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A Quantitative Comparison of Two Common Amphibian Sampling Techniques for Wetlands

Obtaining reliable survey data is critical for amphibian conservation and management. Many techniques are established for collecting quantitative survey data, including call surveys, pitfall arrays, drift fences, box sampling, seining, dipnetting, and aquatic funnel traps (Heyer et al. 1994; Skelly and Richardson 2010). Multiple techniques are often used in management and research to account for different species and life stages (Ryan et al. 2002). These techniques can be passive (pitfalls, aquatic traps, call recording) or active (seining, dipnetting, searches).

In lentic habitats, the most widely used sampling technique is dipnetting (Shaffer et al. 1994), an active sampling technique. Dipnetting is often standardized by unit effort such as time or number of dipnet sweeps. Another common technique for surveying aquatic amphibians is funnel trapping (Heyer et al. 1994),

which is passive. Many types of aquatic funnel traps, both commercially available and hand-made, have been used and evaluated, including those made from galvanized wire (Fronzuto and Verrell 2000), collapsible nylon mesh (Adams et al. 1997), and PVC pipe with acrylic plastic sheeting (Smith and Rettig 1996). Augmentations to funnel traps such as aquatic fences (Willson and Dorcas 2004) and net leads (Buech and Egeland 2002) have also been implemented. Both dipnetting and funnel trapping provide capture-per-unit-effort estimates of relative density (Skelly and Richardson 2010), but differ in terms of action (passive vs. active), cost, and time investment. Crosswhite et al. (1999) compared multiple active and passive sampling methods for terrestrial reptiles and amphibians and found that passive methods captured the greatest number of individuals while active methods were the most efficient in terms of time cost. Few studies have compared active and passive aquatic techniques, but Gunzburger (2007) found that species richness was larger when using passive methods in aquatic habitat.

We compared the efficacy of a standardized dipnetting protocol with aquatic funnel trapping for capturing amphibians in 19 ridge-top ponds in the Daniel Boone National Forest (DBNF), Kentucky. We evaluated each pond using both techniques and surveyed for one, three-day period per month, May–August 2010. Dipnet sweeps were taken every five meters while walking

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the edge of each pond and the mean number of sweeps per pond ranged from 2.8–20.8. A sweep consisted of guiding a d-frame net in a 180-degree arc from the shoreline while jabbing the net into the detrital substrate. Each pond had 1–2 aquatic minnow trap arrays installed that consisted of two traps (4 mm mesh size, 46 × 26 × 26 cm dimensions, 6 cm openings) (Promar, Gardena, California; US \$8.99 [TR-501]) placed on either side of an aquatic drift fence that extended perpendicularly to the pond's shoreline (Willson and Dorcas 2004). The number of traps per pond followed Adams et al. (1997): two traps (one trapping array) were placed for every estimated 25 m² of littoral zone. The two largest ponds (612 m², 440 m²) were more than twice the area of the third largest pond (207 m²), and two trap arrays were installed at these two ponds. The cumulative number of individuals of a species captured during a single, three-day sampling period was used as an index of abundance of that species for the month. All individuals were immediately released unmarked. We did not mark individuals because of our focus on comparing the number of captures, not the number of unique individuals. Six of the nineteen ponds sampled did not hold standing water during one or more of the sampling periods, so amphibian data from these ponds were based on fewer than three sampling events. We performed Wilcoxon signed-rank tests to compare abundance values between the two survey methods for each life stage detected of each species (Quinn et al. 2007).

The total number of captures from each method was large (minnow traps = 5435, dipnetting = 4281) and 13 species were detected (Table 1, Fig. 1). Adults of *Hyla chrysoscelis* (Cope's Gray Treefrog) and *Pseudacris crucifer* (Spring Peeper) were only detected by minnow traps, whereas larvae of *Lithobates palustris* (Pickerel Frog) and *Hemidactylium scutatum* (Four-toed Salamanders) were only detected by dipnetting. Twelve of the 13 species captured were detected at either larval or adult life stage by both methods and Four-toed Salamanders were only detected by dipnetting. Generally, dipnetting captured more caudate larvae than minnow trapping, a similar result to a previous comparison of dipnetting and funnel trapping in streams (Willson and Dorcas 2003).

