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Conservation of Bridle Shiner (Notropis bifrenatus) in Connecticut: Issues in Detecting an Elusive Species

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Conservation of Bridle Shiner (*Notropis bifrenatus*) in Connecticut: Issues in Detecting an Elusive Species

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CHAPTER 1

Using multi-method occupancy estimation models to quantify gear differences in detection probabilities: is electrofishing missing occurrences for a species of concern?

ABSTRACT

Bridle Shiner (*Notropis bifrenatus*) is a small, rare minnow species native to Northeastern streams and lakes. It is declining over most of its range, and currently is listed as a species of concern in Connecticut. Surveys conducted by the Connecticut state environment agency in the 1960s found Bridle Shiner at 56 locations statewide using seine nets. In contrast, surveys conducted by the agency in the 1990s, using backpack electrofishing, detected Bridle Shiner at only 8 locations. Different sampling techniques made it difficult to assess what portion of the observed decline might be a sampling artifact, confounding efforts to assess the actual conservation status. I sampled 18 habitat patches in three Connecticut watersheds in 2012 to determine if seining for Bridle Shiner yielded a higher detection probability than electrofishing. A multi-method occupancy estimation modeling approach, using program PRESENCE, quantified the probability of correctly detecting Bridle Shiner by gear, and as it covaried with habitat features. Electrofishing detection probability was lower and approximately half that of seining. The abundance of Bridle Shiner in the patch was the most supported covariate to detection, and particularly aided detection for electrofishing. Higher mean water velocity improved detection probability of electrofishing and reduced

that of seining. It is possible that the 1990s sampling underestimated the number of populations of Bridle Shiner, and a repeat survey of all historic locations using a seine is recommended.

INTRODUCTION

Evaluation of sampling methods and gears is an ongoing, if not fundamental endeavor in fisheries conservation and management (Bonar et al. 2009). There exists (and likely always will) a need to find the comparatively most precise and least biased options for quantifying the distribution and abundance of fish species and assemblages (Price and Peterson 2010), and recent efforts towards describing and sharing standard methods among fisheries biologists have emphasized the need for comparability and repeatability among sampling efforts through time and across locations (Bonar et al. 2009). However this can become complicated when uncommon habitats require non-standard methods or use of gear, and when making comparisons to historical data collected with different gears than those currently favored (Patton et al. 1998).

Two of the more regularly used gears for freshwater stream fish surveys are seines and electrofishing. Currently, electrofishing is more commonly used than seining and regarded as the most effective gear type when monitoring fish assemblages (Barbour et al. 1998; Poos et al. 2007). Furthermore electrofishing is often the only gear used in wadeable stream-based population assessments and bio-monitoring programs conducted by fisheries management agencies in the United States of America (Kanno et al. 2009). A number of studies have compared seining to backpack electrofishing (Onorato et al. 1998; Poos et al. 2007; Mercado-Silva and

Escandón-Sandoval 2013), and all found the backpack to be preferable. However, in shoreline surveys of larger lotic and embayment habitats where boat electrofishing is favored seines have been recommended (Lapointe et al. 2006; Jurajda et al. 2009).

The need for adequate sampling methods is especially important for rare, and often imperiled species where assessments can be based on determining the number of occurrences within a political jurisdiction. These are then used as criteria for designating conservation status such as threatened, endangered, etc. (e.g., Master 1991; CT DEEP 2010). Occurrences are frequently used in lieu of relative abundances or population estimates because they take less time and effort to generate, particularly for rare species (Kéry and Schmidt 2008). In rivers and streams, occurrences usually represent multiple individuals sampled from a watershed (or different flow paths within larger watersheds), such that a localized, catastrophic extirpation event at one occurrence would not necessarily affect the others.

There is an increasing appreciation of the need to account for imperfect detection, which results in false absences in monitoring datasets (Kéry and Schmidt 2008; Haynes et al. 2013). Absence of a species when it is not observed from a given area is ensured only when the probability of observation is 1, a condition rarely satisfied in surveys, particularly with elusive targets such as fish (Peterson and Bayley 2004; Haynes et al. 2013). Detectability is a complex function of the probability of individual capture (which varies widely with sampling method and effort) fish size, physical habitat, local density, and seasonal or behavioral patterns (Bailey et al. 2004). Rarity itself increases the chances of missing present species

and imperfect detection can become inflated if a species is cryptic (Mackenzie et al. 2006), has particularly discontinuous or patchy distributions (Angermeier et al. 2002) or if habitats used by rare animals are difficult to sample efficiently (Parsley et al. 1989). The probability of detecting a species is therefore an intertwined combination of sampling gear performance and imperfect detection (Bailey et al. 2004) because aspects of both the sampling and target organisms affect eventual detection probabilities (Dolan and Miranda 2003). Imperfect detection can be accounted for in an occupancy-modeling framework using repeated site surveys to generate presence-absence histories for habitat patches (MacKenzie 2005). Presence-absence data has been used for studies of distribution and range (MacKenzie et al. 2003), occupied landscape characteristics (Poos and Jackson 2012) and the spread of invasive species (Kornis and Vander Zanden 2010).

Evidence of global scale declines in minnow populations has highlighted the need for more extensive and rigorous monitoring programs to document species occurrence and detect population changes (Whittier et al. 1997). One of the more drastic minnow declines in the northeastern United States and southeastern Canada is the Bridle Shiner, *Notropis bifrenatus* (Sabo 2000). The Bridle Shiner microhabitat is associated with low velocity areas and stands of aquatic macrophytes, which are used for reproduction and predator avoidance (Jensen and Vokoun 2013). This species can be found in a variety of habitats including anthropogenic ponds, beaver impoundments, swamps, and low-gradient pool reaches of streams (Sabo 2000). Bridle Shiners carry various conservation listings including endangered, threatened or special concern throughout their range (Sabo

2000) and in Connecticut were listed as a species of concern in 2010. Little research has been conducted to offer explanations for the decline, but there is speculation that it may be due to habitat degradation, or changes in predator assemblages over time (Whittier et al. 1997; Sabo 2000).

Fish surveys were conducted by state fisheries biologists during the 1960s with a seine and detected Bridle Shiner at 56 sites across Connecticut. In the 1990s a second statewide survey was conducted using electrofishing backpack units and Bridle Shiner were detected at only eight sites (Jacobs and O'Donnell 2009). Between these surveys, electrofishing technology was rapidly developed and became increasingly popular, particularly with the availability of small lightweight backpack units (Barbour et al. 1998) that allowed researchers to work in higher water velocities, and near large woody debris that precluded seines (Onorato et al. 1998). Because Bridle Shiner frequently occupy habitats that are neither classically lotic nor lentic, and because of our personal experience with backpack electrofishing in swampy areas, I wondered if the switch in sampling gears among the historic and more recent surveys accurately depicted a precipitous range decline or if increased instances of imperfect detection could be responsible for some of the observed reduction.

The objectives of this study were therefore to (1) estimate and compare detection probabilities of Bridle Shiner when using either a seine or electrofishing backpack units within an occupancy-modeling framework and (2) model the influence of population abundance and habitat covariates on detection.

METHODS

Study site

Habitat in our sampling locations was heterogeneous and changed rapidly over short distances. Sampling patches were defined by changes in habitat or barriers that impeded movement of fish such that habitat within a patch was homogeneous and habitat in adjacent patches was heterogeneous. Sampling occurred in late July through August 2012 at 18 habitat patches across Connecticut with 11 patches along the Shunock River in North Stonington, 7 patches along Hall Meadow Brook in Torrington, and the Shepaug River in Goshen and Litchfield (Figure 1). Patches were selected based on documented occupancy and accessibility to both seine and electrofishing. The Shunock River was recently sampled (Jensen and Vokoun 2013), is a local stronghold for Bridle Shiner, and characterized as a low-gradient watercourse featuring heterogeneous habitat ranging from swamps, beaver impoundments and anthropogenic ponds. The Shunock River patches contained mostly meandering slow-flowing river reaches and a few higher gradient pool-riffle complexes. Hall Meadow Brook consisted of ponds and swampy marsh areas whereas the Shepaug River had a few swampy, low-gradient locations interspersed but mainly consisted of pool-riffle geomorphology. Most patches containing Bridle Shiner consisted of dense submerged and emergent vegetation. Substrate was variable from patch to patch, with substrate in ponds and swamps consisting of sand and silt, many sections being too deep and soft to safely wade in entirety.

Sampling design

Sampling followed protocols conducted by state biologists in previous sampling surveys. Backpack electrofishing units (Smith-Root, Inc., Vancouver, Washington, USA) set to output pulsed DC at 0.3 amps were used side by side with two additional persons netting, each equipped with 4 foot handle dip nets with 3.175 mm mesh. Sampling occurred in the upstream direction. Netted fish were held in a bucket until I reached the end of the habitat patch, at which point fish were counted, identified, and measured. A 3.175 mm mesh, 5 m wide bag seine was operated by two individuals walking downstream and captured fish were processed at the end of each seine haul. Due to the heavily vegetated banks and abundance of coarse woody debris, often seine hauls were pursed mid-water rather than being dragged ashore.

While most patches were sampled at one location, 5 larger patches were subsampled twice. Since there was an increased sampling effort at these 5 patches the mean catch per unit effort was calculated and used in analyses. Seine hauls and electrofishing passes were on average 35 m in length and catch per unit effort was defined as the number of Bridle Shiner caught over the number of hauls/passes within a patch. I conducted three surveys and visited each patch once by the seine and once by the backpack units within a given calendar week. Seining and electrofishing surveys occurred 2-3 days apart to minimize the effects of habitat disturbance.

Habitat covariates such as velocity (m/s), depth (cm), conductivity (μS), and water temperature (°C) were monitored once per week. Conductivity and water temperature were measured on site with a YSI 85 meter (Yellow Springs Instrument, Yellow Springs, Ohio) and velocity was taken with a Marsh-McBirney Flo-Mate 2000 (Hach, Loveland, Ohio). I classified sampling patches as lotic or lentic since this could potentially impact gear avoidance behavior and gear performance.

