

Distribution, Abundance, and Habitat Associations of Amphibians and Reptiles at the Home of
Franklin D. Roosevelt National Historic Site, an Urban Protected Area

By

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A thesis
submitted in partial fulfillment
of the requirements for the
Master of Science Degree
State University of New York
College of Environmental Science and Forestry
Syracuse, New York
May 2017

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Acknowledgments

To the people in my life who provided love, kindness, encouragement, support, listening ears, or all of the above during the course of this thesis – and you know who you are – know that I am grateful to you all. This was not a solo voyage.

I would specifically like to thank Dustin for his unwavering confidence, listening ear, and for calmly and repeatedly talking me down from the ledge of quitting. Without you, I would not have finished this. I appreciate you. I am also grateful to my parents, Mike and Kelly Dean, and Grammy, Mary Alice Cayton, for their unequivocal love and encouragement, and for endowing me with a love of critters big and small.

To the people who offered advice, guidance, ideas, constructive critiques, field assistance, logistical support, statistics help, and/or de-lousing assistance: Thank You. You've all taught me so much.

I would like to thank my committee: James Gibbs, Brian Underwood, David Hayes, and Eddie Bevilacqua, and my field assistants: Shaina Ball-Douglas, Gwyn Daunton, and Atiya Harvey.

Funding for this research was provided by the National Park Service. Support for this project was also provided by a SUNY-ESF Graduate Teaching Assistantship.

This research was carried out under New York Dept. of Environmental Conservation Scientific Collection Permit #1792, and SUNY-ESF IACUC permit #150202.

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Abstract

S. H. Dean. Influence of Habitat Associations on the Distribution and Abundance of Amphibians and Reptiles at Home of Franklin D. Roosevelt National Historic Site, an Urban Protected Area, 114 pages, 3 tables, 7 figures, 2017.

Recreational trails fragment habitat within protected areas set aside for amphibian and reptile conservation, potentially generating habitat “edge effects” that affect population persistence and community structure. Lack of research on effects of trails on amphibians and reptiles hinders conservation planning for protected areas. Habitat associations of herpetofauna were examined at the Home of Franklin D. Roosevelt, Hyde Park, New York during 2015 and 2016. Habitat edges exerted a stronger negative influence on relative amphibian abundance than did trails, although trail effects did occur and were primarily driven by loss of microhabitats for amphibians (low sample size precluded assessment for reptiles). Effects of trails and habitat edges were additive. These findings highlight the need to consider recreational trails in habitat planning for conservation of herpetofauna, particularly in urban protected areas.

Key words: *Ambystoma maculatum*; amphibian; edge effect; habitat; *Lithobates sylvaticus*; New York; *Notophthalmus viridescens*; *Plethodon cinereus*; *Pseudacris crucifer*; protected area.

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Chapter I: Introduction and Literature Review

Impacts of Urbanization on Wildlife and Their Habitats

With over 80% of the US population residing in urban areas – an increase of 27 million people from the year 2000 – urbanization is a form of land use change that poses a major threat to biodiversity (USCB 2015; McDonald et al. 2009). The environmental changes associated with urbanization modify animal habitats, behaviors, diversity, and life ecologies. Although threats of urbanization to wildlife are numerous, biologists have disproportionately focused their attentions on pristine environments instead of those in human dominated or disturbed systems (Chiesura 2004; Collins et al. 2000; Martin et al. 2012), leading to knowledge gaps in how to effectively manage species in urbanizing landscapes. Moreover, species living in or near human-dominated environments are under strong and novel selective pressures (Collins et al. 2000). Despite this, fewer than 6% of the ecological literature represents research in suburban or urban areas (Miller and Hobbs 2002), even though as of 2008, 8% of terrestrial vertebrate species on the IUCN Red List are largely imperiled because of urban development (McDonald et al. 2008).

Species diversity is substantially affected by urbanization, and this trend holds true for most vertebrates and insects (McKinney 2008). Knapp et al. (2008) found a higher species diversity of butterflies, lichens, and birds in a rural protected area versus inside natural areas of an adjacent German city. Wetlands embedded within urban areas exhibit up to 60% lower taxonomic richness of amphibians and aquatic reptiles compared to those in more natural rural settings (Johnson et al. 2013; Rubbo and Kiesecker 2001). Species richness may commonly underestimate the impacts of urbanization, because overall richness can mask steep declines in infrequently encountered species (Villasenor et al. 2017). In some cases, researchers have noted not just a lack of richness, but an absence of even common species in developed areas (Foster et al. 2004). One of the results of low-diversity biological communities is that disease transmission can increase among urban-adapted hosts with negative implications for nearby non-urban populations (Bradley and Altizer 2006).

One reason for reduced biological diversity in urban areas is that remaining natural habitats are often smaller than their rural counterparts, which reduces their habitat complexity, thereby

decreasing the available niche space (Hamer and McDonnell 2010). Other research suggests that a main driver of reduced species richness in urban areas is road density, because it alters habitat characteristics that influence dispersal (Johnson et al. 2013). Hydroperiod, a measure of wetland permanence, has also been implicated as a reason for altered species richness, because urban wetlands are more likely to be permanent (Johnson et al. 2013, Rubbo and Kiesecker 2001) and support fish and other non-native predators (Johnson et al. 2013) which are uncommon in wetlands with variable hydroperiods. Light pollution may also be a factor in lowered biological diversity of urban areas because it can disrupt animal physiology, behavior, and ecology of nocturnal species (Perry et al. 2008), and its effects can extend 10 km into protected areas (McDonald et al. 2009). Increased noise exposure from urbanization can compromise predator/prey detection or mating signals, alter animal movement patterns, and increase physiological stress among wildlife (Francis and Barber 2013). Chemical contaminants associated with development such as road de-icing salt and pesticides befoul water bodies, which also contributes to population declines or extirpation of sensitive aquatic organisms (Khan and Law 2005).

Urban landscapes favor generalists and highly adaptable species to the detriment of others. Raccoons (*Procyon lotor*), Opossums (*Didelphis virginiana*), and domestic pets such as dogs (*Canis familiaris*) and cats (*Felis catus*) all benefit from urbanization. Urban-adapted wildlife benefit from direct or indirect food subsidies, such as trash cans, or wildlife feeders (Hamilton et al. 2002), and population increases in these predators are detrimental to other wildlife. For example, Sutherland (2009) witnessed an increase in mesopredators (medium sized predators that serve as both predators and prey) with increasing urbanization, and an opposing trend in amphibians, snakes, and ground-nesting birds along the same gradient. Among Diamondback Terrapin (*Malaclemys terrapin*) nests laid at Jamaica Bay (NY), approximately 95% were depredated by raccoons, a pattern attributed to high levels of food subsidization of raccoons, rats, and gulls (Feinberg and Burke 2003; Ner and Burke 2008).

Other animals, such as domestic cats and dogs, are important sources of mortality for both adult and juvenile lizards in suburbs (Koenig et al. 2002), and this is likely true for snakes and small-bodied and juvenile turtles. Larger-bodied turtles are almost completely protected from all but the largest of animal predators by their size and shell (Brisbin et al. 2008). Humans can also be urban “predators” in the sense that they can remove individuals from a population via

poaching for food or the pet trade (Burger 2001; Garber and Burger 1995) or by accidental mortality from lawnmowers or burning brush piles (Brisbin et al. 2008).

Amphibians, Reptiles, and Urbanization

Amphibians and reptiles are among the most imperiled groups globally, whose populations have seen steep declines due to direct habitat loss, habitat fragmentation, climate change, disease, invasive species, and the pet trade (Alford and Richards 1999; Bohm et al. 2013; Gibbons et al. 2000). Collectively these groups often exhibit traits that make them more vulnerable to urbanization than other taxa (i.e., birds, mammals), such as limited dispersal ability (amphibians), moving large distances during reproductive activities (reptiles), sensitivity to pollutants (amphibians), low recruitment (turtles), reliance on multiple habitat types, exhibiting strong site fidelity, and popularity in the pet trade (Cushman 2006; Hamer and McDonnell 2008; Tsuji et al. 2011). Particularly problematic is that for a high proportion of reptile and amphibian species there is insufficient data to base management and species status decisions. One in five reptiles is globally threatened with extinction and 20% are data deficient (Bohm et al. 2013). For amphibians, 24% lack data ($n = 1545$, Nori and Loyola 2015), with amphibian extinctions approximately 211 times the background extinction rate (McCallum 2007). Urbanization contributes to these species declines because it directly and indirectly creates – and can exacerbate – existing threats (Figure 1.1).

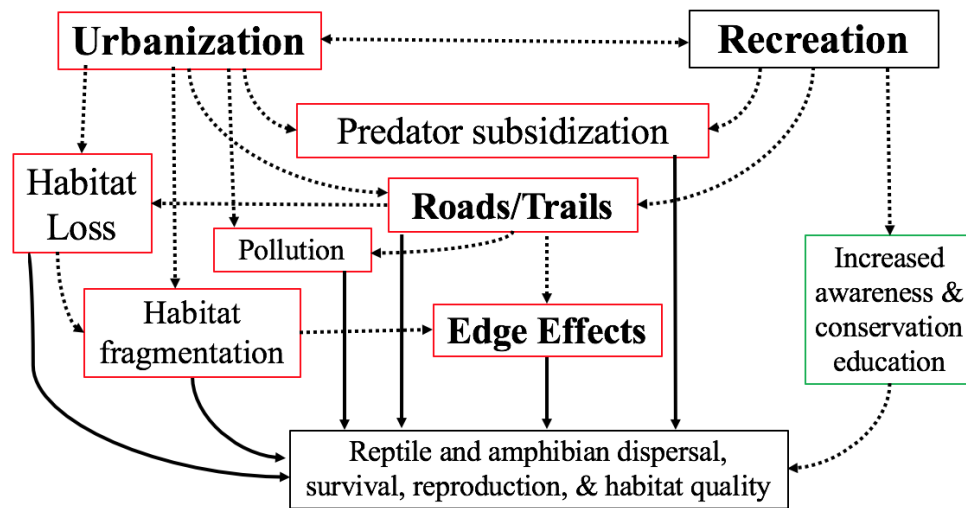


Figure 1.1: Major effects of urbanization that can impact reptiles and amphibians. Positive relationships have dotted lines, negative relationships have solid lines. The majority of these effects negatively impact reptile and amphibians, although recreation results in increased awareness and conservation education, which positively impacts herpetofauna.

Among the most pernicious of direct threats from urbanization is habitat loss and fragmentation, where habitat is depleted via construction, and further lost, fragmented, and degraded through the creation of roads and edge effects. Most habitat lost to urbanization is usually habitat that is lost permanently (McKinney 2002). Populations in habitats that remain are still threatened by fragmentation because amphibians and reptiles have limited dispersal abilities (Cushman 2006), life cycles that are often dependent on multiple habitats (Rubbo and Kiesecker 2005) at large spatial scales (Clark et al. 2008), and acute sensitivities to pollution and microclimatic shifts.

Wildlife that live in urban areas often exhibit different behavior than their non-urban counterparts. For example, suburban dwelling Eastern Long-necked Turtles (*Chelodina longicollis*) moved distances twice as far as a control group on a nature reserve in Australia (Rees et al. 2009). Urban resident Gila Monsters (*Heloderma suspectum*) also tend to move smaller distances, although their movement pattern may have more to do with more abundant prey in desert urban areas due to well-watered lawns and bird feeders (Kwiatkowski et al. 2008). In contrast, Box Turtles (*Terrapene carolina*) move less when in fragmented habitat (Igley et al. 2007). Often human actions are the cause of urbanization impacts, but in some cases the mere presence of humans is a disturbance in and of itself. For example, basking behaviors are greatly

influenced by human disturbance. Peterman and Ryan (2009) noted only juveniles and hatchling turtles used disturbed basking sites, and a third semi-aquatic snakes in a NJ park responded to observer approach by fleeing (Burger 2001).

Reptiles and amphibians show contrasting responses to urbanization. Reptiles and amphibian populations exhibited an inverse relationship along an urban-rural land use gradient, and are thus not equivalent for monitoring purposes (Barrett and Guyer 2008). Specifically, reptiles increased in abundance, while amphibians decreased, as urbanization increased, which is similar to the results of Venegas-Guerrero et al. (2016), who compared amphibian richness in urban areas versus reptile richness in Colombia. The opposite result was found in Australia, with a higher proportion (25 %) of reptiles with a < 50 % probability of persistence in urban areas, versus only 7 % of frogs (Hamer and McDonnell 2010). These conflicting results indicate that the relationship between urbanization and herpetofauna may not be generalizable across all amphibian and reptile taxa, but might be species-specific or determined by certain life history traits. For example, in some reptile species with sizeable home ranges, in which one sex is more vagile than the other, human-induced sex-biased mortality can occur (Nazdrowicz et al. 2008), which can slowly wreak havoc on the demography of long-lived species. Kwiatkowski et al. (2008) found an urban population of Gila Monsters to be female-biased, which may indicate that the male population (the more vagile sex in this species) may have a higher mortality risk.

How amphibians are affected by urbanization is likely a function life history and other factors, including habitat connectivity, propinquity of habitats required for life cycle, amount of nearby impervious surface, and natural vegetation cover (Furman et al. 2016). Salamanders, especially those that are associated with small first or second order streams are disproportionately affected by urbanization, with negative effects on abundance one year post urbanization, and declines as steep as 98% after four years (Price et al. 2012). A comparison of Southern Two-lined Salamanders (*Eurycea cirrigera*) larvae in urban and developing catchments found lower survivorship in urban cohorts, which was likely due to frequent and increased channel flow during flood events (Barrett et al. 2010; Barrett 2009). Increased sedimentation may also be a culprit in decreased occupancy of Southern Two-lined Salamanders and Northern Dusky Salamanders (*Desmognathus fuscus*) (Price et al. 2010).

Urbanization tends to decrease reptile and amphibian diversity. The relationship between urbanization and wetland species richness and diversity is strong and negative among reptiles

(Elston et al. 2016), amphibians (Zhang et al. 2016), and insects, perhaps because of increased likelihood of predation and competition by non-native species (Johnson et al. 2013). For amphibians, low urban diversity can be caused by a loss of wetland connectivity, as this inhibits the ability of immigrants to rescue population losses (Johnson et al. 2013). Infrequently, urbanization increases diversity, such as the relationship found by Ackley et al. (2014), where lizard diversity is highest in affluent suburban areas due to a “luxury effect” of increasing plant richness in carefully cultivated lawns in the desert landscape.

Recreational Impacts on Wildlife and Their Habitats

As the proximity between urban areas and protected areas increase, so does the potential for human interactions with wildlife and their habitats (McDonald et al. 2009). Land managers must balance public demand for nonconsumptive recreational opportunities with undesirable impacts of these activities on the lands and species they manage. Recreational opportunities include many diverse activities such as: hiking, boating, biking, swimming, scuba diving, motorized recreational vehicles, bird watching, rock climbing, nature photography, geocaching, trail running, and dog walking. Environmental impacts of recreation are generally negative and include disturbance of wildlife, trampling of habitats and vegetation, increased erosion, and subsidization of predators; however a positive effect exists, because participants in outdoor recreation are more likely to demonstrate environmentally responsible behavior, and may be more likely to contribute to conservation organizations (Lee 2011; Stewart and Craig 2000; Zaradic et al. 2009). Although recreation is usually an indirect result of urbanization, the opposite relationship exists, such as in cases where remote protected areas become tourist attractions (e.g., Yellowstone National Park), thus encouraging urban development surrounding the protected area (Figure 1).

The severity of the environmental impacts of visitors to protected areas is dependent on at least ten factors, which include: conservation value of the site, site’s resistance to use, its recovery from use, susceptibility to erosion, severity of direct impacts associated with specific activities, severity of indirect impacts, the amount of use, the social and ecological dimension to the timing of use, and the total area affected (Pickering 2010). Recreational ecology, the study of the environmental consequences of outdoor recreation and tourism activities and their effective

management, is a relatively recent discipline that has already established links between human recreation and negative outcomes for wildlife and habitats.

Outdoor recreation has the potential to disturb wildlife, which may result in modified behaviors and fitness, as well as the potential avoidance of preferred habitat (Taylor and Knight 2003). Animals disturbed in one place can remember the experience, and because this behavior is learned, individuals of the same species may differ in their vulnerability to the same disturbance, making anticipating their response a challenge for managers (Cole 1993). In a review of nature-based recreation studies (wildlife viewing, hiking, running, cycling, canoeing, horse riding, and dog walking, n = 69) on birds, 88% of papers from 1978 – 2010 reported negative impacts, including changes in bird physiology, immediate behavior, and abundance, as well as changes in reproductive success (Steven et al. 2011). A comparison of paired protected areas with and without recreation revealed a five-fold decline in native carnivores and a substantial shift in community composition from native to non-native species in areas with recreation (Reed and Merelender 2008), however, predators that remain may use trails and roads as travel corridors (Whittington et al. 2005).

Recreational disturbance can alter animal behavior. Black Bears (*Ursus americanus*) will abandon dens and cubs in response to anthropogenic disturbance (Goodrich and Berger 1994). Dogs alone and people walking dogs on leashes were found to negatively alter not only the behavior of small mammals and birds (Lenth et al. 2006; Miller et al. 2001) at least 25 m and up to 100 m off trail (Lenth et al. 2006), but also abundance of certain bird species up to 100 m away from trails (Miller et al. 1998). Bird nest predation increases near trails (Miller et al. 1998), however, predator type (avian or mammalian) may also influence nest predation close to trails, as Miller and Hobbs (2000) found avian predation increased and mammalian predation decreased with close proximity to trails.

Recreation has the capacity to extirpate populations, as demonstrated by Garber and Burger (1995), who documented the demise of two populations of Wood Turtles (*Glyptemys insculpta*) over the course of 10 years as a result of the opening of the turtles' habitat for recreation likely due to removal of turtles by recreationalists or greater predator presence (e.g., raccoons, crows) as a result of increased human activity. Nocturnal animals may not be affected by recreational activity, especially if activities are limited to daylight hours (Reilly et al. 2017).

Areas that support recreation appear to offer lower quality habitat than those that do not. Recreational areas were found to have lower numbers of trees, species, and amount of canopy cover in Iran (Behjou and Esfahan 2013), however, different modes of recreation may result in varying levels and types of disturbance, of which there is an asymptotic relationship between amount of use and resulting impact (Cole 1993). Although hiking may be perceived to be an environmentally low-impact activity, research shows that this is not often the case. A review by Steven et al. (2011) found that even low impact activities such as walking or hiking, where recreationists do not deliberately disturb animals, can have negative effects on wildlife ranging from reduced reproductive success to changes in physiology. These effects can extend up to 200 m from the trail (Taylor and Knight 2003).

Recreational activities can affect wildlife indirectly through the degradation of their habitat. Nutrient inputs from human waste has direct impacts such as removal of vegetation to dig a hole, but also indirect impacts through the addition of nutrients which can result in species composition due to competitive displacement (Pickering and Hill 2006). Vegetation height, cover, and composition in an Australian grassland can be reduced after a mere two weeks of a hiking treatment (Pickering et al. 2011). More alarming, radial patterns of increased bare ground can develop after a single night of camping and this pattern can be sustained for more than three years, which suggests long recovery patterns even in low impact situations (Cole and Monz 2004). The few long-term studies on habitat recovery from recreation further support the idea of extensive recovery timelines (Monz 2002), as recovery of an herbaceous understory to an “acceptable state” (defined as a plant community similar to that of the surrounding secondary forest and a lack of exotics originally seeded in the recreation areas) takes 11 to 20 years, potentially varying with historical and environmental factors (Gibson et al. 2000).

Mountain biking, although relatively understudied, has detrimental effects on birds and soils. Mountain biking in areas of Golden-cheeked Warbler (*Setophaga chrysoparia*) (IUCN endangered) nesting resulted in a three-fold increase in nest abandonment, 50% increase in size of male territories, and over triple the amount of nest depredation compared to non-biking sites (Davis et al. 2010). In terms of habitat devastation, mountain biking causes soil and vegetation damage from skidding and the construction of unauthorized trails/jumps/bridges (Pickering et al. 2010). Mountain biking has also been shown to reduce vegetation height, cover, and species richness, cause changes in species composition, and increase litter and soil compaction

(Pickering et al. 2011). A major disturbance from bikers is the creation of “technical trail features” (TTF) to enhance their riding experience. These can include created ramps, dirt ramps, riding on logs, ditches, mounds, or creating jumps across logs, and these are often combined with the unauthorized creation of informal trails because bikers seek more challenging trails than those provided (Newsome and Davies 2009; Pickering et al. 2010). The conservation value of recreational areas may be reduced by the presence of large numbers of TTFs because they damage habitat and promote fragmentation from within (Pickering et al. 2010).

Comparisons of the disturbances enacted between hiking and biking seem to be dependent on the taxa studied, though very little research has investigated wildlife response to bikers. Taylor and Knight (2003) found little difference in large mammal response to hikers and mountain bikers. A potential problem arising from bikers, especially because their speed allows them to disturb more wildlife per unit time (Purdy 1987), is the establishment of wildlife avoidance behavior that leads animals to expend energy, leave preferred or prime habitat, and move to other, lower quality habitat (Marion and Wimpey 2007). Because bikers tend to travel greater distances than hikers, they also have an increased ability to further the spread of non-native plant seeds that may be attached to their bike or clothes, more so than hikers alone. Much of the literature on the comparison of these activities focuses on impacts to vegetation, soils, and trail degradation, and these effects on the microhabitats around trails plausibly affect wildlife sensitive to habitat perturbations. Early research on mountain biking impacts found little distinction between the impacts of mountain biking and hiking on trail soils and adjacent vegetation (Thurston and Reader 2001), instead arguing that other factors like trail placement, slope, and soil wetness were more influential in affecting trail erosion and muddiness (Marion and Wimpey 2007, White et al. 2006). More recent studies have indicated more nuanced differences between these activities, such that mountain biking causes more vegetation and soil damage than hiking, but only at high use, and that unlike hiking, mountain biking can result in skid damage and unauthorized building of TTFs (Newsome and Davies 2009; Pickering et al. 2010; 2011; Pickering and Barros 2015). In addition, both can significantly increase the percent of bare ground cover, but hikers tend to trample broader swaths of vegetation than bikers, as hikers are more selective about their footfalls, while bikers are perhaps less discerning (Havlick et al. 2016).

Independent of the activities that occur on them, trails have impacts in and of themselves. Much trail research is plant focused, and the anthropogenic effects of trails on plants include soil erosion, compaction, nitrification, changes in hydrology, trail widening, damage to plants via trampling, negative changes in plant species composition, and the spread of weeds and plant pathogens (Pickering et al. 2010; Ballantyne and Pickering 2015c). Many of these effects likely affect soil moisture and temperature (Barros et al. 2013), thereby altering microhabitats within and adjacent to trails. Trails can also restrict movement for plant species with short dispersal distances (Barros and Pickering 2012). Additionally, plant damage from trails can facilitate the establishment and growth of other species, eventually altering plant community function, composition, and succession (Ballantyne and Pickering 2015a,b); Exotic plant cover is often highest within three meters of trails (LaPaix et al. 2012). Even lightly trampled paths decrease cover from the most abundant plant species and show drastic plant community change (Hamberg et al. 2010). Although short-term trampling can cause reduced plant cover, plant height, and species density, long-term trampling includes all of the above plus a decrease in leaf litter biomass and soil density (Kissling et al. 2009). Poor placement of trails and unclear trail cues can exacerbate trail effects. For example, trails with high slope degrees are more easily eroded, and trails in seasonally wet areas can be widened by hikers seeking to avoid muddy areas (Cole 1993). Trail widening can occur when visitors have difficulty discerning where the trail is located (Svajda et al. 2016; Wimpey 2009).

Impacts of trails can differ based on their design, location, construction, maintenance, and use (Ballantyne and Pickering 2015b). Of note is the difference between formally maintained trails, and informal, visitor-created trails (also called “social trails”). Formal trails can be dirt, gravel, or paved, and are differentiated from informal trails, which usually have a bare ground substrate, because they are created and/or maintained by land stewards. Often, informal trails consist of short cuts to other trails and trails that are created to connect existing tracks (Marion and Wimpey 2007). Physically, formal trails are generally wider than informal trails, and this increased width may be more important than trail type in determining impact (Wimpey 2009; Ballantyne and Pickering 2015b).

Ballantyne and Pickering (2015b) found that both trail types resulted in a similar loss of forest strata, with informal trails exhibiting poorer surface conditions, and poor design and location. Differences that exist between formal and informal trails is also attributable to the

greater spatial proliferation, ergo greater fragmentation capacity, of informal trails (Ballantyne et al. 2014). For example, Mt. Rainier National Park is divided into 26 patches by park trails, and up to 465 patches when informal trails are included (Moskal and Halabisky 2014). Informal trails are problematic because as habitat becomes more fragmented, each patch is exposed to proportionally greater edge effects (Pickering et al. 2012). Although data suggests that when confronted with a choice, visitors choose formalized trails, approximately a third of visitors use informal trails (Walden-Schreiner and Leung 2013). However, trail type comparisons are limited by the dearth of informal trail literature, which comprises only 15% of published trail research (Ballantyne and Pickering 2015c).

As networked structures, roads render vast areas of the landscape road-affected due to their direct and edge effects (Coffin 2007; Maynard et al. 2016); in certain areas such as parks, a similar effect is likely attributable to trails. Although trails are not generally associated with vehicular traffic and the pollutants associated with them, they may prove similar to roads in that: they come in similar widths and substrates, habitat is disturbed and/or destroyed to build them, they are anthropogenic features managed with anthropogenic goals in mind, they create forest edges and associated edge effects (Ballantyne et al. 2014), they contribute to habitat fragmentation, potentially contribute to the spread of invasive species, and they increase impervious surface area (Figure 1.2). Although arguably less threatening than motor vehicles, mountain bikers certainly have the potential to contribute to amphibian and reptile mortality on a smaller scale. Roads will obviously always have greater impacts than trails due to area coverage alone, however, trails have the potential to produce negative impacts.

