TOWSON UNIVERSITY

COLLEGE OF GRADUATE STUDIES AND RESEARCH

SURROGATE HABITATS FOR URBAN MAMMALS: QUANTIFYING SURVIVAL AND OCCUPANCY IN CONSTRUCTED WETLANDS

by

Caitlin M. Graff

A thesis

Presented to the faculty of

Towson University

In partial fulfillment

Of the requirements for the degree

Master of Science

Department of Biological Sciences

Towson University

December 2013

TOWSON UNIVERSITY
OFFICE OF GRADUATE STUDIES

THESIS COMMITTEE APPROVAL FORM

Student's Name

Caitlin Graff

Chair, Thesis Committee

Harald Beck

Signature Typed name

Member Alice O'Connell

Signature Typed name

Member Joel W. Snodgrass

Signature Typed name

Member _____

Signature Typed name

Note: Please attach a description of the affiliation and credentials of any non-Towson University members of the Committee, and the members' *curriculum vitae.*

Approved by

Graduate Program Director 12/11/13

Signature Date

Department Chairperson

12/11/13

Signature Date

Dean of Graduate Studies

12-15-13

Signature Date

Instructions: Type in the appropriate information in the fields above or just put in a space if you only have three committee members; print the form and obtain signatures from your committee chairperson, the Graduate Director(s) and the Chairperson of the Department of Biological Sciences; and submitted this form to the Biology Graduate Program Director(s). They will submit to the School of Graduate Studies.

ACKNOWLEDGMENTS

I thank Harald Beck Ph.D., my major advisor, for guiding me through the research process with enthusiasm and encouragement. I am grateful to the members of my committee, Allan O'Connell Ph.D. and Joel Snodgrass Ph.D., for their intellectual contributions and support. I thank Miguel Fernandes Ph.D. for his valuable insights to Program MARK.

The Towson University Department of Biological Sciences, Fisher College of Science and Mathematics, and Graduate Student Association are recognized for financially supporting my field research and analysis. The entertaining conversation and hard work of Olusumbo Alade, Holly Badin, Jessica Kalisch, Dovid Kozlovsky, Megan Niehas, and Cody Werner were greatly appreciated. Special thanks go to Beth Fitzpatrick, April Marcangeli, Nick Tingley, and John Zaharick.

ABSTRACT

Surrogate Habitats for Urban Mammals: Quantifying Survival and Occupancy in Constructed Wetlands

Caitlin M. Graff

Land management plans require the construction of stormwater ponds to mitigate the effects of urbanization on streams. Stormwater ponds temporarily retain polluted runoff from impervious surfaces, allowing contaminants to settle before the water recharges ground water. With intentions to maintain natural hydrology, temperature, and nutrient inputs, stormwater ponds also attract wildlife. Some stormwater ponds might better support or attract urban wildlife when their hydrology, size, and vegetation are considered. I surveyed the small, meso-, and large mammal community in Baltimore County, Maryland, USA to determine the use of stormwater ponds by mammals. My results suggest that stormwater ponds are being utilized extensively by mammals in urban landscapes and that small changes in management could improve the quality of these surrogate habitats.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

INTRODUCTION

Over 80% of the 313 million US citizens live in an urban setting (Central Intelligence Agency 2010). The concentration of human inhabitants requires large-scale landscape alterations as sophisticated transportation systems and permanent structures are built to meet a diversity of needs (Fahrig 2003, McDonnell and Pickett 1990, McKinney 2002, Pickett and Cadenasso 2008). Urbanization formally describes the conversion and development of the landscape. As a consequence of this continuous process that disrupts succession, an overwhelming amount of terrestrial habitat is directly destroyed, landscapes are heavily fragmented, and non-native invasive species that frequently outcompete natives are introduced (McDonnell and Pickett 1990, McKinney 2002, Vitousek et al. 1997). Fragmentation is detrimental to regional wildlife because it isolates existing habitat, restrains movement and dispersal, reduces available resources, and limits mate access (Fischer and Lindenmayer 2007, Ryall and Fahrig 2006, Saunders et al. 1991). Accordingly, increasing development and expansion of the urban landscape has become a driving force for localized extinction of species (Dickman 1987, McKinney 2002, Olden et al. 2004).

A characteristic of an urbanized landscape is the high proportion of impervious surfaces that replace crucial ecosystems. For instance, wetlands are often dredged, drained, and filled in areas of development, causing a 53% decrease in historical US wetlands (Johnson 1994). Minimal losses of wetland habitat greatly reduce the success of wetland species (Croonquist and Brooks 1991, Gibbs 1993, Gibbs 2000). In addition to

wetlands providing critical habitat, Costanza et al. (1997) showed that they play a role in the rate of erosion, habitat disturbance, water quality, gas regulation, and waste regulation. These valuable ecosystems services stop with the disappearance of wetlands during development. Faulkner (2004) states that impervious surfaces prevent groundwater recharge and cause a flashier stream hydrology, putting stress on the watershed and biota. He in turn states the need for promoting forested wetlands in the urban landscape.

Stormwater ponds (hereafter SWPs) are a type of "Best Management Practice" constructed to restore wetland ecosystem services and preserve the health of the urbanized watershed (Colosimo and Wilcock 2007, Tsihrintzis and Hamid 1997). Most SWPs are built into preexisting depressions and are defined by an inflow culvert, emergency spillway, water basin, containment wall, and terrestrial berm that aid in temporarily retaining runoff from impervious surfaces (Galli 1990, Yeh and Labadie 1997). Designed to settle out suspended solids and contaminants, SWPs regulate the quality and quantity of the runoff entering stream systems (Behera et al. 1999, Galli 1990). Baltimore County, Maryland, USA mitigates watershed stress caused by increased impervious surfaces, stormwater runoff, and erosion by implementing multiple "Best Management Practices", including hundreds of SWPs (MDP 2009).

SWPs can have a secondary function of providing a constructed habitat in developed landscapes, a function which is frequently overlooked by municipal authorities during phases of planning and construction (Schueler 1992). Indeed, SWPs are now

receiving increased attention in urban ecology studies because of their influence on contaminants, habitat complexity, and the extensive communities of amphibians, reptiles, and birds that have been observed (Bishop et al. 2000, Casey et al. 2006, Snodgrass et al. 2008, Sparling et al. 2004). Fragmentation that results from urbanization often puts pressure on existing wildlife to settle in alternative habitats (Crooks 2002). Wildlife can be seen utilizing urban habitats, such as residential properties, unmanaged lots, integrated open space preserves, or constructed land management areas (Cadenasso et al. 2006, McKinney 2008). Animals can be attracted from a network of habitat patches and stream corridors in proximity to the SWPs (Bishop et al. 2000, Brand and Snodgrass 2010). Ng et al. (2004) found that small mammal species, white-tailed deer (*Odocoileus virginianus*), and mesopredators utilize drainage culverts as corridors, which could facilitate their use of SWPs.

As urbanization destroys and converts the remaining forest patches, the SWPs of Baltimore County may serve as surrogate habitats for a variety of wildlife. In this study, I quantified the mammal community of 30 SWPs located in Owings Mills using capture-mark-recapture and presence-absence surveys. Based on small mammal capture histories, I modeled the capture and survival probability for two small mammal species, the deer mouse (*Peromyscus maniculatus*) and white-footed mouse (*P. leucopus*). I also modeled detection probability and site occupancy for meso- and large mammal species. *A priori*, I developed models to test multiple hypotheses about the influence of hydrology, SWP area, surrounding land use, and vegetation on pond use (Dooley and Bowers 1998, Fahrig

and Merriam 1985, Foster and Gaines 1991). Thus, I analyze the role of landscape and SWP characteristics in determining the mammals supported by a number of key SWP habitat characteristics. My results will encourage new land management approaches to aid in species conservation in an urbanized area.

First considering the predicted effects of urbanization on community structure, fragmentation clearly influences behaviors, interactions, and demographics of wildlife populations (McKinney and Lockwood 1999, Olden et al. 2004, Olden 2006). Changes in the patch area, connectivity among patches, and quality of habitat can reduce genetic and species diversity by restricting gene flow among populations and causing the extirpation of local species (McKinney 2002, Olden and Poff 2004). The degree of anthropogenic influence in surrounding land use should limit the presence of some species, but encourage generalist or non-native species, such as house mouse (*Mus musculus*), black rat (*Rattus rattus*), Virginia opossum (*Didelphis virginiana*), feral cats (*Felis catus*), and northern raccoons (*Procyon lotor*), to utilize SWPs as habitat (Chamberlain et al. 2007, Devictor et al. 2008, Gomez et al. 2008). I expected the degree of development to restrict the presence of rare Maryland species, such as the southern bog lemming (*Synaptomys cooperi*), smoky shrew (*Sorex fumeus*), or least-weasel (*Mustela nivalis*) (MDNR 2010).

Connectivity is a critical factor that affects density and movement and is important to the presence and persistence of urban mammals in SWPs (Fahrig 2003, McDonnell and Pickett 1990, Perault and Lomolino 2000). The urban matrix limits resources and habitat availability, which could restrict animal dispersal and accessibility

to SWP habitats (McKinney 2008, Perault and Lomolino 2000). The reduction of large contiguous habitats can force the extirpation of important species such as coyotes (*Canis latrans*) (Crooks 2002, Laliberte and Ripple 2004, Riley et al. 2003). Therefore, SWPs in proximity to other habitat patches, made evident by a high percent forest cover in the surrounding landscape, may have higher use rates by mammals.

A third SWP characteristic that may also affect the mammal community is hydrology. For example, SWPs with more permanent water could provide the necessary resources to sustain opportunistic feeders, like the northern raccoon, or specialists, such as the water shrew (*Sorex palustris*), southeastern star-nosed mole (*Condylura cristata*), or American beaver (*Castor canadensis*). Stormwater ponds that sporadically flood could destroy burrows, nests, or food, therefore impacting the survival of small terrestrial mammals (Anderson et al. 2000, Chamberlain and Leopold 2003, Williams et al. 2001).

Finally, vegetation is known to influence the demography of small (August 1983, Beck et al. 2004, Williams et al. 2002) and large (Bull et al. 2001, Catling et al. 2001, Rea 2003) mammals. Some small mammal species, such as, voles (*Clethrionomys* spp. and *Microtus* spp.) and shrews (*Sorex* spp., *Cryptotis* spp., and *Blarina* spp.) are reliant on the preservation of quality resources and microhabitat preferences, and react to the changes in vegetation often inflicted by land development (Dickman and Doncastor 1987, Dueser and Shugart 1978). Kohn and Walsh (1994) report that large habitat fragments encompass more spatial variation and can provide species with increased habitat types

and resources. Therefore, the larger SWPs with more heterogeneous vegetation should attract and support a more diverse mammal community.

CHAPTER II

Evaluation of the Urban Small Mammal Community and Effects of Landscape and Patch Characteristics on Survival in Stormwater Ponds

Urbanization replaces available habitat with the impervious surfaces of development and often results in the need for stormwater management plans. Without sufficient groundwater recharge, surface runoff creates flashfloods and decreases stream health. Stormwater ponds collect the surface runoff to mitigate flashfloods by allowing recharge, while promoting contaminant retention. With stormwater pond construction, watershed health is improved and a portion of habitat lost to urbanization may be restored for wildlife. It is important to understand what variables impact the species that are being attracted to these new habitat patches. I quantified the small mammal community of stormwater ponds in Baltimore County, Maryland, USA and found eight native species that were tolerant of urbanization, including two specialists (short-tailed shrew and meadow jumping mouse). Few non-native house mouse individuals were captured, suggesting positive prospects for the use of stormwater ponds in the management of urban biodiversity. I also used an information-theoretic approach to model landscape and stormwater pond characteristics that might influence the survival of two species, the deer mouse and white-footed mouse. Models predicted survival probability changed through time (ranging from 0.20 to 0.74), was function of hydrology, and was negatively influenced by shrub cover. The inverse relationship between shrub cover and survival

could be from opportunistic predators or the county's maintenance of overgrown vegetation.

Introduction

Urbanization is a large-scale landscape alteration resulting from the human conversion, destruction, and fragmentation of habitat (Fahrig 2003, McDonnell and Pickett 1990, McKinney 2002, Pickett and Cadenasso 2008). This process reduces resources, connectivity, and quality of habitat available to wildlife (Crooks 2002, Gibbs 2000, Hilderbrand et al. 2010, Laliberte and Ripple 2004, Prugh et al. 2008, Rodewald 2003). The presence of generalist species that are tolerant to human disturbances or well adapted to altered landscapes is a common characteristic of urban wildlife communities (McKinney 2002, Olden 2006). Specifically, small mammal communities may become dominated by fewer native and non-native species in response to urbanization (Bowers and Breland 1996, Cavia et al. 2009, Lowwell and Geis 1983, Sauvajot et al. 1998, Umetsu and Pardini 2007). Species commonly found in urban habitats include deer mouse (*Peromyscus maniculatus*), white-footed mouse (*P. leucopus*), and eastern gray squirrel (*Sciurus carolinensis*) (Dickman 1987, Dickman and Doncaster 1987, Dueser and Shugart 1978). Some urban habitats, for example city parks and vacant lots, can be dominated by non-native invasives such as the house mouse (*Mus musculus*), black rat (*Rattus rattus*), and Norway rat (*R. norvegicus*) (Gomez et al. 2008, Pocock et al. 2004).

Baltimore County, Maryland, implements “Best Management Practices” to offset some negative impacts of urbanization caused by new residential developments, shopping centers, and industrial parks (Baish and Caliri 2009). One management practice is the construction of stormwater ponds (hereafter SWPs), which mitigate flash floods and collect water runoff, thereby allowing the settlement of particulates and pollutants (Colosimo and Wilcock 2007, DEPS 2011). Thus, SWPs improve the overall watershed health. They vary in size, ranging from less than 500 to greater than 20,000 m^2 and are typically elliptic or round in shape. Stormwater ponds may be temporarily flooded for several days after a rain event or have a permanent water source. Some SWPs are frequently maintained to resemble mowed lawns, whereas others are not maintained so that plant succession leads to development of a vegetative community.

With the disappearance of natural habitats, SWPs could function as surrogate habitats for small mammal species. One might expect more generalists that are tolerant of urbanization to utilize SWPs (McKinney and Lockwood 1999). Although some of the wetlands and forest patches of SWPs could support specialists, such as the eastern harvest mouse (*Reithrodontomys humulis*) or water shrew (*Sorex palustris*).

To improve our understanding of how populations adapt to urbanization, I quantified the small mammal community of SWPs using a capture-mark-recapture study. I tested the influence of the surrounding landscape and various SWP characteristics on small mammal survival. Specifically, I hypothesized that increased forest cover in the surrounding landscape would positively influence survival, as a result of increased patch

connectivity (Fahrig and Merriam 1985, Henein and Merriam 1990, Prugh et al. 2008). One rationale behind this hypothesis is that during floods, more forest cover would allow individuals to escape the SWPs and return once more favorable conditions return. Because studies demonstrated an impact of patch size on population demography, I tested the effect of SWP area on survival (August 1983, Kohn and Wlash 1994, Nupp and Swihart 1996, Schweiger et al. 2000). Also, temporary flood events may impact small mammals by submerging their nests, shelter, and food resources (Shure 1971, Williams et al. 2001). Finally, vegetation is known to influence the demography of small mammals (August 1983, Beck et al. 2004, Dickman and Doncastor 1987, Dueser and Shugart 1978, Williams et al. 2002); therefore, I quantified the effects of vegetation heterogeneity and complexity on survival.

Methods

Study Area and Site Selection

All SWPs included in this study were located 24 km northwest of Baltimore City in Owings Mills, Maryland, USA at 39°25’19.52” N 76°46’52.59” E (Figure 1). The 54 km^{2} suburb was designated for Transit Oriented Development (TOD) in Baltimore County, which resulted in concentrated landscape alterations along the Baltimore Beltway (Interstate 695). Owings Mills was zoned for commercial, industrial, and residential development and therefore shifted from a rural landscape to an urban-rural matrix (Groffman et al. 2006, Pickett et al. 2008). Because of large-scale urbanization,

SWPs were constructed to improve stream quality in the Red Run watershed and Chesapeake Bay.

In 2007, 69 out of 170 available SWPs were randomly selected for a series of ecological studies and assigned an alpha numeric name based on their location within the landscape (Gallagher 2009). Using aerial photographs, the author quantified the percent forest cover in a 100 m buffer surrounding the SWPs. Gallagher (2009) also categorized pond hydrology as 1) intermittently exposed (IE) land with a permanent water body, 2) seasonally flooded (SF) basins that filled in wet months and dried through summer or 3) temporarily flooded (TF) basins that filled and drained after rain events. These classifications reflected inundation greater than 90%, 50% to 90%, and less than 50% of the year according to the National Wetland Inventory (NWI) hydrologic modifiers (Cowardin et al. 1979). In this study, I utilized percent forest cover within 100 m buffer of ponds and hydrology classifications to randomly select a subset of 30 SWPs from the original 69 using a random number generator (Microsoft Excel® 2012, Redmond, WA) (Figure 2). To ensure sufficient sites to randomly select from, I categorized SWPs as having less or greater than 40% forest cover within 100 m of ponds (Table 1).

The areas of the SWPs were determined from aerial photographs in ArcView GIS and ranged from 597 to 20,397 m^2 (Gallagher 2009). Stormwater ponds were encircled by a 1 m to 25 m elevated berm, which formed the perimeter, a grassy containment wall surrounding the settling basin, concrete inflow culvert, and emergency spillway (Figure 3). Eleven of these SWPs were managed with planting, seasonal mowing, and periodic

tree removal. The remaining SWPs resembled forests or wetlands and were dominated by forbs (Euphorbiaceae), cattails (*Typha* spp.), grasses (*Poaceae*, *Cyperaceae*, and *Juncaceae* spp.), mile-a-minute weed (*Persicaria perfoliata*), blackberry (*Rubus* spp.), black locusts (*Robinia pseudoacacia*), maples (*Acer* spp.), oaks (*Quercus* spp.), and willows (*Salix* spp.).

Vegetation Heterogeneity and Complexity Survey

In collaboration with Fitzpatrick (2011), I completed vegetation surveys during the summers of 2009 and 2010 by placing transects 10 m apart perpendicular to the longest length of the SWP. We recorded the distances of forbs, grass, and shrub cover types along transects and divided them by the total transect lengths to quantify vegetation heterogeneity and calculate percent cover for the SWP. Starting at 0 m and continuing at 10 m intervals along the transects, we recorded the vertical vegetation complexity as the number of times vegetation came into contact with each 0.25 m section of a 2 m PVC pole (Rotenberry and Wiens 1980). From these data, we calculated an average of all the sections as vegetation density for the entire SWP.

Mammal Recapture Survey

I conducted a capture-mark-recapture (CMR) study from March to December 2010 and from March to November 2011 in order to quantify the SWP small mammal community and estimate survival probabilities for small common species. Using a seven week primary sampling period, I surveyed all 30 SWPs for five occasions in 2010 and four in 2011. I used 5 x 6 x 16.5 cm live traps (Sherman Traps, Inc. © 2010, Tallahassee,

FL) set out in a 15 x 15 m grid that encompassed the dry area within the basin of each SWP. Depending on the size of the SWP and the area covered by standing water, the number of traps ranged from five to 117. I baited traps with a mixture of melted paraffin wax, oats, peanut butter, raisins, and rendered beef suet (Calhoun 1959). I checked traps every 24 hours at sunrise on four consecutive mornings to process all captures, then re-baited and reset the traps.

I identified each captured individual to genus or species, weighed them with a 300 g spring scale (Pesola Ag © 2012, Baar, Switzerland), attached a uniquely numbered 1005-1 monel ear tag (National Band and Tag Company, Newport, KY), sexed them, and assessed reproductive condition before release. I recorded testes as ascended or descended. I palpitated females for current pregnancies and status of the pubic symphysis for previous pregnancies. I also recorded the trap location of all captures and the ear tag number of previously marked individuals. The protocol for handling mammals followed guidelines for use of mammals in research (Sikes et al. 2011).

Capture and Survival Probability Analyses

I pooled daily captures into primary sampling periods to summarize the individuals' capture history over the two year study. Because it was not always possible to distinguish between the deer mouse and white-footed mouse in the field, I combined their capture histories into the genus *Peromyscus* for all statistical analyses in Program Mark version 6.0 (Rich et al. 1996, White and Burnham 1999). To determine the most appropriate CMR model for the analysis, I tested assumptions of the populations being

closed in time to births and deaths and in space to emigration and immigration in CloseTest version 3 (Stanley and Burnham 1999). I used a goodness-of-fit test in Program Release to assess how the most parameterized model (full dependence on sex and time) for capture (p) and survival (φ) probability would fit the data (Burnham et al. 1987, Lebreton et al. 1992). Depending on an estimate for overdispersion ($\hat{c}$) from the Median $\hat{c}$ test, I made adjustments to the model $\hat{c}$ (Anderson et al. 1994, White et al. 2002). The $\hat{c}$ value estimated the extra binomial variation of the most parameterized model, which is caused by small variations in independence or homogeneity of variance (Anderson et al. 1994, Burnham and Anderson 2002).

Multiple hypotheses tested the influence of the surrounding landscape and various SWP characteristics on the capture and survival probability of *Peromyscus* spp. A priori, I developed 40 Cormack-Jolly-Seber (CJS) models for open populations based on the influence of sex of the individual, sampling period (time), and landscape and SWP characteristics on capture and survival probability (Cormack 1964, Jolly 1965, Lebreton et al. 1992, Seber 1965). I first developed a null model of constant (.) capture and survival probabilities to create an over-simplified representation of the data for model comparisons. Next, I held survival probability constant and altered the factor of influence (sex or time) on capture probability before exploring all other combinations (Table 2). Finally, I added covariates to the best approximating models to investigate the influence of landscape and SWP characteristics and more effectively represent the variation in the data (Table 3).

