Cities and the Environment (CATE)

Volume 8
Issue 1 *Urban Wildlife Research in Support of Conservation Management*

Article 4

April 2015

# A Case Study of Urban Streamside Salamander Persistence in Staten Island, NY

Ellen Pehek
*Natural Resources Group. New York City Department of Parks & Recreation*, ellen.pehek@parks.nyc.gov

Susan C. Stanley
*Natural Resources Group. New York City Department of Parks & Recreation*, susan.stanley@parks.nyc.gov

## A Case Study of Urban Streamside Salamander Persistence in Staten Island, NY

We monitored salamander populations in four stream segments on Staten Island, New York, from 2000 to 2012. We found three salamander species in our study. Two streams had all three species: a headwater stream (Reed's Basket Willow) and a third-order stream (BloodrootValley). We found *Eurycea bislineata* and *Desmognathus fuscus* in all streams, although the frequency of occurrence and densities of these species differed markedly among streams. Reed's Basket Willow had significantly greater populations of *E.bislineata* and *D. fuscus* than the other three, higher order, streams. *Pseudotriton ruber* was found only on two occasions each in Reed's Basket Willow and Bloodroot Valley. We found lower population densities than that reported in other studies for both *Eurycea bislineata* and *Desmognathus fuscus*. The maximum density we recorded for *E. bislineata* was 14.4 individuals/$m^2$ on one occasion in one stream and for *D. fuscus* 0.3 individuals/$m^2$ on several occasions. Despite the low densities, and seasonal and yearly variability, the populations have not shown any noticeable trends in the twelve years of our study and appear stable. We measured sediment deposition and found the highest amount deposited in Reed's Basket Willow. Because this stream also has the highest population densities, our results suggest that sediment does not always have a negative impact on streamside salamanders. We measured impervious cover in the watershed and found that it did not correspond to increased salamander densities; Reed's Basket Willow had the highest salamander densities despite having the highest percent impervious cover. However, Reed's had the lowest percent impervious cover in its buffer. The stream with the lowest densities was a second-order stream downstream from a dam in place for at least 80 years at the start of our study. Egbertville Ravine, which lies below a dam constructed in 2003, has not shown a declining trend in population densities, although the 2012 sampling showed a decrease that was not experienced at the other three sites. Within urban areas, local impacts such as stream order, dams and adjacent land cover may obscure effects of landscape scale factors.

### Keywords

Plethodontidae, urban, stream, salamander, sediment, dam, impervious

### Acknowledgements

This study was supported in part by Grant CD982239-01 from the U.S. Environmental Protection Agency, Region 2. We would like to thank the New York Department of Parks and Recreation for allowing us to complete this study, and Marc Matsil for the original idea which led to this research. The following people provided invaluable field assistance: Marielle Anzelone, Sylvia Birns-Swindlehurst, Susannah Cox, Linda D'Anna, Ben D'Avanzo, Martin Junco, Hadas Kushnir, Nic Kotscheubey, Raphael Mazor, James McCarthy and Christina Scully. Reviews by Nancy Falxa-Sonti, Winsor Lowe and three anonymous reviewers significantly improved the manuscript.

## INTRODUCTION

Urban streams, and the fauna that depend on them, are subject to multiple threats. Urbanization causes major changes in stream hydrology, geomorphology, water quality and biotic communities (Baer and Pringle 2000). Urban development leads to an increase in impervious surfaces, which exacerbates fluctuations in velocity and volume of stream flow (Paul and Meyer 2001). In-stream sediment deposition and erosion of banks increases with urban development (Wolman 1967). Water temperature and turbidity increase, and substrate particle size decreases (Walters et al. 2003). These physical and chemical changes alter communities of benthic invertebrates and fish, favoring cosmopolitan species at the expense of endemic species (Paul and Meyer 2001; Scott and Helfman 2001).

One group that has been shown to suffer from urbanization's effects on streams is salamanders that rely on this habitat for breeding, foraging and/or overwintering (hereinafter referred to as "streamside salamanders"). Orser and Shure (1972) were among the first to describe effects of urbanization on streamside salamanders, in this case the northern dusky salamander *(Desmognathus fuscus)*. They found that degree of urbanization, and consequent increases in erosion and scouring, were lower where *D. fuscus* populations were larger. Other, more recent, studies have corroborated the connection between urbanization and decline of streamside salamanders (Barrett and Price 2014). In particular, the percent impervious cover in a stream's watershed and increases in erosion and sedimentation have been implicated in declines in diversity and/or abundance of salamanders (Welsh and Ollivier 1998; Willson and Dorcas 2003; Miller et al. 2007). Fragmentation of populations by infrastructure can cause declines in genetic diversity, prevent salamanders from dispersing to less-populated areas of streams, and disconnect them from habitats used at different times of year or life stages (Munshi-South et al. 2013; Lowe 2003).

In New York City, three species of streamside salamanders have been found historically (Fig. 1): the northern two-lined (*Eurycea bislineata*); the northern dusky (*D. fuscus);* and the northern red salamander (*Pseudotriton ruber).* All three species remain, although numbers of populations are reduced and not all populations are large enough to maintain genetic diversity (Munshi-South et al. 2013). One streamside salamander, *D. fuscus*, appears to be extirpated from adjacent Westchester County, New York and is declining in nearby Fairfield County, Connecticut. Similarly, the southern dusky salamander (*Desmognathus auriculatus*) has suffered declines in the southeastern United States (Miller et al. 2007) and declines were seen in several salamander species in watersheds studied before and after urbanization in North Carolina (Price et al. 2012). It is clear that research and conservation efforts must be undertaken if these species are to persist in a rapidly urbanizing landscape.

As part of our conservation mission, the New York City Parks Department's Natural Resources Group began to inventory streamside salamanders, as well as other amphibians, throughout New York City in the 1990s. Starting in 2000 we intensively monitored streamside salamanders in four stream segments located in New York City parkland on Staten Island, New York. These streams represent a range of orders, conditions, and catchment sizes. We sought to understand population dynamics of streamside salamanders in these segments and determine which stream habitat characteristics were found where salamander diversity was highest, and populations largest. We looked at sediment deposition, watershed land use and presence of dams at the four stream segments as the impacts most likely to affect diversity and population size.

Based on research in other locations, we expected that salamanders would be less abundant where sediment deposition was highest, where impervious cover in the watershed was greatest, and where dams had been constructed. However, when a dam was constructed between two of our study segments in 2003, sampling in 2004 and 2005 showed an apparent benefit to salamander populations downstream, with an increase in density and in the numbers of larvae (Pehek and Mazor 2008). The current study follows the segments above and below the dam five (2008) and nine (2012) years after construction, as well as monitoring the other segments in 2001, 2002, 2008 and 2012.

