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CHARACTERIZING FISH ASSEMBLAGE STRUCTURE IN THE PENOBSCOT RIVER PRIOR TO DAM REMOVAL

By

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B.S. Cornell University, 2007

A THESIS

Submitted in Partial Fulfillment of the

Requirements for the Degree of

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THESIS ACCEPTANCE STATEMENT

On behalf of the Graduate Committee for Ian Kiraly, I affirm that this manuscript is the final and accepted thesis. Signatures of all committee members are on file with the Graduate School at the University of Maine, 42 Stodder Hall, Orono, Maine.

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CHARACTERIZING FISH ASSEMBLAGE STRUCTURE IN THE PENOBSCOT RIVER PRIOR TO DAM REMOVAL

By Ian A. Kiraly

Thesis Advisor: Dr. Stephen Coghlan, Jr.

An Abstract of the Thesis Presented
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The Penobscot River drains the largest watershed in Maine, and once provided spawning and rearing habitats to at least 11 species of diadromous fish. The construction of dams blocked migrations of these fish and likely changed the structure and function of fish assemblages throughout the river. Further alteration to fish assemblage structure likely occurred as a result of habitat fragmentation and alteration. The proposed removal of two main-stem dams, improved upstream fish passage at a third dam, and construction of a fish bypass on dam obstructing a major tributary is anticipated to increase passage of diadromous and resident fishes. To sample fish assemblages within the lower 70 kilometers of the Penobscot River prior to dam removal, we used standardized boat electrofishing methods during both summer and fall in 2010 and 2011 while implementing two sampling designs. Fixed-station sampling on the Penobscot River was conducted at eleven pre-established 1000-meter transects. Stratified-random sampling was conducted among nine strata, at multiple randomly selected 500-meter transects within each stratum. Major tributaries were also sampled along eight fixed-station transects. In total, we captured 61,837 fish of 35 species while sampling 114 kilometers of river and tributary shoreline. Our sampling designs were equivalent in precision and efficiency for encountering species and estimating total species richness; we found no significant differences between sampling designs for the proportional abundance of all species, although the stratified-random design was slightly more efficient for characterizing proportional abundance. We combined data from both sampling designs for further analyses and identified longitudinal patterns of fish assemblage structure within the study area. Distinct fish assemblages were present among river sections bounded by dams, indicating that dams were a major driver of fish assemblage patterns within the river. Alewife (*Alosa pseudoharengus*) and blueback herring (*Alosa aestivalis*) were captured frequently within the tidal river section, but at no locations upriver. *Fundulus* species were also abundant within the tidal river section. Smallmouth bass (*Micropertus dolomieu*) and pumpkinseed (*Lepomis gibbosus*) were most prevalent within Veazie Dam impoundment, along with the free flowing river section immediately upriver. Further upriver, warm-water species such as chain pickerel (*Esox niger*), brown bullhead (*Ameiurus nebulosus*), and yellow perch (*Perca flavescens*), along with cyprinid species such as common shiner (*Luxilus cornutus*) and fallfish (*Semotilus corporalis*) were more prevalent than within any other river section. Patterns of fish assemblage structure did not change considerably during our sampling; we identified relatively few species which contributed to seasonal and annual variability within the main-stem river, including smallmouth bass, white sucker (*Catostomus commersoni*), pumpkinseed, and golden shiner (*Notemigonus crysoleucas*). We predict that many anadromous fish will migrate further upriver after dam removal, potentially causing broad shifts in fish assemblage structure. Improved connectivity among habitats for many fish species could

also change the longitudinal pattern of fish assemblage structure within the river. While it is difficult to predict specific changes to fish assemblages in this large river, such predictions can be tested by future studies to evaluate river rehabilitation success and the recovery of historically important fish species.

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CHAPTER 1

A COMPARISON OF TWO SAMPLING DESIGNS FOR FISH ASSEMBLAGE ASSESSMENT IN THE PENOBSCOT RIVER

Introduction

Characterizing fish assemblage structure is an important component within fisheries research and management. Some assessments are conducted within relatively large ecosystems over multiple seasons or years, and the sampling effort required for researchers to provide repeatable estimates is often unknown. Assessments are particularly difficult in large rivers where longitudinal variation and impacts of dams on fish assemblage structure can be profound. Low levels of sampling effort yield imprecise data, which could result in ambiguous results and poorly informed recommendations for management. Researchers must balance precision with many other considerations, including the potential bias within the sampling design and a limited budget (Hughes and Peck 2008). Therefore, it is important to evaluate precision, efficiency, and bias when designing or choosing a sampling design for fish assemblage assessment.

Sampling designs can vary in both total effort expended and effort expended per site. Comparing species inventories that are derived from sampling at different levels of effort is often challenging because results are dependent on sampling effects (Colwell et al. 2004). Species-accumulation curves can be used effectively to compare species inventories from different survey methods, different habitats, and from different times (Moreno and Halffter 2000; McCune and Grace 2002; Lapointe et al. 2006). Species-accumulation curves show the estimated number of species that can be encountered at any amount of effort up to the total effort expended, and can be projected to predict the total number of species present. The shape of species-accumulation curves is affected by species diversity and relative abundance (Thompson and Withers 2003) as well as species distributions (Kanno et al. 2009). Sampling design can also affect the shape of the curve (Thompson et al. 2007); a more efficient design will produce a curve that rises to an asymptote more quickly than a less efficient design.

Species-accumulation curves are non-linear. As sampling effort increases, the number of species encountered accumulates rapidly (Lyons 1992), as does the precision of species richness estimates. The curve asymptotes at high levels of sampling effort at which all possible species are encountered and the precision of species richness estimates is maximized. At an intermediate level of sampling, the slope of the curve decreases and additional effort yields few new species per unit effort; precision gained per unit effort is also relatively low. Interpretations of species-accumulation curves allow researchers to optimize sampling and make recommendations for further effort (Soberon and Llorente 1993).

Species-accumulation curves utilize presence data and do not incorporate the abundance of each species. Dissimilarity curves provide similar insights to species accumulation curves by plotting the amount of compositional change that occurs with increased sampling effort (McCune and Grace 2002). If the encountered assemblage changes little with increased sampling effort, then community analyses incorporating abundance should yield precise results. Steeply sloped areas of the curves indicate rapidly changing dissimilarity values with increased sampling effort. Alternatively, shallow slopes indicate that dissimilarity values change little as effort increases, indicating that little information on proportional abundance is gained from increased sampling effort. As with species-accumulation curves, dissimilarity curves can be interpreted in order to estimate the sampling effort at which precision and sampling effort are optimized. An efficient sampling design will approach an asymptote more quickly than a less efficient design.

Here, we evaluate two sampling designs implemented to assess fish assemblage structure as part of monitoring the effects of dam removal within the Penobscot River Restoration Project (PRRP). Through the removal of Veazie Dam and Great Works Dam, along with the installation of a fish lift at Milford Dam, the PRRP is anticipated to increase passage of diadromous and resident fishes and improve connectivity among currently fragmented habitat of the main-stem river (PRRT 2011). Data presented here were collected as part of biological monitoring of the main-stem Penobscot River for the PRRP in order to quantify anticipated change due to dam removal and evaluate restoration success. Our results will inform decisions regarding sampling effort and design for future monitoring of fish assemblages in the Penobscot River. Our objectives were to: 1) Determine whether our sampling effort produces repeatable estimates of species richness and proportional abundance. 2) Compare the efficiency between sampling designs. 3) Compare estimates of species richness and proportional abundance between sampling designs.

Study Area

The Penobscot River watershed is the largest in Maine, and the second largest in the New England, draining 2.2 million hectares through more than 8800 kilometers of river and streams (Opperman et al. 2011). This study focuses on the lower 70 km of river (Figure 1.1), which ranges from 170 to 600 meters wide with an average annual discharge of ~440 cubic meters per second during recent years (USGS 2011). This reach contains approximately 257 kilometers of shoreline, and includes freshwater tidal, impounded, and free flowing areas. Excluding impoundments, most areas are relatively heterogeneous in shoreline habitat and flow types. The river is impounded at the head of tide by the Veazie Dam (Figure 1.1), and two other main-stem dams (Great Works and Milford) are also included in the study area, which is bounded upriver by the West Enfield Dam.

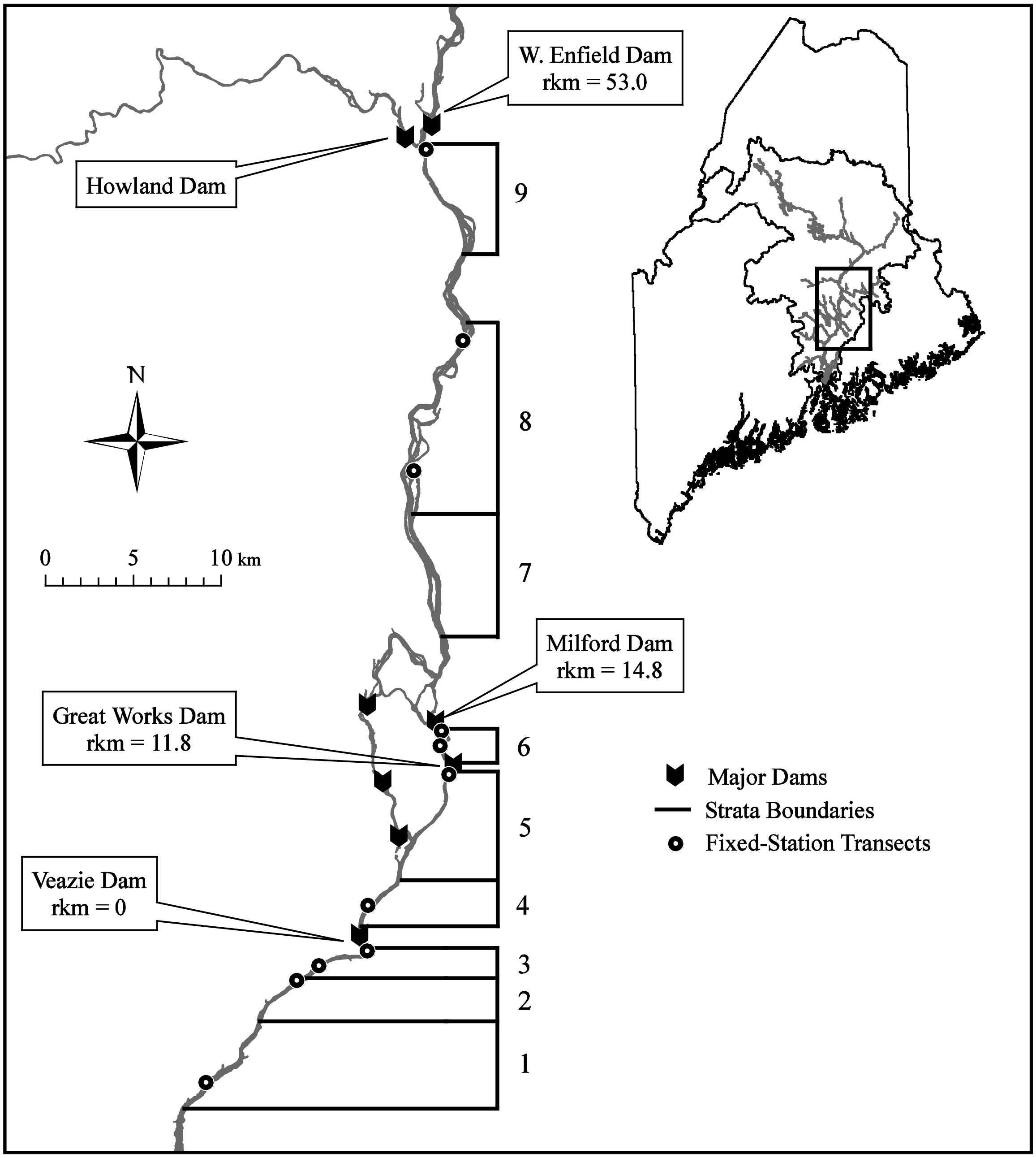

Figure 1.1. Study area. The Penobscot River and locations of main-stem dams (river kilometer = rkm) within our study area, along with boundaries of strata and fixed-station transects where we captured fish via boat electrofishing during 2010 and 2011.

Methods

Sampling Designs

Fixed-station sampling design. The fixed-station design had been implemented as part of an earlier study for two years prior to our data collection (Kleinschmidt Associates 2009a; 2009b); we sampled along transects that were chosen and sampled previously by Kleinschmidt Associates (2009b). The fixed-station sampling design included 11 transects on the main-stem river and eight transects along major tributaries (Figure 1.1), all of which were approximately 1,000 meters in length. Six of the main-stem transects were concentrated in areas above and below dams scheduled for removal (Kleinschmidt Associates 2009b). During the summer 2011 sampling, we divided each fixed transect in half when feasible, to yield data comparable to that collected with the stratified-random design (described below).

Stratified-random sampling design. The stratified-random sampling design was implemented to better account for spatial heterogeneity within the river. We divided the river longitudinally into nine strata (Figure 1.1), the bounds of which were based on dam locations, broad-scale habitat types, and boat access. Using ArcGIS 9.3 (Redland, California), we delineated accessible river shoreline, including shoreline around large islands, into 219 transects approximately 500 meters in length. We selected multiple transects at random from within each stratum; a prioritized list was created to select alternate transects if that area of river was inaccessible by boat.

