Using Changes in Naïve Occupancy to Detect Population Declines in Aquatic Species; Case Study: Stability of Greenhead Shiner in North Carolina

Todd Ewing, *North Carolina Wildlife Resources Commission, Division of Inland Fisheries, 1721 Mail Service Center, Raleigh, NC 27699-1721*

Michael Gangloff, *Department of Biology, Appalachian State University, Box 32027, Boone, NC 28608*

Abstract: Determining population trends for many aquatic species is problematic for most resource agencies because little or no historical information is available on population size nor are resources available for contemporary population estimates. Managers often only have available to them presence-absence data collected by qualitative surveys conducted at intermittent intervals. Changes in naïve occupancy can be used to detect population trends. Naïve occupancy is the ratio of number of sites where a species is detected to total number of sites surveyed, without correcting for imperfect detection. Herein, we present ways to conduct analyses for measuring changes in naïve occupancy using presence/absence data from multiple sources. Required elements include showing measures of uncertainty and statistical analysis (including power analysis). These data can effectively be used to determine population trends for many species in a cost effective and statistically rigorous manner. A case study using the naïve occupancy ratio shows the greenhead shiner (*Notropis chlorocephalus*) has remained stable from the 1960s to the 2000s.

Key words: population trend, power analysis, presence-absence, *Notropis chlorocephalus*

Journal of the Southeastern Association of Fish and Wildlife Agencies 3:1–5

Determining the conservation status of at-risk species is a major responsibility of fish and wildlife agencies. Population trends provide important information for determining population viability and conservation status (O'Grady et al. 2004). Population trend is an important component of most conservation status protocols including the ones utilized by the International Union for the Conservation of Nature (IUCN 2011) and NatureServe (Master et al. 2012), which are, in turn, often used by states when constructing priority species lists or assigning conservation status. However, determining population abundance trends for most species is often difficult. There is usually limited availability of historical information on population sizes for most species (Shaffer et al. 1998, Strayer and Smith 2003, Pollock 2006). Additionally, resources are often lacking to generate contemporary abundance estimates. For example, within the state of North Carolina, there are approximately 200 nongame fish species, 45 crayfish species, 50 mussel species, and 58 species of aquatic snails for which the NC Wildlife Resources Commission has conservation responsibilities. Generating population estimates for such a large number of species is impractical. This lack of information often leads to conservation status determinations that are based on anecdotal information, especially when expert opinion is used to assign status.

Presence-absence data available to managers are often generated in surveys done by naturalists, museum staff, academic researchers, or resource agencies, where the goal was not necessarily to determine abundance but to document the distribution or presence of a species or groups of species (Shaffer et al. 1998). These data are more correctly termed detection/non-detection data. However, herein we use the terminology presence/absence to maintain consistency with work done by others (i.e., Strayer 1999, Strayer and Fetterman 1999, Strayer and Smith 2003, Joseph et al. 2006, Pollock 2006). Presence-absence data can be used in conjunction with basic statistical approaches and is readily available or easily collected; they are a valuable tool to assess apparent changes in species abundance. Changes in naïve occupancy, the proportion of sites where the species is detected (MacKenzie 2005), can be used instead of directly comparing a species' past abundance with current abundance. Stricktly speaking, measuring a change in naïve occupancy tests the hypothesis that the proportion of sites where a species is detected has changed. It does not directly test for changes in abundance (Strayer and Smith 2003, MacKenzie 2005), but since occupancy and abundance are usually strongly correlated the technique can track population changes (Strayer 1999, Strayer and Smith, Joseph et al. 2006, Pollock 2006).