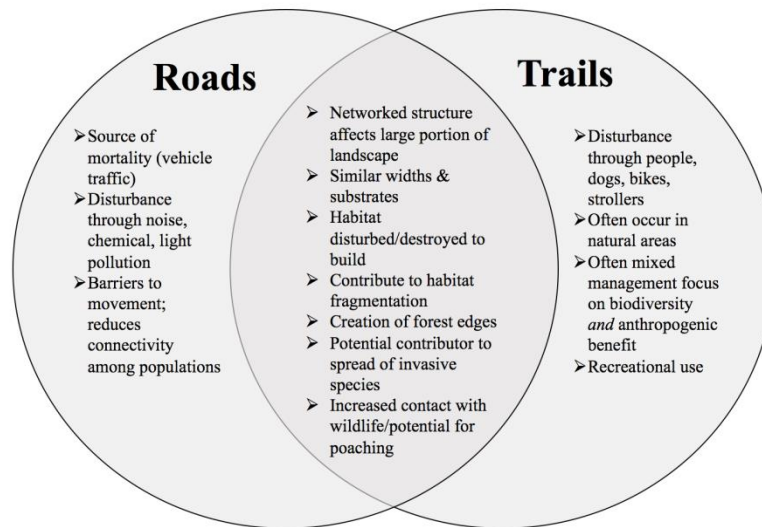


Figure 1.0.2: Comparison of potential impacts of roads and trails on biodiversity. Roads and trails are used by differing groups with different goals, but similarities in their structures and the way they are constructed contribute to habitat destruction and fragmentation.

Recreational Impacts on Reptiles and Amphibians

Recreational trails and roads are analogous in their impacts on wildlife, especially concerning effects on reptiles and amphibians. An extensive body of research documents the negative effects of road building, use, pollution, and the resulting traffic on herpetofauna (e.g., Eigenbrod et al. 2007; Fahrig and Rytwinski 2009; Row et al. 2007), and because many species are small and have low retention times on roads, effect sizes are likely underestimates (Heatwole and Wilkinson 2012). One study found 178 roadkill toads scavenged from a road by dawn, of which there would be no remaining evidence by a 10 am survey (Slater 2002). Road effects on amphibians are substantial enough that the negative effects of roads are equally as severe as the negative effects of deforestation (Fahrig et al. 1995; Eigenbrod et al. 2007). An increase in the density of paved roads up to 2 km from wetlands was negatively correlated with herpetofaunal species richness and newer studies show wetlands 1 km from highways have 150% of the frog richness of those close to highways (Findlay and Houlihan 1997; Hamer 2016). For tortoises, population reductions extended up to 300 m from roads (Peaden et al. 2015). Even unpaved roads have been shown to exhibit similar changes in plant population and community structure along their edges (Arévalo et al. 2008; Delgado et al. 2007).

Roads and the vehicle traffic on them are a source of mortality for all herpetofauna (Fahrig and Rytwinski 2009), and may have disproportionate effects for long-lived and vagile species (Karraker and Gibbs 2011; Gibbs and Steen 2005; Row et al. 2007). Particularly for amphibians, turtles, and small snakes, roads serve as major barriers to movement, reducing connectivity among populations (Andrews and Gibbons 2005; Gibbs and Shriver 2002; Gravel et al. 2012; Proulx et al. 2014) through mortality and behavioral avoidance, although roads may be more similar to a filter than a complete barrier (Hamer 2016). It is possible that wider roads may serve just as a physical barrier, but a psychological barrier, where animals may perceive crossing as too formidable (Heatwole and Wilkinson 2012). While the benefit of road removal for herpetofauna has not been explored in the literature, reductions in chronic erosion would likely improve downstream water quality to the benefit of aquatic wildlife (Switalski et al. 2004).

A pollutant specific to roads is deicing salt, which can travel up to 172 m from highways into wetlands (Karraker et al. 2008). Road deicing salts alter the chemistry of aquatic habitats, resulting in the exclusion of salt intolerant amphibians (Collins and Russell 2009), sublethal effects on embryonic amphibians that hamper their response to competition and predation (Denoel et al. 2010; Karraker and Ruthig 2009), and mortality (Sanzo and Hecnar 2005; Turtle 2000; Winkler and Forte 2011); these effects tend to escalate with increasing NaCl concentration. Certain indirect effects of roads create other concerns. Traffic noise can mask breeding calls, which may explain why some frogs tend to avoid habitat with high levels of noise (Hoskin and Goosem 2010; Cosentino et al. 2014). Additionally, the illegal collection of herpetofauna for the pet trade is likely a result of the expansion of road networks into their habitats (Grandmaison and Frary 2012).

Although overwhelmingly negative, roads do create some marginal benefit to reptiles and amphibians. As ectotherms, some reptiles and amphibians use paved roads for thermoregulation or as migration corridors (Timm and McGarigal 2014), however, this creates an ecological trap via road mortality. Lower turtle nest predation near roads suggests roads could be beneficial because they deter predators, but only in circumstance where the roads do not increase adult turtle mortality (Hamilton et al. 2002). This seems an unlikely reality, as nesting near roads puts females at risk during the journey to and from the nesting site, and also makes dispersal even more so precarious for hatchlings if nests survive.

Very few studies have investigated the specific effect of recreation on herpetofauna, and those studies generally focused on reptiles. In comparing Desert Tortoise (*Gopherus agassizii*) populations in areas of different levels of off-road vehicle use, Bury and Luckenback (2002) found unused areas had almost double the number of live plants, and four times the amount of plant cover, number of tortoises, and active tortoise burrows. For aquatic turtles, boat propeller strikes contribute to poor shell condition, injuries, and assumedly mortality (Bennett and Litzgus 2014; Selman et al. 2013), suggesting that boats may be a major threat to turtles (Bennett and Litzgus 2014). Additionally, turtles at recreation sites bask for shorter periods of time and are more stressed (Selmen et al. 2013). Although the behavior of Eastern Musk Turtles (*Sternotherus odoratus*) was not affected by a recreational waterway, their population demography showed evidence of higher mortality at the recreational site (Laverty et al. 2016). Considering the impact of boating on smaller aquatic reptiles is difficult to quantify as they are less likely to survive propeller strikes, and the logistical challenge of gathering data on strike survival (Bennett and Litzgus 2014; Laverty et al. 2016), it is not surprising that few studies have investigated the relationships between turtles and water based recreation. A single study on Painted Turtle (*Chrysemys picta*) nesting in recreational areas found no effect of human recreation (Bowen and Janzen 2008).

Recreation has negative consequences for snakes, as Burger (2001) observed that a third of basking semi-aquatic snakes in a New Jersey park fled in response to researcher approach, and most alarmingly, made direct observations of snakes being poached, collected, moved, and harassed by park visitors and their dogs. Burger (2001) also noted that most pedestrians do not approach snakes, and generally do not even see them. However, this does not mean that snakes are not aware of recreationists. Eastern Massasaugas (*Sistrurus catenatus*) moved shorter distances and less frequently with increasing exposure to human disturbance (Parent and Weatherhead 2000), and Beale et al. (2016) found Western Diamondback Rattlesnakes (*Crotalus atrox*) were more likely to be concealed in areas frequented by tourists, and are more likely to move during the mating season in highly disturbed areas, probably owing to the overlap of their mating season with peak tourist activity.

Amphibians and nesting reptiles may be at risk near trails due to an increased chance of encountering a predator. Similar to their propensity for roads, carnivore detection is highest on trails (Reilly et al. 2017). In the San Francisco Bay area, all medium and large sized predators

used trails, despite use of trails by recreationists. Most mammals in the study showed no negative spatial or temporal response, suggesting they are able to habituate to predictable and recurrent human recreation (Reilly et al. 2017).

The scope of the effects of recreation likely varies by type of recreation and taxa due to different disturbance sensitivities and life histories. For example, mammals exhibited a 70% likelihood of flushing from on-trail recreationists within 100 m from trails (Taylor and Knight 2003), but a salamander beneath a cover object would likely not have the same response. Other findings support the idea of taxa-specific responses (Ficetola et al. 2007). They found a negative relationship of litter disturbance with reptile and amphibian species richness, and no relationship for small mammals or birds in an urban Italian park. However, species specific management to address impacts of recreational trails is limited because the trail literature is biased towards broad taxa generalizations (Ballantyne and Pickering 2015c).

In contrast to other taxa and amphibians, research indicates that reptiles in particular may actually exhibit an inverse relationship to urbanization-related disturbance. Among a land-use gradient, reptiles favored areas more affected by urbanization, characterized by open vegetation and deeper/warmer waters, while amphibians favored forested areas with a closed-canopy and small shallow streams (Barrett and Guyer 2008). Grundel et al. (2014) and Crosswhite et al. (2004) found similar results, where reptiles were more abundant in open habitats with low canopy cover, and amphibian abundance was greater in forests. Barrett and Guyer (2008) concluded that reptiles are more adaptable to urbanization due to their thicker skin, larger size, and amniotic eggs. In some cases, reptiles may be more influenced by human disturbance because unlike most amphibians, they share the same diurnal activity patterns as recreationists (Ficetola et al. 2007). This specific trait leaves reptiles inhabiting recreational areas vulnerable, because an increased likelihood of crossing paths with recreationists leads to an increased likelihood of illegal poaching (Garber and Burger 1995; Grandmaison and Frary 2012). The removal of reproductive adults from the population can be especially detrimental to turtles because of their low reproductive output. Heatwole and Wilkinson (2012) acknowledge that amphibian poaching also occurs in recreational areas, but at conceivably low levels.

If trails generate a more open canopy, impervious surfaces, and more human disturbance, then reptiles would be expected to be more prevalent in these areas than in the surrounding forest. In line with this, we might expect a low density of reptiles in the forest because they are

choosing alternative habitats for more preferable canopy cover. If trails negatively impact amphibians because of trail edge effects similar to habitat edges effects, we would expect to see a decrease in abundance close to trails. However, multiple influences are at play in urban protected areas, including the presence of habitat edges and roads along with high recreational pressure. How habitat edges, trails, and trail edges interact is unknown, though the overwhelmingly negative body of research from these disciplines individually predict that areas of overlap will also likely have effects at least as detrimental, if not more so, on reptiles and amphibians.

Habitat “Edge Effects” on Wildlife

“Edge effects” refer to the changes in population and community structure that occur at the boundary of two habitats; this is generally a result of habitat fragmentation. The differences in abiotic factors between edges and interior habitats are often the crux of their impact on animal communities, and these can undermine habitat quality and ecosystem services (De Paula et al. 2016). The creation of edges produces differential abiotic gradients that result in a set of primary responses, which lead to secondary responses that further perpetuate edge influence in plant communities (Harper et al. 2005). Edge-to-interior forest gradients in tree cover, soil temperature, light intensity, canopy cover, vapor pressure, and canopy height, permeate 10 to nearly 120 m into the forest (Delgado et al. 2007; De Paula et al. 2016; Harper and MacDonald 2001; Marsh and Beckman 2004; Pohlman et al. 2007), and these abiotic factors can vary temporally and by type of edge (Pohlman et al. 2007). Edges also disrupt vertical stratification of microclimate variables (Didham and Ewers 2014). Interestingly, edge effects can further vary in time of climatic stress; edge effects found by Marsh and Beckman (2004) quadrupled (increase from 20 m to 80 m) during a drought. Biotic gradients exist as well, such as 30 – 45% lower soil microbial activity and biomass 20 m into the forest (Malmivaara-Lamsa et al. 2008).

The magnitude and distance of edge influence may be a function of the contrast in the structure and composition between two edge communities (Eldegard et al. 2015; Harper et al. 2005), landscape context (Donovan et al. 1997), or forest patch size (De Paula et al. 2016). Pohlman et al. (2007) termed this phenomena, where the nature of the matrix between isolated fragments exerts a strong influence on the severity of edge effects, the “matrix effect”. For

example, Gibbs (1998) found the relative permeability of forest-road edges much reduced compared to forest-open land edges and the forest interior.

Unlike transitional edges from field to forest, roads and trails generate twice their length in forest edges (Goosem 2007), which fragment landscapes at small scales (Malmivaara-Lamsa et al. 2008), and these traits can diminish the benefits of land sparing (Lamb et al. 2016). Prior research has shown trail effects to range from 3 to 10 m (LaPaix et al. 2012; Pickering et al. 2012; Godefroid and Koedam 2004), although for salamanders, trail width may drive the magnitude of trail edge effects (Marsh 2007). Road edge effects can reduce reptile and amphibian abundance and diversity 100 m from the forest edge (Maynard et al. 2016). The understanding that trails have edge effects drastically increases the percent of land that is directly affected by their existence (Ballantyne et al. 2014). Currently unknown are the effects of concomitant habitat and trail edges, although these are likely to be frequented the most often by trail visitors (LaPaix et al. 2012).

For amphibians and reptiles, a conservative estimate for edge effect on amphibians from deMaynadier and Hunter (1998) is 25 – 35 m for Red-back Salamander (*Plethodon cinereus*), Spotted Salamander (*Ambystoma maculatum*), Blue-spotted Salamander (*A. laterale*), and Wood Frog (*Lithobates sylvaticus*), but was found to be 34 m for Redback Salamander in Hocking et al. (2013). Of note is a recent study by Maynard et al. (2016) which found edge effects on abundance and diversity of reptiles and amphibians in the Amazonian rainforest extended 100 m from dirt roads closed to traffic. This study also found humidity increased with greater distance from road edge, which may be driving the trend in amphibian distribution. Even partial canopy cover at forest edges will be avoided by some amphibians (deMaynadier and Hunter 1999), and some species can only be found in forest interiors (≥ 200 m into forest) (Ferreira et al. 2016). Although the consensus indicates that amphibians are negatively impacted by edges, Karraker and Welsh (2006) assert that silvicultural edges may contain host habitat for amphibians, however their results showed great variation in capture rates at edges; salamander abundances at edges were similar to interior forests and twice as high as clearcuts

Alternatively, edge effects on reptiles, particularly snakes, generally seem to follow the opposite response pattern of amphibians (Barrett and Guyer 2008; Grundel et al. 2014, and Vanegas-Guerrero et al. 2016), although inconsistent results among studies show this pattern may not hold true for all reptiles (Maynard et al. 2016). Black Ratsnakes (*Pantherophis*

alleghaniensis) exhibited a strong preference for edge habitats, likely for thermoregulatory purposes rather than foraging (Blouin-Demers and Weatherhead 2001; Durner and Gates 1993). For birds, edge habitats can result in higher predation rates (Donovan et al. 1997) and lower nest survival (Peak 2007), which may partially be the result of increased encounters with snakes using edges for thermoregulation (Blouin-Demers and Weatherhead 2001), or other predators that preferentially use edges as travel routes, such as raccoons (Barding and Nelson 2007).

Home of Franklin D. Roosevelt as a Focal Area for Examining Impacts of Urbanization on Amphibians and Reptiles in Protected Areas

The United States National Park Service (NPS) is mandated by the National Park Service Organic Act (1916) to promote and regulate the use of the Federal areas known as national parks, monuments, or reservations ... by such means and measures as conform to the fundamental purpose to conserve the scenery and the natural and historic objects and the wildlife therein and to provide enjoyment of the same in such manner and by such means as will leave them unimpaired for the enjoyment of future generations. Inherently, this “dueling mandate” sets the stage for conflicts between biological and anthropocentric goals that at times may be incompatible, and nowhere is this conflict better illustrated than in parks located in urban areas. One such example is the Home of Franklin D. Roosevelt in Hyde Park, New York.

The Home of Franklin D. Roosevelt National Historic Site (HOFR) and the Eleanor Roosevelt National Historic Site (ELRO) are located in the lower Hudson River Valley of New York State, a region that supports some of the greatest herpetofaunal diversity in the state (Gibbs et al. 2007). The designation of HOFR as a National Historic Site, and location ~9 km from urban Poughkeepsie, New York, (est. pop 31,000, USCB) make it an exceptional example of a shared space. HOFR supports natural areas of cultural and biodiversity significance in a highly developed landscape, bordering roads, businesses, forest, and the suburbs. The increasing awareness and popularity of outdoor recreation as a form of exercise and emotional well-being (Korpela et al. 2014), combined with the recent centennial of the National Park Service and accompanying ‘Find Your Park’ outreach campaign, predict an uptick in future visitors to HOFR. In addition, there has been recent demand for the NPS to increase the number of biking trails within their property. The NPS has a mission to preserve habitats and wildlife for the

enjoyment of people, which can be a difficult task in and of itself, especially with little understanding of the species present or the potential impacts that existing in this particularly setting have upon them.

Herpetofaunal inventory was performed by Klemens (1992) in the late 1980s on HOFR and ELRO as well as the neighboring Vanderbilt Mansion National Historic Site. Since then amphibian and reptile monitoring on all properties has been inconsistent. The previous inventory found Blanding's Turtles, a New York state Threatened species, at HOFR and ELRO, but no Blanding's Turtles have been seen since 2003. Moreover, the National Park Service (NPS) recently added two properties to HOFR – Roosevelt Farm Lane (RFL) (155.72 ha) and Top Cottage (16.15 ha)– which had not been inventoried. RFL is separated from HOFR to the west and ELRO to the east by roads (NY Route 9 and Route 9G), while Top Cottage is directly connected to easternmost side of ELRO. Both RFL and Top Cottage are bordered in places by suburban backyards. All areas support substantial recreational use by hikers, and bikers are allowed on the paved trail (Farm Lane Trail) of RFL. During tourist season, NPS offers a trolley tour that follows the paved Farm Lane Trail from HOFR to Val-Kill Cottage at ELRO.

Study Objectives

This project investigated the impacts of recreational trails on the distribution and abundance of reptiles and amphibians, in comparison and conjunction with habitat edges, within a protected area embedded within an urban landscape. It sought to answer the questions:

1. Do recreational trails affect herpetofaunal distribution and abundance?
2. Do reptiles and amphibians differ in their response to this disturbance?
3. Do trail edges differ from habitat edges in their effects on herpetofaunal distribution and abundance?
4. Do habitat and trail edges positively interact to affect herpetofaunal distribution and abundance?

To answer these questions, reptiles and amphibians were systematically surveyed at Roosevelt Farm Lane and Top Cottage properties three times in 2015, with the intent to document species presence and model species abundance in relation to habitat components, including habitat edges and recreational trails. To further investigate how habitat edges and trails affect reptile and

amphibian distribution, I collected herpetofaunal abundance and habitat data from five types of disturbed habitats (edge, edge + trail, 0, 15, and 50 m from trail) and a control within RFL. Species abundance and habitat covariates were compared to determine if and how these disturbances affect reptiles and amphibians. The aim of conducting these studies was to aid NPS land managers in informed decision making so that limited resources are allotted to maximize positive outcomes for both biological and anthropocentric aspects of this shared space. More broadly, this research documented the effects of recreational trails on reptiles and amphibians for the first time in North America, and investigated whether trails act as edges in regard to herpetofauna and how far their effects extend, as well as how they interact with habitat edges. Additionally, it also adds to the growing body of research on herpetofauna in urban environments.

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Chapter II – The Influence of Habitat Edges and Recreational Trails on the Distribution and Abundance of Amphibians at the Home of Franklin D. Roosevelt National Historic Site

Abstract

Recreational trails fragment habitat within protected areas set aside for amphibian conservation, potentially generating “edge effects” or changes in population and community structure at the boundary between two habitats. Little attention has been paid to the effects of trail edges on amphibians, which hinders conservation planning for protected areas. To address the issue, habitat associations of herpetofauna at the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, New York) were explored among five disturbance types (0, 15, and 50 m from trail, habitat edge, edge + trail) and a control. Habitat edges exerted a much stronger negative influence on amphibians than trail edges, yet trail effects did occur and extended up to 50 m from trails, primarily driven by reduction in leaf litter and coarse woody debris, whereas habitat edge effects were likely driven by change in microclimate. Effects of recreational trails and habitat edges on amphibians was additive. Given effects of trails on amphibians, protected area managers should seek to limit unauthorized trails as well as consider impacts on amphibians in future trail design and planning.

Introduction

Urbanization is a major contributor to global amphibian and reptile declines via habitat loss and fragmentation, the concomitant creation of road networks, edge effects associated with those, and predator subsidization (Alford and Richards 1999, Gibbons et al. 2000). A strong negative relationship exists between urbanization and aquatic assemblage richness and diversity among reptiles (Elston et al. 2016) and amphibians (Zhang et al. 2016). Wetlands – critical habitats for herpetofauna – that are embedded within urban areas host up to 60% lower taxonomic richness of amphibians and aquatic reptiles compared to those in more natural settings (Johnson et al. 2013). Even widespread species are scarce in urban areas (Foster et al. 2004). Amphibians are particularly vulnerable to urbanization because of their limited dispersal abilities (Cushman 2006), life cycles that are often dependent on multiple habitats (Rubbo and Kiesecker 2005) at

large spatial scales (Clark et al. 2008), and acute sensitivities to pollution and microclimatic shifts (Collins and Storfer 2003).

Transportation networks are particularly well developed in urban areas and research suggests that higher road density can negatively drive species richness (Johnson et al. 2013). Roads generate many negative consequences for herpetofauna, including habitat loss and fragmentation, direct mortality, serving as a barrier to movement, noise and chemical pollution, illegal collection of wildlife, and the introduction of non-native species (Heatwole and Wilkinson 2012). For amphibians, the negative effects of roads are equally as severe as the negative effects of habitat loss (Eigenbrod et al. 2007). Particularly for long-lived and vagile species such as turtles and snakes, even small amounts of road mortality can have a disproportionately large effect on population demography (Gibbs and Steen 2005, Row et al. 2007).

A secondary effect of both urbanization and roads are edge effects, which are the changes in population and community structure that occur at the boundary of two habitats. These are often driven by changes from edge to interior in gradients of tree cover, soil temperature, light intensity, canopy cover, vapor pressure, and canopy height, permeating 10 to nearly 120 m into the forest (Delgado et al. 2007, De Paula et al. 2016, Harper and MacDonald 2001, Marsh and Beckman 2004, Pohlman et al. 2007), and these abiotic factors can vary in times of climatic stress (Marsh and Beckman 2004), temporally, and by type of edge (Pohlman et al. 2007). For example, road edge effects can reduce reptile and amphibian abundance and diversity 100 m from the forest edge (Maynard et al. 2016), and habitat edges have been found to affect herpetofauna from 25 m (deMaynadier and Hunter 1998) to more than 200 m into the forest (Ferreira et al. 2016). Unlike habitat edges, roads generate twice their length in edges, effectively doubling the amount of affected habitat (Goosem 2007).

Protected habitats in urban areas can be subject to intense nature-based recreation. Overwhelmingly, outdoor recreation tends to have adverse effects on wildlife, encompassing consequences such as modified behavior and fitness (Taylor and Knight 2003), changes in physiology and abundance (Steven et al. 2011), increased predation (Miller et al. 1998), poaching (Garber and Burger 1995), and lowered habitat quality (Ballantyne and Pickering 2015c). A common element of outdoor recreation is the use of hiking and biking trails. Much of the trail literature is plant and soil focused, and has demonstrated that trails increase soil erosion, compaction, and nitrification, alter hydrology, damage plants via trampling, precipitate negative

changes in plant species composition, and spread weeds and plant pathogens (Pickering et al. 2010, Ballantyne and Pickering 2015c).

Because they share many similarities, trails and roads have analogous impacts on wildlife, especially concerning effects on reptiles and amphibians. Although trails are not generally associated with vehicular traffic and accompanying pollutants, they may prove similar to roads in that: they are networked structures that render vast areas of the landscape affected (Coffin 2007), they come in similar widths and substrates, habitat is disturbed and/or destroyed to build them, they are anthropogenic features managed with anthropogenic goals in mind, they create edges and associated edge effects (Ballantyne et al. 2014), they contribute to habitat fragmentation, potentially contribute to the spread of invasive species, and they increase impervious surface area.

Few studies investigating the effect of recreation have focused on herpetofauna of any kind, and those that exist tend to focus on aquatic reptiles. Several studies have indicated that freshwater turtles are impacted by recreational boating and fishing activities (Bennett and Litzgus 2014; Lavery et al. 2016; Steen et al. 2014). Burger (2001) made direct observations of snakes being captured, collected, moved, and harassed in an urban park by visitors and their dogs, and these types of behaviors may contribute to Eastern Massasaugas (*Sistrurus catenatus*) moving shorter distances in disturbed areas, and less frequently with increasing exposure to human disturbance (Parent and Weatherhead 2000). A single study, Ficetola et al. (2006), has investigated amphibians in a recreational context, and they found a negative relationship of disturbance (measured by leaf litter disturbance i.e., trampling) to reptile and amphibian species richness in an Italian urban park.

The stressors of urbanization and recreation combine in urban parks, which are fragmented habitats that are often close to roads and contain multiple habitat edges within an urbanized landscape. Urban protected areas also experience disturbance from recreationists, bikers, and dogs. Because trails are essentially modified road edges, they further fragment already disturbed habitat. In addition, urban parks may be subject to illegal wildlife collection (Garber and Burger 1995), and the effects of augmented populations of subsidized mesopredators (Ner and Burke 2008). How these combined stressors interact is unknown, although the overwhelmingly negative body of research from these disciplines individually predict that areas of overlap will also likely have effects at least as detrimental, if not more so, on

reptiles and amphibians. Balancing biological considerations with public demand for recreation can be a daunting task for protected area managers.

Historically, reptiles and amphibians are grouped together when studied because of their physiological similarities, which had led to the assumption that they exhibit the same response to stressors; however, the literature indicates amphibians and reptiles respond in contrasting ways to altitude (Brown et al. 2015), timber harvesting (Todd and Andrews 2008, Reinert et al. 2011, Ross et al. 2000), and urban-rural land use gradients (Barrett and Guyer 2008, Crosswhite et al. 2004, Grundel et al. 2014). Reptile populations may increase in more developed areas (Barrett and Guyer 2008, Crosswhite et al. 2004, Grundel et al. 2014), and fragmented habitat like urban parks may offer expanded opportunities for thermoregulation via open canopies and increased impervious surfaces. Although they may be more adaptable to urban disturbance than amphibians, reptiles could be more adversely affected by recreation because reptiles generally share the same diurnal activity patterns as recreationists (Ficetola et al. 2007). This specific trait leaves reptiles in recreational areas vulnerable because increased interactions with recreationists leads to an increased likelihood of illegal poaching (Garber and Burger 1995, Grandmaison and Frary 2012), persecution, and/or harassment (Burger 2001). In contrast, amphibians are generally more sensitive to disturbance, and thus an area affected by multiple disturbances – urbanization, recreation, road effects, and edge effects – is likely to be detrimental to their populations (Furman et al. 2016, Price et al. 2010, 2012, Cosentino et al. 2014).

