



National Capital Region Network 2009 Amphibian Monitoring

Synthesis of 2005-2009 Data

Natural Resource Technical Report NPS/NCRN/NRTR—2011/414



ON THE COVER

Pseudotriton ruber (northern red salamander)

Photograph: courtesy of the USGS Northeast Amphibian Research and Monitoring Initiative

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January 2011

U.S. Department of the Interior
National Park Service
Natural Resource Program Center
Fort Collins, Colorado

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Please cite this publication as:

Campbell Grant, E. H., E. F. Zipkin, and A. B. Brand. 2011. National Capital Region Network 2009 amphibian monitoring: Synthesis of 2005-2009 data. Natural Resource Technical Report. NPS/NCRN/NRTR—2011/414. National Park Service, Fort Collins, Colorado.

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Abstract

This report describes ongoing amphibian monitoring efforts in the National Capital Region Network. Amphibian monitoring was initiated in 2005 and is currently concentrated in Chesapeake and Ohio Canal National Historic Park and Rock Creek Park, with stream (lotic) sampling also occurring in Prince William Forest Park. The objectives of the monitoring program are to develop an efficient long-term sampling design to: (1) describe the current distribution of amphibians and explore factors that may influence occupancy probabilities or distributional patterns, (2) determine if amphibian distributions are changing annually, and if so, explore whether occupancy changes are related to habitat quality and (3) provide information to aid in the generating and testing hypotheses that differentiate among possible causes of long-term changes in the proportion of area occupied among species, habitats, and park areas.

Here, we present analysis from 2005-2009 and describe our findings related to the first two objectives. We discuss our results in the context of the National Capital Region Network's continuing amphibian monitoring program. For the lentic habitats, we detected 12 amphibian species or species complexes in each year of sampling 33 wetlands from the Potomac Gorge area within Chesapeake and Ohio Canal National Historic Park (CHOH). Results from our model indicated only a 23% probability that there were greater than 12 species in the region suggesting that most, if not all species were observed on at least one occasion during the five years of sampling. We found that detection probabilities (the probability of detecting a species, given it is present at a site) were less than one and were strongly influenced by when sampling occurred (i.e., day of the year) but were only mildly variable between years.

We investigated the relationships between wetland species richness and two important limiting variables in amphibian communities--habitat area and hydroperiod. Lentic species richness was generally highest in permanent, followed by semi-permanent, and lowest in temporary wetlands. Total richness (i.e., the average number of species at a wetland that were present in at least one year of sampling) ranged from 2.7 to 11.0 across the set of permanent wetlands (mean: 7.4; median: 9.0), from 3.2 to 10.0 in semi-permanent wetlands (mean: 6.8; median: 7.4) and from 0.3 to 10.0 in temporary wetlands (mean: 2.5; median: 1.4). The mean estimated richness for each year of the survey was variable in each of the wetlands, with some wetlands having fairly constant richness and others experiencing large annual changes. Annual wetland variability in richness varied independently of hydroperiod category. Increasing wetland area generally had a positive effect on species occurrence probabilities but the result was fairly minimal.

For the lotic sampling, we observed four species of stream salamanders in each year of our study: *Desmognathus fuscus*, *Eurycea bislineata*, *E. longicauda*, and *Pseudotriton ruber*. Detection probabilities varied among these species and years. In general, initial occupancy estimates were higher for stream transects in CHOH compared to ROCR, except for *E. bislineata* which had high occupancy probabilities in both parks. *D. fuscus* and *P. ruber* had higher occupancy probabilities at transects near the stream headwaters, but our *a priori* hypothesis that proximity of the stream origin to the park boundary or road would result in lower occupancy probabilities was not well supported for any of the 3 species. We also expected models specifying no turnover (i.e., $\gamma = \epsilon = 0$) to be among the top models for all species, but this was

not the case. However, the estimated rate parameters for turnover were small, and thus their effect on occupancy was negligible. As with our lentic sampling, more years of data, encompassing both wet and dry years, are needed to further elucidate the potential relationship between spatial covariates and site occupancy and related rate parameters for stream salamander species. Finally, we are investigating how dry years change the distribution of a population, which will inform the monitoring program design and future studies of stream salamander populations.

Introduction

The National Capital Region Network (NCRN) Inventory and Monitoring (I&M) Program has identified amphibians as a priority taxonomic group for its monitoring program. The goals of this program are to document at least 90% of the amphibian species in its parks, and to determine whether the integrity and status of amphibian populations are changing over time. As of 2004, all NCRN parks had completed amphibian inventories (Shawn Carter, National Park Service, *personal communication*), and planning for monitoring efforts were initiated. The objectives of the monitoring program are to implement a long-term sampling design, in cooperation with park service biologists and managers, to: (1) describe the annual distribution of amphibians (i.e. estimate initial occupancy probabilities for targeted species) and explore factors that may influence occupancy probabilities or distributional patterns, (2) determine if amphibian distributions are changing annually, and if so, explore whether occupancy changes are related to habitat quality, paying special attention to factors associated with increased urbanization in the region, and (3) provide information to aid in generating and testing hypotheses that differentiate among possible causes of long-term changes in the proportion of area occupied among species, habitats, and park areas.

In 2005 an NCRN-supported pilot study was initiated by USGS Northeast Amphibian Research and Monitoring (NEARMI) personnel to collect detection/nondetection data for amphibians located in terrestrial, lotic (stream) and lentic (wetland) habitats at two parks in the National Capital Region Network (Chesapeake and Ohio Canal National Historical Park, Rock Creek Park). Multiple survey techniques were employed and data from each survey method was analyzed by USGS NEARMI researchers and evaluated by both USGS and NPS personnel (see National Park Service 2007). Based on this evaluation, a decision was made to focus monitoring efforts on amphibians that occupy lentic and lotic habitats (National Park Service 2007). This report details these ongoing monitoring efforts and describes findings based on analysis of the current available data (2005-2009).

Methods

Our study focused on wetlands (lentic habitat) and first-order streams (lotic habitat) in the Chesapeake & Ohio Canal National Historical Park (CHOH), and lotic habitats only in Rock Creek Park (ROCR) and Prince William Forest Park (PRWI). Both CHOH and ROCR are located within the urbanized Washington, DC metropolitan area, and PRWI is located near Triangle, Virginia. Our study at CHOH was conducted in the urbanized, southern section of the park, known as the Potomac Gorge area (Figure 1), while the studies at ROCR and PRWI involved a random sub-sample of all mapped first-order streams which originated within the parks (Figure 2, 3).

