Herpetofaunal Diversity of Alligator River National Wildlife Refuge, North Carolina

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Abstract - In the past century, habitat alteration and fragmentation have increased dramatically, which increases the need for improving our understanding of how species and biological communities react to these modifications. A national strategy on biological diversity has focused attention on how these habitat modifications affect species, especially herpetofauna (i.e., changes in species richness, community evenness and similarity, and dominant/rare species). As part of this strategy, we surveyed Alligator River National Wildlife Refuge, a coastal, mixed second-growth forested swamp (MFS) and pocosin wetland (PW), in North Carolina for amphibians and reptiles from September 2000 to August 2001. We randomly selected three sites (3 x 3 km) in two major habitat types (MFS, PW) and completed random surveys and trapping using transects, quadrats, nighttime aural road surveys, drift fences, canal transects, coverboards, incidental captures, and evening road surveys. We also collected herpetofauna opportunistically throughout the refuge to establish an updated species list. For analysis, we used Shannon-Weiner species diversity (H'), evenness (J'), species richness and species detectability (COMDYN4), and community percent similarity index to determine herpetofaunal community differences. We estimated 39 species in MFS and 32 species in PW ($P < 0.10$). Species detectability was similar between habitats (0.84 to 0.86). More reptilian species (+ 31%) inhabited MFS than PW, but estimated amphibian species richness was identical (17 spp.). H' was higher ($P < 0.0001$) for PW (2.6680) than for MFS (2.1535) because of lower J' in the latter (0.6214 vs. 0.8010). Dominance of three *Rana* species caused lower J' and H' in MFS. Similarity between the communities was 56.6%; we estimated 22–24 species in common for each habitat (95% CI = 18 to 31 spp.). We verified 49 of the 52 herpetofaunal species on the refuge that were known to exist in the area. Restoration of natural water flows may affect herpetofaunal diversity, which may be monitored during a restoration project. Currently, the refuge retains historical levels of herpetofaunal diversity for the region.

Introduction

During the last decade of the 20th century, concerned scientists, resource managers, and policy makers mobilized to contend with losses of biological diversity (Mooney and Gabriel 1995). Previously, Wilson (1988) provided information on the state of global biological diversity and urged everyone to treat the global resource of biodiversity more seriously. He recommended that this resource be preserved, indexed, and used. Human population expansion and land stewardship should not be on opposite

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sides, because biological diversity contains many undiscovered uses for humans. Furthermore, we know that biological diversity is a sensitive indicator of environmental change (Lovejoy 1995). Loss of species and their information for human use is irreversible. While most biological diversity is centered in the tropics or "hotspots" for conservation (Myers 1988, Myers et al. 2000), the southeastern United States also contains important areas of high diversity, especially for herpetofauna. Management and conservation of "coldspots" and hotspots of biodiversity in the Southeast may be just as important as conservation of hotspots in the tropics (Kareiva and Mavier 2003).

As a region, the southeastern United States supports one of the richest diversities of reptiles and amphibians world-wide (Gibbons and Stangel 1999). This region supports the world's highest biological diversity in some taxa, e.g., salamanders (Petranka 1998). Herpetofauna are distributed often by habitat type and abiotic conditions (Fauth et al. 1989, Means and Simberloff 1987, Stevens 1992). Diversity of habitats, among other factors, is one reason the southeastern states support large numbers of herpetofaunal species.

Managed areas, especially public lands, where human influence may directly affect species composition, should be a priority in conservation of herpetofaunal diversity. If herpetofaunal knowledge of a region is incomplete, efforts to protect individual species may have unforeseen impacts on other species or regional biodiversity. Additional threats to reptile and amphibian populations in the region include invasive species, environmental pollution, disease, unsustainable use, and global climate change (Alford and Richards 1999, Gibbons et al. 2000).

Given the state of herpetofaunal threats, especially with a rapidly growing human population in the southeastern United States, there is a need to determine regional herpetofaunal diversity and manage for these species, especially on public lands. Therefore, we surveyed Alligator River National Wildlife Refuge (ARNWR) for one year to establish baseline data of herpetofauna species and to provide ideas for managing and conserving herpetofaunal diversity in ARNWR in the future. Our objectives were to determine the absence or presence of herpetofauna in the two major habitat types of ARNWR, to determine the community composition (species diversity, evenness, species richness, community similarity) for each major habitat, and to provide a herpetofaunal species list for the refuge.