To investigate the impacts of recreational trails and habitat edges in an urban protected area on the distribution and abundance of reptiles and amphibians, I surveyed two areas of an urban protected area for herpetofauna. In the following field season, I compared herpetofaunal composition and habitat variables among a disturbance gradient representing four disturbance types (trails, edges, trails + edges, and interior forest), and different distances from trails (15 m, and 50 m from trails) within Farm Lane to gain a better understanding of the fine scale effects of disturbance on herpetofaunal distribution. My objectives were to answer the questions: (1) Do recreational trails affect herpetofaunal distribution and abundance, and do trail edges differ from habitat edges in their effects? (2) Do reptiles and amphibians differ in their response to these disturbances? And (3) How do habitat and trail edges interact to affect herpetofaunal distribution and abundance?

Methods

Study Area. – These studies were conducted May – August of 2015 and May–July of 2016 at the Home of Franklin D. Roosevelt National Historic Site (HOFR), which is administered by the National Park Service (NPS) in the lower Hudson River Valley of New York State (41.7699° N, 73.9341° W). The survey focused on two recently acquired parts of the property, Roosevelt Farm Lane (RFL) (155.72 ha) and Top Cottage (16.15 ha), acquired in 2007 and 2002, respectively. RFL is located between the main HOFR property and the Home of Eleanor Roosevelt National Historic site (ELRO) (Figure 2.1). Top Cottage is adjacent to the easternmost portion of ELRO. Both areas share boundaries with residential lots, and RFL also shares some boundaries with local businesses. RFL has four formal trails, and Top Cottage has one. Hiking is allowed on trails, and biking is allowed on the paved trail (Farm Lane Trail) at RFL. These areas are within an ecoregion subsection of the Hudson Limestone Valley in the Northern Piedmont ecoregion, which is defined by low mountains and lakes throughout the landscape (Cole et al. 2012). At HOFR, primary soils include: Hoosic gravelly loam, Colonie fine sandy loam, Steep ledgy loam, and Staatsburg gravelly loam (Cole et al. 2012).

Roosevelt Farm Lane is the most intact forest within the entire HOFR site, and is dominated by hemlock-northern hardwood forest, with areas of beech-maple mesic areas with basswood and white ash trees and red maple-black gum swamps. Small patches of old field, pockets of former conifer plantation, and landscaped grounds are also present. Water sources (wetlands, streams, a pond) are abundant in RFL, and stone outcroppings are common. The hemlock-northern hardwood forest and vernal pools within RFL are considered rare and significant by the New York Natural Heritage program (Cole et al. 2012). The average elevation of Farm Lane is 76 m. Top Cottage is situated on a Taconic foothill, which holds the highest elevation on the property (137 m). Unlike RFL, Top Cottage only supports a small segment of stream and a single vernal pool.



Figure 2.1: Map of study area where herpetofauna were surveyed from May – August of 2015 and 2016 at the Home of Franklin D. Roosevelt National Historic Site in the lower Hudson River Valley of New York State. Wetlands are in light gray. Edges are speckled black and white. Insert shows location of Hyde Park within New York State.

Field Methods. – In 2015 reptiles and amphibians were surveyed across the RFL and Top Cottage properties to estimate abundance, and community structure. To do so the property was gridded into 159, ~1 ha parcels (referred to as sampling “units”), and assessed using area-constrained surveys. Two researchers walked roughly four transects ~25 m apart through all sampling units turning over cover objects (logs, pieces of bark, refuse, etc. judged large enough for an amphibian to hide beneath) while also searching for surface-active amphibians and reptiles (including calling amphibians). Hoop trapping was attempted for aquatic reptiles in the single pond and Maritje Kill, but after a week was abandoned due to inadequate water levels. Surveys were replicated three times from May to August 2015 (May 12 – Jun 5, Jun 6 – Jul 6, Jul 7 – Aug 5), and were conducted between 0800 and 1800 hrs. Each individual was georeferenced with a Garmin GPS (GPSMap 78) accurate to 10 m, and track points were automatically recorded every 10 meters while surveying in order to record effort.

NPS provided GIS layers on property boundaries, wetlands, roads, land cover type, and trails, and several of these were updated in 2016 (Table 1). Weather-related covariates were

acquired from the Northeast Regional Climate Center (www.nrcc.cornell.edu); units with multiple survey dates in a single survey had their weather data averaged.

Table 1: Data sources for habitat variables used in modeling herpetofaunal abundance at HOFR in 2015.

Data	Type	Source	Resolution/Scale	Units
Elevation	continuous	NYSGIS Clearinghouse ¹	30x30 m (raster)	meters
Slope	continuous	Elevation layer	30x30 m (raster)	degrees
Aspect	continuous	Elevation layer	30x30 m (raster)	degrees
Land cover type	categorical	USDA Forest Service ²	1:8,000	-
Roads	categorical	NPS ³	1:2,400	-
State boundary	categorical	CUGIR ⁴	1:2,000,000	-
Trails*	categorical	NPS ³	1:2,400	meters
Property boundaries	categorical	Dutchess County Real Property Tax Service Agency	unknown	unknown
Water presence*^	categorical	NYDEC ⁵ & National Wetlands Inventory	1:24,000	meters

¹New York State GIS Clearinghouse (<https://gis.ny.gov/>)

²United States Department of Agriculture

³ National Park Service at Roosevelt-Vanderbilt NHS(Contact:David Hayes, dave_hayes@nps.gov)

⁴ Cornell University Geospatial Information Repository (<http://cugir.mannlib.cornell.edu/>)

⁵ New York State Department of Environmental Conservation

*Original layer(s) updated by S.Dean in 2016

^Water presence = vernal pools, ponds, streams (streams have 0.5 m buffer), and wetlands

In 2016, over 300, 100 m² circular plots were established within RFL in order to investigate the effects of disturbances on the distribution of reptiles and amphibians in relation to six habitat types: Control (interior forest), Edge, Trail, 15 m from trail (T15), 50 m from trail (T50), and “Interaction” (trail + edge) (Figure 2.2). Edge plots were defined as being within 20 m of a habitat edge and Control plots were plots > 50 m from trails and > 20 m from habitat edges. Plot allocation was random, with slight adjustments for topographical barriers (cliff edges, streams, wetlands). Plots were sampled on non-rainy days between the hours of 0800 and 1800

from May to July 2016, and only terrestrial areas were sampled. In addition to habitat data, location of each plot was recorded using a Trimble (Sunnyvale, CA) GeoXH receiver and Nomad data collector at the plot center.

To survey each plot for reptiles and amphibians, the central point of each plot was marked, and flags were placed in the four cardinal directions marking 5.65 m from the center point. Two researchers spent 10 minutes searching the plot for reptiles and amphibians via intermediate-level sampling of Crump and Scott (1994) and Smith and Petranka (2000). Cover objects (e.g., rocks, logs, pieces of bark) that were easily moved and judged to be able to hide a salamander were checked and returned to their original position to minimize habitat disturbance. Species and number of individuals were recorded for all reptiles and amphibians encountered.

In each plot, data were collected on leaf litter cover (%) (LLC) and depth (cm) (LLD), canopy cover (%) (CC), a count of coarse woody debris (CWD) size and coverage (1–5 scale), tree size and type, and basal area (ft²/acre) (BA) following the species survey. Leaf litter coverage was estimated with a 1 m² sampling grid, where each square was equivalent to 5% leaf litter coverage. Leaf litter coverage was collected at four locations, midway between the plot center and boundary flags, and averaged. Litter depth was taken five times to the nearest cm using a metric ruler placed in the middle of the center box of the sampling grid, as well as at the center point next to the GPS pole (deMaynadier and Hunter 1998). Both leaf litter values were averaged at the plot level.

Overstory vegetation of each plot was assessed based on type (deciduous/coniferous) (DeGraaf and Rudis 1990) and measured tree diameter. Diameter at breast height of all trees in the plot was measured and sorted into four categories adapted from the US Forest Service Forest Inventory and Analysis Program (<https://www.fia.fs.fed.us/>) seedling (< 1 in/2.54 cm), sapling (1 – 4.99 in/2.54 – 12.67 cm), poletimber (5 – 9 in/12.7 – 22.86 cm), and sawtimber ($\geq$ 9 in/22.86 cm). A count of shrubs and woody plants $\geq$ 1 m in height in each plot were recorded. Basal area (sq. ft per acre) was measured from the center of the plot with a 10-factor basal area prism (JIM-GEM Rectangular Cruising Prism, Jackson, MS). Coarse woody debris was sorted into large ($\geq$ 10 cm) or small (< 10 cm) categories using a metric ruler (McKenny et al. 2006, Crane 2008) based on the Braun-Blanquet Cover Abundance Scale (Ellenberg and Mueller-Dombois 1974): 5 = $\geq$ 75% cover, 4 = 50-75% cover, 3 = 25-50% cover, 2 = 5-25% cover, and 1 = < 5% cover.

Canopy cover was assessed using fisheye photos taken using a MPOW clip-on iPhone fisheye lens (MPOW Mlens, 180° angle, 0.33X magnification, Cheung Sha, Wan Kln, Hong Kong) and an iPhone 5 (Cupertino, CA) from 1.6 m above the ground (eye level of both researchers), at the halfway points between the plot center and the boundary flags. The software ImageJ (Rasband 2016) was used to convert photos to binary, and a ratio of the number of white pixels to overall pixels (adjusted for circular lens size in a rectangular photo) was converted to a percentage of canopy cover; four photos per plot were averaged.

The width (m) of all trails within RFL was measured at 25 m intervals to calculate total trail coverage of the property. For plots with trails, the area of the trail (m²) was measured by multiplying trail width at the plot center by the length of the trail within the plot.

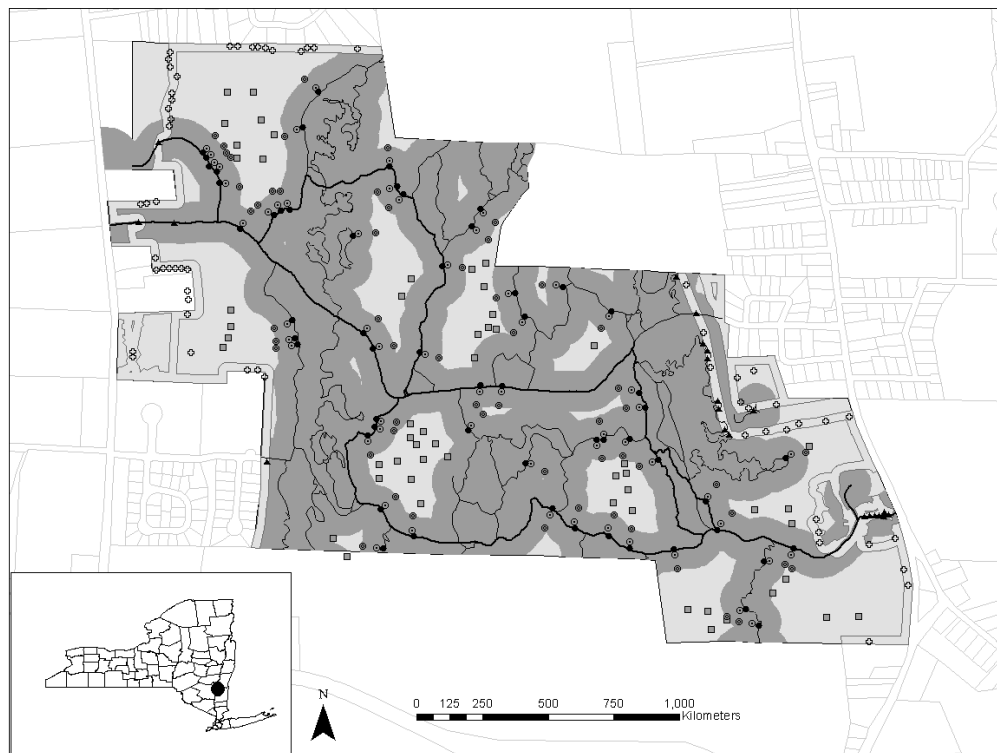


Figure 2.2: Locations of five types of disturbance plots and controls within RFL in 2016. Recreational trails are in black with a gray 50 m buffer, habitat edges are shown as a light gray 20 m buffer around edges. Plot types: Control = squares, Edge = cross, Interaction = triangle, Trail = black circle, T15 = light dotted circle, T50 = dark dotted circle. Inset shows the location of Hyde Park within New York.

Comparisons of Herpetofauna Among Habitat Types. – Counts of herpetofauna and habitat covariates on each plot were compared among the six habitat types using generalized linear

models (package stats, R Core Team 2016). Amphibian and reptile count data were modeled with Poisson error terms and habitat data ln-transformed and modeled with Gaussian error terms. Tukey's honest significant difference (HSD) post-hoc test was used to interpret significant effects from general linearized models (package multcomp, Hothorn et al. 2008).

Abundance Modeling. – Abundance was estimated using Poisson or zero-inflated Poisson N-mixture models via the R package “unmarked” (Fiske and Chandler 2011; Kery and Royle 2016, pcount function) on species with ≥ 30 individuals, including Redback Salamander, Slimy Salamander, Spotted Salamander, Eastern Newt, Four-toed Salamander, Wood Frog, and Spring Peeper, as well as aggregate species groups including Anura sp., Caudata sp., Amphibia sp., and Serpentes sp. Additionally, data on 17 state variables (elevation, aspect, slope, the coefficient of variation of elevation, aspect, and slope, trail length, edge length, percent of: coniferous forest, deciduous forest, early successional forest, developed area, landscaped area, old field, riparian area, wetlands, and aquatic habitat), and five sampling-related variables (maximum temperature, minimum temperature, growing degree days, cm of rain in prior week, cm of rain in prior day) related to detection probabilities were used in models. Models also included an effort variable (the number of auto-sampled GPS points at 10 m intervals collected during surveys within each unit) equal to the inverse of effort to account for differences in effort among sampling units (Kery and Royle 2016).

All models assumed demographic site closure in terms of abundance i.e., no changes in abundance between surveys. Species with small sample sizes (< 30), juvenile and dead animals, as well as individuals found on trails, were excluded from analyses. Models did not include any two variables with Pearson's correlations > 0.30 . Models were compared using Akaike Information Criterion (AIC, Burnham and Anderson 1998). I calculated an estimated overdispersion parameter ($\hat{c}$) to assess goodness-of-fit for top-ranked models. I considered models with $\hat{c} > 2$ to be significantly overdispersed. From high ranked, well-fitting models, I predicted abundances of species in each unit in RFL and Top Cottage. For species with competing models ($\Delta AIC \leq 2.0$), I calculated a composite model via model averaging (package AICcmodavg, Mazerolle 2016).

Results

A total of 5,246 individuals (12 amphibian, 6 reptile species) were found at RFL and Top Cottage (159, ~1 ha units) in 295 search hours (Figure 2.3) during 2015. Together, Redback Salamanders and Wood Frogs accounted for 73% of total species captures. Eastern Newts comprised 8% of captures, and Spotted Salamanders and Spring Peepers each comprised 5% of captures. On average, 18 individuals were encountered per hour.

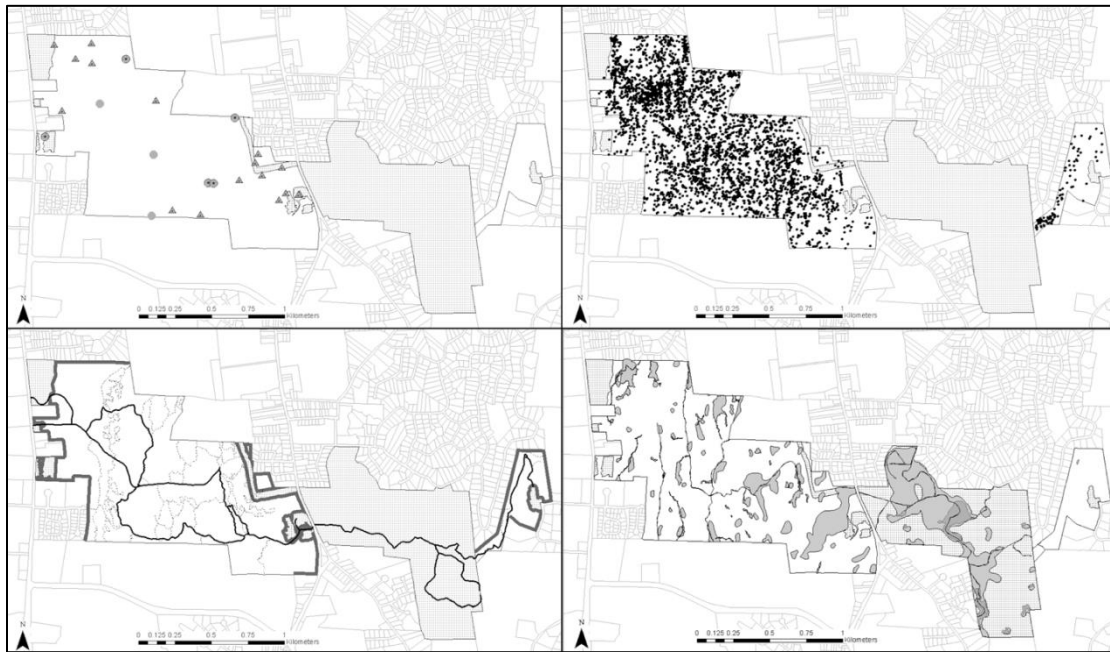


Figure 2.3: Survey results from the 2015 herpetofaunal survey and illustration of disturbances in an urban landscape context. Clockwise from top left: (1) Reptiles found at HOFR during 2015 survey; snakes = triangle, hoop trapped turtle = gray circle, E. Box Turtle = dotted circle. (2) Amphibians found during 2015 survey at HOFR (black dots). (3) Wetlands, streams, and ponds at HOFR, ELRO, and Top Cottage. (4) Edges (dark gray) and both formal (black line) and informal (dotted black line) trails present at HOFR.

Abundance Models. – Models for every species generally contained at least one topographic, land cover, and disturbance variable. Redback Salamander abundance was negatively related to elevation, percent of old field, and trail length (Table 2). Eastern Newt models included the same variables as Redback Salamander models and showed the same relationships, accounting for 99% of the AIC weight. Spring Peeper models were similar to Redback and Eastern Newt models, with the addition of percent of aquatic habitats and edge length. Wood Frog abundance was associated with elevation, deciduous forest, trail length, percentage of aquatic habitat, and edge length. Spring Peepers were influenced by similar variables, except that percent of mixed forest and old field were present as opposed to percent of deciduous forest. The greatest variation in model components was shown by Spotted Salamander, with eight variables (elevation, deciduous forest, mixed forest, old field, percent of aquatic habitat, wetlands, edge length, trail length) appearing in eight models that account for 89% of AIC weight. Trail length and a water variable were present in all Four-toed Salamander models, which also included slope, percent of early successional forest, and edge length, but only accounted for 48% of AIC weight. The top Slimy Salamander model was notable in that it was the only amphibian model to be influenced by percent of developed area. Surprisingly, snakes were influenced by similar variables (elevation mean, developed area, old field, water percentage) as amphibian species, though AIC weight was low for this model (0.56) (Table 2).

Table 2: Highest ranked abundance models for seven amphibians at RFL and Top Cottage in 2015. Model statistics are reported for the top-ranked model and those within two AIC units of the top-ranked model. All models also included a coverage offset variable. λ covariates are site-specific covariates. No detection covariates (p) were included. K is the number of estimated parameters in the model, Δ AIC is the difference in AIC values compared to the top model (models within two AIC units are averaged), w is the Akaike model weight, and $\hat{C}$ is a measure of dispersion and model fit.

Species	P covariate ^a	λ covariates ^b	K	Δ AIC	w	Cum. w	$\hat{C}$
Redback Salamander	Null	Elev/mx.for/old.fld/wetlnd/trail	9	0.00	0.96	0.96	4.41*
Slimy Salamander	Null	Slope/dec.for/dev.area/edge	8	0.00	0.61	0.61	1.39
Four-toed Salamander	Null	Slope/wetlnd/edge/trail	8	0.00	0.24	0.24	1.44
	Null	Wetlnd/trail	6	1.20	0.13	0.37	1.31
	Null	Slope/es.for/water%/trail	8	1.52	0.11	0.48	1.42
Spotted Salamander	Null	Elev/dec.for/wetlnd/edge	8	0.00	0.17	0.17	1.44
	Null	Elev/mx.for/old.fld/trail	8	0.56	0.12	0.29	1.40
	Null	Elev/dec.for/edge/trail	8	0.78	0.12	0.41	1.44
	Null	Elev/dec.for/water%/edge	8	0.79	0.12	0.52	1.48
	Null	Elev/dec.for/edge	7	0.81	0.12	0.63	1.50
	Null	Elev/dec.for/water%/edge/trail	9	1.09	0.10	0.73	1.40
	Null	Elev/mx.for/old.fld	7	1.10	0.10	0.83	1.50
	Null	Elev/water%/edge/trail	8	1.88	0.06	0.89	1.47
Eastern Newt	Null	Elev/mx.for/old.fld	7	0.00	0.44	0.44	2.24*
	Null	Elev/mx.for/old.fld/trail	8	0.40	0.37	0.81	2.05*
	Null	Elev/mx.for/old.fld/wetlnd/trail	9	1.72	0.18	0.99	2.12*
Wood Frog	Null	Elev/dec.for/water%/edge/trail	9	0.00	0.71	0.71	4.48*
Spring Peeper	Null	Elev/mx.for/old.fld/trail	8	0.00	0.33	0.33	1.58
	Null	Elev/water%/edge/trail	8	1.25	0.18	0.51	1.60
	Null	Elev/mx.for/old.fld/wetlnd/trail	9	1.38	0.17	0.68	1.55
Snakes	Null	Elev/dev.area/old.fld/water%	7	0.00	0.56	0.56	0.99

^a Null = naïve detection

^b Elev = mean elevation, Slope = mean slope, Mx.for = % of mixed forest, Dec.for = % of deciduous forest, Es.for = % of early successional forest, Dev.area = % of developed area, Old.fld = % of old field, Water% = % of area composed of water sources, Wetlnd = % of wetland, Edge = length of edge (m), Trail = length of trail (m).

*Model is significantly overdispersed

I attempted to model abundance while accounting for detection (p), however, detection probabilities were much lower than generally accepted detection probabilities (ranging from 0.004–0.15) needed for unbiased modeling (Bailey et al. 2004; O’Connell et al. 2006), and I found that detection models were not sufficiently different from null models, thus abundance was modeled without detection. Previous abundance modeling from six years of salamander surveys by Dodd and Dorazio (2004) indicates that salamander detection can vary wildly, often doubling or tripling p estimates or similarly decreasing between or among years. Counts between surveys were so highly variable that Dodd and Dorazio (2004) questioned the suitability of count data to assess salamander populations. Count data from this research (three surveys/one season) also show stark variability, and this variability likely resulted from two attributes that are specific to amphibians. In abundance modeling, p is binary, where 1 indicates presence (except in cases of misidentification), and 0 can represent either true absence, or a missed detection. An assumption of p is that if animals are truly present, it is possible for a researcher to detect them. This assumption is violated when sampling terrestrial amphibians, because animals are often underground or hiding inside coarse woody debris; i.e., they are present, but not detectable, leading to excess variability in survey counts. An additional issue with amphibians is their close relationship with meteorological patterns (Dodd and Dorazio 2004), such as rainfall, that can have effects on amphibian activity outside of the bounds of a survey period. In this survey precipitation was near normal levels for all but two weeks of the survey period (May – August 2015), however New York saw a much lower than average precipitation rate from March–May of 2015 (NOAA 2016), which may have contributed to drier than normal conditions and the flux in variation. As mentioned previously, counts can vary wildly between years, and counts for 2015 may have been below average.

Disturbance Plots. – Across 304 plots, 791 individuals (789 amphibians, 2 reptiles) of 12 species (6 salamander, 4 frog, 1 turtle, 1 snake) were found in plot surveys. Wood Frogs and Redback Salamanders were the most prevalent species, both occurring in 39% of all plots ($n = 119$, and 120, respectively). Reptile captures consisted of one Box Turtle and a snakeskin of unknown species.

The mean number of individuals found in Control plots (4.20 ± 3.35) was 6.2x higher than Interaction plots, 3.7x higher than Edge, and 1.5x higher than Trail plots (Figure 2.4). Species richness was also much higher for Control ($1.93 \text{ species} \pm 0.18$) than the other plot types nearly tripling in comparison to Edge and Interaction, and still 30% higher than Trails. Frogs appeared to be less sensitive to trail disturbance than salamanders, as frog numbers were similarly low in Edge (3.53 ± 1.10) and Interaction (3.38 ± 0.59) plots, but were 25% more abundant on trails and 30% more abundant in Control plots. Salamanders exhibited a 17 – 30% increase in abundance in Control plots (7.13 ± 2.37) compared to all other disturbance types. There were no differences in species between Interaction and Edge plots.

Plots characterized by habitat disturbance were on average 2x (Trail) to > 5x (Interaction) more likely to yield zero individuals than Control plots (plots with no species, $n = 89$, 29% of plots). All species analyzed individually ($n = 5$) except Eastern Newts showed significantly lower abundance in Edge plots compared to Control plots, with Redback Salamander and Wood Frogs showing the greatest declines of nearly 75%. In Interaction plots, Redback Salamander and Wood Frog counts were even more reduced, with > 90% declines in abundance compared to Control plots, and both Spotted Salamander and Eastern Newt were completely absent.

Plots with trails (Interaction and Trail) showed generally negative relationships with habitat variables. Leaf litter depth (LLD) was halved in plots that have trails (T, I) compared to plots that do not have trails (C, E) (Figure 2.5). Similarly, Trail and Interaction plots showed 25 – 50% less leaf litter coverage than Control and Edge plots. Plots with trails in them (T, I) had lower levels of coarse woody debris, with CWD increasing 45% from Interaction to Trail to Edge to Control. Canopy cover (CC) was generally high across plot types, between 87 and 91%, such that the few significant differences that resulted were not biologically significant. Among all habitat variables except LLC, Trail plots were not significantly different from Interaction plots, although trends indicate that Interaction plots exhibits lower values for all habitat variables.

Table 3: Means ($\pm$ SD) of habitat variables at HOFR in 2016 (southeast NY state) among six plot types: Control (n = 55), Edge (n = 60), Interaction (n = 21), Trail (n = 56), T15 (n = 56), and T50 (n = 56). Control plots were interior forest > 50 m from trails and > 20 m from habitat edges. Edges were within 20 m of habitat edges. Interaction plots were located on trails within 20 m of habitat edges. Trail plots were centered on trails, and T15 and T50 were 15 and 50 m from trails.

