity hotspot. In this study we assessed the influence of several watershed-scale variables (including neighborhood age) and a suite of in-stream measures on the occupancy and abundance of five salamander species. Based on previous research (reviewed in Barrett and Price 2014), we predicted watershed-scale measures of disturbance would be most important in predicting salamander presence or abundance, and that the negative influences of development would be exacerbated in older neighborhoods (Gagne and Fahrig 2010).

METHODS

Site Selection

Selection

This study took place within the Southern Blue Ridge Ecoregion of North Carolina and Tennessee. Elevation in this montane area is between 500 and 2000 m and the climate ranges from temperate to boreal. We selected a total of 27 exurban watersheds and eight forested watersheds to survey (Fig. 2.1). Among the exurban

watersheds our goals were to identify sites with similar amounts of impervious surface that varied in the age of development present within the watershed. For detailed information on site selection, see Chapter 1.
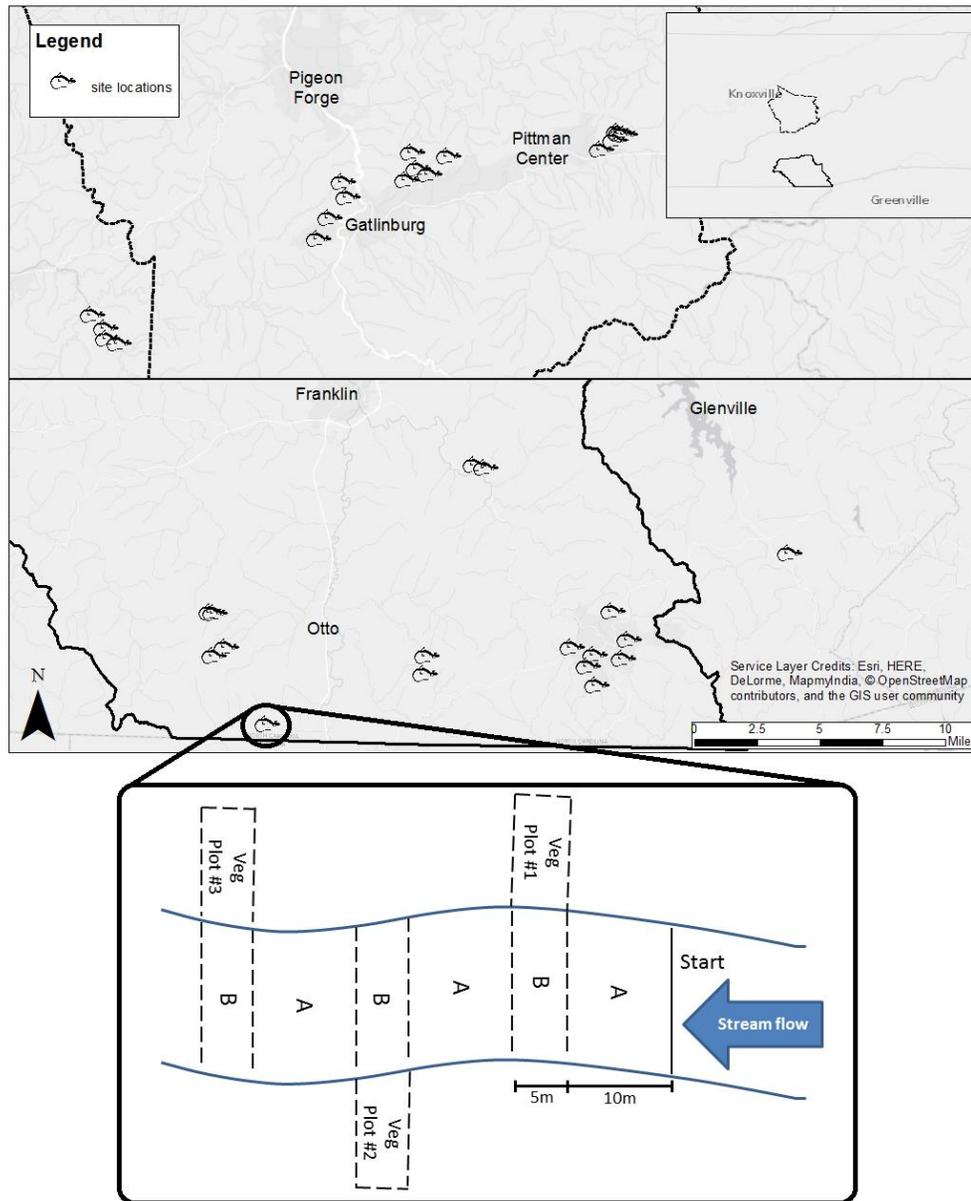
Figure 2.1. Sample locations were located in Sevier County, Tennessee (dashed) and Macon County, North Carolina (solid) with an inset map of the sample design for each site.