Methods

Study area

We conducted herpetofaunal surveys at ARNWR, which is located on the northeastern coast of North Carolina (35.75°N, 75.76°W) in Dare County and a small portion of Hyde County (Fig. 1). ARNWR covers 616 km^2 and occupies most of a peninsula (surrounded by brackish water to saltwater) connected to the mainland by a 5- x 13-km strip of land adjoining the southwestern edge of the refuge (US Fish and Wildlife Service 2004). ARNWR was established to conserve and protect pocosin

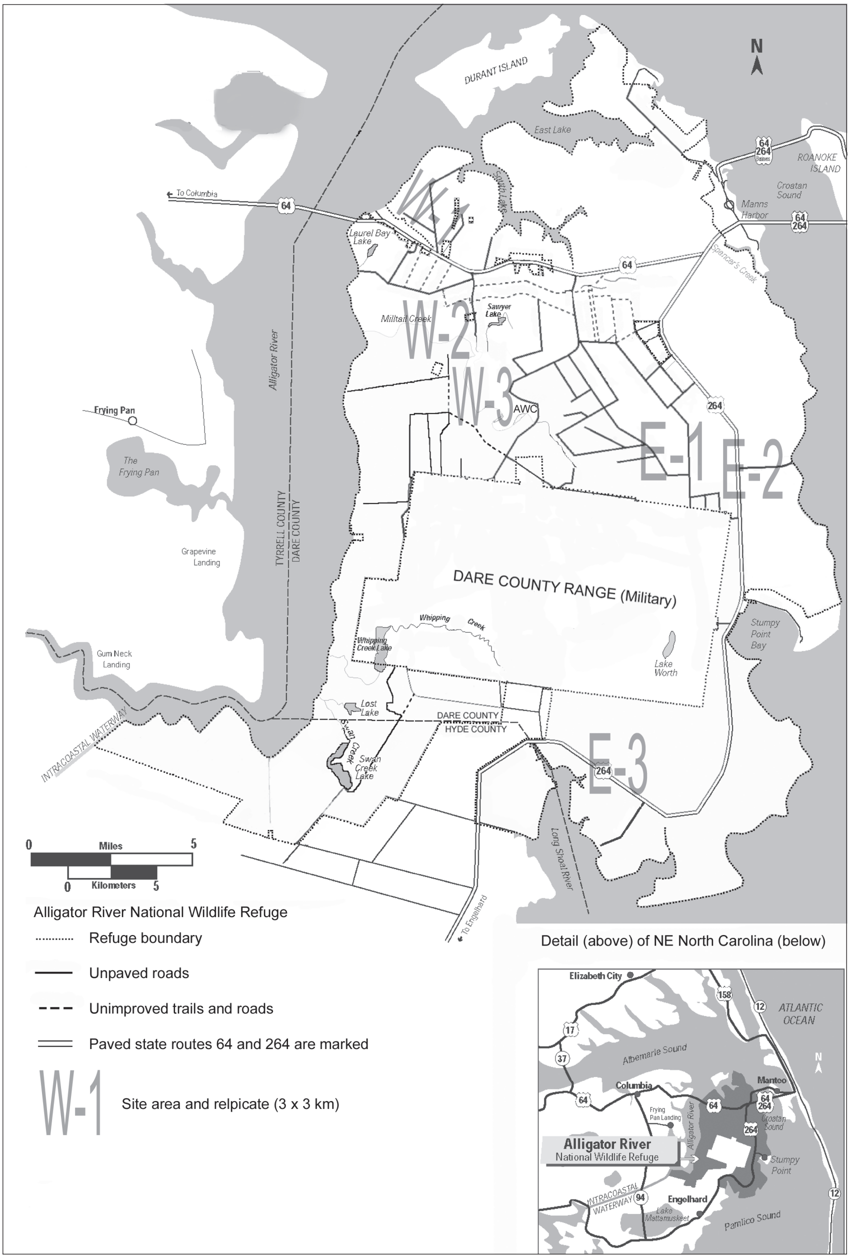

Figure 1. Herpetofaunal study sites of Alligator River National Wildlife Refuge, Dare and Hyde Counties, NC, 2000–2001. W-1 to W-3 mark sites in mixed forest swamp habitat (MFS), E-1 to E-3 mark sites in coastal pocosin wetland habitat (PW), and AWC marks the site for Atlantic white cedar habitat (≈ 1 km x ≈ 1 km, map center).

wetland habitat and its associated wildlife species. Most of the peninsula (ca. 70%) is a wetland in transition from partially ditched or drained areas with altered native vegetation to older forest swamps and pocosins. The remainder of the area is 20% natural wetlands and 10% developed (Sharitz and Gibbons 1982).

Major habitat types on the refuge include second growth, pine-hardwood forest swamps (MFS, western side), pocosin wetlands (PW, eastern side), and to a lesser extent, lowland mixed pine and *Chamaecyparis thyoides* (L.) (Atlantic white cedar). Swamp forests are dominated by *Nyssa sylvatica* var. *biflora* (Walter) Sargent (black gum) and *Acer rubrum* L. (red maple). Planted *Pinus taeda* L. (loblolly pine) and *P. elliottii* Engelm. (slash pine) stands occupy higher drained areas. Pocosin vegetation is variously described as boggy shrublands or flatwoods usually dominated by broad-leaved evergreen shrubs or low trees (Sharitz and Gibbons 1982). Pocosins contain low nutrient, peaty soils that are poorly drained. The low elevation on the refuge, < 1 to 5 m above sea level, allows saltwater invasion (flooded and killed areas), especially in areas peripheral to saltwater habitats and during storm surge events, which creates patchy habitats.

