

SOURCE – SINK DYNAMICS OF ANURANS IN STORMWATER  
BASINS OF NEW JERSEY’S COASTAL PLAIN

by

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## ABSTRACT OF THE THESIS

Source - Sink Dynamics of Anurans in Stormwater Basins of the New Jersey  
Coastal Plain

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Stormwater basins are a commonly employed Best Management Practice designed to deal with the negative effects of runoff from impervious surfaces. They are ubiquitous in the landscape; yet the effect of these basins on faunal assemblages has not been investigated. Stormwater basins have the potential to influence the breeding distribution of anurans by being sources for some species and sinks for others. This study aims to determine which species benefit from the existence of stormwater basins, which species are negatively impacted, and what local and landscape variables are the best predictors of these effects. Thirty-six permanently ponded stormwater basins in southern New Jersey's outer coastal plain were monitored in 2008 for the presence of mating adults and anuran larvae by aural surveys, dip-netting and trapping. Interviews, visual encounters, dip-netting, and traps assessed fish presence. Water temperature, conductivity, and pH were measured throughout the growing season. A 100 foot buffer

area surrounding the basins was divided into managed and unmanaged grass and woody vegetation as well as impervious surface. Two connectivity metrics, distance to a canopied corridor and percent of undeveloped upland within 500 meters, were analyzed in ArcGIS.

Fish were detected in 92% of the basins. Resistance to fish predation distinguished successful species, those with larvae present, from unsuccessful species, those with calling activity but no larval presence. Permanently ponded basins were sources for *Bufo woodhousii fowleri*, and *Rana catesbeiana*, and sinks for *Pseudacris crucifer crucifer*, *Hyla versicolor*, and *Rana clamitans*.

Connectivity to and availability of terrestrial habitat were significant predictors of how many species mated at the basins. The number of species increased as access to and amount of quality terrestrial habitat increased. This suggests that placement of permanently ponded basins near populations of threatened or endangered amphibians susceptible to fish predation is inadvisable. Additionally, as permanent stormwater ponds are sources for *Rana catesbeiana*, in areas where bullfrogs are invasive, basins will likely increase propagule pressure.

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## INTRODUCTION

Stormwater basins are the principal method employed by municipalities in the US and across the globe to deal with the negative effects of stormwater runoff from impervious surfaces (DayWater 2003, CIRIA 2007, Environment Australia 2002). These negative effects include increased stream siltation and erosion, increased in-stream levels of nutrients and contaminants, decreased groundwater recharge, and increased frequency of flooding (US EPA Phase II Rules 2000). Structural basins have been designed to ameliorate these negative effects (NJDEP 2004).

The State of New Jersey has mandated the construction of basins to control stormwater runoff since the early 90's and in more limited cases since the 70's. More stringent restrictions were enacted in 2004, mandating basin construction in more instances (N. J. A. C. 7:8 and N.J.A.C. 7:14A). Commercial and residential development projects with as little as  $\frac{1}{4}$  acre of impervious surface are obliged to construct basins to assist in the recharge of groundwater and address water quality impacts. As a result, these basins are ubiquitous in the landscape wherever recent development has occurred and they are certain to become more so.

Research into storm water basins has so far focused on pollution, flood, and pathogen control (US EPA 1983, Fischer *et al.* 2003; Morgan *et al.* 2005; Marsalek *et al.* 2006; Hogan & Walbridge 2007; Rusciano & Obropta 2007; Hatt *et al.* 2008; Pomeroy *et al.* 2008), nutrient cycling (Datry *et al.* 2003; Zhu *et al.* 2004) and mosquito production (Kwan *et al.* 2008). Tadpole tolerance to sediment toxicity has been investigated in microcosms (Sanzo & Hecnar 2006; Snodgrass *et al.* 2008), but plant community and faunal assemblages in the field have not been investigated.

Urbanization is associated with amphibian declines through the detrimental effects of habitat destruction, fragmentation and degradation (Hamer *et al.* 2002; Stuart *et al.* 2004; Cushman 2006; Crawford & Semlitsch 2008; Harper *et al.* 2008). As components of the urban matrix, stormwater basins may have the potential to further influence the breeding distribution of anurans by being sources for some species and sinks for others. This study was undertaken to determine which species benefit from the existence of stormwater basins, which species are negatively impacted, and what local and landscape variables are the best predictors of these effects.

## BACKGROUND

Anuran communities exploit a range of aquatic habitats that are arrayed along a gradient from ephemeral to permanent ponds. This gradient is theorized to structure tadpole assemblages within ponds in several ways. Species distributions are controlled by trade-offs between larval tolerance to drying in temporary ponds and vulnerability to predation in more permanent ponds (Werner & McPeck 1994; Skelly 1995; Wellborn *et al.* 1996; Babbitt *et al.* 2003; Van Buskirk 2003) or by a permanence-competitive ability trade-off whereby tadpoles are tolerant of pond drying or are strong competitors in more permanent ponds (Smith 1983; Wilbur 1987). A unimodal pattern of species richness that peaks at intermediate hydroperiods is expected and this pattern has been demonstrated in empirical studies (Snodgrass *et al.* 2000; Paton & Crouch 2002; Peltzer & Lajmanovich 2004; Buskirk 2005; but see Babbitt *et al.* 2003). Ponds are treated as discrete patches with characteristics preferred by individual species.

Spatially explicit metapopulation models also treat the pond as a discrete habitat patch, but focus on the extinction and colonization of populations in a regional group of pond patches within a homogeneous matrix (Hecnar & M'Closkey 1996; Marsh & Trenham 2001). Extinction and colonization rates are governed by the size relationship and connectivity among patches (Hanski 1994; Trenham *et al.* 2003). Connectivity is defined as density or distance between patches (Gulve 1994; Marsh *et al.* 1999; Werner *et al.* 2007b). Variations in the characteristics of ponds affect the survivorship and reproductive success of species at a particular patch. Within the regional framework, source-sink dynamics (Pulliam 1988) describe local population variation. Patches or ponds with positive growth rates (birth rates exceed death rates) are defined as sources

while sinks exhibit negative growth rates in the absence of immigration. Consequently, the proportion of source habitat to sink habitat may affect the persistence of the population on a regional scale (Pulliam & Danielson 1991).

Most pond-breeding anurans depend on upland forests for foraging, hibernating and migrating (Stebbins and Cohen 1995). Although the biphasic life history of anurans is not explicitly accounted for in metapopulation models, highly variable recruitment and less variable adult mortality (Alford & Richards 1999) in amphibian populations suggest the pond as patch model to be useful in natural habitats. However, increasing development fragments the matrix surrounding breeding ponds, affecting the permeability (Rubbo & Kiesecker 2005; Cushman 2006b; Eigenbrod *et al.* 2008), suitability (Gray *et al.* 2004; Gibbs *et al.* 2005), and structural complexity (Rubbo & Kiesecker 2005; Gardner *et al.* 2007) of critical upland habitat, as well as altering metapopulation structure (Marsh & Trenham 2001; Smith & Green 2005). Proximity and area of critical upland habitat has become part of the analysis of anuran community structure in both natural and fragmented landscapes (Knutson *et al.* 1999; Guerry & Hunter 2002; Homan *et al.* 2004; Pillsbury & Miller 2008).

In addition to the negative impact on coterminous habitat, urbanization produces subsidies to wildlife that can increase the abundance of species able to take advantage of those subsidies. Species that respond positively to anthropogenic resources are typically generalists, with broad dietary and habitat requirements (McKinney 2002). However, they may also be specialists that have access to a resource readily available in urban environments and tolerant of human presence or active when humans are not (Adams 2006). In this context, stormwater basins may be characterized as habitat subsidies for

those anuran species able to utilize them. Hamer & McDonnell (2008) included habitat creation in the form of garden ponds, ornamental lakes and dams, and retention ponds as a key process that impacts amphibian population dynamics in urban and suburban landscapes.