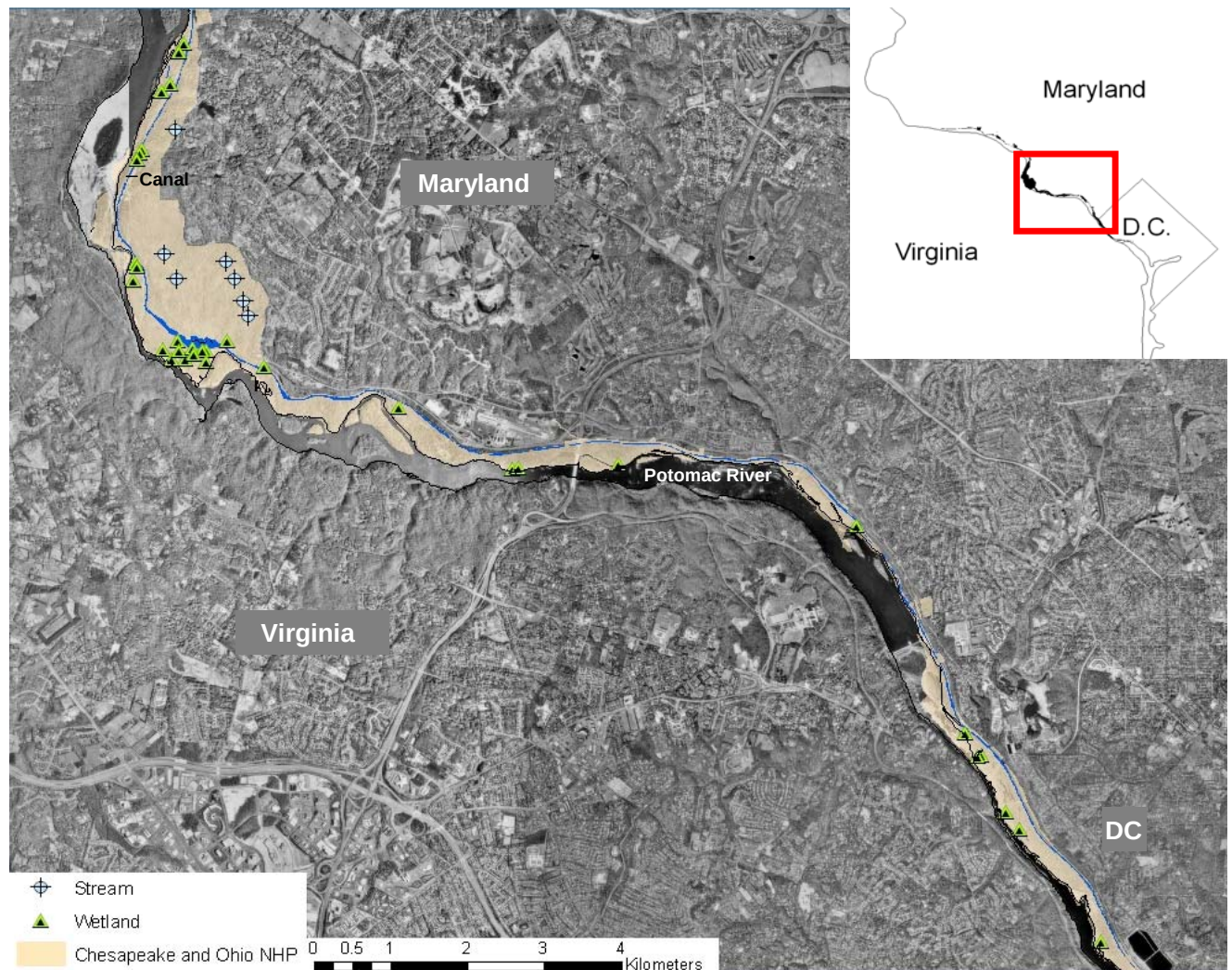


Figure 1. Study area: location of the 33 wetlands and 7 streams surveyed for amphibians in the Chesapeake and Ohio Canal National Historical Park (2005-2009).