**Figure 1a-c.** (a) northern two-lined salamanders (*E. bislineata),* (b) northern dusky salamander *(D. fuscus)* and (c) northern red salamander (*P. ruber).*

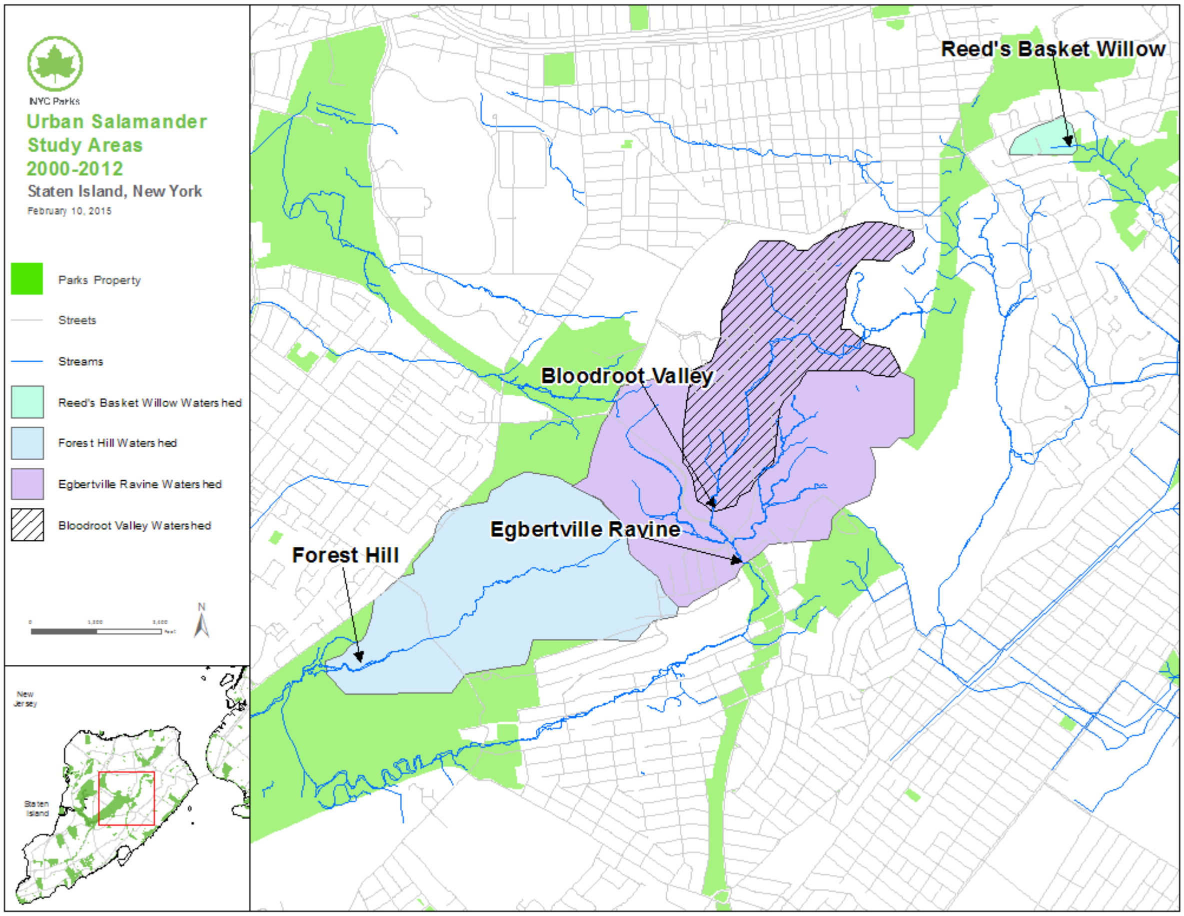
## METHODS

We searched the literature for historic records of amphibians in New York City and re-visited sites with streamside salamander records. We also used aerial photos and topographic maps to identify other likely habitat for streamside salamanders and searched those as well. Starting in 1999 we assessed streams for suitability for long-term monitoring. Although we assessed streams in Queens and Staten Island, we decided to limit our study to Staten Island for ease of conducting field work and similarity of environmental conditions.

We conducted intensive monitoring in four streams, ranging from headwater to 4th order on Staten Island in New York City (Fig. 2): Reed's Basket Willow (Reed's)-1$^{st}$ order; Forest Hill-2$^{nd}$ order; Bloodroot Valley (Bloodroot)-3$^{rd}$ order; and Egbertville Ravine (Egbertville)-4$^{th}$ order. The sites are located in central Staten Island, New York. This highly urbanized region is characterized by a glacial terminal moraine with oak hickory forest and numerous streams and wetlands.

Reed's, which has the smallest watershed, is located near the highest point on Staten Island at 410 feet above sea level (USGS 2014). It is mostly forested, with some residential development. The watersheds of the other streams have a mix of forest, large recreational and hospital complexes, and residential development. Forest Hill is a modified second order stream that currently has two ponds upstream of the study segment that are located on a golf course built in the 1920s. Richmond Creek proper begins in the Egbertville Ravine, which has a road crossing at the top of the study segment and another road running parallel to the stream within 30 m. This is the highest order stream in this study. Bloodroot is a tributary to Richmond Creek, upstream from our Egbertville study section. There are two existing dams at the sites, one historical (Forest Hill) and the other constructed in 2003 between Bloodroot and Egbertville. In 2006, during a dry period, Forest Hill suffered an interruption of flow, with large portions of the stream and its impounded headwaters drying. During the study 3.3 ha was developed for a recreational center in the watersheds of Bloodroot and Egbertville, completed in 2007.

**Figure 2.** Study area on Staten Island, New York, showing watershed outlines in color. Stream segments studied are labeled with a black arrow.




## Salamander Sampling

We measured salamander abundance and length at all four sites using one-meter wide cross-stream belt transects (Connery 2000; Stehman 2000) on sixteen occasions from early spring 2000 to summer 2012. Due to staff limitations, not all streams were sampled on each occasion, and not all seasons in each year. We sampled streams in four seasons; early spring (SP, March-April); late spring/early summer (SPSU, May-June); summer (SU, July-August); and fall (FA, October-November). We sampled four streams in SP and SU 2000. One of these streams, Manor Creek, subsequently dried, so we substituted Forest Hill starting in SP 2001. We sampled Reed’s, Bloodroot, Egbertville and Forest Hill in each season in 2001 and in SP 2002. To study the effects of a dam inserted between two of our stream segments in 2003, we sampled only those two segments (Bloodroot and Egbertville) in all seasons of 2004 and FA of 2005. In 2008 we again sampled all streams in SP, SPSU and FA. In SU 2012, as part of a genetic study, we sampled all streams except Reed’s. We had collected genetic samples from salamanders at Reed’s on May 28, 2010, although we did not standardize our captures by area or time.