Variable	Mean $\pm$ SD					
	Control	Interaction	Edge	Trail	Trail 15 m	Trail 50 m
Basal Area (ft ² /ac)	126 $\pm$ 34.42	107 $\pm$ 38.72	119 $\pm$ 40.38	119 $\pm$ 44.35	121 $\pm$ 36.03	131 $\pm$ 33.61
Tree count w/height $\geq$ 1 m	10.98 $\pm$ 4.43	13.33 $\pm$ 5.32	11.15 $\pm$ 5.95	10.07 $\pm$ 5.32	13.14 $\pm$ 6.22	13.61 $\pm$ 7.96
Small CWD (< 10 cm diameter)	11.13 $\pm$ 4.92	6.91 $\pm$ 4.65	9.37 $\pm$ 5.03	8.63 $\pm$ 3.98	10.55 $\pm$ 4.94	9.14 $\pm$ 4.49
Large CWD ($\geq$ 10 cm diameter)	5.11 $\pm$ 4.31	2.24 $\pm$ 2.23	4.32 $\pm$ 4.62	3.91 $\pm$ 3.83	3.23 $\pm$ 2.93	3.83 $\pm$ 4.43
Total CWD in plot	16.24 $\pm$ 6.47	9.14 $\pm$ 4.51	13.68 $\pm$ 8.14	12.54 $\pm$ 5.96	13.79 $\pm$ 6.68	12.98 $\pm$ 6.32
Leaf litter % coverage	82.20 $\pm$ 13.04	38.32 $\pm$ 23.96	73.49 $\pm$ 23.84	60.17 $\pm$ 25.95	83.92 $\pm$ 15.83	82.48 $\pm$ 16.17
Leaf litter depth (cm)	3.79 $\pm$ 1.02	1.91 $\pm$ 1.04	3.94 $\pm$ 1.10	2.05 $\pm$ 1.03	4.25 $\pm$ 0.97	5.0 $\pm$ 1.23

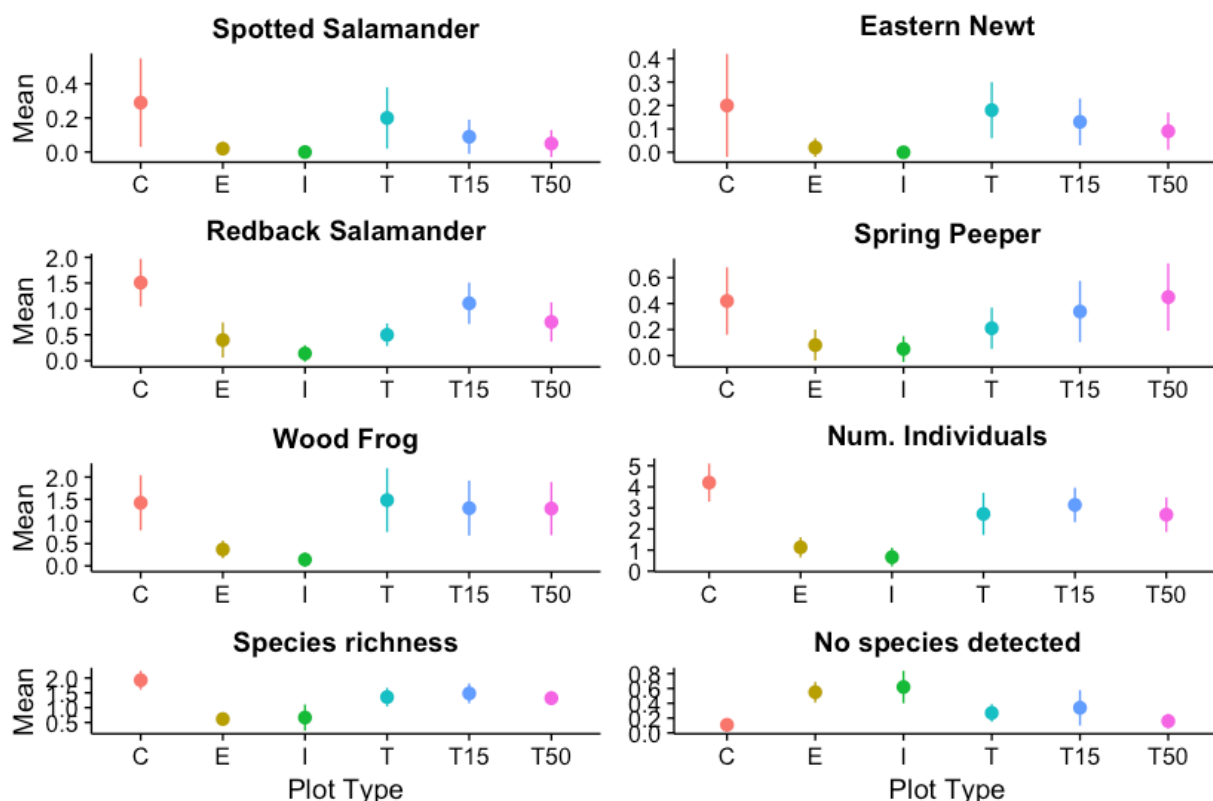


Figure 2.4: Comparison of mean relative abundance among amphibian species found within six plot types at RFL 2016 (points are $\pm$ SE), comparison of species richness among plots ($\pm$ SE), and comparison of the fraction of plots w/no species detected ($\pm$ 95% CI). Plots consisted of Control (n = 55), Edge (n = 60), Interaction (n = 21), Trail (n = 56), T15 (n = 56), and T50 (n = 56). Control plots were interior forest > 50 m from trails and > 20 m from habitat edges. Edges were within 20 m of habitat edges. Interaction plots were located on trails within 20 m of habitat edges. Trail plots were centered on trails, and T15 and T50 were 15 and 50 m from trails.

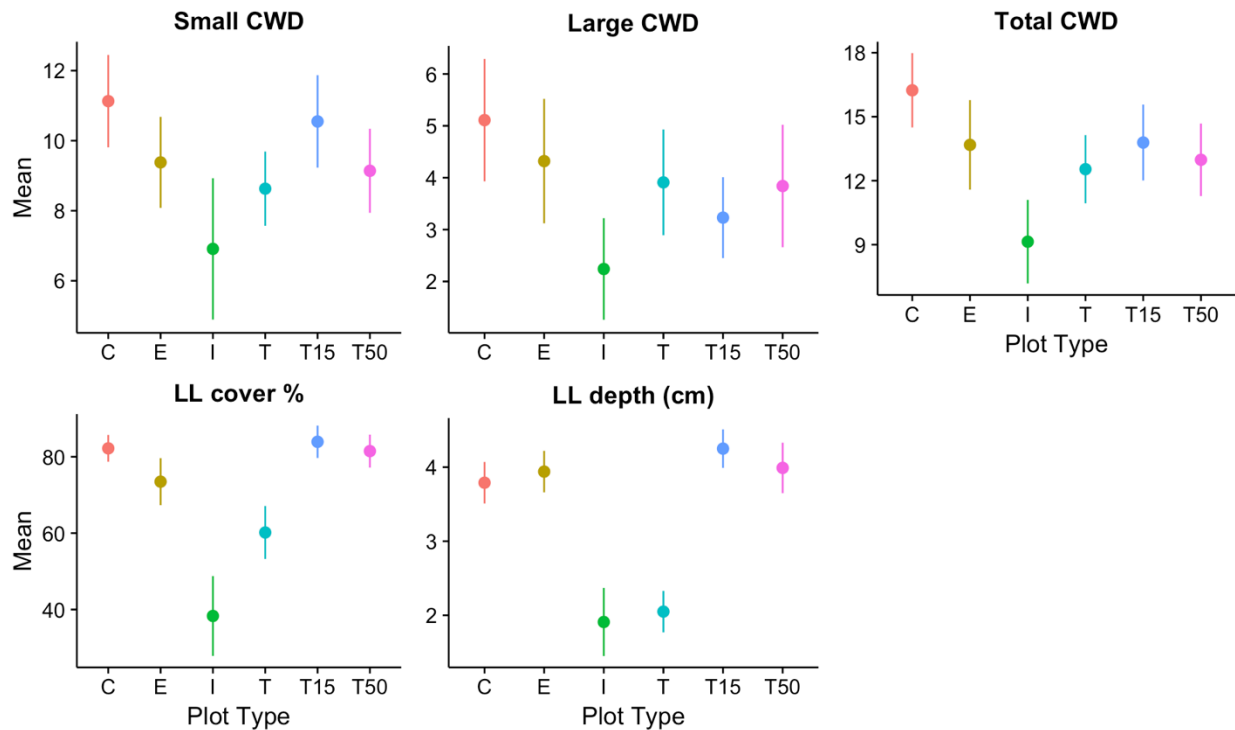


Figure 2.5: Comparisons of habitat variables among plot types surveyed in Roosevelt Farm Lane in 2016. Points are $\pm$ SE. Means of habitat variables at HOFR in 2016 (southeast NY state) among six plot types: Control (n = 55), Edge (n = 60), Interaction (n = 21), Trail (n = 56), T15 (n = 56), and T50 (n = 56). Control plots were interior forest > 50 m from trails and > 20 m from habitat edges. Interaction plots were located on trails within 20 m of habitat edges. Trail plots were centered on trails, and T15 and T50 were 15 and 50 m from trails.

Trail Effect Distance. – Trails supported lower abundances of all salamanders combined (T–C, $p = < 0.001$) and individuals (T–C, $p = < 0.001$) than did Control plots. Additionally, Trail plots held lower mean numbers of all species and groups tested except Wood Frogs, and supported significantly lower means of Redback Salamander ($p < 0.001$). T15 and T50 plots also sustained lower average numbers of all species and groups versus Control plots, except for Spring Peepers in T50 plots, however, all but one of the Control–T15 comparisons were non-significant, and the significant Control–T50 results mirror those of the Control–Trail comparison.

There was no relationship between likelihood of finding no species, or counts of Wood Frogs, Ambystomid salamanders, and Eastern Newts with increasing distance from the trail. In contrast, average counts of Spring Peepers increased with increasing distance from the trail,

surpassing even Control (0.42 ± 0.94) at T50 (0.45 ± 0.95). A less clear relationship existed for *P. cinereus* abundance, which doubled from Trail plots to T15 (0.50 to 1.11), decreased from T15 to T50 (1.11 to 0.75), and doubled again between T50 and Control (0.75 to 1.51). Species richness and the number of individuals found exhibit a similar pattern, with increases between Trail and T15 plots, decreases between T15 and T50, followed by another increase between T50 and Control plots. Anurans as a group show a non-linear pattern similar to species richness, except that instead of increasing between Trail and T15, they decrease in abundance (4.77 to 4.68).

Discussion

In this study of amphibian and reptile communities in an urban protected area, habitat cover, low elevation related to water proximity, and habitat edge-related variables drove much of the amphibian distribution, whereas edge-related variables were the strongest predictor of snake abundance. Of the disturbances investigated in this study, trails had a smaller impact than habitat edges or trails and edges together on amphibian distribution, but their effects could be species-specific and are not always negative. In comparing across different types of disturbance plots, plots containing a disturbance (habitat edge, trail, or both) supported lower abundance and diversity of forest herpetofauna than control plots with no disturbance, confirming that recreational trails generally negatively impact herpetofauna. These effects extend up to 50 m from the trail. From Trail to Control plots, Redback Salamander abundance triples, and the number of individuals (of all species) increases 35%. In addition, trails and habitat edges interact in places of overlap, resulting in further declines in amphibian abundance and richness, to the point of extirpation of certain species. Trail effects were driven by reduced amounts of leaf litter and coarse woody debris near trails, whereas edge effects were likely driven by other microclimate variables that were not measured in this study.

The effects of trails in our models tended to be species-specific and were not particularly strong. Increased elevation and increased slope were generally associated with lower amphibian abundance, probably because low elevations are indicative of water sources such as wetlands, which are associated with herpetofauna (Houlahan and Findlay 2003, Garrah et al. 2015). Wood

Frogs and Slimy Salamanders were the only species associated with increased elevation. Although obligate vernal pool breeders, Wood Frogs are more vagile and surface active, and are likelier able to exploit drier areas than salamanders (Gibbs et al. 2007). As Plethodontids, Slimy Salamanders are fully terrestrial and not wetland dependent, which may be why they were found at higher elevations. The negative association of adult Spotted Salamanders with wetlands and percentage of aquatic habitat likely reflects adult habitat selection patterns of these vernal pool specialists. Higher, negative beta coefficients with edge length and Spotted Salamander abundance suggest habitat edges may be the most important factor driving Spotted Salamander abundance. Spotted Salamanders are probably avoiding wetlands and deciduous forest that also happen to be edge habitats, rather than avoiding these habitats entirely.

In every model it appeared, habitat edge length was negatively associated with abundance. This relationship was true regardless of the presence of trail length in the model, suggesting that for amphibians habitat edge effects are more important in determining abundance than the effects of trail edges. Percent of old field also negatively affected abundance of amphibians, but positively influenced snake abundance. Snakes are generally thought to prefer habitat edges because it provides more basking opportunities (Blouin-Demers and Weatherhead 2001). Developed area, which is also associated with edges, was not a common variable in models, but when present, had strikingly large, negative beta coefficients. In particular, snakes exhibited a strong negative association with developed area, which is contrary to findings for reptiles from previous research (Barrett and Guyer 2008; Grundel et al. 2014) which found reptile abundance increased in habitats associated with developed areas. Reptiles were scarce at Roosevelt Farm Lane and Top Cottage throughout fieldwork for this study, precluding more specific (i.e. to species) analyses of trail effects on reptiles. Though possibly an artifact of the detectability of reptiles which is generally very low (Refsnider et al. 2011) or a location-specific idiosyncrasy, this observation may indicate that urban parks do not provide adequate habitat for reptiles of the Northeast.

Plot surveys support the model results for the negative effects of habitat edges for amphibians. A comparison of Interaction plots to Edge and Trail plots indicate that habitat edges are more of a disturbance than trails in Interaction plots, further shown by the lack of difference in species means between Edge and Interaction plots. Habitat edges have lower soil moisture content than interior areas (Malmivaara-Lamsa et al. 2008), and this makes edge areas poor

habitats for amphibians, especially Plethodontid salamanders, that must maintain a certain moisture level to keep their skin functioning as a respiratory organ (Heatwole 1960). As ectotherms, the activities of all herpetofauna are moderated by temperature, and edge areas generally offer a more variable microclimate, versus a stable microclimate offered in interior forests that is regulated by the forest canopy (Roloff et al. 2011). Changes in amphibian abundance are often reflective of forest invertebrates (Welsh and Droege 2001), and edges may also possibly support lower numbers of amphibian prey, similar to the reduction of invertebrates on road edges found by Delgado et al. (2013).

Previous research by deMaynadier and Hunter (1998) have offered evidence that salamanders are more sensitive to edges than frogs (although both groups have habitat generalists), and they posit that openings in the forest canopy may be especially deleterious for lungless salamanders. Such patterns were evident in this study in which low counts on Edge plots for Redback Salamanders, Spotted Salamanders, and Wood Frogs were observed. DeMaynadier and Hunter (1998) specifically identified Wood Frogs as vulnerable to habitat edges; I also found Wood Frog abundance was significantly decreased in plots containing habitat edges (E, I) but not by trail edges. Smaller edge gaps have effects on microclimates (Pohlman et al. 2009) and can affect even generalist herpetofauna, such as the disparity in skink abundance found by Anderson and Burgin (2002), who observed skink abundance in areas separated by powerline corridors to be half that of sites located at the core of small habitat remnants. Other studies have found juvenile amphibians to also avoid power-line edges (DeMaynadier and Hunter 1999).

The magnitude and distance of edge effects are likely a direct function of the contrast between the communities on either side of the edge (Harper et al. 2005). The gaps provided by trails create disruptions in habitat continuity, and these gaps tend to be small. Loss of some canopy likely affects other microclimatic variables such as ambient light and temperature, but the degree of these effects, as well as other climatic effects such as wind (Pohlman et al. 2009), may be buffered by the surrounding tree canopy and forest. Canopy gaps created by trails may mimic a natural canopy disturbance, such as openings in the canopy via dying trees, streams, or windthrow, and this may be the reason that amphibians previously noted to be sensitive to habitat edges such as Wood Frog and Spotted Salamander, did not show comparable responses to trail edges, and why several species showed positive responses to trails in the models.

Even though Edge plots exhibited vast differences in species' abundances in comparison to Control plots, there were few differences in habitat variables between these habitats. It is interesting that of Edge and Trail plots versus Control, Edge showed greater abundance disparities and Trails exhibited greater habitat disparities. Other studies suggest that habitat edge characteristics not measured in this study may be contributing to the relationship in amphibian abundance between Edge and Control plots. Prior studies have shown changes in soil temperature (Delgado et al. 2007), humidity (Maynard et al. 2016), and predation rates (Temple 1987) at habitat edges, and it is possible that these factors are driving habitat edge differences in amphibian abundance.

The negative effect of trails on other biota have suggested that trails may be detrimental to both habitats and wildlife (Taylor and Knight 2003), but our results suggest this is not always the case. In our models, four species of amphibians were positively associated with trails (Spring Peepers, Wood Frogs, Spotted Salamanders, and Four-toed Salamanders), whereas two species (Eastern Newts and Redback Salamanders) were negatively impacted by trails. To standardize survey effort, animals found on trails were omitted, resulting in the Eastern Newt sample size to be reduced by > 40% (n = 185). Including these animals would have certainly changed the association of Eastern Newts with trails in our models. Prior research by Marsh and Beckman (2004) found fewer Redback Salamanders within 20 m of forest roads, which could be similar to the relationship seen here with trails. Of the three species with sufficient plot disturbance sample sizes that showed positive associations with trails in our models, Spring Peepers and Spotted Salamanders were most abundant at 50 or more meters from the trail while Wood Frog abundances between Trails and Control plots were nearly identical. Spring Peepers showed a trend of increasing abundance with increasing distance from the trail, while Spotted Salamanders decreased from the trail up to 50 m, and increased after 50 m. Spring Peepers may be more affected by trail disturbances than Wood Frogs because they are smaller and more prone to predation. Spotted Salamanders may be benefiting from trail maintenance practices that increase coarse woody debris close to trails, or possibly because trails happen to be located close to preferred habitat such as wetlands.

The plot surveys revealed higher Redback Salamander counts in Control plots versus plots containing a trail (T, I). Both Trail and Interaction plots supported significantly lower total coarse woody debris and leaf litter depth than Control plots and these habitat characteristics may

be driving differences in relative amphibian abundance. Generally, amphibian counts are positively correlated with downed woody material (Hicks and Pearson 2002, McKenny et al. 2006), and this relationship is especially strong with debris that is large and well decayed (Otto et al. 2013). The relationship is partially due to increased invertebrate abundance with larger coarse woody debris (Riffell et al. 2010), and because amphibians use them as refugia and nesting areas. Amphibians may be avoiding trails in favor of other habitats due to lack of coarse woody material. Of note is that formal trails – those that are maintained by land stewards – may have artificially high amounts of coarse woody debris compared to informal trails. Regular trail maintenance, specifically dealing with downed trees that are blocking the trail, generally involves cutting the tree into large chunks and rolling these logs to areas immediately adjacent to the trail. Because these logs tend to be large in size, given decomposition time, they may be attractive to amphibians. When trees are blocking informal trails, the general response from recreationists is to reroute the trail around the offending obstacle, which may or may not offer new woody debris to amphibians, depending on whether the tree is on the ground. In contrast to artificially high amounts of woody debris by trails, studies of edge effects of gravel roads have indicated that cover objects tend to decrease near roads (Marsh and Beckman 2004).

The abundance patterns among amphibians in plots with trails are also partially attributable to differences in leaf litter, which was reduced in areas with trails. Compared to Control and Edge plots, average leaf litter depth was essentially halved in Interaction and Trail plots. Percent coverage of leaf litter is drastically reduced in plots with trails; Trail plots show a 25% reduction in leaf litter coverage, and Interaction plots a 50% reduction. Redback Salamanders are positively associated with leaf litter depth in New York (Pough et al. 1987). Petranka et al. (1993) reported reductions in leaf litter to be the most important variable in determining presence of terrestrial salamanders. Large amounts of leaf litter retain moisture for several days after rain, which, combined with the high canopy cover across the site, creates ample foraging opportunities for amphibians (Harpole and Haas 1999). There is a direct negative relationship between leaf litter and trampling intensity (Kissling et al. 2009), and this effect is clearly evident in areas with trails, particularly dirt trails. Litter disturbance, particularly through trampling, has previously been found to negatively influence amphibian abundance (Ficetola et al. 2007).

Trampling is a direct effect of recreational trail use, and it can result in vegetation loss (Monz 2002). This is particularly important issue with more permanent trails that are also generally

wider, creating a larger gap in the canopy, and thus less leaf litter to begin with. More study is needed to investigate how the relationship between trail type and trail width influences forest amphibians.

Trail effect distance was species-specific and in some cases mirrored habitat changes. There exists a humped distribution between Trail and T50 plots for Redback Salamanders, and this was also mirrored by small coarse woody debris, which indicates that an increase in relative abundance is reflective of the amount of habitat. Whether this is a result of trails generally, or only of more recently formed trails (not distinguished in this study) is unknown. In contrast, the trend for Spotted Salamander and Eastern Newt was to decrease from Trail to T15 to T50, and then to increase within Control plots, which were > 50 m from trails. The only species that appeared to be unaffected by trails was Wood Frogs, which showed very similar abundances between Control and T/T15/T50 plots.

Trails seem to have less of an effect than elevation, edges, and in some cases, forest type and wetlands, and these other variables may be driving the relationship between trails and herpetofaunal abundance. Control areas (i.e., core habitat) are in such limited supply (see Figure 2.2) that they may be saturated, forcing some amphibians to live in sub-optimal habitats that may be within 50 m of trails. Second, trails are not random across the landscape, and it may be the case (see Figure 2.1) that trails are coincidentally located in low-lying areas, near preferred amphibian habitats because recreationists also prefer those areas. Third, not all trails are identical, and formal (e.g., paved/gravel) trails may have a greater impact than informal (dirt) trails. In RFL, formal trails ($n = 4$) have nearly double the average width of informal trails ($n = 40$; 2.38 m vs. 1.22 m, respectively), and cover the same amount of area (1.5 ha), but account for less than half the length of informal trails (5.3 km vs. 12.9 km). Research on road-edge effects suggest that the width of the habitat gap is a good predictor of the magnitude of the edge effects (Marsh 2007). It may be that wider formal trails have a negative influence on amphibian abundance, but informal trails, which are narrower, do not. Alternatively, as I focused on trail length but not trail type, it is important to note that informal trails account for a disproportionate length of trails in the models, but may have a lesser effect than formal trails.

In this study I was unable to account for detection in abundance models due to the low detection estimates. Although my sample size was similar, prior studies estimating amphibian detection have used multiple years of data (Dodd and Dorazio 2004) and/or multiple sampling

types (Bailey et al. 2004) to achieve reasonable estimates of detection. Sampling at Roosevelt Farm Lane and Top Cottage across years in conjunction with other methods such as cover boards or drift fences may allow for detection-corrected abundance modeling in the future.

I focused on the effects of recreational trails themselves, but not on the effects of the recreationists that use them. Noise and vibrations generated from recreationists may alter behavior of herpetofauna, but as of yet this has not been tested in a recreational context. I focused on terrestrial species, but aquatic herpetofauna may also be impacted by recreational trails located close to water sources, perhaps from excess sedimentation from erosion caused by dirt trails (see Price et al. 2012). I was not able to distinguish the effects of bicycling on herpetofauna because bikers and/or tire tracks were encountered on every trail, including trails where biking is technically not allowed. The impact of bicyclers versus hikers on herpetofauna warrants future study. Although trail cameras are present at the eastern and western official entrances to RFL, multiple social trail entrances provide ample opportunities for visitors to enter or exit the property without passing a trail camera, precluding investigation of visitor load on herpetofauna.

The 50 m distance that trails effected salamander abundance may be specific to urbanized areas. As an urban park, HOFR potentially experiences increased predation rates (Feinberg and Burke 2003), heavy stream flows during rain events (Price et al. 2010), increased pollution inputs (Cole 1993), and an overall greater amount of disturbance than more rural sites. Because it is constrained by the development surrounding it, HOFR is also likely smaller than more rural parks. Larger, natural areas that experience less disturbances may have greater amounts of core (interior) habitat to buffer the negative effects of disturbance, and thus trail effects in those areas may penetrate smaller distances. Future research should compare trail effects on herpetofauna in urban versus rural areas to explore these relationships.

Management Implications. – Urban parks play an important role in connecting the public with nature in addition to their biological services, and thus their existence is a valued part of society. However, our results along with previous trail-related research indicate that there are ecological tradeoffs associated with the presence of trails. Although urban park management is context specific, managers can take action to limit the effects of these disturbances on amphibians by

considering trail impacts on amphibians and their preferred habitats in trail design, and in trail reduction and proper trail maintenance strategies (see Marion 2016).

Reducing the number of trails, particularly informal trails, will limit the impacts of recreational activities on herpetofauna by reducing fragmentation and both habitat and animal disturbance. This will in effect create more core habitat that is not negatively influenced by edges or trails. Informal trails are often unplanned, and as such are usually poorly located with respect to terrain and often drastically exceed formal trails in number (Moskal and Halabisky, unpubl. data, 2014); their lack of maintenance increases their potential to degrade habitats (Marion et al. 2006). Additionally, informal trails can create a visually scarred landscape and can potentially lead to liability concerns (e.g., trails created in unsafe areas) (Marion et al. 2006). Managers should strive to keep maintained trails well marked, which generally keeps visitors on formal trails by limiting confusion with informal trails, and also decreases trail width from excessive trampling (Wimpey 2009). Additionally, I recommend that managers restrict the number of trail entrances into their properties, which restricts the number of areas that have particularly negative influences on reptiles and amphibians, but also assists in centralizing visitor flow i.e., keeping visitors on formal trails (Ballantyne et al. 2014).

Conclusion

I document for the first time that recreational trails negatively influence herpetofaunal distribution and abundance in an urban protected area in North America. I investigated how two disturbances, habitat edges, and recreational trails, affected the distribution of herpetofauna in the context of a highly developed area, and found that both disturbances, alone and in combination, affect the distribution and abundance of herpetofauna. Habitat edges negatively impact amphibian populations more so than trail edges, however, in combination their effects are additive. Trail effects can extend up to 50 m from trails but this effect is species-specific, and is perhaps related to forest patch size. Land managers should limit informal trails and trail network entrances to minimize the negative effects of trails on reptiles and amphibians.