Data Analysis

Following Nichols et al. (2008), I employed an occupancy estimation modeling approach derived from detection histories from both gears to compare detection probabilities of the two methods. This approach is a likelihood-based method for estimating proportion of area (or patches) occupied when species detection probabilities are less than 1 (Bailey et al. 2004). While landscape or patch occupancy rates are often the targeted output of these modeling exercises, the multi-method occupancy framework provides an attractive way to quantify and compare detection through use of repeated site surveys.

Occupancy estimation uses two occupancy parameters ψ and θ_t, to model occupancy at two spatial scales. The first parameter ψ represents large-scale occupancy where a species has some probability of being detected given that patch is occupied. The small-scale parameter, θ_t represents the probability that the species is present in the direct vicinity of the gear at sampling occasion t. The product $\psi\theta_t$ represents the probability of small-scale occupancy indicating the presence of at least one individual exposed to detection at sampling occasion t.

Using the 'multi-method, single season' model variant, program PRESENCE 4.1 (Hines 2006) estimated detection probabilities for both gears using the data collected over the three surveys from our 18 patches. Models were created *a priori* and our first candidate set of models (Table 1) was designed to determine if there was a difference in detection between the two sampling methods. This candidate set included models with varying assumptions, and I first considered a null hypothesis that assumed species small scale occupancy was constant across surveys, and detection probabilities were equal for both gears. I investigated the effects of these parameters and compared additional models by allowing θ to vary with time and detection to vary by sampling method. These models were ranked according to AIC_c values (Burnham and Anderson 2001) and I used AIC_c weights to determine the most supported model and considered models within 2 AIC_c units of each other as competing. The most supported model from this first candidate set was then selected to be the single model structure used to determine if abundance and environmental covariates such as depth, velocity, conductivity, and temperature influenced detection probabilities. Patches were categorized as no, low, or high abundance such that patches with ~ 20 or greater percent total of all Bridle Shiner caught across the three surveys were considered high abundance and the remainder as low abundance, unless no fish were captured, which was then categorized as no abundance (Figure 2).

RESULTS

Bridle Shiners were detected in 8 of the 11 patches in the Shunock, in 1 of the 4 patches in the Shepaug and in all 3 patches in Hall Meadow Brook. The seine had perfect detection at 5 patches and the backpack had perfect detection at 2 patches. There was never an instance during any of the surveys where the backpack detected Bridle Shiner and the seine did not. Habitat covariates included an average depth of 46.05 cm with a mean flow of 0.057 m/s, average water temperature was 22.9℃, and conductivity, 103μS.

The most supported model from the first candidate set consisted of differences between methods, and no changes in detectability across the three surveys, during which θ did not vary (Table 1). This model structure was carried forward in subsequent models exploring habitat covariates to detection reported below. The small-scale occupancy parameter (θ) was estimated at 0.77, resulting in detection probability estimates for the backpack of 0.46 and 1.00 for the seine. The value of 1 for the seine was non-intuitive, since the seine only had perfect detection at 5 patches in the raw data. This result has to do with the underpinnings of the multi-method model. If both gears failed to detect the species at a patch the model considers it a true absence, but if one gear detects and one does not, then the patch is considered occupied and θ estimates the portion of the time that the target species was available for detection by the sampling gear. Therefore, given our dataset, the seine had perfect detection since there was never an instance where the backpack detected Bridle Shiner and the seine did not. Perhaps a more intuitive interpretation of these detection probabilities is given by manually fixing θ equal to

1.00. This then changes the assumptions of the model such that Bridle Shiner were considered to be always available to the gears at the time they were used. Doing so returned lower detection probabilities of 0.33 and 0.77 for the backpack and seine respectively (Figure 3).

The model that included abundance was the most supported among those that included covariates to detection (Table 3), and was the strongest single covariate to detection with both the backpack and seine having higher detection in high abundance patches (Figure 4, for parameter estimates see Table 4). The second most supported model included water velocity as a covariate to detection (Table 5). This model garnered about 10% of the available Akaike weight. Models containing other habitat covariates received relatively little support. The relationship to the two sampling methods was inverse, in that high water velocities negatively impacted use of the seine while benefiting use of the backpack and low velocities favored seine use over the backpacks (Figure 5).

DISCUSSION

Through a multi-method, occupancy-modeling framework I documented a clear difference in the probability of detecting Bridle Shiner when sampling with a seine versus backpack electrofishing. I also found, and identified environmental covariates related to gear performance. This information strongly suggests imperfect detection likely biased the 1990s electrofishing surveys in Connecticut and exaggerated the true extent of Bridle Shiner declines.

Abundance was the most supported covariate affecting detection, and was evidenced by both gears having higher detection probabilities in patches with high

abundance of Bridle Shiner. Intuitively local densities should affect the detection probability of most species however they were sampled (Bailey et al. 2004) as greater abundances would increase the likelihood of encounter. Localized population abundance was likely the most important source of heterogeneity among our patches. Although both gears showed similar responses to high abundance, the low abundance patches impacted the backpack more than the seine. At low abundances, the backpack had fewer detections compared to high abundance patches, while the seine detected the species more similarly across high and low patches. This distinction in gear performance is important for future fisheries monitoring programs, as detection at low densities is a desirable trait for sampling programs.

Water velocity was the second most influential covariate and the most supported habitat covariate to detection, where greater water velocities positively affected the backpack and negatively impacted the seine. Higher velocities can aid the visibility of the backpack netters by clearing silt and debris stirred up from the bottom downstream (Rabeni et al. 2009). With higher velocities it is perhaps also more likely a fish will roll over, allowing netters to more easily see the silvery sides of stunned fish. Other studies have shown higher velocities negatively impact seine sampling (Jensen and Vokoun 2013), and the majority of our study patches had relatively low velocities, which in part explains why the seine had higher detection probabilities. Conversely electrofishing does not work well in swampy habitats like those frequently found in our patches where turbid water can easily obscure netting visibility (Barbour et al. 1999). Substrate has been shown to have an impact on

electrofishing with mud and silt reducing efficiency compared to cobble and gravel substrate (Reynolds and Kolz 2012). While the impacts of substrate on sampling efficiency were not directly quantified, a majority of the sampling patches had unconsolidated substrates. Backpack sampling crews reported that netting visibility was an issue in these habitats since not only was the substrate a hindrance but submerged vegetation could potentially obscure stunned fish as well. Seines are known to work well in areas of submerged vegetation coverage (Lapointe et al. 2006) and are not restricted by water quality factors such as turbidity.

The widespread adoption of backpack electrofishing in monitoring programs was partially driven by the need to collect representative fish assemblage data. Seining alone offers an incomplete perspective on the fish assemblage and has known biases (Mercado-Silva and Escandón-Sandoval 2013), and several studies have concluded that electrofishing is superior to seining for describing fish assemblages (Onorato et al. 1998; Mercado-Silva and Escandón-Sandoval 2013). However, it has also been reported that mid-water schooling fish such as cyprinids are more likely to be caught with a seine than with electrofishing (Onorato et al. 1998; Lapointe et al. 2006). There may even be an advantage of some schooling fish to have group fright responses, which can aid in electrofishing shock avoidance but make them more susceptible to seine capture (Reynolds and Kolz 2012). Our study highlights an instance where a species of concern exhibited habitat use patterns that present a serious hindrance to backpack electrofishing gear, and would be better monitored using seine sampling.

I acknowledge both seines and backpack sampling protocols vary among users, often regionally, and perhaps based on past experiences and instruction. Within our study I minimized such difference by having all sampling led by the same individual during each survey. Important differences from routine monitoring were present however, and should be considered in interpreting our results. First, I chose patches based on recent Bridle Shiner occupancy research (Jensen and Vokoun 2013) so I knew from previous sampling where within the stream course these fish would most likely be detected. Previous multi-season, repeated sampling combined with mark-recapture research conducted in the Shunock River found state changes (occupied to not occupied) were rare within patches and that fish moved very little in the system (Jensen and Vokoun 2013). This knowledge may have increased our overall detection probabilities compared to routine monitoring efforts. Patches were selected using a common ground approach using only sampling locations where both gears could be employed, in particular avoiding depths too deep for the backpack. While this preserved the direct comparison aspect of our research, routine monitoring with a seine may have included deeper habitat. I also subsampled the patches with less effort, and covered less area than is reported and recommended in routine monitoring (Barbour et al. 1999; Kanno et al. 2009). Traditional monitoring generally samples longer distances to ensure characterization of the community and encounter rare species, and often pushes fishes to an obstruction or riffle in a stream (Barbour et al. 1999). Since I already knew the occupancy status of our sampling patches, I subsampled primarily for two reasons, to maximize the number of patches I could sample in the study, and to

ensure that variations in stream habitat could be quantified and used as potential covariates to detection. In routine monitoring catches are often lumped along a longer stream reach with little regard to gear effectiveness as stream characteristics change (Brewer and Ellersieck 2011).

We also acknowledge that our sampling design was not able to completely randomize the order of gear usage within patches from week to week due to the higher personnel demands and constraints with backpack electrofishing equipment. However, by allowing a 48-hour delay between samples, it was very unlikely the previous sampling would have influenced the subsequent sample. Other researchers conducting electrofishing surveys report leaving sampling locations undisturbed from as little as 15 minutes (Bowen and Freeman 1998) up to 1 hour (Schwartz and Herricks 2004) between repeat trials. Further, even when experimentally defaunated, research has shown that small areas of habitat return to original fish community conditions in a few days (Peterson and Bayley 1993).