I used an information-theoretic approach, specifically Akaike's information criterion (AIC), to select the best approximating generalized linear model without assuming the inclusion of the true model in the candidate set (Akaike 1973, Anderson and Burnham 1999, Johnson and Omland 2004). Akaike's information criterion measures the expected Kullback-Leibler distance between each model and reality, as well as the amount of information lost with the model's approximation of reality (Anderson and Burnham 1999, Burnham and Anderson 2002, Johnson and Omland 2004). Program Mark ranked the models by corrected AIC (AIC_c), an asymptotically efficient correction for small sample size, which had a lessening effect as sample size became more appropriate for the number of the parameters in the model (Hurvich and Tsai 1989). I then selected the most parsimonious model to minimize the change in AIC_c (ΔAIC_c) and maximize model weight (Anderson et al. 1994). I considered alternative models within two units of the top model for inference because of substantial support (Burnham and Anderson 2002). Selected models supplied maximum-likelihood parameter estimates of both capture and survival probability from slope estimates of the CJS logistic regressions.

Results

Vegetation Heterogeneity and Complexity of Stormwater Ponds

The SWP vegetation was highly variable (Table 4). Five SWPs (E4-2, F7-1, F8-3, F9-1, and I8-1) exhibited very little vegetation heterogeneity or complexity because they lacked a developed understory and more closely resembled lawns with a homogenous

cover type. The remaining 25 SWPs had more heterogeneous and complex vegetation. Maintenance crews from the county and property owners managed the vegetation in 11 ponds. They seeded the steep containment walls of E7-2 and G6-1 and covered the walls with a fibrous mat to reduce erosion. Crews cut down young trees in E4-1 and F4-1, cleared all trees and mowed G6-10 and I8-1, and removed vegetation off the berm in E9-2. Four ponds (E4-2, F7-1, F8-3, and F9-1) were cleared of shrubs and saplings then maintained by routine mowing through the growing season. The non-native invasive common reed (*Phragmites australis*) was present in three SWPs (E7-5, G6-14, and H8-2) and dominated the aquatic vegetation in the basin of E7-5 and H8-2.

Mammal Community of Stormwater Ponds

Over the course of the two year CMR study, I had 1,490 total captures comprised of 739 individuals (Table 5). I captured small mammals in all but one SWP (F8-3), which was one of the smallest SWPs with the lowest amount of vegetative cover. The most species diverse SWP was E10-2 where I captured short-tailed shrew (*Blarina brevicauda*), *Peromyscus* spp., meadow vole (*Microtus pennsylvanicus*), eastern chipmunk (*Tamias striatus*), and eastern cottontail (*Sylvilagus floridanus*).

Peromyscus spp. made up 80% of the captured individuals and had the highest rate of recapture. The SWP H8-2 had 98 captured individuals and 223 total captures, followed by F8-8 with 57 individuals and 75 total captures. I captured *Peromyscus* spp. individuals up to 13 times during the study. One individual was first captured on 4/10/10 and last captured on 11/4/11, marking the longest capture history that spanned 564 days.

Because only *Peromyscus* spp. had sufficient capture and recapture events, I focused on this genus for analysis of survival probability.

Twenty-two pregnant females were captured from May to October 2010 and from April to October 2011, indicating their reproductive period. Over 85% of captured females had an open pubic symphysis and three females gave birth while inside the traps. Of all male captures, 64% had scrotal testes and showed signs of reproduction in every sampling period.

I also observed fleas, ticks, and over 16 botfly larvae positioned under the skin of the hind leg of the mice. Additional insects, such as ants (*Formicidae* spp.), attacked the live individuals and dead specimens while inside the traps. Considering the possible negative impacts of ant colonies on wildlife (Allen et al. 2004, Beck et al. 2004), these ant attacks were considered forms of predation. I also found disemboweled or dismembered *Peromyscus* spp. individuals nearby tampered traps, evidence of predation by northern raccoons (*Procyon lotor*) and feral cats (*Felis catus*).

Capture and Survival Probabilities of Peromyscus

The *Peromyscus* spp. population was open to births, deaths, immigration, and emigration of individuals ($\chi^2 = 487.59_{(14)}$, $P = < 0.000$), therefore, I used the CJS models rather than the robust design for closed populations. The most parameterized model φ(sex * time) *p*(sex * time) for *Peromyscus* spp. fit the data ($\chi^2 = 12.35_{(28)}$, $P = 0.991$) and the calculated $\hat{c}$ estimated little under-dispersion with a confidence interval that bound the measure of one, suggesting no overdispersion ($\hat{c} = 0.99$, SE < 0.00, 95% CI = 0.99 –

1.00). Consequently, I did not make adjustments to the model $\hat{c}$ and selected models using ΔAIC_c values (Anderson et al. 1994). I used two models (Table 6) to test the influence of measured landscape and SWP characteristics (Table 7).

Together, the best approximating models accounted for 8% of the total variation and reflected the importance of vegetation (percent shrub cover) in survival probability. The two models estimated similar parameters, averaging 0.70 ± 0.02 SE for capture and 0.37 ± 0.02 SE for survival probability (Table 8). Although sex was included in the second model to influence capture probability, parameter estimates for males and females have similar values with overlapping confidence intervals. Survival probability estimates were highest in the spring season of April to June 2010 and lowest in the winter season of December 2010 to April 2011 (Figures 4 and 5). The parameter estimates did not vary according to hydrology classification due to overlapping variation among groups (Figures 6 and 7). Survival did decrease rapidly with increasing shrub cover (Figure 8 and 9).

Discussion

Mammal Community of Stormwater Ponds

The results demonstrate that small mammals utilize SWPs as habitat. I observed and captured small mammals that were foraging, nesting, and reproducing in these SWPs. Many of the captured and released individuals used extensive networks of runways and protective shelters, suggesting that much of their home ranges overlap the SWPs, instead of merely entering the SWP on occasion for supplemental resources (Bennett 1990,

Foster and Gaines 1991, Nupp and Swihart 2000). Members of a number of small mammal families have been documented as using habitat remnants in urban areas, including Dasyuridae, Peramelidae, Leporidae, Cricetidae, Muridae, Heteromyidae, and Soricidae (Baker et al. 2003, Dickman and Doncaster 1987, Garden et al. 2007, Sauvajot et al. 1998). Even the most isolated SWPs (E7-4, F4-1, F6-3, G6-5, and G6-10) surrounded by 0% forest cover apparently provided sufficient resources to support a small mammal community. No new species colonized or disappeared from a given SWP over the course of two years, indicating a stable community composition at least under the time frame of this study.

Species encountered in this study were typical to the urbanized mid-Atlantic piedmont region (Dueser and Shugarts 1978, Lackey et al. 1985, Manson et al.1999, Myton 1974, Stickel and Warback 1960) and were generalists that are tolerant of human disturbance (Adler and Wilson 1987, Dickman 1987, McKinney 2006, Seamon and Adler 1996). For example, *Peromyscus* spp. (deer mouse and white-footed mouse) range throughout most of North America and Mexico, utilizing a wide range of habitats including forests, fields, and urbanized areas (Fahrig and Merriam 1985, Lackey et al.1985, Lehmkuhl et al.2008, Peavey et al.1997, Schweiger et al.2000). Furthermore, they have an omnivorous diet (Adler 1987, Brannon 2005, Wywialowski 1987), thus it is not surprising that they were the most abundant species in this study. The second most abundant species, the short-tailed shrew, occurred sympatrically with *Peromyscus* spp. in 18 out of 30 SWPs. Shrew species are restricted to the moist leaf litter of forests,

marshes, and meadows with sufficient invertebrate resources (Ford et al.1994, George et al.1986, Lehmkuhl et al. 2008, Spencer and Pettus 1966, Stickel and Warback 1960). Stormwater ponds that were dominated by grassy vegetation also supported herbivorous species, such as meadow voles. The meadow jumping mouse (*Zapus hudsonius*) and eastern chipmunk were less common captures in the SWPs, although they are not classified as rare (Brannon 2005, Snyder 1982, Whitaker and Wrigley 1972).

Species commonly associated with wet meadows and open fields, such as the eastern mole (*Scalopus aquaticus*), southern red-backed vole (*Clethrionomys gapperi*), or least shrew (*Cryptotis parva*) were not encountered. By utilizing additional trapping techniques, including pitfall traps, I might have sampled these species (Wilson et al. 1996). However, the absence of these species may characterize biotic homogenization of the urban small mammal community, caused by urbanization and the stochastic environmental conditions of SWPs (Baker et al. 2003, Dickman 1987, Lowell and Geis 1983, McKinney 2006, Sauvajot et al. 1998).

Surprisingly, non-native or invasive species that are generally associated with highly disturbed habitats and densely populated urbanized areas did not dominate SWP small mammal communities (Gomez et al. 2008, McKinney 2002, Pocock et al. 2004). Unlike other studies, I did not capture black or Norway rats, and encountered only nine individuals of the house mouse in six SWPs. Previous studies reveal that rat populations rely on waste stations, overgrown lots, vacant buildings, sewers, and railways for food, shelter, runway systems, and nesting sites (Bajomi and Sasvari 1986, Barbehenn 1970,

Childs et al. 1998, Nolte et al. 2003). One reason for their absence in SWPs could be that the surrounding landscape of Owings Mills lacks the qualities of inner-city disturbances because of land planning, community associations, and moderate human population densities.

Besides the intended function for improving watershed health, SWPs also function as valuable surrogate habitats in an urban landscape. Stormwater ponds with increased vegetative heterogeneity and complexity (E3-1, E7-5, E9-2, E10-2, and F7-5) had the most species. First, SWPs support both generalist and specialist small mammal species that are native to the region. Second, these species are ecologically important because they consume seeds and reduce the insect populations (Buckner 1966, Liebhold et al. 2000, Ivan and Swihart 2000). Third, small mammals are food resources for snakes, birds, and carnivorous mammals (Bartoszewicz et al. 2008, Cavallini and Volpi 1996, Dell'Arte et al. 2007, Fitzpatrick 2011, Norrdahl and Korpimaki 1995). Thus, the findings of a moderately diverse small mammal community utilizing and persisting in SWPs may have new implications for species conservation and land management over larger spatial scales.

Capture and Survival Probability of Peromyscus

Capture probability in *Peromyscus* spp. models varied according to two factors of influence. In one model, the parameter estimate was constant and did not vary through time. However, in the second model, sex of the individual was an important influence on the capture probability for *Peromyscus* spp. Although capture estimates between the

sexes have similar ranges in variation, females were in fact captured less frequently than males, which could be related to sexual dimorphisms in habitat use. Males have larger territories, overlapping the territories of multiple potential mates (Lackey et al. 1985, Otsfeld 1990, Wolff 1985). Since reproductive maturity is suppressed by the presence of a parent, males also tend to disperse as juveniles in order to secure a territory and consequently spend more time foraging (Lackey et al. 1985, Myton 1974). Females on the other hand, have very rigid territories used for rearing young, which they aggressively defend from intruders and the risk of infanticide (Ostfeld 1990, Wolff 1993, Wolff and Cicirello 1991).

The same three factors were included in both top models, demonstrating their critical influence on the survival probability of *Peromyscus* spp.. First, survival estimates fluctuated through time. For instance, the winter of 2009 to 2010 was exceptionally harsh with an average monthly precipitation of 126 mm and two months with above average snowfalls resulting in a late spring (NOAA 2009, 2010, 2010). Following this severe winter, there was a higher estimation for the survival probability between the first and second sampling period that took place from April to June 2010, a possible result of cached food resources and shelter through the winter. By comparison, all subsequent surveys showed a reduction in survival probability. Through the summer and fall seasons, predators like the red fox are rearing young and the increased demand for food by their pups requires increased predation on small mammals (Lariviere and Pasitschniak-Arts 1996, Lindstrom 1994, Sargeant 1978). The winter of 2010 to 2011 was relatively mild

with an average precipitation of 61 mm and could have allowed for continued predation, instead of another high survival estimate for the second year's spring (NOAA 2010, 2011, 2011). There was also a four month hiatus in trapping, where individuals could have died or migrated while new individuals were not being captured.

Stormwater pond hydrology was the second factor of influence included in both top models. How did the unpredictable flooding of SWPs impact the survival of *Peromyscus* spp? Runoff was channeled into the SWPs during rain events and even filled their basins during extreme flashfloods of April 2010 and September 2011. Water levels rose as much as 2 m, forcing small mammal residents to temporarily move into the surrounding landscape. Interestingly, residents seemed less impacted by hydrology and were recaptured even after multiple flooding events (Anderson et al. 2000, O'Connell 1989). Therefore, it is not surprising that survival estimates among SWP hydrology classifications did not differ. Despite the intended function and the varied hydrology of SWPs, they did serve as habitats for at least six small mammal species.

Finally, survival probability of *Peromyscus* spp. was negatively influenced by percent shrub cover. Vegetation heterogeneity and complexity are important to *Peromyscus* spp. because it provides cover, shelter, and food resources (Dueser and Shugart 1978, Kitchings and Levy 1981, Williams et al. 2002). For example, Anderson et al. (2003) tested the effect of vegetation complexity on white-footed mouse densities by live-trapping the edge and interiors of habitat patches. *Peromyscus* spp. densities were greatest in the edge habitat, where the most complex herbaceous vegetation structure

occurred and few shrubs occurred. Due to their position alongside the forested riparian buffer and general small size, SWPs can create edge habitats dominated by herbaceous plants and shrubs that small mammals often colonize. The complex vegetation and microclimates inside these SWPs also attract deft predators (Blouin-Demers and Weatherhead 2001, Charland and Gregory 2009, Weatherhead et al. 2003) such as the eastern gartersnake (*Thamnophis sirtalis*), black rat snake (*Elaphe obsolete*), and the northern black racer (*Coluber constrictor*). Blouin-Demers and Weatherhead (2001) found that black rat snakes select edge habitat for thermoregulation, but engage in opportunistic foraging and negatively impacts nesting birds and small mammals. Weatherhead et al. (2010) states that increased predation in fragmented habitats is a result of larger snake populations, rather than increased activity in edge habitats. Stormwater ponds with dense vegetation and overgrown shrubs could become the focus of management plans that destroy available resources or directly harm individuals. Therefore, management regimes that setback plant succession by mowing vegetation and clearing trees could negatively affect the survival probability of *Peromyscus* spp.

Sediments, salts, and metals that accumulate on roadways are washed away by rain events and deposited in SWPs, thereby collecting in a concentrated area for an indeterminate amount of time (Camponelli et al. 2010, Casey et al. 2005). The metal Zn was detected in earthworms (*Lumbricus terrestris*) (Lev et al. 2010), a common food source for various shrew species (Brooks and Doyle 2001, George et al. 1986). As these metals accumulate in the environment, they potentially negatively affect small mammal

populations in SWPs (Johnson et al. 1978, Ma et al. 1991, Sparling et al. 2004). Future studies should test whether SWPs function as a source or sink habitat, or ecological trap for small mammals.

Stormwater ponds are a “Best Management Practice” to benefit the health of the Chesapeake Bay watershed. This study demonstrates that SWPs provide habitat for small mammals in an urban landscape. Brand and Snodgrass (2010) found that amphibians, which utilized SWPs during reproduction, had a higher reproductive success than individuals breeding in natural wetlands because natural wetlands had a greater tendency to dry before metamorphosis was completed. A recent study quantified diversity of SWPs and revealed the presence of 11 species of amphibians, including anurans and caudates (Simon et al. 2009). Fitzpatrick (2011) found 27 bird species occurring in the 69 SWPs in Owings Mills, Maryland. These individuals represented specialist and generalist species, were associated with edge, grassland, and wetland habitat, and utilized SWPs for nesting, foraging, and singing. Given the extensive evidence of wildlife species utilizing SWPs as habitat, management strategies that promote their colonization and persistence should be considered during the planning process. For example, connectivity among SWPs and with natural riparian systems, sufficient size of SWP, planting native aquatic and terrestrial plant species, and promoting plant succession could improve SWP habitat quality for wildlife. With proper management, SWPs could at least partially counteract the loss of wetland habitats in urbanizing areas. As new species are attracted to the

constructed wetlands, SWPs may increase species diversity, becoming critical biological patches and aesthetic islands in the urban landscape.

Table 1. Balanced factorial design of 30stormwater ponds (SWPs) located in Owings Mills, Maryland, USA. Sites were selected for percent forest cover in the surrounding landscape and hydrology, defined as intermittently exposed (IE) land with a permanent water body, seasonally flooded (SF) basins, and temporarily flooded (TF) basins.

%Forest Cover	IE	SF	TF
<40	E4-1	E7-2	E7-4
	E4-4	F4-1	F7-1
	E9-2	F6-3	F8-8
	G6-10	F8-3	G6-5
	G6-12	H7-1	H5-1
>40	H8-2	E4-2	D7-1
	E10-2	E7-5	E3-1
	E9-4	G6-1	E3-2
	F9-1	G8-2	F7-5
	G6-14	I8-1	I8-2

Table 2. Candidate Cormack-Jolly-Seber models for estimating capture (*p*) and survival (φ) probabilities of *Peromyscus* spp. in stormwater ponds of Owings Mills, Maryland, USA. In Program Mark, estimates were modeled as constant (.) or varied by sex or sampling period (time). Model 11 tested for assumptions of model fit and closure.

Number	Model
1	φ(.) *p*(.)
2	φ(.) *p*(sex)
3	φ(.) *p*(sex * time)
4	φ(.) *p*(time)
5	φ(sex) *p*(.)
6	φ(sex) *p*(sex)
7	φ(sex) *p*(sex * time)
8	φ(sex) *p*(time)
9	φ(sex * time) *p*(.)
10	φ(sex * time) *p*(sex)
11	φ(sex * time) *p*(sex * time)
12	φ(sex * time) *p*(time)
13	φ(time) *p*(.)
14	φ(time) *p*(sex)
15	φ(time) *p*(sex * time)
16	φ(time) *p*(time)

Table 3. Candidate Cormack-Jolly-Seber models for estimating capture (*p*) and survival (φ) probabilities of *Peromyscus* spp. in stormwater ponds (SWPs) of Owings Mills, Maryland, USA. In Program Mark, estimates were modeled to vary by hydrology, area, the proportion of forest cover in the surrounding landscape, the proportion of forbs, grass, or shrub cover, and vegetation density inside the SWP.

Number	Model
1	φ *p*(%forbs)
2	φ *p*(%grass)
3	φ *p*(%shrub)
4	φ *p*(veg.density)
5	φ(hydrology) *p*
6	φ(area) *p*
7	φ(%forest) *p*
8	φ(%forbs) *p*
9	φ(%grass) *p*
10	φ(%shrub) *p*
11	φ(veg.density) *p*

Table 4. Summary of landscape and stormwater pond (SWP) characteristics for 30 sites in Owings Mills, Maryland, USA. Surveys occurred during the summers of 2009 and 2010. Hydrology was defined as intermittently exposed (IE), seasonally flooded (SF), or temporarily flooded (TF) SWP basins. Vegetation density is measured in hits per point intercept.

SWP	Hydrology	Area (m^2)	%Forest	%Forbs	%Grass	%Shrub	Vegetation Density
D7-1	TF	1931	0.47	0.75	0.17	0.00	10.00
E3-1	TF	2864	0.84	0.42	0.22	0.03	12.08
E3-2	TF	2558	0.51	0.50	0.19	0.13	13.14
E4-1	IE	1843	0.17	0.23	0.24	0.14	13.37
E4-2	SF	1889	0.71	0.24	0.56	0.01	11.15
E4-4	IE	597	0.38	0.03	0.15	0.01	10.62
E7-2	SF	4858	0.09	0.11	0.52	0.00	5.21
E7-4	TF	1867	0.00	0.51	0.10	0.13	11.56
E7-5	SF	20397	0.61	0.44	0.39	0.04	15.84
E9-2	IE	5536	0.28	0.12	0.06	0.07	8.93
E9-4	IE	6754	0.54	0.38	0.19	0.05	15.38
E10-2	IE	2061	0.51	0.55	0.15	0.01	14.39
F4-1	SF	1696	0.00	0.27	0.09	0.10	13.43
F6-3	SF	1104	0.00	0.08	0.37	0.17	8.25
F7-1	TF	1266	0.37	0.57	0.32	0.02	11.31
F7-5	TF	7963	0.54	0.62	0.26	0.07	14.95
F8-3	SF	669	0.38	0.09	0.02	0.00	3.75
F8-8	TF	3110	0.22	0.65	0.11	0.06	25.71

F9-1	IE	2980	0.66	0.18	0.20	0.00	5.79
G6-1	SF	1825	0.59	0.64	0.08	0.00	15.83
G6-5	TF	648	0.00	0.39	0.17	0.11	21.00
G6-10	IE	4068	0.00	0.31	0.05	0.02	6.72
G6-12	IE	1111	0.25	0.33	0.50	0.00	14.47
G6-14	IE	2446	0.68	0.06	0.13	0.01	9.53
G8-2	SF	5341	0.58	0.49	0.21	0.06	16.32
H5-1	TF	4221	0.17	0.74	0.03	0.10	14.00
H7-1	SF	3185	0.27	0.73	0.18	0.02	22.02
H8-2	IE	10307	0.41	0.23	0.06	0.23	13.18
I8-1	SF	790	0.41	0.50	0.14	0.09	17.36
I8-2	TF	1968	0.78	0.23	0.42	0.11	17.23

Table 5. Capture-mark-recapture summary of urban mammal species in 30 stormwater ponds (SWPs) located in Owings Mills, Maryland, USA. This study utilized live traps over nine sampling periods from March 2010 to November 2011. The eastern cottontail and Virginia opossum individuals were both juveniles.

Species	Individuals	% Recaptured	♀	♂	Deaths	Weight (g)	SWPs
Deer and white-footed mouse (*Peromyscus maniculatus* and *P. leucopus*)	598	0.59	242	301	76	3 - 39	28
Eastern chipmunk (*Tamias striatus*)	15	0.06	-	1	2	75 - 95	6
Eastern cottontail (*Sylvilagus floridanus*)	1	0.00	-	-	0	72	1
House mouse (*Mus musculus*)	9	0.10	5	4	0	9 - 21	6
Meadow jumping mouse (*Zapus hudsonius*)	1	0.00	-	-	1	-	1
Meadow vole (*Microtus pennsylvanicus*)	37	0.03	12	16	6	12 - 57	17
Short-tailed shrew (*Blarina brevicauda*)	77	0.00	-	-	51	10 - 23	18
Virginia opossum (*Didelphis virginiana*)	1	0.00	-	-	0	-	1

Table 6. Ranked Cormack-Jolly-Seber models estimating capture (*p*) and survival (φ) probabilities for *Peromyscus* spp. in stormwater ponds of Owings Mills, Maryland, USA. Estimates were modeled as constant (.) or varied by sex or sampling period (time). *k* = number of parameters in the model; ΔAIC_c = change in the AIC_c value; w_i = model weight; Var = variance explained.