Most refuge roads parallel canals (unpaved ca. 188 km, paved ca. 77 km), which were created by road building during draining of surrounding land for planned or former agricultural use (Braswell and Wiley 1982). Canals vary in size; most are 3–8 m wide, 2–5 m deep, and contain acidic blackwater (tannins). Canals located near the shoreline often connect directly to the brackish-saltwater systems of Alligator River Sound and Croatan Sound. Although no major streams exist on the eastern side of the refuge, Milltail Creek flows west from the center of the refuge to Croatan Sound (Fig. 1). Whipping Creek flows from the south-central area of Dare County Range to the southwestern side of the refuge and Alligator River Sound. Nearby, Swan Creek flows from Swan Lake in the southwestern corner northeast to Alligator River Sound.

Sampling design

We divided the refuge into two areas, a priori, based on major habitat type (MFS, PW) and elevation. MFS varied from 1 to 5 m above sea level. Because of slightly higher elevation, MFS sites receive less saltwater invasion and freshwater inundation than PW sites. PW varied from 0 to 3 m above sea level. PW had more standing water and shorter shrub and tree canopy (2–5 m high) than MFS. In each area, we randomly established three 3- x 3-km sites where we concentrated our sampling efforts (Fig. 1). We selected large sites (900 ha) to prevent past land uses (e.g., 100–200-ha clearcuts) from significantly affecting one site. We located one additional site (≈ 1 x ≈ 1 km) in the largest stand of Atlantic white cedar (AWC; Fig. 1).

Sampling methods

We sampled herpetofauna in September 2000 and again more intensively from March to August 2001. We used a variety of methods to collect and survey, including time- and distance-constrained visual encounter surveys along canals, time-constrained transects and quadrats, drift fences, nighttime frog-call surveys, baited turtle traps, coverboards, and incidental encounters (Campbell and Christman 1982, Dodd 2003, Heyer et al. 1994, Ryan et al. 2002, Vogt and Hine 1982). Incidental encounters occurred mainly on or near roads during travel within the sites. We also searched specific areas (e.g., old dump sites) of ARNWR for herpetofaunal species.

We randomly located canal surveys monthly from May to July in each PW and MFS site (3 x 6) and in the AWC site (n = 21). While walking parallel along canals for 1.6 km, we visually surveyed for all reptiles and amphibians for 60 minutes (Heyer et al. 1994). We randomly located paired transects (parallel and 10 m apart) at 9–11 locations on each site from March to late June (n = 126 transects, including 6 in AWC). Transect surveys began 15 m from adjacent canals, perpendicular to the canals and roads, and proceeded away from the roads. We searched transects for an average of 9 min and 188 m (PW and MFS mean time = 9 min, 95% CI = ± 1 min; PW mean distance = 190 m, 95% CI = ± 25 m; MFS mean distance = 186 m, 95% CI = ± 26 m; distance calculated by Global Positioning System with 2–5 m accuracy). Along transects, we visually searched for all exposed herpetofauna and for those covered by large objects (e.g., logs). At the beginning (near-canal) and end (far-canal) of each transect, we established 10- x 10-m quadrats (n = 252, two times the number of transects). We visually searched quadrats for 10 minutes and removed large ground cover to reveal species that may not have been readily visible otherwise.

We randomly placed three drift fences in each site (n = 21, including 3 in AWC) along unpaved refuge roads. Roads were the only locations with consistently high elevations for drift fences and usually remained unflooded during trapping. Fences were about 2 m off the road. Roads had little or no traffic; therefore, we expected no effects on captures from vehicles. We constructed drift fences with 46-cm high aluminum flashing, 15-m long and buried a 19-L plastic bucket to ground level at each end. Fences were buried 10 cm below ground. We placed funnel traps in the middle of each fence on the side facing the road beginning halfway through the trapping period. Funnel traps consisted of 50-cm diameter by 90-cm long cylinders made from rigid hardware cloth (ca. 4-mm mesh) with a funnel (ca. 7-cm inside opening) at both ends (Heyer et al. 1994). We opened drift fences on a rotating schedule, with three MFS sites (and white cedar site) open for two days (total = 10 days) and the remaining three PW sites open for the next two consecutive days (total = 9 days) for a total of 19 days from April to July. All drift fences were checked early in the day to minimize desiccation of captured individuals.

We conducted nighttime aural surveys monthly from March to July (n = 10, 5 each on PW and MFS unpaved roads) during warm, rainy periods on each site. We drove slowly (16–32 km/h) for ca. two hours and stopped to record all calling anurans using a calling index (1 = individuals can be counted, space and time between calls; 2 = individuals can be distinguished, calls overlapping, 3 = full chorus, calls continuous, constant, and overlapping). We placed turtle traps in canals near each drift fence and checked and rebaited them daily (sardines, chicken, and crabs) for three trap-nights during 8 June–18 July 2005. Four coverboards (1.5 m^2/board) were randomly placed (2 boards near 2 drift fence locations) in each PW and MFS site (n = 12) and at the AWC site (n = 2) in May and were checked once monthly thereafter for three months.