We found the occupancy estimation modeling framework useful for quantifying detection probabilities of a species found in difficult to sample habitats. Multi-method models allowed the simultaneous use of data from both sampling gears for generating inference about method-species detection probabilities. Also our results further highlight the importance of multiple surveys when trying to get an accurate snapshot of occupancy (Pusey et al. 1998; Hense et al. 2010; Haynes et al. 2013). Often management agencies do not have the resources to conduct repeated sampling surveys; however, it is increasingly important to consider multiple surveys to evaluate species occupancy and range. Even though the seine

was found to be about twice as effective as the backpack in Bridle Shiner detection, the seine still failed to detect species presence during some surveys. Presence-absence based on only one sampling occasion is likely to underestimate the true occupancy status of habitats because species detection probabilities are most often not 100%, nor constant across species, time, space or sampling method (Pusey et al. 1998; Bailey et al. 2004).

It is difficult to compare historic surveys to present data when different sampling methods or techniques are used. Most data collection has been standardized at local, state, or provincial levels, but not across North America (Bonar and Hubert, 2002) and I suspect similar gear transitions through time are widespread. Data collected with different techniques have biases inherent to the techniques used, and while it is impossible to fully account for all bias it is still important to understand the sources and to minimize negative effects as much as possible. This is especially pertinent if management decisions are to be made regarding the conservation needs of fishes. While electrofishing is generally perceived as the most efficient, our increased reliance on this single sampling method is likely allowing species not easily sampled by this gear (such as Bridle Shiner) to be imperfectly detected at an increased rate. Local conditions, such as areas with complex habitat, may require consideration of seining as an additional sampling method. Supplemental seining can result in a more accurate assessment of the fish assemblage when done in addition to electrofishing passes (Pusey et al. 1998). Finally, when comparing historical presence-absence data to present surveys it is thus important to consider how species were detected, and if different

gears were used, to acknowledge that the detection probabilities were likely not equal to one another. Appropriate selection of sampling gear is a critical component of a biological study and the landscape of your sampling location should not be overlooked, especially if the habitat changes rapidly over short distances like in the streams sampled for this study. Results of this research have prompted the Inland Fisheries Division of the Connecticut Department of Energy and Environmental Protection to resurvey all known historical locations using a seine. In that ongoing work, some populations thought extirpated have been 'rediscovered' with seine sampling, although an overall decline in occupied locations is evident. In the future,I hope to expand seine sampling to other unsampled areas that contain habitat characteristics selected by Bridle Shiner to better understand declines in this species.

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FIGURES & TABLES

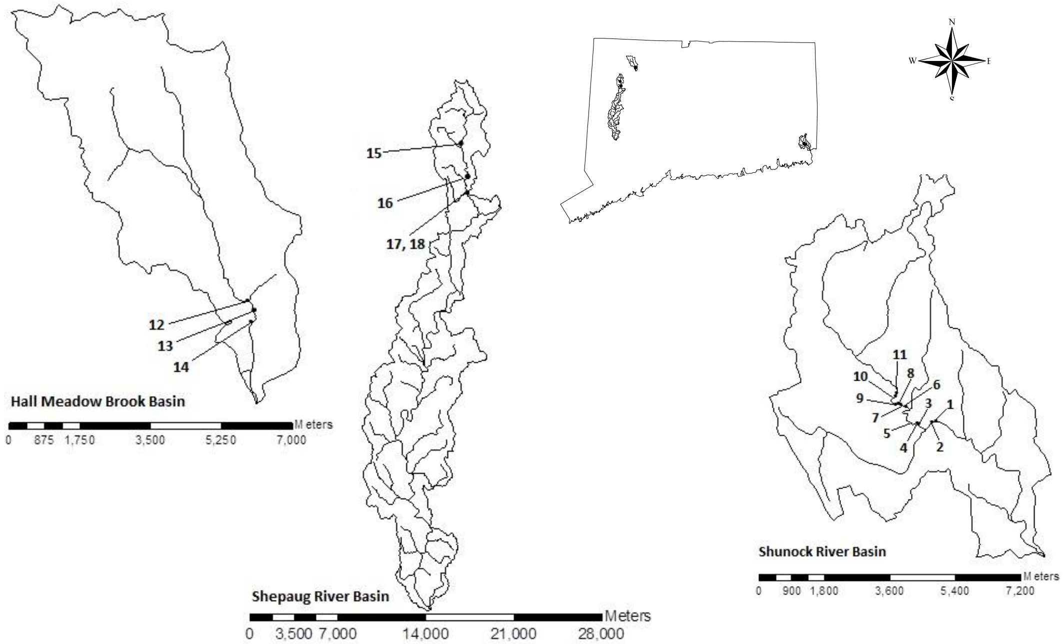

Figure 1. Distribution of sampling patches in the state of Connecticut, United States of America.

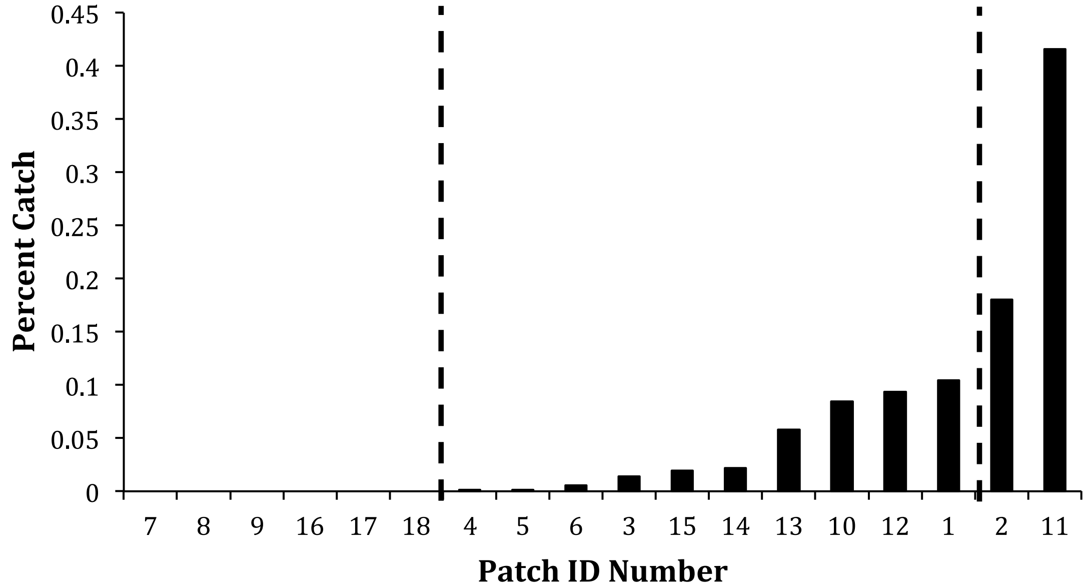

Figure 2. Low and high abundance patches determined from the percent of total Bridle Shiner caught within each patch. The dashed lines distinguish between no abundance (left), low abundance (center) and high abundance (right) patches.

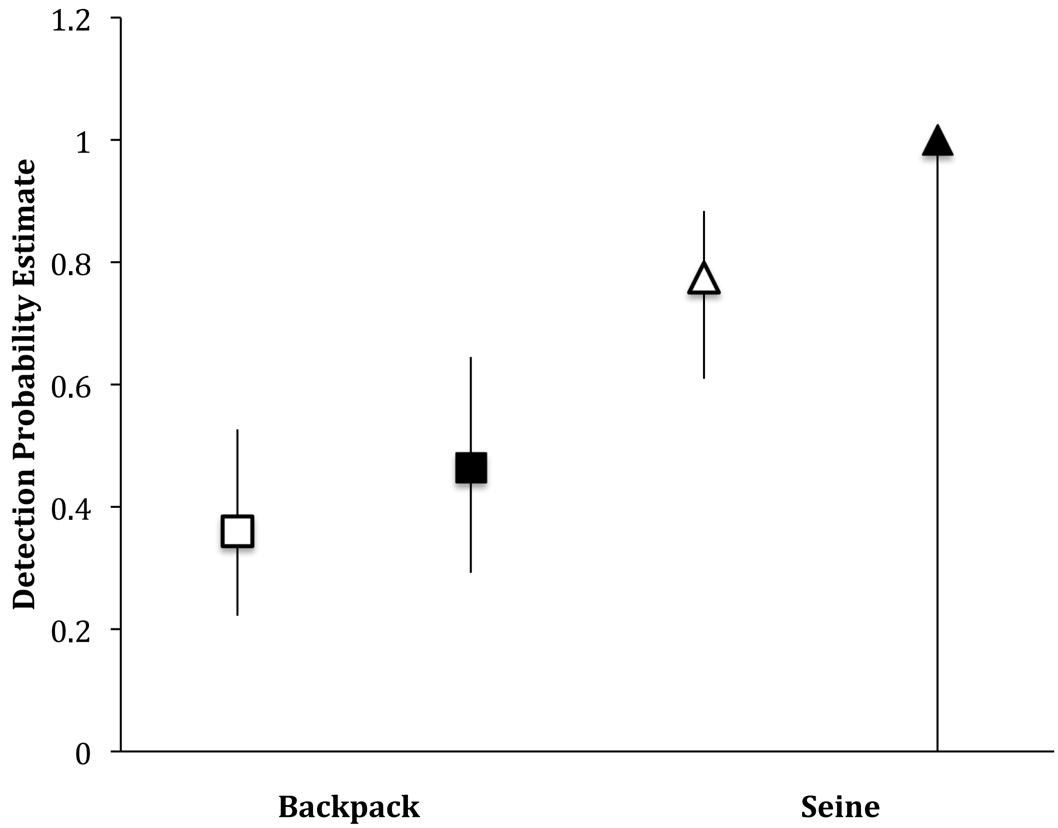

Figure 3. Detection probability estimates for the backpack electrofishing (squares) and seine sampling of Bridle Shiner (triangles). The lower values (open shapes) represent those derived from the most supported model, modified to fix θ at 1 while the higher values (colored shapes) are the unmodified detection probability estimates. Error bars represent 95% confidence intervals.