Over half of the study sites are located within the Pinelands National Reserve (PNR). Comprehensive studies of anuran community composition in PNR have been conducted since 1993 by scientists of the Pinelands Commission (Bunnell & Zampella 1999; Zampella & Bunnell 2000; Zampella *et al.* 2006; Bunnell & Zampella 2008). Water quality degradation characterized by elevated pH and specific conductance in these typically acidic, low nutrient, high dissolved organic 'blackwater' habitats has been associated with basin-wide upland development including agriculture (Zampella *et al.* 2007). The low pH and low nutrient conditions found in stream impoundments of primarily undeveloped forested watersheds, are host to distinctive Pinelands native-fish and native-anuran communities, whereas degraded impoundments tend to support non-Pinelands anuran and fish species, including Bullfrogs (*Rana catesbeiana*), bluegill sunfish (*Lepomis macrochirus*), and other non-Pinelands centrarchids (Zampella *et al.* 2006; Bunnell & Zampella 2008). The work by Bunnell and Zampella (2008) suggests that species, such as the Carpenter Frog (*Rana virgatipes*), found restricted to undisturbed Pinelands (Pine Barrens) habitats, may be especially vulnerable to watershed disturbance as well as competition/predation from non-Pinelands species such as Bullfrogs.

## HYPOTHESES

- 1) Species richness (as defined by the presence of anuran larvae) will be greater in temporary stormwater ponds as opposed to permanently ponded basins. Work in this area has shown that natural ponds with intermediate hydroperiods (i.e. temporary or vernal pools) have the highest species richness of amphibians (Snodgrass *et al.* 2000). Too short a hydroperiod prevents anuran metamorphosis whereas permanent waters are colonized by or stocked with predators such as fish, predatory insects, and large frogs.
- 2) Fish presence will have a negative effect on anuran richness. Reduced species richness has been correlated with the presence of large predators such as fish (Wellborn *et al.* 1996; Hecnar & McLoskey 1997; Snodgrass *et al.* 2000). Stocking large, permanent pools with fish for recreational purposes as well as mosquito control is common in permanently ponded stormwater basins.
- 3) Anuran presence will be influenced by local and landscape characteristics that affect water chemistry and the quality of terrestrial habitat. Anuran richness will be positively correlated with residential versus commercial land use of the drainage area, percent of emergent vegetation in the basins, percent of unmanaged near-shore vegetation, and connectivity to forested upland. Richness will also be positively correlated to the size and age of the basin due to a direct relationship to the quantity of resources and by increasing the probability of colonization over time.
- 4) Permanently ponded basins are sinks for anuran species other than bullfrogs (*Rana catesbeiana*) and sources for bullfrogs. In this context, a sink is defined as a pond where mating calls are heard, but no larvae are found. A source is defined as a pond where larvae are found. It is expected that a majority of permanently ponded basins will

be stocked with fish. *R. catesbeiana* has been shown to be resistant to predation by several fish species (Kats *et al.* 1988; Kurzava & Morin 1998; Gunzburger & Travis 2005). In addition, bullfrog populations are facilitated by centrarchid sunfish in their native and introduced range (Werner & McPeck 1994; Adams *et al.* 2003). This positive indirect effect is accomplished through two pathways. Sunfish feed on large invertebrate predators that are a major mortality force on bullfrogs and they also feed on other anuran larvae, which are competitors for bullfrog larvae.

## METHODS

### Identification of Study Basins:

The study area is within Ocean County, New Jersey from Brick Township in the north to Stafford Township in the south. Route 9 approximately defines the eastern boarder. The western border follows a diagonal from 74° 14' 16" in the north to 74° 19' 30" in the south near Manahawkin. This section of Ocean County is part of the outer coastal plain physiographic region.

Study basins were chosen from the 2002 NJDEP Land Use/Land Cover Update map and the 2006 New Jersey high-resolution aerial orthophotos to identify a subset of basins with ponded waters. Basins that are impoundments within streams were excluded, as this placement is no longer permitted in New Jersey. Basins that are features in golf courses were also excluded due to the difficulty of receiving permission from landowners. Basins owned by the Department of Transportation were excluded as ponded basins in the study area were located in cloverleafs with very high traffic volume. Basins without ponded water in March were also excluded. The choice of basins was limited to those for which permission was granted or access available. Admittance was available to 43 basins. Two of these were later excluded from the study due to excessive travel time. An additional two basins did not have inflow or outflow structures and were excluded. Thirty-nine basins remained in the study. My terminology for stormwater basins is defined according to hydroperiod. Temporary basins or ponds hold water less than the entire growing season. Permanently ponded basins or permanent ponds refer to stormwater basins that hold water the entire year. Engineering nomenclature such as detention or retention basin will not be used as many detention basins designed to drain



completely do not perform as built (Gregory et al. 2006).

The Pinelands National Reserve (PNR) covers a large portion of the study area. Nineteen sites are within PNR, albeit most within designated growth, development, village or town areas. One site is within a designated PNR forested area. Basins are located in highly urbanized landscapes with only scattered forest remnants as well as on the edges of developed areas next to large tracts of forest.

### Species Survey

Anuran mating presence was determined by nine vocalization surveys conducted from April 8, 2008 until July 9, 2008 at least once every two weeks with the exception of one basin. Vocal surveys for that basin, Aldi, began on May 31, 2008. Care was taken to choose optimal weather conditions for mating choruses. The number of calling individuals heard in a three-minute period was estimated using a ranking system where 0 = none, 1 = 1 calling, 2 = 2-5 calling, 3 = 6-10 calling, and 4 = >10 calling individuals. Vocalizations were taped on a digital recorder with a Sennheiser shotgun microphone and archived for validation. Tadpole presence was established by dipnetting and trapping. Dipnetting took place three times from May 21, 2008 to July 3, 2008. Ten one cubic meter sweeps were sampled at all microhabitats within each pond with a 12" x 7" D-ring dipnet. Species accumulation curves at three locations concluded that ten 1m<sup>3</sup> sweeps were sufficient to adequately sample each location. Three minnow traps were placed in each basin for 24 hours at the end of June or beginning of July. Tadpole specimens that could not be identified on site were taken to New Brunswick for identification at Dr.

Peter Morin's lab. Small individuals were raised in aquarium tanks until they could be identified, then returned to the field from their original collection point.

#### Within Site Characteristics (Table 1)

Fish presence was assessed by visual encounters, presence in minnow traps and dipnets, as well as interviews with managers, anglers, and the New Jersey Mosquito Commission.

Temperature pH, and conductivity levels were measured three times (May, June-July, and October 2008) with a Hanna HI 98129 combination tester (resolution 0.1°C, 0.01 pH, 1µS/cm). Water was sampled one foot below the surface approximately three meters from the shore. Three exceptions occurred: the May sample was missed for two ponds and the October sample was missed for one pond.

Pond area was measured by walking along the basin perimeter with a global positioning system. Both units used ArcPad software (ESRI, Redlands, CA) to interface with ArcGIS. The Trimble Pathfinder GPS receiver was used with differential correction for an accuracy of 2 to 5 meters. The Magellan MobileMapper 6 performs with a real-time accuracy of 2 to 5 meters. Satellite photos supplemented the GPS data when the results were inaccurate due to inaccessibility or canopy cover (NJDEP 2002 leaf-off and 2006 leaf-on orthophotos). Area was calculated in ArcGIS. Stormwater ponds were visited at least once every three weeks from April through October 2008 to determine pond permanence with the exception of Aldi basin, which commenced observations the end of May. Inspections were made before rain events to ensure accuracy. A pond was considered permanent if water depth did not drop below 10 cm.