Model	*k*	AIC_c	ΔAIC_c	w_i	Model Likelihood	Var
phi(time) p(.)	9	2424.42	0.00	0.62	1.00	0.02
phi(time) p(sex)	10	2425.83	1.41	0.31	0.50	0.02
phi(time) p(time)	15	2428.85	4.43	0.07	0.11	0.02
phi(sex * time) p(.)	17	2436.15	11.73	0.00	0.00	0.02
phi(time) p(sex * time)	20	2436.20	11.77	0.00	0.00	0.02
phi(sex * time) p(sex)	18	2438.10	13.68	0.00	0.00	0.02
phi(sex * time) p(time)	22	2439.10	14.68	0.00	0.00	0.02
phi(sex * time) p(sex * time)	26	2445.20	20.78	0.00	0.00	0.03
phi(.) p(time)	9	2457.50	33.08	0.00	0.00	0.01
phi(sex) p(sex)	4	2458.64	34.22	0.00	0.00	0.00
phi(.) p(.)	2	2458.75	34.32	0.00	0.00	0.00
phi(sex) p(time)	10	2459.23	34.81	0.00	0.00	0.01
phi(sex) p(.)	3	2459.36	34.93	0.00	0.00	0.00
phi(.) p(sex)	3	2460.09	35.67	0.00	0.00	0.00
phi(.) p(sex * time)	17	2471.49	47.06	0.00	0.00	0.01
phi(sex) p(sex * time)	18	2472.64	48.22	0.00	0.00	0.01

Table 7. All ranked Cormack-Jolly-Seber models using landscape and stormwater pond (SWP) characteristics to estimate capture (*p*) and survival (φ) probabilities for *Peromyscus* spp. in Owings Mills, Maryland, USA. In Program Mark, estimates were modeled as constant (.) or varied by sex, sampling period (time), hydrology, area, the proportion of forest cover in the surrounding landscape, the proportion of forbs, grass, or shrub cover, and vegetation density inside the SWP. *k* = number of parameters in the model; ΔAIC_c = change in the AIC_c value; w_i = model weight; Var = variance explained.

Model	*k*	AIC_c	ΔAIC_c	w_i	Model Likelihood	Var
phi(time + hydrology + shrub) p(.)	13	2382.62	0.00	0.66	1.00	0.04
phi(time + hydrology + shrub) p(sex)	14	2383.97	1.35	0.34	0.51	0.04
phi(time + hydrology) p(.)	12	2403.06	20.44	0.00	0.00	0.03
phi(time + hydrology) p(sex)	13	2404.44	21.82	0.00	0.00	0.03
phi(time + shrub) p(.)	10	2408.13	25.51	0.00	0.00	0.03
phi(time + shrub) p(sex)	11	2409.51	26.89	0.00	0.00	0.03
phi(time + grass) p(.)	10	2416.32	33.70	0.00	0.00	0.02
phi(time + grass) p(sex)	11	2417.68	35.06	0.00	0.00	0.02
phi(time + forbs) p(.)	10	2419.79	37.17	0.00	0.00	0.02
phi(time + forbs) p(sex)	11	2421.19	38.58	0.00	0.00	0.02
phi(time) p(.)	9	2424.42	41.81	0.00	0.00	0.02
phi(time) p(forbs)	10	2424.48	41.86	0.00	0.00	0.02
phi(time) p(veg.density)	10	2425.02	42.40	0.00	0.00	0.02
phi(time + area) p(.)	10	2425.31	42.69	0.00	0.00	0.02

phi(time) p(grass)	10	2425.53	42.91	0.00	0.00	0.02
phi(time) p(sex)	10	2425.83	43.21	0.00	0.00	0.02
phi(time) p(sex + forbs)	11	2425.88	43.26	0.00	0.00	0.02
phi(time + forest) p(.)	10	2426.18	43.56	0.00	0.00	0.02
phi(time + veg.density) p(.)	10	2426.35	43.73	0.00	0.00	0.02
phi(time) p(shrub)	10	2426.43	43.82	0.00	0.00	0.02
phi(time) p(sex + veg.density)	11	2426.53	43.91	0.00	0.00	0.02
phi(time + area) p(sex)	11	2426.73	44.11	0.00	0.00	0.02
phi(time) p(sex + grass)	11	2426.73	44.11	0.00	0.00	0.02
phi(time + forest) p(sex)	11	2427.58	44.96	0.00	0.00	0.02
phi(time + veg.density) p(sex)	11	2427.76	45.14	0.00	0.00	0.02
phi(time) p(sex + shrub)	11	2427.85	45.23	0.00	0.00	0.02
phi(time) p(time)	15	2428.85	46.23	0.00	0.00	0.02
phi(sex * time) p(.)	17	2436.15	53.53	0.00	0.00	0.02
phi(time) p(sex * time)	20	2436.20	53.58	0.00	0.00	0.02
phi(sex * time) p(sex)	18	2438.10	55.48	0.00	0.00	0.02
phi(sex * time) p(time)	22	2439.10	56.48	0.00	0.00	0.02
phi(sex * time) p(sex * time)	26	2445.20	62.58	0.00	0.00	0.03
phi(.) p(time)	9	2457.50	74.88	0.00	0.00	0.01
phi(sex) p(sex)	4	2458.64	76.03	0.00	0.00	0.00
phi(.) p(.)	2	2458.75	76.13	0.00	0.00	0.00
phi(sex) p(time)	10	2459.23	76.61	0.00	0.00	0.01
phi(sex) p(.)	3	2459.36	76.74	0.00	0.00	0.00
phi(.) p(sex)	3	2460.09	77.47	0.00	0.00	0.00
phi(.) p(sex * time)	17	2471.49	88.87	0.00	0.00	0.01
phi(sex) p(sex * time)	18	2472.64	90.02	0.00	0.00	0.01

Table 8. Parameter estimates of capture (*p*) and survival (φ) probabilities for *Peromyscus* spp. in 30 stormwater ponds (SWPs) located in Owings Mills, Maryland, USA. These estimates were modeled to vary by sex, sampling period (time), vegetation density, or the proportion of grass cover inside the SWP. SE = standard error; 95% CI = 95% confidence interval.

Model	Parameter	Estimate	SE	95% CI
phi(time + hydrology + shrub) p(.)	φ_1	0.66	0.04	0.58-0.72
	φ_2	0.44	0.04	0.36-0.53
	φ_3	0.39	0.05	0.31-0.49
	φ_4	0.32	0.05	0.24-0.42
	φ_5	0.20	0.03	0.14-0.28
	φ_6	0.28	0.06	0.18-0.41
	φ_7	0.35	0.07	0.23-0.49
	φ_8	0.34	0.07	0.23-0.48
	φ_{IE}	0.35	0.03	0.30-0.41
	φ_{TF}	0.40	0.03	0.34-0.47
	φ_{SF}	0.35	0.03	0.30-0.42
	p	0.69	0.04	0.61-0.77
phi(time + hydrology + shrub) p(sex)	φ_1	0.66	0.04	0.58-0.72
	φ_2	0.44	0.04	0.36-0.53
	φ_3	0.40	0.05	0.31-0.49
	φ_4	0.32	0.05	0.24-0.42
	φ_5	0.20	0.03	0.14-0.28
	φ_6	0.28	0.06	0.18-0.41
	φ_7	0.35	0.07	0.23-0.49
	φ_8	0.34	0.07	0.23-0.48
	φ_{IE}	0.35	0.03	0.30-0.41
	φ_{TF}	0.40	0.03	0.34-0.47

φ_{SF}	0.35	0.03	0.30-0.42
p_{Female}	0.67	0.05	0.58-0.76
p_{Male}	0.74	0.06	0.60-0.84

Figure 1. Map of Baltimore County including Owings Mills, MD. County lines are shown in grey, the Red Run watershed in orange, and impervious surface cover in pink.

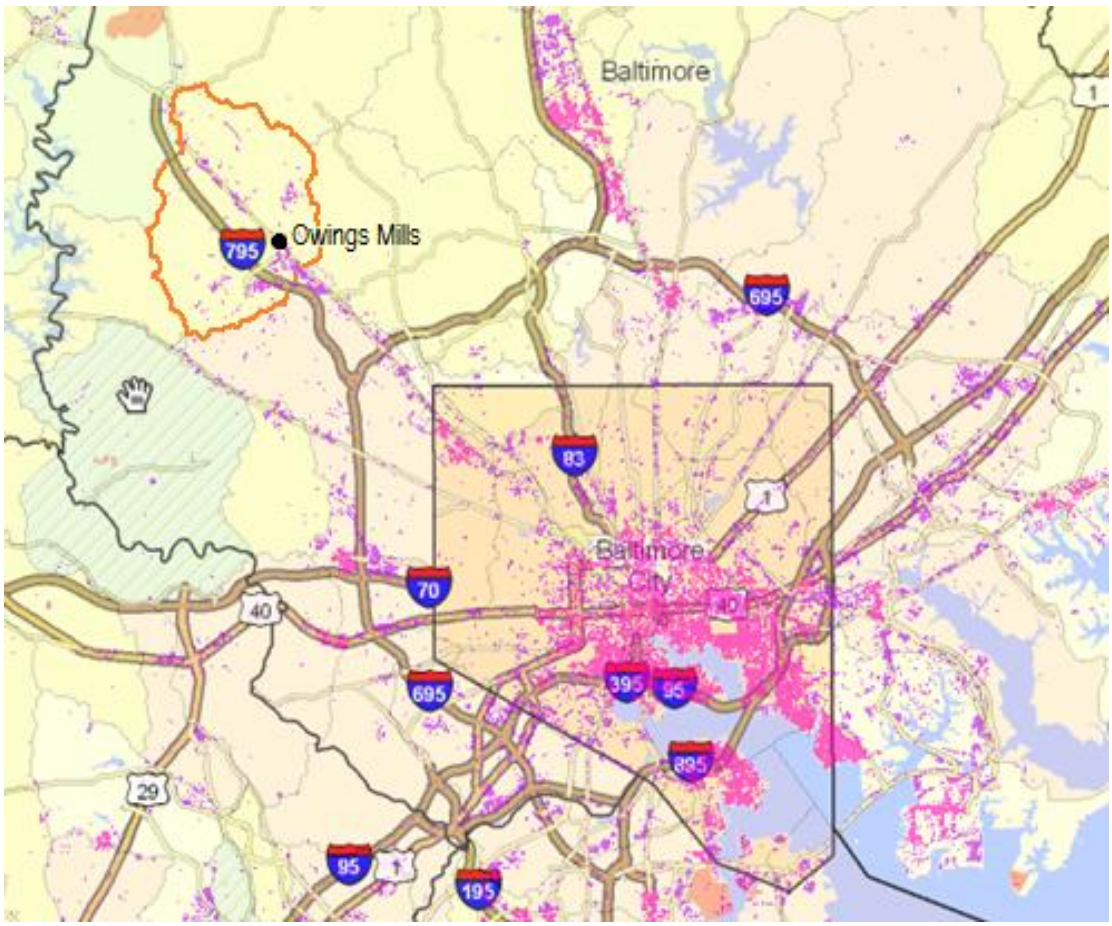


Figure 2. Satellite image (Google © 2011, Mountain View, CA) of the riparian buffer, development, and 30 randomly selected stormwater ponds (SWPs) in Owings Mills, Maryland, USA.

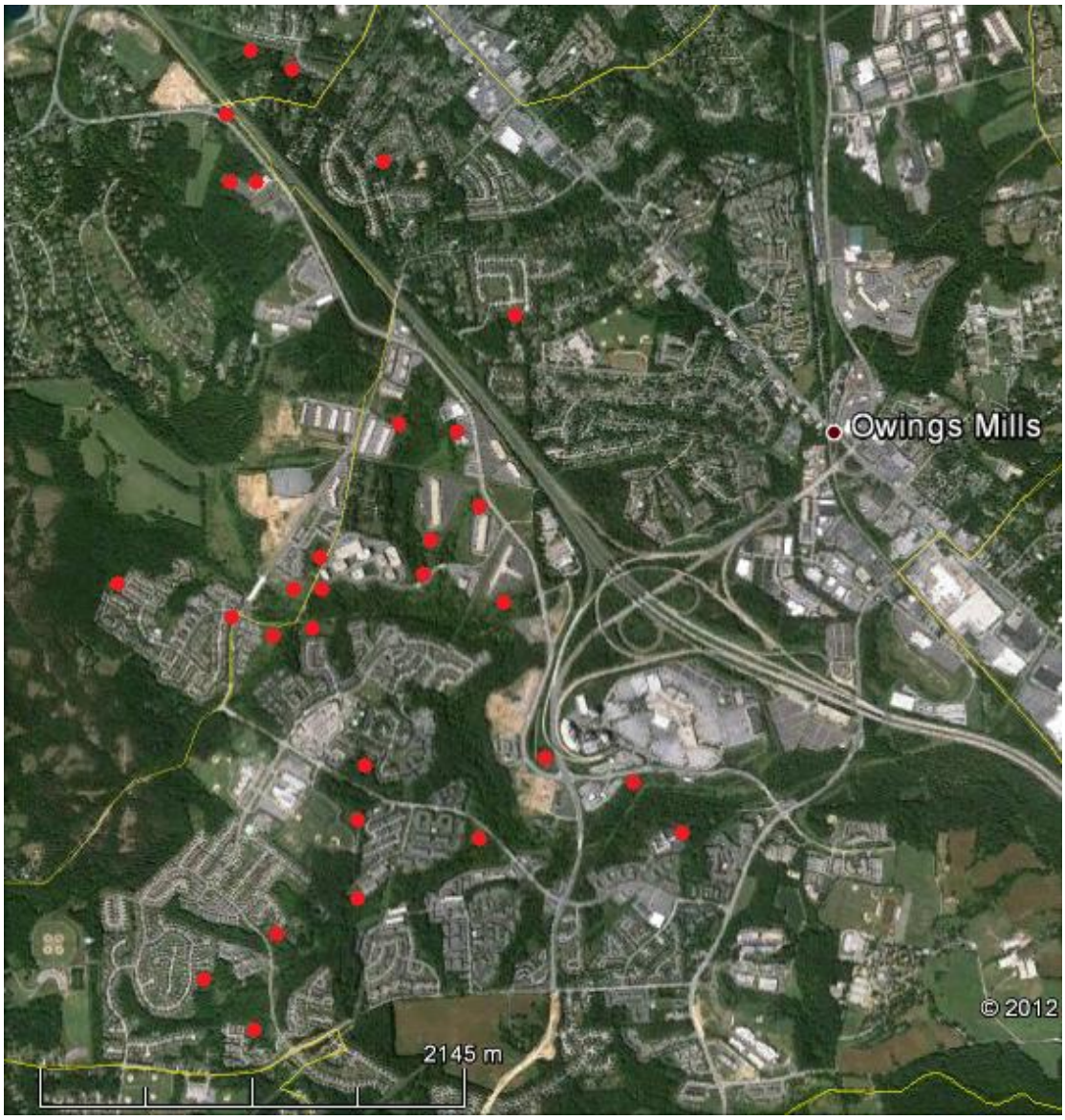


Figure 3. Typical stormwater pond architecture. Water enters through a concrete pipe to fill the basin. In an extreme rain event, excess water flows into the emergency spillway instead of compromising the integrity of the walls or overflowing the catchment system.

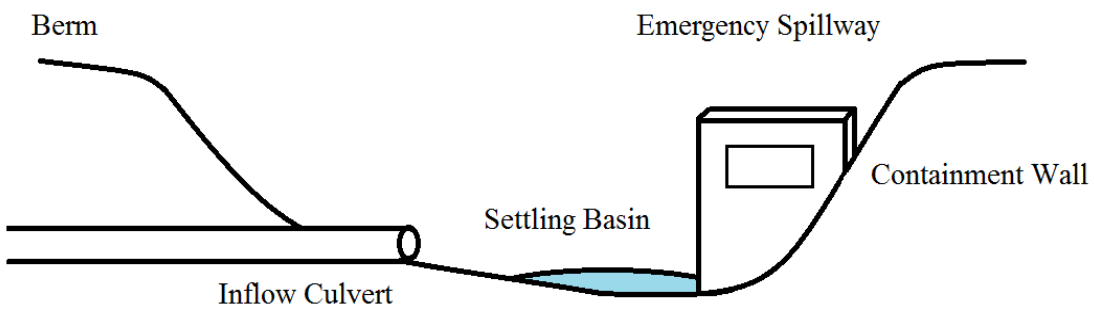


Figure 4. Relationship between time and estimated survival probability of *Peromyscus* spp. from model φ (time + hydrology + %shrub) *p*(.), where *p* is capture probability and φ is survival probability. The solid line represents the parameter estimate and the dotted lines are the upper and lower 95% confidence intervals. phi_1 = April to May 2010; phi_2 = June to July 2010; phi_3 = July to August 2010; phi_4 = September to October 2010; phi_5 = November 2010 to March 2011; phi_6 = April to May 2011; phi_7 = June to July 2011; phi_8 = .August to September 2011.

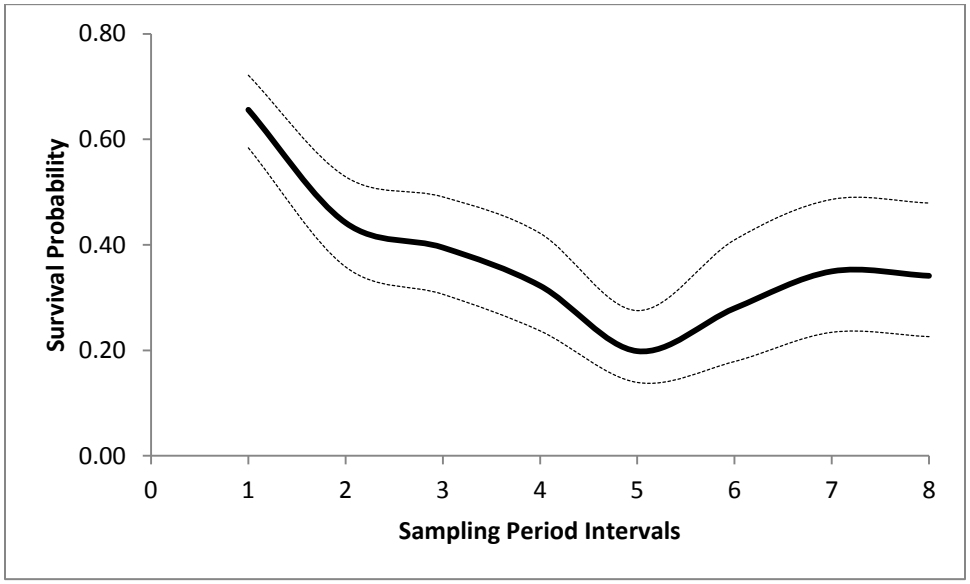


Figure 5. Relationship between time and estimated survival probability of *Peromyscus* spp. from model φ(time + hydrology + %shrub) *p*(sex), where *p* is capture probability and φ is survival probability. The solid line represents the parameter estimate and the dotted lines are the upper and lower 95% confidence intervals. phi_1 = April to May 2010; phi_2 = June to July 2010; phi_3 = July to August 2010; phi_4 = September to October 2010; phi_5 = November 2010 to March 2011; phi_6 = April to May 2011; phi_7 = June to July 2011; phi_8 = .August to September 2011.

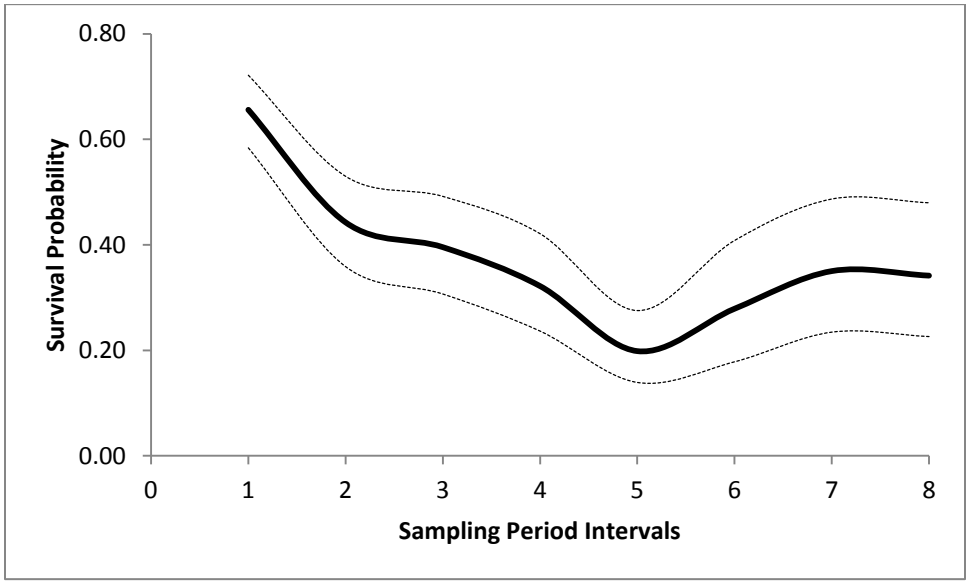


Figure 6. Relationship between the stormwater pond hydrology and estimated survival probability of *Peromyscus* spp. from model φ (time + hydrology + %shrub) *p*(.), where *p* is capture probability and φ is survival probability.