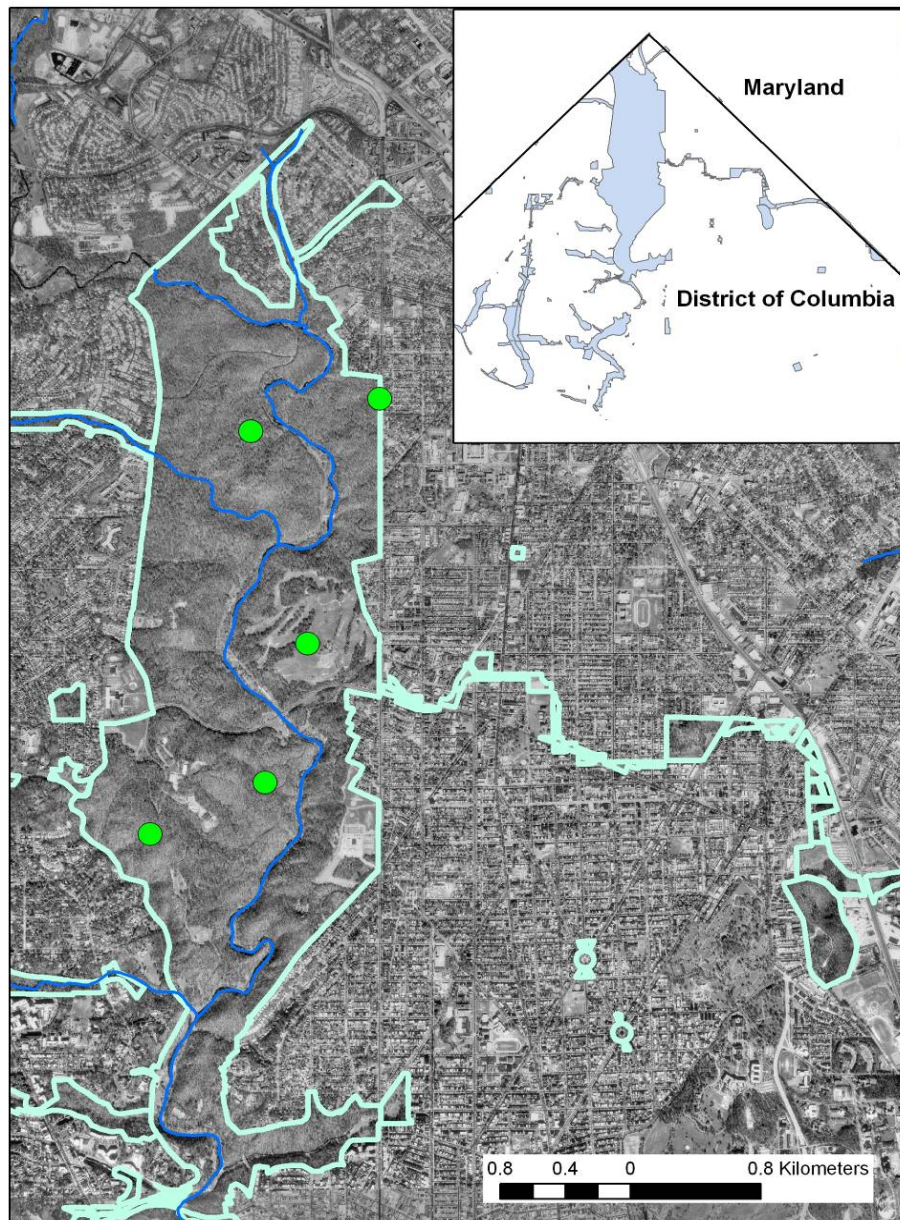


Figure 2. Study area: location of the 5 streams surveyed for amphibians in Rock Creek Park (2005-2009).

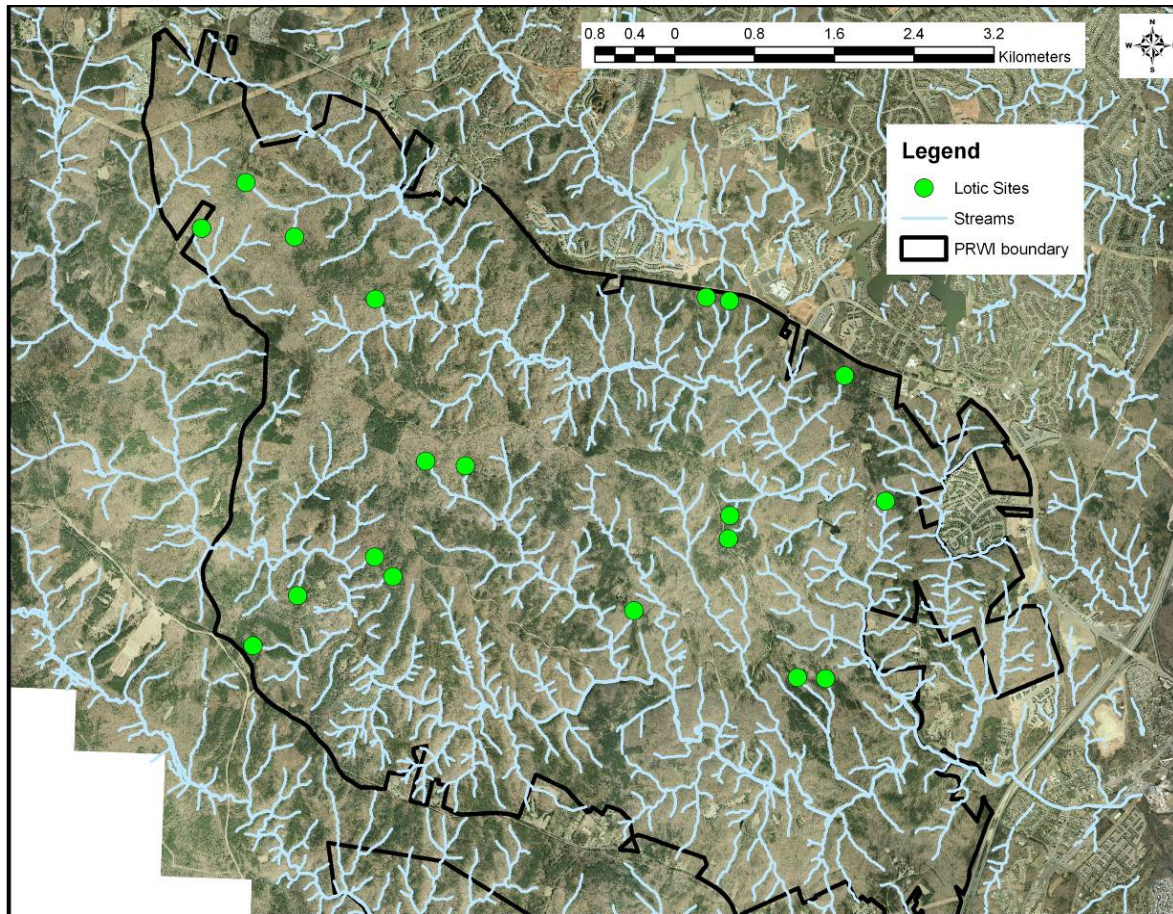


Figure 3. Study area: location of the 19 streams surveyed for amphibians in Prince William Forest Park (2005-2009).

Lentic Habitats

Site Selection

Our sample frame consisted of all known (mapped) wetlands in the Potomac Gorge area of CHOH. For the purposes of this study we limit our definition of wetlands to those that are isolated, palustrine, and less than 0.4 hectare in area. These constraints are based primarily on sampling logistics, but also reflect the primary breeding habitats of lentic amphibians. Our sample frame consisted of 169 wetlands of which we selected a random sub-sample of 33 wetlands (representing 15 temporary, 10 semi-permanent, and 8 permanent hydroperiods).

Field Methods

The 33 wetlands were visited on four occasions from March-July for five years (2005- 2009; 20 visits to each site over the study duration). Two detection methods were used to document the presence of amphibian species during these visits: visual encounter or dip-net surveys. We conducted visual encounter surveys early in the year to target amphibian egg masses and breeding adults. Observers walked along the perimeter of the wetland, recording all life phases of

amphibian species observed in the wetland, or under terrestrial cover (objects >6 cm) within a meter of the wetland edge. Dip-net surveys were conducted later in the season to target larval amphibians. Every sampling visit was conducted by two independent observers: observers began their surveys at opposite sides of a wetland site, navigated around the pond in the same direction, and did not communicate during the survey.

Several covariates were also measured during the surveys, and we focused on the effects of wetland hydroperiod and wetland surface area (m²) on amphibian occupancy. Hydroperiod is the amount of time that a wetland holds water. Wetlands were classified into one of three hydroperiod categories: temporary (dry every year), semi-permanent (dry in some years) or permanent (usually hold water for multiple years) using GIS maps provided by the NCRN I&M program. For more details on the sampling process refer to Mattfeldt et al. (2009).