There are two basic ways a population can decline. It can become locally extirpated from entire sites or it may experience declines across its range but not necessarily become extirpated from a site (Strayer 1999, Pollock 2006). Modeling studies by several researchers have demonstrated that presence-absence data can be used to detect both types of population decline (Strayer 1999, Joseph et al. 2006, Pollock 2006). Because presence-absence data can detect situations where a species declines but does not become locally extirpated, it is important not to correct for imperfect detection since this would obscure local population declines. For most species, detection probabilities are < 1 (i.e., < 100%), thus a species will often not be detected when it is present. MacKenzie (2005) discusses ways to correct for imperfect detection probability using repeat surveys. This may be costly, causing a reduction in the the number of sites that can be surveyed. For most species, we recommend using naïve occupancy because this metric is easily calculated from the type of presence-absence data typically available to managers.

Presence-absence surveys are less sensitive than abundance-based surveys to natural fluctuations in abundances and are typically more cost effective because they require less time to conduct than most abundance-based surveys. Furthermore, they can be used to compare information from historical surveys, that may not have estimated species abundance, with more intensive, recent surveys (Shaffer et al. 1998, Joseph et al. 2006, Pollock 2006). However, presence-absence surveys are usually less powerful at detecting population decline than abundance-based surveys (Strayer 1999).

Shaffer et al. (1998) recognized two ways to characterize studies to document changes in occupancy. The first of these are "fixed effect" studies wherein historical sites are re-sampled. This approach is also described as monitoring or repeated-measure studies. In a second, "random effects" approach, a different but presumably equal set of sites are sampled at a later time period (Shaffer et al. 1998). A second distinction between survey types involves species selection. In "fixed" surveys, usually one species is the target of the investigation. In "random" surveys, multiple species may be targeted and the focus of the study is usually assemblage or community composition or species composition (Shaffer et al. 1998). The choice of fixed versus random locations or single versus multiple species surveys has major implications for statistical power and the ability of surveys to detect changes in occupancy.

There are numerous statistical biases that may arise from site selection when using presence-absence surveys to assess species declines. Re-surveys often concentrate on sites where a species was previously detected (Strayer and Smith 2003). The problem with drawing conclusions from such studies is that this design fails to consider that the species may now occur in some sites where it was previously not detected and abundance may therefore be stable or increasing (Strayer and Smith 2003). A solution is to resurvey all sites previously surveyed, whether the target species was present historically or not (Strayer and Smith 2003).

Sampling methodology is another important aspect of survey design (Strayer and Smith 2003). As noted earlier, low population size decreases the observability of a species at a given site reducing its chances of being observed during a survey (Joseph et al. 2006). However, other factors may also influence the change a species is observed such as time of year sampling occurs, flow conditions, turbidity, etc. Additionally, if the sampling technique used in more recent surveys is more thorough than what was used for historical surveys, such data may impair the ability to detect actual declines. The converse is also true; if more recent surveys are less through than past surveys, this may lead managers to believe that stable taxa are declining. Hence, researchers should ensure that consistent sampling techniques are being used during each survey.

When reporting occupancy, as with reporting any sample statistic, a measure of uncertainty should be reported as well. Confidence intervals (CIs) are a useful and easy measure of uncertainty. Because occupancy data have a binomial distribution (i.e., a species is either detected or not detected), CIs can be calculated using either the binomial distribution or the normal approximation (Krebs 1999, Zar 1999). The statistical tests used to analyze changes in spatial occupancy between two sampling periods depend on the sampling/re-sampling regime. With fixed sampling sites (i.e., repeated measures or monitoring design) McNemar's Test is appropriate because the sampling sites are not independent (Zar 1999, Strayer and Smith 2003, Pollock 2006). If sample sizes are small, a binary test is most appropriate (Zar 1999). When re-sampling randomly-selected sites (i.e., survey method), a contingency table analysis is appropriate using the χ^2, Fisher's Exact, or Z-tests (Zar 1999). If specifically testing for a decline (or increase) in spatial occupancy, a one-tailed test can be used. If testing for any change in spatial occupancy, then a two-tailed test is appropriate.