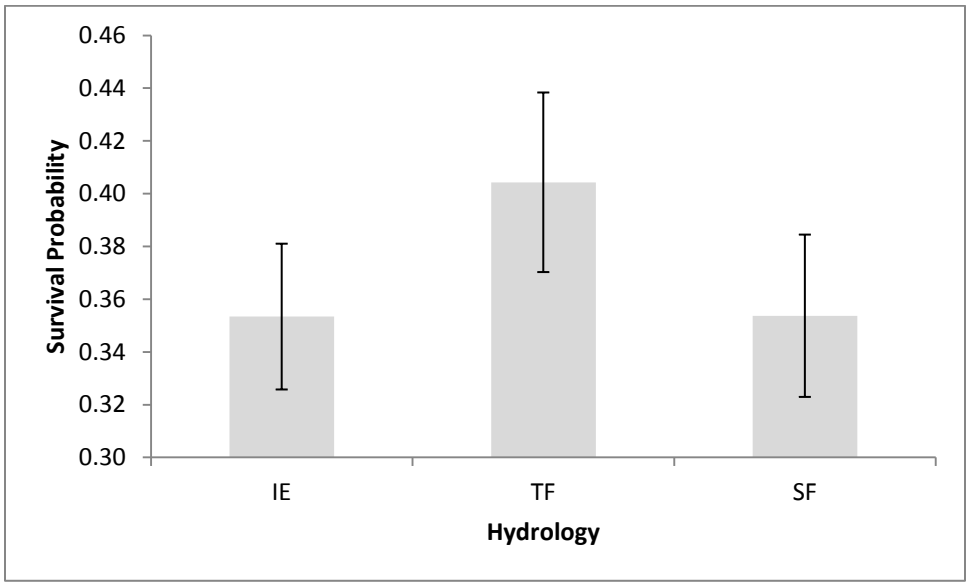


Figure 7. Relationship between the stormwater pond hydrology and estimated survival probability of *Peromyscus* spp. from model φ (time + hydrology + %shrub) *p*(sex), where *p* is capture probability and φ is survival probability.

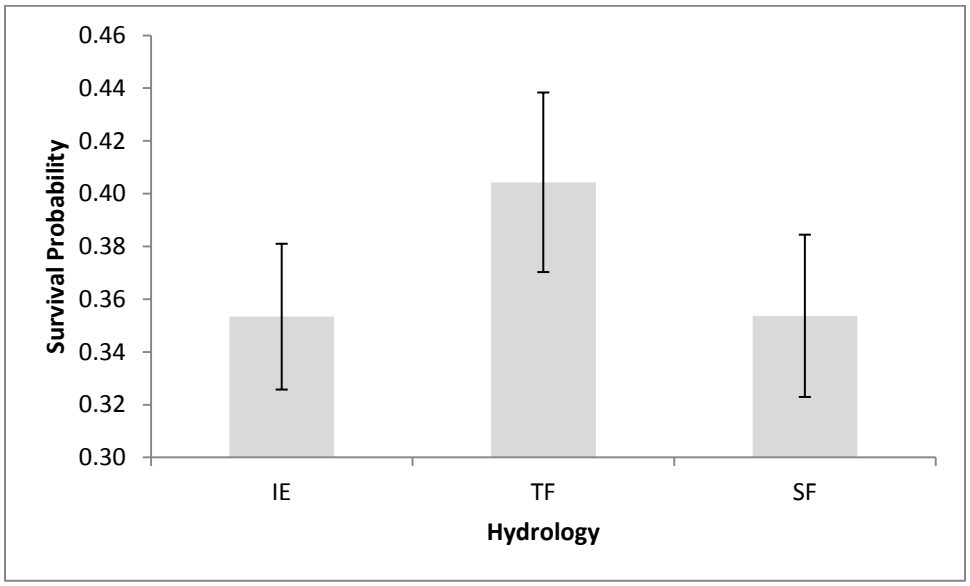


Figure 8. Relationship between the percent shrub cover inside the stormwater pond and estimated survival probability of *Peromyscus* spp. from model φ (time + hydrology + %shrub) *p*(.), where *p* is capture probability and φ is survival probability. The solid line represents the parameter estimate and the dotted lines are the upper and lower 95% confidence intervals.

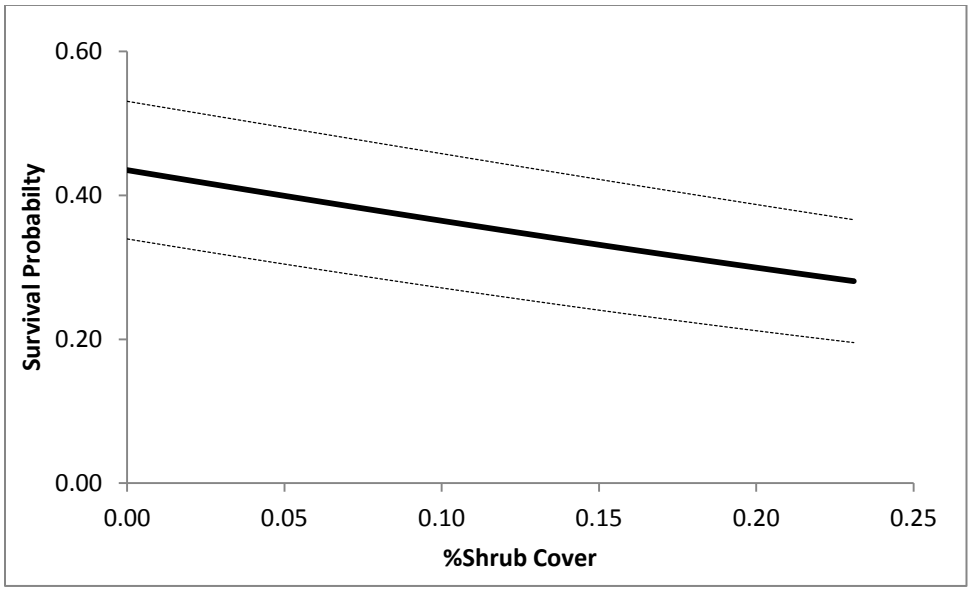
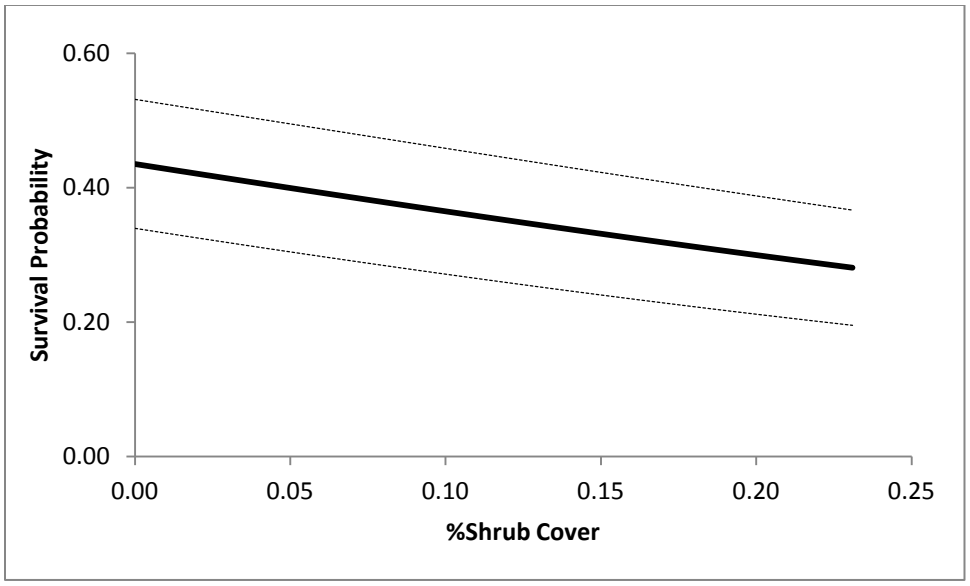


Figure 9. Relationship between the percent shrub cover inside the stormwater pond and estimated survival probability of *Peromyscus* spp. from model φ (time + hydrology + %shrub) *p*(sex), where *p* is capture probability and φ is survival probability. The solid line represents the parameter estimate and the dotted lines are the upper and lower 95% confidence intervals.



CHAPTER III

Evaluation of the Urban Meso and Large Mammal Community and Effects of Landscape and Patch Characteristics on Occupancy in Stormwater Ponds

Stormwater ponds are a “Best management Practice” incorporated into land development protocols to lessen the negative effects of urbanization on the watershed. They collect surface runoff, preventing large quantities of contaminated water from directly entering stream systems. These constructed wetlands improve watershed health, but could also lessen habitat loss during urbanization by providing habitat for wildlife. Using multiple detection methods, I quantified the meso- and large mammal community that uses stormwater ponds in Baltimore County, Maryland, USA and detected 10 generalist species that are tolerant of urbanization. For eight of these species, I used an information-theoretic approach to model landscape and stormwater pond characteristics that might influence their occupancy. Gray squirrel occupancy (1 ± 0.00 SE) was influenced by time, most likely because of sparse detections. I was unable to distinguish influential landscape and stormwater pond characteristics for the eastern cottontail, groundhog, and white-tailed deer because of a lack of variation in the presence-absence data and sparse detections. The occupancy of carnivores such as feral cats (0.41 ± 0.02 SE), Virginia opossum (0.27 ± 0.02 SE), northern raccoon (0.98 ± 0.02 SE), and red fox (0.64 ± 0.02 SE), was negatively affected by the percent forest cover in the surrounding landscape, suggesting the reduced reliance on stormwater ponds as habitat when more forest is available in the immediate landscape.

Introduction

Urbanization is the development of the landscape that results in the conversion, destruction, and fragmentation of available habitat (Fahrig 2003, McDonnell and Pickett 1990, McKinney 2002, Pickett and Cadenasso 2008). These large-scale alterations impact local wildlife by reducing habitat connectivity and quality, as well as resource availability (Crooks 2002, Gibbs 2000, Hilderbrand et al. 2010, Rodewald 2003). For example, Laliberte and Ripple (2004) found that several species of ungulates and carnivores contracted their ranges or were locally extirpated in human-dominated landscapes. In a response to habitat loss and connectivity, large carnivores that require expansive home ranges can be extirpated, whereas mesopredators may experience less predation and competition (Crooks 2002). This "mesopredator release" (*sensu* Soulé et al. 1988) can result in increased mesopredator populations and has been documented across many urban mammal communities (Crooks 2002, Prugh et al. 2008, Ritchie and Johnson 2009). Crooks and Soulé (1999) found that larger forest fragments supported coyotes (*Canis latrans*) and their abundance was a strong predictor of smaller mesopredator abundance. This negative effect of top predator loss was realized when the absence of coyotes in smaller forest fragments resulted in increased mesopredator abundance, increased predation of scrub-breeding birds, and decreased bird diversity as compared to forests that supported coyotes (Crooks and Soulé 1999).

Studies also indicate urbanization can lead to changes in the meso- and large mammal community, shifting it towards more tolerant native and non-native species (Bowers and Breland 1996, DeVault et al. 2011, George and Crooks 2006, Reeves et al. 2005). **S**pecies commonly found in these urban habitats include groundhog (*Marmota monax*), white-tailed deer (*Odocoileus virginianus*), feral or domestic cat (*Felis catus*), striped skunk (*Mephitis mephitis*), Virginia opossum (*Didelphis virginianus*), northern raccoon (*Procyon lotor*), and red fox (*Vulpes vulpes*) (Reeves et al. 2005, Schiller and Horn 1997, Sinclair et al. 2005). Species such as elk (*Cervus elaphus*), river otter (*Lontra canadensis*), gray wolf (*Canis lupus*), and bear species (*Ursus* spp.) that require large continuous habitats (i.e. forests) and sufficient resources are mostly absent from urbanized landscapes (Laliberte and Ripple 2004). As a result, the urban communities are often composed of generalist species that are tolerant of human disturbances or well adapted to altered landscapes (Devictor et al. 2008, Olden and Poff 2004, Spear and Chown 2008, White and Kerr 2007**).** McKinney and Lockwood (1999) described the loss of specialists, expansion of generalists, and the general homogenization of regional faunas.

Urbanization also leads to the destruction of natural wetlands and an increase in impervious surfaces, resulting in changes in hydrology and water quality (Costanza et al. 1997, Faulkner 2004, Gibbs 2000). To limit some of the negative impacts of urbanization and improve overall watershed health, Baltimore County, Maryland implements “Best Management Practices” (Baish and Caliri 2009). One management practice is the

construction of stormwater ponds (hereafter SWPs), which mitigate flash floods and collect water runoff, thereby allowing particulates and pollutants to settle out before reaching the streams and eventually the Chesapeake Bay (Colosimo and Wilcock 2007, DEPS 2011). Typically, SWPs are elliptical or round depressions that vary in hydrology, area, and vegetation. For example, the SWP basins can be flooded for days, a given season, or year round. Vegetation in some SWPs can be frequently maintained to resemble mowed lawns, whereas less frequent maintenance regimes allow plant succession to resemble natural wetlands or forest patches.

With the disappearance of natural habitats, SWPs could function as supplemental habitats for mammal species. One might expect that more generalist species that are tolerant of urbanization (McKinney and Lockwood 1999) could also utilize SWPs. However, some of the constructed wetlands and forest patches of SWPs could support specialists such as the common muskrat (*Ondatra zibthicus*), American beaver (*Castor canadensis*), or weasel species (*Mustela* spp.).

To improve our understanding of how SWPs might serve as wildlife habitat, I quantified the meso- and large mammal community of SWPs using indirect monitoring techniques in a presence-absence study. In addition, I tested the influence of the surrounding landscape and various SWP characteristics on site occupancy. Specifically, I hypothesized that increased forest cover in the surrounding landscape would positively influence occupancy as a result of patch connectivity (Fahrig and Merriam 1985, Henein and Merriam 1990, Prugh et al. 2008). One rationale behind this hypothesis was that

mammals could use forested corridors to access multiple SWPs and their various habitat types to access resources. Because studies demonstrated an impact of patch size on population demography (Crooks 2002, Laliberte and Ripple 2004), I tested the effect of SWP area on occupancy. Furthermore, temporary flood events may impact mammals by compromising burrows, reducing cover, and submerging food resources (Ahlers et al. 2010, Reichman and Smith 1990). Finally, because vegetation heterogeneity and complexity are known to influence the demography of mammals (Bull et al. 2001, Catling et al. 2001, Rea 2003), I quantified their effects on occupancy.

Methods

Study Area and Site Selection

All SWPs included in this study were located 24 km northwest of Baltimore City in Owings Mills, Maryland (U.S.A.) at 39°25’19.52” N 76°46’52.59” E (Figure 1). The 54 km^{2} suburb was designated for commercial, industrial, and residential development (MDOT 2012). As requested by Baltimore County’s “Best Management Practices” (BMPs), these large-scale landscape alterations required the construction of SWPs to protect stream quality in the Red Run watershed and Chesapeake Bay. In 2007, Gallagher (2009) randomly selected 69 out of 170 available SWPs for an environmental impact study. He also quantified the hydrology, area, and percent forest cover in the surrounding landscape of the SWPs (Chapter II). Hydrology classifications included 1) intermittently exposed (IE) land with a permanent water body, 2) seasonally flooded (SF) basins that

filled in wet months and dried through summer or 3) temporarily flooded (TF) basins that filled and drained after rain events (Cowardin et al. 1979). The percent forest cover was measured at a 100 m buffer around the perimeter of each SWP.

In this study, I utilized Gallagher's (2009) hydrology and percent forest cover (above and below 40%) classifications to randomly select a subset of 30 SWPs from the original 69 (for location of the SWPs see Figure 2 in Chapter II). The area of these SWPs ranged from 597 to 20,397 m^2 (for individual SWP characteristics see Table 4 in Chapter II). A 1 to 25 m elevated berm encircled the SWPs, which formed the perimeter and grassy containment wall surrounding the settling basin. Common SWP structures included a concrete inflow culvert, emergency spillway, and fence. The vegetation in 11 of these SWPs was maintained by planting, seasonal mowing, and periodic tree removal. The remaining 19 SWPs resembled forests or wetlands and were dominated by forbs (Euphorbiaceae), cattails (*Typha* spp.), grasses (*Poaceae*, *Cyperaceae*, and *Juncaceae* spp.), mile-a-minute weed (*Persicaria perfoliata*), blackberry (*Rubus* spp.), black locusts (*Robinia pseudoacacia*), maples (*Acer* spp.), oaks (*Quercus* spp.), and willows (*Salix* spp.).

Vegetation Heterogeneity and Complexity Survey

In collaboration with Fitzpatrick (2011), I completed vegetation heterogeneity and complexity surveys during the summer of 2009 and 2010. Perpendicular to the longest length of the SWP, we placed transects 10 m apart and recorded the distances of each vegetation cover type. Cover types included forbs, grasses, and shrubs. We totaled the

distances per cover type and divided by the total transect lengths to calculate percent cover for the SWP heterogeneity. Starting at 0 m and continuing at 10 m intervals along the same transects, we recorded the vertical complexity as the number of times vegetation came into contact with each 0.25 m section of a 2 m PVC pole (Rotenberry and Wiens 1980). From these data, we calculated an average of all the sections as vegetation density for the entire SWP.

Mammal Presence-absence Survey

Because of differences in body size (eastern gray squirrel, *Sciurus carolinensis*, to white-tailed deer), nocturnal behavior, and difficulty in sampling elusive mammals, I used multiple detection methods to quantify the meso- and large mammal community utilizing SWPs and make inferences about the proportion of sites occupied by species (Gompper et al. 2006, Nichols et al. 2008, O'Connell et al. 2006, Wilson et al. 1996). I conducted a presence-absence study from March 2010 to January 2011 and from March 2011 to January 2012 in SWPs located in Owings Mills, Maryland, USA. I surveyed all 30 SWPs during a seven week primary sampling period for five occasions in the first year and for logistic reasons only four in the second, with the exception of five second year occasions for four SWPs (E7-5, H8-2, I8-1, and I8-2).

In the first three occasions of the first sampling year, I set a trap array of a single Moultrie Feeders Game Spy D-50 Flash Digital Trail Camera (EBSCO Industries, Inc. © 2012, Birmingham, AL) and three 23 x 23 x 81 cm plywood cubby boxes in 25 SWPs (Zielinski 1995). One SWP (F7-1) had this array for only the first sampling period before

I removed the camera on subsequent surveys due to possibilities of theft. I sampled 4 SWPs (F6-3, F8-3, G6-5, and G6-10) with only the cubby boxes for the same reason. During the remaining sampling periods, 25 SWPs had only a single camera trap and five SWPs (F6-3, F7-1, F8-3, G6-5, and G6-10) had only the three cubby boxes due to possibilities of theft.

I centered camera traps in the dry area of the SWP basin along established animal trails and baited them with venison and a scent lure (Cronk's Lures, Wiscasset, ME). I arranged the three cubby boxes in the dry area surrounding the basin in a triangle formation to maximize the distance among the track plates within them. Track plates consisted of carbon treated metal flashing fitted with contact paper adhesive side up (Belant 2003). I baited the track plates with dry cat food, fish sauce, and a scent lure and operated both methods for a week at a time before collecting contact paper for track analysis and transporting the equipment to another SWP.

Detection Probability and Occupancy Analyses

I analyzed photographs and tracks for the presence of mammals and identified individual detections to species (Elbroch 2003, Halfpenny 1986, Murie and Elbroch 2005, Rezendes 1999). To ensure sufficient sample size and test models for species with similar ecological requirements, I combined the detected individuals into squirrel, herbivore, and carnivore groups. For analysis, I used gray squirrel daily detections because of their limited detections throughout the study, but pooled herbivore and carnivore daily detections into primary sampling periods. I input these encounter histories

into Program Presence version 4.0 (Hines 2006) to test the influence of landscape and SWP characteristics on occupancy.

I considered imperfect detection probability (p) in the maximum-likelihood estimation of site occupancy (ψ) because of the possibility that SWPs were occupied by a detected species (ψ x p), occupied by an undetected species (ψ x [1- p]), or not occupied ($1 - \psi$) (MacKenzie et al. 2002). I used a goodness-of-fit test on a parametric bootstrap analysis (with 1,000 iterations) to test if the most parameterized model would fit the data (MacKenzie and Bailey 2004). Specifically, the most parameterized model ψ(species) p(method + time) showed an influence of detection method and sampling period on detection probability and species on occupancy. I calculated an estimate for overdispersion ($\hat{c}$) to make adjustments to the model $\hat{c}$ if overdispersion was present and $\hat{c}$ was greater than 1 (MacKenzie et al. 2006).

I built a priori single-season, generalized linear models for each group based on the biologically meaningful combinations of covariates. For example, each group does not contain multiple species. Also, occupancy cannot be logically influenced by detection method. These models tested the influence of detection method, time of sampling period, and landscape and SWP characteristics on detection probability and occupancy. Similar to the Cormack-Jolly-Seber (CJS) model approach for the capture-mark-recapture (CMR) study, I first developed a null model of detection probability and occupancy being constant through time (.) to create an over-simplified representation of the data for model comparison. Next, I held occupancy constant and altered the covariate of influence on

detection probability before developing all biologically meaningful combinations (Table 9). I then investigated the influence of landscape and SWP characteristics (Table 10), thereby developing models that more effectively represented the variation in the data.

Program Presence uses an information-theoretic approach to rank the models according to Akiake's information criterion (AIC). This AIC value measures the expected Kullback-Leibler distance between each model and the data, as well as the amount of information lost with the model's approximation (Anderson and Burnham 1999, Burnham and Anderson 2002, Johnson and Omland 2004). I selected the most parsimonious model to minimize the change in corrected AIC (ΔAIC_c) or change in corrected quasi- AIC ($\Delta QAIC_c$) if there was overdispersion (Anderson et al. 1994). This asymptotically corrected for small sample size, with a lessening effect as sample size became more appropriate for the number of the parameters in the model (Hurvich and Tsai 1989). I considered alternative models within two units of the top model for inference because of substantial support (Burnham and Anderson 2002). These models supplied maximum-likelihood parameter estimates of both detection probability and occupancy from slope estimates of the single-season logistic regressions.

Results

Vegetation Heterogeneity and Complexity of Stormwater Ponds

The SWP vegetation was highly variable in heterogeneity and complexity. Five SWPs (E4-2, F7-1, F8-3, F9-1, and I8-1) lacked a developed understory and more closely

resembled lawns with a homogenous cover type. The remaining 25 SWPs had more complex vegetation, including large stands of the non-native invasive common reed (*Phragmites australis*) in three SWPs (E7-5, G6-14, and H8-2). Maintenance crews were worked regularly in 11 ponds. For example they seeded the steep containment walls and covered them with a fibrous mat to reduce erosion in E7-2 and G6-1. Crews cut down young trees in E4-1 and F4-1, cleared all trees and mowed G6-10 and I8-1, and removed vegetation off the berm in E9-2. Four ponds (E4-2, F7-1, F8-3, and F9-1) were cleared of shrubs and saplings then maintained by routine mowing through the growing season.