Fish collection

Single-pass daytime boat electrofishing surveys (Curry et al. 2009) were conducted in the summer (June) and the fall (September-October) during 2010 and 2011. We electrofished only if discharge was less than 425 cubic meters per second at West Enfield, ME (USGS gauge 01034500) and when water temperatures were below 22°C as measured at the start of each transect. We used a 17.5-foot (5.5-meter) Lowe (Lebanon, Missouri) Roughneck aluminum boat equipped with Smith Root (Vancouver, Washington) electrofishing equipment, including two booms with 6-dropper anode arrays, and a GPP 5.0 electrofishing system. We installed custom cathode dropper arrays near and along the bow of the boat. Metal conduit encased half of the droppers in order to increase the cathode surface area (~30,755cm^2); the purpose of this design was to reduce fish injury and mortality. The electrofishing unit was operated using pulsed DC at 60 Hz and 30-40 percent of power, as required to capture fish successfully while limiting injury; settings were chosen to maximize power transfer (Reynolds 1999). Two netters captured shocked fish with Duraframe (Viola, Wisconsin) dip nets; all net bags were constructed of 4.8 mm mesh.

Surveys were conducted by maneuvering the boat parallel and close to shore and fishing in a downstream direction, at a speed equal to or slightly greater than the current. Pockets, eddies, and shoreline were sampled by maneuvering the boat perpendicular or at an angle to shore. Habitat structure (e.g., boulders, large woody debris, and vegetation) were fished thoroughly as well. All fish that were captured were identified to species and measured to the nearest millimeter and tenth of a gram. If age 0 or small fish (length < 80mm) of any species were captured in high abundance (n > 50), these fish were separated by size class, counted, and mass was measured for batches, with length taken to the nearest millimeter for the smallest and the largest specimens in a batch. This method was implemented to collect required data from these specimens while reducing mortality and processing time. Due to endangered species permitting restrictions, we did not attempt to net adult Atlantic salmon *Salmo salar*, Atlantic sturgeon *Acipenser oxyrhynchus*, or shortnose sturgeon *Acipenser brevirostrum*, but rather noted their occurrence visually and considered each encounter as a "capture" for data analysis below. Estimated mass for Atlantic salmon observed in 2010 was calculated by approximating size and year class (Dube et al. 2010) and using historical (Baum 1997) and recent (Bacon et al. 2009) length-mass data. Similar methods were used to estimate mass of Atlantic salmon during 2011, but mass data were available from fish that were captured in the Penobscot River (O. Cox, Maine Department of Marine Resources, unpublished data). Sturgeon mass was estimated using length-frequency and length-mass data provided by G. Zydlewski and M. Altenritter (University of Maine, unpublished data).

Data Analysis

Dataset. Age 1 smallmouth bass *Micropterus dolomieu* (< 30mm) and white sucker *Catostomus commersoni* (< 40mm) were removed from the summer sampling data prior to analyses because the growth of these specimens necessary to recruit to our gear (> 25mm) appeared to be variable among strata for the duration of the summer sampling; by fall, these fish were large enough to be captured reliably within all strata. Catch per unit effort (CPUE) and mass per unit effort (MPUE) for each species was calculated for each stratum and for each fixed-station transect by dividing the total catch or mass by the total length of shoreline electrofished, as measured between start and end GPS coordinates using orthoimagery in ArcGIS 9.3.

Species-accumulation curves. Sample-based species-accumulation curves show the average species richness that is calculated for all subsets of total site effort (the number of transects), as opposed to individual-based species-accumulation curves which show the average species richness calculated for all subsets of the total number of individuals encountered (Kindt and Coe 2005). We constructed sample-based species-accumulation curves using the exact method, which uses analytical formulae to calculate average species richness at each level of effort (Colwell et al. 2004; Kindt and Coe 2005; Oksanen et al. 2011). Standard error of the estimated species richness calculations were not conditional on the empirical data and were derived under the assumption that the first-order jackknife accurately estimates the total species richness. The exact method replaces random re-sampling methods often used to create sample-based species accumulation curves, and provides useful confidence intervals (Colwell et al. 2004). Transect length was different between sampling designs; therefore, all species accumulation curves were scaled by kilometers of electrofishing in order to facilitate direct comparisons of effort (Kindt and Coe 2005). All curves were constructed using the statistical package R (R Development Core Team 2010) and the BiodiversityR and Vegan libraries (Kindt and Coe 2005; Oksanen et al. 2011). Curves were inspected visually for asymptotic behavior. Error bars representing 95% confidence intervals were also plotted and inspected visually across different species accumulation curves; curves and locations along curves with confidence interval overlap were considered similar (Colwell et al. 2004).

Dissimilarity curves. Our dataset contains information on the abundance of each species, but species-accumulation curves only incorporate presence or absence of any species within each transect. Dissimilarity curves were created by randomly re-sampling subsets of the data, and calculating the dissimilarity between the centroid of each subset and the centroid of the total sample (McCune and Grace 2002). High dissimilarity values indicate that subsamples are dissimilar in the abundance of species in comparison to the whole sample. Dissimilarity curves for each sampling event and sampling design were constructed using PC ORD software (McCune and Mefford 1999), and were based on Bray-Curtis dissimilarity measures (McCune and Grace 2002), thereby incorporating proportional abundance of each species. We calculated dissimilarity curves using CPUE data for each species from each sampling design. These curves were then plotted along an *x*-axis scaled to kilometers of electrofishing.

Slopes of curves. We used smoothed spline functions to fit all seasonal species-accumulation and dissimilarity curves and calculated the first derivatives at all levels of effort along the curves (R Development Core Team 2010; Maechler et al. 2011). The first derivative values from dissimilarity curves can be interpreted as percent slope because the Bray-Curtis dissimilarity values on which the curves were based are percentages (McCune and Grace 2002). Species-accumulation curve derivatives were calculated as a change in richness, given a change in effort, and were converted to a percent form for comparison with dissimilarity curve slopes. The relation between the absolute value of the dissimilarity derivatives and percent species-accumulation derivatives for each sampling design was then determined using least squares linear regression (R Development Core Team 2010), and compared with a 1:1 relation in order to analyze the efficiency of each sampling design at assessing abundance-based, assemblage level characteristics when compared to the number of species encountered. A regression with a slope of greater than 1.0 and an intercept below zero indicates more efficient sampling of proportional abundance when compared to species richness (e.g. the slope of the dissimilarity curve is steeper than that of the associated species-accumulation curve for low levels of effort, but shallower than that of the species-accumulation curve at higher levels effort). The opposite holds true for a slope less than 1.0 and an intercept above zero.

Estimation of Total Species Richness. There are a variety of methods for estimating the total estimated species richness (S_{est}) within an area (Colwell and Coddington 1994). Because evaluating S_{est} through extrapolation using asymptotic values of models is unreliable (Palmer 1990; Hortal et al. 2006), we estimated total species richness for both sampling designs from every sampling event with a nonparametric first-order jackknife estimator (Palmer 1990; McCune and Grace 2002; Kindt and Coe 2005). All calculations of S_{est} were performed using the statistical package R with the Vegan library (R Development Core Team 2010; Oksanen et al. 2011).

Abundance and mass estimates. We estimated mean abundance (N/km) and mass (kg/km) for each species and all fish combined using the statistical package R (R Development Core Team 2010) and the survey library (Lumley 2011) for each sampling design and event. Estimates derived from stratified-random sampling were calculated using inverse-probability weights for each stratum(Lumley 2004); we calculated weights as the inverse of the number of transects surveyed within a stratum divided by the total number of transects located within that stratum. The estimates derived from fixed-station sampling were also calculated using sampling weights similar to the stratified-random design, under the assumption that the fixed-station transects are representative of the strata we have delineated.

Comparison of proportional CPUE. Dissimilarity among sampling designs was evaluated by using multi-response permutation procedures (MRPP) complemented with Indicator Species Analysis. MRPP is a non-parametric method that tests for differences in assemblages among groups; it yields a p-value and an A-statistic, both of which must be used to assess dissimilarity (McCune and Grace 2002). The p-value is the likelihood that an observed difference is due to chance, whereas the chance corrected within-group agreement (A), also known as the effect size, describes within-group homogeneity (McCune and Grace 2002). We performed MRPP computations with PC ORD software (McCune and Mefford 1999) after rank transforming the distance matrix. Because many additional analyses is at the stratum scale, as opposed to the transect scale, we combined catch data from transects for both sampling designs from within each stratum and standardized by kilometers of electrofishing. Calculations for MRPP analyses were based on Bray-Curtis dissimilarity, the same measure that was used to create our dissimilarity curves. We conducted Indicator Species Analysis (Dufrene and Legendre 1997) on significant ($\alpha = 0.05$) MRPP comparisons to identify potential bias within our sampling designs. Indicator Species Analysis provides an Indicator Value (IV) and a p-value, along with the relative abundance relative frequency of each species in each group; it is often used in conjunction with MRPP (McCune and Grace 2002), and was performed using PC ORD software (McCune and Mefford 1999). The p-values for Indicator

Species Analysis were calculated using a Monte Carlo test of significance based on 1000 permutations.

Results

Fish collection

Over all sampling events, 88 kilometers of shoreline was surveyed and 45,874 fish were captured that were suitable for analysis. Sampling effort for each event within each design ranged from 9.0 to 15.7 kilometers of electrofishing, except for fall 2011 fixed-station sampling when only 3.0 kilometers of shoreline was electrofished. We encountered 34 species total; 31 species within the stratified-random sampling design, and 30 species within the fixed-station sampling design (Table 1.1). Four design-unique species were encountered using stratified-random sampling; three were encountered using fixed-station sampling (Table 1.1). All design-unique species were encountered in very low abundances and in few places (n = 1-10). One of these species, creek chub, was slightly more abundant (n = 10) than the other design-unique species, and was found along seven stratified-random transects.

Table 1.1. CPUE and MPUE of all species captured on the Penobscot River.

Species	Mean CPUE (n/km) Stratified-Random	Mean CPUE (n/km) Fixed-Station	Mean MPUE (g/km) Stratified-Random	Mean MPUE (g/km) Fixed-Station
Common Shiner *Luxilus cornutus*	258.3	64.3	380	96
Fallfish *Semotilus corporalis*	239.9	65.0	687	269
Redbreast Sunfish *Lepomis auritus*	58.0	29.7	1,011	534
White Sucker *Catostomus commersoni*	34.6	18.5	824	3,510
Smallmouth Bass *Micropterus dolomieu*	85.6	45.9	2,928	3,253
Pumpkinseed *Lepomis gibbosus*	17.1	14.5	116	139
Golden Shiner *Notemigonus crysoleucas*	15.7	6.5	32	18
American Eel *Anguilla rostrata*	15.0	9.4	1,528	1,096
Chain Pickerel *Esox niger*	8.1	1.9	582	234
Sea Lamprey *Petromyzon marinus*	5.1	2.5	83	10
Yellow Perch *Perca flavescens*	4.7	3.0	105	63
Brown Bullhead *Ameiurus nebulosus*	3.0	2.0	449	322
Banded Killifish *Fundulus diaphanus*	7.3	1.0	18	5
Alewife *Alosa pseudoharengus*	3.2	2.3	440	51
Burbot *Lota lota*	1.2	0.7	71	28
Blueback Herring *Alosa aestivalis*	1.6	2.0	47	111
Mummichog *Fundulus heteroclitus*	0.2	0.1	< 1	< 1
Creek Chub *Semotilus atromaculatus*	0.2	0	< 1	0
Eastern Silvery Minnow *Hybognatus regius*	0.7	0.2	1	0
Black Crappie *Pomoxis nigromaculatus*	0.2	0.2	2	1
White Perch *Morone americana*	0.7	< 0.1	7	1
Largemouth Bass *Micropterus salmoides*	0.1	0.2	3	1
Longnose Sucker *Catostomus catostomus*	0	< 0.1	0	37
Atlantic Salmon *Salmo salar*	< 0.1	0.5	271	1,446
Brook Trout *Salvelinus fontinalis*	< 0.1	0	1	0
Blacknose Dace *Rhinichthys atratulus*	< 0.1	0	< 1	0
Blacknose Shiner *Notropis heterolepis*	< 0.1	< 0.1	< 1	< 1
Finescale Dace *Phoxinus neogaeus*	< 0.1	0	< 1	0
Northern Redbelly Dace *Phoxinus eos*	< 0.1	< 0.1	< 1	< 1
American Shad *Alosa sapidissima*	< 0.1	< 0.1	24	24
Threespine Stickleback *Gasterosteus aculeatus*	< 0.1	0	< 1	0
Ninespine Stickleback *Pungitius pungitius*	< 0.1	< 0.1	< 1	< 1
Sturgeon spp. *Acipenser spp.*	0	< 0.1	0	287
Striped Bass *Morone saxatilis*	0	< 0.1	0	94

Curve and slope comparisons

All curves appear to be approaching an asymptote, although none of them are fully asymptotic (Figure 1.2), similar to findings in other studies (Angermeier and Smogor 1995; Kanno et al. 2009); this is typical of curves derived from data collected as a representative sample (Blocksom et al. 2009). Curves begin to approach an asymptote after ~5 kilometers of electrofishing. Sampling during fall 2011 at fixed-station transects produced the only species-accumulation curve that did not show any asymptotic behavior (Figure 1.2). The 95% confidence intervals of stratified-random and fixed-station sampling for each event and at all levels of effort for the species-accumulation curve overlapped considerably (Figure 1.2).

The fixed-station design during fall 2011 also produced the only dissimilarity curve that did not exhibit asymptotic behavior (Figure 1.2). Dissimilarity curves declined more rapidly and leveled out more completely under the stratified-random design than under the fixed-station sampling design (Figure 1.2) during most sampling events. When the derivatives of seasonal species-accumulation curves were plotted and regressed against derivatives of dissimilarity curves (Figure 1.3), the stratified-random design exhibited a slope of 1.103 and an intercept of -0.0066 ($R^2 = 0.9676$, n = 70), and the fixed-station design exhibited a slope of 0.886 and an intercept of 0.0039 ($R^2 = 0.9815$, n = 38).