We chose a 50 m segment at each site, based on suitability of habitat for streamside salamanders, results of preliminary searches for salamanders, and accessibility. The bottom of each 50 m segment was permanently marked as a "zero" point. On sampling occasions, ten transect locations were randomly chosen within the 50 m segment for each stream and sampling date. Transects were 1 m wide and extended 1 m landward of the waterline on each side. Length of each transect was recorded, along with the number of observers, water and air temperature and humidity. Observers caught salamanders by placing a steel mesh strainer downstream of each cover object and removing the object (Heyer 1994; Jung 2001). All cover objects that could be lifted were sampled in this way, and the stream bottom was observed afterwards to ensure all fauna were caught. The number of minutes elapsed during sampling were recorded for each transect. All salamanders were recorded by life-stage (larva, juvenile, adult) and measured (snout-vent length in cm). We calculated a relative density by dividing the number of captures in a transect by the transect's area to give a number per square meter for each species. We tested differences between sites in densities and snout-vent lengths of *E. bislineata* and *D. fuscus* using a repeated measures linear mixed model SYSTAT 13 (SYSTAT Software, Inc. 2009).

### Sediment Deposition

We sampled sediment in all stream segments in July 2001 and July 2008. In July of 2004 we sampled sediment in Bloodroot and Egbertville as part of our study on the effects of dam construction. We constructed sediment traps from 3.75" diameter steel cans filled with washed, commercially available, Delaware River stone. The stone was sorted using a 1/2" wire mesh so only gravel > 1/2" was used in the sediment traps. All traps were filled with stone and then weighed to the nearest 0.1 g. We generated random distances from the zero point of the bottom of our 50 m segment and from the right bank for trap placement. We installed 10 steel can sediment traps in each site. Traps were buried in the substrate so that the rim was flush with the surface. To measure stormflow sediment deposition, the traps remained in place for one week during which a rain event was predicted. Rain gauges at each site, installed under an open canopy, measured the amount of rain received locally during stormflow sediment measurements. Upon return to the laboratory, we air-dried the contents, after which we weighed the sample and then removed the river stone. Each sample of air-dried sediment was weighed to the nearest 0.1 g. We calculated the amount of sediment deposited per day in the stream and the amount of sediment deposited per millimeter rain. We tested differences among the streams in sediment deposition using a repeated measures linear mixed model SYSTAT 13 (SYSTAT Software, Inc. 2009).

### Land Use Calculations

We delineated watersheds for each site in ArcGIS 10.1. We then digitized land cover and calculated percent impervious cover in each watershed and in a 30 m buffer, for 1996, 2006 and 2010. We were limited in our choice of years by availability of aerial orthophotos, so we chose the years prior to the most full sampling years (1996 for 2001, 2006 for 2008 and 2010 for 2012).

## RESULTS

### Status of Known Salamander Populations

From the literature and our own sampling, we became aware of 18 populations of *E. bislineata* that were reported in 1980 or more recently; based on our surveys, six are likely extirpated. Of 10 reported populations of *D. fuscus*, three are probably extirpated. *Pseudotriton ruber* has been found at five sites in this time period, but always in low numbers. However, none of these populations has been extirpated and one population discovered in 2012 had not been previously reported.

### Salamander Species Richness and Density at Monitoring Sites

We found all three salamander species at two of our monitoring sites, Bloodroot and Reed's, between 2000 and 2012. Only *E. bislineata* was consistently found at all sites, although densities varied considerably over time and between sites (Fig. 3, Table 1). *Desmognathus fuscus* was found consistently only at Reed's (Fig. 4, Table 1). At Bloodroot densities were low and *D. fuscus* were not found on many sampling occasions. Very few *D. fuscus* were found at Egbertville and Forest Hill, and none since spring 2001. Reed's had significantly higher densities of both *E. bislineata* ($F = 46.857$, $p < 0.001$) and *D. fuscus* ($F = 32.912$, $p < 0.001$) than any other site. *Pseudotriton ruber* were found on two occasions each at Bloodroot and Reed's, with five or less individuals found on each occasion (Fig. 5, Table 1).

**Figure 3**. Mean densities of *E. bislineata* in four stream segments on Staten Island, New York, 2000 to 2012. Sampling occasions are represented by a seasonal code followed by the year. SP = early spring, SPSU = late spring/early summer, SU = mid-summer, FA = fall. ◇ ◇ = Bloodroot; • = Egbertville; △ = Forest Hill; ■ = Reed's. Error bars represent standard error.

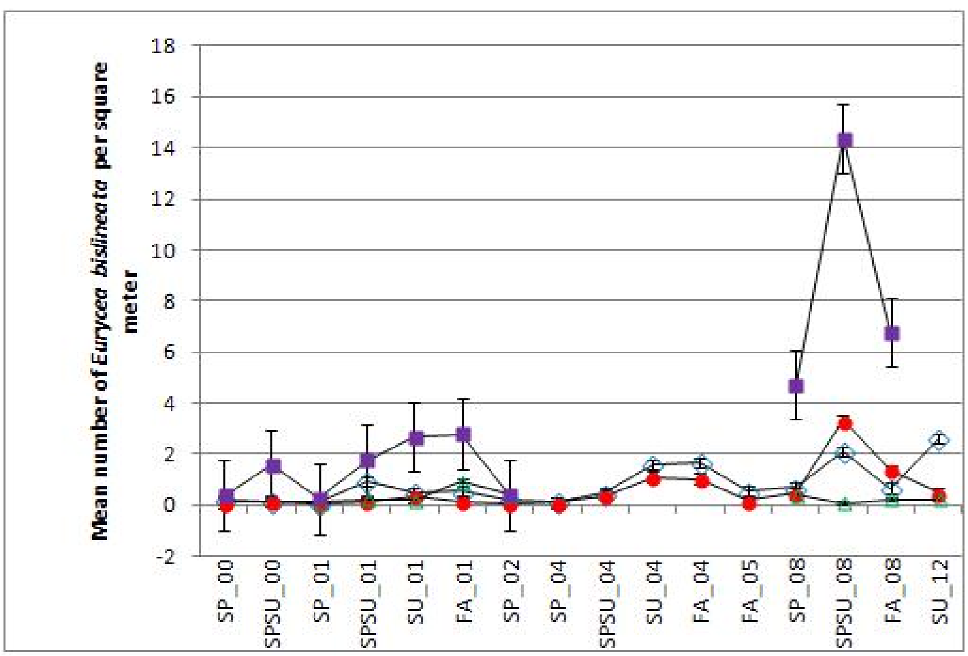


**Figure 4**. Mean densities of *D. fuscus* in four stream segments on Staten Island, New York, 2000 to 2012. Sampling occasions are represented by a seasonal code followed by the year. SP = early spring, SPSU = late spring/early summer, SU = mid-summer, FA = fall. ◇ = Bloodroot; • = Egbertville; △ = Forest Hill; ■ = Reed's. Error bars represent standard error.