We used tax parcel information to identify the age of each individual structure within the watershed for each stream. These ages were then averaged to assign an age for the exurban development. Exurban housing ranged in age from four to 42 years (mean = 25.99 yrs) across the 27 watersheds with development. We calculated the percentage of impervious surface coverage for each watershed by obtaining 2014 leaf off aerial imagery (0.65 meter resolution) from the counties containing our study areas. We hand-delineated polygons around all impervious surfaces and calculated the percent of the watershed they covered. We calculated distance to impervious surface using the "near" tool in ARCGIS 10.2 (ESRI, Redlands, CA) by measuring the distance from stream sample plots to the nearest edge of an impervious surface polygon.

Field Methods

From May – June we made three visits to 35 sites in 2014 and made three additional visits in 2015. Upon our first visit to each study stream we established a 45-meter transect along the length of the stream, which was then divided into three 10-m sections, each separated by a 5-m section (Fig. 1). We sampled salamanders by dip netting in the entire wetted area of the 5-m sections in one pass by a field team lined up perpendicular to the stream (Quin et al. 2007).We also placed two leaf litter bags haphazardly in each of the three 10-m sections to capture larvae (Pauley and Little 1998). We sampled for salamanders three times during each of the two field seasons. Dip netting was performed on the first and second site visit, and leaf litter bags were placed on the first visit and then checked on the second and third visits. Leaf litter bags are 1" plastic mesh bags that are one square-foot in size and filled with leaf litter from the riparian area.

During sampling occasions we recorded the number of captures and identity for all salamanders.

At each stream we measured wetted width, maximum depth, bank height, percent undercut banks, and composition of streambed material within each of the 5-m sections. We measured streambed composition as percentage of sedimentation, pebbles, gravel, rock, and bedrock (to the nearest 5%). A YSI Sonde 600R (YSI Ohio, USA) was used to measure water pH, temperature, salinity, conductivity, and dissolved oxygen in each of the 5-meter sections once for each field season.

We measured percent canopy cover three times in the middle of the stream using a densitometer on the first site visit during the first field season. Along each of the 5-m sections, we measured 10 m from the bank of the stream to establish a 50 m^2 vegetation plot (Fig. 2.1). The plot was measured on the right side of the first section, the left side of the second section, and the right side of the third section. Streams were sampled three times each during two field seasons. We estimated percentage of ground covered by course woody debris, vegetation, and bare ground (defined as rock or soil not covered by vegetation) within each plot to the nearest 5%. We considered any fallen limb or tree larger than 10 cm to be course woody debris. We estimated basal area within an acre (40.47 dam^2) of each section using a 10 BAF basal area prism. Basal area is an estimation of the average amount of an area, one acre, which is occupied by tree stems.

Analyses

We used multivariate multiple regression to examine the influence of three watershed-scale variables (percent impervious surface, neighborhood age, and distance to

impervious surface) on a suite of uncorrelated (R<0.7) local scale response variables

(Table 2.1). We used Type I sum of squares for model evaluation. When a watershed–

scale variable was found to significantly predict local factors ($\alpha < 0.10$), we used least–

squares linear regression to identify specific bivariate relationships that were statistically

significant ($\alpha \leq 0.05$). We used the less stringent α for the multivariate test because we

considered this portion of the analysis exploratory in nature. All linear models were run

in Program R (R Core Team 2013).

Table 2.1. List of watershed-scale predictor variables and local-scale response variables
used in the multivariate multiple regression analysis.

Watershed Scale Predictor Variables	Local Scale Response Variables
% Impervious Surface	Canopy Cover
Neighborhood Age	% Coarse Woody Debris
Distance to Impervious Surface	Stream Depth
	% Undercut Banks
	Stream Bank Height
	% Pebble
	% Gravel and Cobble
	Salinity
	Dissolved Oxygen
	Water pH

Changes in salamander assemblage across sites were measured in terms of

occupancy and abundance. We began the analysis by first standardizing all covariates.