Mammal Community of Stormwater Ponds

Ten mammal species were detected during the two year presence-absence study (Table 11). The most diverse SWPs were E3-1 and H5-1 with eight species (gray squirrel, eastern cottontail (*Sylvilagus floridanus*), groundhog, white-tailed deer, domestic cat, Virginia opossum, northern raccoon, and red fox) and the least diverse was F6-3 with 2 species (gray squirrel and northern raccoon).

Groundhogs and white-tailed deer were frequently detected in the majority of SWPs in over eight sampling periods. Over half of SWPs contained discarded or active groundhog burrows and I often saw foraging individuals use them for cover. I observed white-tailed deer inside SWPs on numerous occasions and encountered a herd of over eight individuals in D7-1, E7-5, and F7-5. I found deer beds and accidentally flushed fawns out of the high vegetation in E7-5 and F7-5. Both herbivores seem to be absent from small SWPs with intact fences and short, steep slopes. Conversely, the eastern

chipmunk (*Tamias striatus*) was detected only during the first sampling period in a single SWP, even though 15 individuals were live captured in subsequent periods during a CMR study (Chapter II). Also captured in the CMR study was a subadult eastern cottontail. The eastern chipmunk, eastern cottontail, and white-tailed deer were detected only by the camera traps, whereas gray squirrel and groundhog were detected by both methods. Juvenile and adult muskrats were observed in G6-14, but were not detected by either method.

Two carnivores were present in nearly every sampling period and were the most frequently detected species in the entire study. Northern raccoons were detected in 29 out of 30 SWPs and red foxes were encountered in 26 SWPs. Interestingly, these two mesopredator species occurred sympatrically in 18 SWPs. I observed adult red foxes, active burrows, or discarded remains of prey in more than five SWPs. In E9-4, two kits interacted with each other in front of their burrow. Virginia opossums were detected less frequently in only 12 SWPs in six sampling periods. They were not detected during the 5th, 9th, and 10th sampling periods, which coincide with the winters of 2010 and 2011. I did capture a juvenile Virginia opossum in SWP H5-1 during the CMR study (Chapter II). Cats were detected in the majority of sampling periods and in over half of the SWPs. Although it was not possible to distinguish between feral and domesticated individuals, I observed cats hunting or fighting in four SWPs. A single domestic dog (*Canis lupus familiaris*) was detected by camera trap in one SWP, which was in close proximity to residential developments.

Detection Probability and Occupancy of Gray Squirrels

For the gray squirrel analysis, the most parameterized model ψ(.) *p*(method + time) did not fit the data for a single-season occupancy model (χ^2 = 3820494.54, *P* = 0.011). An estimate for overdispersion ($\hat{c}$ = 12.02) was calculated and the model $\hat{c}$ was adjusted accordingly. Therefore, models were ranked according to $QAIC_c$ values (Table 12) and the model ψ(.) *p*(time) was the best approximating model for testing covariates. Out of the 15 total candidate models, I again identified ψ(.) *p*(time) as the only top model, accounting for 12% of the total variation in the data (Table 13). Detection probability fluctuated greatly through time and averaged 0.01 ± 0.00 SE (Table 14), although the range from 0.00 to 0.07 was not large. The naïve occupancy estimate of 0.29 did not take into consideration imperfect detections and was therefore much lower than the real occupancy estimate (1.00 ± 0.00 SE), which remained constant.

Detection Probability and Occupancy of Herbivores

The most parameterized model ψ(species) *p*(method + time) for eastern cottontail, groundhog, and white-tailed deer detections did not fit the data (χ^2 = 8990.83, *P* = 0.01). Because overdispersion ($\hat{c}$) was 5.87, the adjusted model $\hat{c}$ was utilized. Therefore, models were ranked according to $QAIC_c$ values instead of AIC_c values and the top four models (Table 15) were used to test the effects of covariates (Burnham and Anderson 2002). I then identified 48 top models showing equal influence of multiple landscape and SWP characteristics on detection and occupancy (Table 16). These models suggested that every covariate was equally influential on detection probability and occupancy and

accounted for less than 3% of the variation in the data with each model. Average detection probability (0.26 ± 0.00 SE) and naïve occupancy (0.49) estimates were much higher than that of gray squirrels. Considering imperfect detection in parameter estimates, the average eastern cottontail occupancy (0.35 ± 0.01 SE) was the lowest, followed by groundhog (0.50 ± 0.00 SE) and white-tailed deer (0.86 ± 0.00 SE).

Detection Probability and Occupancy of Carnivores

The most parameterized model ψ(species) *p*(method + time) for cat, Virginia opossum, northern raccoon, and red fox detections did fit the data for a single-season occupancy model (χ^2 = 1236.67, P = 0.335). Because there was no overdispersion ($\hat{c}$ = 0.92), no adjustment to the model $\hat{c}$ was needed and AIC_c values were used for model selection (Burnham and Anderson 2002). The model ψ(species) *p*(method) was selected to test the influence of the measured covariates on estimated parameters (Table 17). After modeling the covariates, I identified ψ(species + %forest) p(method) as the best approximating model, explaining 6% of the variation in the carnivore data (Table 18). Parameter estimates (Table 19) for detection probabilities averaged 0.33 ± 0.01 SE and were higher for track plates than for camera traps. Naïve occupancy was 0.50 for all species, but was species specific when considering imperfect detection. Virginia opossum (0.27 ± 0.02 SE) had the lowest occupancy, followed by cat (0.41 ± 0.02 SE), red fox (0.64 ± 0.02 SE), and northern raccoon (0.98 ± 0.01 SE). Occupancy for all species declined with increased forest cover in the surrounding landscape, cats being most resilient to the affect (Figure 11).

Discussion

Mammal Community of Stormwater Ponds

The need for multiple detection methods in quantifying the meso- and large mammal community of SWPs was apparent in this study. For example, eastern cottontails and white-tailed deer were not detected by the cubby boxes, but by camera traps. Similarly, O'Connell et al. (2006) demonstrated the influence of different detection methods on species-specific detection probabilities for terrestrial mammals (see also Bailey et al. 2004, MacKenzie et al. 2002, Nichols et al. 2008, Weller 2008). In their study, river otter, coyote, and white-tailed deer were not detected by cubby boxes, but were frequently encountered with camera traps.

Over the course of the two year study, I detected 10 different meso- and large mammal species utilizing SWPs. In fact, mammals occurred in every SWP, regardless of hydrology, area, vegetation heterogeneity, and complexity. Even the most isolated SWPs (E7-4, F4-1, F6-3, G6-5, and G6-10) surrounded by 0% forest cover attracted multiple mammal species typical to the mid-Atlantic region (Fuller and DeStefano 2003, Kelly and Holub 2008, Reeves et al. 2005). Northern raccoons and red foxes utilize a variety of habitats including field, forest, and urban habitats (Beasley et al. 2011, Chamberlain et al. 2002, DeVault et al. 2011). These generalists can take advantage of a wide range of resources, including food discarded by humans (Dell'Arte et al. 2007, Larivière and Pasitschniak-Arts 1996, Lotze and Anderson 1979). For example, Prange et al. (2004) demonstrated that increased artificial food resources (restaurant, park, and residential

trash receptacles) resulted in decreased home range for urban northern raccoons as compared to suburban and rural populations. Even large herbivores, such as the white-tailed deer, are common to habitat remnants and urban parks that provide sufficient woody cover (DeStefano and DeGraaf 2003, Grund et al. 2002, Rondeau and Conrad 2003). These species are well adapted to the urban matrix, thus it is not surprising to encounter them in SWPs.

One obvious reason why mammals may utilize SWPs is the availability of food resources. For example, oak and walnut trees inside the SWPs produced fruits, which were consumed by several mammal species. Inside SWPs, I encountered northern raccoon fecal samples containing blackberry and wineberry seeds. I frequently observed eastern cottontails, groundhogs, and white-tailed deer herds foraging on SWP vegetation. There was also indirect evidence suggesting that SWPs could provide carnivores with prey items. For instance, I often observed northern raccoon prints along the standing water of SWP basins that contained amphibians. Stover (2012) suggested that northern raccoons might be major predators of red-winged blackbird (*Agelaius phoeniceus*) eggs and nestlings in SWPs.

Another reason for the use of SWPs could be the availability of complex vegetation. For example, fawns and eastern cottontails frequently hid in the dense vegetation. Similarly, Althoff et al. (1997) highlight the importance of dense vegetation cover for eastern cottontails to conserve energy, reduce heat loss, and avoid predators.

Stormwater ponds with sufficient vegetation complexity may function as refuge within a simplified urban landscape.

Several mammal species also utilized SWPs for natal sites and rearing young. I encountered kits, fawns, and juvenile muskrats, as well as red fox dens, groundhog burrows, or muskrat bank burrows in at least 22 SWPs. Groundhog burrows are of particular importance because it is suggested that multiple species, including eastern cottontails, striped skunks, Virginia opossums, northern raccoons, and red foxes may use a portion of their burrows during winter (Hossler et al. 1994, Schmeltz and Whitaker 1977, Swihart and Picone 1995).

Besides the intended function for improving watershed health, SWPs can also function as habitat for meso- and large mammals. Depending on the surrounding matrix, mammals could incorporate a single or multiple SWPs into their home range. Stormwater ponds that provide sufficient food resources, vegetation cover, and nesting sites can play a vital role in sustaining urban mammal species.

Detection Probability and Occupancy of Mammal Groups

Occupancy models for the squirrel, herbivore, and carnivore groups had varied success. Contrary to the success of modeling carnivore occupancy, the squirrel and herbivore groups had high estimates of overdispersion, most likely due to having small variations in the data from the most parameterized models and too many parameters in the model without enough detection data to support them. The gray squirrels had very low detection probabilities throughout the 70 days of sampling, including 45 days with no

detections. Because daily detections were not collapsed into sampling periods, gray squirrels went undetected on over half of the days. This resulted in a very simple top model where time had a substantial effect on detection probability and occupancy remained constant. Similarly, eastern cottontails and white-tailed deer were detected only by camera traps, which resulted in the sparse encounter histories for the herbivore models. A possible lack of variation in where the herbivores were detected resulted in the inability to identify a few covariates that impact detection probability and occupancy. This resulted in every landscape and SWP characteristic having an equal influence on the parameter estimates. Considering the overdispersion in the squirrel and herbivore groups, inferences are less meaningful for ecological and conservation applications. Therefore, future studies are needed to better differentiate the effects of landscape and SWP characteristics on squirrel and herbivore occupancy.

Carnivores, on the other hand, were detected by both camera traps and track plates, which ultimately showed different efficiencies in detection probability. Track plates had a higher probability of detecting carnivores than the camera traps. In general, detection probability can also vary by track plate design (Loukmas et al. 2002), camera models, and settings (Kelly and Holub 2008, Swann et al. 2004).

Out of the seven measured landscape and SWP characteristics, only percent forest cover in the surrounding landscape influenced carnivore occupancy. As vegetation inside the SWPs would likely provide numerous resources for carnivores, the vegetation in the surrounding landscape is more influential in providing access to multiple habitat patches.

But, contrary to my hypothesis, carnivore occupancy decreased with increased forest cover in the surrounding landscape. Fahrig (2003) reviewed the literature on the effects of the landscape on biodiversity and found inconsistent trends that lend support to the surprising negative effects of increased forest. The mechanism behind this response could be determined by the quality of the forested riparian buffer decreasing carnivore reliance on SWP cover, shelter, and food resources. In a meta-analysis study, Prugh et al. (2008) concluded that cover type in the surrounding landscape was more influential than patch size or isolation on wildlife. Baltimore County protects forested riparian buffers within the urban landscape to improve watershed health (Arnold and Gibbons 1996, Lowrance et al. 1997, Schueler 1994). Perhaps with low forest cover, the importance of SWPs is magnified as they act like stepping stone habitats that supplement carnivores and connect individuals to larger forest fragments.

Carnivore occupancy estimates were also influenced by species. Interestingly, average occupancy for urban Virginia opossum and northern raccoon closely resembled, but were slightly higher than estimates from a similar study in a continuous forest of a coastal national park (O'Connell et al. 2006). These findings confirm that these species are well adapted to the urban landscape (Lotze and Anderson 1979, McManus 1974). Estimates for the red fox were drastically higher in this study and could be attributed to the absence of a larger competitor, the coyote (O'Connell et al. 2006). The high occupancy of Virginia opossums, northern raccoons, and red foxes in SWPs could have resulted from mesopredator release (Crooks 2002, Prugh et al. 2008, Soulé et al. 1988).

Eagan et al. (2011) studied the effects of northern raccoon abundance on the white-footed mouse density in a human-altered environment. As compared to the control, white-footed mouse density significantly increased in response to the reduced northern raccoon abundance of the manipulated populations. Meserve et al. (1993) and Reid et al. (1995) also emphasized the connection between habitat fragmentation, increased mesopredator abundance, and a limited prey population.

Future studies could focus on the function of SWPs in the abundance and reproductive success of mesopredators for further insights to the mesopredator release hypothesis for which little empirical evidence is available (Gehrt and Clark 2003, Gehrt and Prange 2007). Current management strategies implement the demolition of mammal burrows, which could negatively impact the survival of juvenile and adults of multiple species, including mesopredators. However, the notion that these structures compromise the integrity of the SWP should be tested, considering location of burrow, SWP hydrology, and projections of flood severity.

SWPs are a “Best Management Practice” for improving the Chesapeake Bay watershed health. Results from this study clearly demonstrate that SWPs also function as habitats for meso- and large mammals. Previous studies also indicate that SWPs support small mammal (Chapter II), amphibian (Brand and Snodgrass 2010, Simon et al. 2009), and bird communities (Fitzpatrick 2011, Stover 2012). Stormwater ponds could function as an ecological refuge for numerous vertebrate species in a highly disturbed urban setting. Therefore, strategies that further promote the utilization of SWPs and improve

their habitat quality should be considered during the planning and management process. For example, percent forest cover in the surrounding landscape, connectivity among SWPs, proximity to natural riparian buffers, planting native aquatic and terrestrial plant species, and promoting plant succession. With proper management, SWPs could attract new species, increase species diversity, and become constructed habitats with a high ecological value in the urban landscape.

Table 9. Candidate single-season models for estimating detection probability (*p*) and occupancy (ψ) of meso- and large mammals in stormwater ponds of Owings Mills, Maryland, USA. In Program Presence, estimates were modeled as constant (.) or varied by detection method, sampling period (time), or species, if applicable. The assumption of model fit was tested using model three for the gray squirrel group and model seven for the herbivores and carnivores because these groups contained multiple species.

Number	Model
1	ψ(.) *p*(.)
2	ψ(.) *p*(method)
3	ψ(.) *p*(method + time)
4	ψ(.) *p*(time)
5	ψ(species) *p*(.)
6	ψ(species) *p*(method)
7	ψ(species) *p*(method + time)
8	ψ(species) *p*(time)

Table 10. Candidate single-season models for estimating detection probability (*p*) and occupancy (ψ) of meso- and large mammals in stormwater ponds (SWPs) of Owings Mills, Maryland, USA. In Program Presence, estimates were modeled to vary by hydrology, area, the proportion of forest cover in the surrounding landscape, the proportion of forbs, grass, or shrub cover, and vegetation density inside the SWP.

Number	Model
1	ψ *p*(%forbs)
2	ψ *p*(%grass)
3	ψ *p*(%shrub)
4	ψ *p*(veg.density)
5	ψ(hydrology) *p*
6	ψ(area) *p*
7	ψ(%forest) *p*
8	ψ(%forbs) *p*
9	ψ(%grass) *p*
10	ψ(%shrub) *p*
11	ψ(veg.density) *p*

Table 11. Presence-absence summary of urban mammal species in 30 stormwater ponds (SWPs) located in Owings Mills, Maryland, USA. This survey utilized camera traps and track plates over 10 sampling periods from March 2010 to January 2012. Because of low detections, domestic dog and eastern chipmunk were not included in a group for modeling occupancy.

Group	Species	Sampling Periods	Camera Detections	Track Plate Detections	SWPs
-	Domestic dog (*Canis lupus familiarus)*	1	1	0	1
-	Eastern chipmunk (*Tamias striatus*)	1	1	0	1
Squirrel	Eastern gray squirrel (*Sciurus carolinensis*)	9	5	8	13
Herbivore	Eastern cottontail (*Sylvilagus floridanus*)	8	8	0	9
	Groundhog (*Marmota monax*)	8	7	5	18
	White-tailed deer (*Odocoileus virginianus*)	10	10	0	22
Carnivore	Domestic cat (*Felis catus*)	9	7	7	16
	Northern raccoon (*Procyon lotor*)	9	9	9	29
	Red fox (*Vulpes vulpes*)	10	10	4	26
	Virginia opossum (*Didelphis virginiana*)	6	1	6	12

Table 12. Ranked single-season models for estimating eastern gray squirrel detection probability (*p*) and occupancy (ψ) in stormwater ponds of Owings Mills, Maryland, USA. Estimates were modeled as constant (.) or varied by detection method, or sampling period (time). *k* = number of parameters in the model; $\Delta QAIC_c$ = change in the $QAIC_c$ value; w_i = model weight; Var = variance explained.

Model	*k*	$QAIC_c$	$\Delta QAIC_c$	w_i	Model Likelihood	Var
ψ(.) *p*(time)	71	-473.95	0.00	1.00	1.00	0.12
ψ(.) *p*(method + time)	72	-452.91	21.04	0.00	0.00	0.18
ψ(.) *p*(.)	2	30.36	504.31	0.00	0.00	0.00
ψ(.) *p*(method)	3	33.90	507.85	0.00	0.00	-0.05

Table 13. All ranked single-season models using landscape and stormwater pond (SWP) characteristics to estimate eastern gray squirrel detection probability (*p*) and occupancy (ψ) in Owings Mills, Maryland, USA. In Program Presence, estimates were modeled as constant (.) or varied by detection method, sampling period (time), hydrology, area, the proportion of forest cover in the surrounding landscape, the proportion of forbs, grass, or shrub cover, and vegetation density inside the SWP. *k* = number of parameters in the model; ΔAIC_c = change in the AIC_c value; w_i = model weight; Var = variance explained.

Model	*k*	$QAIC_c$	$\Delta QAIC_c$	w_i	Model Likelihood	Var
ψ(.) *p*(time)	71	-649.32	0.00	1.00	1.00	0.12
ψ(%forbs) *p*(time)	72	-628.18	21.14	0.00	0.00	0.12
ψ(%grass) *p*(time)	72	-628.18	21.14	0.00	0.00	0.12
ψ(%shrub) *p*(time)	72	-628.18	21.14	0.00	0.00	0.12
ψ(.) *p*(time + %grass)	72	-628.18	21.14	0.00	0.00	0.12
ψ(%forest) *p*(time)	72	-628.18	21.14	0.00	0.00	0.12
ψ(.) *p*(time + %shrub)	72	-628.18	21.14	0.00	0.00	0.12
ψ(.) *p*(time + veg.density)	72	-628.18	21.14	0.00	0.00	0.12
ψ(.) *p*(method + time)	72	-628.08	21.23	0.00	0.00	0.18
ψ(veg.density) *p*(time)	72	-628.08	21.23	0.00	0.00	0.18
ψ(area) *p*(time)	72	-628.07	21.24	0.00	0.00	0.19
ψ(.) *p*(time + %forbs)	72	-628.07	21.24	0.00	0.00	0.19
ψ(hydrology) *p*(time)	73	-609.61	39.70	0.00	0.00	0.12
ψ(.) *p*(.)	2	0.79	650.11	0.00	0.00	0.00
ψ(.) *p*(method)	3	1.10	650.42	0.00	0.00	-0.05

Table 14. Parameter estimates for eastern gray squirrel detection probability (*p*) and occupancy (ψ) in 30 stormwater ponds located in Owings Mills, Maryland, USA. These estimates were modeled as constant (.) or to vary by sampling period (time). SE = standard error; 95% CI = 95% confidence interval.