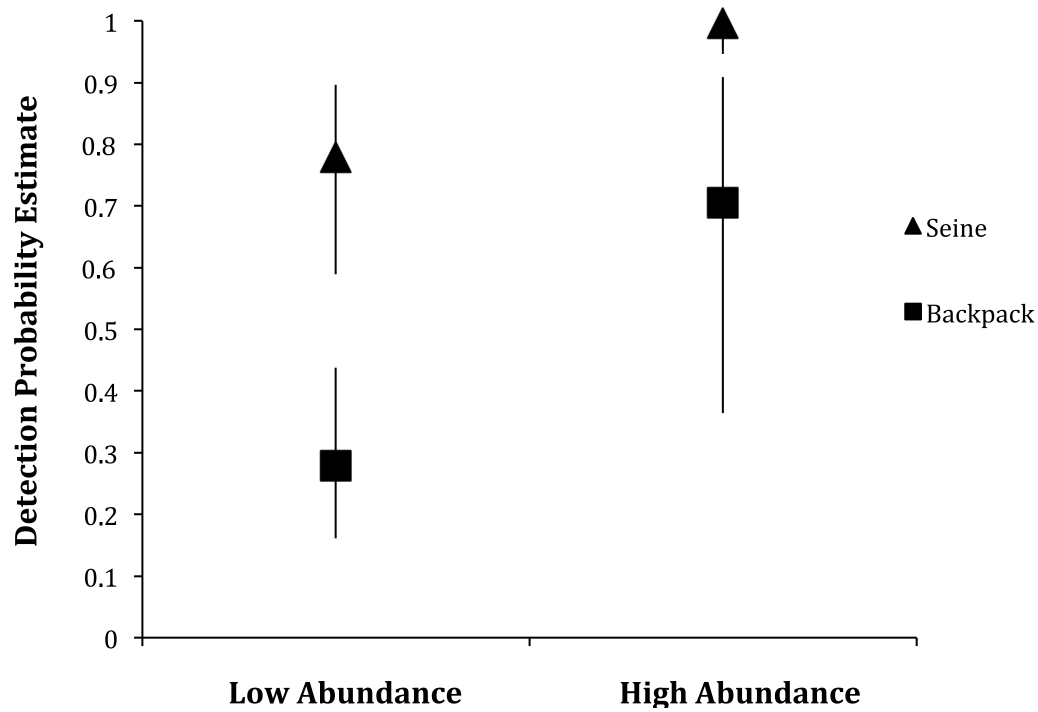

Figure 4. The effect of low and high abundance on the detection probability estimates of seine and electrofishing backpack units sampling for Bridle Shiner. Error bars represent 95% confidence intervals.

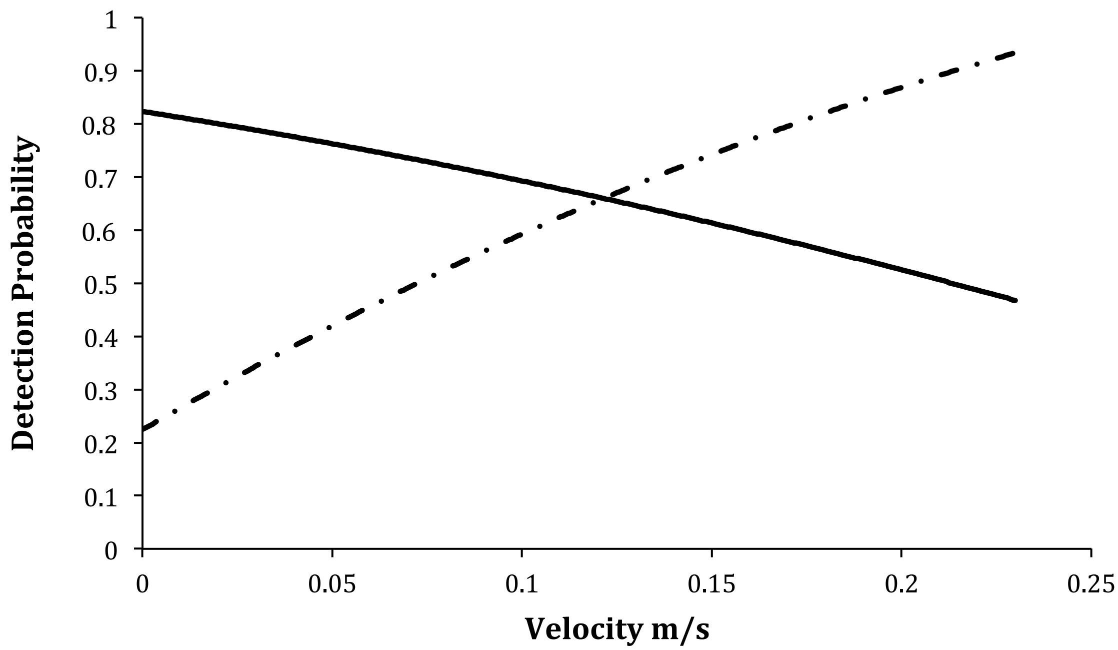

Figure 5. The effect of velocity (m/s) on detection probability estimates of seine (solid) and electrofishing backpack units (dashed) sampling for Bridle Shiner.

Table 1. First candidate set of models to test whether θ varied by time, and detection (p) varied by method, and when these parameters were held constant (.). Model rankings are based on Akaike's Information Criterion adjusted for sample size (AIC_c), differences in AIC_c (ΔAIC_c), model weight (w_i), model likelihood, and number of parameters (K).

Model	AIC_c	ΔAIC_c	w_i	Likelihood	K
Ψ,θ(.),p(method)	108.10	0.00	0.9024	1.0000	4
Ψ,θ(time),p(method)	112.55	4.45	0.0975	0.1081	6
Ψ,θ(.),p(.)	126.63	18.53	0.0001	0.0001	3
Ψ,θ(time),p(.)	130.92	22.82	0.0000	0.0000	5

Table 2. Parameter estimates from the most supported model (θ(.), p(method))from the first candidate set (Table 1) to test if there is a difference in sampling methods for Bridle Shiner.

Parameter	Estimate	Std. Error
θ	1.1974	0.4196
Backpack	-0.143101	0.3789
Seine	29.5117	699858.202

Table 3. Ranking of models with covariates based on Akaike's Information Criterion adjusted for sample size (AIC_c), differences in AIC_c (ΔAIC_c), model weight (W_i), model likelihood, and number of parameters (K).

Detection Model	AIC_c	ΔAIC_c	W_i	Likelihood	K
Ψ,θ,p(method + abundance)	94.96	0.00	0.8936	1.0000	6
Ψ,θ,p(method + velocity)	99.39	4.43	0.0975	0.1092	6
Ψ,θ,p(method + waterbody)	105.72	10.76	0.0041	0.0046	6
Ψ,θ,p(method)	107.44	12.48	0.0017	0.0019	4
Ψ,θ,p(method + depth)	107.89	12.93	0.0014	0.0016	6
Ψ,θ,p(method + temperature)	108.12	13.16	0.0012	0.0014	6
Ψ,θ,p(method + conductivity)	110.80	15.84	0.0003	0.0004	6

Table 4. Parameter estimates from the most supported model θ (p(method*abundance)) from the model set with covariates (Table 3) to test what covariates influence detection of Bridle Shiner.

Parameter	Estimate	Std. Error
θ	1.2009	0.4025
Backpack	-2.2561	0.7606
Seine	-47.2503	NA
Backpack*Abundance	1.5842	0.6538
Seine*Abundance	72.6024	NA

Table 5. Parameter estimates for the most supported habitat covariate (θ (.), p(m*velocity)) from the second model set (Table 3).

Parameter	Estimate	Std. Error
θ	1.1974	0.4196
Backpack	2.4960	1.2542
Seine	30.5025	NA
Backpack*Velocity	5.1220	2.1347
Backpack*Seine	-3.4936	NA

CHAPTER 2

Landscape correlates of localized extirpation in Bridle Shiner (*Notropis bifrenatus*)

ABSTRACT

Bridle Shiner (*Notropis bifrenatus*) is apparently declining over most of its range and is currently listed as a species of concern in Connecticut. Recent research indicated the apparent decline of Bridle Shiner in the state was in part due to changes in sampling gear used for statewide surveys. Seining used 50 years ago was demonstrably more effective at capturing Bridle Shiner than the currently favored and more frequently used electrofishing gear. The present study is a reevaluation of the species distribution in light of this recent finding. I seined at all known historic sites in Connecticut and found that some populations once thought to be extirpated are in fact extant. Nonetheless, Bridle Shiner has a sharply reduced range in Connecticut, in that the number of site occurrences has declined 60% over 50 years. Using geospatial tools I identified landscape-scale habitat measures that were potential correlates of extirpation. Using logistic regression, I investigated metrics associated with land cover change, such as impervious surfaces and those indicative of habitat fragmentation and patch isolation. I found that the current Bridle Shiner distribution in Connecticut can be explained by areas of high forest cover and low impervious cover. My results provide needed context on declines in this species and potential avenues for conservation actions.

INTRODUCTION

Fisheries biologists use analyses of species occurrences to reveal the influence of landscape and anthropogenic factors on fish distribution and community structure (Bayley and Peterson, 2001). Anthropogenic impacts such as habitat loss, degradation and fragmentation, climate change, changing land-use patterns, and introductions of nonnative species are a few of the factors that interact to influence species abundance and distribution, and have been increasingly prevalent over recent decades (Fagan et al. 2002). Knowledge of these impacts becomes particularly important as development continues at a rapid pace throughout the United States since increased development has both direct and indirect effects on the physical (Booth et al. 2002), chemical (Dietz and Clausen 2008), and biological characteristics of streams (Stanfield and Kilgour 2006). Research has found that stream health is negatively correlated with the amount of urbanization in its surrounding watershed (Bellucci et al. 2013) as urban runoff carries toxic contaminants, nutrients and sediments, pathogens and debris. Impervious surfaces lead to increased variation in stream flow, increased stream temperatures, and destabilization of the channel (Booth et al. 2002). Research has found these stressors negatively impact biological communities when percent impervious cover reaches 8-20%, and habitats become irreparably altered in the range of 25-60% (Miltner, 2004).