Percent of emergent and floating vegetation was visually estimated once in September of 2008. Five transects were created across the shortest axis of the pond in ArcGIS 9.3 software (Environmental Systems Research Institute, Redlands, California, USA) then transferred to a GPS unit. Presence of emergent or floating vegetation was measured along the length of each transect either physically or with a digital rangefinder (Leupold RX-II). The length of all populated sections were added together then divided by the total length of all five transects.

One site was locked before emergent vegetation could be measured. Therefore, I estimated the vegetation along the ArcGIS transects from the 2006 digital orthophotography (8-bit RGB leaf-on photos with a 2 meter resolution). I verified accuracy by estimating coverage from the photos at five additional basins with vegetation. I calculated the error between the photo estimate and the ground estimate and reduced the estimate of coverage on the locked basin accordingly.

Positive correlations between amphibian species richness and the percent of native trees along the perimeter of natural ponds have been found in past surveys (Hecnar & M'Closkey 1998; Werner *et al.* 2007a). Vegetation within 100 ft of the basin was categorized by management regime and cover type. I used the 2006 New Jersey orthophotography in ArcGIS to divide the buffer area into polygons with a minimum mapping unit of 50m<sup>2</sup>. The categories were: impervious surface, managed grass, managed woody vegetation, unmanaged grass, and unmanaged woody vegetation. Unmanaged grass was not mown, but consisted of a mix of turf grass, native grasses and herbs. Unmanaged woody vegetation was defined by the presence of natural litter.

Ground truth validation of vegetation polygons was conducted by generating thirty random points for each category except managed grass, which consisted of only sixteen polygons. The points were located in the field via GPS. I walked a 30 meter transect toward the center of the polygon establishing the landscape category at each three meter interval. A contingency error matrix of the validation survey resulted in 85% of the classified data to be correct. The classification of the buffer areas was corrected according to the ground truth information before final calculations of percent cover. Polygon areas were calculated in ArcGIS then the data was imported into Microsoft Excel (2004) for determination of percent cover.

Depth of ponded water was measured at least once every three weeks from April 9, 2008 until October 12, 2008. Beginning July, hydrology was assessed prior to rain events and as far from a previous rain event as possible. Basins with less than 10 cm of water were considered temporary; all others were considered permanently ponded.

Specific age information for all basins was unavailable. Use of the 2006, 2002, 1995 aerial orthophotographs (NJDEP), and 1986 and 1972 aerial photographs (Ocean County Soil Conservation Department) resulted in a small set of uneven bins. The youngest basins were between two and six years old. The oldest were older than 23 years, but younger than 36. Due to the high error inherent in this variable, it was removed from the analysis.

Whenever possible interviews with managers were undertaken to ascertain what treatments, such as algaecides, herbicides, or pesticides were added to the water and surrounding areas or whether any other issue was known that might contribute to the condition of the pond. Chemical algaecide use (copper sulfate) was added to the analysis

as a factor because a distinct lack of activity was observed in the seven basins where copper sulfate was known to have been applied.

#### Landscape Variables (Table 1)

Pond-breeding anurans depend on upland forests to complete their lifecycle (Stebbins and Cohen 1995). In addition, moist microclimates are essential to avoid dehydration during dispersal, emigration, and immigration (Duellman and Trueb 1986). Measures of landscape composition and configuration have been associated with the likelihood of anuran occurrence at natural breeding ponds (Hecnar & M'Closkey 1998; Guerry & Hunter 2002). In urban settings an inhospitable matrix often surrounds stormwater ponds. Consequently, viable access to forested uplands is likely an important predictive factor for anuran presence at stormwater ponds (Ray *et al.* 2002; Herrmann *et al.* 2005).

Connectivity was characterized by two metrics: distance to corridor and percent of undeveloped upland. Both variables used 2002 NJDEP landuse/landcover maps modified by data in the 2006 orthophotos. Distance to corridor was measured as the shortest distance from the pond perimeter to any minimally interrupted continuous tree canopy that terminated at a forest. Terminal forests were at least 1.48 acre. One short break in the tree canopy occurred at two sites: over a tertiary access road and a suburban backyard. Distances were calculated in ArcGIS with the COSTools extension 1.0 (City of Scottsdale GIS).

Percent of undeveloped upland was calculated within 500 meters of the basins. Although maximum distances traveled by anurans far exceeds this number, the frequency

distribution of anuran movement is highly skewed; shorter distances are more frequent (Smith & Green 2005, 2006; Rittenhouse & Semlitsch 2007). The landscape within 1000 meters of study basins incorporated highly isolated patches of remnant forest, which would estimate a higher proportion of undeveloped upland than is feasibly accessible. Five hundred meters was adopted as a buffer distance from Price et al. (2005). It is below 1000 meters, yet still within their landscape scale metrics. All areas defined by the NJDEP as forest or wetland remained in the calculation with the exception of those categorized as disturbed wetland, cemetery on wetland, managed wetland in maintained lawn greenspace, managed wetland in built-up maintained recreation area, and those developed since 2002.

Work by several authors has shown road traffic to be destructive to amphibian populations by increasing mortality rates (Hels & Buchwald 2001; Mazerolle 2004; Elzanowski *et al.* 2009). Hels and Buchwald (2001) calculated amphibian mortality risk by traffic load (average total vehicles per day) and amphibian movement rate. Major and secondary roadways bisected the 500 meter buffer surrounding many basins in this study. I used the probability of mortality rates developed by Hels and Buchwald (Figure 1) to reduce the available forested upland across major highways (as defined by NJDEP) and major secondary roads. The Hels and Buchwald 2001 study concluded that traffic in excess of 15,000 per day resulted in a mortality rate close to 100%. Therefore, I eliminated forested areas from the 500 meter buffer if they were located across roads with traffic counts above 15,000. For major secondary roads with lower traffic volumes I averaged the mortality probability of both slow and fast moving species from Figure 5 of their paper. This resulted in a reduction of 65.5% of upland across roads at six basins and

61.5% at three additional basins. Traffic load was obtained from NJDOT and a consulting firm for Stafford County at locations on the specific road crossing my sites or at a road feeding into the road crossing my site within 1,400 meters. One road had no nearby data. I used the traffic count from a road within the study that was similar in so far as number of lanes, amount, and kind of development. Areas were computed in ArcGIS and percent cover was calculated in Excel.

### Data Analysis

Determination of whether stormwater basins are sources or sinks for individual species was conducted with an exact binomial test. I tested the probability of the source - sink ratios being more extreme than 1:1 in a one-tailed test.

Correlation analysis was used to measure the strength of the relationship between permutations of species richness (e.g., the number of species mating) and all continuous variables. Transformation did not improve the departure from normality evidenced by most variables, consequently Spearman's rank correlation was chosen as a nonparametric test of association to test the correlation between the number of species mating and size of basin, emergent cover, percent of undeveloped upland within 500 meters, distance to corridor, average conductivity, average temperature, average pH, and the five local landscape variables. All analysis was completed on untransformed data. To reduce Type 1 error associated with multiple testing, the sequential Bonferonni significance level adjustment was applied to the correlations.

Kruskal-Wallis tests were performed to examine the effect of land use (residential/commercial) and algaecide use on the number of species mating. A Brown

and Forsythe test, which is also referred to as a Modified Levene test, was used to test the equality of the variances.

Spatial autocorrelation between the number of species mating and the distance between basins was examined via a variogram. The variogram illustrates how differences in a measured variable  $Z$  vary as the distances between the points at which  $Z$  is measured increase. The pairwise distances between the centroids of all basins were calculated. A lag distance of 1,000 feet was chosen, which separated the pairwise distances into 9 bins. Although this resulted in some bins with small numbers of pairs, which can lead to unreliable variances, the shorter, more relevant distances were addressed. The variance of the pairwise differences in the number of species mating was calculated for each distance bin. Due to the late addition of Aldi pond in the vocal surveys, it was excluded from all analysis of multiple species.