Analytical Methods

We built a hierarchical, multi-species occurrence model to estimate occupancy dynamics as well as community-level richness for amphibians within the National Parks we monitored. An advantage of the hierarchical modeling framework over typical species richness analyses is that it accounts for both species-level effects as well as aggregated effects of habitat on the community as a whole (Kéry & Royle 2008, 2009). This leads to a more efficient use of available data and increased precision in occupancy estimates, especially for infrequently observed species. Results from 2005-2009 data will be used to make predictions for the upcoming field season (2010), which will allow us to validate the accuracy of the model to forecast amphibian species occurrences.

Our hierarchical model summarizes community-level attributes through species-specific estimates of occupancy, detection, and colonization (Dorazio and Royle 2005, Dorazio et al. 2006, Royle and Dorazio 2008, Kéry et al. 2009). Under this analysis framework, occurrence models for all individual species are linked together in a multi-level, or hierarchical model placing them in a context of related community-level responses to environmental factors (Dorazio et al. 2006, Kéry and Royle 2008). Sharing information across species leads to increased precision of parameter estimates and enhanced understanding of both species-specific occupancy dynamics as well as the composite dynamics of the entire community. This approach is especially useful for infrequently observed species for which estimates of occupancy, colonization, and detection would otherwise be unattainable (Mattfeldt et al. 2009, Zipkin et al. 2009).

The model is based on the survey-specific detection/non-detection records of all n species. The observations, $x_{j,i,t,k}$, denote detection ($x=1$) or non-detection ($x=0$) of species i ($1,2,\dots,n$) at site j ($1,2,\dots,33$) in year t ($1,2,\dots,5$) during sampling occasion k ($1,2,\dots,8$). We assumed that occupancy and colonization were species-specific processes, dependent on the habitat characteristics of individual wetlands. As in previous applications of this method, we assumed that occupancy for species i at site j in year t (denoted $z_{j,i,t}$) was a latent variable (i.e., imperfectly observed) such that $z_{j,i,t} = 1$ when species i was present and zero otherwise. If a species was detected at a specific wetland in a given year, then it could be determined that the species was in fact present.

However, if a species was not observed, it could be because either the species was present and went undetected or the species was truly absent. Thus, the repeated sampling protocol ($k > 1$) was used to differentiate non-detection from true absences by estimating the detection probability for each species during sampling occasions.

We specified the species-specific occurrence model by assuming that occupancy was a Bernoulli random variable, $z_{i,j,t} \sim \text{Bern}(\psi_{i,j,t})$, with probability $\psi_{i,j,t}$ for species i in site j , during year t .

The probability that a given species was present at a specific location was assumed to be dependent upon the area and hydroperiod of the wetland. We included a continuous covariate on wetland area, where area was standardized to have a mean of zero with a standard deviation of one. We also included two additional covariates to estimate the deviation (in relation to temporary wetlands) in occurrence dynamics for species in semi-permanent (a data vector, $semi$ in which $semi_j = 1$ if site j is a semi-permanent wetland and zero otherwise) and permanent wetlands (a similarly defined data vector, $perm$, indicating if a wetland j was a permanent). For the first year of sampling ($t=1$), we modeled $\psi_{i,j,1}$ using a logit-link function (Kéry and Royle 2009, Russell et al. 2009):

$$\text{logit}(\psi_{i,j,1}) = u_i + \alpha 1_i semi_j + \alpha 2_i perm_j + \alpha 3_i area_j.$$

The parameter u_i is the probability of occupancy (on the logit scale) for species i in temporary wetlands with mean area. The parameters $\alpha 1_i$ and $\alpha 2_i$ are the difference in estimated occupancy in semi-permanent and permanent wetlands, respectively. The coefficient $\alpha 3_i$ is the effect of area on occupancy for species i , respectively. For $t=2, \dots, 5$, we modified the model, such that $\psi_{i,j,t}$ for species i in site j was also dependent upon whether or not that species was present the previous year (Ruiz-Gutierrez et al. in press, Kéry et al. 2009):

$$\text{logit}(\psi_{i,j,t \geq 2}) = u_i z_{j,i,t-1} + w_i (1 - z_{j,i,t-1}) + \alpha 1_i semi_j + \alpha 2_i perm_j + \alpha 3_i area_j$$

Thus w_i is the occupancy probability (on the logit scale) for species i when it was not present in the previous year ($z_{j,i,t-1} = 0$). When a species was in fact present in the previous year ($z_{j,i,t-1} = 1$) then the probability of occupancy simplifies to the original model for the first year of sampling. We thus define the probability of occupancy (u_i) as a combination of initial occupancy (the probability that species i was present at site j in the first year of sampling), and continued occupancy (the probability that species i was present at site j given it was present the previous year) (Royle and Kéry 2007). The probability of colonization (w_i on the logit scale) is defined as the probability that species i was present at site j in year t , given that it was absent at site j in year $t-1$.

The detection model was similarly assumed to be a Bernoulli random variable,

$$x_{i,j,t,k} \sim \text{Bern}(p_{i,j,t,k} \cdot z_{i,j,t}), \text{ where } p_{i,j,t,k} \text{ is the detection probability of species } i \text{ for the } k^{\text{th}}$$

sampling period in year t at site j , given that species i was in fact present at site j . When species i was not present, detection was a fixed zero since $z_{i,j,t} = 0$. We assumed that species identity, sampling day (i.e., day of year: 1-365), and year could all affect detection. We included both linear ($\beta 1_i$) and quadratic ($\beta 2_i$) effects of sample day as species specific effects on detection. We also included a year effect, which we assumed to be constant across species, by estimating a potential shift in detection for each year as measured in relation to the first year, 2005 in which $t=1$ (e.g., the covariate $\beta 3$ is the estimated effect of $y2006_t$, a vector of length t , with a one for $t=2$, year 2006, and a zero otherwise).

$$\text{logit}(p_{i,j,t,k}) = v_i + \beta 1_i \text{day}_{j,k,t} + \beta 2_i \text{day}_{j,k,t}^2 + \beta 3 y2006_t + \beta 4 y2007_t + \beta 5 y2008_t + \beta 6 y2009_t$$

In the community hierarchical level of the model, the species-specific parameters were linked together through a common distribution (Royle and Dorazio 2008). Specifically, species were linked together by assuming that their parameter values for occupancy (as well as colonization and detection) were drawn from a common distribution dependent on habitat type. For example, we assumed that $u_i \sim N(\mu_u, \sigma_u)$ where μ_u is mean occupancy across all species and σ_u is the standard deviation (among species). We similarly defined the hyper-parameters (the mean and standard deviations – across species) of occupancy, colonization, detection, and each of the covariates.