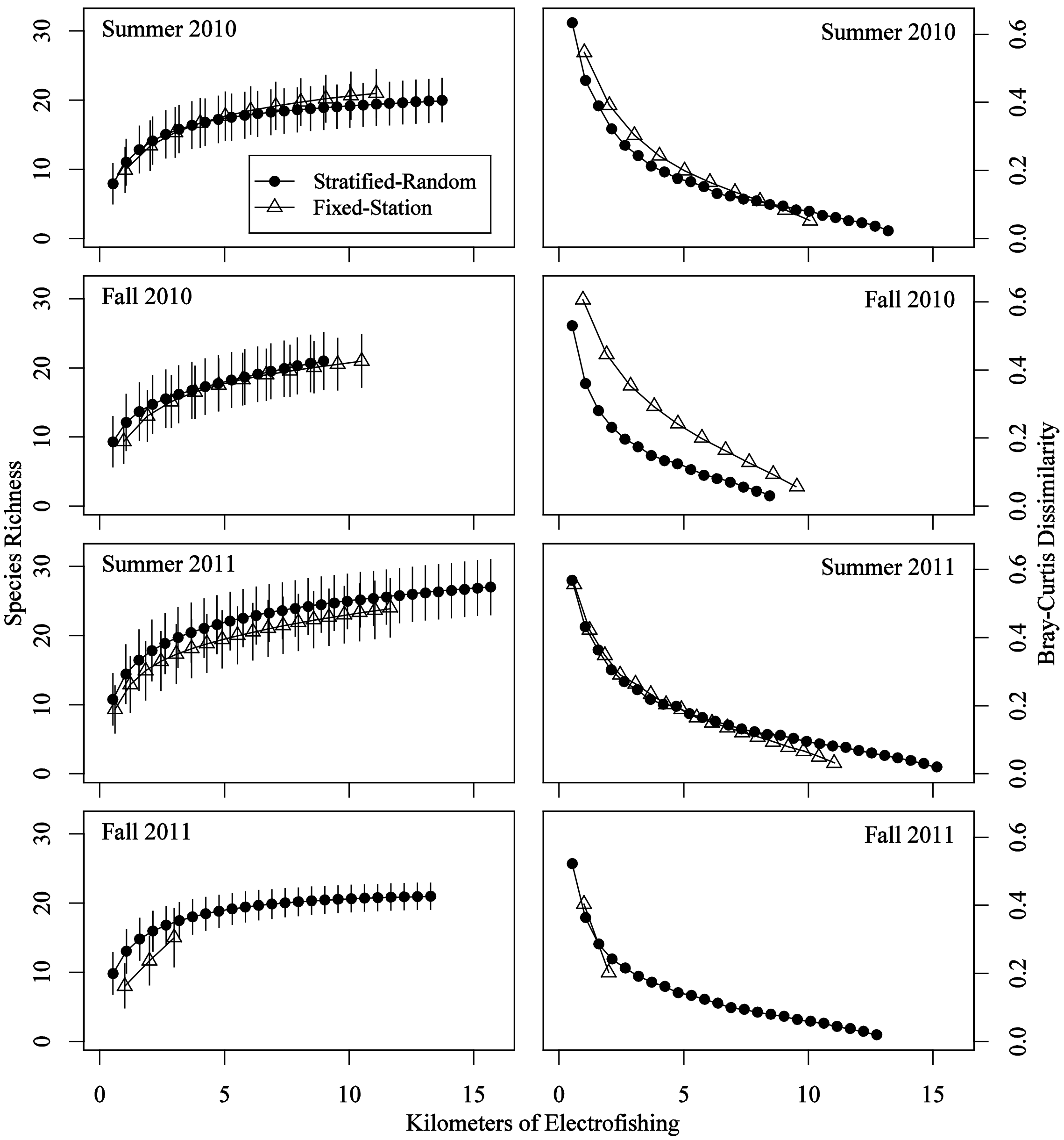

Figure 1.2. Species-accumulation and dissimilarity curves. Data were derived from stratified-random and fixed-station sampling designs on the Penobscot River. Error bars in species-accumulation plots represent 95% confidence intervals.

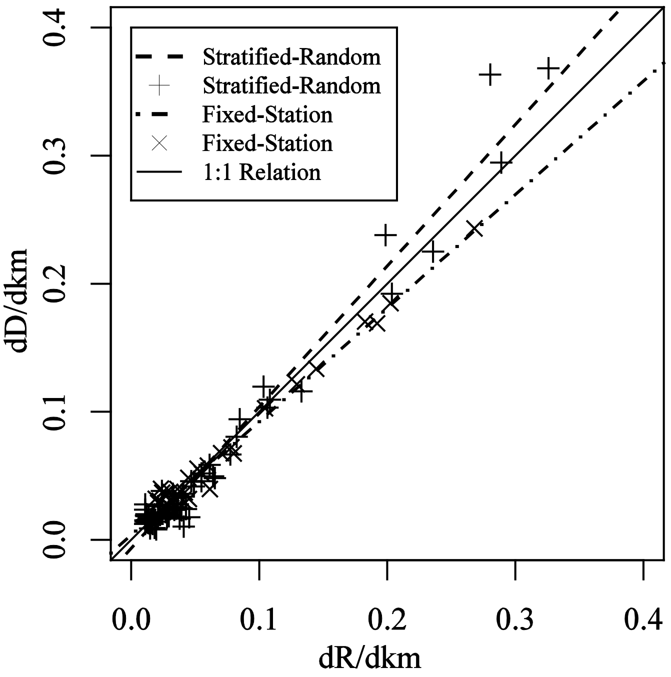

Figure 1.3. Linear relationships comparing the slopes of species-accumulation curves to those from dissimilarity curves. Slopes for species-accumulation curves are shown on the x-axis (dR/dkm), whereas slopes for dissimilarity curves are shown on the y-axis (dD/dkm).

Estimates of total species richness, abundance, and mass

Observed species richness (S_{obs}) for each sampling design within each sampling event ranged from 15 to 27 species, whereas S_{est} ranged from 22.0 to 31.8 species (Figure 1.4). Estimated abundance for the most numerous species such as common shiner and fallfish is consistently higher for the stratified-random design (Table 1.2), although uncommon (0.1 < n/km < 2) or rare (n/km < 0.1) species estimates were comparable between the sampling designs. Estimated mass was considerably higher for the fixed-station design for white sucker and Atlantic salmon (Table 1.2). Total estimated abundance for all species combined was higher for the stratified-random design, especially during fall 2010 sampling (Figure 1.5), whereas total estimated mass was similar between designs for all sampling events.

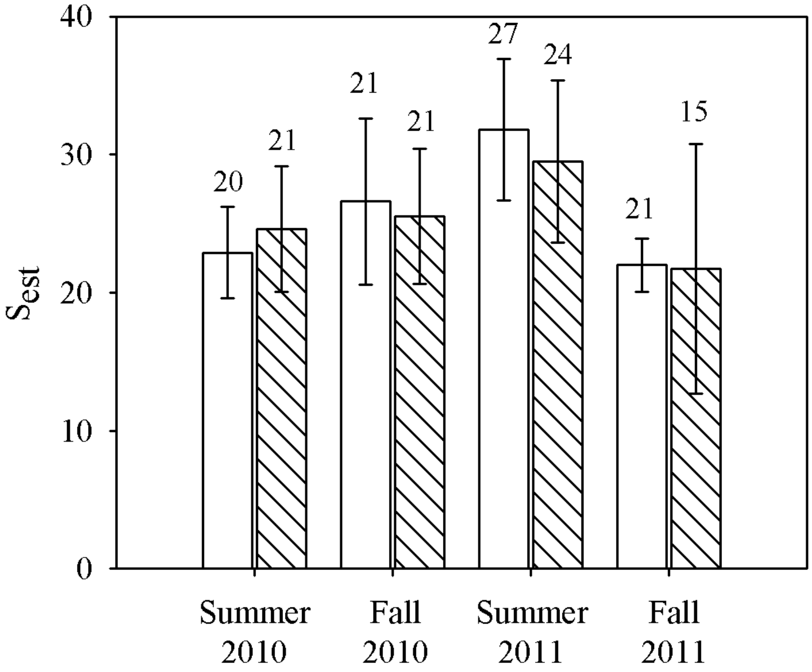

Figure 1.4. Total estimated species richness. Data were derived from boat electrofishing while using stratified-random and fixed-station sampling designs on the Penobscot River. Estimated richness (Sest) was calculated using the first order jackknife; estimates are for all four sampling seasons during 2010 and 2011. The number of species captured is shown above each bar; error bars represent 2SE.

Table 1.2. Mean summer abundance and mass estimates for fish in the Penobscot River, 2010-2011.

	Abundance (N/km)		Mass (kg/km)	
Species	Stratified-Random	Fixed-Station	Stratified-Random	Fixed-Station
Common Shiner	192.9	65.5	378	217
Fallfish	189.0	64.8	812	485
Redbreast Sunfish	38.0	12.8	1,424	744
White Sucker	35.2	20.7	1,314	3,822
Smallmouth Bass	27.2	28.5	2,577	2,405
Pumpkinseed	19.3	19.9	75	76
Golden Shiner	18.1	9.2	24	17
American Eel	14.0	9.2	1,758	1,185
Chain Pickerel	9.4	3.0	574	353
Sea Lamprey	8.5	2.3	125	21
Yellow Perch	3.7	4.8	114	78
Brown Bullhead	3.5	2.6	724	812
Banded Killifish	3.0	0.9	7	1
Alewife	2.1	0.8	298	43
Burbot	1.6	0.6	114	41
Blueback Herring	1.6	4.0	38	75
Mummichog	0.3	< 0.1	< 1	< 1
Creek Chub	0.3	0	< 1	0
Eastern Silvery Minnow	0.2	0	< 1	0
Black Crappie	0.2	0.3	1	2
White Perch	0.2	< 0.1	2	< 1
Atlantic Salmon	< 0.1	0.9	2	1,810
Brook Trout	< 0.1	0.0	< 1	0
Blacknose Dace	< 0.1	0	< 1	0
Blacknose Shiner	< 0.1	< 0.1	< 1	< 1
Finescale Dace	< 0.1	0	< 1	0
Northern Redbelly Dace	< 0.1	< 0.1	< 1	< 1
American Shad	< 0.1	< 0.1	30	14
Threespine Stickleback	< 0.1	0	< 1	< 1
Largemouth Bass	0	< 0.1	0	< 1
Longnose Sucker	0	< 0.1	0	45
Ninespine Stickleback	0	< 0.1	0	< 1
Sturgeon spp.	0	0	0	0
Striped Bass	0	< 0.1	0	55

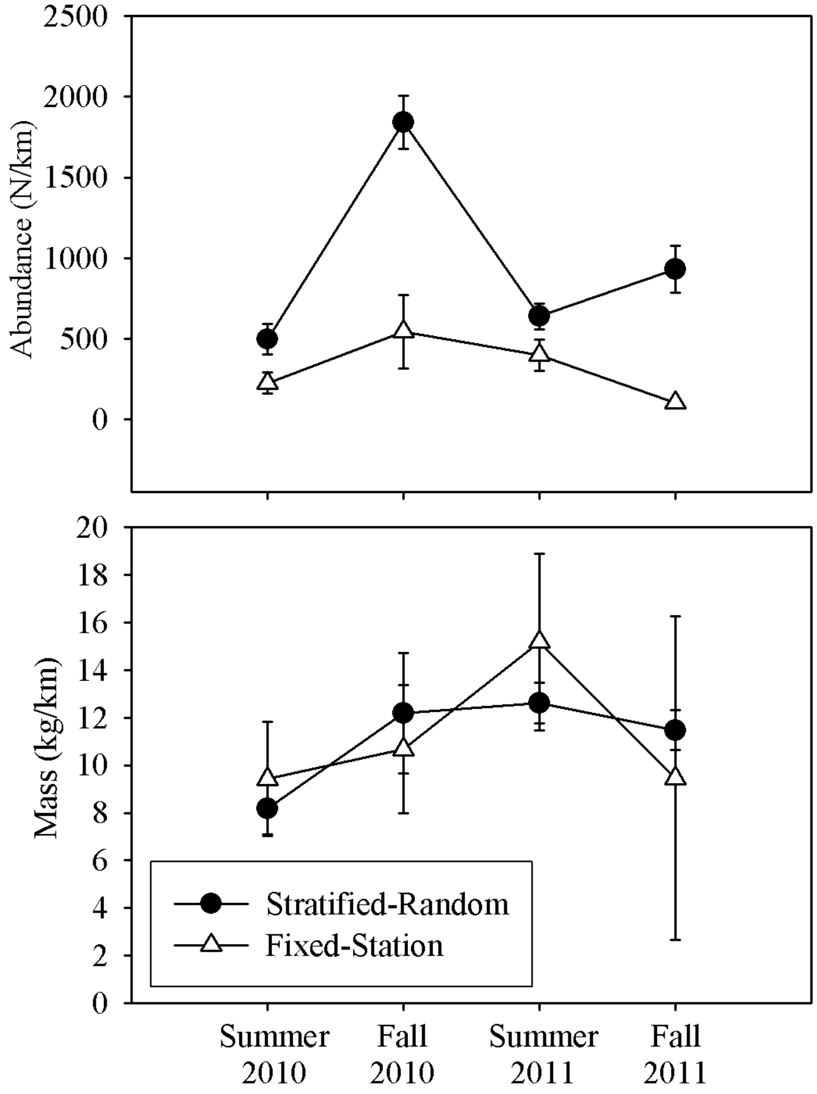

Figure 1.5. Estimates of total abundance and mass of fish in the Penobscot River, 2010-2011 for stratified-random and fixed-station sampling designs. Error bars represent 1SE.