Before evaluating the factors influencing salamander occupancy or abundance, the

influence of three detection covariates were identified: the high temperature for the date

of sampling, Julien date of sampling, and sampling method (dip netting or leaf litter

traps). We modeled occupancy and abundance as a function of three watershed scale and

ten uncorrelated site scale covariates (site elevation plus those variables listed in Table 1).

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Occupancy and abundance covariates were initially evaluated separately, and then

variables from top models (ΔAIC < 2.0) were combined to test for additive and

multiplicative interactions between variables. We constructed each model such that it

contained four sampling occasions: two dip net and two leaf litter trap samples. Because

a neighborhood age could not be assigned to our eight forested sites, we first evaluated

the influence of neighborhood age by constructing candidate model sets among the 27

sites with exurban development. For those model sets where neighborhood age did not

emerge as a strong predictor or occupancy or abundance (ΔAIC < 4), we then used all 35

sites for further evaluation of habitat factors influencing salamanders.

We ran single season occupancy models using program Presence (Hines 2006)

only for mud salamanders (*Pseudotriton montanus*) because they had adequate detections

to fit an occupancy model, but insufficient detections to estimate abundance. Due to

concerns over accuracy of species identification for mud salamanders, we only used 2015

data in the analysis. Occupancy is an instantaneous measure of the distribution of a

population. This kind of model allows for variable levels of imperfect detection of a

species that may be present at a site, and allows for incorporation of covariates to test

hypothesis about what drives presence. At a site a species may be present and detected,

present but not detected, or absent. Because the final two scenarios cannot be

distinguished, detection probability must be estimated. This is done by recording

detection and non-detection data across multiple site visits during a short period of time.

It is assumed that there is no colonization or extinction during this period (Mackenzie et

al. 2006). Single season models incorporate two variables; ψ_i, the probability a species

occupies a site; and p_{ij} the probability a species is detected at a site on the *j*th survey given it is present.

For species with sufficient captures to generate parameter estimates for abundance, we ran N-mixture abundance models (Royle 2004) using the unmarked package (Fiske and Chandler 2011) in Program R (R core team 2013) by using the pcount function. We analyzed the two field seasons separately and report here on the results of two single season models for black-bellied salamanders and Blue Ridge two-lined salamanders because identification of these species was certain, while only 2015 data were used for seal salamanders. We ran abundance models using count data for four sampling occasions each season. N-mixture abundance models estimate detection probability (p) and site-level abundance (λ), and users are able to evaluate the models where both of these parameter vary as a function of site covariates. Both occupancy and abundance models assume that the population is closed during a season and that counts between sites (streams) are independent.

RESULTS

Linear regressions

Multivariate multiple regression revealed that local response variables were not strongly predicted by landscape-scale predictor variables; however, neighborhood age and impervious surface had a marginally significant relationship to local-scale measures ($p = 0.06$ and $p = 0.08$ respectively). Subsequent bivariate linear models evaluating the effect of neighborhood age on local environmental measures revealed only one

significant relationship with each predictor; the older neighborhoods we studied tended to have higher stream banks, and areas with more impervious surface had higher banks.

Occupancy and Abundance Models

The detection probability of mud salamanders was heavily influenced by date of sampling, with detections increasing as the season progressed. As a result of this effect, all models exploring occupancy covariates for this species included Julian date as a detection covariate. Several local-scale variables offered competitive explanations of mud salamander occupancy probability (Table 2.2). The models with both low ΔAIC (< 2) and high model weight (> 0.20) included DO, the percent of undercut bank, and site elevation. Mud salamanders preferred streams at lower elevation sites that had lower levels of dissolved oxygen. They were also associated with sites that had a higher percentage of undercut banks (Fig. 2.2).

Table 2.2. Occupancy model results for season two data on mud salamanders. Models with a ΔAIC of less than 4 are shown. Detection covariate for each model was Julien date.