We recorded each reptile or amphibian species incidentally encountered and its area (PW, MFS, or AWC) and site (1, 2, or 3 for PW and MFS only). Voucher specimens consisting of individuals found dead or that were found injured were deposited at The University of Georgia Museum of Natural History, Athens, GA. We used both published (Braswell and Wiley 1982, Palmer and Braswell 1995) and unpublished (North Carolina State Museum Natural Sciences voucher specimens) museum records to create a species list of reptiles and amphibians occurring in counties adjacent to ARNWR. A species list, generated by pooling all ARNWR survey data, was then compared with the regional species list.

Data analysis

We used species-richness analysis developed by Nichols et al. (1998a, 1998b) and Hines et al. (1999), which accounts for differences in detection rates between PW and MFS areas, to estimate richness and compare community composition between the PW and MFS sites of ARNWR. Program COMDYN4 (available at: www.mbr-pwrc.usgs.gov/software.html) was used to calculate these estimates (means, SE, and 95% confidence intervals [Hines et al. 1999]). Variance estimates were calculated by bootstrapping data 200 times using a default seed. A chi-square goodness of fit test was used to determine if the capture-recapture model M_h (COMDYN) was the most appropriate model for the data ($P > 0.10$) (Nichols et al. 1998a). This model permits heterogeneous detection probabilities among species. A chi-square goodness of fit test was also used to determine if species detection probabilities differed between areas ($P < 0.05$). We made every effort to spend equal effort searching each area using the same techniques (Nichols et al. 1998a, 1998b). We used all surveys combined (except incidental captures) within each site to calculate species presence or absence in the sites for use in COMDYN4. Amphibian and reptilian species richness was analyzed separately with COMDYN4 to determine differences between PW and MFS. We also analyzed near- and far-canal quadrat data for differences in species richness using COMDYN4 to check for edge effect.

We calculated species diversity using the Shannon-Wiener index (H') and evenness index (J'), which varies from 0 (completely uneven distribution of individuals by species) to 1 (all species have equal number of individuals) (Pielou 1966). J' equals H' divided by the maximum for H', given the number of species and individuals surveyed (Lloyd and Ghelardi 1964 in Krebs 1999). PW and MFS areas were compared by combining all sampling methods except nighttime aural surveys (no individual count data) to produce a sample of species and individuals collected with the same effort and methods. We tested differences between H' using Hutcheson's (1970) method.

We used a community percentage similarity index (Renkonen 1938 in Krebs 1999, Whittaker 1975) to determine differences in PW and MFS herpetofaunal communities using the sample of species and individuals collected for diversity analysis. This simple index is one of the best quantitative similarity coefficients available (Wolda 1981).

Results

We found a total of 39 species present at 7 ARNWR sites, consisting of 28 reptiles and 11 amphibians. Of 39 species that we observed elsewhere, incidental captures located 38, canal searches located 29 species, drift fences trapped 16 species, and nighttime aural and transect/quadrat surveys each discovered 14 species, while searches of coverboards revealed only 2 species (Table 1). Turtle trapping located 6 of 7 aquatic turtle species on the refuge (Table 1). We found an additional 10 species on the refuge, but these were located by searching outside of our 7 sites in selected habitats (e.g., old dumps). Accordingly, we found a total of 49 species on ARNWR during 2000–2001 (Appendix A). The mainland adjacent to ARNWR contains 14 more species than the refuge itself, for a total 63 species in the region (28% more than at ARNWR [Appendix B]; Braswell and Wiley 1982, Palmer and Braswell 1995). Of these additional species, 6 are amphibians and 8 are reptiles.

Using COMDYN4, we estimated 39 species in MFS and 32 species in PW sites (Table 2). We found similar numbers of species in common for MFS (26 spp.) and PW (24 spp., Table 2). COMDYN's Model M_h from Nichols et al. (1998a) adequately fit ($P > 0.10$) all data for these estimates (Table 2).

We found no difference in species-detectability probabilities between MFS and PW (Table 2). Therefore, we used actual numbers of species

Table 1. Number of amphibian and reptile species and relative abundance by survey method, Alligator River National Wildlife Refuge, NC, 2000–2001.

Survey method	Number of species	Number of individuals	Number of amphibian species	Number of reptilian species
Incidental capture	38	317	10	28
Canal transect	29	110	12	17
Drift fences	16	240	7	9
Transects and quadrats	14	347	5	9
Nighttime aural	14	ND[A]	14	0
Turtle traps	6	36	0	6
Coverboards	2	3	1	1

[A]No data - Numbers based on call index: 1 = individuals can be counted; 2 = calls of individuals can be distinguished, but there is some overlapping calls; 3 = continuous chorus, individual calls not heard.

observed (more powerful) to calculate differences between PW and MFS areas (altLAMBDA in COMDYN4, Eq. 3 in Nichols et al. 1998a). The difference between MFS and PW sites, measured as a ratio of species richness (MFS/PW sites), indicated that species richness was similar (Table 2). COMDYN4 does not provide 90% confidence intervals, which was acceptable in this study and would indicate that more species (19% or ratio of 1.19, Table 2) inhabited MFS than PW, based on a more precise estimate using Eq. 3 in Nichols et al. (1998a).