Proportional abundance

The MRPP analysis indicated that there was no significant difference ($\alpha = 0.05$) in CPUE between our sampling designs during summer 2010, fall 2010, and summer 2011 seasons (Table 1.3). A significant difference ($p = 0.04$) between sampling designs was present during the fall 2011 sampling, which was the only sampling event where our species-accumulation and dissimilarity curves indicated that effort for the fixed-station design did not reach the minimum effort required for repeatable results. No significant differences were present for MPUE (Table 1.3). Indicator Species Analysis for fall 2011 sampling revealed three significant ($\alpha = 0.05$) indicator species during the fall 2011 sampling: fallfish (IV = 97; $p = 0.05$), common shiner (IV = 94; $p = 0.023$), and smallmouth bass (IV = 81; $p = 0.035$). These indicator values are relatively high and are associated with the stratified-random sampling; fallfish and smallmouth bass were present in all strata within both sampling designs, but the proportional CPUE of those species was much higher within the stratified-random design. Additionally, common shiner were present in all of the strata within the stratified-random design but were captured within only one stratum of the fixed-station design; the proportional CPUE of common shiner was also much higher within the stratified-random design.

Table 1.3. Pair-wise comparisons of fish assemblage CPUE and MPUE. Data were derived from boat electrofishing while using stratified-random and fixed-station sampling designs on the Penobscot River, 2010-2011. An asterisk denotes a significant result (MRPP; $P \leq 0.05$). The change corrected within-group agreement (A) from MRPP analyses is also listed.

	CPUE		MPUE	
Sampling Event	A	p	A	p
Summer 2010	-0.019	0.59	0.025	0.15
Fall 2010	-0.001	0.42	0.004	0.38
Summer 2011	-0.034	0.78	-0.003	0.47
Fall 2011	0.143	0.04*	-0.052	0.81

Discussion

Sampling effort

Regardless of sampling event, all curves begin to approach an asymptote at ~5 kilometers of electrofishing. We feel that this is the minimum level of effort in order to produce repeatable estimates of fish assemblage structure in the Penobscot River. Results from the fixed-station sampling from fall 2011 when only 3 kilometers of shoreline were sampled illustrate how sampling below this minimum can result in lower precision. Increased effort beyond 5 kilometers per sampling event did not increase precision considerably, but this does not imply that increasing sampling effort beyond this minimum is not useful. Additional project objectives such as the capture of rare species would necessitate sampling effort far past the minimum required for repeatable whole-assemblage estimates. The high degree of overlap of confidence intervals between species-accumulation curves indicates that both sampling designs are similar in the accumulation of species with increased effort; one design does not have a clear advantage over the other in terms of efficiency. Our sampling designs produced slightly different shaped curves (Figure 1.2), possibly due to differences in species distributions that were sampled by each design (Angermeier and Smogor 1995; Kanno et al. 2009), which could result from sampling design effects. The sampling designs differ in transect length and method of site selection, these differences alone could produce curves with a different shape (Scheiner 2003; Chapman and Underwood 2009).

The stratified-random design was also similar to the fixed-station design when abundance is incorporated. Additionally, dissimilarity curves are steeper initially and level out more completely than the species-accumulation curve for each sampling event under the stratified-random design (Figure 1.3). This indicates that less sampling effort is necessary to characterize proportional abundance than is necessary to determine species richness, similar to findings by Angermeier and Smogor (1995). The opposite pattern is shown for the fixed-station design, with dissimilarity curves that exhibit shallower slopes at low sampling effort than species accumulation curves, indicating less efficient sampling of proportional abundance relative to species richness. Dissimilarity curves for the fixed-station design also exhibit steeper slopes than associated species-accumulation curves at higher levels of sampling, which could indicate insufficient or ineffective sampling of the assemblage as a whole.

Estimated species richness, abundance, and mass

The study area, the main-stem Penobscot River, is an open system with fish emigration and immigration possible from coastal systems, tributaries, and areas of the river that are inaccessible or too deep to survey with our sampling methods. Fish assemblage structure could vary seasonally and annually according to life-history and habitat requirements of each species. Our estimates of total species richness (S_{est}) for each sampling event are relatively similar between sampling designs (Figure 1.4). The fall 2011 sampling under the fixed-station design was the only sampling event where the species-accumulation curve did not begin to show asymptotic behavior; the standard error for S_{est} derived from fixed-station sampling was very large, although the estimated richness value was similar to the estimate from the stratified-random design (Figure 1.4). The large standard error values overlap the estimate from the summer 2011 data considerably, which did not occur under the stratified-random design. If more transects were sampled within the fixed-station design during fall 2011 and the species-accumulation curve exhibited asymptotic behavior, precision of S_{est} would have been greater. Results might have been more similar to the stratified-random design in terms of identifying seasonal variability.

The abundance estimate for each species derived from stratified-random transects is unbiased, therefore, differences between the stratified-random and fixed-station estimates show the potential for bias within the fixed-transect design. We observed (Table 1.1) and estimated (Table 1.2) large numbers of common shiner and fallfish within the stratified-random design relative to the fixed-transect design. Because no data were collected above Great Works Dam during fall 2011 for the fixed-station design, bias during the fall may have been caused by a lack of sampling rather than an inherent bias within the design. However, summer abundance estimates should have been similar between sampling designs, since the minimum sampling effort for precise estimates was reached for both designs during all summer sampling events, yet bias is apparent within the fixed-station design during the summer (Table 1.2). The fixed-station design appears to be biased low for estimating the abundance of minnow species such as fallfish and common shiner, but biased high for estimating the mass of white sucker and Atlantic salmon in the Penobscot River. Additionally, total estimated abundance for the stratified-random design is consistently higher than the fixed-station design, especially during fall sampling when we captured many age 0 fish. This bias within the fixed-station design is likely due to transect location; because four of the eleven fixed-station transects are in close proximity to the base of dams where adult migratory fish are more likely to be captured while habitat for small or juvenile fish may be underrepresented. Alternatively, the stratified-random design better accounts for habitat heterogeneity and is not biased in terms of the location of transects.

Our MRPP results indicate no statistically significant differences in CPUE between the stratified-random and the fixed-station designs for all sampling events, except for the fall 2011 sampling (Table 1.3). The Indicator Species Analysis describes which species may have differed between sampling designs. The three transects that were electrofished during the fall 2011 under the fixed-station design were limited to areas downstream of Great Works Dam (Figure 1.1). The species with the highest indicator values were typically encountered more frequently and in greater abundance within areas upstream of Great Works Dam. Our significant MRPP result for fall 2011 is likely due to a downriver bias that was present for the fixed-station design during that sampling event, which resulted from low sampling effort located in only part of the river system rather than bias inherent to the sampling design. If fixed-station transects above Great Works Dam had been electrofished during fall 2011, it may not have produced a significant MRPP result when compared to the stratified-random design.

Conclusion

Fixed-station sampling designs have been recommended over randomized sampling methods due to potentially higher power to detect changes in catch per unit effort (Quist et al. 2006) and also as a logistical alternative (King et al. 1981). In our study, neither sampling design was noticeably advantageous over the other for encountering species, but the stratified-random design provided abundance estimates with less potential for bias and was more efficient at characterizing proportional abundance.

Comparisons of species richness and proportional abundance estimates indicate that our sampling designs yield similar results, suggesting that the particular choice of fixed-station transects did not lead to appreciable bias for whole-assemblage assessment. We would note, however, that this does not imply that any choice of fixed transects would provide unbiased estimates for the entire reach sampled. Moreover, anticipated changes to fish assemblage structure following dam removal on the Penobscot River could potentially alter how representative fixed sites are of the whole study reach. It would be unfortunate for any fish assemblage monitoring program if changes to fish assemblages occurred but were not documented due to bias of a sampling design through time, even if that sampling design is not biased within a given sampling event. As such we feel that the stratified-random sampling design is preferable for quantifying fish assemblages in the Penobscot River and detecting changes anticipated to occur after dam removal.

CHAPTER 2

AN ASSESSMENT OF FISH ASSEMBLAGE STRUCTURE IN THE PENOBSCOT RIVER AND MAJOR TRIBUTARIES PRIOR TO DAM REMOVAL

Introduction

Dams affect the distribution and abundance of fishes through fragmentation and alteration of habitat. They fragment habitat by impeding movements of fishes within a river system (Gehrke et al. 2002; Burroughs et al. 2010), potentially restricting access to spawning, rearing, feeding, or refuge habitat. One of the most publicized effects is restricted passage of spawning anadromous fishes (Beasley and Hightower 2000; Maret and Mebane 2005; Sprankle 2005), which affects not only the distribution and abundance of those species, but food web dynamics and nutrient cycling within freshwater ecosystems (Saunders et al. 2006; MacAvoy et al. 2009). Resident fish movements are impeded by dams as well, resulting in changes to assemblage structure through isolation of populations and restriction of access to habitats essential to fish at different life stages (Porto et al. 1999; Lienesch et al. 2000; Burroughs et al. 2010). Dams convert lotic habitat to lentic habitat (Kanehl et al. 1997; Santucci et al. 2005), which favors generalist and piscivorous species (Guenther and Spacie 2006) and could result in the invasion of riverine areas from impoundments by these species (Erman 1973; Martinez et al. 1994). They also alter flow and thermal regimes, along with water chemistry, further altering fish assemblages (Bain et al. 1988; Lessard and Hayes 2003; Quinn and Kwak 2003).

Historically, 10 species of anadromous fishes were native to the Penobscot River and could access hundreds of kilometers of river, stream, and lake habitat for spawning and/or juvenile rearing (Saunders et al. 2006). The construction of more than 100 dams on the river and tributaries has limited the distribution of many anadromous species to lower portions of the river; these species have subsequently declined in abundance and some are nearly extirpated (Saunders et al. 2006). Resident fish assemblages above and below the dams likely have changed as well, as suggested by results of several other studies (Quinn and Kwak 2003; Guenther and Spacie 2006; Catalano et al. 2007), although overall changes to fish assemblage structure due to dams on the Penobscot River are unknown. The Penobscot River Restoration Project (PRRP) is anticipated to increase passage of anadromous and resident fish and improve connectivity among currently fragmented habitats within the Penobscot River watershed through the removal of the two farthest downstream dams coupled with the installation of a fish lift at the third main-stem dam and a fish bypass around a dam on a major tributary (Opperman et al. 2011; PRRT 2011).

Despite widespread damming of rivers (Dynesius and Nilsso 1994), the effects of dams and dam removal projects on large rivers are understudied, with most research focused on smaller rivers and streams or upper-watershed areas of large rivers (e.g. Connolly and Brenkman 2008; Burroughs et al. 2010; Gardner et al. 2011). Additionally, evaluations of the effects of dam removal for recent projects on large rivers have relied on anecdotal evidence rather than scientific assessment because often, few data are available before and/or after dam removal (Babbitt 2002). The PRRP provides a valuable opportunity to study the effects of dams, and eventually dam removal, on fish assemblage structure within a large river. Our goal was to characterize fish assemblage structure in the Penobscot River and major tributaries prior to dam removal by focusing on the distribution and abundance of fishes, along with variability of fish assemblage patterns among years and seasons.

Study Area

The Penobscot River watershed is the largest in Maine, and the second largest in the New England, draining 2.2 million primarily forested hectares through more than 8,800 kilometers of river and streams (Opperman et al. 2011). Our study focused on the lower 70 km of river (Figure 2.1), which ranges from 170 to 600 meters wide with an average annual discharge of ~440 cubic meters per second during recent years (USGS 2012a). This river reach contained approximately 257 kilometers of shoreline, and included freshwater tidal, impounded, and free-flowing areas. Excluding relatively small impoundments, most areas were heterogeneous in shoreline habitat and flow types. The river was impounded at the head of tide by the Veazie Dam (Figure 2.1); two other main-stem dams (Great Works and Milford) were also included in the main-stem study area, which was bounded on the upstream end by the West Enfield Dam and Howland Dam. Major tributaries were also sampled, some of which drain into the Penobscot River within the main-stem study area; others drain into the river farther upstream in the watershed (Figure 2.1).

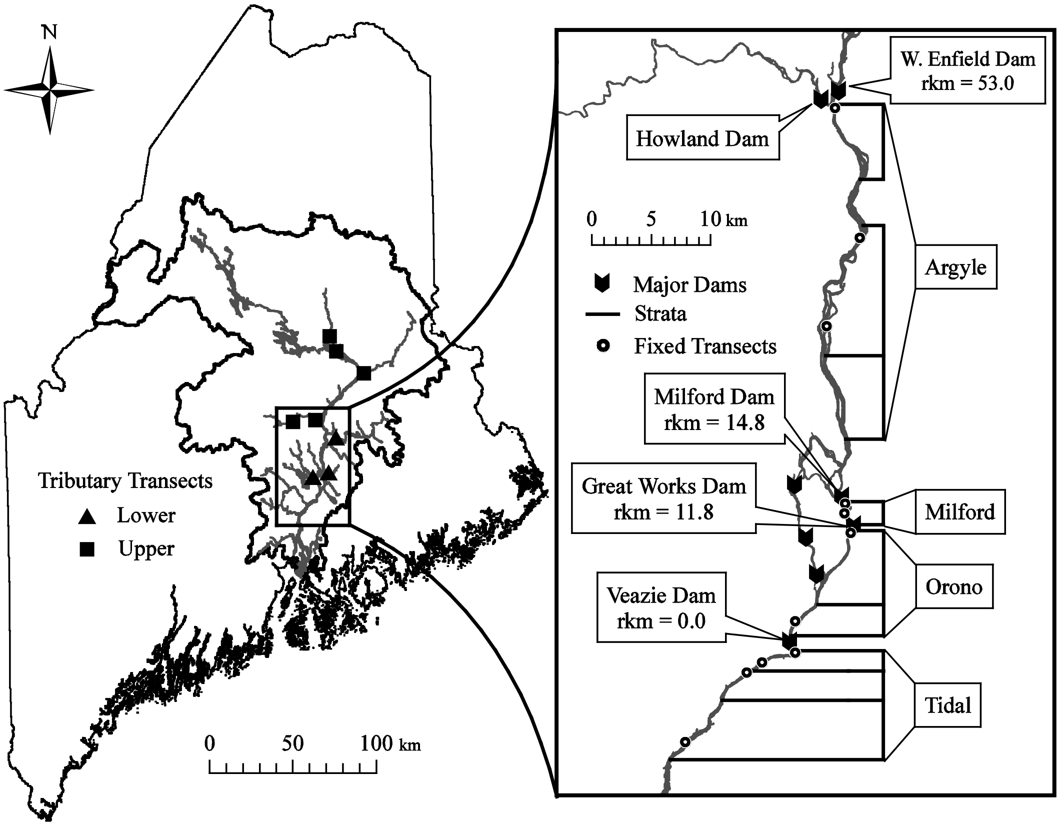

Figure 2.1. Study area. The Penobscot watershed, including the locations of major dams, main-stem strata boundaries and river sections, along with tributary transects.