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Appendix I: Survey and Mapped Abundance Model Results from 2015 Herpetofaunal Survey at the Home of Franklin D. Roosevelt National Historic Site

Table A1.1: Species totals by search effort for 2015 survey of Roosevelt Farm Lane and Top Cottage. Survey consisted of 295 search hours and 46 hoop trap nights.

Species	Individuals	% of total individuals	Individuals per search hour
Redback Salamander	2249	0.43	7.6
Slimy Salamander	104	0.02	0.35
Spotted Salamander	240	0.05	0.81
Four-toed Salamander	43	0.008	0.15
Two-lined Salamander	9	0.002	0.03
Marbled Salamander	31	0.006	0.11
Eastern Newt	441	0.08	1.49
Gray Treefrog	37	0.007	0.13
American Bullfrog	5	0.001	0.01
American Toad	6	0.001	0.02
Green Frog	51	0.01	0.17
Wood Frog	1572	0.30	5.33
Spring Peeper	224	0.05	0.76
E. Box Turtle	6	0.001	0.02
Black Ratsnake	12	0.002	0.04
Black Racer	2	0.0004	0.007
Gartersnake	3	0.0006	0.01
Painted Turtle	3	0.0006	0.07 (46 trap nights)
Snapping Turtle	2	0.0004	0.008 (46 trap nights)

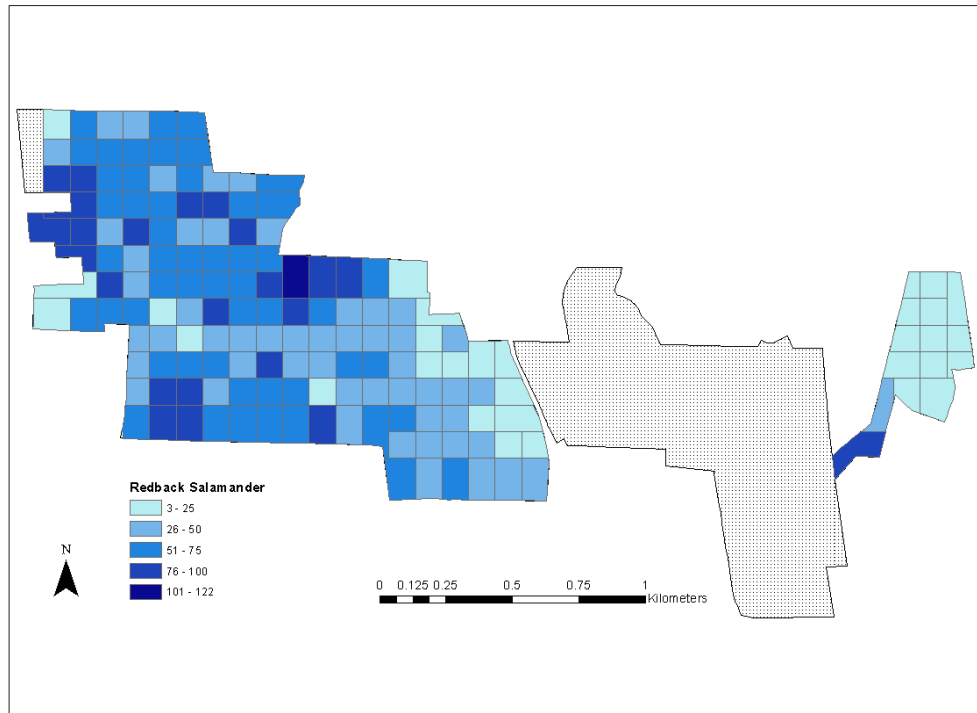
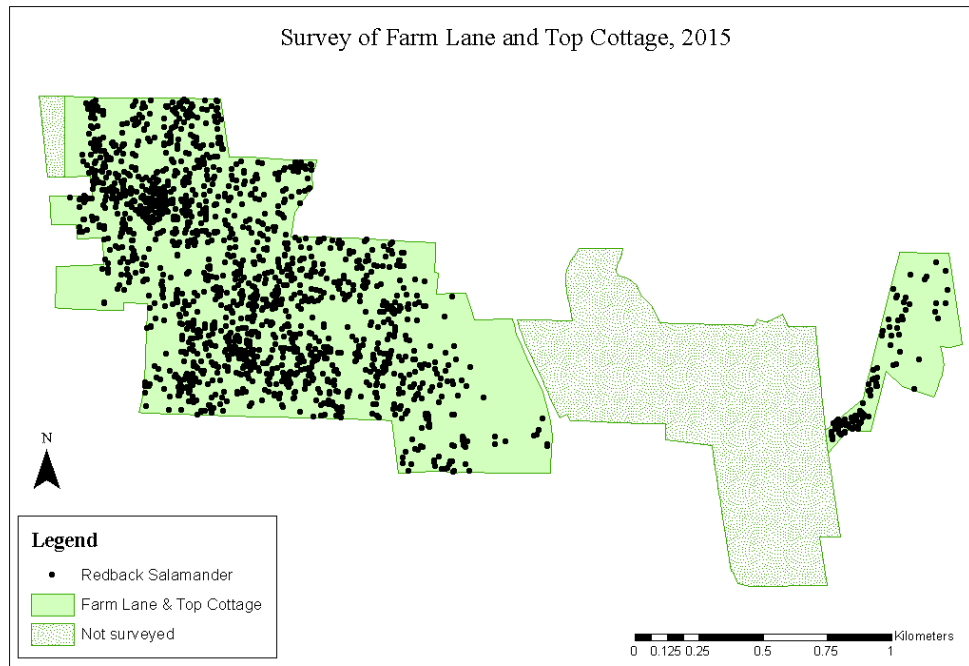


Figure A1.1: Predicted abundances at Farm Lane and Top Cottage at the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY) for Redback Salamander. Darker blue indicates a higher relative abundance.



Redback Salamander *Plethodon cinereus*

Figure A1.2: Redback Salamander locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).

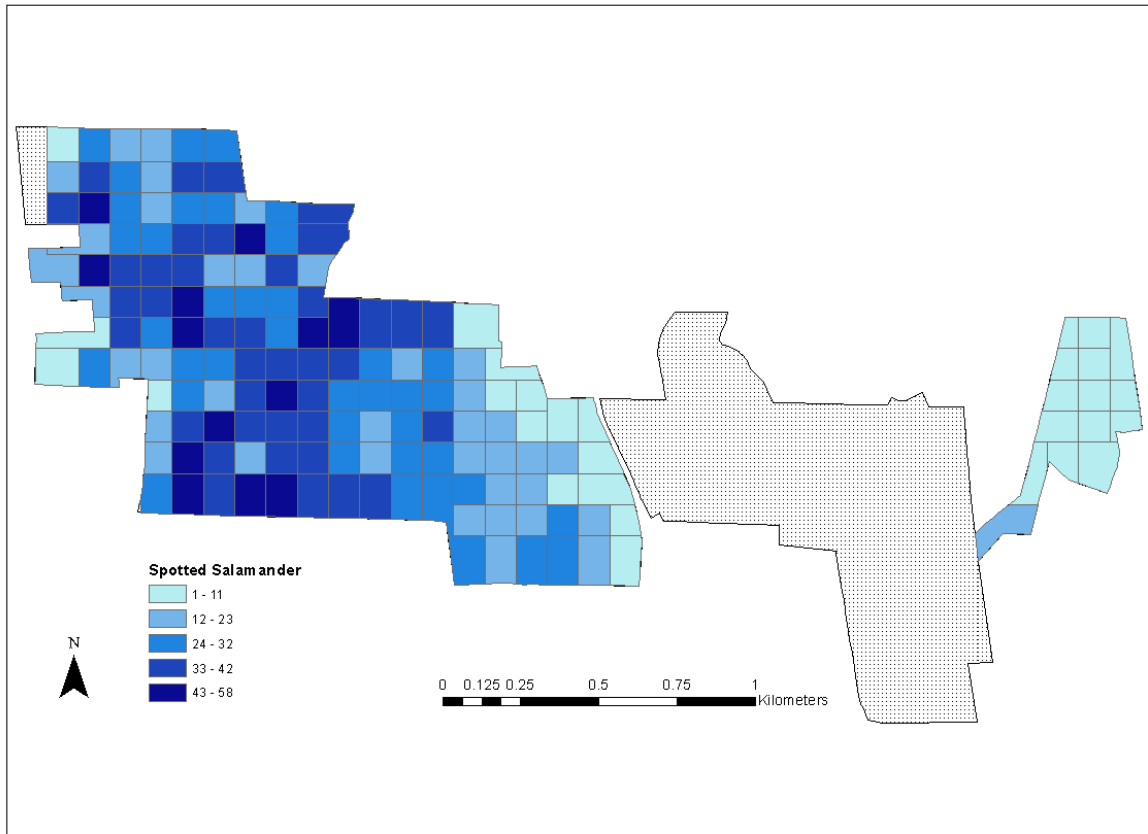
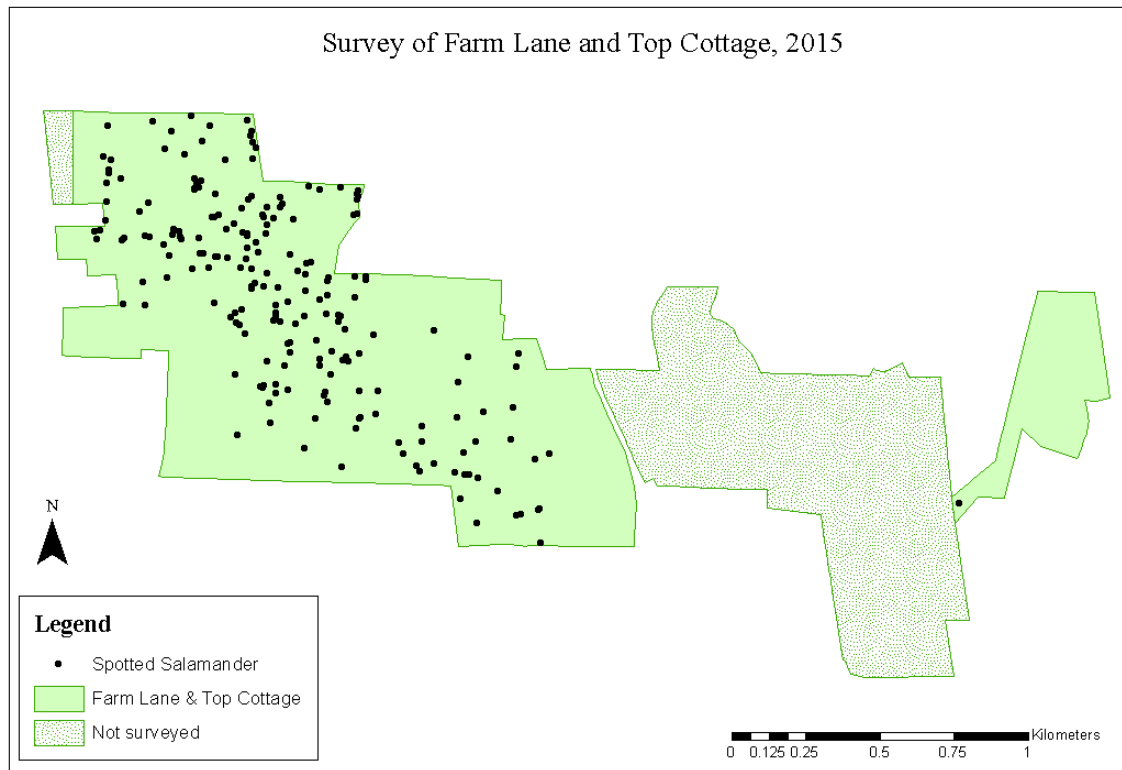


Figure A1.3: Predicted abundances at Farm Lane and Top Cottage at the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY) for Spotted Salamander. Darker blue indicates a higher predicted relative abundance.



Spotted Salamander

Ambystoma maculatum

Figure A1.4: Spotted Salamander locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).

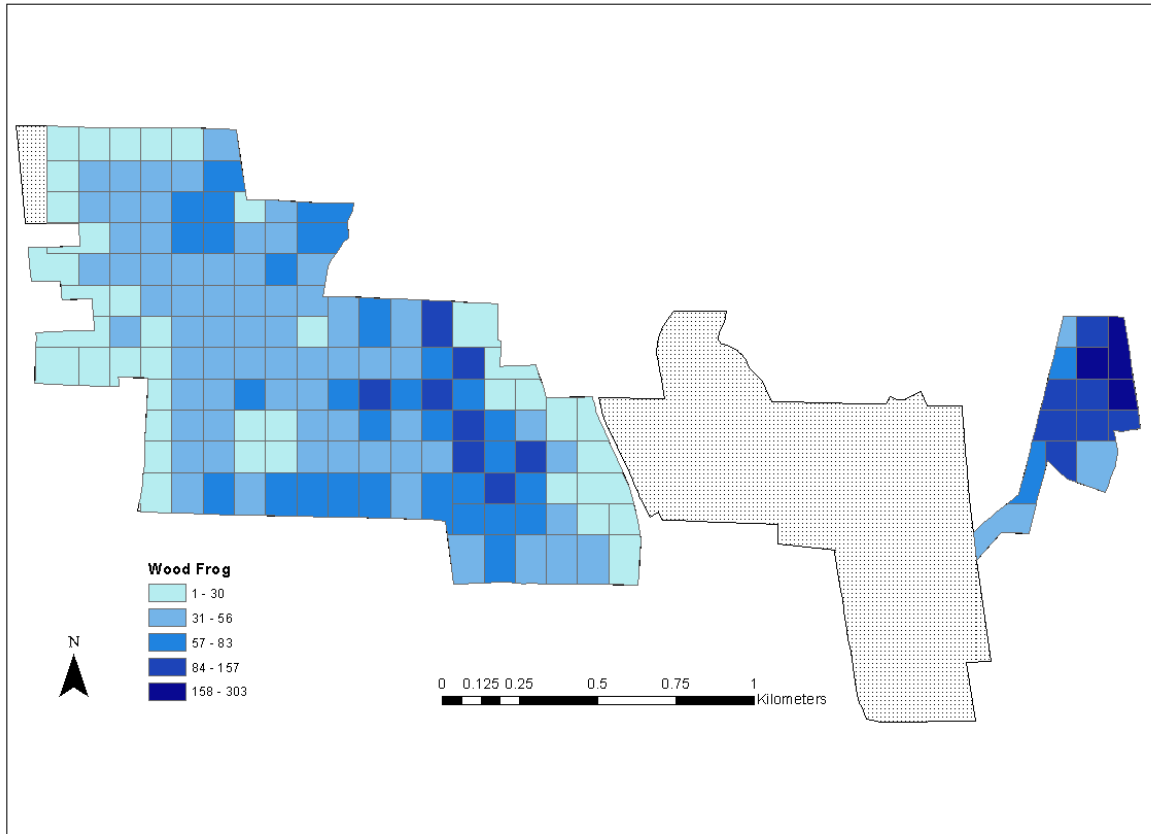
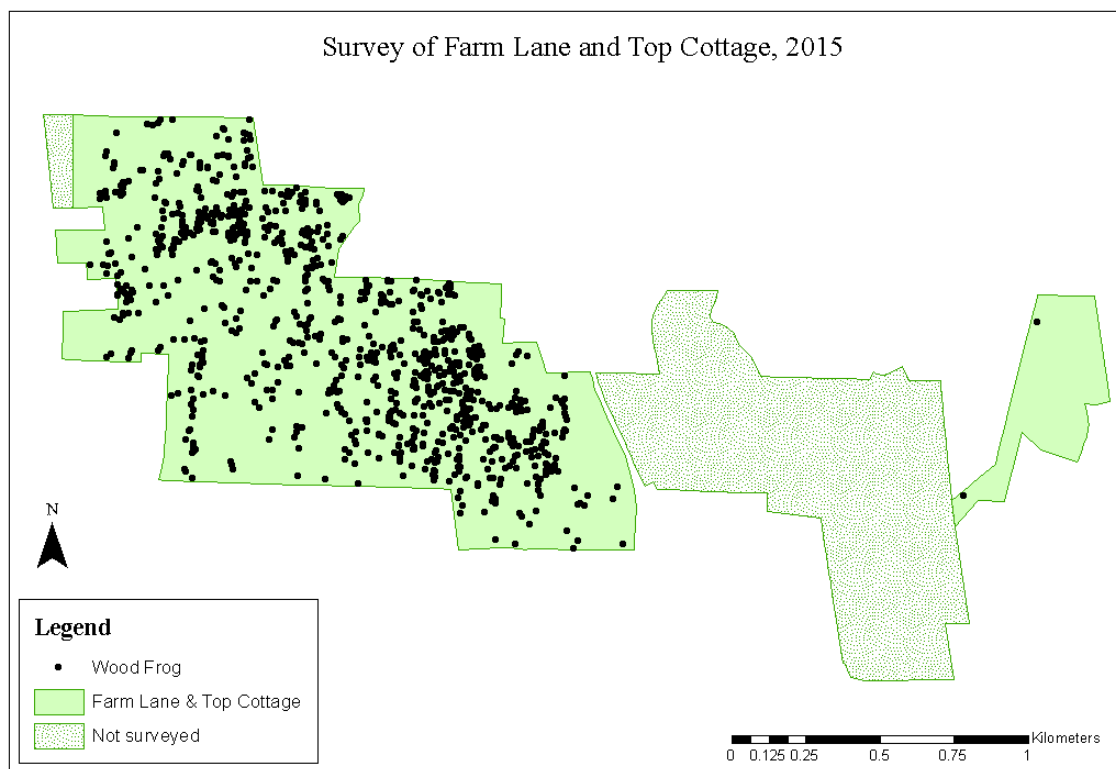


Figure A1.5: Predicted abundances at Farm Lane and Top Cottage at the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY) for Wood Frog. Darker blue indicates a higher predicted relative abundance.



Wood Frog
Lithobates sylvaticus

Figure A1.6: Wood Frog locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).

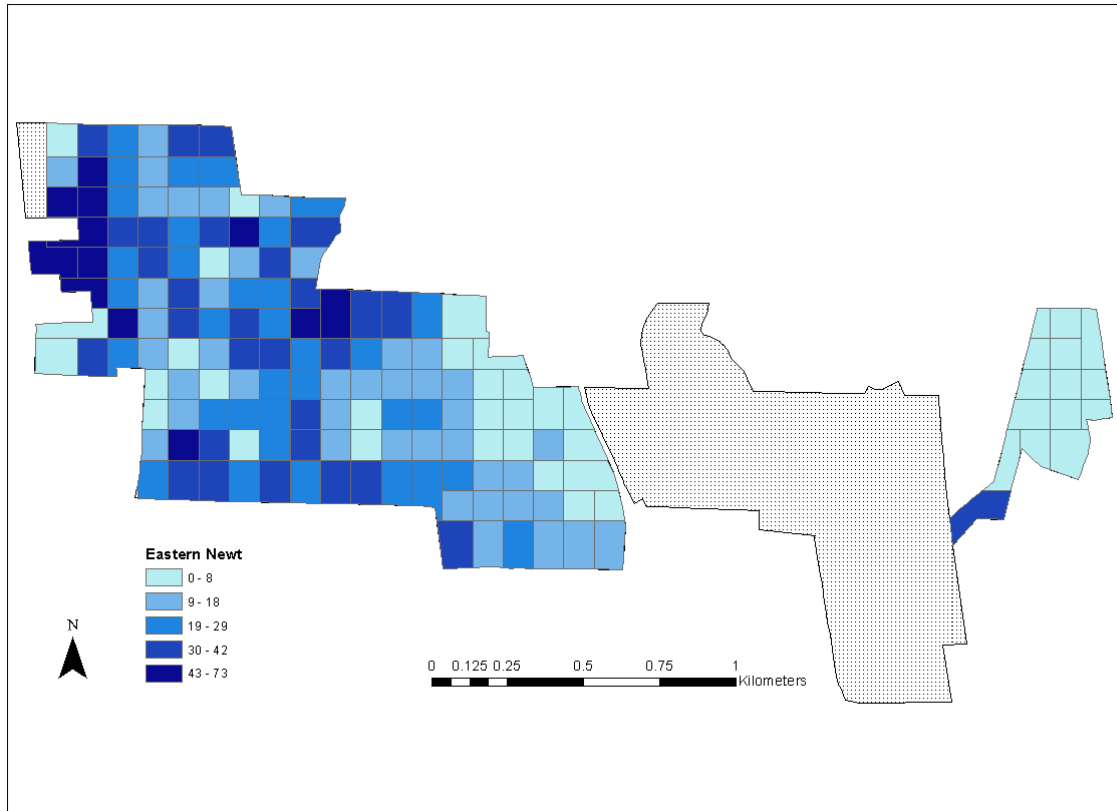
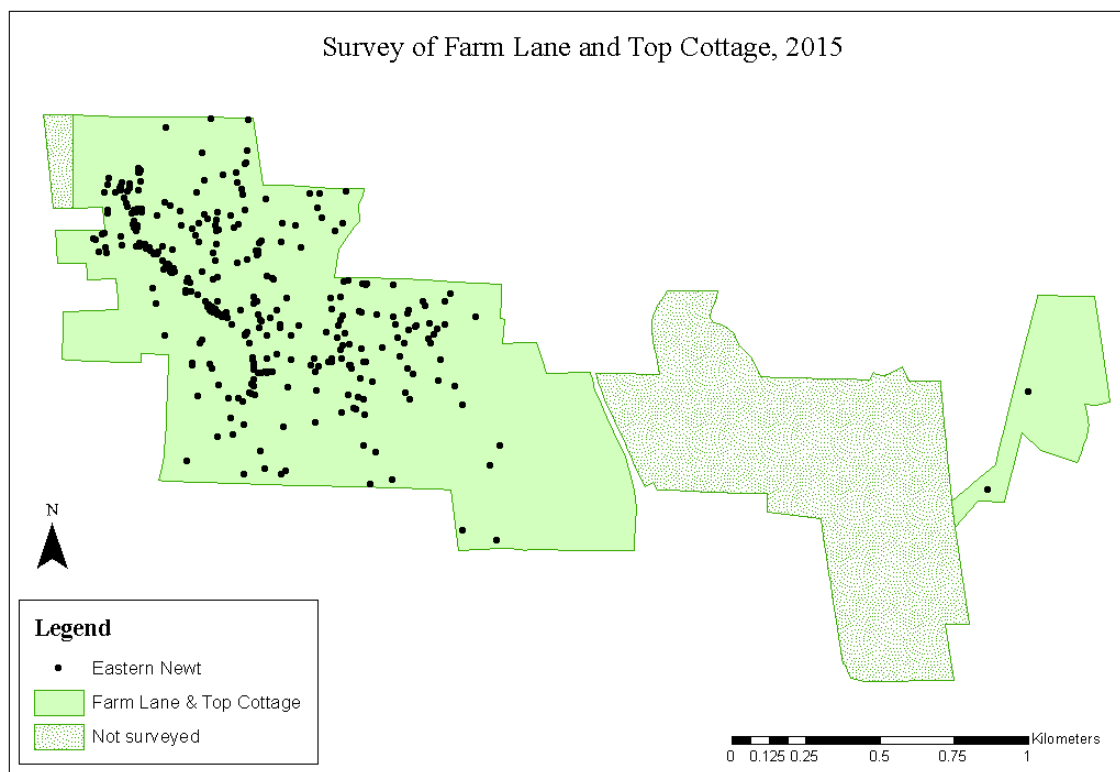


Figure A1.7: Predicted abundances at Farm Lane and Top Cottage at the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY) for Eastern Newt. Darker blue indicates a higher predicted relative abundance.



Eastern Newt
Notophthalmus viridescens

Figure A1.8: Eastern Newt locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).