Further analysis by amphibian and reptilian classes revealed almost identical species-richness estimates for amphibians in MFS (18 spp) and PW (17 spp) sites. However, we found 31% more reptilian species in MFSs (17 spp.) than in PWs (13 spp.) using Eq. 3 (Nichols et al. 1998a and Table 3). The ratio of species richness (MFS/PW), 1.31, was estimated using actual species-richness samples because there was no difference in detection probability. The ratio's 95% CI indicated significantly more reptile species were found in MSF compared to PW (Table 3).

Because of potential edge effects from the canal on quadrat and transect surveys, we analyzed far- and near-canal quadrats separately for species richness (COMDYN4). Observed species richness between these quadrats was identical (spp. = 4). For far- and near-canal quadrats, species-richness estimates were also similar (far-canal quadrat: 5 species [95% CI = 5 to15 spp.], near-canal quadrat: 7 species [95% CI = 6 to 18 spp.]). Species in common for near- and far-canal quadrats were estimated at 5 (95% CI = 3 to 8) and 5 (95% CI = 1 to 15), respectively.

Table 2. Estimates of quantities for herpetofaunal community parameters[A] of pocosin wetlands (PW) and mixed forest swamp (MFS) sites of Alligator River National Wildlife Refuge, NC, 2000–2001.

Quantity	Number sampled	Estimate	SE	95% CI
Species richness, $N_{(PW)}$	28	32.38	3.55	28.00–41.29
Species richness, $N_{(MFS)}$	33	39.21	4.04	33.00–47.75
Species present in MFS and observed PW	22	25.76	3.85	19.50–35.34
Species present in PW and observed MFS	22	23.99	3.51	18.00–31.7
Proportion of PW species present in MFS	-	0.92	0.10	0.67–1.00
Proportion of MFS species present in PW	-	0.73	0.11	0.56–1.00
Estimated ratio of species richness estimated as $N_{(MFS)}/N_{(PW)}$	-	1.21	0.16	0.93–1.54
Estimated ratio of species richness (MFS/PW) estimated using actual species richness sample[B]	-	1.19	0.12	0.94–1.38
Species not present in PW, present in MFS	9	9.42	5.15	0.00–20.46
Estimate species detection probability (PW)	1.0[C]	0.86	0.09	0.68–1.00
Estimate species detection probability (MFS)	1.0[C]	0.84	0.09	0.69–1.00

[A]Estimates calculated using the program COMDYN4 (Hines et al. 1999); Model M_h fit all data ($P > 0.10$) for detection frequencies (Nichols et al. 1998a).
[B]Actual sample data may be used to calculate this ratio because there was no difference in species detection probability between PW and MFS sites (Eq. 3 in Nichols et al. 1998a).
[C]By assumption.

Herpetofaunal communities differed in species composition between MFS and PW sites based on COMDYN4. An estimated 9 species present in MFS sites were not present in PW sites (Table 2). The actual samples indicated 9 species present in MFSs that were not present in PWs and 4 species present in PWs that were not found in MFS. The estimated fraction of PW species found in MFS was 92%, whereas the reciprocal fraction of MFS species found in PW was 73%. Confidence intervals for both estimated fractions, however, overlapped considerably (Table 2, proportion parameter). The herpetofaunal community similarity index between MFS and pocosin was 0.566, indicating a moderate difference in species and numbers of individuals between them (0 = no similarity, 1 = completely similar species and numbers). We detected no additional species in the AWC site (n = 1) than in MFS and PW sites. AWC had few species and low numbers (12 spp. and 38 individuals for same collection methods, compared to an average of 18 and 24 spp. in PW and MFS sites [n = 3], respectively).

Contrary to species-richness analysis (above), we discovered higher H' ($P < 0.0001$) in PW sites (2.6680, $e^{H'} = 14.41$) compared to MFS sites (2.1535, $e^{H'} = 8.61$). J' was lower in MFSs (0.6212) compared to PWs (0.8010). Dominance by *Rana clamitans*, *R. sphenocephla utricularia*, and *R. virgatipes* caused lower J' in MFSs; therefore, H' was lower than expected for the same number of species. These 3 ranid species accounted for 34.3, 21.7, and 14.6% of the individuals surveyed in forest swamp sites (70.6% of total individuals surveyed, n = 809). Also, each of these species was captured or observed predominantly by different methods (*R. clamitans*—transects and quadrats, *R. sphenocephla utriculuria*—canal, and *R. virgatipes*—drift fences).

Table 3. Estimates of quantities for reptilian community parameters[A] of wetland pocosins (PW) and mixed forest swamp (MFS) sites of Alligator River National Wildlife Refuge, NC, 2000–2001.