Methods

Sampling Designs

Fixed-station sampling design. The fixed-station design had been implemented as part of an earlier study for two years prior to our data collection (Kleinschmidt Associates 2009a; 2009b); we sampled along transects that were chosen and sampled previously by Kleinschmidt Associates (2009b). The fixed-station sampling design included 11 transects on the main-stem river and eight transects along major tributaries (Figure 2.1), all of which were approximately 1,000 meters in length. Six of the main-stem transects were concentrated in areas above and below dams scheduled for removal (Kleinschmidt Associates 2009b). During the summer 2011 sampling, we divided each fixed transect in half when feasible, to yield data comparable to that collected with the stratified-random design (described below).

Stratified-random sampling design. The stratified-random sampling design was implemented to improve spatial coverage and account for heterogeneity within the main-stem river, and was not conducted on any tributaries. We divided the river longitudinally into nine strata (Figure 2.1), the bounds of which were based on dam locations, broad-scale habitat types, and boat access. Using ArcGIS 9.3 (Redlands, California), we delineated the river shoreline, including shoreline around large islands, into 219 transects approximately 500 meters in length. We selected multiple transects at random from within each stratum; a prioritized list was created to select alternate transects if that area of river was inaccessible by boat.

River Sections

To describe fish assemblage structure in relation to dam locations, we divided the river into four sections: Tidal, Orono, Milford, and Argyle (Figure 2.1). All river sections were bounded on both ends by dams except for the Tidal section which was only bounded on the upriver end, and there were no dams obstructing the main-stem river within any section. Each river section contained between one and three strata (Figure 2.1). The Tidal river section contained all three tidal strata, the Orono section contained one impounded and one free-flowing strata, the Milford section contained one impounded strata, and the Argyle section contained three free-flowing strata. Tributaries were grouped according to whether they drainage locations and the presence of additional dams between our main-stem strata and tributary transects. Tributary transects were considered “Lower” if the tributary drained directly into our main-stem strata with no dams between the transect and our strata. All Lower tributary transects were located three dams above the head of tide; these included transects on Pushaw Stream, Sunkhaze Stream, and the Passadumkeag River. Tributary transects were considered “Upper” if the tributary drained into the main-stem river were higher in the watershed, or if dams were present between a tributary transect and the main-stem strata. All Upper tributary transects were located four or five dams above the head of tide; these included transects on the Piscataquis River, Mattawamkeag River, and East Branch Penobscot River.

Fish Collection

Prior to sampling, we measured water temperature and specific conductivity, and recorded GPS coordinates at the start and end of each transect, along with seconds of electrofishing after sampling was complete. Single-pass daytime boat electrofishing surveys (Curry et al. 2009) were conducted in the summer (June) and the fall (September-October) during 2010 and 2011, for a total of four discrete sampling events. We electrofished on the Penobscot River only if discharge was less than 425 cubic meters per second at West Enfield, ME (USGS Gage 01034500) and when water temperatures were below 22°C as measured at the start of each transect.

On the Penobscot River and the largest tributaries, we used a 17.5-foot (5.5-m) Lowe (Lebanon, Missouri) Roughneck aluminum boat equipped with Smith Root (Vancouver, Washington) electrofishing equipment, including two booms with 6-dropper anode arrays, and a GPP 5.0 electrofishing system. On smaller tributaries, we used a 14-foot (4.3-m) Sea Eagle (Port Jefferson, New York) inflatable raft equipped with a Smith Root (Vancouver, Washington) GPP 2.5 electrofishing system and a custom anode array similar to that used by Maine Department of Inland Fisheries and Wildlife biologists (J. Dembeck, MDIFW, pers, comm.). On both vessels, we installed custom cathode dropper arrays near and along the bow of the boat. Metal conduit encased half of the droppers in order to increase the cathode surface area (~30,755cm^2) in order to homogenize the electric field and reduce fish injury and mortality. The electrofishing units were operated using pulsed DC at 60 Hz and 30-40 percent of power, as required to capture fish successfully while limiting injury; settings were chosen to maximize power transfer (Reynolds 1999). Two netters captured shocked fish with Duraframe (Viola, Wisconsin) dip nets of multiple designs; all net bags were constructed of 4.8 mm mesh. Surveys were conducted by maneuvering the boat parallel and close to shore and fishing in a downstream direction, at a speed equal to or slightly greater than the current. Pockets, eddies, and shoreline were sampled thoroughly by maneuvering the boat perpendicular or at an angle to shore. Habitat structure (e.g., boulders, large woody debris, and vegetation) were fished thoroughly as well.

All fish that were captured were identified to species and measured to the nearest millimeter and tenth of a gram. If age 0 or small fish (length < 80mm) of any species were captured in high abundance (n > 50), these fish were separated by size class, counted, and mass was measured for batches, with length taken to the nearest millimeter for the smallest and the largest specimens in a batch. This method was implemented to collect required data from these specimens while reducing mortality and processing time. Due to endangered species permitting restrictions, we did not attempt to net adult Atlantic salmon *Salmo salar*, Atlantic sturgeon *Acipenser oxyrhynchus*, or shortnose sturgeon *Acipenser brevirostrum*, but rather noted their occurrence visually and considered each encounter as a "capture" for data analysis below. Estimated mass for Atlantic salmon observed in 2010 was calculated by approximating size and year class (Dube et al. 2010) and using historical (Baum 1997) and recent (Bacon et al. 2009) length-mass data. Similar methods were used to estimate mass of Atlantic salmon during 2011, but mass data were available from fish that were captured in the Penobscot River (O. Cox, Maine Department of Marine Resources, unpublished data). Sturgeon mass was estimated using length-frequency and length-mass data provided by G. Zydlewski and M. Altenritter (University of Maine, unpublished data).

Data Analysis

Dataset. Age 0 smallmouth bass *Micropterus dolomieu* (< 30mm) and white sucker *Catostomus commersoni* (< 40mm) were removed from the summer sampling data prior to analyses. The growth of these specimens necessary to recruit to our gear (>

25mm) appeared to be inconsistent among strata for the duration of the summer sampling; by fall, these fish were large enough to be captured reliably within all strata. Previous analyses in Chapter 1 indicated that species richness and proportional abundance results from fixed-station and stratified-random sampling were similar; therefore, we combined data from both sampling designs in further calculations (described below) by considering fixed transects as part of the stratified-random design.

Catch and mass per unit effort. Both CPUE (catch per unit effort) and MPUE (mass per unit effort) were analyzed to explore potential differences in patterns that may exist between the two measurements. Analyses for CPUE pertain to species which are most abundant, often small fish, while those for MPUE pertain to larger fish which are usually less abundant but are also important within aquatic ecosystems. We calculated CPUE and MPUE of each species for each stratum and each tributary classification by dividing the total catch or mass by the total length of shoreline electrofished, as measured between start and end GPS coordinates using orthoimagery in ArcGIS 9.3. The sample mean and variance were also calculated for total CPUE and MPUE within each stratum and tributary classification by averaging sampling seasons (i.e. summer and fall samplings). To identify longitudinal patterns, we plotted total CPUE and MPUE against river kilometer using the midpoint of each stratum for the main-stem river. Because tributary transects varied in relative location within the watershed, we did not attempt to identify longitudinal patterns in CPUE and MPUE among tributaries; thus, tributary data remain categorical (Lower and Upper).

Multivariate ordination. Fish assemblage structure was analyzed using a variety of multivariate methods; all multivariate analyses were performed with PC ORD software (McCune and Grace 2002) after a fourth-root transformation of CPUE and MPUE. Fourth-root transformations reduce the effects of numerically large values, and increase the contribution from rare species, focusing attention on the whole assemblage rather than on species dominating abundance or mass (Clarke 1993; Goodsell and Connell 2002). Non-metric multidimensional scaling (NMS) was performed using PC ORD software (McCune and Mefford 1999; McCune and Grace 2002). Ordinations were based on Bray-Curtis dissimilarity, which is considered to be the most reliable distance measure for NMS ordination of assemblage structure (Clarke 1993). Dimensionality was determined by following the procedure by McCune and Grace (2002): performing the analysis with a random start, a stability criteria of 1.0×10^{-5}, and incorporating 40 runs of real data with 50 runs of randomized data. After a stable solution was found, the ordination was conducted with one run of real data; we determined the number of ordination axes by balancing reduction in stress with ease of interpretation (Clarke 1993; McCune and Grace 2002). We incorporated all sampling events into one NMS ordination each for CPUE and MPUE in order to identify variability in spatial patterns among sampling events; PC ORD provided Kendall's Tau coefficients (T) with which we determined the direction and strength of correlations between each species and both axes.

MRPP and Indicator Species Analysis. We used multi-response permutation procedures (MRPP) based on rank-transformed Bray-Curtis dissimilarity to identify significant ($\alpha = 0.05$) differences of fish assemblages among sampling events and also among river sections within the Penobscot River. When comparing sampling events, we considered all strata from a given sampling event as a group (i.e. all nine strata during summer 2010 vs. all nine strata during summer 2011) regardless of location in relation to dams; whereas when comparing river sections, we considered all strata within a given river section for all sampling events as a group (i.e. Tidal = three tidal strata from each of the four sampling events analyzed together as a group). Tributary data were not included in MRPP analyses because many tributary transects were not sampled during every sampling event due to flow and time constraints. MRPP is a non-parametric method which tests for differences in assemblages among groups; it calculates a p-value and an A-statistic, both of which must be used to assess dissimilarity (McCune and Grace 2002). The p-value is the likelihood that an observed difference is due to chance, while the chance corrected within-group agreement (A), also known as the effect size, describes within-group homogeneity compared to the random expectation (McCune and Grace 2002). When all items within groups are identical, $A = 1$, whereas if heterogeneity equals expectation by chance then $A = 0$; however, $A < 0$ if there is less agreement within groups than expected by chance (McCune and Grace 2002). A rank transformation of the distance matrix was performed so that results were analogous to our NMS ordinations (McCune and Grace 2002). All groups within CPUE and MPUE were analyzed simultaneously, followed by subsequent pair-wise comparisons given a significant result.

Indicator Species Analysis (Dufrene and Legendre 1997) is a method often used in conjunction with MRPP (McCune and Grace 2002) and was performed on significant pair-wise MRPP results. Indicator species are relatively more abundant or frequent within a group when compared to other groups and thus describe differences among groups. Rare species are not typically indicators; however, it is not necessary for an indicator species to be dominant within a group. Indicator Species Analysis provides an Indicator Value (IV = 0-100), along with p-values which were calculated using a Monte Carlo test of significance based on 1000 permutations and were considered significant if $p \leq 0.05$.

Results

Catch and Mass

Over all four sampling events, 61,837 fish of 35 species (Table 2.1) were captured; 45,874 of these fish were captured in the main-stem Penobscot River (88 kilometers of electrofishing), while 15,963 were captured in tributaries (26 kilometers of electrofishing). Sampling within the Penobscot River accounted for 34 species, with slimy sculpin as the only species captured within tributaries (one transect) but not in the main-stem river. The most numerically abundant species captured were fallfish and common shiner, whereas smallmouth bass and white sucker contributed the most to total mass (Table 2.1). Of all species captured, seven were anadromous, making up 1.4% of the total catch by numbers and 4.4% of the total catch by mass. The single catadromous species captured, American eel, accounted for 1.8% of the total catch by numbers but 12.8% of catch by mass. There were also seven introduced species, making up 13.8% of catch by numbers and 36.3% of catch by mass; of these species, smallmouth bass was dominant for catch by both numbers and mass. The majority of species (22) were captured both above and below Veazie Dam. Seven species were captured downriver of Veazie Dam but at no locations upriver: American shad, alewife, blueback herring, striped bass, sturgeon, mummichog and black crappie. Six species were captured upriver of Veazie Dam (tributaries included), but at no locations downriver: burbot, finescale dace, longnose sucker, ninespine stickleback, northern redbelly dace, and slimy sculpin. Finescale dace, ninespine stickleback, and slimy sculpin were captured only above Milford Dam.

CPUE was relatively low within strata downstream of Great Works Dam and was similar between seasons (Figure 2.2). We recorded relatively high CPUE within the three strata above Milford dam; two of these strata exhibited high seasonal variability, with greater CPUE during fall sampling. CPUE was low within the Milford stratum during the summer but relatively high during the fall. Total MPUE increased from downriver to upriver within the tidal strata, but was relatively low within the impounded stratum immediately upriver of Veazie Dam (Figure 2.2); this pattern was evident during both summer and fall samplings. The greatest MPUE was recorded within the free-flowing stratum upriver of the Veazie Dam impoundment, but declined upriver of Great Works Dam another impoundment. MPUE was moderate and similar among strata and season above Milford Dam, including Lower and Upper tributary transects (Figure 2.2).

Table 2.1. Fish captured by boat electrofishing in the Penobscot River and major tributaries, 2010-2011. Origin (N = Native; I = Introduced) and general life history for each species are also shown. An asterisk indicates that MPUE is an estimate rather than a measurement.