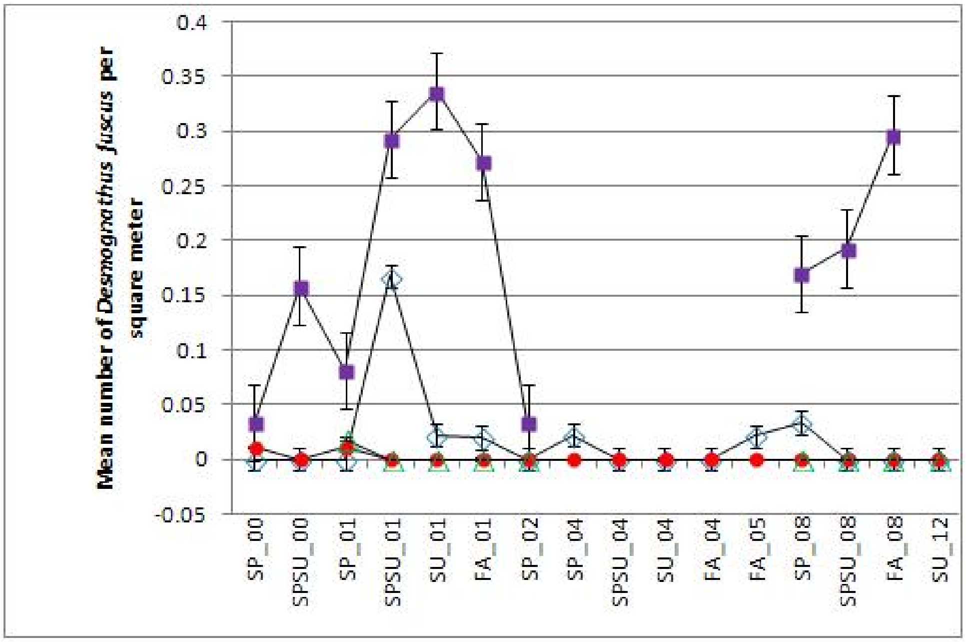
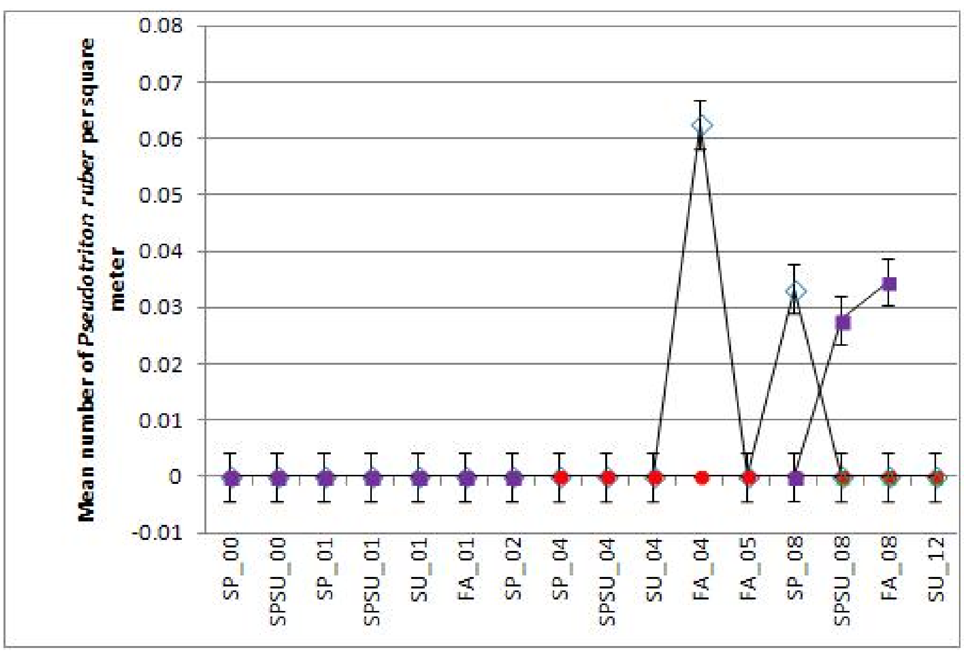



**Figure 5.** Mean densities of *P. ruber* in four stream segments on Staten Island, New York, 2000 to 2012. Sampling occasions are represented by a seasonal code followed by the year. SP = early spring, SPSU = late spring/early summer, SU = mid-summer, FA = fall. ◇ = Bloodroot; • = Egbertville; △ = Forest Hill; ■ = Reed's. Error bars represent standard error.

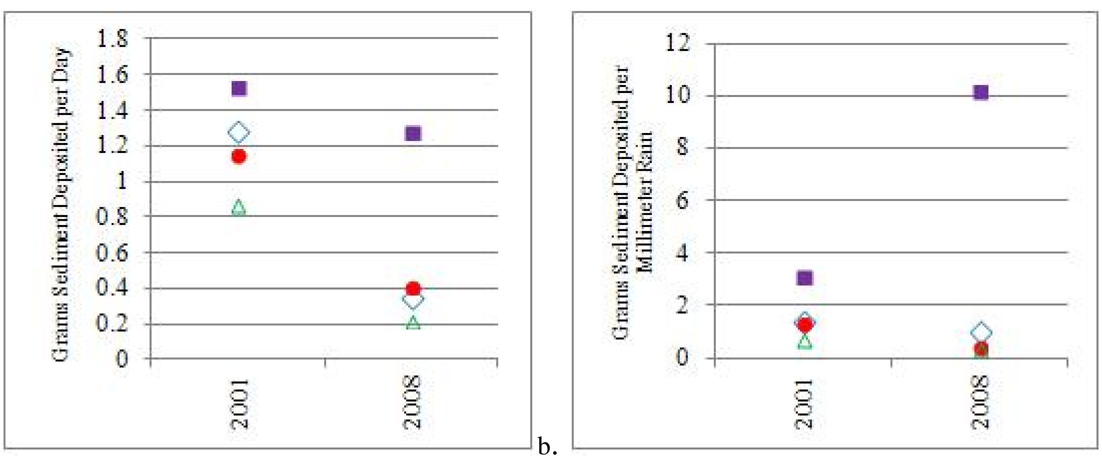


Salamander densities varied by season and year. For both *E. bislineata* and *D. fuscus*, densities were lowest in early spring, higher in late spring/early summer, and usually highest in summer and fall. The few captures of *P. ruber* did not show any seasonal pattern. The greatest densities of *E. bislineata* at Forest Hill occurred in fall 2001, at Reed's and Egbertville in late spring/early summer 2008, and at Bloodroot in summer 2012. Although not a standardized sample, 21 *E. bislineata* were captured in 2010 at Reed's during genetic sampling and we believe that population is secure. *Desmognathus fuscus* numbers at Reed's were greatest in 2001 (summer), but were not much lower in 2008 (fall). Although again not a standardized sample, the 32 *D. fuscus* captured at Reed's in 2012 is a greater number than captured in any previous year. At Bloodroot, in contrast, *D. fuscus* densities were highest in 2001 (late spring/early summer), were low or absent in subsequent years, and duskies have not been found there since spring 2008. *Desmognathus fuscus* were only found on one occasion each at Egbertville and Forest Hill, both in spring, and have not been found at either site since 2001.