Quantifying changes in spatial occupancy is generally less powerful than count- or abundance-based surveys for detecting population trends (Strayer 1999, Pollock 2006). Therefore studies examining changes in population trend using occupancy must be rigorously designed such that they have sufficient power to avoid type II errors (i.e., failing to detect significant changes; Peterman 1990, Cobb et al. 1996). Prior to sampling, investigators should determine a minimum sample size to ensure that a certain level of power is attained. We recommend a power of at least 0.8 (probability of a type II error, $\beta = 0.2$). Given the low power of using changes in occupancy to detect a population decline, using a less conservative alpha $(\alpha) = 0.1$ should also be considered (Peterman 1990). Zar (1999) provides formulae for determining sample sizes for both random and fixed sampling designs to attain a given level of statistical power or effect size.

The North Carolina Wildlife Resources Commission's Division of Inland Fisheries is preparing to implement a protocol using change of abundance as a means to assess fish, mussel, crayfish and aquatic snail populations and determine conservation status.

Due to the constraints discussed above, changes in naïve occupancy will often need to substitute for change of abundance to track population trend. Herein, we describe the elements required for change of abundance surveys designed to detect population declines, and then we provide some examples using a stream fish, the greenhead shiner *Notropis chlorocephalus*, in North Carolina.

Methods

Power Analysis

In order to demonstrate the potential utility of occupancy to detect abundance changes in a re-sampling design, we generated sample-size curves for three different effect sizes: reductions of 30%, 50%, and 80%. These effect sizes were chosen because they are the population reduction thresholds used by the IUCN (2011) to categorize a species as vulnerable, endangered, or critically endangered respectively. For these sample-size curves, $\alpha = 0.1$ and $\beta = 0.2$ were used. These sample-size curves were based on McNemar's test for the fixed resampling design and the Z-test for the random re-sampling design. Power and sample size calculations for the Z-test were conducted using Statistix version 10 (Analytical Software, Tallahassee, Florida, 2013). Power and sample size calculations for the McNemar's test were conducted using a web-based application (www.statstodo.com/SSizMcNemar_Pgm.php).

Case Study

The greenhead shiner is a member of the minnow family (Cyprinidae) and endemic to the Catawba River basin of North and South Carolina. Extensive surveys (137 different sites sampled) were conducted in the Catawba River Basin by the NCWRC in 1963 (Louder 1964). First- through sixth-order streams were sampled using rotenone. Many of the identifications of nongame fish by Louder (1964) were incorrect, potentially limiting the usefulness of these surveys. However, Starnes and Hogue (2011) reexamined the voucher specimens, corrected the mis-identifications, and updated the taxonomy. One of the key findings of this reassessment was the determination that all vouchered specimens of the redlip shiner (*Notropis chiliticus*) from Louder (1964) were in fact greenhead shiners.

The North Carolina Department of Environment and Natural Resources (NCDENR) routinely samples stream fishes across North Carolina, using backpack electrofishing to sample second- through fourth-order streams. NCDENR sampled 42 sites in the 1990s and 63 sites in the 2000s in the Catawba River Basin (NCDENR 2013). Thirty of the sites sampled in the 1990s were re-sampled in the 2000s. We used corrected (Starnes and Hogue 2011) data from Louder (1964) and NCDENR (2013) to assess changes in greenhead shiner site occupancy from the 1960s to the 2000s and the 1990s to the 2000s. When comparing the 1960s to the 2000s, we used a random resampling design because very few of the sampling sites were the same. In order to ensure valid comparisons between the 1960s and the 2000s, only second- through fourth-order streams were analyzed from the corrected (Starnes and Hogue 2011) data, which allowed us to re-examine 91 of the 137 sites sampled in 1963. The null hypothesis of no change in occupancy of greenhead shiner from the 1960s to the 2000s was tested using Z-test (Zar 1999). To determine if there was a change in occupancy between the 1990s and 2000s, a fixed re-sampling design was used. The 30 sites sampled by NCDENR during both time periods were compared using McNemar's test. Because both of these comparisons were made using existing data, *a priori* power analysis was not conducted, but power was evaluated in a post hoc manner. McNemar's test and the Z-test were performed using Statistix. $P \leq 0.1$ was used to determine statistical significance for all comparisons.