Species	Code	n	kg	Freq.	Origin	Life History
Common Shiner *Luxilus cornutus*	CSH	18554	27.3	0.68	N	Resident
Fallfish *Semotilus corporalis*	FF	15717	50.5	0.90	N	Resident
Smallmouth Bass *Micropterus dolomieu*	SMB	6733	303.4	0.96	I	Resident
Golden Shiner *Notemigonus crysoleucas*	GSH	5211	9.8	0.47	N	Resident
Redbreast Sunfish *Lepomis auritus*	RBS	3923	82.5	0.92	N	Resident
White Sucker *Catostomus commersoni*	WS	3465	212.7	0.74	N	Resident
Pumpkinseed *Lepomis gibbosus*	PS	3056	24.3	0.68	N	Resident
American Eel *Anguilla rostrata*	EEL	1140	133.9	0.85	N	Catadromous
Yellow Perch *Perca flavescens*	YP	961	23.3	0.49	I	Resident
Chain Pickerel *Esox niger*	CHP	698	53.7	0.61	I	Resident
Brown Bullhead *Ameiurus nebulosus*	BBH	567	65.1	0.43	N	Resident
Banded Killifish *Fundulus diaphanus*	BKF	517	1.2	0.29	N	Resident
Sea Lamprey *Petromyzon marinus*	LAM	429	5.5	0.44	N	Anadromous
Alewife *Alosa pseudoharengus*	ALE	224	26.6	0.15	N	Anadromous
Blueback Herring *Alosa aestivalis*	HER	192	7.7	0.12	N	Anadromous
Burbot *Lota lota*	CSK	166	11.3	0.24	N	Resident
Eastern Silvery Minnow *Hybognatus regius*	ESM	68	0.1	0.08	I	Resident
Mummichog *Fundulus heteroclitus*	MUM	52	0.1	0.03	N	Resident
White Perch *Morone americana*	WP	40	0.5	0.08	N	Resident
Atlantic Salmon *Salmo salar*	ATS	27	81.5*	0.08	N	Anadromous
Largemouth Bass *Micropterus salmoides*	LMB	19	0.3	0.07	I	Resident
Black Crappie *Pomoxis nigromaculatus*	CRA	18	0.2	0.07	I	Resident
Creek Chub *Semotilus atromaculatus*	CRC	16	<0.1	0.06	N	Resident
Ninespine Stickleback *Pungitius pungitius*	NSS	9	<0.1	0.03	I	Resident
Blacknose Dace *Rhinichthys atratulus*	BND	6	<0.1	0.04	N	Resident
Threespine Stickleback *Gasterosteus*	TSS	6	<0.1	0.03	N	Resident
Slimy Sculpin *Cottus cognatus*	SSC	5	<0.1	<0.01	N	Resident
Blacknose Shiner *Notropis heterolepis*	BNS	4	<0.1	0.02	N	Resident
American Shad *Alosa sapidissima*	SHD	3	2.5	0.02	N	Anadromous
Longnose Sucker *Catostomus catostomus*	LNS	3	1.5	0.02	N	Resident
Brook Trout *Salvelinus fontinalis*	BKT	2	0.1	0.01	N	Resident
Northern Redbelly Dace *Phoxinus eos*	RBD	2	<0.1	0.01	N	Resident
Striped Bass *Morone saxatilis*	STB	2	4.2	<0.01	N	Anadromous
Finescale Dace *Phoxinus neogaeus*	FSD	1	<0.1	<0.01	N	Resident
Sturgeon spp. *Acipenser spp.*	SGN	1	3.4*	<0.01	N	Anadromous

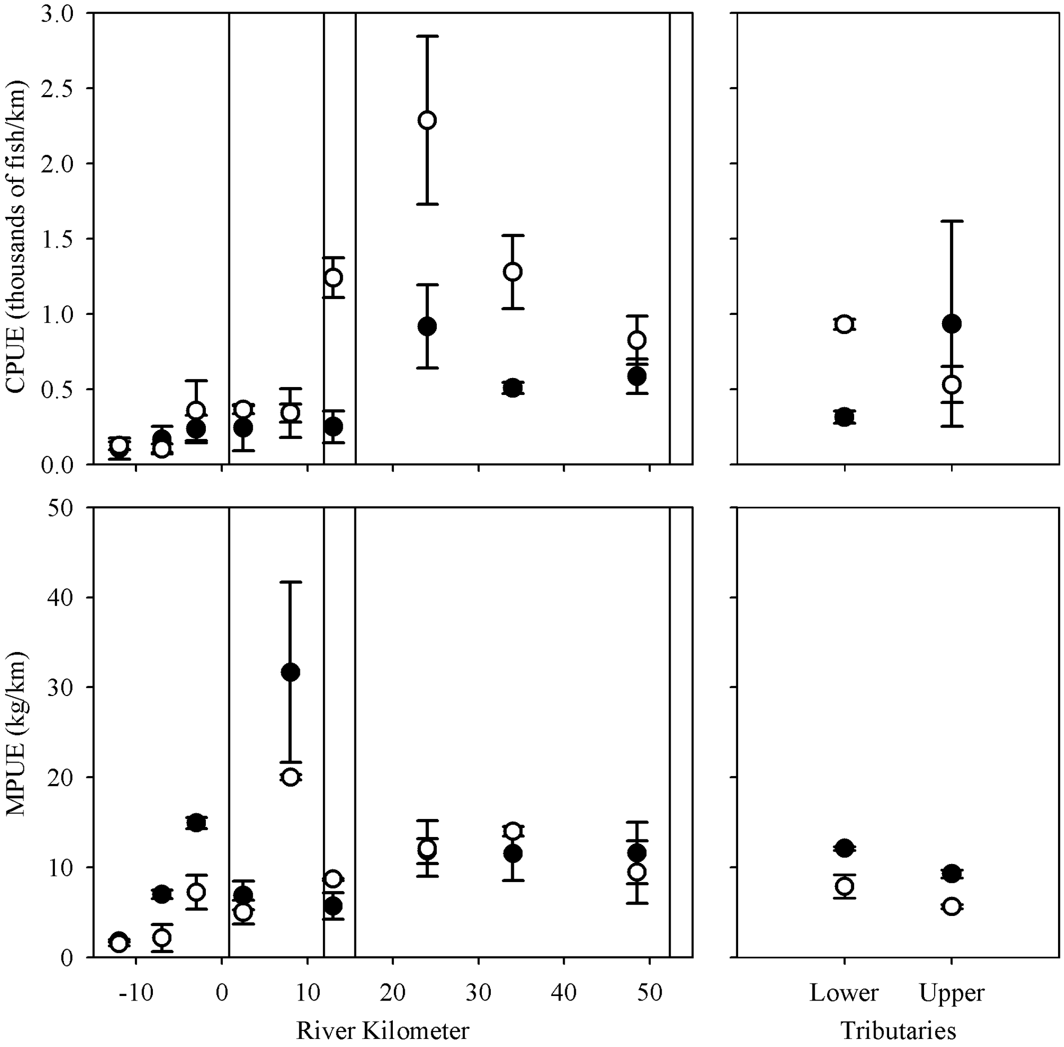

Figure 2.2. Mean CPUE and MPUE of all fish captured by boat electrofishing during 2010 and 2011 on the Penobscot River and major tributaries. Closed circles represent early summer while open circles represent fall sampling. Locations of dams are indicated by vertical bars within each panel; river kilometer zero is located at Veazie Dam and the head of tide. Error bars represent 1SE.

Multivariate Ordination

We obtained stable, two-dimensional NMS ordinations (Figure 2.3) for CPUE (final stress = 15.7) and MPUE (final stress = 16.3) in 46 and 56 iterations respectively. The solution for CPUE explained 87.9% of the variance. Axis 1 accounted for 47.9% of the variance, and was correlated most positively with brown bullhead (T = 0.54), yellow perch (T = 0.48), white sucker (T = 0.46) and chain pickerel (T = 0.44) (Figure 2.4); it was correlated most negatively with alewife (T = -0.40), blueback herring (T = -0.28), and mummichog (T = -0.27). Axis 2 accounted for 40.0% of the variance, and was correlated most positively with burbot (T = 0.45), smallmouth bass (T = 0.30) and longnose sucker (T = 0.23) (Figure 2.4); it was correlated most negatively with golden shiner (T = -0.65), brown bullhead (T = -0.49), pumpkinseed (T = -0.43) and eastern silvery minnow (T = -41).

The solution for MPUE explained 87.8% of the variance. Axis 1 accounted for 54.6% of the variance, and was correlated most positively with brown bullhead (T = 0.73), yellow perch (T = 0.58), golden shiner (T = 0.57) and chain pickerel (T = 0.39) (Figure 2.5); negative correlations with axis 1 were relatively weak, but the most negative species correlations were mummichog (T = -0.19) and alewife (T = -0.17). Axis 2 accounted for 33.2% of the variance, and was correlated most positively with burbot (T = 0.59), white sucker (T = 0.33), common shiner (T = 0.24) and fallfish (T = 0.23) (Figure 2.5); it was correlated most negatively with alewife (T = -0.46), banded killifish (T = -0.42), blueback herring (T = -0.381) and pumpkinseed (T = -0.38).

CPUE and MPUE ordinations show similar patterns, where there was a clear progression in the assemblage structure longitudinally and large differences among the various tributary types. Strata within the Tidal section of the river grouped together but were relatively variable and distant in ordination space from strata within the Argyle river section. Strata within the Orono and Milford sections were grouped relatively tightly and positioned between the Tidal and Argyle river sections in ordination space, which corresponds to their spatial position in the landscape. Lower river tributaries were consistently grouped tightly (Figure 2.3) and were separated from main-stem strata, characteristic of warmwater species (Figures 2.4 and 2.5), with the exception of one lower tributary which grouped with the Argyle river section (Figure 2.3); the only tributary transect which was above four dams but was relatively close to the study area grouped consistently within the Argyle river section, whereas other Upper tributary transects grouped together (Figure 2.3) and were separated from the main-stem strata in the opposite direction from lower tributaries, characteristic of a coolwater assemblage (Figures 2.4 and 2.5).

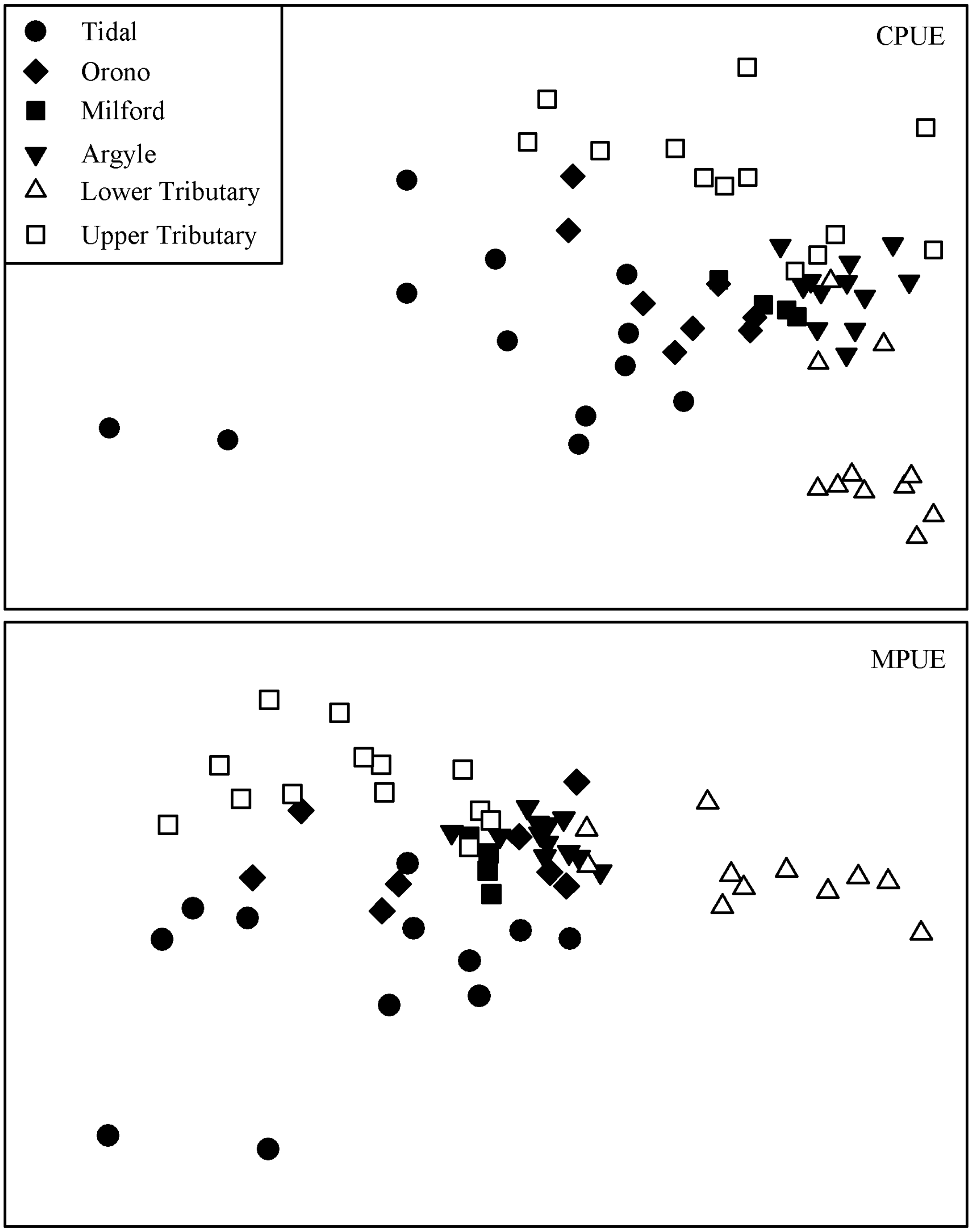

Figure 2.3. NMS ordinations of CPUE and MPUE. Data were from all fish captured by boat electrofishing on the Penobscot River and major tributaries during 2010 and 2011. Ordinations were based on Bray-Curtis dissimilarity. Symbols represent river sections and tributary transects presented in Figure 2.1.