Accurately documenting these threats is important for rare and imperiled species where the number of occurrences determines conservation designation (Master 1991, CT DEEP 2010). Declining biodiversity and increasing rates of

extinction are the subjects of intensive investigation and the mitigation of these losses are among the fundamental goals of conservation biology (Burkhead 2012). Extinction rates are higher in aquatic ecosystems than surrounding terrestrial areas (Ricciardi and Rassmusen 1999) perhaps because they are susceptible to the multiple stressors that accumulate in watercourses. Since 1900 nearly 57 species and subspecies of North American freshwater fishes have become extinct (Burkhead, 2012). The conservation efforts for threatened or endangered species are challenging because the factors that relate to their decline often are unknown (Worthington et al. 2013) and communities are rarely structured by a single factor.

Species that are rare in terms of limited range, frequency of occurrence, or local abundance often have a greater likelihood of extinction than more common species. A commonly held belief is that the fewer occurrences a species has or the more fragmented its distribution is, the more vulnerable that species should be to localized extirpation and extinction (Fagan et al. 2002). Having clumped occurrences helps species to endure localized extirpation events, by increasing the chance that a site will be recolonized. Hanski (1991) demonstrated that the movement of individuals between patches contributes to survival and distribution of species. Fish populations that are more isolated will be more susceptible to local extirpation and the sites where they occur will be less likely to be recolonized.

Biodiversity loss is evident in minnow populations and has highlighted the need for increased monitoring to document species occurrence and detect population changes (Whittier et al. 1997). One of the more drastic minnow declines in the northeastern United States and southeastern Canada is the Bridle Shiner,

Notropis bifrenatus (Sabo 2000). The Bridle Shiner's microhabitat is associated with low velocity areas and stands of aquatic macrophytes, which are used for reproduction and predator avoidance (Jensen and Vokoun 2013). This species can be found in a variety of habitats including anthropogenic ponds, beaver impoundments, swamps, and low-gradient pool reaches of streams (Sabo 2000). Bridle Shiners carry various conservation listings including endangered, threatened or special concern throughout their range (Sabo 2000) and in Connecticut were listed as a species of concern in 2010. Little research has been conducted to offer explanations for the decline, but there is speculation that it may be due to habitat degradation, or changes in predator assemblages over time (Whittier et al. 1997; Sabo 2000).

A comparison of Bridle Shiner fish surveys done in the early 1960s and the 1990s showed a marked reduction in distribution for Connecticut (Jacobs and O'Donnell 2009). Of 29 lakes identified with Bridle Shiner in the 1960s only 11 lakes still had populations during the 1990s sampling period. Even more dramatic was the decline in stream populations. Of 56 stream samples in the early 1960s only 8 stream locations were found during the 1990s sampling surveys. This pattern matches a similar trend reported in other eastern US states (NatureServe 2007).

Subsequent to the data collected in the 1990s, questions were raised about how much of the documented decline could be an artifact of sampling methods. Recent work by Jensen and Vokoun (2013) showed, using patch occupancy modeling, that seine sampling can accurately determine the presence of Bridle

Shiner. Previously I compared electrofishing to seining and found electrofishing was half as efficient in detecting Bridle Shiner compared to the seine (Pregler et al. in review).

Based on this gear comparison research my objective in the current study was to re-visit all sites with historical sampling with a seine in the summer of 2013 in an attempt to uncover additional populations that may have been imperfectly detected by previous electrofishing surveys. I did not expect to find Bridle Shiner at all of the extirpated locations so my second objective was to identify landscape-scale correlates of extirpation using geospatial tools to understand the causes of range reduction. This approach provides much needed context on declines in this species and potential avenues for conservation actions.

METHODS

Study area

The study area included all known historical locations of Bridle Shiner in Connecticut (Figure 1), which included streams, ponds, and lakes. These locations were identified from surveys conducted by the Connecticut Department of Energy & Environmental Protection in the 1960s using a seine, and an additional 1990s survey conducted with electrofishing gear. 63 historical sites with extirpated populations were re-sampled in summer 2013 with a seine to determine if additional Bridle Shiner populations could be uncovered, since a study conducted by Pregler et al. (2014 *in review*) found seining superior to electrofishing in the detection of Bridle Shiner. This sampling was done to ensure missed occurrences in

previous sampling events would not bias my future analyses to understand what has contributed to the current Bridle Shiner distribution. While 82 percent of historical extirpated sites were re-surveyed, the remainder could not be visited due to constraints on access. In some locations I was unable to accurately identify where the historic survey took place, so these sites were omitted from the survey and my analyses. Due to logistical constraints and a short sampling season I only sampled these historic sites once, and relied on detection probabilities from the study conducted by Pregler et al. (in review) which demonstrated that Bridle Shiner can be detected using a seine about 80% of the time. Because occupied Bridle Shiner habitat is characterized by slow flowing areas with silty substrate and submerged vegetation (Jensen and Vokoun 2013), I sampled vegetated areas of the littoral zone in lakes, and slow flowing vegetated reaches of streams, including impoundments. I used a 3.175 mm mesh, 5 m-wide bag seine operated by two individuals, and processed captured fish at the end of each seine haul. All seine hauls were about 35 m in length, and the number of hauls at a given site ranged from 3-12 depending on the amount of submerged vegetated habitat available. The 2013 sampling season extended from late July through August since this is when Bridle Shiner are most susceptible to capture (Jensen and Vokoun 2013).

Analysis

To understand and identify correlates of Bridle Shiner decline in Connecticut I compared two competing hypotheses using an information-theoretic approach (Burnham and Anderson 2002). I first hypothesized that changes in land-use over time were responsible for the decline. I used a space-for-time substitution in which

I used present land cover, since quantified land-cover data was not available for Connecticut prior to 1985. My second hypothesis attributed the decline to habitat fragmentation caused by dams, a population's position in the watershed, and distance to nearest historical site (patch isolation).

These covariates were quantified using geospatial data layers in Geographic Information System (GIS) (ArcMap10.1), and all environmental data were downloaded from online sources (Table 1). I calculated percent land cover within a 250 m riparian buffer surrounding the upstream watershed of a site. Total impervious cover was used as a metric of development, and I divided the remaining land cover into forest and non-forest categories. Non-forest included cover such as agriculture, and barren land. Development was excluded from the non-forest category since it was already accounted for using total impervious cover. Correlation between percentage landscape coverages, such as impervious and forest cover, are common and can be problematic in regression modeling. Impervious cover and forest cover were not correlated in the data, and both covariates had a variance inflation factor of 1.014. I determined the total number of dams within a 2 km radius of a site. We coded watershed position as a categorical variable to determine if extirpations were more frequent in the headwaters or downstream regions and determined using Strahler stream order (Strahler 1957). Patch isolation was a categorical variable, and categories were determined by measuring the waterway distance from one site to the nearest site where there was a population record in the historical surveys. After all distances were measured they were ranked from low to high, and categories 1 through 3 were determined by

natural breaks in the data (Figure 2). A final category of 4 was given to sites that were completely isolated and had no nearest historic site within a watershed.

To best explain Bridle Shiner occurrence, I used the presence-absence data set, and constructed two model groups using the covariates from each hypothesis. I restricted my analysis to presence-absence data rather than abundance, since previous sampling used different types of sampling gear. I also had an additional categorical covariate separating sites into lotic and lentic waterbodies since there was a combination of streams, lakes, and ponds. I performed analyses using logistic regression and models were evaluated using Akaike's Information Criterion (AIC_c) adjusted for small sample size (Burnham and Anderson 2002). Models were considered to be competing if within two AICc units of one another. I calculated variance inflation factors to determine if multicollinearity was present (Zuur et al. 2010). In order to better visualize land cover trends I sorted the percent forest and impervious cover from low to high and binned the range of observed percentages within each cover type into five groups such that each bin contained approximately the same number of observations (~20-22). I then calculated the mean of the binned observations both for the percent of cover type (x-axis) and the percent of sites that were occupied by Bridle Shiner (y-axis) and created plots.

A post-hoc model set was fitted combining the most supported covariates from both hypotheses to explore if a more supported model might emerge. I also included an interaction term between waterbody type and watershed position in the post-hoc model set to capture potential differences in how stream and lake watershed position might explain Bridle Shiner occurrences.

RESULTS

A total of 99 sites were included in my analysis with 63 historical extirpated sites (41 stream sites and 13 lake sites) sampled in the 2013 survey. The remaining 36 sites were historical extant Bridle Shiner populations. Out of the 63 historical sites sampled in the 2013 survey I found that 9 of these extirpated sites had Bridle Shiner populations; 6 in streams and 3 in lakes.

The candidate model set supported the land cover hypothesis more than the habitat fragmentation hypothesis (Table 2). Models including land cover parameters carried 96.63% total weight versus models including habitat fragmentation parameters which only had 2.35% of total available weight. The most supported model in this candidate set contained a combination of forest, impervious cover, and waterbody, and carried 63.06% of the total model weight (parameter estimates, Table 3). Percent occupancy of sites increased with more forest cover and less impervious cover (Figure 3), and binned plots of percent forest (Figure 4) and impervious cover (Figure 5) help to better visualize this trend. There was not much model support for patch isolation or dam covariates. While the habitat fragmentation hypothesis had low support, the single most supported covariate within that model group was watershed position, and models including watershed position had 52% of the total weight in comparison to other models within the habitat fragmentation model group. Occupied locations were higher in the watershed (lower stream order values) compared to unoccupied sites, (Figure 6) and this difference was more pronounced in streams than in lakes (Figure 7).