Quantity	Number sampled	Estimate	SE	95% CI
Species richness, $N_{(PW)}$	13	15.71	2.63	13.00–22.59
Species richness, $N_{(MFS)}$	17	19.02	2.34	17.00–25.25
Species present in MFS and observed PW	9	12.20	2.72	7.33–17.72
Species present in PW and observed MFS	9	11.00	2.94	7.33–17.72
Proportion of PW species present in MFS	-	0.94	0.15	0.53–1.00
Proportion of MFS species present in PW	-	0.64	0.17	0.33–1.00
Estimated ratio of species richness estimated as $N_{(MFS)} / N_{(PW)}$	-	1.21	0.21	0.82–1.66
Estimated ratio of species richness $N_{(MFS)} / N_{(PW)}$ estimated using actual species richness sample[B]	-	1.31	0.17	1.00–1.70
Species not present in PW, present in MFS	8	4.28	3.60	0.00–13.09
Estimate species detection probability (PW)	1.0[C]	0.83	0.12	0.57–1.00
Estimate species detection probability (MFS)	1.0[C]	0.89	0.10	0.66–1.00

[A]Estimates calculated using the program COMDYN4 (Hines et al. 1999); Model M_h fit all data ($P > 0.10$) for detection frequencies (Nichols et al. 1998a).

[B]Actual sample data may be used to calculate this ratio because there was no difference in species detection probability between PW and MFS sites (Eq. 3 in Nichols et al. 1998a).

[C]By assumption.

Discussion

Our one-year survey, using a variety of methods, encountered the majority of previously recorded reptile and amphibian species (49 of the 52 species) of ARNWR (Braswell and Wiley 1982; North Carolina State Museum of Natural Sciences, unpubl. specimen records; Palmer and Braswell 1995). Previous records were accumulated from numerous surveys and a longer time period.

By far, most species found were surveyed incidentally within the seven randomly selected sites; however, this method did not provide appropriate random and standardized samples. Using incidental surveys to quickly determine species richness with limited information on relative abundance and evenness excludes information on species dominance, which could elucidate problems, e.g., pollution causing high dominance by a few species (Wilhm 1967). Diversity, expressed as H', depends on the number of species and evenness (Pielou 1975).

General collecting in appropriate herpetofaunal habitat, as well as incidental searches, can provide useful species lists (Scott 1994). In our study, these searches found 10 more species than in random searches of seven large, randomly selected areas. Over time, general collecting will provide gamma-diversity, i.e., landscape-scale regional diversity (Odum and Barrett 2005; Whittaker 1960, 1975). General short-term collecting, however, is not a substitute for more quantitative methods (Scott 1994). Standardizing and randomizing the effort in these searches (see Amphibian Research and Monitoring [ARMI] at www.edc2.usgs.gov/armi/, Dodd 2003) would provide information useful for estimating species richness using COMDYN4 (Hines et al. 1999; Nichols et al. 1998a,b), in addition to completing a species list for the management area.

Conversely, coverboards, turtle traps, and nighttime aural surveys found the lowest number of species mainly because of less intensive sampling efforts. These methods, however, provided more species richness (e.g., frog species undetected by other methods), but little information on relative abundance (e.g., call count index). By using transects and quadrats, drift fences, and canal surveys, we discovered a moderate number of species and some of the highest numbers of individuals per species with more intensive sampling efforts than above (Table 2). Time-constrained surveys, such as transect and canal visual surveys, may provide the most efficient and least costly method to monitor herpetofaunal diversity at ARNWR (see summary in Dodd 2003).

Overall, MFS and PW sites contain different herpetofaunal communities when compared in species relative abundance, species richness, H', J', and species composition. However, amphibian species richness and composition was almost identical for PW (estimated 17 spp., 95% CI = 15–21) and MFS (estimated 18 spp., 95% CI = 16–23). The same 15 anurans were found in both areas (Appendix A), and only one salamander species, *Plethodon chlorobryonis*, was found in MFS. Contrary to this similarity and to herpetofaunal species richness differences from COMDYN4, however, was the H' difference caused by dominance of three *Rana* species, which caused lower evenness, J', and therefore lower H' in MFS than in PW. *Rana* is widespread worldwide and many species are abundant, especially *R. clamitans* and *R. utricularia* (Conant and Collins 1998). Dominance by *Rana* may be caused by an unnatural environment, disturbance, or lack of salamander predators in the community (Corn and Bury 1989; Morin 1981, 1986).

Based on greater species richness of reptiles in MFS than PW, we suggest that this area is important in maintaining herpetofauna species richness on ARNWR. MFS habitats have more vertical structure (e.g., older, taller trees) and higher elevations with more pockets of upland habitats (horizontal heterogeneity) than PW, which may explain higher reptile species richness. More reptiles than amphibians occupied mixed forest habitat than clearcuts in a similar South Carolina study (Ryan et al. 2002). MFS sites may also have had less effect from saltwater intrusion from storms and wild fires, which regularly setback plant succession and exclude species associated with older forests at lower elevations in PW along the eastern edge of ARNWR (Gibbons and Coker 1978) .