Seven different technique comparisons were significantly different (Wilcoxon Sign-Rank Test, $\alpha = 0.05$; Table 1). Minnow trapping, a passive technique, was significantly more effective at capturing adult *L. clamitans* (Green Frogs) and adult *Notophthalmus viridescens* (Red-spotted Newts). Dipnetting, an active technique, was significantly more effective at capturing larval *H. scutatum*, larval *Ambystoma maculatum* (Spotted Salamanders), larval *A. jeffersonianum* (Jefferson Salamander), larval *P. crucifer*, and larval *N. viridescens*.

Some capture biases may be a product of a particular species' behaviors or morphology. Generally, those species caught in larger numbers using minnow traps were bigger and more mobile than those caught in larger numbers using dipnetting. Alternatively dipnetting may also be the preferred method for species like *H. scutatum*, which are difficult to catch in traps due to their small size, low breeding output, and relatively low larval activity pattern (Harris 2005).

Notophthalmus viridescens were often found in traps breeding in large numbers, most likely due to chemical cues released by females that attract males for reproduction (Dawley 1984), suggesting that species using chemical cues to identify potential mates may be efficiently caught using passive traps. However, if only one sex responds to the attractant, there may be a bias in the sex ratio captured. Trapping was inappropriate for species such as *Anaxyrus americanus* (American Toads), *P. crucifer*, and

H. scutatum, which were often small enough to move through the mesh of the minnow traps, but large enough to be captured by the mesh in the dipnets. This study also suggests the difficulty in capturing adult, large frogs using dipnetting, likely due to the combination of their tendency to quickly jump away when startled and the speed with which they swim.

Because abundances for some species and life stages were not different between techniques, our study indicates that for such species (Table 1, Fig. 1), decisions about which techniques to use can be based on amount of time and equipment available to the researcher. Dipnetting is more efficient in terms of equipment and time and would often be the preferred method. However, as we have shown for a number of species, a species' attributes and life stage can bias its probability of capture by a given survey method. Hence, the behavior, size, and other aspects of life history should be considered before choosing a sampling technique.

Overall, our results reinforce the need to make species-specific decisions during the evaluation of surveying protocols. Conclusions based on aquatic trapping data alone would underestimate the abundance of ambystomatid salamanders and

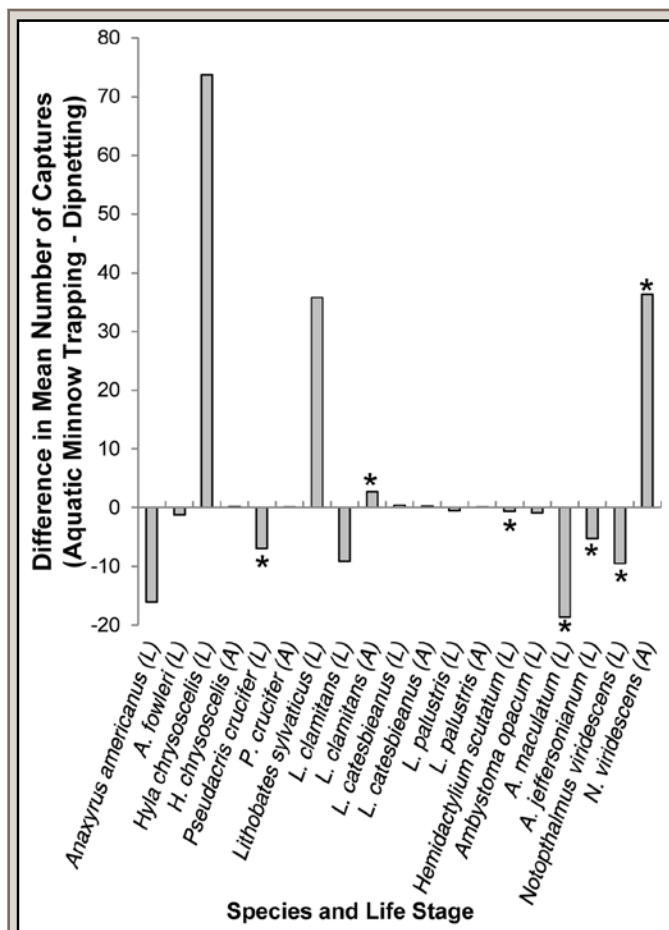


FIG. 1. Difference in mean number of amphibian captures (aquatic minnow trapping - dipnetting) from 19 ponds in the Daniel Boone National Forest, Kentucky, May–August 2010. Positive bars indicate more individuals captured using minnow trapping and negative bars indicate more individuals captured using dipnetting. Asterisks represent significant comparisons of mean number of captures between aquatic minnow trapping and dipnetting techniques using Wilcoxon sign-rank tests (Table 1). Life stage: A = adults, L = larvae.