## Sediment Deposition

Both measures of sediment (per day, and per mm rain) differed significantly among sites (per day: $F = 21.535$, $p < 0.001$; per mm rain: $F = 11.339$, $p < 0.001$). Reed's had the highest amount of sediment deposited in sediment traps in both 2001 and 2008 (Fig. 6). Deposition at the other sites was lower, and did not differ much among the sites. Forest Hill had the lowest amount of sediment in both 2001 and 2008. Bloodroot and Egbertville had nearly the same amount of sediment deposited in 2001, 2004, and 2008.

**Figure 6a-b.** Sediment deposition from 10 sediment traps each in 4 Staten Island, New York, streams, July 2001 and July 2008. (a) Amount of sediment collected/day (mg/day), (b) Amount of sediment collected per millimeter of rainfall (mg/mm rainfall). ◇ = Bloodroot; ● = Egbertville; △ = Forest Hill; ■ = Reed's.




a. b.

## Impervious Cover

Egbertville had the largest watershed (223 hectares) and the third greatest percent impervious cover in the watershed (Table 2). Reed's had the smallest watershed (5.6 hectares) and the greatest percent impervious cover. Forest Hill had the second largest watershed (139 hectares) and the lowest percent impervious cover. Bloodroot's watershed was 97 hectares, all falling within Egbertville's watershed, and percent impervious cover was slightly higher than

Egbertville's. Impervious cover rose 1% in Forest Hill's watershed, and between 1.9 and 2.2% in the other watersheds, from 1996 to 2010.

The pattern differs in the 30 m buffer (Table 2). The percent impervious cover was much lower in the buffers of all streams than it was in their watersheds. Egbertville had the highest percent impervious cover in its buffer, at 2.8% in 2006 and 2010. In all three years Bloodroot had 1.1% and Forest Hill had 1.5% impervious in the buffer. Reed's had 0% impervious cover in its buffer in all years.

## DISCUSSION

The apparent disappearance of *E. bislineata* and/or *D. fuscus* from several sites in New York City where they had been known in 1980, or more recently, is troubling. *Eurycea bislineata* disappeared from one site after a drought year when a stream in the Bronx dried nearly completely. We have been unable to find salamanders there even though subsequent years have been much wetter. We also have not been able to find *E. bislineata* at any of the Manhattan sites where it had been reported. The continued existence of some populations, however, is both surprising and heartening. For example, the population of *D. fuscus* discovered by Gans in 1944 (Gans 1945) in Manhattan does not appear to have decreased in size since we first visited it in 2005. The continued presence of *P. ruber* in a spring-fed stream within a golf course is another example. It may be important for conservation to understand how some populations persist in what seem to be unsuitable habitats.

In our monitoring sites, densities of *E. bislineata* are generally lower than densities reported for other sites (10 to 63 larvae/$m^2$, Barrett and Price 2014). On only one occasion did density of *E. bislineata* fall within this range; in SPSU 2008 we found 14.4 larvae/$m^2$ at Reed's. Although low and variable, densities of *E. bislineata* appear to be stable overall. However, the drop in density at Egbertville in 2012 with a corresponding rise at Bloodroot that year may reflect delayed effects of dam construction that were not evident in 2005 or 2008. Despite consistently low densities at Forest Hill, *E. bislineata* does not appear to be declining there.

*Desmognathus fuscus*, in contrast, has been absent on at least some occasions from all sites except Reed's. Densities are much lower than those reported by Barrett and Price (2014), with the greatest density being 0.3 larvae/$m^2$ at Reed's in SPSU, SU and FA 2001 and FA 2008. Duskies show no decreasing trend at Reed's, but may be in decline at Bloodroot. The lack of any captures of this species since spring 2001 at either Egbertville or Forest Hill most likely indicates extirpation.

Because *P. ruber* may spend large portions of its life cycle in underground or piped streams, we do not feel our surveys adequately represent population sizes or trends. Although always found in low numbers, we have not, however, found it to be extirpated from any of the sites from which it is known in the City. Its use of underground refugia may have somewhat sheltered it from urban impacts.

The data from this study would be much improved if we employed mark-recapture methods and included detection probabilities in analyses. Several authors have found that data from serial counts with low replication have high variability and biases due to sampling technique, observer or other variables. (Mazerolle et al. 2007; Price et al. 2012). Thus, serial counts without incorporating differences in detectability may not reflect actual abundances. In a study comparing counts without marking and mark-recapture methods, mark-recapture gave population estimates orders of magnitude higher than simple counts (Nowakowski and Maerz 2009). We feel, although our data may underestimate population sizes, our data do represent real differences among sites and seasons. Our study used higher replication than the above studies, we limited our sampling to riffle and step-pool habitats within each stream, and we did not sample within 24 hours after rainfall, thus eliminating some of the bias due to habitat differences.

Against our expectations, the most-productive stream for all three salamander species, Reed's, had the highest amounts of sediment deposition (primarily sand) of the four sites. The second most productive, Bloodroot, had a high percent sand in pebble counts of the substrate compared to Forest Hill and Egbertville (E. Pehek, unpublished data). This finding contrasts with evidence from other studies of effects of sediment on streamside salamanders (Peterman and Semlitsch 2009; Miller et al. 2007; Barrett et al. 2010) although a recent study found negligible effects of fine sediment on larval *Eurycea wilderae* and *Desmognathus quadramaculatus* (Keitzer and Goforth 2012). Urban salamanders in our area are persisting despite high sediment erosion and deposition, perhaps due to behavioral or other modifications or regional differences. For example, salamanders may retreat to seepage areas adjacent to Reed's and Bloodroot during high flow events. Most research on the effects of sedimentation on streamside salamanders, including Keitzer and Goforth (2012), has been conducted in the southern Appalachians. Little is known about potential differences in habitat preferences between mid-Atlantic and southern Appalachian salamanders.