Bayesian analysis of the model was carried out using the method of data augmentation described in Royle et al. (2007) and Kéry and Royle (2009), which allows for estimation of the number of species in the community that were unobserved (either locally or never detected) during the sampling process. Analysis by data augmentation assumes a uniform $(0, M)$ prior for N , the “true” species richness, where M is a fixed constant chosen to be much greater than the number of observed species (n) and such that the resulting posterior distribution is not truncated.

Implementation of the model with a uniform prior is done by augmenting the data set with $M-n$ all-zero encounter histories. Then the model for the augmented data set is a zero-inflated version of a model where the actual number of species in the community (N) is known (Royle et al. 2007, Kéry and Royle 2009). The occurrence process is modified so that

$$z(i, j) \sim \text{Bern}(\psi_{i,j,t} \cdot w_i) \text{ where } w_i \sim \text{Bern}(\Omega) \text{ for species}$$

$i = 1, 2, \dots, n, n+1, n+2, \dots, N, N+1, N+2, \dots, M$. The interpretation of this modified occurrence process is that if $w_i = 1$ (corresponding to species that were observed or that were unobserved but available for sampling), the probability of occurrence is simply $\psi_{i,j,t}$. If $w_i = 0$ (indicating that a species was unavailable for sampling), then occurrence is zero by definition (i.e., a structural zero). The model is now modified to estimate the parameter Ω . The value of M need only be large enough to not truncate the posterior distribution of N , which can be assessed by running short initial trials. Interpretation of the posterior of N must be done cautiously. It is not necessarily the number of species that occur in a particular landscape; rather, it is equivalent to the asymptote of a species accumulation curve (Kéry and Royle 2009). In this context, N is the

intrinsic amphibian carrying capacity of the park, suggesting the possible number of species that could occur in regions with similar wetland characteristics and configuration.

We analyzed the model using the programs R and WinBUGS (Spiegelhalter et al. 2003). We used independent, diffuse proper prior distributions for the community-level hyper-parameters. We ran three chains of length 5,000 after a burn-in of 5,000 and thinned the posterior chains by 5. Convergence was assessed using the R-hat statistic, which examines the variance ratio of the MCMC algorithm within and between chains across iterations (Gelman and Hill 2007).

Lotic Habitats

Site Selection

Streams were chosen from all known headwater, first order streams in each of three parks in the National Capital Region Network [Chesapeake and Ohio Canal National Historical Park (CHOH), Rock Creek Park (ROCR) and Prince William Forest Park (PRWI)]. Based on GIS-mapped streams (provided in digital form by NPS for CHOH and PRWI and mapped in the field in 2005 for ROCR), streams were classified as having confluent first order streams of at least 150 m in length ('branched'), or having a single first order branch ('unbranched') before meeting a higher order stream. We then attempted to randomly select an equal number of branched and unbranched streams from the set of mapped streams from each park; field surveys revealed that only CHOH and PRWI had branched streams according to our definition. Seven streams were randomly chosen at CHOH (Figure 1), five streams at ROCR (Figure 2), and 53 streams at PWRI (Figure 3). We surveyed PRWI streams in 2006, and at that time only 19 streams had sufficient surface water flow to warrant sampling for stream salamanders. These 19 streams were sampled in 2006, 2008, and 2009; all CHOH and ROCR streams were sampled annually from 2005-2009.

Field Methods

We surveyed streams for salamanders in the genera *Desmognathus*, *Eurycea*, and *Pseudotriton*. Each stream was surveyed using two pairs of 15x3 m transects (surveying 2 m on the bank and 1 m in the water). The first transect pair was located at the stream origin, with the second pair located 100 m from the end of the first pair along the same stream (145 m from the stream origin). Each pair of transects (i.e., a 'site') was visited twice (once each in June and July) by two observers. We found that searching the leaf litter along with turning cover objects (e.g., rocks and logs) is suitable for detecting larval *P. ruber*, which has a low detection probability when turning cover alone (Mattfeldt and Grant 2007). We included detections of larval *P. ruber* from 2005 leaf litterbag surveys in the analysis presented here: surveys in 2006-2009 incorporate leaf litter searches every 0.5 m along the transect, in addition to turning cover objects (see National Park Service 2007 SOP#6 for details). We conducted two passes of each transect on each visit, and summed the detections for our analysis. For each site, we recorded covariates that were likely to influence stream salamander occupancy and rate parameter probabilities including: transect location (distinguishing whether the transect pair was located at the stream origin or not) and park (CHOH, PRWI, or ROCR).

Analytical Methods

We performed our analyses for three species, *Desmognathus fuscus*, *Eurycea bislineata* and *Pseudotriton ruber*. We use the multi-season modeling approach of MacKenzie et al. (2003, 2006) and analyzed species-specific data from paired transects on 31 streams (seven streams at CHOH, nineteen streams at PRWI and five streams at ROCR) from 2005 – 2009. Each year was considered a 'season', with two sampling occasions (June and July) within each season. We assumed that each site was closed to changes in occupancy within a season (i.e., no local colonization or extinction of a species), but we investigated the possibility of local colonization or extinction between seasons (years).

Our modeling allowed for different initial occupancy at the three parks. We also investigated whether γ or ε were zero (γ only), nonzero but constant over years, or varied by year. We expected *a priori* models with no colonization (i.e. $\gamma = 0$) to have the most support, given assumptions of population stability of stream salamanders in natural ecosystems by previous authors (e.g., Davic and Welsh 2004, Hairston et al. 1993).

Finally, using the most general occupancy and rate parameter structure for each species (i.e. the model with the most parameters), we modeled detection probability as a function of transect location (at the headwaters vs. 150 m from the stream origin) and season (year) for all species.

Results

Lentic Habitats

Twelve species were observed during sampling. The model estimated only a 23% probability that there were greater than 12 species in the region suggesting that most, if not all, species were observed on at least one occasion during the five years of sampling. Occupancy probabilities were variable across species and habitat type (i.e., temporary, semi-permanent, and permanent wetlands; Table 1A). Eight species had highest occupancies in permanent wetlands and four had highest occupancy probabilities in semi-permanent wetlands. Colonization probabilities also varied widely, suggesting that some species are more mobile than others (Table 1B). As expected, all species had lower colonization probabilities compared to continued occupancy (i.e., a species was less likely to use a wetland if it had not been there in the previous year).