COMDYN4 provided models (M_h, χ^2, $P > 0.10$) suitable for estimating herpetofaunal species richness differences between two major areas of ARNWR (Nichols 1998a, 1998b). Species presence/absence sampling allows a wide variety of field methods to be used to calculate species-richness estimates for monitoring of herpetofaunal diversity or community comparisons. COMDYN4 provides information that can be used to determine changes in areas with time, e.g., local extinctions, but can also determine differences within the herpetofaunal classes, e.g., reptile species richness, or other subgroups (Boulinier et al. 1998). If species detection probabilities are similar, as in this study, use of species-richness count data will provide a more precise estimate of species richness and differences between areas (Eq. 3 in Nichols 1998a). This was the case in two species-richness comparisons, which found differences in richness (> 90% CI). ARNWR herpetofaunal community percent similarity index (56.6%) between the two areas provided information similar to estimates from COMDYN4, possibly because of similarities in species-detection probabilities between the areas. These estimates along with one year of general collecting tallied 49 of 52 species known in the area. Herpetofaunal relative abundance by species, however, indicated that the Shannon index, H', provides additional information from the evenness index, J', that species richness analysis alone did not reveal. Methods that incorporate estimates of species richness and relative abundance (abundance distribution) may be beneficial for community analysis in some instances when relative abundance affects diversity as expressed in H' and J' (Nichols and Conroy 1996).

Conservation Implications

MFS and PW are important for maintaining herpetofauna species diversity and should be conserved and managed appropriately so that species composition of the refuge is maintained or increased, especially for rare species in the region. This is critical, because we know little about individual movements of most reptile and amphibian species between habitats (e.g., Bodie and Semlitsch 2000, Buhlmann and Gibbons 2001, Semlitsch 1998, Semlitsch and Bodie 2003) and the potential metapopulation dynamics of these distinct areas. Managers and scientists should strive to view the landscape as a whole, maintain the natural levels of spatial and temporal heterogeneity by using management of resources to match natural patterns, maintain the landscape (not increase fragmentation), and be attentive to the effect of critical thresholds, i.e., maintain natural habitat abundance (Pearson et al. 1996).

ARNWR is a "coldspot" on a regional scale of herpetofaunal diversity, which contains a subset of species present in the surrounding region, but it has a large functioning ecosystem with unique attributes, such as a large, top-level carnivore, *Canis rufus* Audubon and Bachman (red wolf; Kareiva and Mavier 2003). This wetland is important because of its size (661 km^2) and protected status (USFWS). It will provide habitat for many herpetofaunal species and habitats may improve further with restoration of natural water flow and management (Semlitsch 2000, Wilson 1995). Because ARNWR is connected to the mainland region by a relatively small (5- x 13-km) land bridge managed for the most part by the Department of Defense (Dare County Range), it is possible that management in this corridor or land bridge may affect the chances of successful herpetofaunal colonization of ARNWR. Colonization, therefore, especially after local extinctions (e.g., caused by saltwater) may depend on habitat management on Dare County Range. Most of the species absent from the refuge cannot successfully disperse over large bodies of brackish and salt water that surrounds ARNWR on three sides (Gibbons and Coker 1978).

The discussion of the causes of decline for herpetofaunal species have been ongoing for decades (Alford and Richards 1999, Gibbons et al. 2000) and have reached a high point with recent nationally broadcast news of worldwide losses of amphibians (Stokstad 2004). These declines will only be reversed by monitoring herpetofauna and by further research with involvement of managers. ARNWR has the potential to increase its herpetofaunal diversity by 13 species from the mainland near the wetland peninsula. Potentially, management could provide the energy necessary for increasing herpetofauna biodiversity with a moderate subsidy-stress (Odum et al. 1979). Monitoring of herpetofauna may also provide information to help prevent collapses of herpetofaunal communities, such as we have seen in other regions of the United States (e.g., Drost and Fellers 1996).

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Appendix A. List of herpetofaunal species for Alligator River National Wildlife Refuge, Dare and Hyde County, NC, August 2001–September, 2002[A,].