The relationship between all habitat variables and the presence or absence of individual species found mating at the basins was explored to identify which species contributed to the general model or whether the presence of some species was better described by other variables. Stepwise selection of variables is generally used when the objective is to describe differences among groups in the most parsimonious manner possible (McGarigal *et al.* 2000). I conducted a forward stepwise logistic regression for all individual species using a significance level of  $\alpha = 0.05$  for all criteria. Fit of the models was assessed with the Hosmer-Lemeshow (H-L) statistic. According to the H-L statistic, a good model produces a nonsignificant result (Tabachnick & Fidell 2001). Some researchers find stepwise methods to have severe drawbacks. Variables excluded in the final model may sometimes be more important than those included in the model by



the stepwise process. This happens because the criteria for inclusion in the model is simply that one variable is a better predictor regardless of the magnitude of improvement. Very small differences may be insignificant to the question being asked (Meyers 2006). I employed a classification and regression tree analysis for individual species to support the logistic regression analysis (CART 6.0, Salford Systems, San Diego, CA).

Classification and regression trees are a valuable complement to traditional statistical techniques due to their flexibility to handle a broad range of response types and their invariance to monotonic transformations of the explanatory variables (De'ath & Fabricius 2000). A classification or regression tree is built through an iterative process of splitting the data into partitions, and then splitting it up further on each of the branches. At each split the data is partitioned into two mutually exclusive groups, each of which is as homogeneous as possible. The objective is to partition the response into homogeneous groups, but also to keep the tree reasonably small. Excepting CART analysis, all tests were conducted in SAS 9.2 (SAS Institute, Inc., Cary, NC).

## RESULTS

### Stormwater Basins as Sources or Sinks for Anuran Species

The overwhelming majority of basins were permanently ponded (36/39). In addition, one of the three temporary ponds had water quality issues due to petrochemical contamination. Therefore, the analysis was conducted on the sample of permanently ponded basins (n = 36).

A total of seven species vocalized at the basins: *Bufo woodhousii fowleri* (fowler's toad), *Rana catesbeiana* (bullfrog), *Pseudacris crucifer crucifer* (spring peeper), *Hyla versicolor* (northern gray treefrog), *Rana clamitans* (green frog), *Rana palustris* (pickerel frog), and *Rana sphenoccephala* (southern leopard frog). All species are considered wide ranging in the state of New Jersey (Gessner and Stiles 2001). Two groups clearly emerged from the exact binomial test: *Bufo woodhousii fowleri* and *Rana catesbeiana* successfully reproduced in the basins, *Pseudacris crucifer crucifer*, *Hyla versicolor*, and *Rana clamitans* did not. *R. clamitans* did not have a significant result due to small sample size, but did not successfully reproduce at any basins. *Rana palustris* and *Rana sphenoccephala* were untestable with only one occurrence. (See Table 2 for counts and statistics, Figures 2 and 3 for frequency.)

### Number of Species Mating in Permanently Ponded Stormwater Basins

The species richness of anurans that successfully reproduced had very little variation. Therefore, the measured variables were tested to ascertain if any contributed to a greater number of species mating at the basins. In addition, lack of variation in the data excluded fish presence from statistical tests. Fish inhabited thirty-three out of thirty-six

basins. Highly predatory fish such as largemouth bass (*Micropterus salmoides*), catfish (*Ictalurus or Ameiurus spp.*) and/or sunfish (*Lepomis spp.*) were found in twenty-seven basins. *Gambusia* species were found in two additional basins. Other predators were sighted such as crayfish, turtles, *Belostoma spp.*, and Anisoptera naiads. With the exception of Anisoptera naiads, predators other than fish inhabited a maximum of two basins each.

Spearman's rank correlation found distance to corridor was negatively correlated with the number of species mating ( $r = -0.47474$ ,  $p = 0.0040$ , See Figure 4) as was the percent of managed grass ( $r = -0.39978$ ,  $p = 0.0173$ ). Conversely, percent undeveloped upland had a significant positive correlation with the number of species mating ( $r = 0.54913$ ,  $p = 0.0006$ , See Figure 5), as did average temperature ( $r = 0.34800$ ,  $p = 0.0405$ ) and percent of unmanaged woody vegetation ( $r = 0.33790$ ,  $p = 0.0471$ ). With corrections for multiple tests, only the percent of undeveloped upland and distance to corridor remained statistically significant. Distance to corridor and percent of undeveloped upland were not significantly correlated ( $p = 0.1713$ ).

Based on Kruskal-Wallis tests, the mean rank of commercial versus residential basins was not significantly different ( $H = 1.4311$ ,  $df = 1$ ,  $p = 0.2316$ ). However, the mean rank of species mating in basins without chemical algaecides was significantly higher than that of species mating in basins with algaecide use ( $H = 10.2610$ ,  $df = 1$ ,  $p = 0.0014$ ). Only one *R. catesbeiana* individual called at one time and no tadpoles of vocalizing *B. woodhousii fowleri* were found.

The variogram indicates spatial autocorrelation at basins in most lag classes until an average distance of 5,836 feet between basins is reached. The exception is the third lag

class with only one pair, which is an unreliable measure (Fig. 6, Table 3). No correction for autocorrelation was employed as the alpha values utilized in the correlation analysis were already robust ( $\alpha = 0.004167$ ).

### Individual Species Presence and Basin Variables

Percent of undeveloped upland within 500 meters was the only variable that contributed to the stepwise logistic regression model for *P. crucifer* (H-L = 0.2893; Wald Chi-Square = 6.4993, df = 1, p = 0.0108, Odds ratio = 1.094). There is a 9.4% higher odds of *P. crucifer* mating per each 1% increase of undeveloped upland (Table 4).

The classification tree for *P. crucifer* created a major split with eight of the ten occurrences of *P. crucifer* at ponds less than 6.3 meters from a corridor and with undeveloped upland greater than 12%. Spring peepers also mated in two basins with average conductivity less than 52 $\mu$ S/cm and distance to corridor greater than 6.3 meters. Due to the fact that an importance factor of 100 was assigned by the CART analysis to distance to corridor and an extreme outlier appeared in the data, a Mann-Whitney U Test was performed to establish that the median distance to corridor of basins with *P. crucifer* mating was lower than ponds without *P. crucifer* mating (U = 92.0000, Z = -3.2848, p = 0.0010).

CART analysis determined that the percent of managed woods within 100 ft of the basin was the single variable contributing to the presence of *H. versicolor*. All occurrences of *H. versicolor* were in basins with less than 1.5% of managed woody vegetation. Stepwise logistic regression placed then removed this single variable from the model. A logistic regression with the single variable found a significant effect (Wald Chi-

Square = 11.3535,  $df = 1$   $p = 0.0008$ ) with an odds ratio of 0.328 indicating a 32.8% decrease in the odds of *H. versicolor* mating per 1% increase in managed woody vegetation (Table 5).

*R. clamitans* appeared at only three basins. The stepwise logistic regression created a model with one variable, average temperature. However, the validity of the model was questioned due to complete separation of the data points. A binary logistic procedure on distance to corridor and percent of undeveloped upland found undeveloped upland to be statistically significant (Wald Chi-Square = 5.4722,  $df = 1$ ,  $p = 0.0193$ , Odds ratio = 1.082; Table 6). Quasi-complete separation of the data points was detected in the procedure for distance to corridor. The CART model's error was 1.182, which indicates that it is worse than random. Too few presence points may exist for this species to return meaningful information.

*R. catesbeiana* mated in sixty-four percent of the basins. Stepwise logistic regression found distance to corridor and percent of managed woods to be significant (H-L = 0.7155; distance to corridor: Wald Chi-Square = 5.4915,  $df = 1$ ,  $p = 0.0191$ , Odds ratio = 0.955; managed woody vegetation: Wald Chi-Square = 3.9042,  $df = 1$ ,  $p = 0.0482$ , Odds ratio = 1.276; See Table 7). The effect of distance to corridor is negative, but very small. The odds of *R. catesbeiana* mating increase with a higher percentage of managed woody vegetation surrounding the basin. The CART model for the *R. catesbeiana* described the main split for presence points to be stormwater ponds with undeveloped upland greater than 11% and managed woody vegetation greater than 50%. The error of the model was somewhat high at 0.602. A binary logistic regression of presence/absence

of *R. catesbeiana* and percent of undeveloped upland did not return a significant effect ( $p = 0.0953$ ).