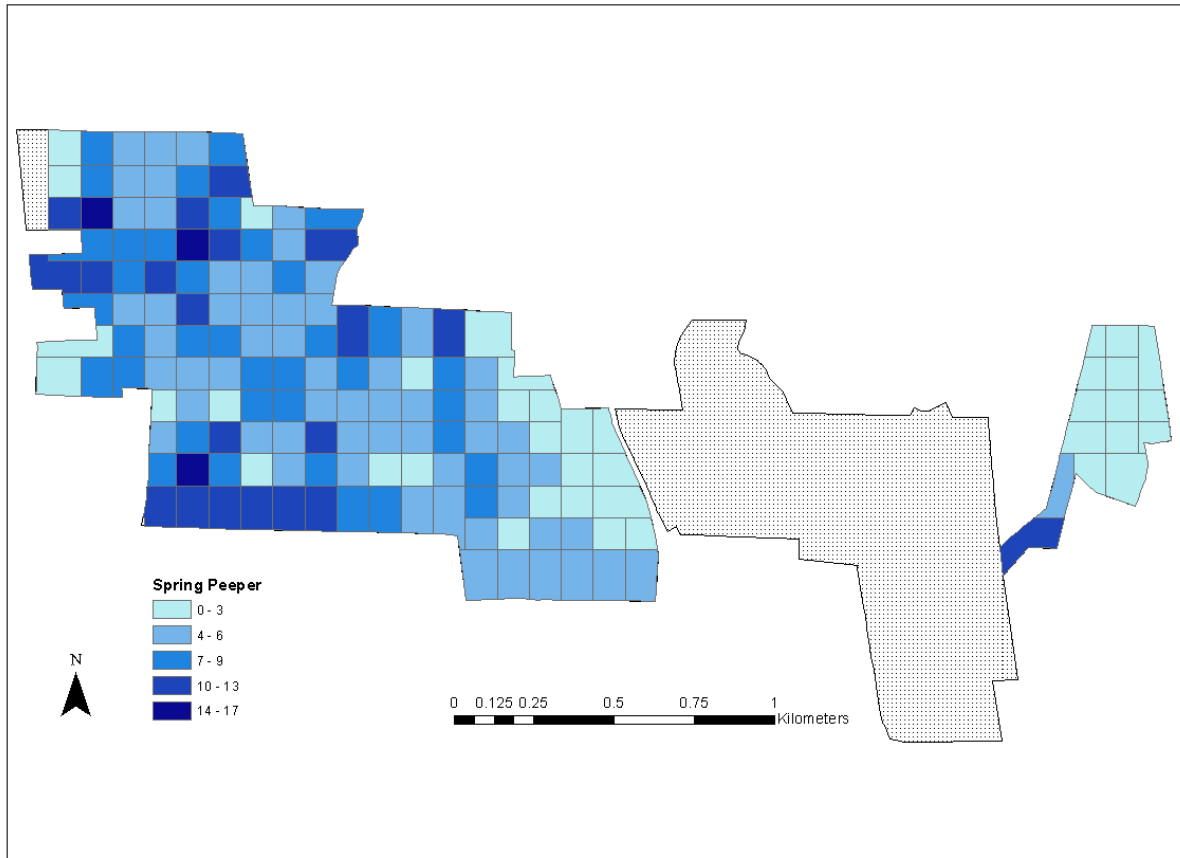
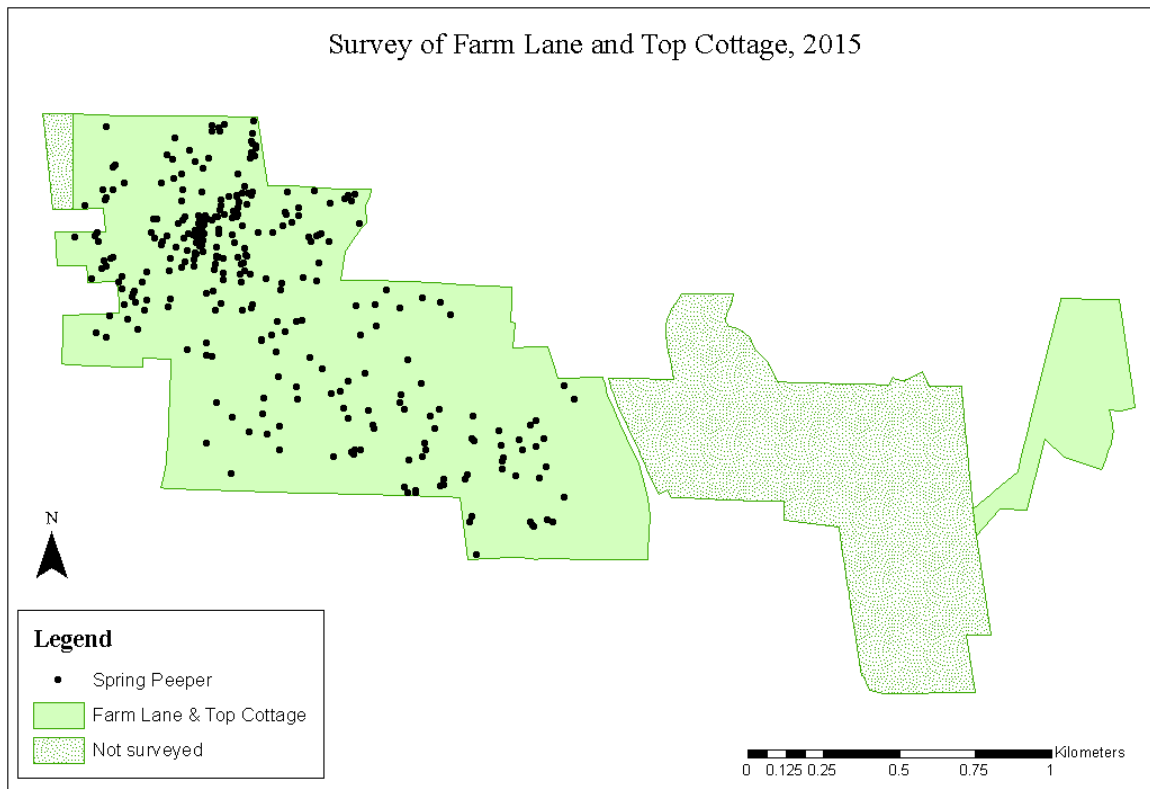


Figure A1.9: Predicted abundances at Farm Lane and Top Cottage at the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY) for Spring Peeper. Darker blue indicates a higher predicted relative abundance.



Spring Peeper
Pseudacris crucifer

Figure A1.10: Spring Peeper locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).

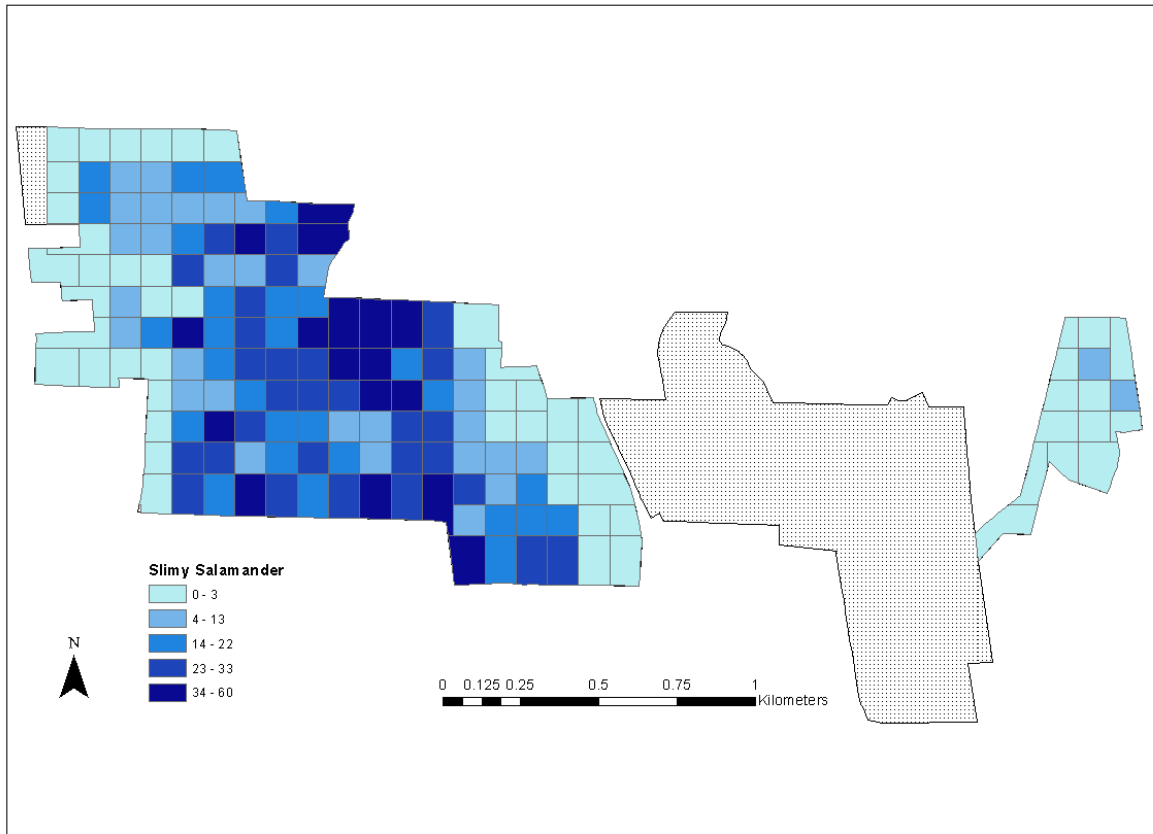
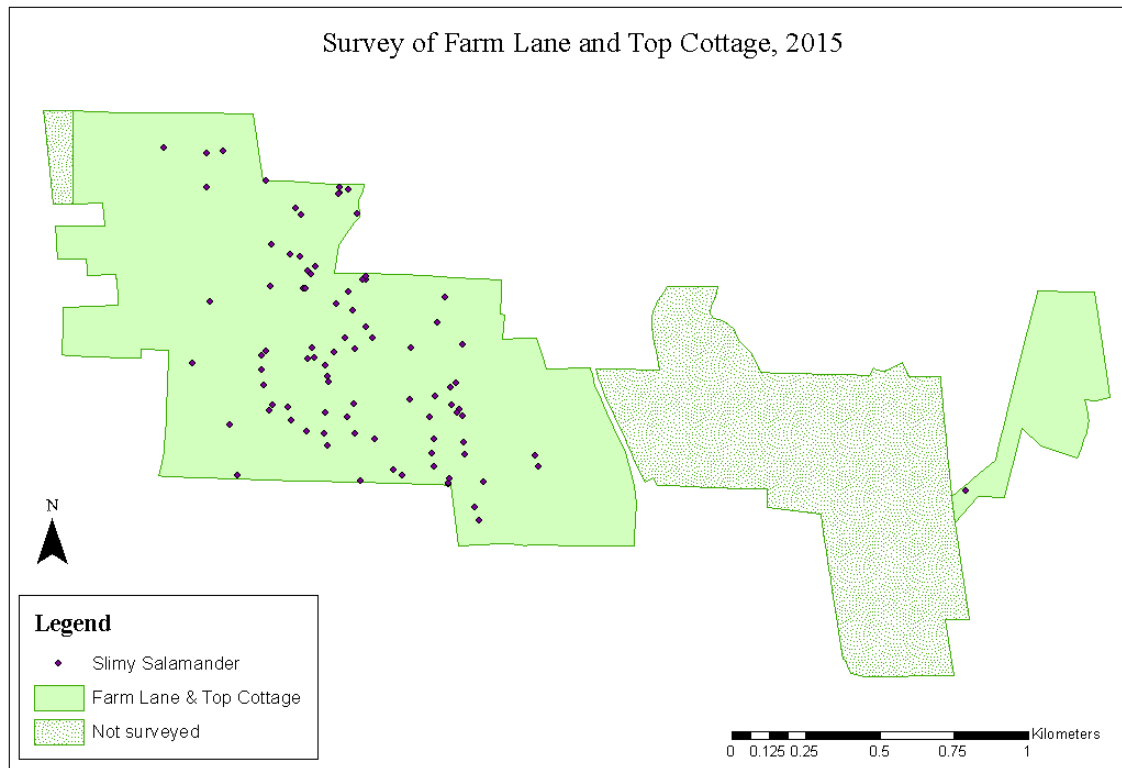


Figure A1.11: Predicted abundances at Farm Lane and Top Cottage at the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY) for Slimy Salamander. Darker blue indicates a higher predicted relative abundance.



Slimy Salamander
Plethodon glutinosus

Figure A1.12: Slimy Salamander locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).

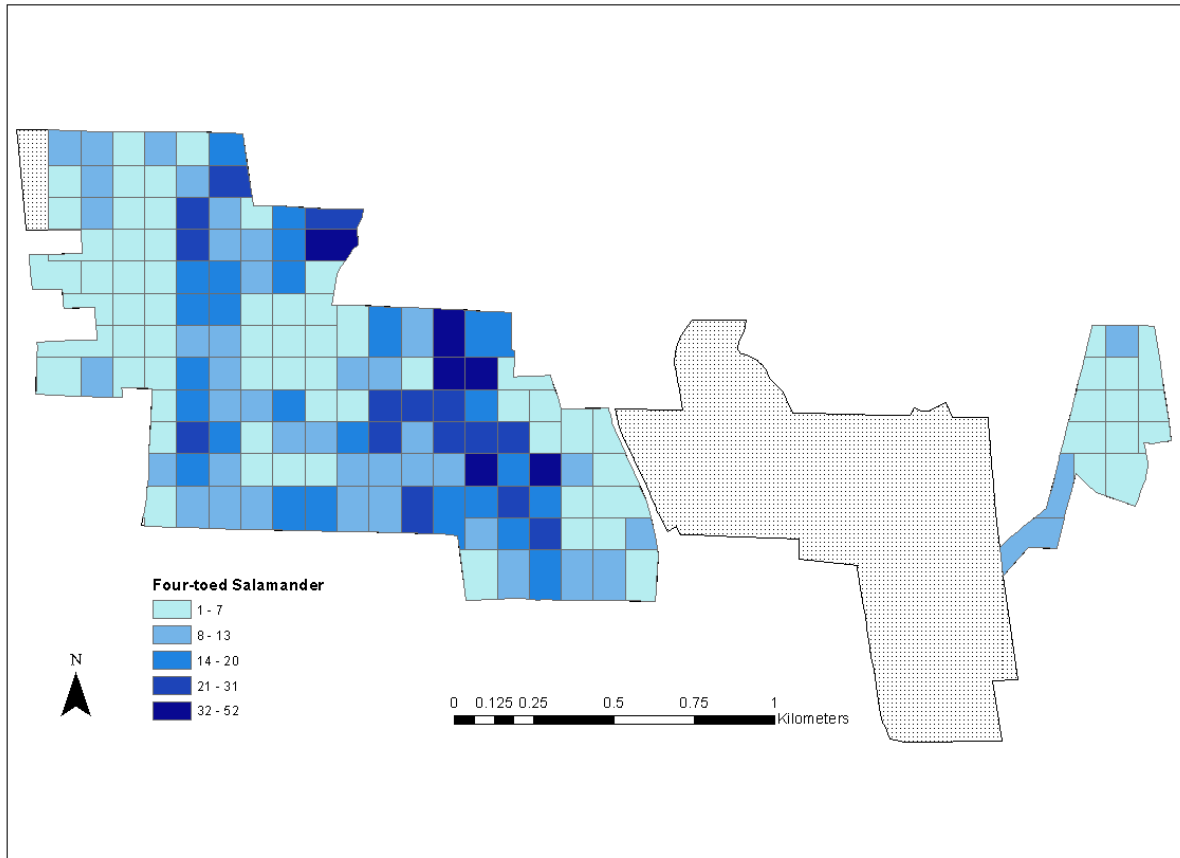
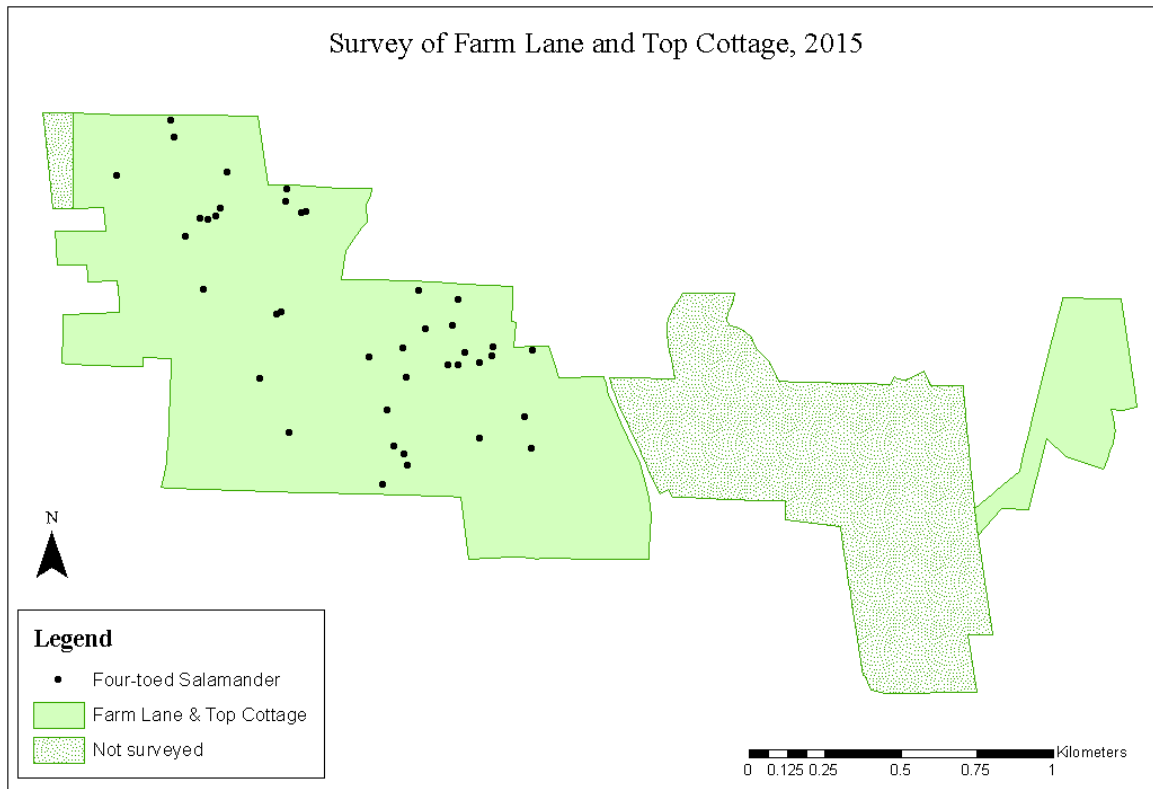


Figure A1.13: Predicted abundances at Farm Lane and Top Cottage at the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY) for Four-toed Salamander. Darker blue indicates a higher predicted relative abundance.



Four-toed Salamander
Hemidactylium scutatum

Figure A1.14: Four-toed Salamander locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).

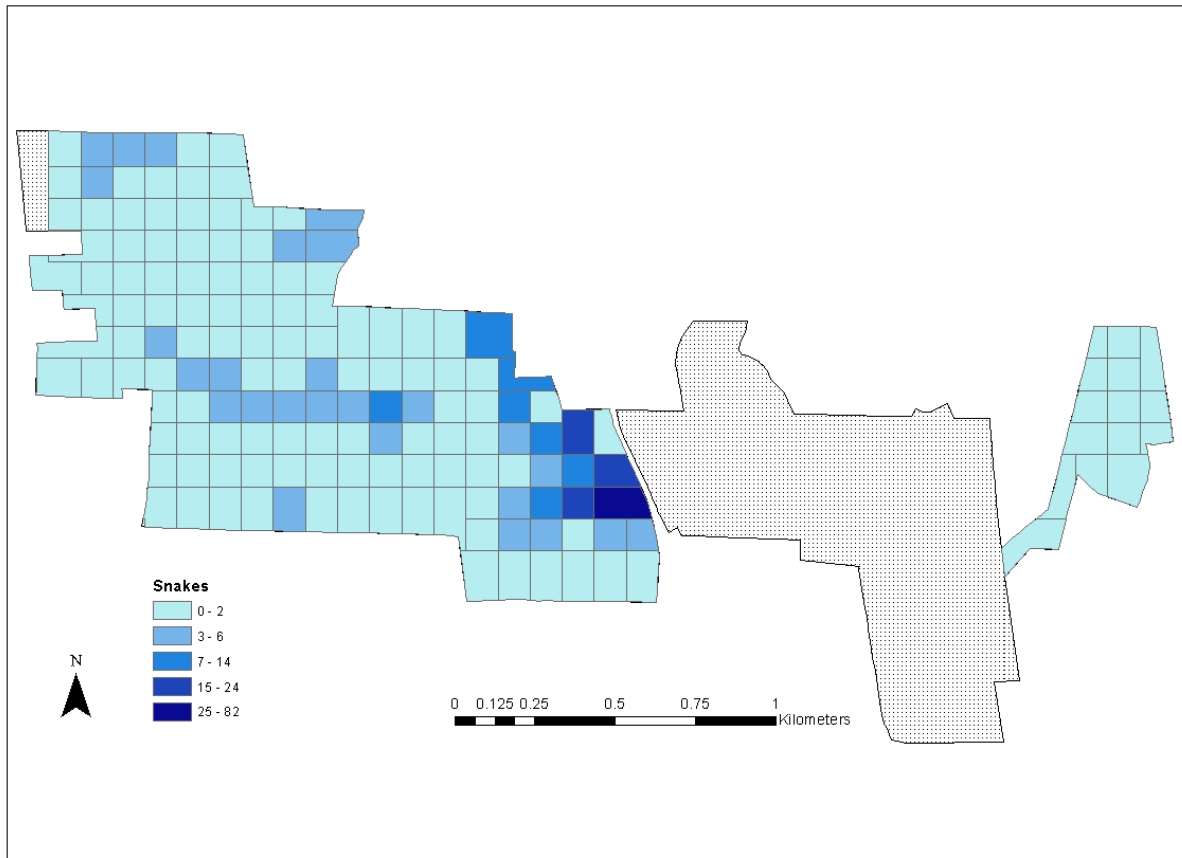


Figure A1.15: Predicted abundances at Farm Lane and Top Cottage at the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY) for all snake species. Darker blue indicates a higher predicted relative abundance.

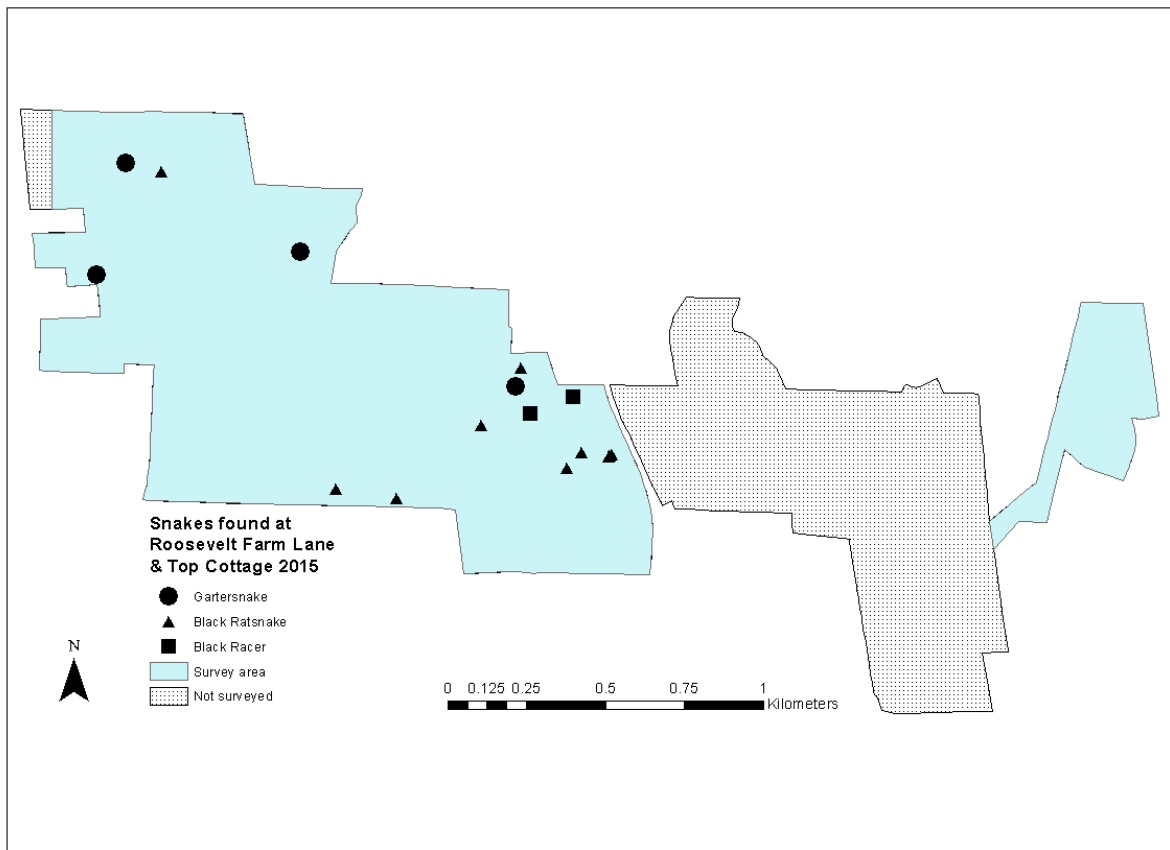


Figure A1.16: Snake locations of three species from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).

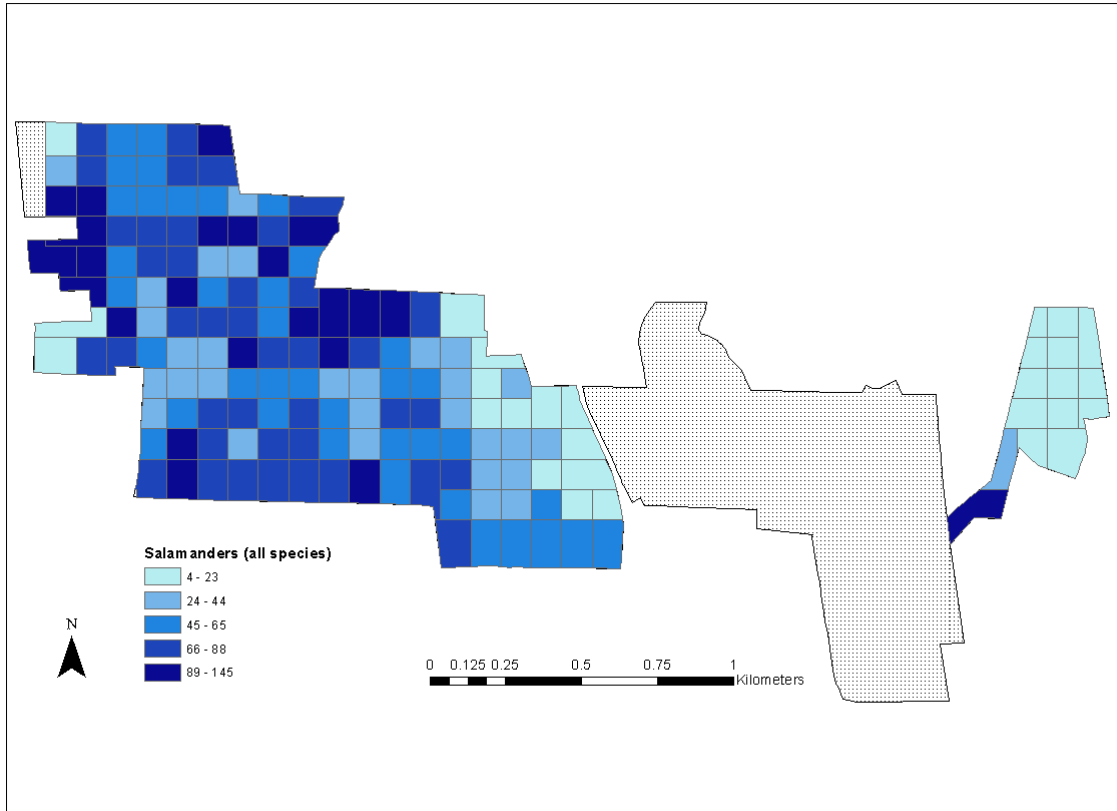
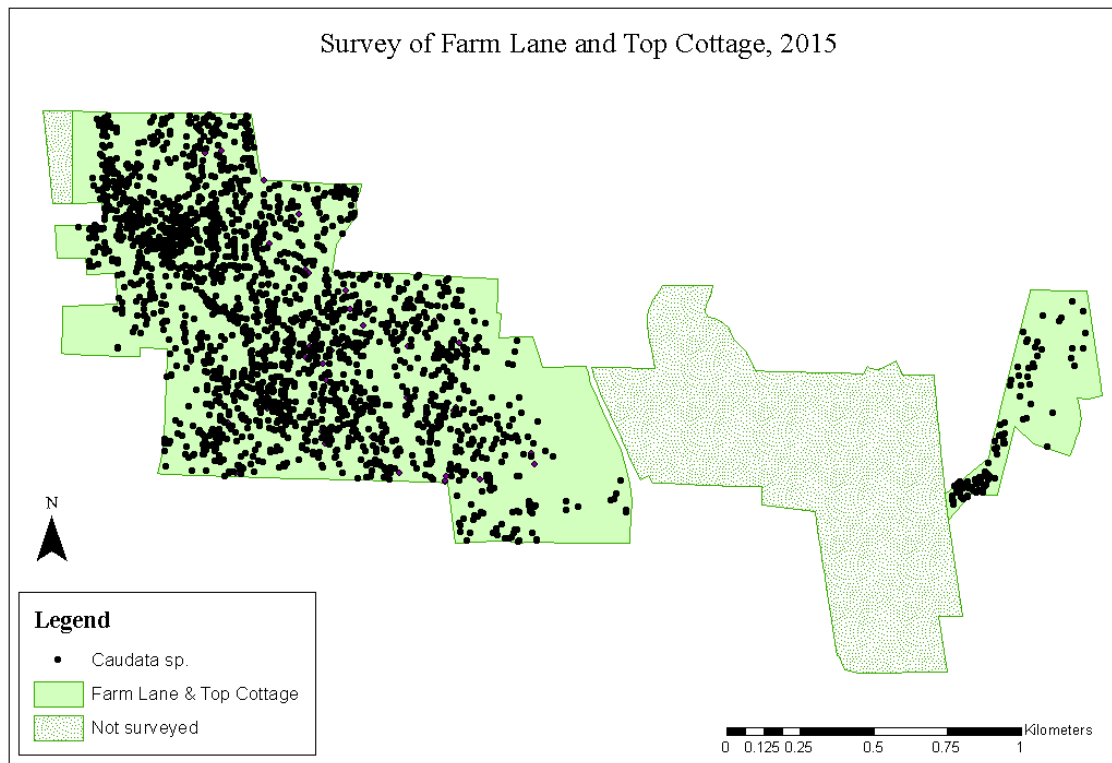


Figure A1.17: Predicted abundances at Farm Lane and Top Cottage at the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY) for all salamanders. Darker blue indicates a higher predicted relative abundance.