TABLE 1. Mean abundance (mean no.) and standard error of 13 amphibian species by life stage among 19 ridge-top ponds in the Daniel Boone National Forest, Kentucky, May–August 2010.

Species	Aquatic Minnow Traps			Dipnetting			Wilcoxon ^b	
	n ^a	mean no.	SE	n ^a	mean no.	SE	Z	P
<i>Anaxyrus americanus</i> - Larvae	2	1.58	1.33	2	17.63	14.35	-1.34	0.180
<i>An. fowleri</i> - Larvae	2	0.32	0.23	3	1.47	1.21	-0.45	0.655
<i>Hyla chrysoscelis</i> - Larvae	8	84.42	77.77	7	10.68	4.51	-0.53	0.594
<i>H. chrysoscelis</i> - Adults	2	0.11	0.07	0	0.00	0.00	-1.41	0.157
<i>Pseudacris crucifer</i> - Larvae	2	0.79	0.60	6	7.68	3.69	-2.20	0.028
<i>P. crucifer</i> - Adults	1	0.05	0.05	0	0.00	0.00	-1.00	0.317
<i>Lithobates sylvaticus</i> - Larvae	4	135.26	76.74	4	99.47	59.57	-1.46	0.144
<i>L. clamitans</i> - Larvae	11	4.47	1.19	11	13.53	6.48	-1.51	0.130
<i>L. clamitans</i> - Adults	12	2.95	1.18	4	0.21	0.10	-2.85	0.004
<i>L. catesbeianus</i> - Larvae	8	5.16	2.02	9	4.74	1.68	-0.26	0.798
<i>L. catesbeianus</i> - Adults	3	0.37	0.23	2	0.11	0.07	-1.29	0.197
<i>L. palustris</i> - Larvae	0	0.00	0.00	2	0.47	0.38	-1.34	0.180
<i>L. palustris</i> - Adults	3	0.16	0.09	1	0.11	0.11	-0.38	0.705
<i>Hemidactylium scutatum</i> - Larvae	0	0.00	0.00	6	0.58	0.28	-2.26	0.024
<i>Ambystoma opacum</i> - Larvae	5	2.05	1.14	3	2.89	2.42	-0.27	0.768
<i>Am. maculatum</i> - Larvae	15	13.68	3.51	16	32.26	8.73	-3.07	0.002
<i>Am. jeffersonianum</i> - Larvae	14	3.00	0.83	13	8.21	2.55	-2.73	0.006
<i>Notophthalmus viridescens</i> - Larvae	8	3.84	1.68	12	13.31	7.49	-2.94	0.003
<i>N. viridescens</i> - Adults	17	48.32	13.63	13	11.95	2.82	-3.62	<0.001

^a Sample sizes (n) represent the number of wetlands of the 19 total where the indicated species and life stage were detected.

^b Wilcoxon paired sample test Z values and probabilities are from individual comparisons of each species and life stage by capture method.

overestimate the abundance of *N. viridescens*, and conclusions based on dipnetting alone would underestimate the abundance of *L. clamitans* adults. Hence, when evaluating amphibian populations, it is appropriate to include measures of detection probability to strengthen conclusions from count data (Dodd and Dorazio 2004). Additionally, this study supports the use of active and passive survey methods together when developing long-term monitoring of entire amphibian communities as well as providing evidence supporting active or passive survey methods chosen for a specific species or life stage.

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Cephalopod Ingestion by Juvenile Green Sea Turtles (*Chelonia mydas*): Predatory or Scavenging Behavior?