Most lake sites were situated on low order streams while there was a larger range in order among the stream sites. The post-hoc model set showed an improved model with the addition of an interaction of watershed position and waterbody to the previously supported model (Table 4). The interaction between streams and watershed position has a more negative effect on Bridle Shiner occurrence compared to the interaction between watershed position and lakes (Table 5). I multi-model averaged parameter estimates from the two most supported models because the second model in the post hoc set carried ~20% model weight indicating substantial model selection uncertainty (Burnham and Anderson 2002) (Table 6).

DISCUSSION

Overall, Bridle Shiner populations have undergone a substantial decline in Connecticut. Although I uncovered additional Bridle Shiner populations using a seine, an overwhelming 60 percent range reduction was apparent over the past 50 years. Extant populations are more likely to persist in areas of high forest cover and low impervious cover, and there is also evidence that populations are more likely to have become extirpated from higher ordered streams.

In streams, Bridle Shiners were frequently found in impoundments with submerged aquatic vegetation and silt substrate. I also observed that associated aquatic vegetation was variable and included milfoils (*Myriophyllum*) and pondweeds (*Potamogeton,* sp.) (submerged weeds), smartweeds (*Polygonum* sp.) (emergent weeds), and even filamentous algae. Bridle Shiner co-occurred with a variety of fish species, including predators like Largemouth Bass (*Micropterus*

salmoides) and Pickerel (*Esox* sp.). Since 4 out of my 6 rediscovered stream populations were found in small impoundments, this led me to question whether or not the 1990s sampling events were targeting the correct habitats. It is possible these swampy, heavily vegetated areas would likely have been avoided by electrofishing surveys given their difficulty to sample. Historical 1960s seining data records had all species recorded under an umbrella location without specific habitat details pertaining to each species captured. There may have been some type of habitat selection bias by samplers in the 1990s surveys such that when faced with the decision of electrofishing in a swampy impoundment or backwater, versus a riffle-pool sequence the latter was chosen.

I found that the land cover and land use change related hypothesis was more supported than the habitat fragmentation hypothesis. Bridle Shiners in Connecticut tend to be found in riparian areas with high forest cover, and low impervious cover. When total model weights were calculated for model parameters, forest and impervious cover were equally important with model weights of 80.78% and 79.42% respectively. Landscape alteration related decline has been observed in other species such as Brook Trout (*Salvelinus fontinalis*) where forest clearing and agriculture led to the degradation of streams' physical and chemical habitat conditions, and in turn decreased fish distribution and abundance (Stranko et al. 2008). However, while Bridle Shiner is another example of a species that is negatively impacted by landscape alterations it is possible the mechanisms of decline are different compared to other classically intolerant species such as Brook Trout. Bridle Shiners are typically not viewed as an indicator species because they

live in swampy habitats and are exposed to and are able to tolerate varying dissolved oxygen levels, fluctuating temperatures, and sediment loads. They do, however, have specific habitat requirements such as submerged vegetation for reproduction and predator avoidance (Sabo 2000). Changes in land use, and the associated decline in forest and increase in impervious cover, could act negatively on submerged vegetation. Developed lake shorelines tend to have an increased frequency of winter drawdowns and herbicide treatments, which can result in limited availability of submerged vegetation habitat for fish species. Furthermore, recent studies demonstrate that human development of lake shores alters the physical habitat and nutrient cycles (Scheuerell and Schindler 2004). There is a decrease in the spatial aggregation of fishes with increased shoreline development by humans, reflecting a loss of refugia and resource heterogeneity that favors aggregation among fishes. In streams, vegetation availability is more stochastic, but as the floodplain of streams becomes more developed there is a more flashy flow regime and scour, which results in less vegetation. Additionally, because streams tend to have a stochastic environment they may be more susceptible to vegetation loss. These stream locations could potentially be sink populations rather than sources with the lakes and impoundments housing stronger Bridle Shiner populations due to lentic environments having more stable vegetated habitat. Streams in disturbed catchments tend to respond faster and more severely to storm events, have lower base flows during dry seasons, and be wider, shallower, more polluted and warmer than streams in undisturbed catchments (Booth et al. 2002). Runoff from the impervious surfaces in these watersheds continues to be a

major cause of degradation in the nation's waterways (US EPA 2001) and cities tend to be built on low-slope, formerly agricultural land, rather than high-slope formerly forested land (Wenger et al. 2008). Percent impervious cover is typically low in natural landscapes, intermediate in agricultural landscapes, and high in urban landscapes. Furthermore, additional research conducted in Connecticut has illustrated that there has been a significant amount of riparian development over the past 30 years (Wilson and Arnold 2011). As development and land use has transformed natural forest cover to agricultural and urban landscapes the resulting increased impervious cover has led to less absorption of precipitation into soils, and increased overland flow (Stanfield 2006).

This research potentially highlights areas that might be good for future conservation and re-introductions and thus better informs land use policy decisions in Connecticut. This also provides valuable insight to other states interested in re-surveying for Bridle Shiner and highlights the type of habitats and environments in which to look for them. However while this information is useful there needs to be a range wide investigation of the Bridle Shiner decline since Connecticut is a small section of the Bridle Shiner range, and my results should be investigated across the range since results may vary from state to state given their land cover compositions. It is also possible states with less funding for fish research may be more susceptible to the imperfect detection problem since there would be a lack of resources for exhaustive sampling efforts in habitats where Bridle Shiner persist.

While this finding does not demonstrate the direct mechanisms of Bridle Shiner decline, this study does demonstrate that landscape level correlates of decline are

apparent and adds to the evidence linking changes in land use and impervious cover to species decline. More site-specific research is needed to quantify the direct mechanisms to decline and it will be important to understand what effect site disturbances such as drawdowns and herbicide treatments can have on Bridle Shiner populations. Especially since Bridle Shiner only live 2-3 years (Sabo 2000), a one-year drawdown that disrupts habitat for reproduction could have a significant impact on the population and the loss of vegetative cover also might make it more susceptible to visual predators. Other speculation has hypothesized that invasive plant species might contribute to the decline (Massachusetts Division of Fisheries and Wildlife 2008). Through my research, however, I have observed Bridle Shiner in a variety of macrophyte compositions including invasive species such as *Myriophyllum heterophyllum*. Such observations suggest that future research on macrophyte composition as it relates especially to Bridle Shiner suitability for reproduction and other habitat influences on survival are needed. I recommend that Bridle Shiner remains a species of concern in Connecticut until more research is conducted to understand the direct mechanisms of decline.

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FIGURES AND TABLES

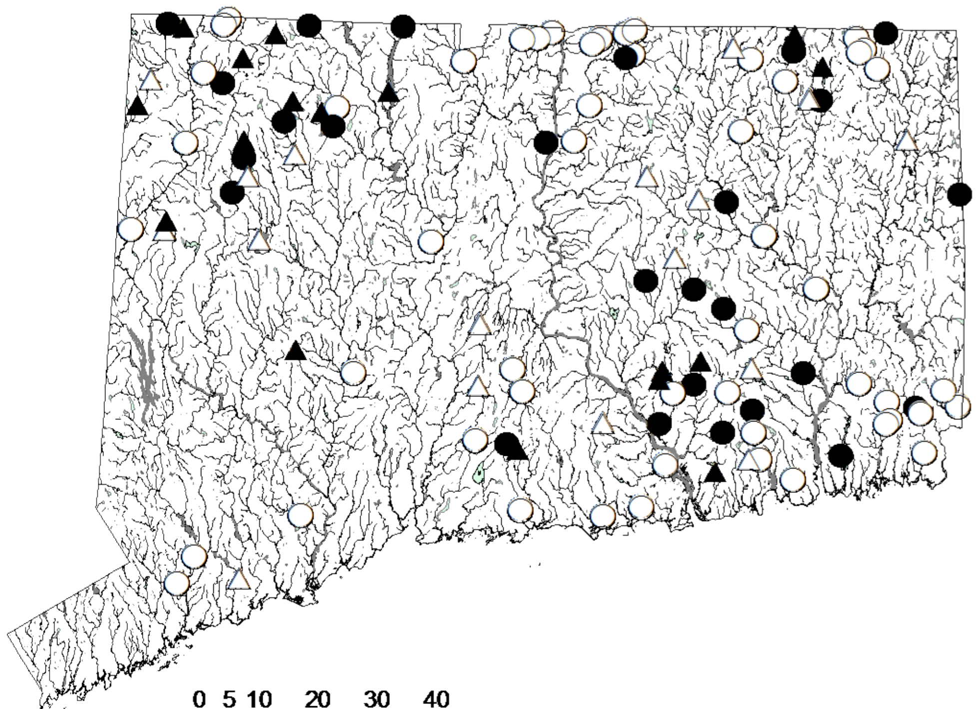


Figure 1. Bridle Shiner distribution in Connecticut, USA prior to 2013 survey season. Extant (filled) and extirpated (hollow) Bridle Shiner lake (triangles) and stream (circles) populations (adapted from Jacobs and O'Donnell 2009).