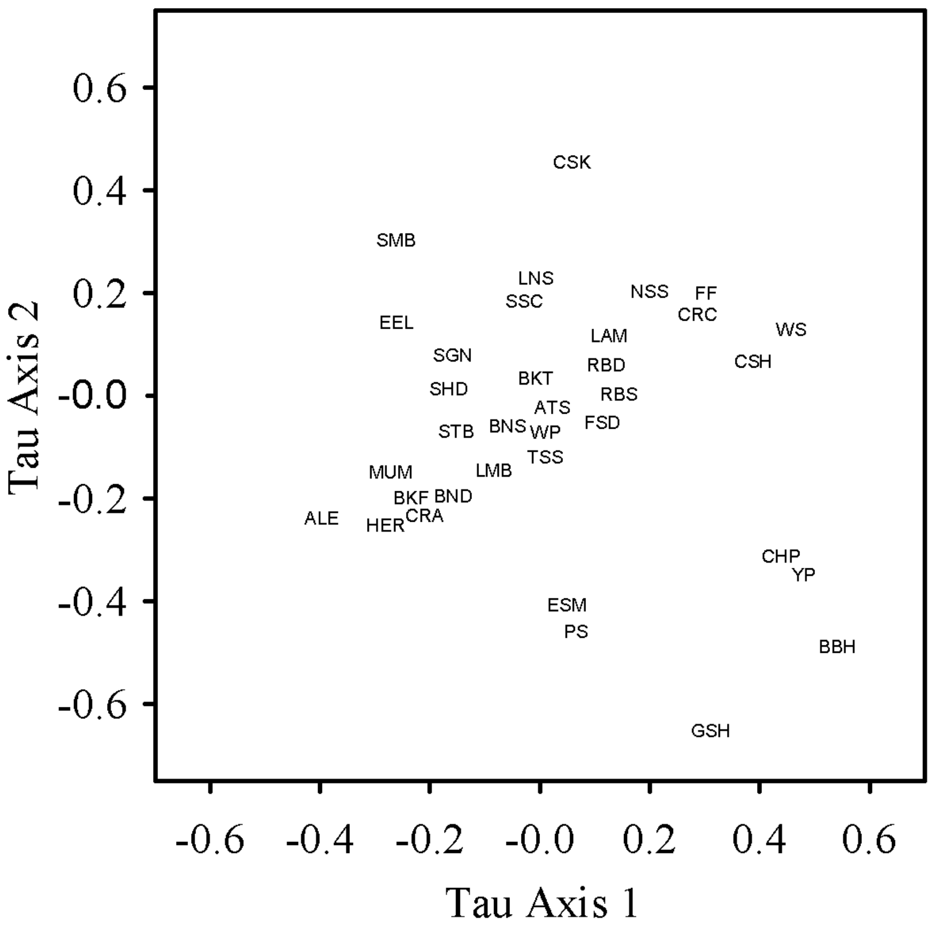

Figure 2.4. Kendall's Tau correlations with NMS ordination axes for CPUE.

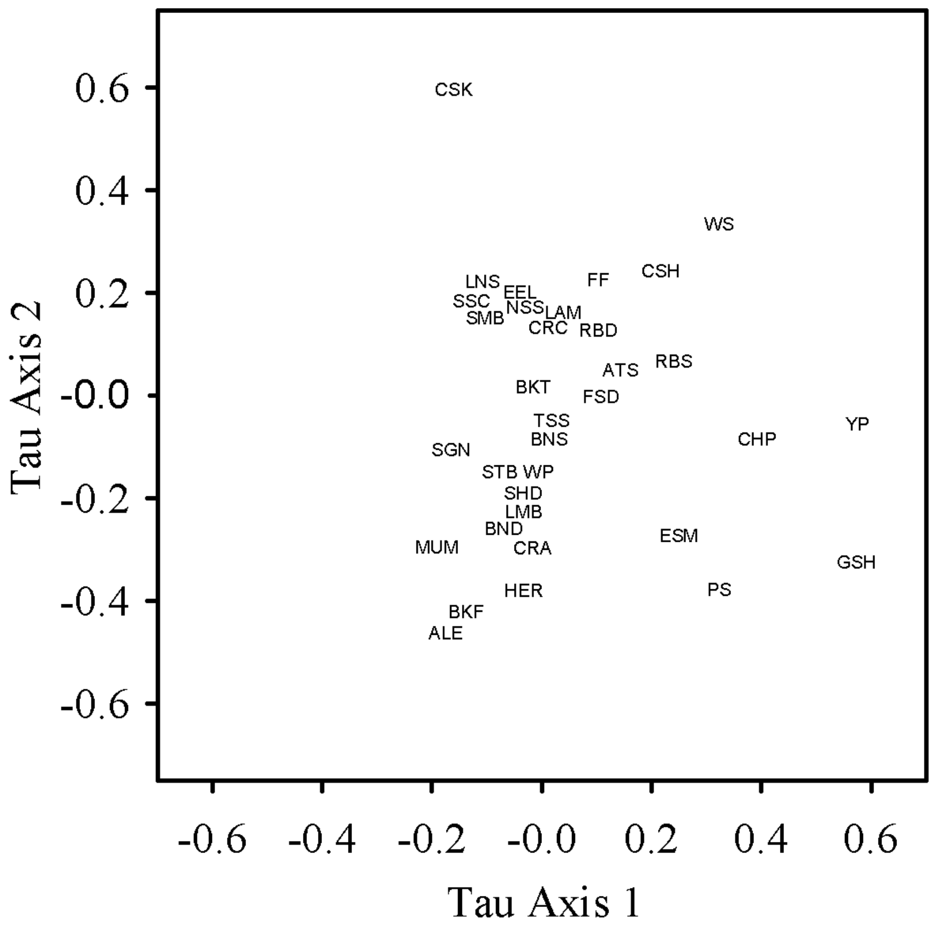

Figure 2.5. Kendall's Tau correlations with NMS ordination axes for MPUE.

MRPP and Indicator Species Analysis

A comparison among all sampling events on Penobscot River yielded significant MRPP results for CPUE (A = 0.064; p = 0.02) whereas analysis of MPUE did not (A = 0.012; p = 0.26) Pair-wise MRPP comparisons of CPUE were significant when summer 2010 sampling was compared with any other sampling event; no other comparisons produced significant differences (Table 2.2). Indicator Species Analysis revealed only one significant indicator of CPUE for the comparison between summer 2010 and fall 2010: smallmouth bass (IV = 61.2; p = 0.008) occurred within all strata during both sampling events, but relative CPUE of this species was much higher during fall 2010. A comparison of summer 2010 and 2011 sampling events yielded four significant indicator species: golden shiner (IV = 71.9, p = 0.003), smallmouth bass (IV = 57.5, p = 0.006), sea lamprey (IV = 68.1, p = 0.011), and pumpkinseed (IV = 64.1, p = 0.011). Smallmouth bass and pumpkinseed were captured within all strata, and indicator values were based on increases in relative CPUE between summer 2010 and 2011. Golden shiner and sea lamprey were captured within six and five strata respectively during summer 2010, but within all nine strata during summer 2011; both of these species also increased in relative abundance between these sampling events. Three significant indicator species described differences between summer 2010 and fall 2011: pumpkinseed (IV = 61; p = 0.001), smallmouth bass (IV = 63; p = 0.001), and white sucker (IV = 62; p = 0.049). Similar to comparisons with other seasons, pumpkinseed and smallmouth bass were captured within all strata during both sampling events but exhibited greater CPUE during fall 2011 relative to summer 2010. White sucker exhibited the opposite pattern, declining in CPUE from 2010 to 2011; relative frequency of white sucker also decreased, having been captured within all strata during summer 2010 and seven out of nine strata during fall 2011.

Table 2.2. MRPP results for pair-wise CPUE comparisons among fish assemblages from all sampling events on the Penobscot River, 2010-2011. Comparison numbers indicate early summer 2010 (1), fall 2010 (2), early summer 2011 (3), and fall 2011 (4).

Comparison	A	p
1 vs. 2	0.058	0.04
1 vs. 3	0.064	0.04
1 vs. 4	0.074	0.03
2 vs. 3	0.045	0.07
2 vs. 4	-0.008	0.51
3 vs. 4	0.006	0.34

Results from MRPP indicated that the fish assemblage differed among all river sections for CPUE (A = 0.303; p = 0.000) and MPUE (A = 0.241; p = 0.000). Results also differed among all pair-wise comparisons of river sections (Table 2.3). Indicator Species Analysis results were similar for CPUE and MPUE (Tables 2.4 and 2.5), although large-bodied piscivorous species such as smallmouth bass, chain pickerel, and American eel were indicators for more river sections when MPUE was analyzed (Table 2.5). Alewife, blueback herring, and banded killifish were consistent indicators of the tidal section of river while pumpkinseed and smallmouth bass were consistent indicators for the Orono river section, especially for MPUE. In comparisons to downriver sections, the strongest indictors within the Milford river section included common shiner, fallfish, and yellow perch; banded killifish were also an indicator species that differentiated this section from other sections immediately upriver or downriver. Many indicator species were present within the Argyle river section, especially in comparison to sections below Great Works Dam (Orono; Tidal) (Tables 2.4 and 2.5).

Table 2.3. MRPP results for pair-wise comparisons among fish assemblages within river sections on the Penobscot River during 2010-2011.

	CPUE		MPUE	
Comparison	A	p	A	p
Tidal vs. Orono	0.140	0.000	0.118	0.001
Tidal vs. Milford	0.147	0.001	0.106	0.009
Tidal vs. Argyle	0.282	0.000	0.228	0.000
Orono vs. Milford	0.107	0.046	0.104	0.042
Orono vs. Argyle	0.256	0.000	0.187	0.000
Milford vs. Argyle	0.092	0.026	0.126	0.004

Table 2.4. Species which described pair-wise assemblage differences in CPUE among river sections on the Penobscot River during 2010 and 2011. River sections listed within the table contain higher, significant ($p \leq 0.05$) indicator values for a species; maximum indicator values are shown in parentheses.

	Comparisons of river sections					
Species	Tidal vs. Orono	Tidal vs. Milford	Tidal vs. Argyle	Orono vs. Milford	Orono vs. Argyle	Milford vs. Argyle
ALE	Tidal(75)	Tidal(75)	Tidal(75)			
BBH			Argyle(75)		Argyle(67)	Argyle(68)
BKF	Tidal(70)		Tidal(66)	Milford(74)		Milford(69)
CHP			Argyle(75)		Argyle(69)	
CSH		Milford(76)	Argyle(77)		Argyle(71)	
CSK		Milford(50)	Argyle(58)		Argyle(52)	
FF	Orono(59)	Milford(65)	Argyle(69)	Milford(57)	Argyle(62)	
HER			Tidal(50)			
LAM			Argyle(64)		Argyle(67)	Argyle(66)
PS				Orono(63)	Orono(61)	
RBS	Orono(60)		Argyle(63)			
SMB	Orono(58)					
WS		Milford(68)	Argyle(73)		Argyle(65)	
YP			Argyle(70)	Milford(75)	Argyle(80)	

Table 2.5. Species which described pair-wise assemblage differences in MPUE among river sections on the Penobscot River during 2010 and 2011. River sections listed within the table contain higher, significant ($p \leq 0.05$) indicator values for a species; maximum indicator values are shown in parentheses.

	Comparisons among river sections					
Species	Tidal vs. Orono	Tidal vs. Milford	Tidal vs. Argyle	Orono vs. Milford	Orono vs. Argyle	Milford vs. Argyle
ALE	Tidal(75)	Tidal(75)	Tidal(75)			
BBH			Argyle(81)		Argyle(67)	Argyle(69)
BKF	Tidal(71)		Tidal(67)			
CHP			Argyle(67)		Argyle(68)	Argyle(62)
CSH		Milford(74)	Argyle(79)		Argyle(71)	
CSK			Argyle(58)			
FF	Orono(60)	Milford(62)	Argyle(67)		Argyle(58)	Argyle(56)
HER	Tidal(50)		Argyle(50)			
LAM						Argyle(65)
PS				Orono(62)	Orono(61)	
RBS			Argyle(60)			
SMB	Orono(60)			Orono(56)	Orono(55)	
WS			Argyle(71)			Argyle(57)
YP			Argyle(75)	Milford(76)	Argyle(81)	
EEL			Argyle(57)			

Discussion

Considerable differences were present among river sections. Most species that we captured within tidal waters but at no locations upriver were anadromous; adults of these species once migrated up the Penobscot River to spawn in great abundances (Saunders et al. 2006), potentially driving ecosystem function through the delivery of marine derived nutrients (Hicks et al. 2005; Walters et al. 2009) and altering assemblage interactions either directly or through their progeny (Hanson and Curry 2005; Kiffney et al. 2009). Alewife and blueback herring were consistent indicators for the Tidal river section; these species were responsible for distinguishing the Tidal section from all other river sections within our Indicator Species Analyses along with separating tidal strata from those above Veazie Dam in our NMS ordination. Atlantic salmon and sea lamprey were the only anadromous species encountered upstream of Veazie Dam. Although both of these species were distributed throughout the study area and within tributaries, passage through dams by these species was likely restricted as well. Atlantic salmon were rarely encountered and most adults were observed in areas below Veazie Dam and Great Works Dam. Sea lamprey were primarily immature individuals and contributed to CPUE to a greater extent than MPUE; the number of adult sea lamprey accessing spawning habitats upriver is unknown. It is possible that our capture methods are ineffective (either in location or timing) for adult sea lamprey, or that relatively few adults are accessing upriver areas but are able to produce moderate numbers of juveniles.