All salamanders

Ambystoma maculatum, *A. opacum*, *Eurycea bislineata*, *Hemidactylium scutatum*, *Notophthalmus viridescens*,
Plethodon cinereus, *P. glutinosus*

Figure A1.18: All salamander locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).

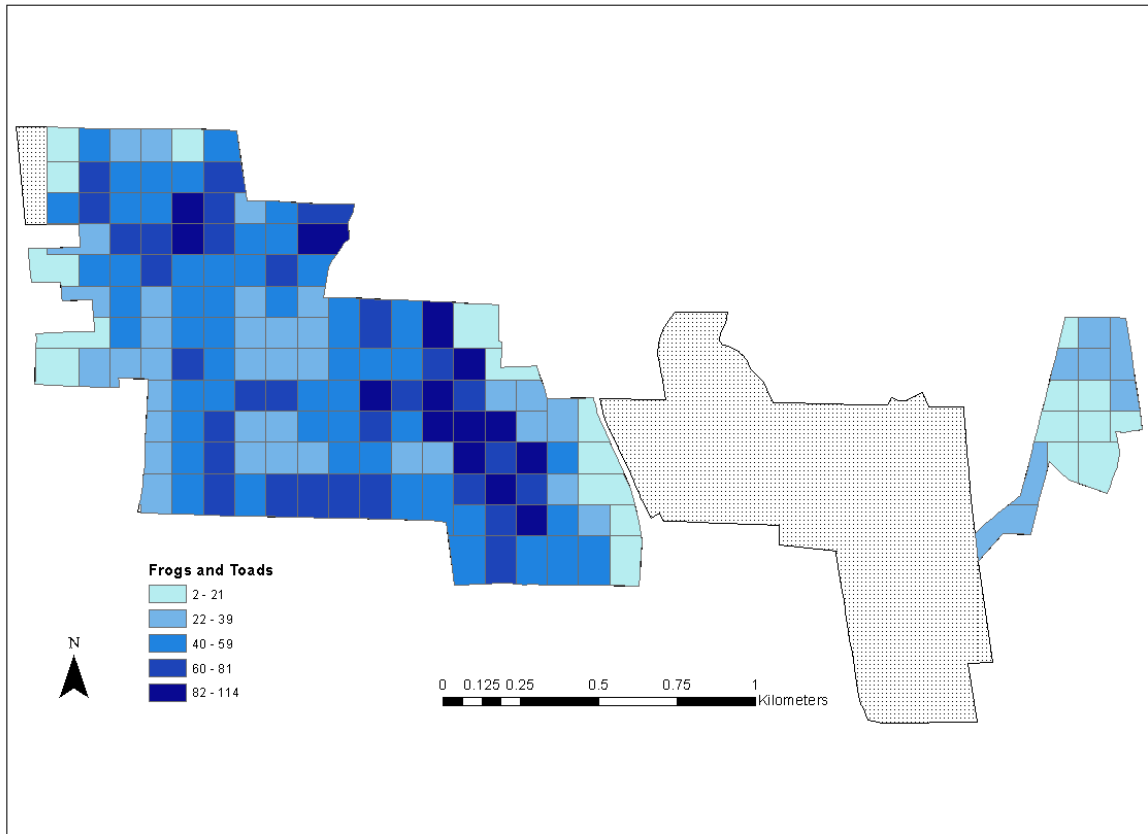
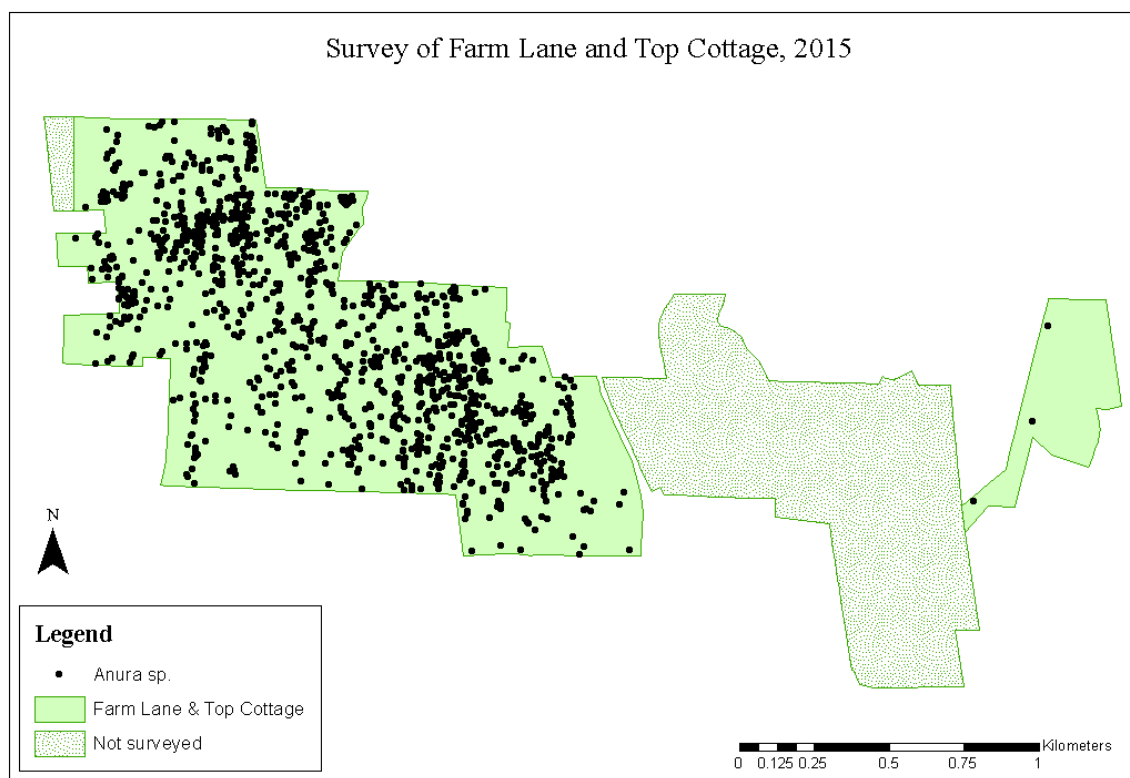


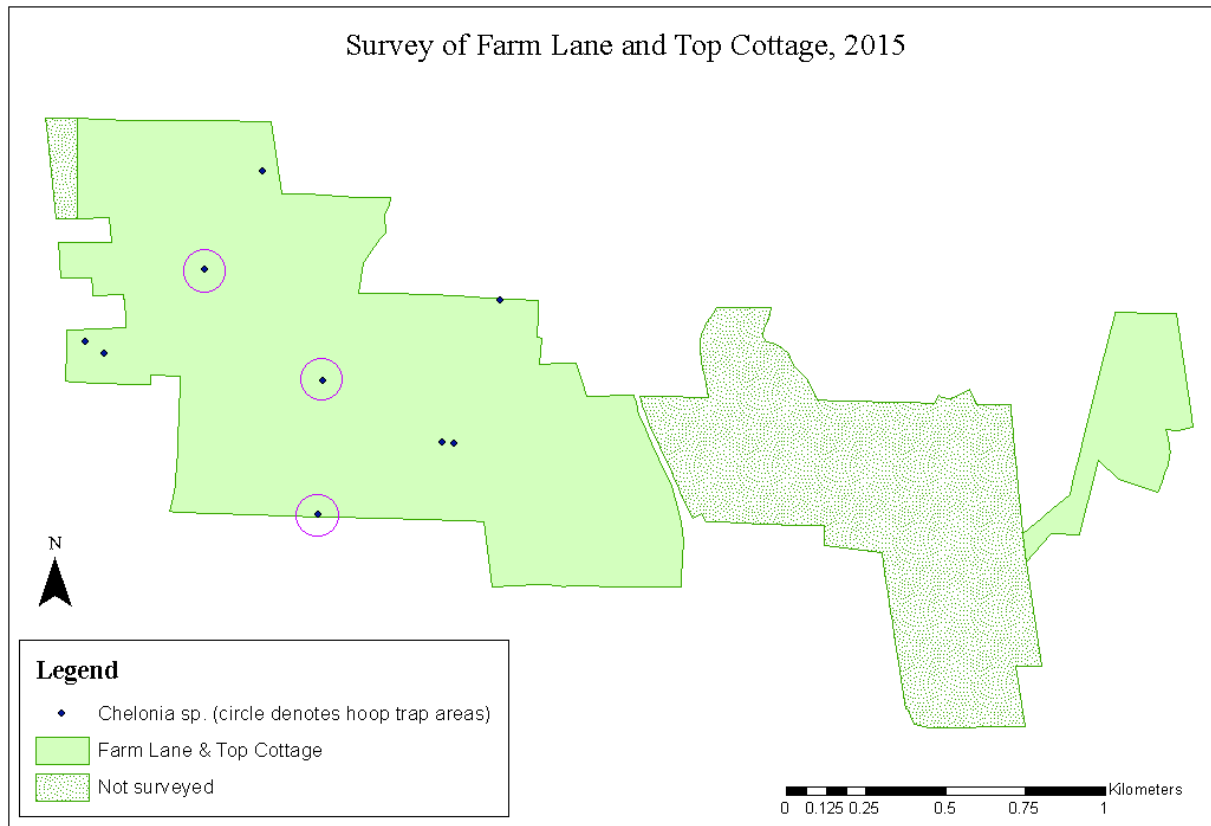
Figure A1.19: Predicted abundances at Farm Lane and Top Cottage at the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY) for all frogs and toads. Darker blue indicates a higher predicted relative abundance.



All frogs

Anaxyrus americanus, *Lithobates catesbeianus*, *L. clamitans*, *Hyla versicolor*, *L. sylvaticus*, *Pseudacris crucifer*

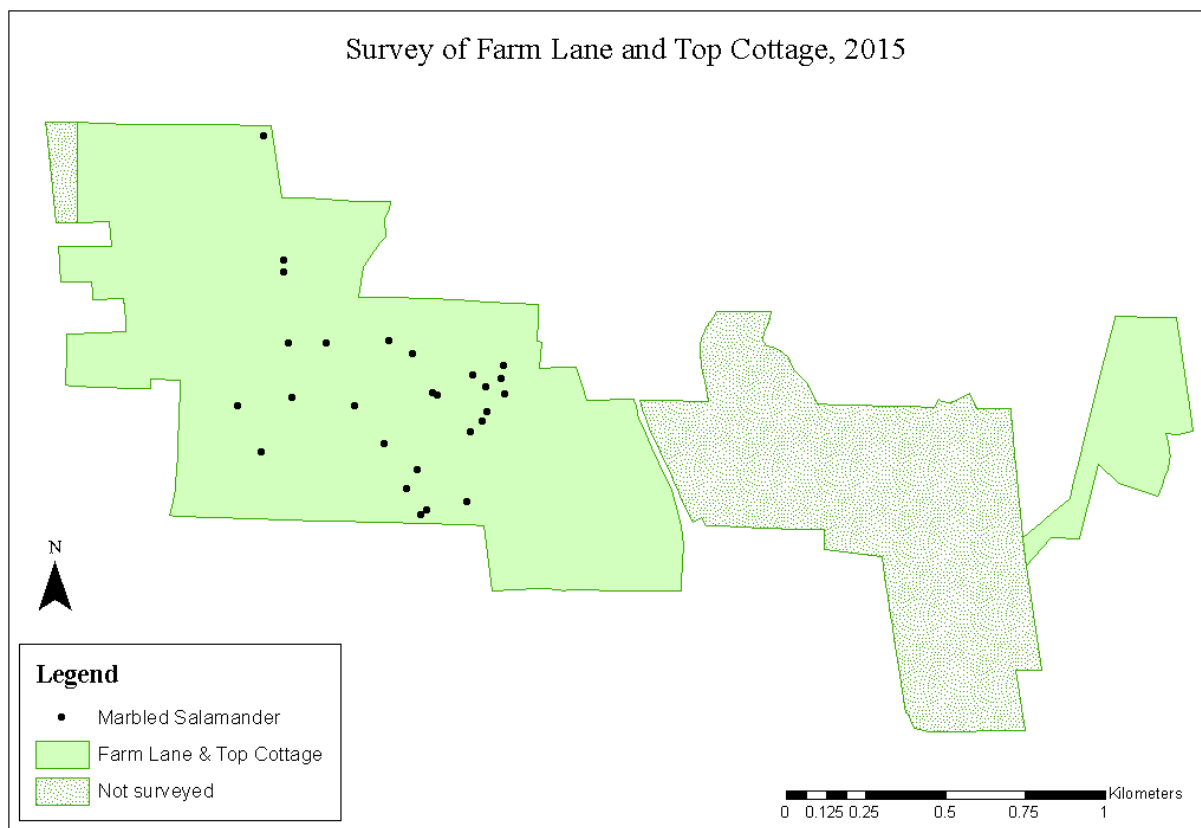
Figure A1.20: All frog locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).



All turtles

Terrapene carolina carolina, *Chrysemys picta*, & *Chelydra serpentina*

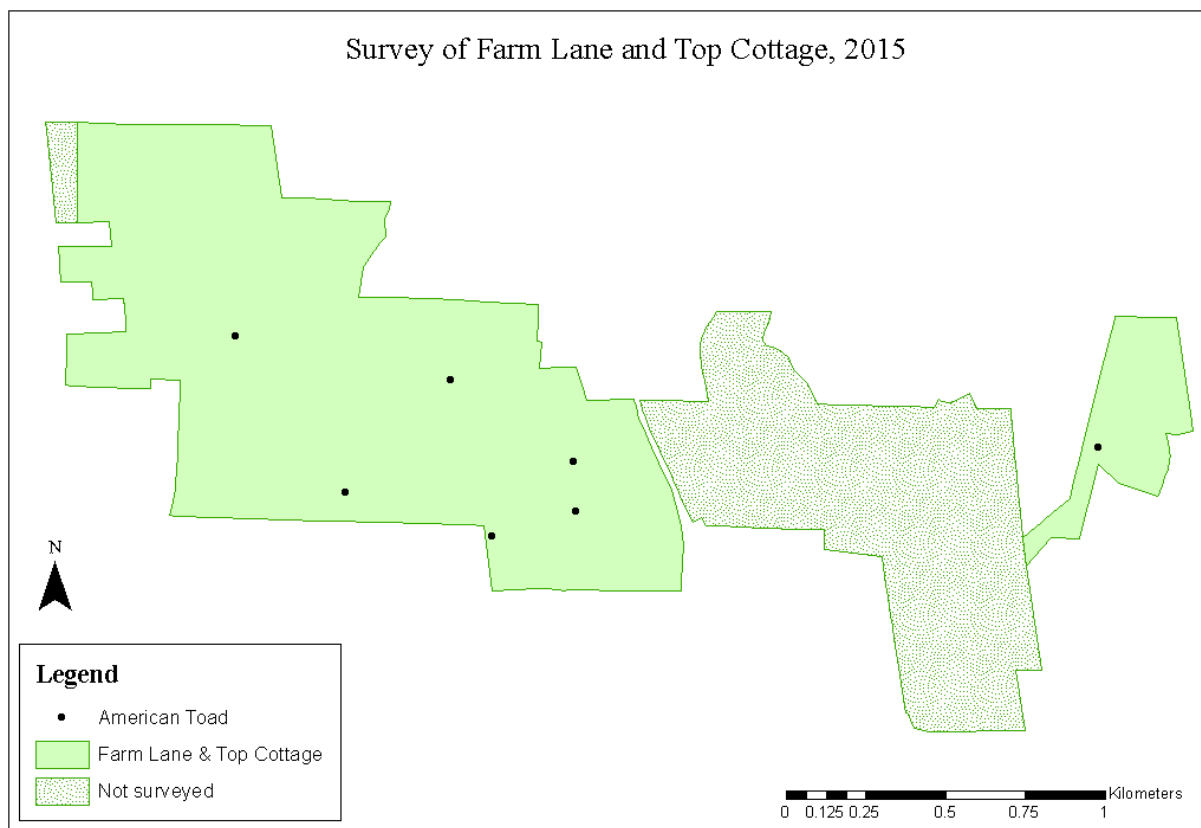
Figure A1.21: All turtle locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY). Circled areas denote hoop trap sites.



Marbled Salamander

Ambystoma opacum

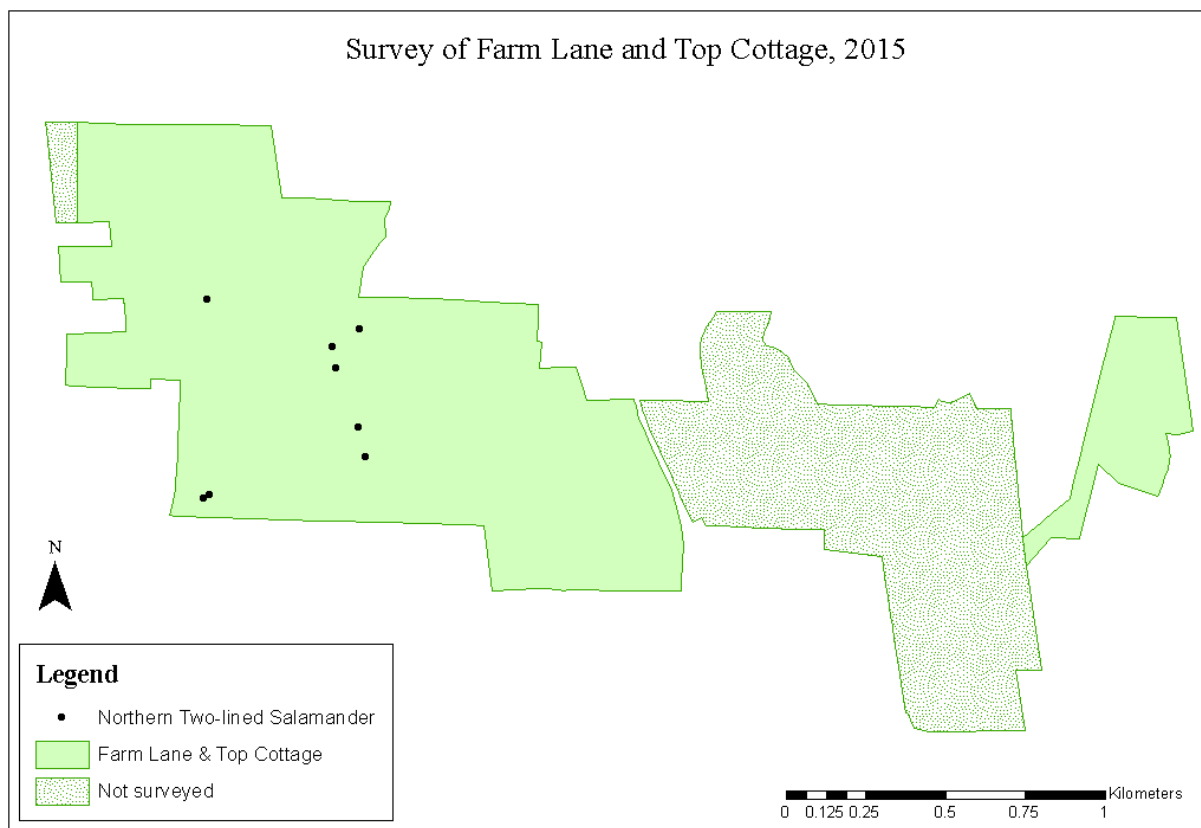
Figure A1.22: Marbled Salamander locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).



American Toad

Anaxyrus americanus

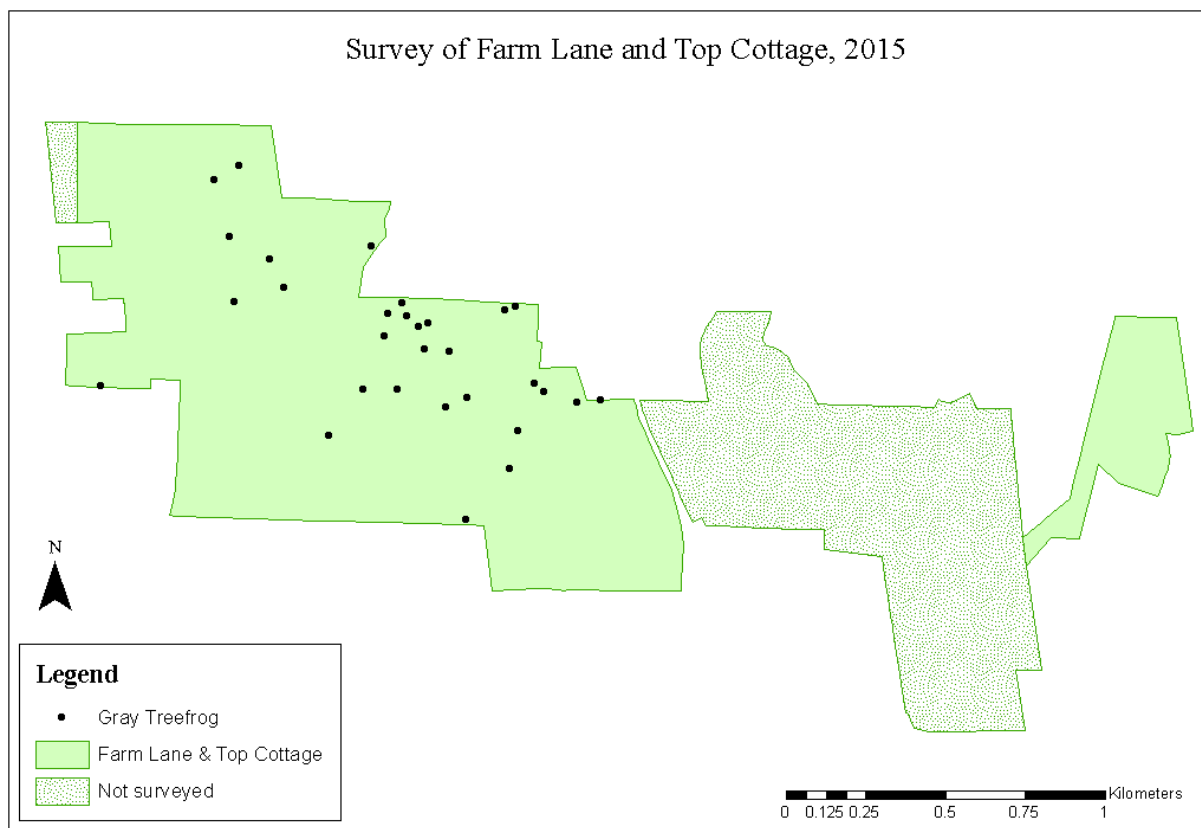
Figure A1.23: American Toad locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).