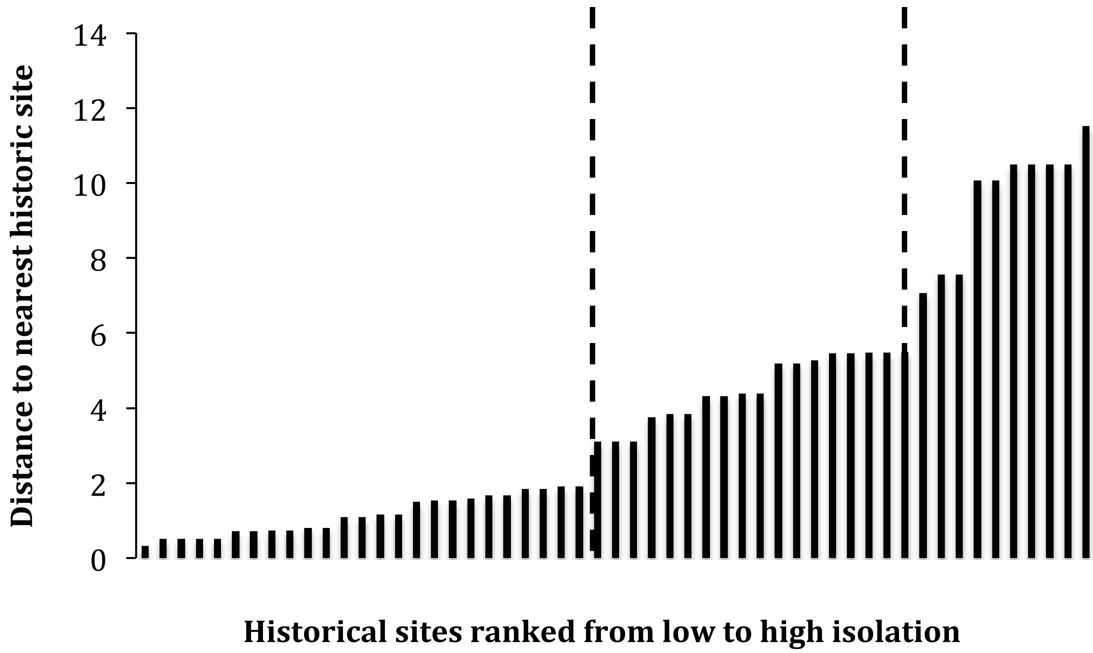
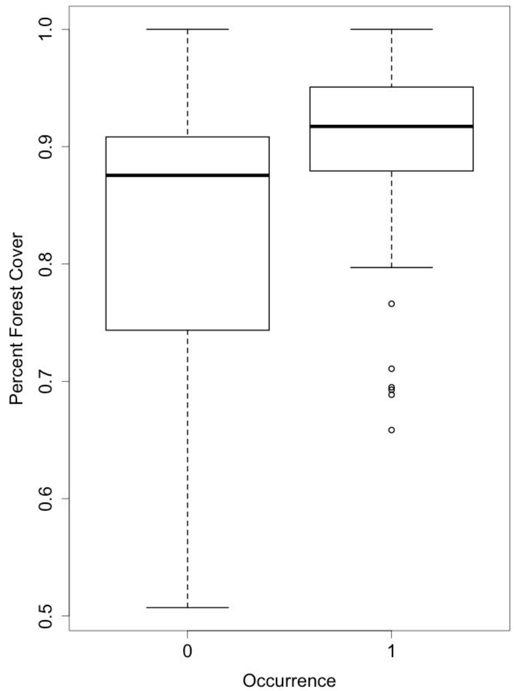
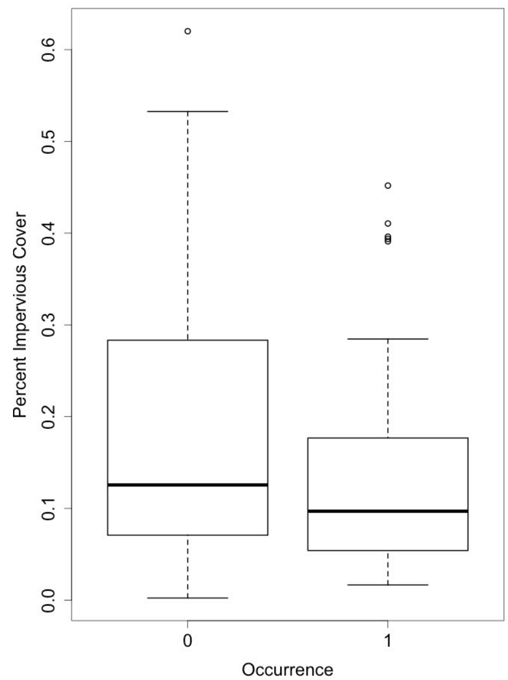


Figure 2. Distance to nearest Bridle Shiner historical site, ranked from low to high isolation. Sites to the left of the leftmost dashed line were placed in patch isolation category 1, those between the dashed lines in category 2, and those to the right of the rightmost dashed line in category 3.



a.



b.

Figure 3. The effects of percent forest (a) and impervious cover (b) on Bridle Shiner occurrence in Connecticut.

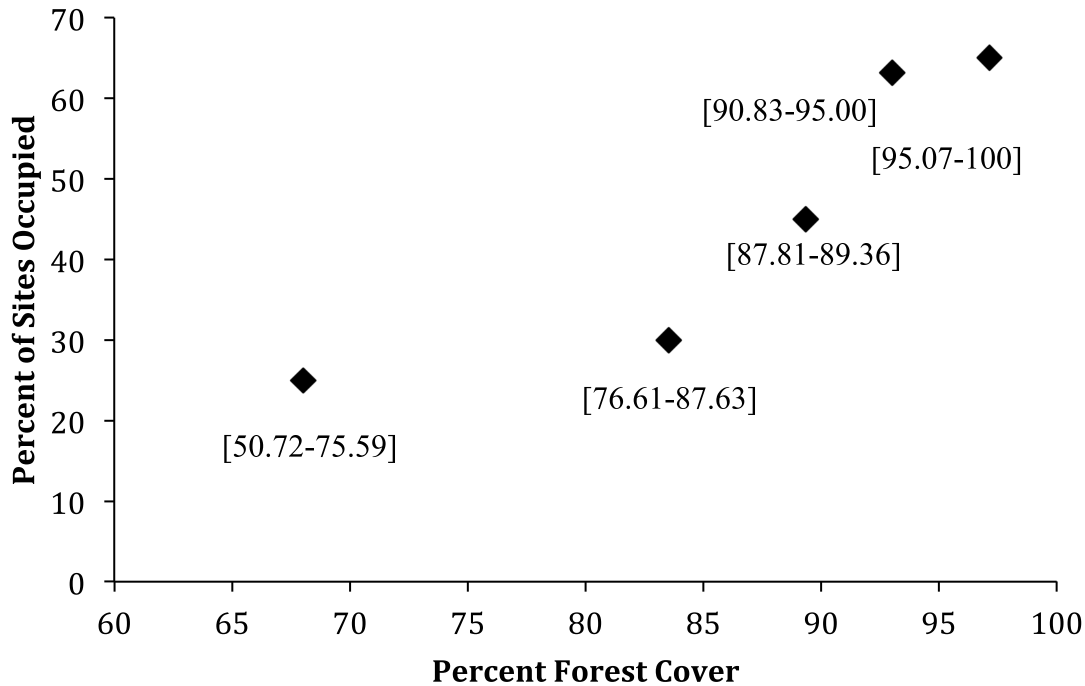


Figure 4. How percent of Bridle Shiner sites occupied changes with increasing percent forest cover. Each point represents a binned range of sites with their occurrences averaged. The numbers underneath each point represents the range of forest cover included in that bin.

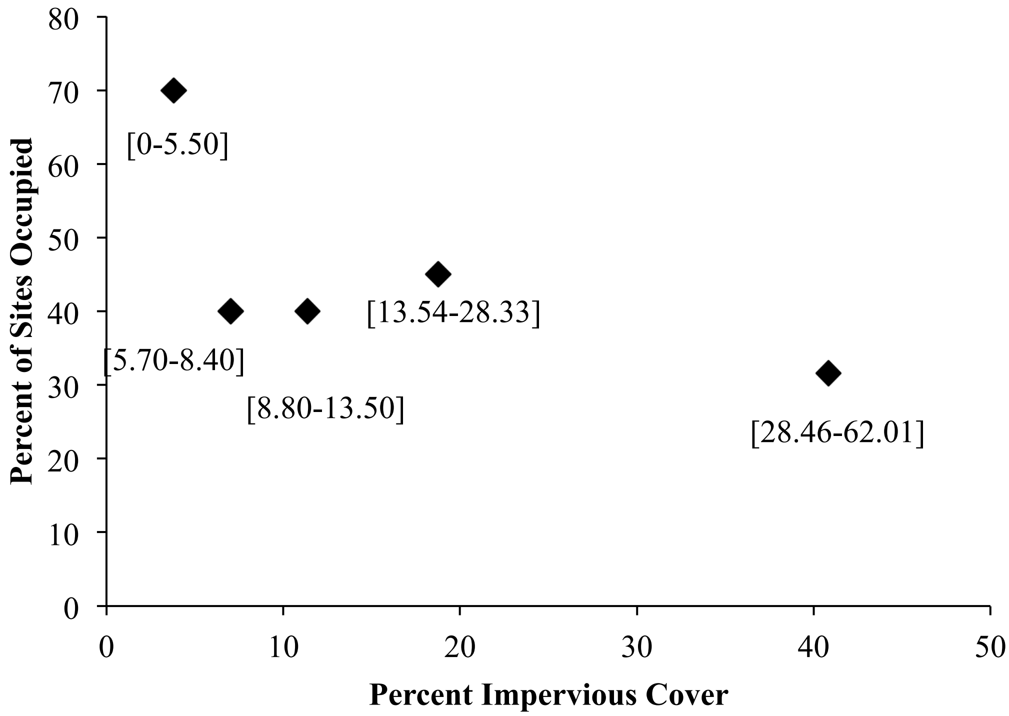


Figure 5. How percent of Bridle Shiner sites occupied changes with increasing percent impervious cover. Each point represents a binned range of sites with their occurrences averaged. The numbers underneath each point represents the range of impervious cover included in that bin.