Our expectation that a larger percent impervious cover in a watershed would correspond to lower salamander densities was not borne out. Forest Hill has the fewest salamanders, but the smallest percent impervious cover in its watershed. Reed's has the most salamanders and the highest percent impervious cover in its watershed. Impervious cover in the 30 m buffer has a closer correspondence with salamander densities. The most productive stream, Reed's, has the least percent impervious in its buffer. However, the stream with the greatest percent impervious in its buffer, Egbertville, does not have the lowest salamander densities. A study of four stream segments is small, however, so we will need to expand our study to include more stream segments, and replicates of each segment order, to make firm conclusions about the relationship of salamander density to impervious cover in New York City.

Stream order appears to have the clearest relationship with salamander densities among these four stream segments, with the lowest-order stream, Reed's Basket Willow, having the highest densities of salamanders by far. Among the other three streams, the greatest impact appears to be the historical dam above Forest Hill. From its percent impervious cover, and the fact that it is only a second-order stream, we would expect Forest Hill to have larger populations than Egbertville, but the long-standing dam apparently has stronger effects on salamander populations than percent impervious cover. Walsh et al. (2005) note that the effects of impervious cover on stream condition may be obscured by other strong stressors, such as sewage or other pollutants.

Nine years after construction, we may not have yet seen the full results of the dam above Egbertville. A delay in biotic change downstream of dams has been shown for invertebrates (Petts 1987). The community at Egbertville may not have yet reached a stable state; the community below Forest Hill has had at least 80 years to equilibrate. Continued monitoring will reveal whether the long-term effects of the dam on the salamanders in Egbertville will be negative, positive, or neutral.

Another aspect of the new dam that should be further investigated is the fragmentation caused by dam construction. Because streamside salamanders use different portions of a stream for egg-laying, for larvae to grow to maturity, and for overwintering, access between upstream and downstream is important (Jackson 2003; Ashton and Ashton 1978). Additionally, disconnection of Egbertville and Bloodroot, which had no other habitat connectivity besides the stream and culvert under the adjacent road, may lead to loss of genetic diversity. We have collected genetic samples and hope to use these to assess the strength of different barrier types (e.g., roads, culverts, dams) and the amount of time it takes for genetic divergence to occur after fragmentation of populations. If we understand which barriers are most disruptive, we can begin to remove or mitigate those barriers to re-connect populations.

That the headwater stream had a much greater abundance of salamanders than all other streams in this study highlights the importance of protecting headwater streams and their connectivity to higher order streams for the conservation of biological diversity in urbanizing areas. Headwater streams may serve as highly-productive “nurseries” for the more-vulnerable life stage (larvae) of salamanders because of the lack of fish and invertebrate predators (Lowe et al. 2004; Peterman et al. 2008). Headwater streams may also provide the only remaining habitat for both adults and larvae in an urbanized area; they are responsible for the persistence of several salamander populations in unlikely places in New York City. Our two *D. fuscus* populations in Manhattan persist due to the presence of headwater streams on steep, undevelopable slopes. *Pseudotriton ruber* continues to live in a headwater stream with a very small wooded and shrubby buffer on a manicured golf course. Without the continued protection of headwaters, urban areas will likely lose both these species, and populations of *E. bislineata* would be reduced in size.

In addition to providing habitat for salamanders, headwater streams increase overall biodiversity, assist with flood mitigation, improve water quality, and facilitate nutrient recycling (Gomi et al. 2002; Lowe and Likens 2005; Alexander et al. 2007; Meyer et al. 2007; Nadeau and Rains 2007). Unfortunately, many headwaters are not mapped, and thus not considered when development plans are made (Brooks and Colburn 2011). A majority of headwaters are buried in some urban areas (Meyer and Wallace 2001; Elmore and Kaushal 2008). Many have been converted to farm ponds, as our Forest Hill site originally was (Davis 1892). Others are subject to erosion, sedimentation and pollution from stormwater overflows (Meyer and Wallace 2001), which can be mitigated through use of green infrastructure such as bioswales, bioretention facilities, green roofs, infiltration devices, or forested buffers (Medina et al. 2004; Moore and Palmer 2005; Sweeney and Blaine 2007). We recommend thorough mapping and assessment of remaining headwaters, and management to reduce impacts in these watersheds, such as limiting impervious cover, installing infiltration structures or swales adjacent to new development, maintaining or restoring forest, and seeking ways to reconnect headwaters to downstream habitat.

## APPENDIX

**Table 1**. Means and standard deviations of densities of *Eurycea bislineata*, *Desmognathus fuscus* and *Pseudotriton ruber* in four stream segments on Staten Island, New York, 2000 to 2012. Dashes for Forest Hill in SP and SP/SU 2000 represent no data collected because the stream was dry.

| Year | Season | Site | *Eurycea bislineata* | *Desmognathus fuscus* | *Pseudotriton ruber* |
|---|---|---|---|---|---|
| 2000 | SP | Bloodroot | 0.2 (0.35) | 0 | 0 |
| 2000 | SP | Egbertville | 0.1 (0.14) | 0 | 0 |
| 2000 | SP | Forest Hill | - | - | - |
| 2000 | SP | Reed's | 0.37 (0.51) | 0.03 (0.11) | 0 |
| 2000 | SPSU | Bloodroot | 0.13 (0.29) | 0 | 0 |
| 2000 | SPSU | Egbertville | 0.15 (0.15) | 0 | 0 |
| 2000 | SPSU | Forest Hill | - | - | - |
| 2000 | SPSU | Reed's | 1.58 (0.71) | 0.16 (0.19) | 0 |
| 2001 | SP | Bloodroot | 0.09 (0.15) | 0 | 0 |
| 2001 | SP | Egbertville | 0.07 (0.1) | 0.01 (0.04) | 0 |
| 2001 | SP | Forest Hill | 0.1 (0.14) | 0 | 0 |
| 2001 | SP | Reed's | 0.25 (0.32) | 0.08 (0.18) | 0 |
| 2001 | SPSU | Bloodroot | 0.94 (0.88) | 0.17 (0.43) | 0 |
| 2001 | SPSU | Egbertville | 0.12 (0.11) | 0.02 (0.07) | 0 |
| 2001 | SPSU | Forest Hill | 0.22 (0.31) | 0 | 0 |
| 2001 | SPSU | Reed's | 1.75 (1.62) | 0.3 (0.43) | 0 |
| 2001 | SU | Bloodroot | 0.51 (0.37) | 0 | 0 |
| 2001 | SU | Egbertville | 0.33 (0.38) | 0 | 0 |
| 2001 | SU | Forest Hill | 0.19 (0.29) | 0 | 0 |
| 2001 | SU | Reed's | 2.67 (1.81) | 0.34 (0.51) | 0 |
| 2001 | FA | Bloodroot | 0.53 (0.41) | 0.02 (0.06) | 0 |
| 2001 | FA | Egbertville | 0.14 (0.16) | 0 | 0 |
| 2001 | FA | Forest Hill | 0.94 (0.9) | 0 | 0 |
| 2001 | FA | Reed's | 2.77 (1.71) | 0.27 (0.64) | 0 |
| 2002 | SP | Bloodroot | 0.21 (0.29) | 0 | 0 |
| 2002 | SP | Egbertville | 0.05 (0.08) | 0 | 0 |
| 2002 | SP | Forest Hill | 0.42 (0.36) | 0 | 0 |
| 2002 | SP | Reed's | 0.38 (0.24) | 0.03 (0.11) | 0 |
| 2004 | SP | Bloodroot | 0.14 (0.16) | 0.02 (0.07) | 0 |
| 2004 | SP | Egbertville | 0.1 (0.17) | 0 | 0 |
| 2004 | SPSU | Bloodroot | 0.47 (0.4) | 0 | 0 |
| 2004 | SPSU | Egbertville | 0.36 (0.47) | 0 | 0 |
| 2004 | SU | Bloodroot | 1.6 (1.83) | 0 | 0 |