Results

Power Considerations

Sample size requirements for a given power were much greater for a random-survey design than for a fixed-survey design (Figure 1). When initial occupancy is low (i.e., 0.1) more than 800 sample sites would be needed to detect a 30% decline using a random design whereas only about 200 sites would be needed to detect a 30% decline using a fixed design. Detecting smaller effect sizes requires much larger data sets and sample sizes. Power to detect declines increased with effect size.

The Case Study

Greenhead shiners were collected from 47 of 91 sites in 1963 for a naïve occupancy rate of 0.52 (CI, 0.43 to 0.63) and from 39 of 63 sites in the 2000s for a naïve occupancy rate of 0.62 (CI, 0.51 to 0.72). Spatial occupancy of greenhead shiners was similar between the 1960s and the 2000s ($Z = -1.10$, $P = 0.2734$; Figure 2). In both the 1990s and 2000s, greenhead shiners were found at 19 of 30 sites for a naïve occupancy rate of 0.63 (CI, 0.47 to 0.78). Greenhead shiners were absent in the 2000s from three sites where they were collected in the 1990s. However, they were found in three locations in the 2000s where they were previously absent in the 1990s. These results indicate that there was no significant change in spatial occupancy of greenhead shiners from the 1990s to the 2000s ($\chi^2 = 0$, $df = 1$, $P > 0.5$; Figure 2).

Discussion

Similar to the results of Strayer (1999), our data indicate that power to detect changes in naïve occupancy is relatively low, espe-

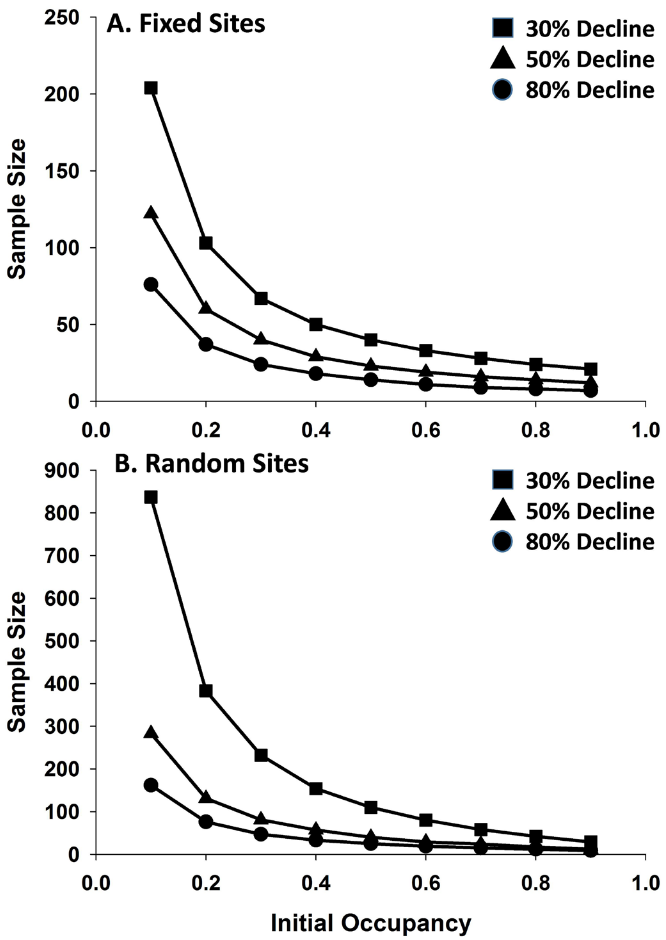

Figure 1. Sample sizes needed to detect changes in occupancy for a random resampling design (A) and for a fixed resampling design (B) using $\alpha = 0.1$ and $\beta = 0.2$.