Model	Parameter	Estimate	SE	95% CI
ψ(.) *p*(time)	ψ	1.00	0.00	1.00 - 1.00
	p_1	0.02	0.02	0.00 - 0.12
	p_2	0.02	0.02	0.00 - 0.12
	p_3	0.00	0.00	0.00 - 1.00
	p_4	0.00	0.00	0.00 - 1.00
	p_5	0.00	0.00	0.00 - 1.00
	p_6	0.00	0.00	0.00 - 1.00
	p_7	0.00	0.00	0.00 - 1.00
	p_8	0.04	0.03	0.01 - 0.13
	p_9	0.02	0.02	0.00 - 0.12
	p_{10}	0.02	0.02	0.00 - 0.12
	p_{11}	0.04	0.03	0.01 - 0.13
	p_{12}	0.00	0.00	0.00 - 1.00
	p_{13}	0.00	0.00	0.00 - 1.00
	p_{14}	0.00	0.00	0.00 - 1.00
	p_{15}	0.02	0.02	0.00 - 0.12
	p_{16}	0.02	0.02	0.00 - 0.12
	p_{17}	0.04	0.03	0.01 - 0.13
	p_{18}	0.04	0.03	0.01 - 0.13
	p_{19}	0.02	0.02	0.00 - 0.12
	p_{20}	0.02	0.02	0.00 - 0.12
	p_{21}	0.04	0.03	0.01 - 0.13
	p_{22}	0.03	0.03	0.00 - 0.20
	p_{23}	0.03	0.03	0.00 - 0.20

p_{24}	0.03	0.03	0.00 - 0.20
p_{25}	0.03	0.03	0.00 - 0.20
p_{26}	0.00	0.00	0.00 - 0.00
p_{27}	0.00	0.00	0.00 - 0.00
p_{28}	0.00	0.00	0.00 - 0.00
p_{29}	0.03	0.03	0.00 - 0.20
p_{30}	0.00	0.00	0.00 - 0.00
p_{31}	0.00	0.00	0.00 - 0.00
p_{32}	0.00	0.00	0.00 - 0.00
p_{33}	0.00	0.00	0.00 - 0.00
p_{34}	0.00	0.00	0.00 - 0.00
p_{35}	0.00	0.00	0.00 - 0.00
p_{36}	0.03	0.03	0.00 - 0.20
p_{37}	0.00	0.00	0.00 - 0.00
p_{38}	0.00	0.00	0.00 - 0.00
p_{39}	0.00	0.00	0.00 - 0.00
p_{40}	0.00	0.00	0.00 - 0.00
p_{41}	0.00	0.00	0.00 - 0.00
p_{42}	0.00	0.00	0.00 - 0.00
p_{43}	0.07	0.05	0.02 - 0.23
p_{44}	0.00	0.00	0.00 - 0.00
p_{45}	0.00	0.00	0.00 - 0.00
p_{46}	0.00	0.00	0.00 - 0.00
p_{47}	0.00	0.00	0.00 - 0.00
p_{48}	0.00	0.00	0.00 - 0.00
p_{49}	0.00	0.00	0.00 - 0.00
p_{50}	0.03	0.03	0.00 - 0.20
p_{51}	0.00	0.00	0.00 - 0.00
p_{52}	0.00	0.00	0.00 - 0.00
p_{53}	0.00	0.00	0.00 - 0.00

p_{54}	0.00	0.00	0.00 - 0.00
p_{55}	0.00	0.00	0.00 - 0.00
p_{56}	0.00	0.00	0.00 - 0.00
p_{57}	0.03	0.03	0.00 - 0.20
p_{58}	0.03	0.03	0.00 - 0.20
p_{59}	0.03	0.03	0.00 - 0.20
p_{60}	0.03	0.03	0.00 - 0.20
p_{61}	0.00	0.00	0.00 - 0.00
p_{62}	0.00	0.00	0.00 - 0.00
p_{63}	0.00	0.00	0.00 - 0.00
p_{64}	0.00	0.00	0.00 - 0.00
p_{65}	0.00	0.00	0.00 - 0.00
p_{66}	0.00	0.00	0.00 - 0.00
p_{67}	0.00	0.00	0.00 - 0.00
p_{68}	0.00	0.00	0.00 - 0.00
p_{69}	0.00	0.00	0.00 - 0.00
p_{70}	0.00	0.00	0.00 - 0.00

Table 15. Ranked single-season models for estimating herbivore detection probability (*p*) and occupancy (ψ) in stormwater ponds of Owings Mills, Maryland, USA. Estimates were modeled as constant (.) or varied by detection method, sampling period (time), or species. *k* = number of parameters in the model; $\Delta QAIC_c$ = change in the $QAIC_c$ value; w_i = model weight; Var = variance explained.

Model	*k*	$QAIC_c$	$\Delta QAIC_c$	w_i	Model Likelihood	Var
ψ(.) *p*(.)	2	1.22	0.00	0.24	1.00	0.00
ψ(.) *p*(method)	3	1.36	0.14	0.22	0.93	0.00
ψ(species) *p*(.)	4	1.52	0.31	0.20	0.86	0.02
ψ(species) *p*(method)	5	1.75	0.54	0.18	0.77	0.03
ψ(.) *p*(time)	11	4.11	2.89	0.06	0.24	0.03
ψ(.) *p*(method + time)	12	4.67	3.45	0.04	0.18	0.04
ψ(species) *p*(time)	13	5.27	4.06	0.03	0.13	0.05
ψ(species) *p*(method + time)	14	5.94	4.73	0.02	0.09	0.06

Table 16. All ranked single-season models using landscape and stormwater pond (SWP) characteristics to estimate herbivore detection probability (*p*) and occupancy (ψ) in Owings Mills, Maryland, USA. In Program Presence, estimates were modeled as constant (.) or varied by detection method, sampling period (time), species, hydrology, area, the proportion of forest cover in the surrounding landscape, the proportion of forbs, grass, or shrub cover, and vegetation density inside the SWP. *k* = number of parameters in the model; ΔAIC_c = change in the AIC_c value; w_i = model weight; Var = variance explained.

Model	*k*	$QAIC_c$	$\Delta QAIC_c$	w_i	Model Likelihood	Var
ψ(.) *p*(.)	2	1.22	0.00	0.03	1.00	0.00
ψ(.) *p*(method)	3	1.36	0.14	0.02	0.93	0.00
ψ(.) *p*(%grass)	3	1.36	0.14	0.02	0.93	0.00
ψ(%forbs) *p*(.)	3	1.36	0.14	0.02	0.93	0.00
ψ(%forest) *p*(.)	3	1.36	0.14	0.02	0.93	0.00
ψ(%shrub) *p*(.)	3	1.36	0.14	0.02	0.93	0.00
ψ(.) *p*(%shrub)	3	1.36	0.15	0.02	0.93	0.00
ψ(.) *p*(%forbs)	3	1.36	0.15	0.02	0.93	0.00
ψ(veg.density) *p*(.)	3	1.36	0.15	0.02	0.93	0.00
ψ(%grass) *p*(.)	3	1.36	0.15	0.02	0.93	0.00
ψ(area) *p*(.)	3	1.36	0.15	0.02	0.93	0.00
ψ(.) *p*(veg.density)	3	1.36	0.15	0.02	0.93	0.00

ψ(species) *p*(.)	4	1.52	0.31	0.02	0.86	0.02
ψ(.) *p*(method + %grass)	4	1.54	0.32	0.02	0.85	0.01
ψ(%forbs) *p*(method)	4	1.54	0.33	0.02	0.85	0.01
ψ(%forest) *p*(method)	4	1.54	0.33	0.02	0.85	0.01
ψ(%shrub) *p*(method)	4	1.54	0.33	0.02	0.85	0.01
ψ(.) *p*(method + %shrub)	4	1.54	0.33	0.02	0.85	0.01
ψ(.) *p*(method + %forbs)	4	1.54	0.33	0.02	0.85	0.01
ψ(veg.density) *p*(method)	4	1.54	0.33	0.02	0.85	0.00
ψ(%grass) *p*(method)	4	1.54	0.33	0.02	0.85	0.00
ψ(.) *p*(method + veg.density)	4	1.54	0.33	0.02	0.85	0.00
ψ(area) *p*(method)	4	1.54	0.33	0.02	0.85	0.00
ψ(hydrology) *p*(.)	4	1.55	0.33	0.02	0.85	0.00
ψ(species) *p*(%grass)	5	1.75	0.54	0.02	0.77	0.03
ψ(species) *p*(method)	5	1.75	0.54	0.02	0.77	0.03
ψ(species + %forbs) *p*(.)	5	1.75	0.54	0.02	0.76	0.03
ψ(species + %forest) *p*(.)	5	1.75	0.54	0.02	0.76	0.03
ψ(species + %shrub) *p*(.)	5	1.75	0.54	0.02	0.76	0.03
ψ(species) *p*(%shrub)	5	1.75	0.54	0.02	0.76	0.02
ψ(species) *p*(%forbs)	5	1.75	0.54	0.02	0.76	0.02
ψ(species + %grass) *p*(.)	5	1.75	0.54	0.02	0.76	0.02
ψ(species + veg.density) *p*(.)	5	1.75	0.54	0.02	0.76	0.02
ψ(species + area) *p*(.)	5	1.75	0.54	0.02	0.76	0.02
ψ(species) *p*(veg.density)	5	1.75	0.54	0.02	0.76	0.02

ψ(hydrology) *p*(method)	5	1.77	0.56	0.02	0.76	0.01
ψ(species) *p*(method + %grass)	6	2.02	0.80	0.02	0.67	0.03
ψ(species + %forbs) *p*(method)	6	2.02	0.81	0.02	0.67	0.03
ψ(species + %forest) *p*(method)	6	2.02	0.81	0.02	0.67	0.03
ψ(species + %shrub) *p*(method)	6	2.02	0.81	0.02	0.67	0.03
ψ(species) *p*(method + %shrub)	6	2.02	0.81	0.02	0.67	0.03
ψ(species) *p*(method + %forbs)	6	2.02	0.81	0.02	0.67	0.03
ψ(species + veg.density) *p*(method)	6	2.03	0.81	0.02	0.67	0.03
ψ(species + %grass) *p*(method)	6	2.03	0.81	0.02	0.67	0.03
ψ(species) *p*(method + veg.density)	6	2.03	0.81	0.02	0.67	0.03
ψ(species + area) *p*(method)	6	2.03	0.81	0.02	0.67	0.03
ψ(species + hydrology) *p*(.)	6	2.03	0.81	0.02	0.67	0.02
ψ(species + hydrology) *p*(method)	7	2.34	1.13	0.01	0.57	0.03
ψ(.) *p*(time)	11	4.11	2.89	0.01	0.24	0.03
ψ(.) *p*(method + time)	12	4.67	3.45	0.00	0.18	0.04
ψ(species) *p*(time)	13	5.27	4.06	0.00	0.13	0.05
ψ(species) *p*(method + time)	14	5.94	4.73	0.00	0.09	0.06

Table 17. Ranked single-season models for carnivore detection probability (*p*) and occupancy (ψ) in stormwater ponds of Owings Mills, Maryland, USA. Estimates were modeled as constant (.) or varied by detection method, sampling period (time), or species. *k* = number of parameters in the model; ΔAIC_c = change in the AIC_c value; w_i = model weight; Var = variance explained.

Model	*k*	AIC_c	ΔAIC_c	w_i	Model Likelihood	Var
ψ(species) *p*(method)	6	1200.22	0.00	1.00	1.00	0.06
ψ(species) *p*(.)	5	1215.46	15.24	0.00	0.00	0.04
ψ(.) *p*(method)	3	1227.48	27.26	0.00	0.00	0.02
ψ(.) *p*(.)	2	1239.02	38.81	0.00	0.00	0.00
ψ(species) *p*(method + time)	15	1251.43	51.21	0.00	0.00	0.07
ψ(species) *p*(time)	14	1260.85	60.64	0.00	0.00	0.06
ψ(.) *p*(method + time)	12	1279.63	79.42	0.00	0.00	0.03
ψ(.) *p*(time)	11	1286.07	85.85	0.00	0.00	0.02

Table 18. All ranked single-season models using landscape and stormwater pond (SWP) characteristics to estimate carnivore detection probability (*p*) and occupancy (ψ) in Owings Mills, Maryland, USA. In Program Presence, estimates were modeled as constant (.) or varied by detection method, sampling period (time), species, hydrology, area, the proportion of forest cover in the surrounding landscape, the proportion of forbs, grass, or shrub cover, and vegetation density inside the SWP. *k* = number of parameters in the model; ΔAIC_c = change in the AIC_c value; w_i = model weight; Var = variance explained.

Model	*k*	AIC_c	ΔAIC_c	w_i	Model Likelihood	Var
ψ(species + %forest) *p*(method)	7	1196.46	0.00	0.82	1.00	0.06
ψ(species) *p*(method)	6	1200.22	3.76	0.13	0.15	0.06
ψ(species + %shrub) *p*(method)	7	1204.85	8.39	0.01	0.02	0.06
ψ(species + area) *p*(method)	7	1205.64	9.18	0.01	0.01	0.06
ψ(species) *p*(method + %grass)	7	1206.45	9.99	0.01	0.01	0.06
ψ(species) *p*(method + %shrub)	7	1206.63	10.17	0.01	0.01	0.06
ψ(species + %forbs) *p*(method)	7	1206.89	10.43	0.00	0.01	0.06
ψ(species + %grass) *p*(method)	7	1207.31	10.85	0.00	0.00	0.06
ψ(species) *p*(method + veg.density)	7	1207.43	10.97	0.00	0.00	0.06
ψ(species + veg.density) *p*(method)	7	1207.84	11.38	0.00	0.00	0.06
ψ(species) *p*(method + %forbs)	7	1207.85	11.39	0.00	0.00	0.06
ψ(species + hydrology) *p*(method)	8	1210.68	14.22	0.00	0.00	0.06

ψ(species) *p*(.)	5	1215.46	19.00	0.00	0.00	0.04
ψ(.) *p*(method)	3	1227.48	31.02	0.00	0.00	0.02
ψ(.) *p*(.)	2	1239.02	42.56	0.00	0.00	0.00
ψ(species) *p*(method + time)	15	1251.43	54.97	0.00	0.00	0.07
ψ(species) *p*(time)	14	1260.85	64.39	0.00	0.00	0.06
ψ(.) *p*(method + time)	12	1279.63	83.17	0.00	0.00	0.03
ψ(.) *p*(time)	11	1286.07	89.61	0.00	0.00	0.02

Table 19. Parameter estimates for carnivore detection probability (*p*) and occupancy (ψ) in 30 stormwater ponds (SWPs) located in Owings Mills, Maryland, USA. These estimates were modeled to vary by detection method, species, or the proportion of forest in the landscape surrounding the SWP. SE = standard error; 95% CI = 95% confidence interval.

Model	Parameter	Estimate	SE	95% CI
ψ(species + %forest) *p*(method)	$\psi_{D7\text{-}1\ Cat}$	0.36	0.08	0.23 - 0.52
	$\psi_{D7\text{-}1\ Fox}$	0.61	0.09	0.43 – 0.76
	$\psi_{D7\text{-}1\ Opossum}$	0.98	0.05	0.20 – 1.00
	$\psi_{D7\text{-}1\ Raccoon}$	0.98	0.05	0.20 – 1.00
	$\psi_{E10\text{-}2\ Cat}$	0.34	0.08	0.21 – 0.50
	$\psi_{E10\text{-}2\ Fox}$	0.58	0.09	0.40 – 0.74
	$\psi_{E10\text{-}2\ Opossum}$	0.20	0.06	0.11 – 0.35
	$\psi_{E10\text{-}2\ Raccoon}$	0.98	0.05	0.19 – 1.00
	$\psi_{E3\text{-}1\ Cat}$	0.17	0.07	0.07 – 0.36
	$\psi_{E3\text{-}1\ Fox}$	0.36	0.11	0.18 – 0.59
	$\psi_{E3\text{-}1\ Opossum}$	0.09	0.05	0.03 – 0.24
	$\psi_{E3\text{-}1\ Raccoon}$	0.95	0.13	0.10 – 1.00
	$\psi_{E3\text{-}2\ Cat}$	0.34	0.08	0.21 – 0.50
	$\psi_{E3\text{-}2\ Fox}$	0.58	0.09	0.40- 0.74
	$\psi_{E3\text{-}2\ Opossum}$	0.20	0.06	0.11 – 0.35

$\psi_{E3\text{-}2\ Raccoon}$	0.98	0.06	0.19 – 1.00
$\psi_{E4\text{-}1\ Cat}$	0.56	0.09	0.38 – 0.72
$\psi_{E4\text{-}1\ Fox}$	0.78	0.08	0.59 – 0.89
$\psi_{E4\text{-}1\ Opossum}$	0.39	0.09	0.23 – 0.57
$\psi_{E4\text{-}1\ Raccoon}$	0.99	0.02	0.33 – 1.00
$\psi_{E4\text{-}2\ Cat}$	0.23	0.08	0.11 – 0.41
$\psi_{E4\text{-}2\ Fox}$	0.45	0.10	0.26 – 0.65
$\psi_{E4\text{-}2\ Opossum}$	0.13	0.05	0.06 – 0.27
$\psi_{E4\text{-}2\ Raccoon}$	0.96	0.09	0.13 – 1.00
$\psi_{E4\text{-}4\ Cat}$	0.42	0.08	0.27 – 0.57
$\psi_{E4\text{-}4\ Fox}$	0.66	0.08	0.48 – 0.80
$\psi_{E4\text{-}4\ Opossum}$	0.26	0.07	0.15 – 0.42
$\psi_{E4\text{-}4\ Raccoon}$	0.99	0.04	0.24 – 1.00
$\psi_{E7\text{-}2\ Cat}$	0.61	0.10	0.41 – 0.78
$\psi_{E7\text{-}2\ Fox}$	0.81	0.08	0.62 – 0.92
$\psi_{E7\text{-}2\ Opossum}$	0.44	0.10	0.26 – 0.64
$\psi_{E7\text{-}2\ Raccoon}$	0.99	0.02	0.37 – 1.00
$\psi_{E7\text{-}4\ Cat}$	0.67	0.10	0.45 – 0.83
$\psi_{E7\text{-}4\ Fox}$	0.85	0.07	0.65 – 0.94
$\psi_{E7\text{-}4\ Opossum}$	0.50	0.11	0.29 – 0.71
$\psi_{E7\text{-}4\ Raccoon}$	0.99	0.02	0.42 – 1.00
$\psi_{E7\text{-}5\ Cat}$	0.25	0.08	0.13 – 0.42
$\psi_{E7\text{-}5\ Fox}$	0.48	0.10	0.29 – 0.67

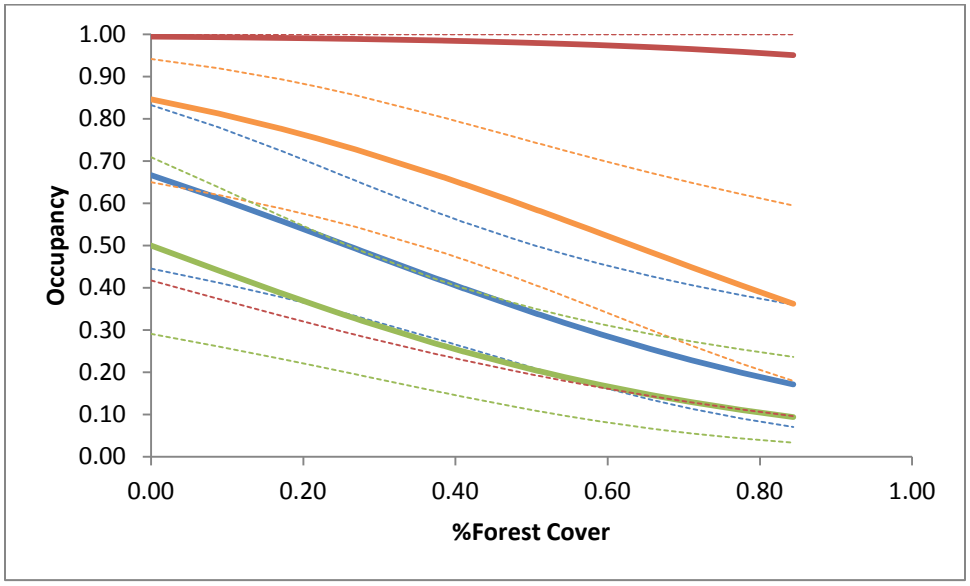
$\psi_{E7\text{-}5\ Opossum}$	0.14	0.06	0.06 – 0.29
$\psi_{E7\text{-}5\ Raccoon}$	0.97	0.08	0.14 – 1.00
$\psi_{E9\text{-}2\ Cat}$	0.49	0.08	0.33 – 0.65
$\psi_{E9\text{-}2\ Fox}$	0.72	0.08	0.54 – 0.85
$\psi_{E9\text{-}2\ Opossum}$	0.32	0.08	0.19 – 0.49
$\psi_{E9\text{-}2\ Raccoon}$	0.99	0.03	0.28 – 1.00
$\psi_{E9\text{-}4\ Cat}$	0.32	0.08	0.19 – 0.48
$\psi_{E9\text{-}4\ Fox}$	0.56	0.09	0.38 – 0.73
$\psi_{E9\text{-}4\ Opossum}$	0.19	0.06	0.10 – 0.34
$\psi_{E9\text{-}4\ Raccoon}$	0.98	0.06	0.18 – 1.00
$\psi_{F4\text{-}1\ Cat}$	0.67	0.10	0.45 – 0.83
$\psi_{F4\text{-}1\ Fox}$	0.85	0.07	0.65 – 0.94
$\psi_{F4\text{-}1\ Opossum}$	0.50	0.11	0.29 – 0.71
$\psi_{F4\text{-}1\ Raccoon}$	0.99	0.02	0.42 – 1.00
$\psi_{F6\text{-}3\ Cat}$	0.67	0.10	0.45 – 0.83
$\psi_{F6\text{-}3\ Fox}$	0.85	0.07	0.65 – 0.94
$\psi_{F6\text{-}3\ Opossum}$	0.50	0.11	0.29 – 0.71
$\psi_{F6\text{-}3\ Raccoon}$	0.99	0.02	0.42 – 1.00
$\psi_{F7\text{-}1\ Cat}$	0.42	0.08	0.28 – 0.58
$\psi_{F7\text{-}1\ Fox}$	0.67	0.08	0.49 – 0.81
$\psi_{F7\text{-}1\ Opossum}$	0.27	0.07	0.16 – 0.42
$\psi_{F7\text{-}1\ Raccoon}$	0.99	0.04	0.24 – 1.00
$\psi_{F7\text{-}5\ Cat}$	0.32	0.08	0.19 – 0.48

$\psi_{F7\text{-}5\ Fox}$	0.56	0.09	0.38 – 0.73
$\psi_{F7\text{-}5\ Opossum}$	0.19	0.06	0.10 – 0.33
$\psi_{F7\text{-}5\ Raccoon}$	0.98	0.06	0.18 – 1.00
$\psi_{F8\text{-}3\ Cat}$	0.42	0.08	0.27 – 0.57
$\psi_{F8\text{-}3\ Fox}$	0.66	0.08	0.48 – 0.80
$\psi_{F8\text{-}3\ Opossum}$	0.26	0.07	0.15 – 0.42
$\psi_{F8\text{-}3\ Raccoon}$	0.99	0.04	0.24 – 1.00
$\psi_{F8\text{-}8\ Cat}$	0.53	0.09	0.36 – 0.69
$\psi_{F8\text{-}8\ Fox}$	0.75	0.08	0.57 – 0.88
$\psi_{F8\text{-}8\ Opossum}$	0.36	0.08	0.21 – 0.53
$\psi_{F8\text{-}8\ Raccoon}$	0.99	0.03	0.31 – 1.00
$\psi_{F9\text{-}1\ Cat}$	0.26	0.08	0.14 – 0.43
$\psi_{F9\text{-}1\ Fox}$	0.48	0.10	0.30 – 0.67
$\psi_{F9\text{-}1\ Opossum}$	0.15	0.06	0.07 – 0.29
$\psi_{F9\text{-}1\ Raccoon}$	0.97	0.08	0.14 – 1.00
$\psi_{G6\text{-}1\ Cat}$	0.29	0.08	0.17 – 0.46
$\psi_{G6\text{-}1\ Fox}$	0.53	0.09	0.35 – 0.70
$\psi_{G6\text{-}1\ Opossum}$	0.17	0.06	0.08 – 0.31
$\psi_{G6\text{-}1\ Raccoon}$	0.97	0.07	0.16 – 1.00
$\psi_{G6\text{-}10\ Cat}$	0.67	0.10	0.45 – 0.83
$\psi_{G6\text{-}10\ Fox}$	0.85	0.07	0.65 – 0.94
$\psi_{G6\text{-}10\ Opossum}$	0.50	0.11	0.29 – 0.71
$\psi_{G6\text{-}10\ Raccoon}$	0.99	0.02	0.42 – 1.00