(Table 1, continued)

| | | | | | |
|---|---|---|---|---|---|
| 2004 | SU | Egbertville | 1.09 (1.01) | 0 | 0 |
| 2004 | FA | Bloodroot | 1.66 (0.76) | 0 | 0.63 (0.198) |
| 2004 | FA | Egbertville | 1.01 (1.61) | 0 | 0 |
| 2005 | FA | Bloodroot | 0.53 (0.4) | 0.02 (0.07) | 0 |
| 2005 | FA | Egbertville | 0.18 (0.22) | 0 | 0 |
| 2008 | SP | Bloodroot | 0.67 (0.74) | 0.03 (0.11) | 0.03 (0.11) |
| 2008 | SP | Egbertville | 0.45 (0.42) | 0 | 0 |
| 2008 | SP | Forest Hill | 0.42 (0.39) | 0 | 0 |
| 2008 | SP | Reed’s | 4.73 (2.6) | 0.17 (0.2) | 0 |
| 2008 | SPSU | Bloodroot | 2.09 (1.48) | 0 | 0 |
| 2008 | SPSU | Egbertville | 3.33 (1.58) | 0 | 0 |
| 2008 | SPSU | Forest Hill | 0.07 (0.11) | 0 | 0 |
| 2008 | SPSU | Reed’s | 14.39 (5.62) | 0.19 (0.18) | 0.03 (0.09) |
| 2008 | FA | Bloodroot | 0.66 (0.83) | 0 | 0 |
| 2008 | FA | Egbertville | 1.37 (1.08) | 0 | 0 |
| 2008 | FA | Forest Hill | 0.22 (0.21) | 0 | 0 |
| 2008 | FA | Reed’s | 6.76 (2.32) | 0.3 (0.3) | 0.03 (0.11) |
| 2012 | SU | Bloodroot | 2.6 (2.1) | 0 | 0 |
| 2012 | SU | Egbertville | 0.46 (0.36) | 0 | 0 |
| 2012 | SU | Forest Hill | 0.23 (0.29) | 0 | 0 |

**Table 2.** Percent impervious cover in the watersheds of four stream segments, and in a 30 m buffer around each stream segment, in 1996, 2006 and 2010. Streams are located on Staten Island, New York.

| Site | Watershed Area | **1996** Impervious Cover | | **2006** Impervious Cover | | **2010** Impervious Cover | |
|---|---|---|---|---|---|---|---|
| | (ha) | (ha) | (%) | (ha) | (%) | (ha) | (%) |
| Bloodroot | 96.9 | 12.0 | 12.4 | 13.5 | 13.9 | 14.2 | 14.7 |
| Egbertville | 223.0 | 19.8 | 8.9 | 23.2 | 10.4 | 24.2 | 10.8 |
| Forest Hill | 136.3 | 4.4 | 3.2 | 5.0 | 3.7 | 5.7 | 4.2 |
| Reed’s | 5.6 | 1.1 | 19.6 | 1.1 | 19.6 | 1.2 | 21.4 |
| **Site** | **Buffer Area** | **1996** Impervious Cover | | **2006** Impervious Cover | | **2010** Impervious Cover | |
| | (ha) | (ha) | (%) | (ha) | (%) | (ha) | (%) |
| Bloodroot | 18.1 | 0.2 | 1.1 | 0.2 | 1.1 | 0.2 | 1.1 |
| Egbertville | 49.9 | 1.3 | 2.6 | 1.4 | 2.8 | 1.4 | 2.8 |
| Forest Hill | 13.2 | 0.2 | 1.5 | 0.2 | 1.5 | 0.2 | 1.5 |
| Reed’s | | 1.1 | 0 | 0 | 0 | 0 | 0 |

**LITERATURE CITED**


Alexander, R. B., E. W. Boyer, R. A. Smith, G. E. Schwarz and R. B. Moore. 2007. The role of headwater streams in downstream water quality. Journal of the American Water Resources Association 43:41-59.

Ashton, R.E. and P.S. Ashton. 1978. Movements and winter behavior of *Eurycea bislineata* (Amphibia, Urodela, Plethodontidae). Journal of Herpetology 12:295–298.

Baer, K. E. and C. M. Pringle. 2000. Special problems of urban river conservation: The encroaching megalopolis, pp. 385-402. In: Boon, P.J., B.R. Davies and G.E. Petts (Eds.). Global Perspectives on River Conservation: Science, Policy, and Practice. Jon Wiley & Sons, Inc. Chichester, UK.

Barrett, K. and S. J. Price. 2014. Urbanization and stream salamanders: A review, conservation options, and research needs. Freshwater Science 33:927-940.

Barrett, K., B. S. Helms, C. Guyer and J. E. Schoonover. 2010. Linking process to pattern: Causes of stream-breeding amphibian decline in urbanized watersheds. Biological Conservation 143:1998-2005.

Brooks, R. T. and E. A. Colburn. 2011. Extent and channel morphology of unmapped headwater stream segments of the Quabbin watershed, Massachusetts. Journal of the American Water Resources Association 47:158-168.

Connery, B. 2000. Acadia National Park stream salamander survey. Fourth Conference of the North American Amphibian Monitoring Program, Eastern Streamside Salamander Meeting. State College, PA. Accessed 04/29/2003.

Davis, W. T. 1892. Days Afield on Staten Island (1994 reprint of original). Staten Island Institute of Arts and Sciences, New York, NY.

Elmore, A. J. and S. S. Kaushal. 2008. Disappearing headwaters: Patterns of stream burial due to urbanization. Frontiers in Ecology and the Environment 6:308-312.