Species[B]	Common name[B]
Frogs and toads	
Acris gryllus (LeConte)	Southern Cricket Frog
Bufo fowleri Hinckley	Fowler's Toad
Bufo terrestris (Bonnaterre)	Southern Toad
Gastrophryne carolinensis (Holbrook)	Eastern Narrow-mouthed Toad
Hyla chrysoscelis Cope	Cope's Gray Treefrog
Hyla cinerea (Schneider)	Green Treefrog
Hyla femoralis Bosc	Pine Woods Treefrog
Hyla squirella Bosc	Squirrel Treefrog
Pseudacris brimleyi Brandt and Walker	Brimley's Chorus Frog
Pseudacris crucifer (Wied-Neuwied)	Spring Peeper
Pseudacris ocularis (Bosc and Daudin)	Little Grass Frog
Rana catesbeiana Shaw	American Bullfrog
Rana clamitans Latreille	Green Frog
Rana sphenocephala utricularia Harlan	Southern Leopard Frog
Rana virgatipes Cope	Carpenter Frog
Salamanders	
Amphiuma means Garden	Two-toed Amphiuma
Plethodon chlorobryonis Mittleman	Atlantic Coast Slimy Salamander
Lizards	
Eumeces inexpectatus Taylor	Southeastern Five-lined Skink
Eumeces laticeps (Schneider)	Broadhead Skink
Sceloporus undulatus[C] (Bosc and Daudin in Sonnini and Latreille)	Eastern Fence Lizard[C]
Scincella lateralis (Say)	Little Brown Skink
Turtles	
Chelydra serpentina serpentina (Linnaeus)	Eastern Snapping Turtle
Chrysemys picta (Schneider)	Painted Turtle
Clemmys guttata (Schneider)	Spotted Turtle
Kinosternon subrubrum (Lacepede)	Eastern Mud Turtle
Pseudemys rubriventris (Leconte)	Northern Red-bellied Cooter
Sternotherus odoratus (Latreille)	Stinkpot
Terrapene carolina (Linnaeus)	Eastern Box Turtle
Trachemys scripta (Schoepff)	Pond Slider
Snakes	
Agkistrodon contortrix (Linnaeus)	Copperhead
Agkistrodon piscivorus (Lacépède)	Cottonmouth
Carphophis amoenus (Say)	Eastern Wormsnake
Coluber constrictor Linnaeus	Eastern Racer
Crotalus horridus Linnaeus	Timber Rattlesnake ("Canebrake")
Diadophis punctatus (Linnaeus)	Ring-necked Snake
Elaphe guttata (Linnaeus)	Cornsnake

Species[B]	Common name[B]
Elaphe obsoleta (Say)	Eastern Ratsnake
Farancia erytrogramma (Palisot de Beauvois)	Rainbow Snake
Lampropeltis getula (Linnaeus)	Common Kingsnake
Lampropeltis triangulum triangulum x *elapsoides* (Lacépède)	"Coastal Plain Milksnake"[D]
Nerodia erythrogaster erythrogaster (Forster)	Red-bellied Watersnake
Nerodia fasciata (Linnaeus)	Southern Watersnake
Nerodia taxispilota (Holbrook)	Brown Watersnake
Opheodrys aestivus (Linnaeus)	Rough Greensnake
Regina rigida (Say)	Glossy Crayfish Snake
Storeria dekayi dekayi (Holbrook)	Northern Brownsnake
Storeria occipitomaculata (Storer)	Red-bellied Snake
Thamnophis sauritus (Linnaeus)	Eastern Ribbonsnake
Thamnophis sirtalis (Linnaeus)	Common Gartersnake
Alligators and Crocodiles	
Alligator mississippiensis (Daudin)	American Alligator

[A]Species not found during 2000–2001 survey, but previously documented from ARNWR by North Carolina State Museum of Natural Sciences herpetologists: *Farancia abacura* (Holbrook) (Red-bellied Mud Snake), *Seminatrix pygaea* (Cope) (Black Swampsnake), *Stereochilus marginatus* (Hallowell) (Many-lined Salamander) (Braswell and Wiley 1982; North Carolina State Museum of Natural Sciences, unpubl. specimen records; Palmer and Braswell 1995).

[B]Scientific names, authorities, and common names according to Crother (2001).

[C]Possibly seen, not confirmed; therefore not counted in species list for Alligator River National Wildlife Refuge.

[D]Thought to be an intergrade between the scarlet kingsnake and the Eastern Milk Snake.

Appendix B. List of 14 amphibian and reptile species found in the counties surrounding the peninsular Alligator River National Wildlife Refuge, i.e., mainland Dare County, NC, that do not occur on the peninsula itself (Braswell and Wiley 1982, Palmer and Braswell 1995).

Species[A]	Common name[A]
Frogs and toads	
Scaphiopus holbrookii (Harlan)	Eastern Spadefoot
Salamanders	
Ambystoma maculatum (Shaw)	Spotted Salamander
Ambystoma opacum (Gavenhorst)	Marbled Salamander
Eurycea guttolineata (Holbrook)	Three-lined Salamander
Notophthalmus viridescens viridescens (Rafinesque)	Red-spotted Newt
Pseudotriton montanus Baird	Mud Salamander
Lizards	
Anolis carolinensis (Voigt)	Green Anole
Eumeces fasciatus (Linnaeus)	Common Five-lined Skink
Ophisaurus ventralis (Linnaeus)	Eastern Glass Lizard
Sceloporus undulatus (Bosc and Daudin in Sonnini and Latreille)	Eastern Fence Lizard
Snakes	
Lampropeltis calligaster (Harlan)	Yellow-bellied Kingsnake
Storeria occipitomaculata (Storer)	Red-bellied Snake
Virginia striatula (Linnaeus)	Rough Earthsnake
Virginia valeriae Baird and Girard	Smooth Earthsnake

[A]Scientific names, authorities, and common names according to Crother (2001).