All Sites			
Model	**ΔAIC**	**AIC weight**	**relationship**
DO+elevation	0	0.29	Negative
DO+depth	0.32	0.24	Negative
Undercut+depth	0.49	0.22	Positive
DO+undercut	2.48	0.08	Negative
DO	2.7	0.07	Negative
Undercut	3.64	0.05	Positive
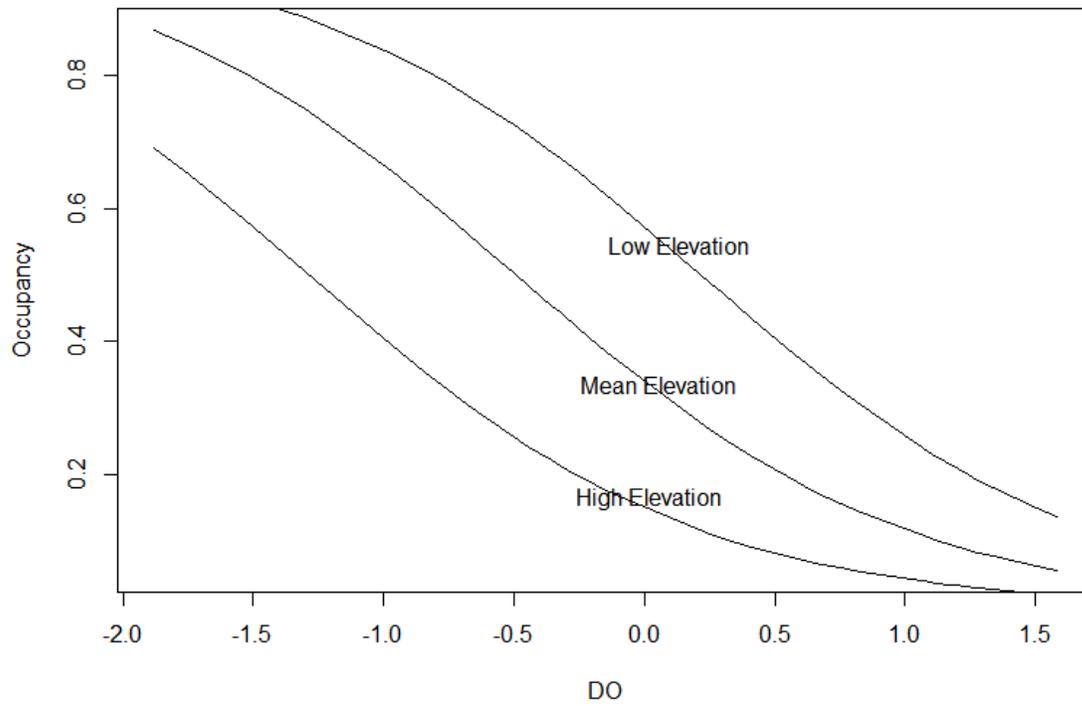
Figure 2.2. Plot of the top model for occupancy of mud salamanders in Western North Carolina and East Tennessee, which included both elevation (plotted here at the mean, 1st, and 3rd quartiles) and dissolved oxygen (standardized values are shown). X-axis values are standardized, actual values ranged between 85 and 100 percent dissolved oxygen.

Detection probability of black-bellied salamanders was most influenced by sampling method in that detection increased using the dip net method, which was then incorporated into all subsequent models of abundance. The only well-supported model for black-bellied salamanders was one in which abundance decreased with increasing salinity. This model had the most support in both 2014 and 2015 (Table 2.3; Fig 2.3a, 2.3b). Blue Ridge two-lined salamander detection probability was also a function of sampling method, with most being found by dip netting. Subsequent models of abundance that included this detection covariate revealed different top explanatory

variables between years. In 2014 DO best predicted Blue Ridge two-lined salamander

abundance (negative relationship, Fig. 2.3c), and in 2015 a model in which abundance

increased with percentage of impervious surface was the best supported (Table 2.4, Fig.

2.3d). Seal salamander detection probability was a function of sampling method. None of

the covariates we evaluated in the 2015 data emerged as better predictors of abundance

than a null model in which abundance was assumed to be equal across all sites.