Table 1. Species-species estimates of occupancy and colonization in temporary, semi-permanent and permanent wetland in CHOH.

A.

Species	Occupancy					
	Temporary		Semi-permanent		Permanent	
	mean	sd	mean	sd	mean	sd
<i>Ambystoma maculatum</i>	0.423	0.075	0.717	0.067	0.670	0.094
<i>Ambystoma opacum</i>	0.063	0.034	0.179	0.080	0.155	0.085
<i>Bufo americanus/fowleri</i>	0.186	0.054	0.490	0.081	0.659	0.082
<i>Hemidactylium scutatum</i>	0.057	0.033	0.185	0.083	0.094	0.076
<i>Hyla versicolor/chrysoscelis</i>	0.106	0.046	0.267	0.082	0.398	0.099
<i>Notophthalmus viridescens</i>	0.070	0.039	0.183	0.083	0.683	0.096
<i>Pseudacris crucifer</i>	0.143	0.047	0.432	0.083	0.464	0.097
<i>Rana catesbeiana</i>	0.224	0.063	0.571	0.077	0.785	0.069
<i>Rana clamitans</i>	0.391	0.072	0.731	0.059	0.836	0.057
<i>Rana palustris</i>	0.182	0.060	0.478	0.090	0.707	0.084
<i>Rana sphenoccephala</i>	0.095	0.038	0.298	0.081	0.538	0.090
<i>Rana sylvatica</i>	0.384	0.075	0.736	0.063	0.496	0.119

B.

Species	Colonization					
	Temporary		Semi-permanent		Permanent	
	mean	sd	mean	sd	mean	sd
<i>Ambystoma maculatum</i>	0.026	0.014	0.085	0.041	0.072	0.040
<i>Ambystoma opacum</i>	0.010	0.007	0.032	0.020	0.028	0.022
<i>Bufo americanus/fowleri</i>	0.091	0.033	0.297	0.075	0.460	0.110
<i>Hemidactylium scutatum</i>	0.004	0.004	0.016	0.015	0.007	0.009
<i>Hyla versicolor/chrysoscelis</i>	0.066	0.028	0.179	0.055	0.287	0.089
<i>Notophthalmus viridescens</i>	0.003	0.003	0.010	0.009	0.090	0.060
<i>Pseudacris crucifer</i>	0.052	0.021	0.200	0.058	0.223	0.072
<i>Rana catesbeiana</i>	0.048	0.022	0.189	0.065	0.392	0.123
<i>Rana clamitans</i>	0.092	0.034	0.298	0.085	0.448	0.122
<i>Rana palustris</i>	0.017	0.010	0.067	0.035	0.160	0.078
<i>Rana sphenoccephala</i>	0.046	0.019	0.165	0.051	0.355	0.096
<i>Rana sylvatica</i>	0.044	0.020	0.171	0.065	0.070	0.036

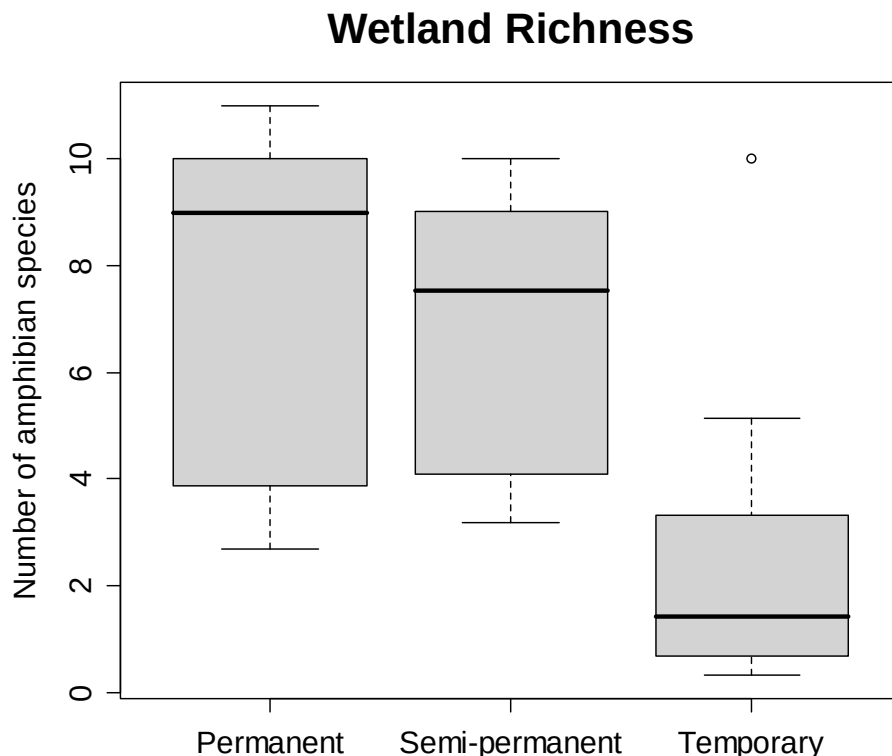


Figure 4. Variation in species richness (total number of species observed over all sampling years) in individual wetlands by hydroperiod: permanent, semi-permanent, and temporary. The black line shows the median richness, the grey box represents the inter-quartile range, and the whiskers extend to the 95% interior range.

Species specific detection probabilities were strongly influenced by sampling day but were only mildly variable between years. Most species had wide ranges in their probability of detection over the sampling season, with some species increasing in detectability over the season and others decreasing. Yet, detection at each species peak, was fairly high (mean was greater than 0.60 for all species). These results suggest that the extended time frame of the sampling season (March through July) is necessary to capture the occupancy dynamics of each species. If sampling had been limited to a shorter time frame, some species may have been missed, which would bias estimates of occurrences.

Species richness was generally highest in permanent, followed by semi-permanent, and lowest in temporary wetlands (Figure 4). Total richness (i.e., the average number of species at a wetland that were present in at least one year sampling) ranged from 2.7 to 11.0 in permanent wetlands (mean: 7.4; median: 9.0), from 3.2 to 10.0 in semi-permanent wetlands (mean: 6.8; median: 7.4) and from 0.3 to 10.0 in temporary wetlands (mean: 2.5; median: 1.4). The mean estimated richness for each year of the survey was variable, with some wetlands having fairly constant richness and others experiencing large changes (Table 2).