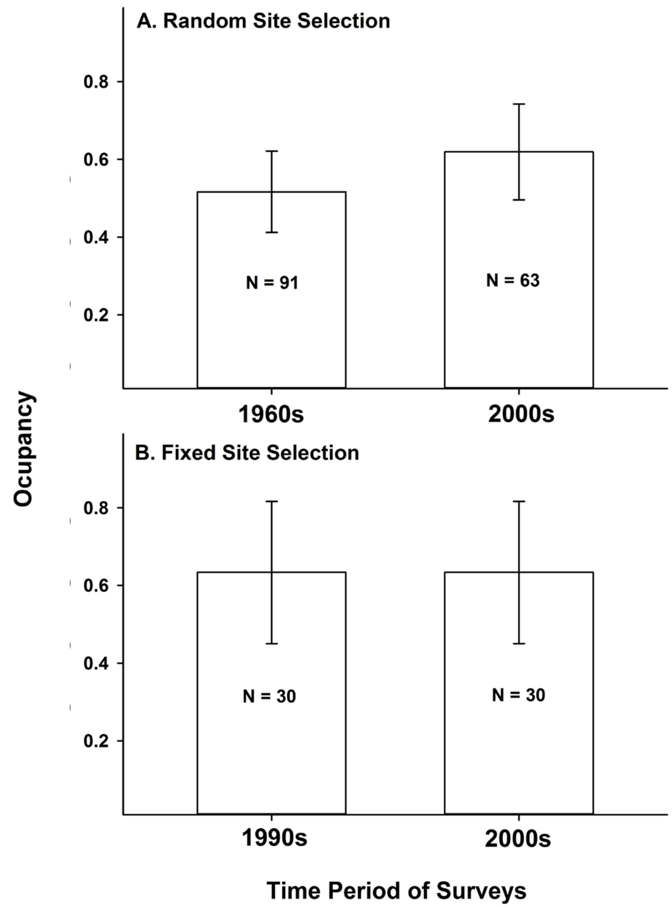

Figure 2. Comparison of changes of naïve occupancy between the 1960s versus the 2000s (A) and the 1990s versus the 2000s (B) for the greenhead shiner in the Catawba River Basin of North Carolina. Error bars represent 90% confidence intervals.

cially when trying to detect small-scale changes or changes when initial occupancy is low. However, use of a fixed sampling design instead of a random design can increase power and greatly reduce the required sample size. In our case study, the fixed design required roughly 75% fewer samples to detect a 30% decline in occupancy. Therefore fixed sampling designs should be employed by agencies and conservation researchers whenever possible.

The greenhead shiner appears to be stable in North Carolina. We observed no detectable difference in occupancy for this species between the 1960s and 2000s or between the 1990s and 2000s, but occupancy variance during the intervals between samples were not accounted for with our design. Also, none of the data sets used in these comparisons calculated abundance or density so it is not possible to quantify changes in site-scale abundance from these data. However, by measuring occupancy, we can be reasonably certain that abundance has not changed substantially. One weakness of this comparison is its relatively low power. We calculated that our power to detect a 30% population decline was only 0.67, below the target of 0.8. However, there was little evidence for a change in the occupancy of greenhead shiners in North Carolina between the two time periods, and increased power likely would not have led to a different conclusion. Likewise, there was no difference in naïve occupancy for greenhead shiner between the 1990s and 2000s. Despite the low number of sites available for 1990–2000s comparisons, we had substantially more power ($1 - \beta > 0.85$) than comparisons between 1963 and the 2000s. This illustrates the advantage of using a fixed sampling design when possible. Both long- and short-term spatial occupancy trends indicate that the greenhead shiner is stable throughout its range in North Carolina.

Assigning conservation status is often done using incomplete distributional or ecological knowledge. This is often justified under the 'precautionary principle' whereby a species is considered threatened until compelling evidence shows otherwise, which may be particularly useful for data-poor species (IUCN 2011).