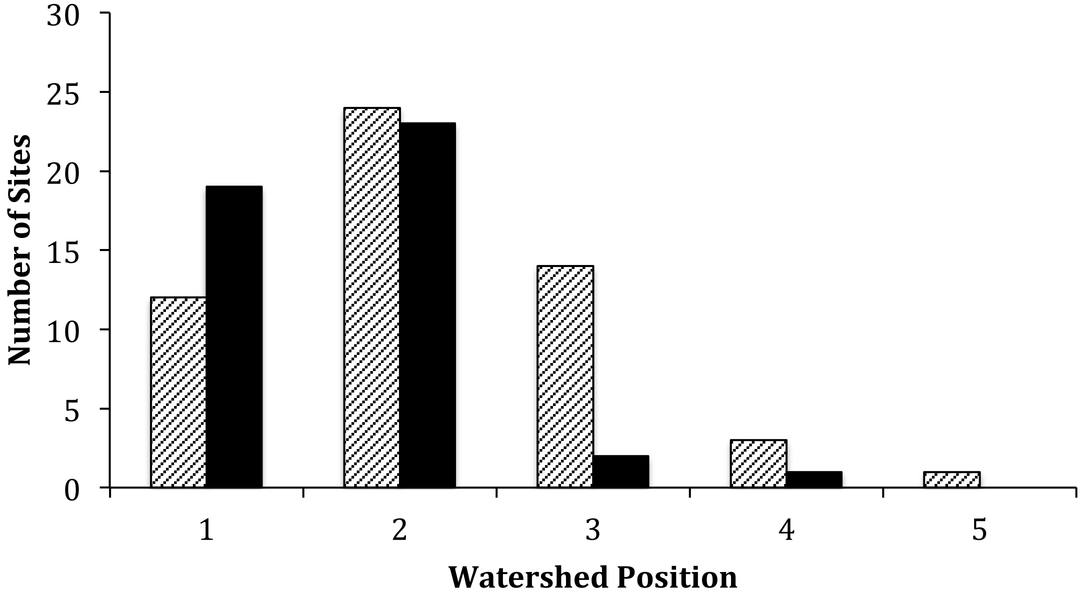
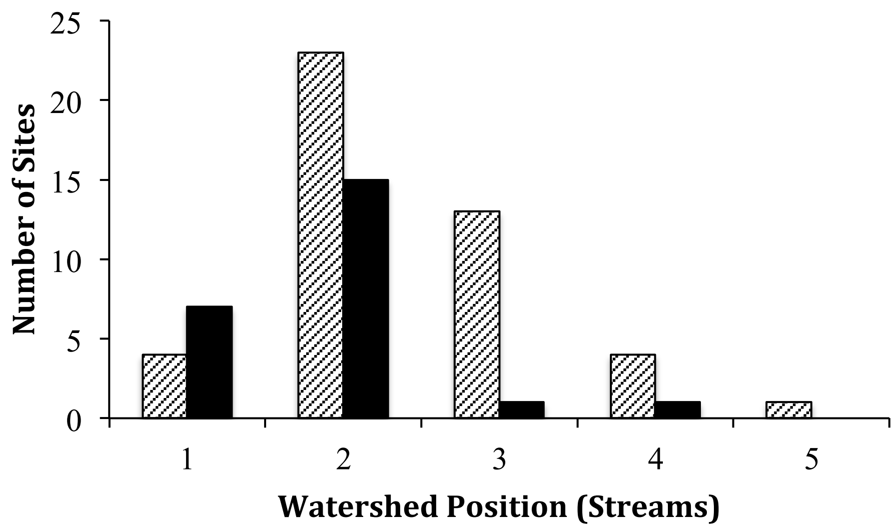


Figure 6. Number of sites in each watershed position category for extirpated (lined bars) and extant (black bars) Bridle Shiner populations.

a.



b.

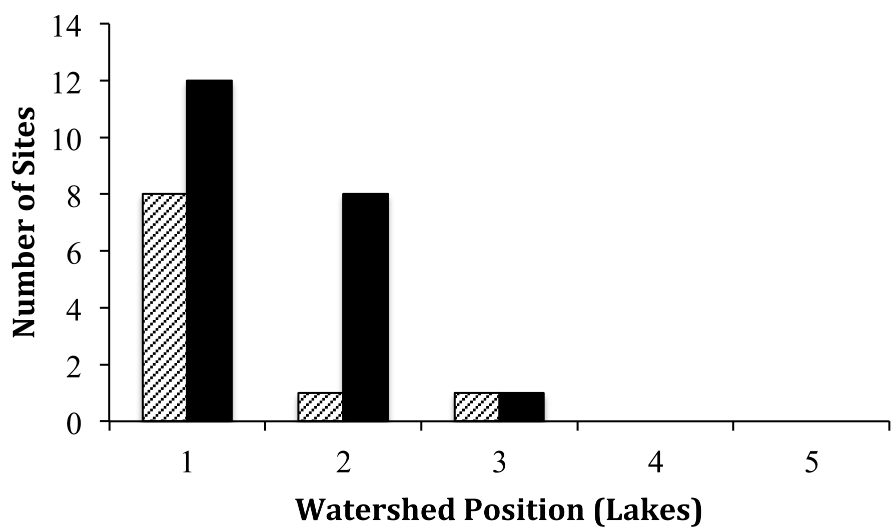


Figure 7. Comparison of watershed position in streams (a.) and lakes (b.) in extirpated (lined bars) and extant (black bars) Bridle Shiner populations.

Table 1. Source and description of landscape and habitat fragmentation covariates used in model construction.

Model Covariate	**Description**	**Source**
Land cover	National land cover database. Resolution 30 m.	Homer et al. (2001)
Impervious cover	National land cover database. Resolution 30 m.	Homer et al. (2001)
Hydrography	National Hydrography Dataset	McKay et al. (2014)
Stream order	Modified Strahler Stream Order	NHDplus (2014)
Number of dams	GIS point shapefile	CT DEEP (2013)

Table 2. First candidate set of models to compare the two hypotheses to determine if land cover types (1) or habitat fragmentation (2) models better describe Bridle Shiner decline. Model rankings are based on Akaike's Information Criterion adjusted for sample size (AIC_c), differences in AIC_c (ΔAIC_c), model weight (w_i), and number of parameters (K).

Hypothesis	**Model**	**AICc**	**ΔAICc**	**w_i**	**K**
1	Forest + Impervious + Waterbody type	123.2	0.0	0.6306	4
1	Forest + Waterbody type	125.9	2.8	0.1572	3
1	Impervious + Waterbody type	126.0	2.8	0.1561	3
1	Forest	131.1	7.9	0.0121	2
	Waterbody type	131.4	8.2	0.0102	2
2	Watershed position + Waterbody type	131.5	8.3	0.0099	6
1	Forest + Impervious	131.6	8.4	0.0093	3
2	Dams + Waterbody type	131.8	8.6	0.0085	3
2	Watershed position	133.7	10.6	0.0032	5
2	Patch isolation + Waterbody type	136.7	13.5	<0.001	5
2	Watershed position + Dams + Patch isolation + Waterbody type	136.9	13.8	<0.001	10
1	Impervious	137.1	14.0	<0.001	2
2	Dams + Patch isolation + Waterbody type	137.3	14.1	<0.001	6
2	Watershed position + Dams + Patch isolation	138.7	15.5	<0.001	9
2	Dams	140.3	17.1	<0.001	2
2	Dams + Patch isolation	144.6	21.5	<0.001	5

Table 3. Parameter estimates and variance inflation factors (vif) for the most supported model (Forest + Impervious + Waterbody type) from the first candidate set (Table 2) to describe Bridle Shiner decline in Connecticut.

Parameter	**Estimate**	**SE**	**P**	**vif**
Landcover type				
Forest	0.5745	0.2726	0.03505	1.007449
Impervious	-0.5671	0.2708	0.03626	1.238966
Waterbody type				
Stream	-1.6690	0.5484	0.00234	1.231187
Lake	N/A	N/A	N/A	N/A

N/A indicates no estimate is available since category is used as reference in coefficient determinations.

Table 4. Post-hoc model set combining covariates from both hypotheses. Model rankings are based on Akaike's Information Criterion adjusted for sample size (AIC_c), differences in AIC_c (ΔAIC_c), model weight (w_i), and number of parameters (K).

Model	**AICc**	**ΔAICc**	**w_i**	**K**
Watershed position:Waterbody type + Forest + Impervious	117.9	0	0.7423	10
Watershed position + Forest + Impervious + Waterbody type	120.6	2.7	0.1925	8
Forest + Impervious + Waterbody type	123.2	5.2	0.0546	4
Watershed position + Forest + Waterbody type	126.7	8.7	0.0094	7
Watershed position:Waterbody type	130.7	12.7	0.0013	8

Table 5. Parameter estimates and variance inflation factors (vif) from most supported model (Watershed position:Waterbody type + Forest + Impervious) in the post-hoc model set (Table 4) to describe Bridle Shiner decline in Connecticut.

Parameter	**Estimate**	**SE**	**P**	**vif**
Landcover type				
Forest	0.7264	0.3246	0.02526	1.178067
Impervious	-0.8777	0.3109	0.00475	1.429349
Watershed position				
1	N/A	N/A	N/A	N/A
2	1.744	1.2551	0.16467	6.202753
3	-2.163	1.6156	0.18062	3.302484
4	-1.8278	1.4855	0.21854	1.342307
5	-17.8538	1455.3978	0.99021	1.000000
Waterbody type				
Stream	0.4442	0.9567	0.64241	3.099166
Lake	N/A	N/A	N/A	N/A
Interactions				
2 : stream	-3.89	1.6034	0.01526	10.191545
3 : stream	-2.015	2.0259	0.31992	3.490453
4 : stream	-	-	-	-
5 : stream	-	-	-	-

N/A indicates no estimate is available since category is used as reference in coefficient determinations.

Table 6. Model averaged estimates from the top two models in table 4.

Model parameter	Averaged coefficient
Landcover type	
Forest	-0.8615
Impervious	0.6941
Watershed position	
1	N/A
2	1.2685
3	-2.2850
4	-1.5810
5	-17.5711
Waterbody type	
Stream	0.0669
Lake	N/A
Interactions	
2 : Stream	-3.0889
3 : Stream	-1.6000
4 : Stream	0
5 : Stream	0

N/A indicates no estimate is available since category is used as reference in coefficient determinations.