Table 2.3. Abundance model results for the black-bellied salamander in Western North Carolina and East Tennessee. The top two models are shown.

	model	delta AIC	estimate	weight
season 1	salinity	0	-0.94	0.9
	elevation	4.46	0.58	0.1
season 2	salinity	0	-1.02	1
	depth	39.11	0.53	0

Table 2.4. Abundance model results for the Blue Ridge two-lined salamander in Western North Carolina and East Tennessee. The top two models are shown.

	model	delta AIC	relationship	weight
season 1	DO	0	Negative	1
	Gravel and rock	23.42	Negative	0
season 2	% Impervious Surface	0	Positive	1
	Gravel and rock	17.61	Negative	0
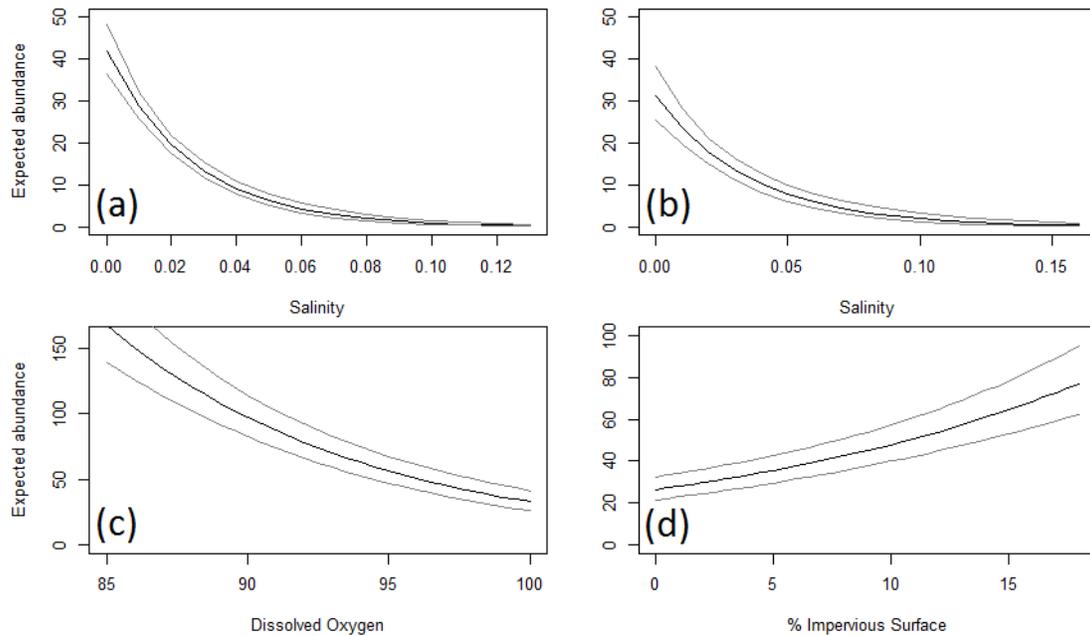
Figure 2.3. Plots of the top abundance model for black-bellied salamanders for 2014 (a) and 2015 (b), along with Blue Ridge two-lined salamanders for 2014 (c) and 2015 (d). The middle like is the mean abundance value, while the outer lines represent the 95% confidence interval. X-axis values are standardized, actual values were 0 – 0.16 g/L, 86.9 – 98.8 %, and 0 – 17.7 % for salinity, percent dissolved oxygen, and percent impervious surface respectively.

DISCUSSION

While our data suggest salamander assemblages are usually not directly predicted by watershed-scale variables, two of these larger-scale measures, neighborhood age and distance to impervious surface, marginally predicted local scale environmental variables. Nevertheless, further evaluation of this relationship indicated only bank height increased significantly with increasing amounts of impervious surface and in older neighborhoods. Overall, these results indicate that the mere presence of an exurban development in Southern Appalachia may not alter stream characteristics.

Although streams return to pre-disturbance conditions in the 20 – 60 years after a timber harvest (Demaynadier and Hunter 1995, Crawford and Semlitsch 2008, Homyack and Haas 2009), we did not find a similar salamander recovery with time in exurban neighborhoods. Neighborhood age never significantly predicted salamander abundance or occupancy, or local-site habitat variables. This is likely related to key differences exhibited by exurban developments in the presence of impervious surface. These surfaces represent continual disturbance, which leads to a cascading complex of stressors (Burcher et al. 2007, Barrett and Price 2014). The ways in which these stressors interact very likely varies among watersheds and their effects vary by species. For example, increasing salinity (likely from road salting) had negative impacts on one salamander species, but did not appear as an important predictor for others. Salinity, among other stressors, would not exist in forested watersheds managed for timber.