However, the precautionary approach has been criticized as being unscientific because speculative conclusions may be drawn from non-standardized comparisons of spatio-temporal and population data (Webb 2008). Our methodology can provide resource managers with a simple, yet evidentiary-based, approach to determine the conservation status of species by describing population trends that might normally not be evident from traditional techniques or qualitative 'expert' opinions. These methods should result in more accurate status determinations and more appropriate allocation of scarce conservation resources to at-risk species.

Acknowledgments

This manuscript was greatly improved with helpful comments from David Cobb, John Jones, Gary Pandolfi, Dan Proud and Lynn Siefferman as well as other members of the Gangloff and Siefferman labs at Appalachian State University.

Literature Cited

Cobb, D. T., G. L. Sprandel, and D. E. Runde. 1996. Statistical power in analyses of population trend data. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies 50:294–302.

International Union for the Conservation of Nature (IUCN). 2011. Guidelines for using the IUCN Red List categories and criteria Version 9.0. Gland, Switzerland.

Joseph, L. N., S. A. Field, C. Wilcox, and H. P. Possingham. 2006. Presence-absence versus abundance data for monitoring threatened species. Conservation Biology 20:1679–1687.

Krebs, C. J. 1999. Ecological Methodology 2nd edition. Benjamin/Cummings, Menlo Park, California.

Louder, D. E. 1964. Survey and classification of the Catawba River and tributaries, North Carolina. Final Report, Federal Aid in Fish Restoration, Job I-H, Project F-14-R. North Carolina Wildlife Resources Commission, Raleigh.

MacKenzie, D. I. 2005. What are the issues with presence-absence data for wildlife managers? Journal of Wildlife Management 69:849–860.

Master, L. L., D. Faber-Langendoen, R. Bittman, G. A. Hammerson, B. Heidel, L. Ramsay, K. Snow, A. Teucher, and A. Tomaino. 2012. NatureServe Conservation Status Assessments: Factors for Evaluating Species and Ecosystem Risk. NatureServe, Arlington, Virginia.

North Carolina Department of Environment and Natural Resources (NCDENR). 2013. Standard Operating Procedures Biological Monitoring-Stream Fish Community Assessment Program Version 5. Raleigh, North Carolina.

O'Grady, J. J., D. H. Reed, B. W. Brook, and R. Frankham. 2004. What are the best correlates of predicted extinction risk? Biological Conservation 118:513–520.

Peterman, R. M. 1990. Statistical power analysis can improve fisheries research and management. Canadian Journal of Fisheries and Aquatic Sciences 47:2–15.

Pollock, J. E. 2006. Detecting population declines over large areas with presence-absence, time-to-encounter, and count survey methods. Conservation Biology 20:882–892.

Shaffer, H. B., R. N. Fisher, and C. Davidson. 1998. The role of natural history collections in documenting species declines. Trends in Ecology and Evolution 13:27–30.

Starnes, W. C. and G. M. Hogue. 2011. Curation and databasing of voucher collections from the North Carolina Wildlife Resources Commission 1960s Statewide Surveys of Fishes. Report to the North Carolina Wildlife Resources Commission. North Carolina State Museum of Natural Sciences. Raleigh.

Strayer, D. L. 1999. Statistical power of presence-absence data to detect population declines. Conservation Biology 13:1034–1038.

_____ and A. R. Fetterman. 1999. Changes in the distribution of freshwater mussels (Unionidae) in the upper Susquehanna River basin, 1955–1965 to 1996–1997. American Midland Naturalist 142:328–339.

_____ and D. R. Smith. 2003. A guide to sampling freshwater mussel populations. American Fisheries Society, Monograph 8, Bethesda, Maryland.

Webb, G. J. W. 2008. The dilemma of accuracy in IUCN Red List categories, as exemplified by the hawksbill turtles *Eretmochelys imbricata*. Endangered Species Research 6:161–172.

Zar, J. H. 1999. Biostatistical analysis, 4th edition, Prentice-Hall, Upper Saddle River, New Jersey.