Table 2. Estimated mean number of species by site, 2005-09.

Site	2005	2006	2007	2008	2009
Carderock Lot 3	4	2.05	2.23	2	5.01
CHW05	1.01	1.04	2.04	0.1	1.05
CHW06	1	1.01	0.23	0.02	0.01
CHW09	0.03	0.21	0.3	1	0.02
DC045	7.02	6.12	8.03	7.01	7.08
GoBack	0.01	0.56	1.41	0.18	1.02
MD014	0.09	0.88	0.54	0.72	0.04
MD014A	0.08	0.76	0.47	0.63	0.04
MD025	6.04	6.04	7	4.09	6.08
MD028	6.03	4.08	2.08	2.14	6.09
MD039	0.06	0.59	0.4	0.2	0.04
MD041	0.06	0.61	0.42	0.21	0.02
MD042	1.06	0.66	0.44	1.04	0.03
MD052	4.01	3.12	1.48	0	1.03
MD053	7	4.02	6.01	3	5.01
MD115	2.01	0.06	1.26	3	1.01
MD118	0.04	0.06	0.26	0.02	0.01
MD119B	7	3.07	7.01	8	9
MD128	6.08	8.08	6.11	8	9.03
MD129	8.07	7.24	9.01	8.04	7.25
MD143	6.1	4.14	3.11	9	6.26
MD144	8.02	10	6.02	8.02	5.13
MD145	6.01	3.05	3.06	3.01	3.07
MD146	7.02	5.03	6.04	3.01	5.03
MD150	3.05	4.06	4.02	2.02	3.12
MD155	6.01	4.04	2.06	4	4.08
MD159	1.02	0.32	0.24	0.11	0.01
MD204	2.06	2.06	2.04	0.03	2.11
MD205	2.04	3.23	2.04	2.02	1.25
MD211	2.01	0.23	1.02	2.02	2.03
MD220	4.04	5.08	3.04	3.02	3.16
MD240	0.02	0.32	2.05	2.01	0.11
MD256X	0.01	0.52	0.34	0	0.02

Lotic Habitats

We observed four species of stream salamanders in each year of our study: *Desmognathus fuscus*, *Eurycea bislineata*, *E. longicauda*, and *Pseudotriton ruber*. We did not encounter *E. longicauda* with enough frequency to include in the formal analysis. Due to the time period of sampling, we did not detect all age classes of all species, though all species were available for detection. Therefore, we combined detections among all age class for our multi-season occupancy analysis of *D. fuscus*, *E. bislineata* and *P. ruber*.

The best model for *D. fuscus* included occupancy as a function of park and colonization as a function of park or year (model uncertainty). The best model for *P. ruber* included a constant occupancy, independent of park. The best model for *E. bislineata* included occupancy and colonization as a function of park.

Detection probability for *D. fuscus* was variable among years (estimates from top model: $\hat{p}^{2005} = 0.68 + 0.07$; $\hat{p}^{2006} = 0.71 + 0.05$; $\hat{p}^{2007} = 0.82 + 0.05$; $\hat{p}^{2008} = 0.71 + 0.06$; $\hat{p}^{2009} = 0.52 + 0.07$). There was no consistent set of covariates which were related to colonization or extinction probabilities, though estimates across the top model set suggest that colonization was higher than extinction over our study period in ROCR, and lower than or equal to extinction (essentially zero) in CHOH and PRWI. The estimates for our rate parameters seem to suggest little change in occupancy at PRWI and CHOH, but a small increase in ROCR occupancy over the study period (Table 3).

Table 3. Estimates for initial occupancy (2005), extinction and colonization rates for three species of stream salamander in CHOH, PRWI and ROCR, from 2005-2009. Initial occupancy (2005) was modeled as park-specific, and year-specific annual estimates of occupancy were calculated as the balance of extinctions and colonizations, given occupancy in the previous year, separately for each park.

Rock Creek Park

Species	$\hat{\psi}_{2005}$ (SE)	$\hat{\psi}_{2006}$ (SE)	$\hat{\psi}_{2007}$ (SE)	$\hat{\psi}_{2008}$ (SE)	$\hat{\psi}_{2009}$ (SE)	$\hat{\epsilon}$ (SE)	$\hat{\gamma}$ (SE)
<i>Desmognathus fuscus</i>	0.40 (0.22)	0.49 (0.18)	0.56 (0.17)	0.61 (0.17)	0.65 (0.18)	0.07 (0.04)	0.20 (0.12)
<i>Pseudotriton ruber</i>	0.53 (0.15)	0.55 (0.12)	0.58 (0.09)	0.60 (0.09)	0.63 (0.09)	0.0 (-)	0.06 (0.08)
<i>Eurycea bislineata</i>	1.0 (0.0)	0.94 (0.03)	0.95 (0.02)	0.95 (0.02)	0.95 (0.02)	0.06 (0.03)	1.0 (-)

C&O Canal NHP

Species	$\hat{\psi}_{2005}$ (SE)	$\hat{\psi}_{2006}$ (SE)	$\hat{\psi}_{2007}$ (SE)	$\hat{\psi}_{2008}$ (SE)	$\hat{\psi}_{2009}$ (SE)	$\hat{\epsilon}$ (SE)	$\hat{\gamma}$ (SE)
<i>Desmognathus fuscus</i>	0.89 (0.10)	0.83 (0.10)	0.77 (0.11)	0.72 (0.12)	0.68 (0.13)	0.07 (0.04)	0.0 (0.0)
<i>Pseudotriton ruber</i>	0.53 (0.15)	0.55 (0.12)	0.58 (0.09)	0.60 (0.09)	0.63 (0.09)	0.0 (-)	0.06 (0.08)
<i>Eurycea bislineata</i>	0.77 (0.14)	0.73 (0.13)	0.69 (0.13)	0.66 (0.13)	0.62 (0.13)	0.06 (0.03)	0.0 (0.0)

Prince William Forest Park

Species	$\hat{\psi}_{2005}$ (SE)	$\hat{\psi}_{2006}$ (SE)	$\hat{\psi}_{2007}$ (SE)	$\hat{\psi}_{2008}$ (SE)	$\hat{\psi}_{2009}$ (SE)	$\hat{\epsilon}$ (SE)	$\hat{\gamma}$ (SE)
<i>Desmognathus fuscus</i>	0.11 (0.08)	0.11 (0.07)	0.10 (0.07)	0.09 (0.06)	0.09 (0.06)	0.07 (0.04)	0.0 (0.0)
<i>Pseudotriton ruber</i>	0.53 (0.15)	0.55 (0.12)	0.58 (0.09)	0.60 (0.09)	0.63 (0.09)	0.0 (0.0)	0.06 (0.08)
<i>Eurycea bislineata</i>	1.0 (0.09)	1.0 (0.0)	1.0 (0.0)	1.0 (0.0)	1.0 (0.0)	0.06 (0.03)	1.0 (-)