$\psi_{G6\text{-}12\ Cat}$	0.50	0.09	0.34 – 0.66
$\psi_{G6\text{-}12\ Fox}$	0.74	0.08	0.55 – 0.86
$\psi_{G6\text{-}12\ Opossum}$	0.34	0.08	0.20 – 0.50
$\psi_{G6\text{-}12\ Raccoon}$	0.99	0.03	0.30 – 1.00
$\psi_{G6\text{-}14\ Cat}$	0.24	0.08	0.13 – 0.42
$\psi_{G6\text{-}14\ Fox}$	0.47	0.10	0.28 – 0.66
$\psi_{G6\text{-}14\ Opossum}$	0.14	0.05	0.06 – 0.28
$\psi_{G6\text{-}14\ Raccoon}$	0.97	0.08	0.14 – 1.00
$\psi_{G6\text{-}5\ Cat}$	0.67	0.10	0.45 – 0.83
$\psi_{G6\text{-}5\ Fox}$	0.85	0.07	0.65 – 0.94
$\psi_{G6\text{-}5\ Opossum}$	0.50	0.11	0.29 – 0.71
$\psi_{G6\text{-}5\ Raccoon}$	0.99	0.02	0.42 – 1.00
$\psi_{G8\text{-}2\ Cat}$	0.30	0.08	0.17 – 0.46
$\psi_{G8\text{-}2\ Fox}$	0.54	0.09	0.36 – 0.71
$\psi_{G8\text{-}2\ Opossum}$	0.18	0.06	0.09 – 0.32
$\psi_{G8\text{-}2\ Raccoon}$	0.98	0.07	0.17 – 1.00
$\psi_{H5\text{-}1\ Cat}$	0.56	0.09	0.38 – 0.73
$\psi_{H5\text{-}1\ Fox}$	0.78	0.08	0.59 – 0.89
$\psi_{H5\text{-}1\ Opossum}$	0.39	0.09	0.23 – 0.57
$\psi_{H5\text{-}1\ Raccoon}$	0.99	0.02	0.33 – 1.00
$\psi_{H7\text{-}1\ Cat}$	0.49	0.08	0.33 – 0.65
$\psi_{H7\text{-}1\ Fox}$	0.73	0.08	0.54 – 0.86
$\psi_{H7\text{-}1\ Opossum}$	0.33	0.08	0.20 – 0.49

$\psi_{\text{H7-1 Raccoon}}$	0.99	0.03	0.29 – 1.00
$\psi_{\text{H8-2 Cat}}$	0.40	0.08	0.26 – 0.56
$\psi_{\text{H8-2 Fox}}$	0.64	0.09	0.47 – 0.79
$\psi_{\text{H8-2 Opossum}}$	0.25	0.07	0.14 – 0.40
$\psi_{\text{H8-2 Raccoon}}$	0.98	0.04	0.23 - 1.00
$\psi_{\text{I8-1 Cat}}$	0.40	0.08	0.26 – 0.55
$\psi_{\text{I8-1 Fox}}$	0.64	0.09	0.47 – 0.79
$\psi_{\text{I8-1 Opossum}}$	0.25	0.07	0.14 – 0.40
$\psi_{\text{I8-1 Raccoon}}$	0.98	0.04	0.23 – 1.00
$\psi_{\text{I8-2 Cat}}$	0.20	0.07	0.09 – 0.38
$\psi_{\text{I8-2 Fox}}$	0.40	0.11	0.22 – 0.62
$\psi_{\text{I8-2 Opossum}}$	0.11	0.05	0.04 – 0.25
$\psi_{\text{I8-2 Raccoon}}$	0.96	0.11	0.11 – 1.00
p_{camera}	0.23	0.02	0.19 – 0.28
$p_{track\ plates}$	0.42	0.03	0.36 – 0.49

Figure 10. Relationship between the proportion of forest cover in the surrounding landscape period and estimated occupancy of carnivores from model ψ(species + %forest) p(method), where *p* is detection probability and ψ is occupancy. The solid lines represent the parameter estimates (feral cat is blue, northern raccoon is red, red fox is orange, and Virginia opossum is green) and the dotted lines are the upper and lower 95% confidence intervals



LITERATURE CITED

Adler, G.H., and M.L. Wilson. 1987. Demography of a habitat generalist, the white-footed mouse, in a heterogeneous environment. Ecology 68: 1785-1796.

Ahlers, A.A., R.L. Schooley, E.J. Heske, and M.A. Mitchell. 2010. Effects of flooding and riparian buffers on survival of muskrats (*Ondatra zibethicus*) across a flashiness gradient. Canadian Journal of Zoology 88: 1011-1020.

Akaike, H. 1973. Information theory and an extension of the maximum likelihood principle, 2nd International Symposium of Information Theory, B.N. Petrov and F. Csaki, eds., Akademiai Kiado, Budapest, 267-281.

Allen, C.R., D.M. Epperson, and A.S. Garmestani. 2004 Red Imported Fire Ant impacts on wildlife: A decade of research. American Midland Naturalist 152: 88-103.

Althoff, D.P., G.L. Storm, and D.R. Dewalle. 1997. Daytime habitat selection by cottontails in central Pennsylvania. The Journal of Wildlife Management 61: 450-459.

Anderson, C.S., A.B. Cady, and D.B. Meikle. Effects of vegetation structure and edge habitat on the density and distribution of white-footed mice (*Peromyscus leucopus*) in small and large forest patches. Canadian Journal of Zoology 81: 897-904.

Anderson, D.C., K.R. Wilson, M.S. Miller, and M. Falck. 2000. Movement patterns of riparian small mammals during predictable floodplain inundation. Journal of Mammalogy 81: 1087-1099.

Anderson, D.R., and K.P. Burnham. 1999. Understanding information criteria for selecting among capture-recapture or ring recovery models. Bird Study 46: S515-S521.

Anderson, D.R., K.P. Burnham, and G.C. White. 1994. AIC model selection in overdispersed capture-recapture data. Ecology 75: 1780-1793.

Arnold Jr., C.L., and C.J. Gibbons. 1996. Impervious surface coverage: The emergence of a key environmental indicator. Journal of the American Planning Association 62: 243-258.

August, P.V. 1983. The role of complexity and heterogeneity in structuring tropical mammal communities. Ecology 64: 1495-1507.

Bailey, L.L., T.R. Simons, and K.H. Pollock. 2004. Estimating site occupancy and species detection probability parameters for terrestrial salamanders. Ecological Applications 14: 692-702.

Baish, A.S., and M.J. Caliri. 2009. Average nutrient and sediment effluent removal efficiencies for stormwater best management practices implemented in Maryland 1984-2002. Baltimore, MD: Johns Hopkins University.

Bajomi, D., and K. Sasvari. 1986. Results of eight years' examination of the habitats of residual urban Norway rat populations after eradication. Proceedings of the 12th Vertebrate Pest Conference 66-74.

Baker, P.J., R.J. Ansell, P.A.A. Dodds, C.E. Webber, and S. Harris. 2003. Factors affecting the distribution of small mammals in an urban area. Mammal Review 33: 95-100.

Barbehenn, K.R. 1970. Notes on the ecology of sewer rats in St. Louis. Proceedings of the 4th Vertebrate Pest Conference 19-22.

Bartoszewicz, M., H. Okarma, A. Zalewski, and J. Szczesna. 2008. Ecology of the raccoon (*Procyon lotor*) from western Poland. Annales Zoologici Fennici 45: 291-298.

Beasley, J.C., Z.H. Olson, G. Dharmarajan, T.S. Eagan II, and O.E. Rhodes Jr. 2011. Spatio-temporal variation in the demographic attributes of a generalist mesopredator. Landscape Ecology 26: 937-950.

Beck, H., M.S. Gaines, J.E. Hines, and J.D. Nichols. 2004. Comparative dynamics of small mammal populations in treefall gaps and surrounding understory within Amazonian rainforest. Oikos 106: 27-38.

Behera, P.K., F. Papa, and B.J. Adams. 1999. Optimization of regional storm-water management systems. Journal of Water Resources Planning and Management 125:107-114.

Belant, J. L. 2003. Comparison of 3 tracking mediums for detecting forest carnivores. Wildlife Society Bulletin 31: 744-747.

Bennett, A.F. 1990. Habitat corridors and the conservation of small mammals in a fragmented forest environment. Landscape Ecology 4: 109-122.

Bishop, C.A., J. Struger, D.R. Barton, L.J. Shirose, L. Dunn, A.L. Land, and D. Shepherd. 2000. Contamination and wildlife communities in stormwater detention ponds in Guelph and the greater Toronto area, Ontario, 1997 and 1998.Part 1—Wildlife communities. Water Quality Research Journal of Canada 35: 399-435.

Blouin-Fdemers, G., and P.J. Weatherhead. 2001. Habitat use by black snakes (*Elaphe obsoleta obsoleta*) in fragmented forests. Ecology 82: 2882-2896.

Bowers, M.A., and B, Breland. 1996. Foraging of gray squirrels on an urban-rural gradient: Use of the GUD to assess anthropogenic impact. Ecological Applications 6: 1135-1142.

Brand, A.B., and J.W. Snodgrass. 2010. Value of artificial habitats for amphibian reproduction in altered landscapes. Conservation Biology 24: 295-301.

Brannon, M.P. 2005. Distribution and microhabitat of the woodland jumping mouse, *Napaeozapus insignis*, and the white-footed mouse, *Peromyscus leucopus*, in the southern Appalachians. Southeastern Naturalist 4: 479-486.

Brooks, R.T., and K.L. Doyle. 2001. Shrew species richness and abundance in relation to vernal pond habitat in southern New England. Northeastern Naturalist 8: 137-148.

Buckner, C.H. 1966. The role of vertebrate predators in the biological control of forest insects. Annual Review of Entomology 11: 449-470.

Bull, E.L., T.R. Torgersen, and T.L. Wertz. 2001. The importance of vegetation, insects, and neonate ungulates in black bear diet in northeastern Oregon. Northwest Science 75: 244-253.

Burnham, K.P., and D.R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Springer, New York, New York. 67, 446p.

Burnham, K.P., D.R. Anderson, G.C. White, C. Brownie, and K.H. Pollock. 1987. Design and analysis methods for fish survival experiments based on release-recapture. American Fisheries Society, Bethesda, Maryland. 737p.

Cadenasso, M.L., S.T.A. Pickett, and M.J. Grove. 2006. Integrative approaches to investigating human-natural systems: The Baltimore ecosystem study. Natures Sciences Sociétés 14: 4-14.

Calhoun, J.B. 1959. Revised sampling procedure of the North American census of small mammals (NACSM)-Population dynamics of vertebrates. U.S. Department of Health, Education, and Welfare, Mimeo Administrative Publication10.

Camponelli, K.M., S.M. Lev, J.W. Snodgrass, E.R. Landa, and R.E. Casey. 2010. Chemical fractionation of Cu and Zn in stormwater, roadway dust and stormwater pond sediments. Environmental Pollution 158: 2143-2149.

Casey, R.E., A.N. Shaw, L.R. Massal, and J.W. Snodgrass. 2005. Multimedia evaluation of trace metal distribution within stormwater retention ponds in suburban Maryland, USA. Bulletin of Environmental Contamination and Toxicology 74: 273-280.

Casey, R.E., J.A. Simon, S. Atueyi, J.W. Snodgrass, N. Karouna-Renie, and D.W Sparling. 2006. Temporal trends of trace metals in sediment and invertebrates from stormwater management ponds. Water, Air, and Soil Pollution 178: 69–77.

Catling, P.C., N. Coops, and R.J. Burt. 2001. The distribution and abundance of ground-dwelling mammals in relation to time since wildfire and vegetation structure in southeastern Australia. Wildlife Research 28: 555-565.

Cavallini, P., and T. Volpi. 1996. Variation in the diet of the red fox in a Mediterranean area. Revue d'ecologie 51: 173-189.

Cavia, R., G.R. Cueto, and O.V. Suarez. 2009. Changes in rodent communities according to the landscape structure in an urban ecosystem. Landscape and Urban Planning 90: 11-19.

Central Intelligence Agency. 2010. The World Factbook: United States. <https://www.cia.gov/library/publications/the-world-factbook/geos/us.html>. Accessed 2012 Jan 15.

Chamberlain, M.J., J. Austin, B.D. Leopold, and L.W. Burger Jr. 2007. Effects of landscape composition and structure on core use areas of raccoons (*Procyon lotor*) in a prairie landscape. American Midland Naturalist 158:113-122.

Chamberlain, M.J., L.M. Connor, and B.D. Leopold. 2002. Seasonal habitat selection by raccoons (*Procyon lotor*) in intensively managed pine forests of central Mississippi. American Midland Naturalist 147: 102-108.

Chamberlain, M.J., and B.D. Leopold. 2003. Effects of a flood on relative abundance and

diversity of small mammals in a regenerating bottomland hardwood forest. The Southwestern Naturalist 48: 306-309.

Charland, M.B., and P.T. Gregory. 2009. Movements and habitat use in gravid and non-gravid female garter snakes (Colubridae: *Thamnophis*). Journal of Zoology 236: 543-561.

Childs, J.E., S.L. McLafferty, R. Sadek, G.L Miller, A.S. Khan, E.R. DuPree, R. Advani, J.N. Mills, and G.E. Glass. 1998. American Journal of Epidemiology 148: 78-87.

Colosimo, M.F., and P.R. Wilcock. 2007. Alluvial sedimentation and erosion in an urbanizing watershed, Gwynns Falls, Maryland. Journal of the American Water Resources Association 43: 499-521.

Cormack, R.M. 1964. Estimates of survival from the sighting of marked animals. Biometrika 51: 429-438.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O'Neill, J. Paruelo, R.G. Raskin, P. Sutton, M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature 387: 253-260.

Cowardin, L.M., V. Carter, F.C. Golet, and E.T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. United States Department of Interior, Fish and Wildlife Service. Washington, D.C.

Crooks, K.R. 2002. Relative sensitivities of mammalian carnivores to habitat fragmentation. Conservation Biology 16: 488-502.

Crooks, K.R., and M.E. Soulé. 1999. Mesopredator release and avifaunal extinctions in a fragmented system. Nature 400: 563-566.

Croonquist, M.J., and R.P. Brooks. 1991. Use of avian and mammalian guilds as indicators of cumulative impacts in riparian-wetland areas. Environmental Management 15: 701-714.

Dell'Arte, G.L., T. Laaksonen, K. Norrdahl, E. Korpimaki. 2007. Variation in the diet composition of a generalist predator, the red fox, in relation to season and density of main prey. Acta Oecologica 31: 276-281.

DEPS 2011. Upper Gwynns Falls Final Report: Small Watershed Action Plan. Department of Environmental Protection and Sustainability, Baltimore County, MD.

DeStefano, S., and R.M. DeGraaf. 2003. Exploring the ecology of suburban wildlife. Frontiers in Ecology and the Environment 1: 95-101.

DeVault, T.L., Z.H. Olson, J.C. Beasley, O.E. Rhodes, Jr. 2011. Mesopredator dominate competition for carrion in an agricultural landscape. Basic and Applied Ecology 12: 268-274.

Devictor, V., R. Julliard, J. Clavel, F. Jiguet, A. Lee, and D. Couvet. 2008. Functional biotic homogenization of bird communities in disturbed landscapes. Global Ecology and Biogeography 17: 252-261.

Dickman, C.R. 1987. Habitat fragmentation and vertebrate species richness in an urban environment. Journal of Applied Ecology 24: 337-351.

Dickman, C.R., and C.P. Doncaster. 1987. The ecology of small mammals in urban habitats. I. Populations in a patchy environment. Journal of Animal Ecology 56: 629-640.

Dooley Jr., J.L., and M.A. Bowers. 1998. Demographic responses to habitat fragmentation: Experimental tests at the landscape and patch scale. Ecology 79: 969-980.

Dueser, R.D., and H.H. Shugart, Jr. 1978. Microhabitats in a forest-floor small mammal fauna. Ecology 59: 89-98.

Eagan II, T.S., J.C. Beasley, Z.H. Olson, and O.E. Rhodes, Jr. 2011. Impacts of generalist mesopredators on the demography of small-mammal populations in fragmented landscapes. Canadian Journal of Zoology 89: 724-731.

Elbroch, M. 2003. Mammal tracks and sign: A guide to North American species. Stackpole Books, Mechanicsburg, Pennsylvania, USA.

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. Annual Review of Ecology, Evolution and Systematics 34: 487-515.

Fahrig, L., and G. Merriam. 1985. Habitat patch connectivity and population survival. Ecology 66: 1762-1768.

Faulkner, S. 2004. Urbanization impacts on the structure and function of forested wetlands. Urban Ecosystems 7: 89-106.

Fischer, J., and D.B. Lindenmayer. 2007. Landscape modification and habitat fragmentation: a synthesis. Global Ecology and Biogeography 16: 265-280.

Fitzpatrick, B.A. 2011. Species Richness and Community Structure of Breeding Bird Communities in Artificial Urban Wetlands. M.S. Thesis, Towson University, Towson, Maryland.

Ford, W.M., J. Laerm, D.C. Weinand, and K.G. Barker. 1994. Abundance and distribution of shrews and other small mammals in the Chattahoochee National Forest of Georgia. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies 48: 310-320.

Foster, J., and M.S. Gaines. 1991. The effects of a successional habitat mosaic on a small mammal community. Ecology 72: 1358-1373.

Fuller, T.K., and S. DeStefano. 2003. Relative importance of early-successional forests and shrubland habitats to mammals in the northeastern United States. Forest Ecology and Management 185: 75-79.

Gallagher, M.T. 2009. Wood frog (*Rana sylvatica*) use and reproductive success in stormwater ponds of varying contaminant levels. M.S. Thesis, Towson University, Towson, Maryland.

Galli, J. 1990. Thermal impacts associated with urbanization and storm water management best management practices. Metropolitan Washington Council of Governments. Prepared for Maryland Department of the Environment, Baltimore, MD.

Garden, J.G., C.A. McAlpine, H.P. Possingham, and D.N. Jones. 2007. Habitat structure is more important than vegetation composition for local-level management of

native terrestrial reptile and small mammal species living in urban remnants: A case study from Brisbane, Australia. Austral Ecology 32: 669-685.

Gehrt, S. D., and W. R. Clark. 2003. Raccoons, Coyotes, and Reflections on the Mesopredator Release Hypothesis. Wildlife Society Bulletin 31: 836-842.

Gehrt, S. D., and S. Prange. 2007. Interference competition between coyotes and raccoons: a test of the mesopredator release hypothesis. Behavioral Ecology. 18:204-214.

George, S.B., J.R. Choate, and H.H. Genoways. 1986. *Blarina brevicauda*. Mammalian Species 261: 1-9.

George, S.L., and K.R. Crooks. 2006. Recreation and large mammal activity in an urban nature reserve. Biological Conservation 133: 107-117.

Gibbs, J.P. 1993. Importance of small wetlands for the persistence of local populations of wetland-associated animals. Wetlands 13: 25-31.

Gibbs, J.P. 2000. Wetland loss and biodiversity conservation. Conservation Biology 14: 314-317.

Gomez, M.D., J. Priotto, M.C. Provensal, A. Steinmann, E. Castillo, and J.J. Polop. 2008. A population study of house mice (*Mus musculus*) inhabiting different habitats in an Argentine urban area. International Biodeterioration and Biodegradation 62: 270-273.

Gomez, M.D., M.C. Provensal, and J.J. Polop. 2008. Effect of interspecific competition on *Mus musculus* in an urban area. Journal of Pest Science 81: 235-240.

Gompper, M.E., R.W. Kays, J.C. Ray, S.D. Lapoint, D.A. Bogan, and J.R. Cryan. 2006. A comparison of noninvasive techniques to survey carnivore communities in northeastern North America. Wildlife Society Bulletin 34: 1142-1151.

Groffman, P.M., R.V. Pouyat, M.L. Cadenasso, W.C. Zipperer, K. Szlavecz, I.D. Yesilonis, L.E. Brand, and G.S. Brush. 2006. Land use and natural soil controls on plant community composition and soil nitrogen and carbon dynamics in urban and rural forests. Forest Ecology and Management 236: 177-192.

Grund, M.D., J.B. McAninch, and E.P. Wiggers. 2002. Seasonal movements and habitat use of female white-tailed deer associated with an urban park. The Journal of Wildlife Management 66: 123-130.

Halfpenny, J. 1986. A field guide to mammal tracking in North America. Second edition. Johnson Publishing Company, Boulder, Colorado, USA.

Henein, K., and G. Merriam. 1990. The elements of connectivity where corridor quality is variable. Landscape Ecology 4: 157-170.

Hilderbrand, R.H., R.M Utz, S.A. Stranko, and R.L. Raesly. 2010. Applying thresholds to forest potential biodiversity loss from human development. Journal of North American Benthological Society 29: 1009-1016.

Hines, J.E. 2006. PRESENCE2- Software to estimate patch occupancy and related parameters. USGS-PWRC. http://www.mbr-wrc.usgs.gov/software/presence.html.

Hossler, R.J., J.B, McAninch, and J.D. Harder. 1994. Maternal denning behavior and survival of juveniles in opossums in southeastern New York. Journal of Mammalogy 75: 60-70.

Hurvich, C.M., and C. Tsai. 1989. Regression and time series model selection in small samples. Biometrika. 76: 297-307.

Ivan, J.S., and R.K. Swihart. 2000. Selection of mast by granivorous rodents of the central hardwood forest region. Journal of Mammalogy 81: 549-562.