The species we sampled varied greatly in their response to environmental variation. Mud salamanders tended to have a higher probability of occupancy at our lower elevation sites. In addition to this factor, the species appears to be more commonly found at sites with a high percentage of undercut banks. In the Blue-Ridge ecoregion mud salamanders have been shown to tolerate disturbance, but they appear to be less tolerant in the piedmont ecoregion (Surasinghe and Baldwin 2015). In contrast, black-bellied salamanders are thought to be indicators of less disturbed habitat, (Surasinghe and Baldwin 2015). Our model results support the claim, and implicates salinity as the driver of black-bellied salamander decline in exurban watersheds. Stream salinity in mountainous areas almost certainly increases due to the use of road salt in the winter.

Kelly et al. (2008) showed that up to 91% of salinity in rural streams could be attributed to road deicing, and road salt not only persisted beyond the application period, but water salinity increased over time. Howard and Haynes (1993) showed that only 45% of the salt applied to roads each year escaped the watershed, the rest was retained and slowly leaked out with the ground water. Abundance relationships for Blue Ridge two-lined salamanders differed by year; however, in both years species abundance increased with typical indictors of higher disturbance (high % impervious surface and low DO). Black-bellied salamanders may reduce abundance of Blue Ridge two-lined salamanders in less disturbed habitats (Beachy 1997); thereby explaining high abundance of Blue Ridge two-lined salamanders in disturbed areas where black-bellied salamanders are reduced.

Other studies have shown impervious surface to be an important influence on salamander populations (Gange and Fahrig 2010, Barrett and Price 2014); however, our results were not congruent, other than a positive relationship between the amount of impervious surface and the abundance of Blue Ridge two-lined salamanders. Our data show that it is most often local-scale measures of the environment that explain our estimates of occupancy and abundance. It is possible that in these steep slope, low order streams watershed-scale variables become less important and local-scale habitat becomes the driving influence (Cecala et al. 2013, Cecala et al. 2014). Roth et al. (1996) showed that as stream watershed size decreases, the compounding effects of altered land-use appear to become less significant relative to local habitat. Montane exurban developments are typically found along ridge-lines and higher elevations, and are more associated with these smaller streams. But, because it is known that compounding

influences of land-use change become more important at larger spatial scales (Roth et al. 1996), land managers and future researchers should consider the size of the watershed when making management decisions. It is also important to acknowledge that there is variation in occupancy and abundance even in forested sites and that previous disturbance history may also be playing a role in urbanized watersheds.

Conclusion

Exurban neighborhoods in the Southern Appalachian experience different conditions and stressors than other urbanized areas of the southeastern United States, especially because they receive more annual precipitation (SOURCE?). Our data align with other studies in that drivers to changes in salamander assemblages are complex and non-singular (Burcher et al. 2007, Barrett et al. 2010b, Barrett and Price 2014). Our data show that local habitat has a much stronger influence on stream salamander populations than watershed-scale variables such as neighborhood age and impervious surface. Land owners and developers who aim to maintain stream salamander assemblages similar to forested sites must consider forested riparian buffers, maintaining heterogeneous stream substrate, and reducing water salinity. The amount of impervious surface within the watershed and the distance between impervious surfaces and streams may indirectly influence salinity and stream substrate. A larger, forested buffer could reduce the potential for road salt to enter streams by increasing soil moisture and thus buffering capacity. Land managers need to develop Best Management Practices (BMPs) for future exurban development planning, especially in regard to impervious surface and salinity. For example, salinity could be reduced by only salting roads in times that people need to

use them such as the holiday season in late November to early January. Working with the

home owners' associations may provide the easiest way to develop management

strategies for exurban neighborhoods.

APPENDICES

Appendix A

List of Species Encountered

Common Name	Scientific Name
Black-bellied salamander	*Desmognathus quadramaculatus*
Seal salamander	*Desmognathus monticola*
Northern dusky salamander	*Desmognathus fuscus*
Blue-Ridge two lined salamander	*Eurycea wilderae*
Ocoee salamander	*Desmognathus ocoee*
Mud salamander	*Pseudotriton montanus*
Red salamander	*Pseudotriton ruber*
Spring salamander	*Gyrinophilus porphyriticus*
Southern Appalachian salamander	*Plethodon teyahalee*
Southern gray-cheeked salamander	*Plethodon metcalfi*

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