Northern Two-lined Salamander

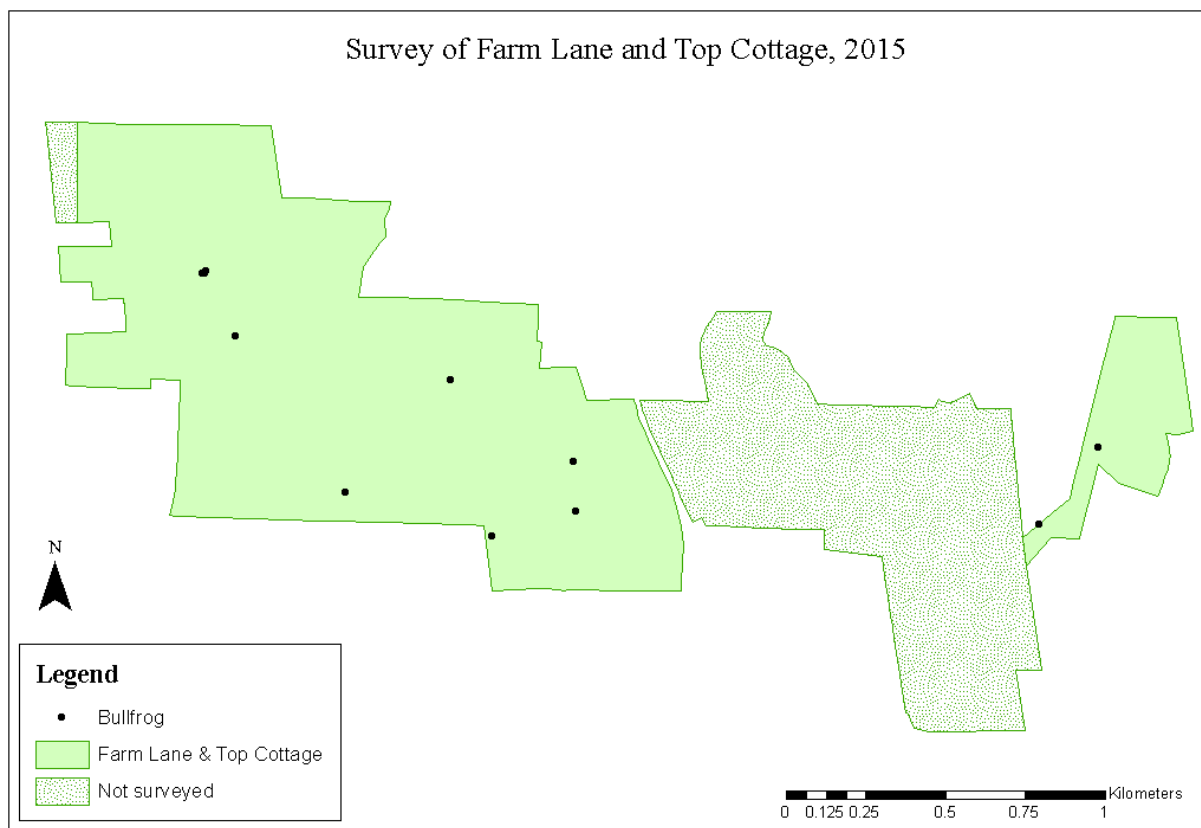
Eurycea bislineata

Figure A1.24: Northern Two-lined Salamander locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).



Gray Treefrog (calls only)
Hyla versicolor

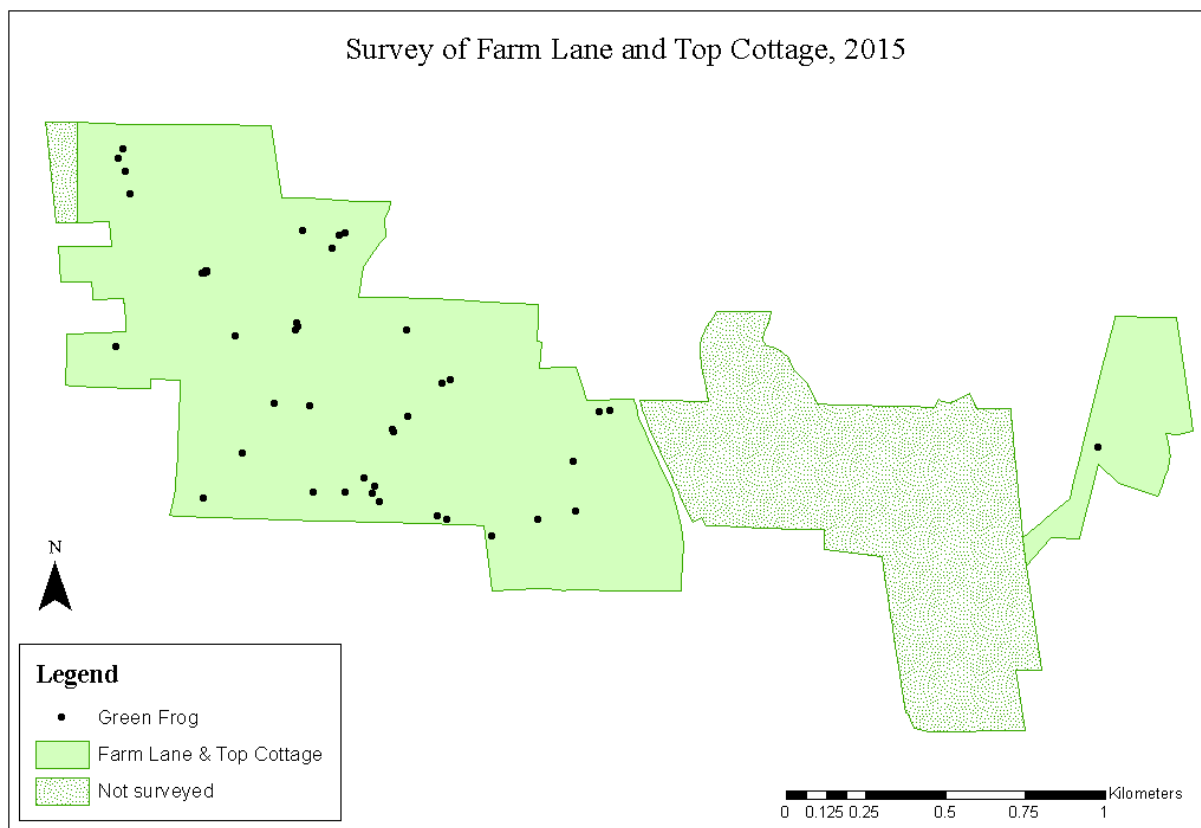
Figure A1.25: Gray Treefrog call locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).



Bullfrog

Lithobates catesbeianus

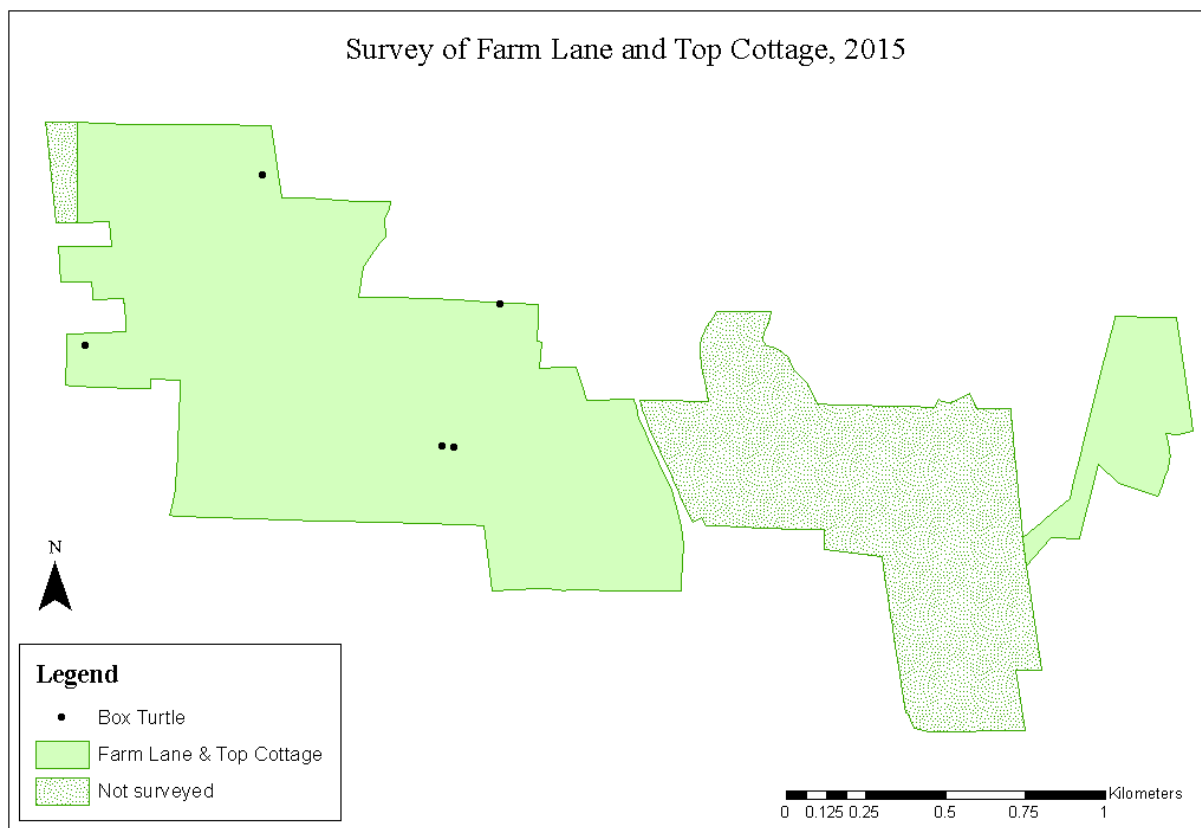
Figure A1.26: American Bullfrog locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).



Green Frog

Lithobates clamitans

Figure A1.27: Green Frog locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).



Eastern Box Turtle

Terrapene carolina carolina

Figure A1.28: Eastern Box Turtle locations from the 2015 herpetological survey at Farm Lane and Top Cottage of the Home of Franklin D. Roosevelt National Historic Site (Hyde Park, NY).

Table A1.2: Amphibian species and snakes, individuals found, and detection probabilities for the effort-only model from HOFR 2015.

Scientific name	Common name	Individuals	Detection [SE]
<i>Ambystoma maculatum</i>	Spotted Salamander	231	0.02 [0.04]
<i>Hemidactylium scutatum</i>	Four-toed Salamander	42	0.004 [0.002]
<i>Lithobates sylvaticus</i>	Wood Frog	1110	0.03 [0.004]
<i>Notophthalmus viridescens</i>	Eastern Newt	259	0.02 [0.004]
<i>Plethodon cinereus</i>	Redback Salamander	2228	0.15 [0.02]
<i>Plethodon glutinosus</i>	Slimy Salamander	101	0.02 [0.04]
<i>Pseudacris crucifer</i>	Spring Peeper	242	0.08 [0.04]
Serpentes spp.	Snakes	16	0.03 [0.06]

Table A1.3: Highest ranked abundance models for three amphibian groups at RFL and Top Cottage in 2015. Model statistics are reported for the top-ranked model and those within two AIC units of the top-ranked model. All models also included a coverage offset variable. P covariates are detection covariates, while λ covariates are site-specific covariates. K is the number of estimated parameters in the model, ΔAIC is the difference in AIC values compared to the top model (models within two AIC units are averaged), w is the Akaike model weight, and $\hat{C}$ is a measure of dispersion and model fit.

Species	P covariate	λ covariates ^a	K	ΔAIC	w	Cum. w	$\hat{C}$
Anura sp.	None	Slope/dec.for/dev.area/water%/edge/trail	11	0.00	0.68	0.68	4.27*
Caudata sp.	None	Elev.m/mx.for/old.fld	8	0.00	0.61	0.61	4.36*
	None	Elev.m/mx.for/old.fld/trail	9	1.69	0.25	0.86	4.31*
Amphibia sp.	None	Elev.m/old.fld/edge/trail	8	0.00	0.37	0.37	4.39*

^a Elev.m = mean elevation, Slope.m = mean slope, Mx.for = % of mixed forest, Dec.for = % of deciduous forest, Dev.area = % of developed area, Old.fld = % of old field, Water% = % of area composed of aquatic habitat, Edge = length of edge (m), Trail = length of trail (m).
 *Model is significantly overdispersed

Appendix II: Results of 2016 Disturbance Plot Comparisons at Roosevelt Farm Lane, Hyde Park,
New York.

Table A2.1: Results of generalized linear models for amphibian species with > 30 individuals
from 2016 disturbance plot comparisons

Species GLM results

Spotted Salamander (n = 36)				
Parameter	Estimate	S.E.	z	P(> z)
C (Intercept)	-1.23	0.25	-4.94	< 0.001
E	-2.85	1.03	-2.78	0.006
I	-17.07	1247.67	-0.01	0.98
T	-0.39	0.39	-1.00	0.32
T15	-1.18	0.51	-2.31	0.02
T50	-1.69	0.63	-2.69	0.007
Redback Salamander (n = 242)				
C (Intercept)	0.41	0.11	3.75	< 0.001
E	-1.3278	0.23	-5.73	< 0.001
I	-2.36	0.59	-4.01	< 0.001
T	-1.10	0.22	-5.06	< 0.001
T15	-0.31	0.17	-1.85	0.06
T50	-0.70	0.19	-3.69	< 0.001
Eastern Newt (n = 34)				
C (Intercept)	-1.61	0.30	-5.338	< 0.001
E	-2.48	1.04	-2.38	0.02
I	-16.69	1247.67	-0.01	0.99
T	-0.11	0.43	-0.26	0.80
T15	-0.47	0.48	-0.97	0.33
T50	-0.81	0.54	-1.50	0.13
Slimy Salamander (n = 16)				
C (Intercept)	-2.4	0.45	-5.36	< 0.001
E	-1.00	0.84	-1.20	0.23
I	-15.90	1247.67	-0.01	0.99
T	-0.93	0.84	-1.12	0.26
T15	0.16	0.61	0.27	0.79
T50	-1.63	1.10	-1.49	0.14
Four-toed Salamander (n = 11)				
C (Intercept)	-3.31	0.71	-4.69	< 0.001
E	1.17	0.80	1.45	0.15
I	0.27	1.22	0.22	0.83
T	-16.99	2076.87	-0.008	0.99
T15	-0.71	1.22	-0.581	0.56
T50	-16.99	2076.87	-0.008	0.99
Marbled Salamander (n = 5)				
C (Intercept)	-22.3	5697	-0.004	1.0
E	18.21	5697	0.003	1.0
I	19.95	5697	0.004	1.0
T	18.97	5697	0.003	1.0

T15	- < 0.001	8020	0.0	1.0
T50	- < 0.001	8020	0.0	1.0

American Toad (n = 4)

C (Intercept)	-4.0	1.0	-4.01	< 0.001
E	-17.30	3308.07	-0.005	1.0
I	0.96	1.41	0.68	0.50
T	-0.02	1.41	-0.01	0.99
T15	-0.02	1.41	-0.01	0.99
T50	-17.30	3424.18	-0.005	1.0

Spring Peeper (n = 85)

C (Intercept)	-0.87	0.21	-4.18	< 0.001
E	-1.61	0.49	-3.27	0.001
I	-2.17	1.02	-2.13	0.03
T	-0.66	0.36	-1.88	0.06
T15	-0.21	0.31	-0.67	0.50
T50	-0.07	0.29	0.23	0.82

Wood Frog (n = 331)

C (Intercept)	0.35	0.11	3.09	0.002
E	-1.35	0.24	-5.60	< 0.001
I	-2.29	0.59	-3.90	< 0.001
T	0.04	0.16	0.28	0.78
T15	-0.08	0.16	-0.52	0.60
T50	-0.10	0.16	-0.60	0.55

Gray Treefrog (n = 22)

C (Intercept)	-2.06	0.38	-5.45	< 0.001
E	-0.42	0.59	-0.72	0.47
I	0.12	0.69	0.17	0.87
T	-0.87	0.69	-1.25	0.21
T15	-1.96	1.07	-1.84	0.07
T50	-0.87	0.69	-1.25	0.21

Species Absent (no species present) (n = 89)

C (Intercept)	-2.22	0.41	-5.43	< 0.001
E	1.62	0.44	3.65	< 0.001
I	1.74	0.49	3.52	< 0.001
T	0.90	0.48	1.86	0.06
T15	0.76	0.49	1.53	0.13
T50	0.39	0.53	0.74	0.46

Table A2.2: Mean ($\pm$ SD) counts of common species (those with ≥ 34 occurrences) at HOFR (Dutchess County, NY) in 2016 among six plot types: Control (n = 55), Edge (n = 60), Interaction (n = 21), Trail (n = 56), T15 (n = 56), and T50 (n = 56). Control plots were interior forest > 50 m from trails and > 20 m from habitat edges. Edges were within 20 m of habitat edges. Interaction plots were located on trails within 20 m of habitat edges. Trail plots were centered on trails, and T15 and T50 were 15 and 50 m from trails.

Species	n	Mean $\pm$ SD					
		Control	Interaction	Edge	Trail	Trail 15 m	Trail 50 m
Wood Frog	331	1.42 $\pm$ 2.29	0.14 $\pm$ 0.36	0.37 $\pm$ 0.78	1.48 $\pm$ 2.66	1.30 $\pm$ 2.30	1.29 $\pm$ 2.25
Spring Peeper	85	0.42 $\pm$ 0.94	0.05 $\pm$ 0.22	0.08 $\pm$ 0.42	0.21 $\pm$ 0.62	0.34 $\pm$ 0.88	0.45 $\pm$ 0.95
All frogs and toads	442	4.98 $\pm$ 2.82	3.38 $\pm$ 0.59	3.53 $\pm$ 1.10	4.77 $\pm$ 2.88	4.68 $\pm$ 2.56	4.79 $\pm$ 2.81
Spotted Salamander	36	0.29 $\pm$ 0.98	0 $\pm$ 0	0.02 $\pm$ 0.13	0.20 $\pm$ 0.64	0.09 $\pm$ 0.35	0.05 $\pm$ 0.30
All Ambystoma	41	1.29 $\pm$ 0.98	1.10 $\pm$ 0.30	1.03 $\pm$ 0.50	1.23 $\pm$ 0.74	1.09 $\pm$ 0.35	1.05 $\pm$ 0.30
Eastern Newt	34	0.20 $\pm$ 0.85	0 $\pm$ 0	0.02 $\pm$ 0.13	0.18 $\pm$ 0.43	0.13 $\pm$ 0.38	0.09 $\pm$ 0.29
Redback Salamander	242	1.51 $\pm$ 1.71	0.14 $\pm$ 0.36	0.4 $\pm$ 1.30	0.50 $\pm$ 0.83	1.11 $\pm$ 1.50	0.75 $\pm$ 1.44
All salamanders	344	7.13 $\pm$ 2.37	5.29 $\pm$ 0.56	5.60 $\pm$ 1.58	5.95 $\pm$ 1.29	6.45 $\pm$ 1.64	5.91 $\pm$ 1.43
Absent (fraction of plots with no species detections) w/95% CI	89	0.11 $\pm$ 0.08	0.62 $\pm$ 0.21	0.55 $\pm$ 0.12	0.27 $\pm$ 0.11	0.34 $\pm$ 0.11	0.16 $\pm$ 0.10
Total individuals	791	4.20 $\pm$ 3.35	0.67 $\pm$ 1.02	1.13 $\pm$ 1.86	2.71 $\pm$ 3.73	3.14 $\pm$ 3.05	2.68 $\pm$ 3.07
Species richness	791	1.93 $\pm$ 0.18	0.67 $\pm$ 1.02	0.62 $\pm$ 0.85	1.36 $\pm$ 1.18	1.48 $\pm$ 1.25	1.32 $\pm$ 0.86

Table A2.3: Results of Tukey's HSD pairwise comparisons for 10 species of amphibians in six plot types at the Home of Franklin D. Roosevelt in 2016. Control (n = 55), Edge (n = 60), Interaction (n = 21), Trail (n = 56), T15 (n = 56), and T50 (n = 56). Control plots were interior forest > 50 m from trails and > 20 m from habitat edges. Edges were within 20 m of habitat edges. Interaction plots were located on trails within 20 m of habitat edges. Trail plots were centered on trails, and T15 and T50 were 15 and 50 m from trails. Significant p values are in bold.

	Spotted Sal.	Marbled Sal.	Amer Toad	Four- toed Sal.	Gray Treefrog	Wood Frog	East. Newt	Redback Sal.	Slimy Sal.	Spring Peeper
n	36	5	4	11	22	331	34	242	16	85
Rel.	Tukey's HSD P(> z)									
E - C	0.04	1.0	1.0	0.61	0.97	<0.001	0.12	<0.001	0.8	0.01
I - C	1.0	1.0	0.98	1.0	1.0	0.001	1.0	<0.001	1.0	0.24
T - C	0.89	1.0	1.0	1.0	0.8	0.99	1.0	<0.001	0.84	0.38
T15 - C	0.14	1.0	1.0	0.98	0.42	0.99	0.9	0.4	1.0	0.98
T50 - C	0.06	1.0	1.0	1.0	0.8	0.98	0.6	0.002	0.61	0.99
I - E	1.0	0.63	1.0	0.94	0.97	0.61	1.0	0.51	1.0	0.99
T - E	0.12	0.98	1.0	1.0	0.99	<0.001	0.16	0.96	1.0	0.44
T15 - E	0.57	1.0	1.0	0.4	0.71	<0.001	0.34	<0.001	0.65	0.04
T50 - E	0.88	1.0	1.0	1.0	0.99	<0.001	0.58	0.12	0.99	0.006
T - I	1.0	0.89	0.97	1.0	0.82	<0.001	1.0	0.28	1.0	0.67
T15 - I	1.0	1.0	0.97	0.97	0.45	0.001	1.0	0.005	1.0	0.36
T50 - I	1.0	1.0	1.0	1.0	0.82	0.002	1.0	0.05	1.0	0.21
T15 - T	0.62	1.0	1.0	1.0	0.92	0.96	0.97	0.005	0.71	0.79
T50 - T	0.28	1.0	1.0	1.0	1.0	0.94	0.75	0.52	0.99	0.26
T50-T15	-0.51	0.73	-0.69	1.0	0.92	1.0	0.99	0.34	0.49	0.93

Table A2.4: Results of Tukey's HSD pairwise comparisons for five groups of amphibians in six plot types at the Home of Franklin D. Roosevelt in 2016. Control (n = 55), Edge (n = 60), Interaction (n = 21), Trail (n = 56), T15 (n = 56), and T50 (n = 56). Control plots were interior forest > 50 m from trails and > 20 m from habitat edges. Edges were within 20 m of habitat edges. Interaction plots were located on trails within 20 m of habitat edges. Trail plots were centered on trails, and T15 and T50 were 15 and 50 m from trails. Significant p values are in bold.

	Salamanders	Frogs and Toads	Ambystoma sp.	Plethodon sp.	Uncommon salamanders
n	344	442	41	258	32
Relationship	Tukey's HSD P(> z)				
E - C	< 0.001	< 0.001	0.04	< 0.001	0.99
I - C	< 0.001	< 0.001	0.64	< 0.001	1.0
T - C	< 0.001	0.95	0.98	< 0.001	0.93
T15 - C	0.07	0.83	0.17	0.49	1.0
T50 - C	< 0.001	0.97	0.06	< 0.001	0.42
I - E	0.51	0.95	0.89	0.42	1.0
T - E	0.25	< 0.001	0.09	0.96	0.69
T15 - E	< 0.001	< 0.001	0.83	< 0.001	0.99
T50 - E	0.36	< 0.001	0.99	0.17	0.25
T - I	0.05	< 0.001	0.83	0.22	0.94
T15 - I	0.001	< 0.001	1.0	0.003	1.0
T50 - I	0.06	< 0.001	0.98	0.04	0.44
T15 - T	0.14	0.99	0.43	0.002	0.94
T50 - T	0.99	1.0	0.18	0.62	0.8
T50-T15	0.08	0.99	0.97	0.15	0.43
Uncommon Salamanders = Four-toed Salamander, Slimy Salamander, and Marbled Salamander.					

Table A2.5: Results of Tukey's HSD pairwise comparisons for eight habitat variables in six plot types at the Home of Franklin D. Roosevelt in 2016. Control (n = 55), Edge (n = 60), Interaction (n = 21), Trail (n = 56), T15 (n = 56), and T50 (n = 56). Control plots were interior forest > 50 m from trails and > 20 m from habitat edges. Edges were within 20 m of habitat edges. Interaction plots were located on trails within 20 m of habitat edges. Trail plots were centered on trails, and T15 and T50 were 15 and 50 m from trails. Significant p values are in bold.

	Basal Area	Tree total	Small CWD	Large CWD	Total CWD	LL depth	LL % cover	Canopy cover %
Relationship	Tukey's HSD P(> z)							
E - C	0.69	0.99	0.31	0.32	0.08	0.99	0.21	0.96
I - C	0.15	0.92	<0.001	0.01	<0.001	<0.001	<0.001	0.58
T - C	0.78	0.61	0.09	0.19	0.03	<0.001	<0.001	1.0
T15 - C	0.98	0.78	0.98	0.08	0.26	0.43	1.0	0.99
T50 - C	0.97	0.79	0.38	0.26	0.10	0.99	0.99	0.002
I - E	0.74	0.81	0.08	0.41	0.07	<0.001	<0.001	0.22
T - E	1.0	0.81	0.99	0.99	0.99	<0.001	0.03	0.95
T15 - E	0.97	0.53	0.71	0.98	0.99	0.74	0.16	0.81
T50 - E	0.23	0.54	1.0	1.0	1.0	1.0	0.34	0.02
T - I	0.69	0.26	0.23	0.57	0.15	0.97	<0.001	0.6
T15 - I	0.41	1.0	0.003	0.75	0.02	<0.001	<0.001	0.77
T50 - I	0.03	1.0	0.07	0.49	0.06	<0.001	<0.001	<0.001
T15 - T	0.99	0.04	0.34	0.99	0.95	<0.001	<0.001	0.99
T50 - T	0.32	0.04	0.98	1.0	0.99	<0.001	<0.001	0.001
T50-T15	0.69	1.0	0.78	0.99	0.99	0.79	0.99	<0.001

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PUBLICATIONS

- Yuan, M. L., Dean, S. H., Longo, A. V., Rothermel, B. B., Tuberville, T. D., & Zamudio, K. R. (2015). Kinship, inbreeding and fine-scale spatial structure influence gut microbiota in a hindgut-fermenting tortoise. *Molecular Ecology*, 24(10), 2521-2536.

CONFERENCE PRESENTATIONS *Indicates presenter

- Dean, S. H*, Gibbs, J. P., and H. B. Underwood. Influence of habitat edges and recreational trails on the distribution and abundance of amphibians in an urban protected area. Joint Meeting of Ichthyologists and Herpetologists, Austin, TX. 2017 [talk]
Dean, S. H*, Rothermel, B. B., & Yuan, M. L. Effects of age and size on Gopher Tortoise (*Gopherus polyphemus*) courtship and mating behaviors. Joint Meeting of Ichthyologists and Herpetologists, Chattanooga, TN. 2014. [talk]
Jones*, L., Dean, S. H*, & Rothermel, B. B. Digging Deeper and Longer: Archbold Biological Station's 46-year History of Gopher Tortoise Research. Turtle Survival Alliance, St. Louis, MO. 2013. [poster]

OUTREACH

- Dean, S. H. & Kamener, G. 2013. Getting to Know the Gopher Tortoise: Burrow Explorers. <http://www.youtube.com/watch?v=6bvB9owIJ9M> [educational YouTube video for children]