Detection probability for *P. ruber* was similarly variable among years (estimate from top model: $\hat{p}_{2005} = 0.14 + 0.94$; $\hat{p}_{2006} = 0.47 + 2.53$; $\hat{p}_{2007} = 0.29 + 1.74$; $\hat{p}_{2008} = 0.26 + 1.79$; $\hat{p}_{2009} = 0.17 + 1.28$; standard error estimates were proportionately large for this model). Initial occupancy was similar across the three parks (Table 3). Extinction probability estimates were essentially zero, and colonization probability estimates were small with proportionately large standard errors. As with *D. fuscus*, the estimates for our rate parameters would suggest little overall change in occupancy over the study period.

Detection probability for *E. bislineata* was also variable among years (estimate from top model: $\hat{p}_{2005} = 0.54 + 0.07$; $\hat{p}_{2006} = 0.83 + 0.03$; $\hat{p}_{2007} = 0.79 + 0.05$; $\hat{p}_{2008} = 0.72 + 0.04$; $\hat{p}_{2009} = 0.68 + 0.03$).

2009 = $0.63 + 0.04$). In general this species had high occupancy probabilities across all three parks (Table 3). Because of high initial occupancy estimates and the estimates for our rate parameters, our results suggest little overall change in *E. bislineata* occupancy over the five years (as with *D. fuscus* and *P. ruber*).

Discussion

Lentic Habitats

Hydroperiod is one of the primary local habitat characteristics associated with amphibian presence or species richness (Skelly et al. 1999, Babbitt et al. 2003, Church 2008). Our analysis indicates the pattern also holds for pond-breeding amphibians found in CHOH. Our results suggest that wetland hydroperiod is a limiting factor in determining the distribution of many of the wetland breeding amphibians found at CHOH (Table 1, Figure 2). The hydroperiod of a wetland may vary due to changes outside of the park (watershed modifications, water diversion, or global climate change) or changes within the park (i.e. changes in the canal water level). All species experienced some annual turnover (Table 1B), indicating that the same sites are not occupied each year. Using only naïve or implicit estimates of occupancy would suggest little change in these wetland systems and would mask important changes in the distribution of these species among wetland sites (MacKenzie et al. 2006). Given our relatively small sample size it will likely require more than five years of data to discern factors that influence rate parameters (Bailey et al. 2007, Mackenzie et al. 2006, Field et al. 2007, Mattfeldt et al. 2009).

Future Directions: Lentic habitat

Wetland hydrology affects habitat availability for pond-breeding amphibians. It may be beneficial in the future to simultaneously model both habitat availability, then conditional on available habitat, estimate amphibian occupancy probabilities. Developing this type of joint model is a current area of research by J. Nichols of USGS and D. MacKenzie of Proteus Wildlife Research Consultants (personal communication). We anticipate being able to apply these types of models to lentic data collected at CHOH in the near future.

Lotic Habitats

Although we expected models specifying no turnover (i.e., $\gamma = \varepsilon = 0$) to be among the top models (i.e., $\Delta AIC < 2.0$) for all species, we found little support for this expectation for any species, as models with $\gamma = 0$ and/or $\varepsilon = 0$ had low Akaike weights relative to other models. Both extinction and colonization were estimated near zero for all species in all parks, except *Desmognathus fuscus* in Rock Creek Park. Across the Mid Atlantic region, 2006 - 2009 were abnormally dry years, and this may have contributed to local extinctions by reducing the amount of suitable habitat available to stream salamander populations. More years of data, encompassing both wet and dry years, are needed to further elucidate the potential relationship between spatial covariates and site occupancy and related rate parameters. Models which can incorporate dynamics of habitat availability and occupancy are forthcoming (J. Nichols and D. MacKenzie, *personal communication*).

Future Directions: Lotic habitat

Stream salamander populations are assumed to be stable in undisturbed systems (Hairston et al. 1993). There is evidence that large scale, permanent land use change may be related to

population declines of some stream salamander species (Price et al. 2006). Further, theoretical (e.g., Fagan et al. 2010, Grant *in press*) and empirical (Lowe and Bolger 2002, Grant et al. 2009) work finds that population stability in stream systems may be related to the complexity of the stream network (i.e., the number and location of intersecting stream reaches). This would suggest that management for populations of animals living in stream networks must be considered at the catchment scale, instead of most historic stream management approaches which focus on the stream reach scale (Hassett et al. 2005). Future investigation within the National Capital Region , combined with other research efforts in the Northeastern United States, will allow us to investigate these relationships more fully. Further, as we note for both the lotic and lentic data, additional years of data collection will allow us to improve these estimates, and better discriminate among covariates which may affect these rate parameters.

Conclusions

Lentic Habitats

We recommend continuing to survey wetland habitats each year. In this preliminary study (2005-2009), the estimates of occupancy probabilities over time were relatively constant; however, the occupancy state did change among wetlands. In other words, some sites were colonized and others failed to support breeding and/or foraging activity (i.e. went locally extinct; Mattfeldt et al. 2009). At this point we do not know if extinction and colonization probabilities are constant over time or vary widely between years due to environmental conditions (i.e. rainfall, hydroperiod) or changes in these conditions due to urbanization. We note that more sites (wetlands) are needed to reduce the uncertainty among models, and to improve estimates of the rate parameters. In order to expand sampling into additional parks in the National Capital Region, a complete map of lentic habitats is needed for each park.

Lotic Habitats

We recommend continued surveying for lotic amphibians, to assess whether rate parameters are constant (or vary over time) and to elucidate the relationships between occupancy, rate parameters, and site-specific covariates. We note that more sites (streams) are needed to reduce the uncertainty among models, and to improve estimates of the rate parameters. One major limitation in increasing the current number of streams monitored is obtaining accurate maps of small habitat features, including headwater streams and tributaries, for parks in the National Capital Region (NCR). Such a map is needed for expansion into areas such as Catoclin Mountain Park and one or more of the historic parks in the NCR.

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NPS 800/106410, January 2011

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