Johnson, J.B., and K.S. Omland. 2004. Model selection in ecology and evolution. TRENDS in Ecology and Evolution 19: 101-108.

Johnson, M.S., R.D. Roberts, M. Hutton, and M.J. Inskip. 1978. Distribution of lead, zinc and cadmium in small mammals from polluted environments. Oikos 30: 153-159.

Johnson, C.A. 1994. Cumulative impacts to wetlands. Wetlands 14: 49-55.

Jolly, G.M. 1965. Explicit estimates from capture-recapture data with both death and immigration-stochastic model. Biometrika 52: 225-247.

Kelly, M.J., and E.L. Holub. 2008. Camera trapping of carnivores: Trap success among camera types and across species, and habitat selection by species, on Salt Pond Mountain, Giles, County, Virginia. Northeastern Naturalist 15: 249-262.

Kitchings, J.T., and D.J. Levy. 1981. Habitat patterns in a small mammal community. Journal of Mammalogy 62: 814-820.

Kohn, D.D., and D.M. Walsh. 1994. Plant species richness- the effect of island size and habitat diversity. Journal of Ecology 82: 367-377.

Lackey, J.A., D.G. Huckaby, and B.G. Orniston. 1985. *Peromyscus leucopus*. Mammalian Species 247: 1-10.

Laliberte, A.S., and W.J. Ripple. 2004. Range contractions of North American carnivores and ungulates. Bioscience 54: 123-138.

Larivière, S., and M. Pasitschniak-Arts. 1996. *Vulpes vulpes*. Mammalian Species 537: 1-11.

Lebreton, J.D., K.P. Burnham, J. Colbert, and D.R. Anderson.1992. Modeling survival and testing biological hypotheses using marked animals: a unified approach with case studies. Ecological Monographs 62:67-118.

Lehmkuhl, J.F., R.D. Peffer, and M.A. O'Connell. 2008. Riparian and upland small mammals on the east slope of the Cascade Range, Washington. Northwest Science 82: 94-107.

Lev, S.M., N. Matthies, J.W. Snodgrass, R.E. Casey, and D.R. Ownby. 2010. Effects of zinc exposure on earthworms, *Lumbricus terrestris*, in an artificial soil. Bulletin of Environmental Contamination and Toxicology 84: 687-691.

Liebhold, A, J. Elkinton, D. Williams, and R. Muzika. 2000. What causes outbreaks of the gypsy moth in North America? Population Ecology 42: 257-266.

Lindstrom, E.R. 1994. Large prey for small cubs- on crucial resources of a boreal red fox population. Ecography 17: 17-22.

Lotze, J., and S. Anderson. 1979. *Procyon lotor*. Mammalian Species 119: 1-8.

Loukmas, J.J., D.T. Mayack, and M.E. Richmond. 2002. Track plate enclosures: Box designs affecting attractiveness to riparian mammals. American Midland Naturalist 149: 219– 224.

Lowell, W.A, and A.D. Geis. 1983. Effects of roads on small mammals. Journal of Applied Ecology 20: 403-415.

Lowrance, R., L.S. Altier, J.D. Newbold, R.R. Schnabel, P.M. Groffman, J.M. Denver, D.L. Correll, J.W. Gilliam, J.L. Robinson, R.B. Brinsfield, K.W. Staver, W. Lucas, and A.H. Todd. 1997. Water quality functions of riparian forest buffers in Chesapeake Bay Watersheds. Environmental Management 21: 687-712.

Ma, W., W. Denneman, and J. Faber. 1991. Hazardous exposure of ground-living small mammals to cadmium and lead in contaminated terrestrial ecosystems. Archives of Environmental Contamination and Toxicology 20: 266-270.

MacKenzie, D.I., J.D. Nichols, G.B. Lachman, S. Droege, J.A. Royle, and C.A. Langtimm. 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83:2248-2255.

MacKenzie, D.I., J.D. Nichols, J.A. Royle, K.H. Pollock, L.L. Bailey, and J.E. Hines. 2006. Occupancy estimation and modeling: Inferring patterns and dynamics of species occurrence. Elsevier Academic Press, Burlington, MA.

MacKenzie, D.I., and L.L. Bailey. 2004. Assessing the fit of site-occupancy models. Journal of Agriculture, Biological and Ecological Statistics 9:300-318.

Manson, RH., R.S. Ostfeld, and C.D. Canham. 1999. Responses of a small mammal community to heterogeneity along forest-old-field edges. Landscape Ecology 14: 355-367.

McDonnell, M.J., and S.T.A. Pickett. 1990. Ecosystem structure and function along urban-rural gradients: An unexploited opportunity for ecology. Ecology 71: 1232-1237.

McKinney, M.L. 2008. Effects of urbanization on species richness: A review of plants and animals. Urban Ecosystems 11: 161-176.

McKinney, M.L. 2006. Urbanization as a major cause of biotic homogenization. Biological Conservation 127: 247-260.

McKinney, M.L. 2002. Urbanization, biodiversity, and conservation. BioScience 52: 883-890.

McKinney, M.L., and J.L. Lockwood. 1999. Biotic homogenization: A few winners replacing many losers in the next mass extinction. Trends in Ecology and Evolution 14: 450-453.

McManus, J.J. 1974. *Didelphis virginiana*. Mammalian Species 40: 1-6.

MDNR. 2010. Rare, threatened, and endangered animals of Maryland. Maryland Department of Natural Resources Annapolis, Maryland, USA.

MDOT. 2012. Maryland Department of Transportation Office of Real Estate. <http://www.mdot- realestate.org/tod.asp> Accessed 2012 Dec 17.

MDP. 2009. Maryland's urban stormwater best management practices by era. Maryland Department of Planning Baltimore, Maryland, USA.

Meserve, P.L., J.R. Gutiérrez, and F.M. Jaksic. 1993. Effects of vertebrate predation on a caviomorph rodent, the degu (*Octodon degus*), in a semiarid thorn scrub community in Chile. Oecologia 94: 153-158.

Murie, O.J., and M. Elbroch. 2005. A field guide to animal tracks. Third edition. Houghton Mifflin Company, New York, New York, USA.

Myton, B. 1974. Utilization of space by *Peromyscus leucopus* and other small mammals. Ecology 55: 277-290.

Ng, S.J., J.W. Dolea, R.M. Sauvajotb, S.P.D. Riley, and T.J. Valone. 2004. Use of highway undercrossings by wildlife in southern California. Biological Conservation 115: 499–507.

Nichols, J.D., L.L. Bailey, A.F. O'Connell Jr., N.W. Talancy, E.H. Campbell Grant, A.T. Gilbert, E.M. Annand, T.P. Husband, and J.E. Hines. 2008. Multi-scale occupancy estimation and modeling using multiple detection methods. Journal of Applied Ecology 45: 1321-1329.

NOAA. 2011. February 2011 monthly climatic data for the world. National Climatic Data Center, Asheville, NC. 1-88.

NOAA. 2011. January 2011 monthly climatic data for the world. National Climatic Data Center, Asheville, NC. 1-94.

NOAA. 2010. December 2010 monthly climatic data for the world. National Climatic Data Center, Asheville, NC. 1-96.

NOAA. 2010. February 2010 monthly climatic data for the world. National Climatic Data Center, Asheville, NC. 1-110.

NOAA. 2010. January 2010 monthly climatic data for the world. National Climatic Data Center, Asheville, NC. 1-103.

NOAA. 2009. December 2009 monthly climatic data for the world. National Climatic Data Center, Asheville, NC. 1-106.

Nolte, D., D. Bergman, and J. Townsend. 2003. Roof rat invasion of an urban desert island. In: Singleton GR, LA Hinds, CJ Krebs & DM Spratt (eds) Rats, mice and people: rodent, biology and management. Australian Centre for International Agricultural Research, Canberra, Australia. 481-484.

Norrdahl, K., and E. Korpimaki. 1995. Effects of predator removal on vertebrate prey populations: Birds of prey and small mammals. Oecologia 103: 241-248.

Nupp, T.E., and R.K. Swihart. 2000. Landscape-level correlates of small-mammal assemblages in forest fragments of farmland. Journal of Mammalogy 81: 512-526.

Nupp, T.E, and R.K. Swihart. 1996. Effect of forest patch area on population attributes of white-footed mice (*Peromyscus leucopus*) in fragmented landscapes. Canadian Journal of Zoology 74: 467-472.

O'Connell, A.F. Jr., N.W. Talancy, L.L. Bailey, J.R. Sauer, R. Cook, A.T. Gilbert. 2006. Estimating site occupancy and detection probability parameters for meso- and

large mammals in a coastal ecosystem. The Journal of Wildlife Management 70: 1625-1633.

O'Connell, M.A. 1989. Population dynamics of neotropical small mammals in seasonal habitats. Journal of Mammalogy 70: 532-548.

Olden, J.D. 2006. Biotic homogenization: a new research agenda for conservation biogeography. Journal of Biogeography 33:2027-2039.

Olden, J.D., and N.L. Poff. 2004. Ecological processes driving biotic homogenization: Testing a mechanistic model using fish faunas. Ecology 85: 1867-1875.

Olden, J.D., N.L. Poff, M.R. Douglas, M.E. Douglas, and K.D. Fausch. 2004. Ecological and evolutionary consequences of biotic homogenization. Trends in Ecology and Evolution 19: 18-24.

Ostfeld, R.S. 1990. The ecology of territoriality in small mammals. Trends in Ecology and Evolution 5: 411-415.

Peavey, C.A., R.S. Lane, and J.E. Kleinjan. 1997. Role of small mammals in the ecology of *Borrelia burgdorferi* in a peri-urban park in north coastal California. Experimental and Applied Acarology 21: 569-584.

Perault, D.R., and M.V. Lomolino. 2000. Corridors and mammal community structure across a fragmented, old-growth forest landscape. Ecological Monographs 70: 401-422.

Pickett, S.T.A., and M.L. Cadenasso. 2008. Linking ecological and built components of urban mosaics: An open cycle of ecological design. Journal of Ecology 96: 8-12.

Pickett, S.A., M.L. Cadenasso, J.M. Grove, P.M. Groffman, L.E. Band, C.G. Boone, W.R. Burch Jr., C.S.B. Grimmond, J. Hom, J.C. Jenkins. N.L. Law, C.H. Nilon, R.V. Pouyat, K. Szlavecz, P.S. Warren, M.A. Wilson. 2008. Beyond urban legends: An emerging framework of urban ecology, as illustrated by the Baltimore Ecosystem Study. BioScience 58: 139-150.

Pocock, M.J.O., J.B. Searle, and P.C.L. White. 2004. Adaptations of animals to commensal habitats: Population dynamics of house mice *Mus musculus domesticus* on farms. Journal of Animal Ecology 73: 878-888.

Prange, S., S.D. Gehrt, and E.P. Wiggers. 2004. Influences of anthropogenic resources in raccoon (*Procyon lotor*) movements and spatial distribution. Journal of Mammalogy 85: 483-490.

Prugh, L.R., K.E. Hodges, A.R.E. Sinclair, and J.S. Brashares. 2008. Effect of habitat area and isolation on fragmented animal populations. Proceedings of the National Academy of Sciences 105: 20770-20775.

Rea, R.V. 2003. Modifying roadside vegetation management practices to reduce vehicular collisions with moose *Alces alces*. Wildlife Biology 9: 81-91.

Reeves, W.K., M.P. Nelder, and J.A. Korecki. 2005. *Bartonella* and *Rickettsia* in fleas and lice from mammals in South Carolina, U.S.A. Journal of Vector Ecology 30: 310-315.

Reichman, O.J., and S.C. Smith. 1990. Burrows and Burrowing behavior by mammals. Current Mammalogy 2:197-244.

Reid, D.G., C.J. Krebs, A. Kenney. 1995. Limitation of collared lemming population growth at low densities by predation mortality. Oikos 73: 387-398.

Rezendes, P. 1999. Tracking and the art of seeing: How to read animal tracks and sign. HarperCollins Publishers, Inc., New York, New York, USA.

Rich, S.M., C.W. Kilpatrick, J.L. Shippee, and K.L. Crowell. 1996. Morphological differentiation and identification of *Peromyscus leucopus* and *P. maniculatus* in northeastern North America. Journal of Mammalogy 77: 985-991.

Riley, S.P.D, R.M. Sauvajot, T.K. Fuller, E.C.York, D.A. Kamradt, C. Bromley, and R.K. Wayne. 2003. Effects of urbanization and habitat fragmentation on bobcats and coyotes in southern California. Conservation Biology 17: 566-576.

Ritchie, E.G., and C.N. Johnson. 2009. Predator interactions, mesopredator release and biodiversity conservation. Ecology Letters 12: 982-998.

Rodewald, A.D. 2003. The importance of land uses within the landscape matrix. Wildlife Society Bulletin 31: 586-592.

Rondeau, D., and J.M. Conrad. 2003. Managing urban deer. American Journal of Agricultural Economics 85: 266-281.

Rotenberry, J.T., and J.A.Wiens. 1980. Habitat structure, patchiness, and avian communities in North American steppe vegetation: A multivariate analysis. Ecology 61: 1228-1250.

Ryall, K.L., and L. Fahrig. 2006. Response of predators to loss and fragmentation of prey habitat: A review of theory. Ecology 87: 1086-1093.

Sargeant, A.B. 1978. Red fox demands and implications to prairie duck production. The Journal of Wildlife Management 42: 520-527.

Saunders, D.A., R.T. Hobbs, and C.R. Margules. 1991. Biological consequences of ecosystem fragmentation: A review. Conservation Biology 5:18-32.

Sauvajot, R.M., M. Buechner, D.A. Kamradt, and C.M. Schonewald. 1998. Patterns of human disturbance and response by small mammals and birds in chaparral near urban development. Urban Ecosystems 2: 279-297.

Schiller, A., and S.P. Horn. 1997. Wildlife conservation in urban greenways of the mid-southeastern United States. Urban Ecosystems 1: 103-116.

Schmeltz, L.L., and J.O. Whitaker, Jr. 1977. Use of woodchuck burrows by woodchucks and other animals. Transactions of the Kentucky Academy of Science 38:79-82.

Schueler, T.R. 1994. The importance of imperviousness. Watershed Protection Techniques 1: 100-111.

Schweiger, E.W, J.E. Diffenforfer, R.D. Holt, R. Pierotti, and M.S. Gaines. 2000. The interaction of habitat fragmentation, plant, and small mammal succession in an old field. Ecological Monographs 70:383-400.

Seber, G.A.F. 1965. A note on the multiple-recapture census. Biometrika 52: 249-259.

Seamon, J.O., and G.H. Adler. 1996. Population performance of generalist and specialist rodents along habitat gradients. Canadian Journal of Zoology 74: 1130-1139.

Shure, D.J. 1971. Tidal flooding dynamics: Its influence on small mammals in barrier beach marshes. American Midland Naturalist 85: 36-44.

Sikes, R.S., W.L. Gannon, and the Animal Care and Use Committee of Mammalogists. 2011. Guidelines of the American Society of Mammalogists for the use of wild mammals in research. Journal of Mammalogy 92: 235-253.

Simon, J.A., J.W. Snodgrass, R.E. Casey, and D.W. Sparling. 2009. Spatial correlates of amphibian use of constructed wetlands in an urban landscape. Landscape Ecology 24: 361-373.

Sinclair, K.E., G.R. Hess, C.E. Mooman, and J.H. Mason. 2005. Mammalian nest predators respond to greenway width, landscape context and habitat structure. Landscape and Urban Planning 71: 277-293.

Snodgrass, J.W., R.E. Casey, D. Joseph, and J.A. Simon. 2008. Microcosm investigations of stormwater pond sediment toxicity to embryonic and larval amphibians: Variation in sensitivity among species. Environmental Pollution 154: 291-297.

Snyder, D.P. 1982. *Tamias striatus*. Mammalian Species 168: 1-8.

Soulé , M.E., D.T. Bolger, A.C. Alberts, J. Wright, M. Sorice, and S. Hill. 1988. Reconstructed dynamics of rapid extinctions of chaparral-requiring birds in urban habitat islands. Conservation Biology 2: 75-92.

Sparling, D.W., J.D. Eisemann, and W. Kuenzel. 2004. Contaminant exposure and effects in red-winged blackbirds inhabiting stormwater retention ponds. Environmental Management 33: 719-729.

Spear, D., and S.L. Chown. 2008. Taxonomic homogenization in ungulates: Patterns and mechanisms at local and global scales. Journal of Biogeography 35: 1962-1975.

Spencer, A.W., and D. Pettus. 1966. Habitat preferences of five sympatric species of long-tailed shrews. Ecology 47: 677-683.

Stanley, T.R., and K.P. Burnham. 1999. A closure test for time-specific capture-recapture data. Environmental and Ecological Statistics 6: 197-209.

Stickel, L.F., and O. Warbach. 1960. Small-mammal populations of a Maryland woodlot, 1949-1954. Ecology 41: 269-286.

Stover, M.R. 2012. Red-winged blackbird reproduction in stormwater ponds in an urban and suburban landscape: Evidence for ecological traps. M.S. Thesis, Towson University, Towson, Maryland.

Swann, D.E., C.C. Hass, D.C. Dalton, and S.A. Wolf. 2004. Infrared-triggered cameras for detecting wildlife: An evaluation and review. Wildlife Society Bulletin 32:357-365.

Swihart, R.K, and P.M. Picone. 1995. Use of woodchuck burrows by small mammals in agricultural habitats. American Midland Naturalist 133: 360-363.

Tsihrintzis, V.A., and R. Hamid. 1997. Modeling and management of urban stormwater runoff quality: A review. Water Resources Management 11: 137-164.

Umetsu, F., and R. Pardini. 2007. Small mammals in a mosaic of forest remnants and anthropogenic habitats- evaluating matrix quality in an Atlantic forest landscape. Landscape Ecology 22: 517-530.

Vitousek, P.M., C.M. D'Antonio, L.L. Loope, M. Rejmanek, and R. Westbrooks. 1997. Introduced species: A significant component of human-caused global change. New Zealand Journal of Ecology 21: 1-16.

Weatherhead, P.J., G. Blouin-Demers, and K.M. Cavey. 2003. Seasonal and prey-size dietary patterns of black ratsnakes (*Elaphe obsoleta obsoleta*). American Midland Naturalist 150: 275-281.

Weatherhead, P.J., G.L.F. Carfagno, J.H. Sperry, J.D. Brawn, and S.K. Robinson. 2010. Linking snake behavior to nest predation in a Midwestern bird community. Ecological Applications 20: 234-241.

Weller, T.J.. 2008. Using occupancy estimation to assess the effectiveness of a regional multiple-species conservation plan: Bats in the Pacific Northwest. Biological Conservation 141: 2279-2289.

Whitaker J.O., Jr., and W.E. Wrigley. 1972. *Napaeozapus insignis*. Mammalian Species 14: 1-6.

White, G.C., and K.P. Burnham. 1999. Program MARK: Survival estimation from populations of marked animals. Bird Study 46 Supplement, 120-138.

White, G.C., K.P. Burnham, and D.R. Anderson. 2002. "Advanced features of Program MARK" in Integrating people and wildlife for a sustainable future: Proceedings of the Second International Wildlife Management Congress, ed R. Fields, Bethesda, MD.

White, P.J.T., and J.T. Kerr. 2007. Human impacts on environment-diversity relationships: Evidence for biotic homogenization from butterfly species richness patterns. Global Ecology and Biogeography 16: 290-299.

Williams, S.E., H. Marsh, and J. Winter. 2002. Spatial scale, species diversity, and habitat structure: Small mammals in Australian tropical rain forest. Ecology 83: 1317-1329.

Williams, A.K., M.J. Ratnaswamy, and R.B. Renken. 2001. Impacts of a flood on small mammal populations of lower Missouri River floodplain forests. American Midland Naturalist 146: 217-221.

Wilson, D.E., F.R. Cole, J.D. Nichols, R. Rudran, and M.S. Foster. 1996. Measuring and monitoring biological diversity: Standard methods for mammals. Smithsonian Institution Press. p 115-116, 157, 170-172.

Wolff, J.O. 1985. The effects of density, food, and interspecific interference on home range size in *Peromyscus leucopus* and *Peromyscus maniculatus*. Canadian Journal of Zoology 63: 2657-2662.

Wolff, J.O. 1993. Why are female small mammals territorial? Oikos 68: 364-370.

Wolff, J.O. and D.M. Cicirello. 1991. Comparative paternal and infanticidal behavior of sympatric white-footed mice (*Peromyscus leucopus noveboracensis*) and deer mice (*P. maniculatus nubiterrae*). Behavioral Ecology 2: 38-45.

Wywialowski, A.P. 1987. Habitat structure and predators: choices and consequences for rodent habitat specialist and generalists. Oecologia 72: 39-45.

Yeh, C.H., and J.W. Labadie. 1997. Multiobjective watershed-level planning of storm water detention systems. Journal of Water Resources Planning and Management 123: 336-343.

Zielinski, W.J., and T. E. Kucera. 1995. American marten, fisher, lynx, and wolverine: Survey methods for their detection. Albany, CA: U.S. Department of Agriculture, Forest Service 157: 67-86.

CURRICULUM VITAE
Caitlin M. Graff

106 East Susquehanna Avenue, #2
Towson, Maryland 21286

(215) 896-7465
cmgraff101@gmail.com

EXPERIENCE:
8/12-12/12 – Assistant Professor, Towson University, Department of Biological Sciences
3/12-Present – Weekend Program Coordinator, Baltimore County Recreation and Parks
8/09–5/12 – Teaching Assistant, Towson University, Department of Biological Sciences
5/09-8/09 – Field Technician, University of Washington
7/08-10/08 – Animal Attendant, Philadelphia Zoo

EDUCATION:
Towson University, Towson, Maryland 21252
M.S. Biology
Date to be conferred: 3/13

Albright College, Reading, Pennsylvania 19604
B.S. Environmental Science
8/06–5/08

Montgomery County Community College, Blue Bell, Pennsylvania 19422
A.A. General Studies
8/04–5/06

HONORS AND AWARDS:
2008 - Albright College Departmental Award in Biology
2007 - Alpha Epsilon Delta National Honor Society

PROFESSIONAL SOCIETIES:
2011-Present - Society of Women Environmental Professionals
2011-Present - The Wildlife Society