Gans, C. 1945. Occurrence of the dusky salamander on Manhattan. Copeia 2:118.

Gomi, T., R. C. Sidle and J. S. Richardson. 2002. Understanding processes and downstream linkages of headwater systems. BioScience 52:905-916.

Heyer, W.R., M.L. Donnelly, R.W. McDiarmid, L.C. Hayek and M.S. Foster. 1994. Measuring and monitoring biological diversity: Standard methods for amphibians. Smithsonian Institution Press, Washington, D.C.

Jackson, S.D. 2003. Ecological considerations in the design of river and stream crossings, pp. 20–29. In: Irwin, C.L., P. Garrett, and K. P. McDermott (Eds.). Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment, Raleigh, NC.

Jung, R. E. 2001. Streamside salamander inventory and monitoring: Northeast refuges and parks, Summer/Fall 2001. Northeast Amphibian Research and Monitoring Initiative. USGS Patuxent Wildlife Research Center. Laurel, MD.

Keitzer, S. C. and R. R. Goforth. 2012. Response of stream-breeding salamander larvae to sediment deposition in southern Appalachian (U.S.A.) headwater streams. Freshwater Biology 57:1535-1544.

Lowe, W. H., K. H. Nislow and D. T. Bolger. 2004. Stage-specific and interactive effects of sedimentation and trout on a headwater stream salamander. Ecological Applications 14:164–172.

Lowe, W. H. 2003. Linking dispersal to local population dynamics: A case study using a headwater salamander system. Ecology 84:2145-2154.

Lowe, W. H. and G. E. Likens. 2005. Moving headwater streams to the head of the class. BioScience 55:196-197.

Mazarolle, M. J., L. L. Bailey, W. L. Kendall, J. A. Royle, S. J. Converse and J. D.Medina, D. E., P. Graham, J. Thorpe, A. Patwardhan and T. Hare. 2004. Modeling low impact development hydrology: Case studies using the LIFE™ model. Proceedings of the Water Environment Federation. Watershed 2004:524-535.

Meyer, J. L., D. L. Strayer, J. B. Wallace, S. L. Eggert, G. S. Helfman and N. E. Leonard. 2007. The contribution of headwater streams to biodiversity in river networks. Journal of the American Water Resources Association 43:86-103.

Miller, J. E., G. R. Hess and C. E. Moorman. 2007. Southern two-lined salamanders in urbanizing watersheds. Urban Ecosystems 10:73-85.

Moore, A. A. and M. A. Palmer. 2005. Invertebrate biodiversity in agricultural and urban headwater streams: Implications for conservation and management. Ecological Applications 15:1169-1177.

Munshi-South, J., Y. Zak and E. Pehek. 2013. Conservation genetics of extremely isolated urban populations of the northern dusky salamander *(Desmognathus fuscus)* in New York City. PeerJ 1:e64.

Nadeau, T. L. and M. C. Rains. 2007. Hydrological connectivity of headwaters to downstream waters: Introduction to the featured collection. Journal of the American Water Resources Association 43:1-4.

Nichols. 2007. Making great leaps forward: Accounting for detectability in herpetological field studies. Journal of Herpetology 41:672–689.

Nowakowski, A. J. and J. C. Maerz. 2009. Estimation of larval stream salamander densities in three proximate streams in the Georgia Piedmont. Journal of Herpetology 43:503-509.

Orser, P. N. and D. J. Shure. 1972. Effects of urbanization on the salamander Desmognathus fuscus fuscus. Ecology 53:1148-1154.

Paul, M. J. and J. L. Meyer. 2001. Streams in the urban landscape. Annual review of Ecology and Systematics 32:333-365.

Pehek, E. and R. Mazor. 2008. Effect of a stormwater impoundment on streamside salamander populations on Staten Island, New York, pp 85-94. In: Mitchell, J.C., R.E. Jung-Brown and B. Bartholomew. Urban Herpetology. *Herpetological Conservation* 3. Society for the Study of Amphibians and Reptiles, Salt Lake City, UT.

Peterman, W. E., J. A. Crawford and R. D. Semlitsch. 2008. Productivity and significance of headwater streams: Population structure and biomass of the black-bellied salamander *(Desmognathus quadramaculatus*). Freshwater Biology 53:347-357.

Peterman, W. E. and R. D. Semlitsch. 2009. Efficacy of riparian buffers in mitigating local population declines and the effects of even-aged timber harvest on larval salamanders. Forest Ecology and Management 257:8-14.

Petts, G. E. 1987. Time scales for ecological change in regulated rivers, pp. 257–266. In: Craig, J. F. and J. B. Kemper (Eds.). Regulated Streams: Advances in Ecology. Plenum Press, New York, NY.

Price, S. J., R. A. Browne and M. E. Dorcas. 2012. Evaluation the effects of urbanization on salamander abundances using a before-after control-impact design. Freshwater Biology 57:193- 203.

Scott, M. C. and G. S. Helfman. 2001. Native invasions, homogenization, and the mismeasure of integrity of fish assemblages. Fisheries 26:6-15.

Stehman, S. V. 2000. Estimating density from surveys employing unequal-area belt transects. Wetlands 20:512-519.

Sweeney, B.W. and J.G. Blaine. 2007. Resurrecting the in-stream side of riparian forests. Journal of Contemporary Water Research and Education 136:17-27.

SYSTAT Software, Inc., 2009. SYSTAT 13. SYSTAT Software, Inc., Chicago, IL.

U. S. Geological Survey. 2014. Geology of National Parks, 3D and Photographic Tours: Staten Island Serpentinite. http://3dparks.wr.usgs.gov/nyc/parks/loc7.htm. Accessed 05/24/2014.

Utz, R. M. and R. H. Hilderbrand. 2011. Interregional variation in urbanization-induced geomorphic change and macroinvertebrate habitat colonization in headwater streams. Journal of the North American Benthological Society 30:25-37.

Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman and R. P. Morgan. 2005. The urban stream syndrome: Current knowledge and the search for a cure. Journal of the North American Benthological Society 24:706-723.

Walters, D. M., D. S. Leigh and A. B. Bearden. 2003. Urbanization, sedimentation, and the homogenization of fish assemblages in the Etowah River Basin, USA. Hydrobiologia 494:5-10.

Welsh, H. H. and L. M. Ollivier. 1998. Stream amphibians as indicators of ecosystem stress: A case study from California's redwoods. Ecological Applications 8:1118-1132.

Willson, J. D., and M. E. Dorcas. 2003. Effects of habitat disturbance on stream salamanders: Implications for buffer zones and watershed management. Conservation Biology 17:763-771.

Wolman, M. G. 1967. A cycle of sedimentation and erosion in urban river channels. Geographiska Annaler 49:385-395.