The stepwise logistic regression for presence of *R. catesbeiana* tadpoles returned no effects at the 0.05 significant level.

*B. woodhousii fowleri* was the most frequent visitor to the stormwater basins. Average temperature was the only variable found in both the CART and logistic models (H-L 0.5206; Wald Chi-Square = 6.0929,  $df = 1$ ,  $p = 0.0136$ , Odds ratio = 3.549, See Table 8). The CART analysis found the majority of presence points to be above 22.88° C, which supports the positive odds ratio in the logistic analysis. The average temperature in ponds with *B. w. fowleri* mating was only slightly higher than the average temperature of all ponds (23.63°C versus 23.20°C). This would seem to be too slight to be meaningful. However, the average temperature at ponds where *B. w. fowleri* tadpoles were found continued the trend (23.88°C). In addition, the stepwise logistic regression for presence of *B. woodhousii fowleri* tadpoles placed average temperature as the only variable in the model (H-L 0.0629; Wald Chi-Square = 6.8083,  $df = 1$ ,  $p = 0.0091$ , Odds ratio = 2.385, See Table 9).

## DISCUSSION

The overwhelming presence of fish in the permanent ponds dictated whether a species' larvae were able to persist in these basins. Both *B. woodhousii fowleri* and *R. catesbeiana*, the only species that were found with successful recruitment in the permanent stormwater ponds, elsewhere have displayed resistance to predation by fish (Kats *et al.* 1988; Kurzava & Morin 1998; Gunzburger & Travis 2005). In addition, an experimental study by Kurzava and Morin (1998) found that fish significantly decreased relative abundances of *H. versicolor* and *P. crucifer*. *Gambusia* species have been implicated in the exclusion of endangered native anurans (Hamer *et al.* 2002). Other studies have found direct and indirect effects of fish presence on larval anuran fitness (Werner & McPeck 1994; Lawler *et al.* 1999; Adams *et al.* 2003; Boone *et al.* 2007; Pope *et al.* 2008). These studies generally found increased *R. catesbeiana* larval survival in the presence of fish and/or decreased survival or fitness for other *Rana* species in fish presence.

Werner and McPeck (1994) found green frogs to be unpalatable to sunfish (but see Kats *et al.* 1988). Although sunfish were the most common fish in the basins, the three ponds where green frogs vocalized contained bass and crayfish, bass and turtles, or turtles alone in addition to sunfish. It is difficult to attribute the lack of green frog larvae wholly to fish presence owing to the range of predators and the small sample of presence points.

Anurans require both aquatic and terrestrial habitats to complete their life cycles. This study established that connectivity to quality terrestrial habitat was a significant predictor of the number of species mating in these basins. It supports the findings of

Eigenbrod et al. (2008). They presented a new measure of the combined effects of roads and habitat amount by defining accessible habitat as the amount of habitat that can be reached from a pond without crossing a road. This study quantified roads as a major barrier to upland movement via traffic load and found the quantity of accessible habitat within 500 meters to be a significant predictor of the number of species mating. This study also found another measure of connectivity, distance to a canopied corridor that terminated at a forest, to be a significant predictor of the number of species mating in stormwater basins. This metric reflects the difference between the siting of natural ponds and structural stormwater basins.

Natural ponds usually occur in wetlands or as a part of a lotic system protected by a buffer area however modest. An assumption exists that portions of the landscape matrix are sufficiently hospitable to allow dispersal, immigration and emigration. Barriers such as roads or agricultural fields are sometimes used as a measure of connectivity by measuring a break in connectivity (Trenham *et al.* 2003; Ficetola & De Bernardi 2004; Parris 2006), but most studies use distance alone or density as the defining metric for connectivity (Ficetola & De Bernardi 2004; Werner *et al.* 2007b; Semlitsch 2008). These animals are small and unable to travel long distances without the threat of desiccation. Therefore, dense configurations of ponds with mandated buffer areas allow greater dispersal and rescue effects since the ponds may be used as stepping stones. Stormwater basins are often built within a sea of turfgrass, housing, and roads, far from forest or shrub cover. Access to basins then requires an adequate corridor to facilitate connectivity.

The distance to a canopied corridor was an important predictor for spring peepers (*P. crucifer*). Basins utilized by *P. crucifer* were less than 15 meters from a corridor with



the exception of one outlier at 49 meters. Although not statistically significant due to uneven sample size, green frogs (*R. clamitans*) were only found where the distance to corridor effectively equaled zero. The bullfrog (*R. catesbeiana*) was also less likely to mate at ponds with large distances between the pond and a canopied corridor although the effect was very small. Bullfrogs are known to be capable of long dispersal distances (Bury & Whelan 1984). However, the quality of the terrain crossed as well as the distance is likely to be a factor (Ray *et al.* 2002). Both the Pickerel Frog (*Rana palustris*) and the Southern Leopard Frog (*Rana sphenoccephala*) were present at a basin very close to a corridor.

Connectivity measured as increased percentages of forested landscape has been found to strongly affect the occurrence and density of many anuran species in fragmented landscapes (Hecnar & M'Closkey 1998; Knutson *et al.* 1999; Guerry & Hunter 2002; Buskirk 2005). This study found the percent of undeveloped upland within 500 meters to be a significant predictor of spring peeper and green frog presence. Other studies in mixed urban/rural areas supported this finding and concluded that green frog and spring peeper occurrence was negatively influenced by the relative area of developed land and positively associated with forested upland (Price 2005, Guerry and Hunter 2002).

Connectivity variables were insignificant for the northern gray treefrog (*H. versicolor*) and the Fowler's toad (*B. woodhousii fowleri*). Fowler's toads occur in areas with loose, well-drained gravelly or sandy soils and establish small home ranges (Klemens 1993). Smith & Green (2006) recaptured nearly 70% of the toads in their study within 100 m of their initial capture site. Toads appear capable of utilizing the drier, open habitats that immediately surround most basins. Studies of the northern gray treefrog

consistently limit the migration and dispersal distances of this anuran to 200 meters (Johnson & Semlitsch 2003; Smith & Green 2005). My study found a negative association with the percent of managed woody vegetation surrounding the basin. This could be an artifact of the minimum mapping area used (50m<sup>2</sup>), which did not capture small clumps or single trees. Another explanation is that *H. versicolor* may be dependent on woody vegetation within a moderate distance of the basins and is sensitive to management practices. *H. versicolor* has been found to be highly sensitive to low concentrations of a popular broad-spectrum pesticide (Relyea & Mills 2001).

Local habitat factors were also of consequence to the bullfrog. The bullfrog appears to cope well with managed vegetation and its presence is positively associated with the percent of managed woody vegetation. As a year round resident of ponds (Bury and Whelan 1984, Hulse et al. 2001), local forage is crucial. The bullfrog may be less sensitive to management practices as tadpoles are able to survive and thrive in the presence of the same broad-spectrum insecticide that decimates *H. versicolor* (Boone & Semlitsch 2003; Boone *et al.* 2007).