Immediately after emerging from eggs on sandy beaches, most sea turtle hatchlings disperse into the sea to enter a pelagic life-phase that may last several years (Meylan and Meylan 1999). During this stage, individuals are believed to associate with convergent oceanic fronts which accumulate floating structures (e.g., debris or algal mats such as *Sargassum* or *Macrocystis*; Nichols et al. 2001) that concentrate small pelagic animals (Carr 1987). Recent studies on the diet of post-hatchling Green Sea Turtles (*Chelonia mydas*) in the Pacific Ocean found no evidence of the association of this species with algal mats, but confirmed the importance of pelagic organisms in the diet of these animals (Boyle and Limpus 2008; Parker et al. 2011).

Asides from young turtles, pelagic ecosystems are comprised of many other organisms, including roaming predators like tuna, billfish, sharks, and dolphins (Dambacher et al. 2010). Oceanic cephalopods (e.g., squids) are also important components of pelagic food chains and serve as food for most of these predators (Clarke 1996; Croxall and Prince 1996; Klages 1996; Smale 1996) as well as for opportunistic scavengers (Croxall and Prince 1994). Because Green Sea Turtles seem to act as opportunists during their open ocean stage of life (Boyle and Limpus 2008), cephalopods might constitute as a complementary food source to their normal diets of cnidarians, gastropods, and crustaceans (Boyle and Limpus 2008; Parker et al. 2011).

Pelagic cephalopods have already been reported in the Green Sea Turtle's diet (e.g., Parker et al. 2011; Seminoff et al. 2002). For example, Parker et al. (2011) considered the presence of fisheries-caught squids in the diet of oceanic Green Sea Turtles as evidence of opportunistic feeding by the turtles on fishing-gear catches. However, implications of these observations

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and other possible explanations of how turtles may eat pelagic cephalopods have remained poorly discussed topics in the literature.

The southern region of Brazil (Fig. 1) suffers direct influence of the Subtropical Convergence, an encounter of the cold-water, nutrient-rich Falklands Current with the warm-water, oligotrophic Brazil Current (Castro and Miranda 1998). Hence, the region is the southern limit of occurrence of many tropical marine species, including fishes (Carvalho-Filho 1999) and mangrove trees (Sobrinho et al. 1969). Its rocky reefs, mangroves, estuaries, bays, lagoons, and oceanic waters are also important feeding grounds for marine turtles, especially the Green Sea Turtle, *Chelonia mydas* (Almeida et al. 2011; Bugoni et al. 2003; Guebert-Bartholo et al. 2011). The only genetic assessment of a coastal green turtle juvenile population from southern Brazil indicated a mixed stock population, composed mainly from the rookeries of Ascension and Aves islands (Proietti et al. 2009).

Here we report the occurrence of cephalopod beaks in the gastrointestinal tracts of stranded juvenile *Chelonia mydas* in South Brazil and discuss possibilities regarding when and how the turtles ingested the cephalopods. To achieve this objective we consider how life-history traits could have influenced the

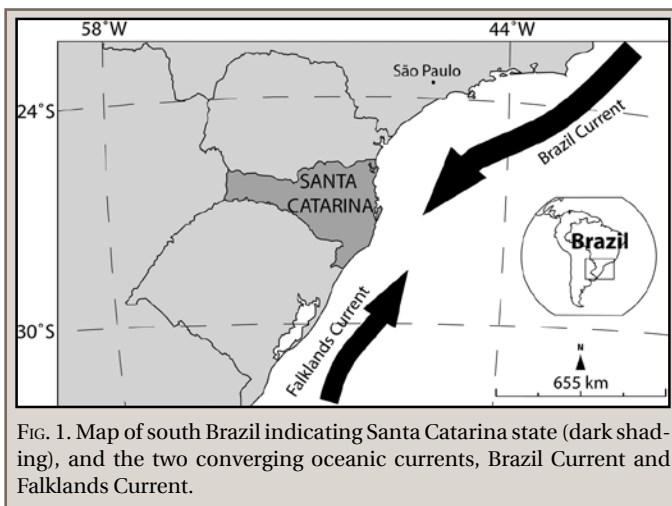


FIG. 1. Map of south Brazil indicating Santa Catarina state (dark shading), and the two converging oceanic currents, Brazil Current and Falklands Current.

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