Stormwater basins are built, in part, to filter contaminants and ameliorate the downstream effects of runoff water heated by impervious surfaces (NJDEP 2004). Several researchers have found anuran larvae to experience lethal and sublethal effects as the result of metal and/or chloride concentrations consistent with stormwater basin conditions (Sanzo & Hecnar 2006; Snodgrass *et al.* 2008). However, no significant correlation with the number of species mating in this study could be established. Nor was conductance a significant variable for presence of individual species as adults or larvae (*B. w. fowleri* and *R. catesbeiana*). Also, the hypothesis that differences in the use of the

drainage area (commercial versus residential) would have an effect on anuran species richness was not supported. High conductance as a consequence of road salt use may have had a negative effect on larval survival, but the effect was masked by fish predation. Average temperature had a positive effect on Fowler's toads mating at basins as well as the presence of their larva. Breden (1988) described breeding ponds used by the toads as shallow with sandy bottoms and gradually sloping banks. The association with higher temperatures may indicate this preference for shallow basins.

Extreme management practices such as the use of chemical algacides was a significant predictor of the number of species mating. Odors given off by algal blooms are suspected to be important in the orientation of the breeding migrations in some anurans (Savage, 1961). A number of olfactometer tests in various species of anurans, (Grubb, 1973, 1975) have documented the ability of migrating animals to orient towards odors from their breeding ponds. Although lack of algal odors may explain the low number of species mating, Grubb's work on Fowler's toad (1973) questions this assumption. However, the exact cues in this study were not specified, leaving the possibility open that olfactory cues other than those from algae are important to *B. woodhousii fowleri*.

Data for this study was gathered over the course of one year, therefore a case must be made for whether the results would be valid in multiple years. The hydroperiods of all ponds analyzed in this study were permanent in 2008. Permanent hydrology is essential to the presence of fish, which largely drove the results of this study. Temperature and precipitation in 2008 was very close to the annual average for the area. South Jersey 2008 monthly mean precipitation deviation from normal was +0.08 inches and monthly mean

temperature deviation from normal was +0.8°F. In addition, weather over the seven months of the study period was drier and hotter (-2.62 inches of precipitation and +3.5°F, Office of the New Jersey State Climatologist, Rutgers University, 54 Joyce Kilmer Ave, Piscataway, NJ 08854) indicating that these basins are permanent over most years under natural conditions. Furthermore, most (15/23) of the residential basins are equipped with aerators that require a minimum depth of water to operate successfully. Management typically fills the basins before they would naturally dry.

Population turnover of amphibians is poorly understood, but thought to be highly dynamic at discrete ponds in natural systems (Semlitsch et al. 1996, Trenham et al. 2003). A recent seven year study at the University of Michigan's E. S. George Reserve by Werner et al. (2007b) found the interannual turnover in species composition of larval amphibian communities to be negatively related to hydroperiod in the absence of fish. The authors also suggest that the turnover they observed was likely a result of local extinctions.

Many of the species in this study exhibit breeding site fidelity and all are wide ranging species. Breeding site fidelity in adults is strong for *P. crucifer*, and *H. vericolor* and significant for *B. woohousii fowleri* (Smith and Green 2006; Trenham et al. 2003). *R. clamitans*, was found at the same site over 60% of the time if it was present in a previous year (Trenham et al. 2003). Although post metamorphic and adult bullfrogs are known to migrate large distances, it may be limited to a small percentage in the absence of drought or adverse conditions (Jameson, 1956; Willis et al. 1956; Govindarajulu et al. 2005). Thus, in spite of legendary bullfrog migration distances, extirpation of the population from a permanent stormwater pond is unlikely unless extreme changes in management

practice take place such as the use of chemical algaecides. In light of these facts, the results of this study are likely to be consistent over multiple years.

## CONCLUSIONS

The small number of temporary basins in the sample prevented an analysis of species richness as it relates to temporary versus permanently ponded basins.

Additionally, high fish presence in the sample constrained an analysis of species richness and fish absence. Fish presence had a direct effect on which species can utilize permanent ponds as sources. Permanently ponded basins are sources for bullfrogs, but are also sources for Fowler's toads. Both species are resistant to fish predation and are able to utilize suboptimal local habitat. Anurans whose larvae are susceptible to fish predation yet mated in these basins did not successfully produce tadpoles. Permanent ponds are sinks for spring peepers and northern gray treefrogs. Although not statistically significant due to small sample size, Green Frogs, Pickerel Frogs and Southern Leopard Frogs did not produce offspring in permanently ponded basins.

Local factors such as size or age of basins, use of drainage area, percent of emergent vegetation, or percent of unmanaged near-shore vegetation was not significantly associated with the number of species mating at these basins. Connectivity to forested upland had a significant association with the number of species mating at each basin. As access to and amount of quality terrestrial habitat increase, so did the number of species mating.

Best management practice recommendations would first of all suggest building one of the alternative structural designs for stormwater basins that do not hold water permanently. Utilizing non-structural solutions for stormwater such as rain gardens is also an alternative. Care must be taken to assure all basins perform as built, as planned infiltration basins have been shown to fail (Gregory et al. 2006). Alternatively, basins

might incorporate an engineered drain that would be operated once a year. This would prevent unwanted fish stocking, yet limit the time during which the pond might be considered unsightly. Timing is crucial, however. Draining would be best in late summer after most species have metamorphosed. The placement of wet basins near populations of threatened or endangered amphibians susceptible to fish predation is inadvisable.

Considering that breeding choruses in natural landscapes are absent or undetectable in most years (Trenham *et al.* 2003), permanent stormwater basins may be sinks for more species than those detected in this study. Where bullfrogs are considered invasive, permanent ponds will likely exacerbate the problem by increasing propagule pressure.

## FUTURE CONSIDERATIONS

Smart growth or dense development, is seen by many governmental agencies and local communities as the best case scenario for species persistence in natural landscapes.

Theoretically the maximum amount of habitat is available for wildlife, yet development continues in a directed manner. However, standard or current building practices may produce indirect effects that impact anuran populations far from the edges of a development. In our attempt to create more natural built landscapes, we may be creating ecological traps where animals choose attractive, but lower quality habitat (Battin 2004). This may be the case for some stormwater basins. Although temporarily ponded stormwater basins could be sinks for fewer species, this is not proven. High fish presence in this sample of ponds may have masked the effect of other variables. Pollutants, siltation, and eutrophication may have significant deleterious effects on mating anurans and offspring (Sanzo & Hecnar 2006; Snodgrass *et al.* 2008) and should be investigated further. Short hydroperiods are likely common in other structural stormwater basins and the effect on anuran communities should be examined.

Abundance data on those species able to utilize stormwater basins should be collected and experiments designed to assess if increased competition and predation impact anuran populations in nearby wetlands.

Models for best management practices should be created and geographically targeted to preserve the persistence of historic anuran assemblages.



**Table 1.** Definitions of the local and landscape variables used in the analyses of anuran occurrence at permanently ponded stormwater basins.

Variable	Definition
Fish presence	Presence/absence
Industrial/Residential	Landuse of drainage area for stormwater basin
Temperature	Average (0.1°C resolution)
pH	Average (0.01 pH resolution)
Conductivity	Average (1 µS/cm resolution)
Pond area	Surface area of water (m <sup>2</sup> )
Emergent and Floating vegetation	Proportion of pond covered by aquatic vegetation
Vegetation within 100 ft of basin	The proportion of the landscape area populated by the particular cover type (Min. mapping unit 50m <sup>2</sup> )
Impermeable surface	Paved or built surfaces and bare ground
Managed grass	Turfgrass and/or mowed native grasses and herbs
Managed woody vegetation	Shrubs and/or trees without natural litter present
Unmanaged grass	Unmown native grasses, herbs, and turfgrass
Unmanaged woody vegetation	Shrubs and/or trees with natural litter present
Algaecide Use	Chemical algaecide (copper sulfate as active ingredient) applied at least once during season
Undeveloped Upland	Proportion of forest or wetland area within 500 meters. Reduced by probability of mortality from road kill at high traffic roads (>7,000 average vehicles per day).
Distance to Corridor	Shortest distance from the basin perimeter to any minimally interrupted (< 6 m) continuous tree canopy that terminated at a forest or forest remnant > 1.48 acre.

**Table 2.** Results of exact binomial test with expected frequencies of 1:1 (one-tailed test) to determine status of stormwater basins as sources or sinks for anuran species.

Species	Source Basins	Sink Basins	Pr ≤ p
<i>B. woodhousii fowleri</i>	22	7	0.0041*
<i>R. catesbeiana</i>	16	7	0.0466*
<i>P. crucifer</i>	1	9	0.0107*
<i>H. versicolor</i>	0	9	0.0020*
<i>R. clamitans</i>	0	3	0.1250
<i>R. palustris</i>	0	1	-----
<i>R. sphenoccephala</i>	0	1	-----

\*(significant result)

**Table 3.** Semivariogram table for number of anuran species mating and distance between basins (feet).

Lag Class	Pair Count	Average Distance (ft)	Average Species Mating	Semivariance	Covariance
0	6	279	1.41667	0.583	0.47531
1	15	949	1.73333	0.333	0.33827
2	5	1800	2.30000	0.700	0.54568
3	1	2719	2.00000	2.000	-0.98765
4	4	4083	2.62500	0.125	1.37346
5	8	4938	1.50000	0.375	0.62346
6	13	5836	1.84615	1.154	0.73884
7	2	7175	3.50000	2.000	1.17901
8	7	7995	2.14286	1.429	-0.30511

**Table 4.** Logistic regression results for *P. crucifer* mating in permanently ponded stormwater basins.

Logistic Regression <i>P. crucifer</i> Mating Maximum Likelihood Estimates					
Parameter	DF	Estimate	Standard Error	Wald Chi-Square	Pr > ChiSq
Undeveloped upland	1	0.0902	0.0354	6.4993	0.0108
			Odds Ratio	95% Wald Confidence Limits	
			1.094	1.021	1.173

**Table 5.** Logistic regression results for *H. versicolor* mating in permanently ponded stormwater basins.

Binary Logistic Regression <i>H. versicolor</i> Mating Maximum Likelihood Estimates					
Parameter	DF	Estimate	Standard Error	Chi-Square	Pr > ChiSq
Managed woody vegetation	1	-1.1138	0.9463	11.3535	0.0008
			Odds Ratio	95% Wald Confidence Limits	
			0.328	0.051	2.098

**Table 6.** Logistic regression results for *R. clamitans* mating in permanently ponded stormwater basins.

Binary Logistic Regression <i>R. clamitans</i> Mating Maximum Likelihood Estimates					
Parameter	DF	Estimate	Standard Error	Chi-Square	Pr > ChiSq
Undeveloped upland	1	0.0792	0.0338	7.0276	0.0008
			Odds Ratio	95% Wald Confidence Limits	
			1.082	1.013	1.157

**Table 7.** Logistic regression results for *R. catesbeiana* mating in permanently ponded stormwater basins.

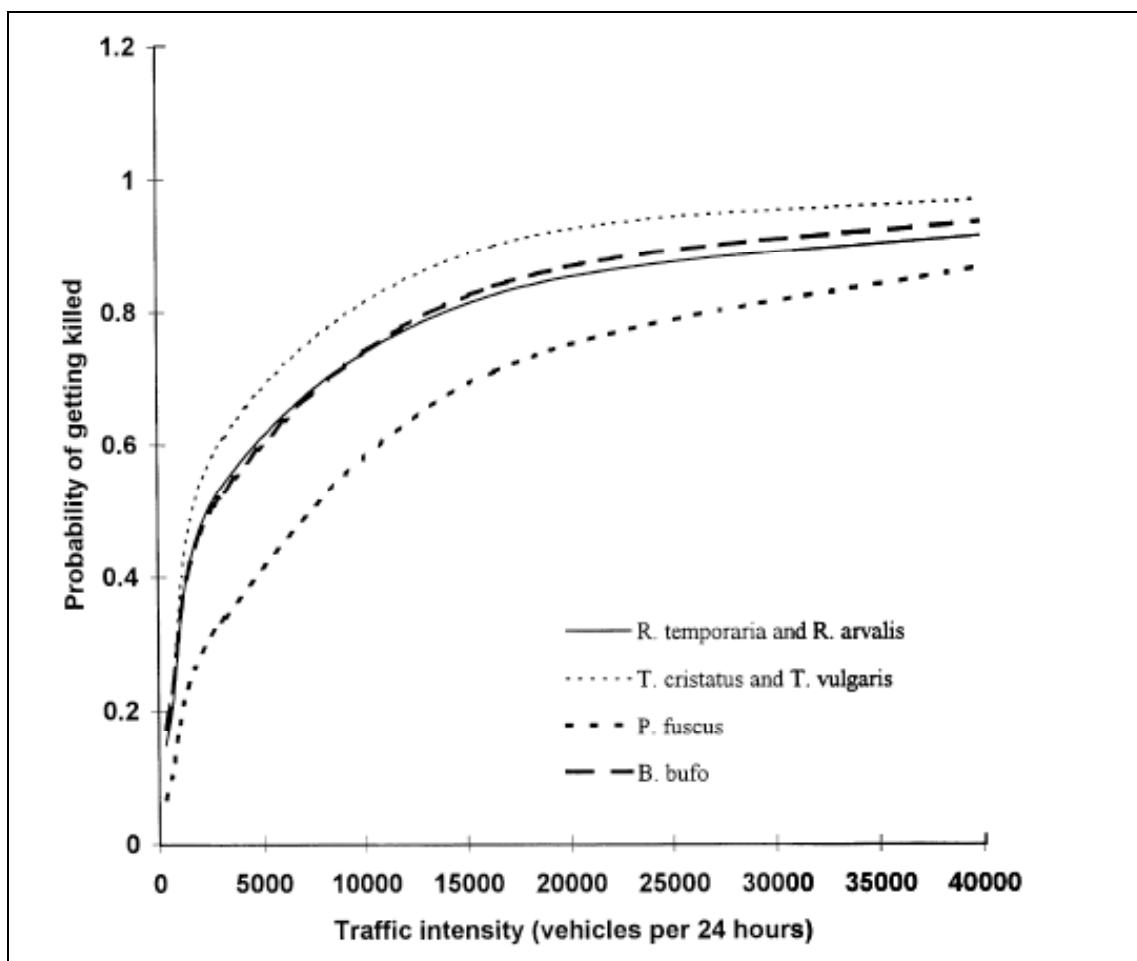
Logistic Regression <i>Rana catesbeiana</i> Mating Maximum Likelihood Estimates					
Parameter	DF	Estimate	Standard Error	Wald Chi-Square	Pr > ChiSq
Managed woody vegetation	1	0.0413	0.0311	3.9042	0.0482
			Odds Ratio	95% Wald Confidence Limits	
			0.955	0.991	0.999
Distance to Corridor	1	-0.0111	0.00555	5.4915	0.0191
			Odds Ratio	95% Wald Confidence Limits	
			1.276	1.002	1.626

**Table 8.** Logistic regression results for *B. woodhousii fowleri* mating in permanently ponded stormwater basins.

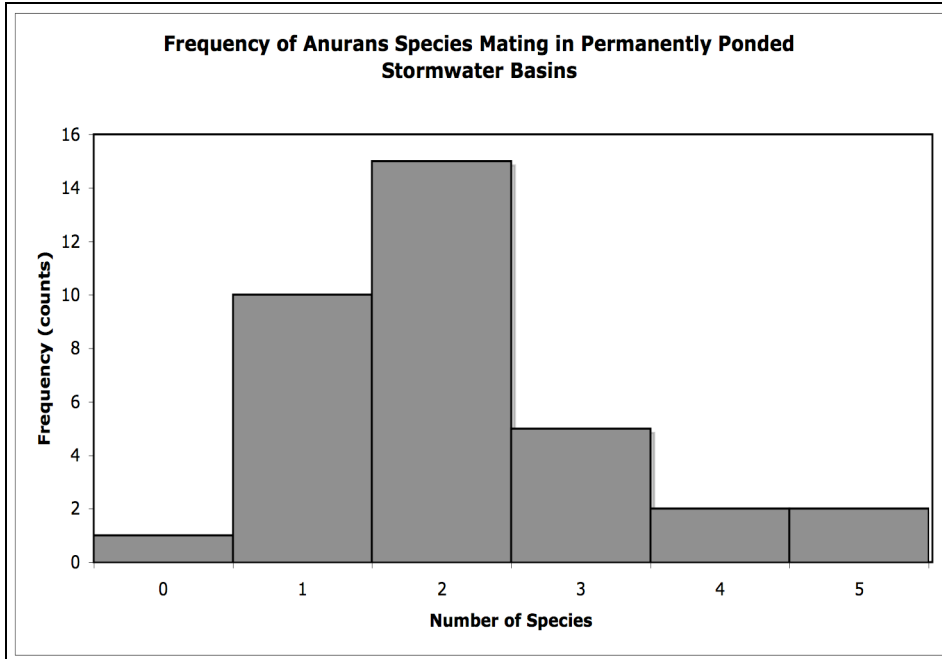
Logistic Regression <i>B. woodhousii. fowleri</i> Mating Maximum Likelihood Estimates					
Parameter	DF	Estimate	Standard Error	Wald Chi-Square	Pr > ChiSq
Average Temperature	1	1.2668	0.5132	6.0929	0.0136
			Odds Ratio	95% Wald Confidence Limits	
			3.549	1.298	9.705

**Table 9.** Logistic regression results for *B. woodhousii fowleri* tadpole presence in permanently ponded stormwater basins.

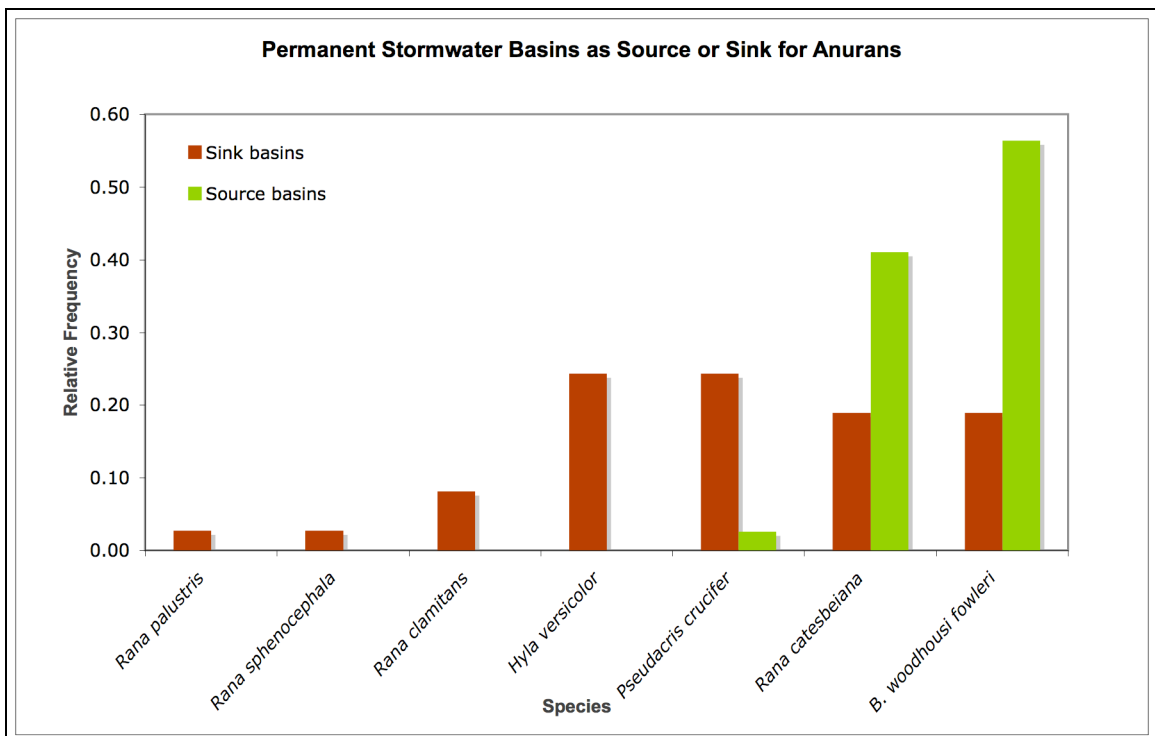
Logistic Regression <i>B. woodhousii. fowleri</i> tadpole presence Maximum Likelihood Estimates					
Parameter	DF	Estimate	Standard Error	Wald Chi-Square	Pr > ChiSq
Average Temperature	1	0.8693	0.3331	6.8083	0.0091
			Odds Ratio	95% Wald Confidence Limits	
			2.385	1.241	4.582



**Figure 1.** Probability of amphibian mortality from road traffic as a function of traffic intensity from Hels and Buchwald 2001, Figure 5.

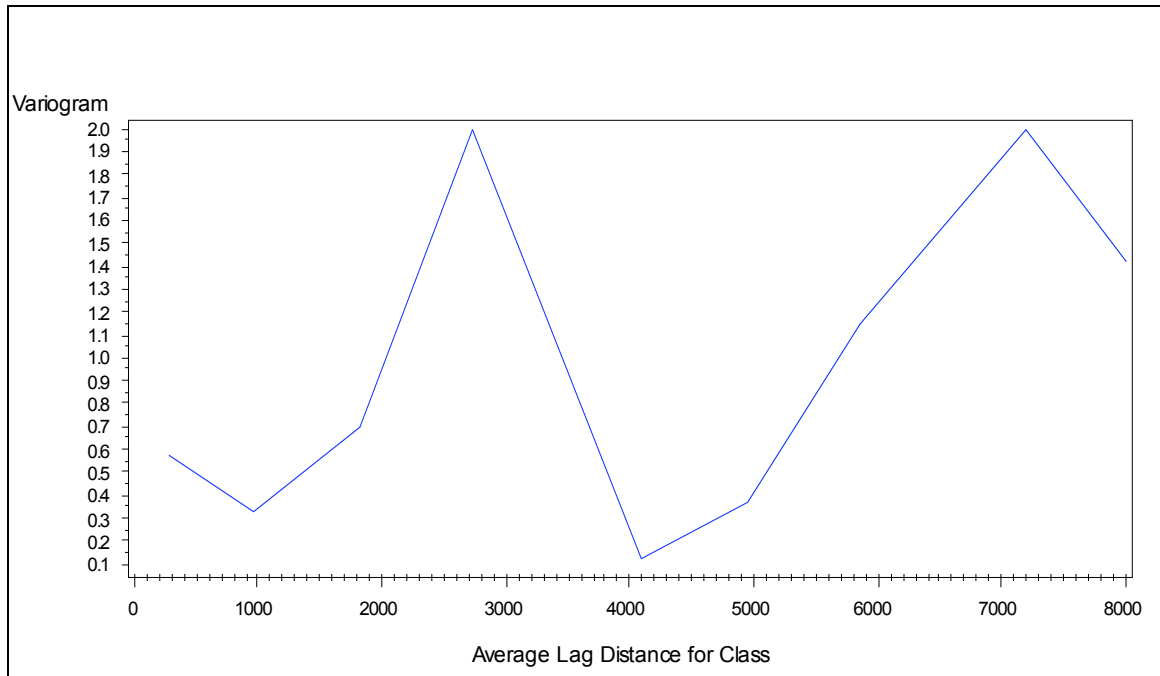


**Figure 2.** Frequency diagram of the number of species mating per stormwater basin.



**Figure 3.** Relative frequency of individual anuran species present in source basins (larvae present) and sink basins (calling activity, but no larvae present).





**Figure 6.** Semivariogram of the number of species mating and distance between stormwater ponds (distance in feet). Nine lag distance classes.



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