Fish assemblages in the main-stem Penobscot River exhibited distinct longitudinal patterns in structure; tributary fish assemblages were nearly always distinct from the main-stem river. The pattern of increasing total CPUE with distance upriver coincides with greater abundances of common shiner, fallfish, brown bullhead, yellow perch, and chain pickerel. Increasing MPUE up to Veazie Dam, especially during the summer sampling, likely resulted from increased concentrations of anadromous fish such as alewife and blueback herring attempting to pass the dam. We found low MPUE within the two impounded strata, both of which were characterized by low habitat heterogeneity, with the greatest MPUE observed within the free-flowing river section upriver of Veazie Impoundment.

It should be noted that we captured smallmouth bass and chain pickerel frequently; these non-native predators were widely distributed throughout the Penobscot River and were likely influencing fish assemblages through top-down effects (Van den Ende 1993; Weidel et al. 2007). Very high MPUE within the stratum above Veazie Impoundment (Figure 2.2) coincided with smallmouth bass as an indicator for MPUE within the Orono river section (Table 2.5), potentially indicating high-quality habitat for smallmouth bass there. The Orono river section was characterized by two distinct but connected habitats: a relatively small impounded area and a longer, free-flowing reach immediately upriver. Smallmouth bass may have been spawning within the impoundment and moving into free-flowing reaches after rearing (Erman 1973; Penczak et al. 2012). Additionally, adult smallmouth bass may have moved between lotic and lentic habitats, utilizing the impoundment during the winter and the free-flowing section during the summer (Langhurst and Schoenike 1990). The presence of Great Works Dam at the upriver boundary of this river section could also concentrate adult fish and result in high MPUE due to the inability of these fish to distribute throughout upriver reaches. The next section upriver, the Milford river section, was impounded by Great Works Dam but was not connected to a large amount of lotic habitat upriver due to obstruction by Milford Dam, and does not appear to be as productive in terms of MPUE for smallmouth bass. Direct effects of dams on chain pickerel are not evident within the Penobscot River, although less apparent effects on this species could emerge after dam removal.

Overall, we observed similar spatial patterns in fish distribution between years and seasons. We observed some variability of species composition among sampling events for CPUE on the main-stem river, which resulted from fluctuating CPUE within certain strata for relatively few species: smallmouth bass, pumpkinseed, golden shiner, white sucker, and sea lamprey (Figure 2.6). Annual variability in environmental conditions combined with interspecific interactions may account for these fluctuations.

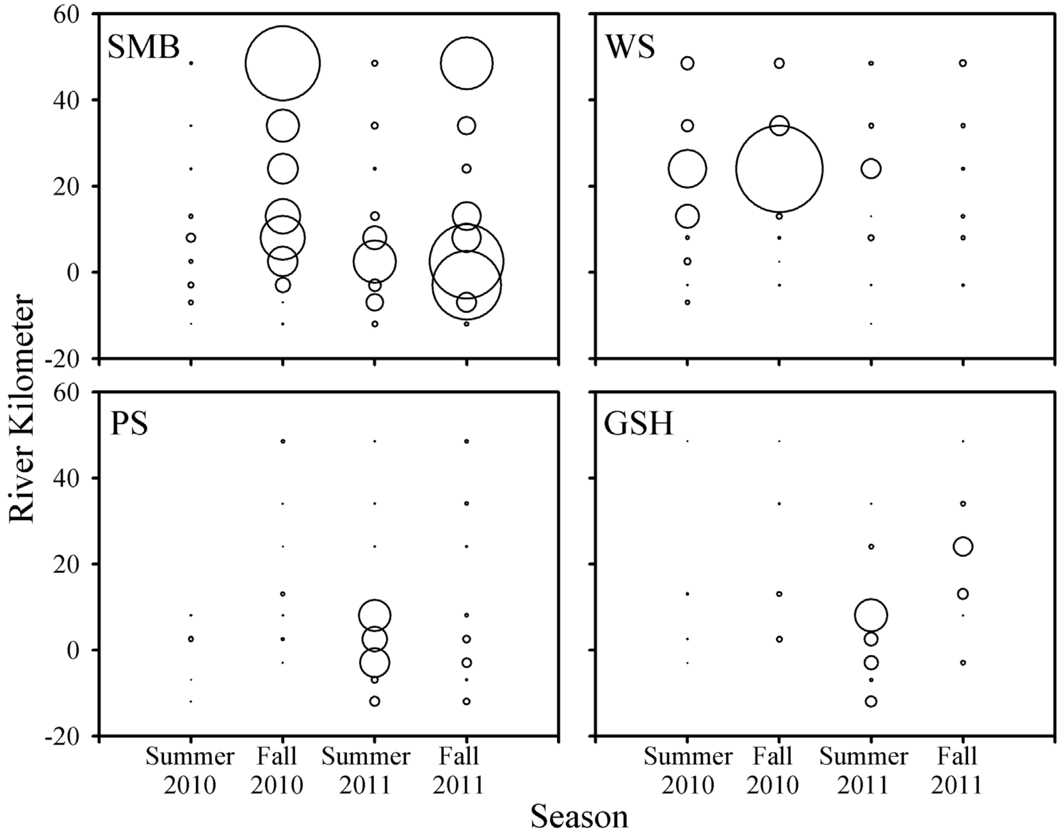

Figure 2.6. Longitudinal patterns of boat electrofishing CPUE for fish species which were indicators for assemblage differences between early summer 2010 and all other sampling events on the Penobscot River.

The one-season lag between the increase in CPUE for smallmouth bass and increases in CPUE for pumpkinseed and golden shiner may be attributed to sampling effects; age 0 smallmouth bass were catchable via boat electrofishing during the fall 2010 sampling whereas the other two species were not large enough to recruit to our gear until the following year's summer sampling event. Alternatively, white sucker declined after fall 2010. The reasons for this are unknown, although environmental conditions during 2011 may have been unfavorable for juvenile survival, or large numbers of smallmouth bass could have reduced the abundance of this species through predation (Weidel et al. 2007). Although sea lamprey CPUE varied between summer samplings, these fish were primarily juveniles that reside within the substrate for multiple years (Beamish 1980) and results from this species may be unreliable due to capture difficulties.

Although it is natural for river systems to exhibit longitudinal gradients in ecosystem structure and function (Naiman et al. 1987), the Penobscot River has been impacted by large, main-stem dams that are impeding passage and fragmenting habitat for a variety of fish species. Our assessment of fish assemblage structure describes the longitudinal patterns and current indicators of seasonal variability within the Penobscot River and major tributaries prior to dam removal; data collected after dam removal can be compared to our findings in order to evaluate success of the PRRP. Dams and associated impoundments on the Penobscot River encompass a relatively small area within the ecosystem, but their effects are considerable and likely reach far upriver and even into marine ecosystems. The effects of removing these dams and improving fish passage in the Penobscot River will not be known until after the PRRP has been completed; however, with improved habitat connectivity (both within freshwater and between marine and freshwater habitats) and reduced lotic-lentic interactions, patterns of fish assemblage structure could change considerably.

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APPENDICES

APPENDIX A. SPECIES-ACCUMULATION CURVES BY STRATUM

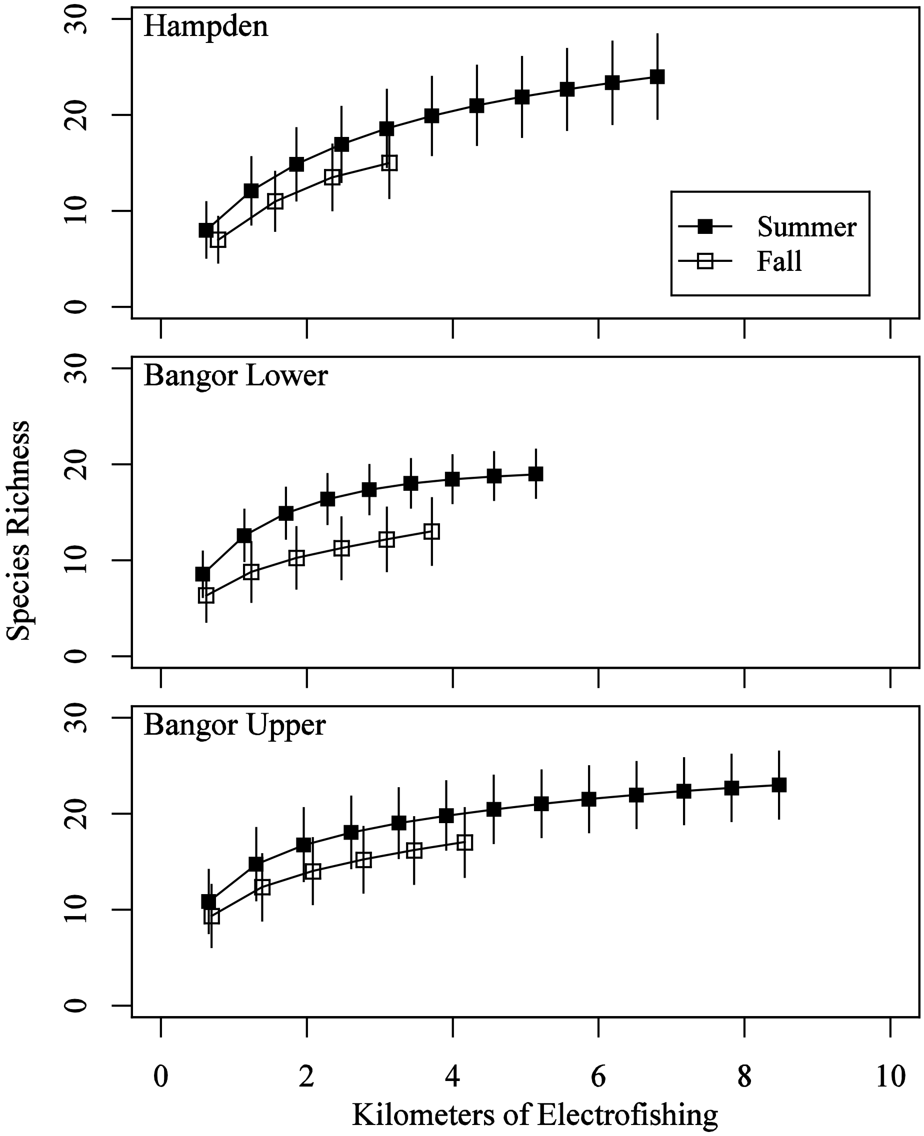

Figure A.1. Species-accumulation curves for summer and fall sampling within each tidal stratum. Curves were constructed using 2010 and 2011 boat electrofishing data on the Penobscot River.

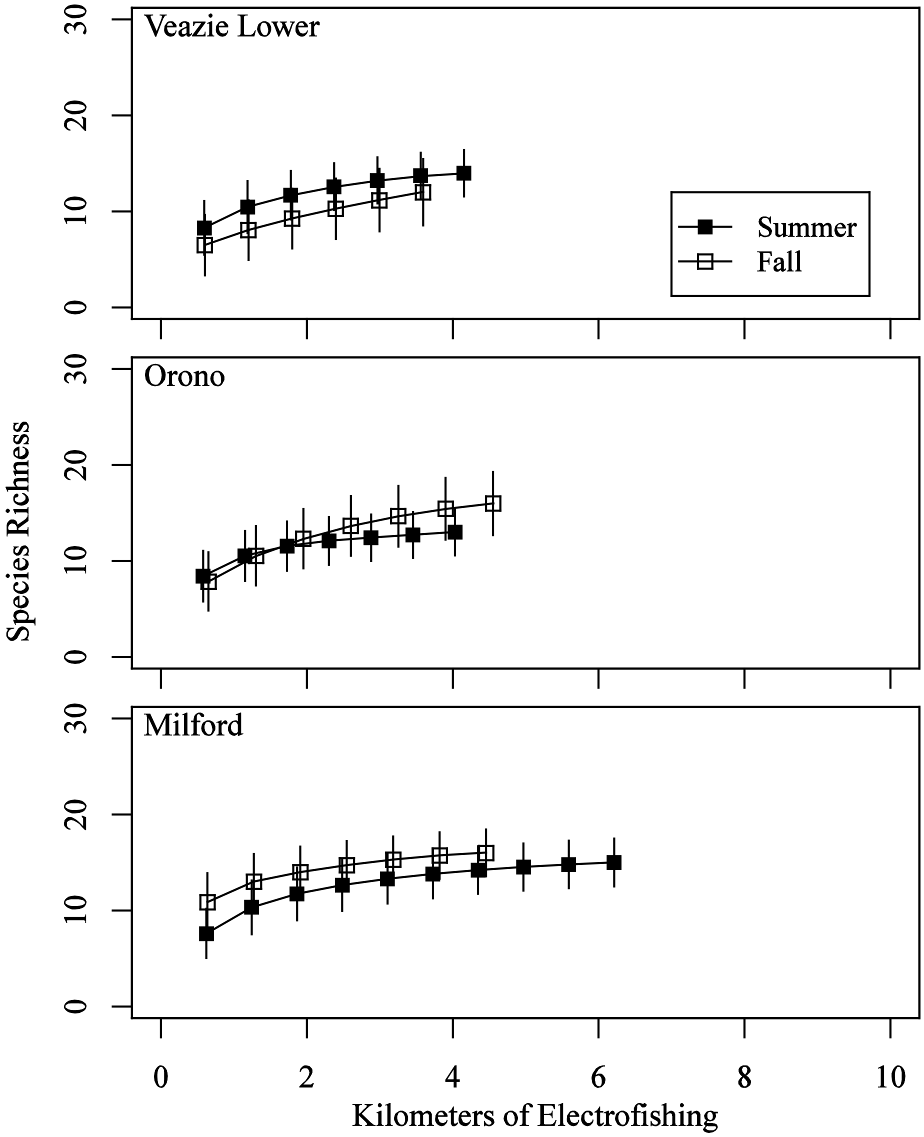

Figure A.2. Species-accumulation curves for summer and fall sampling within each stratum between Veazie Dam and Milford Dam. Curves were constructed using 2010 and 2011 boat electrofishing data on the Penobscot River.

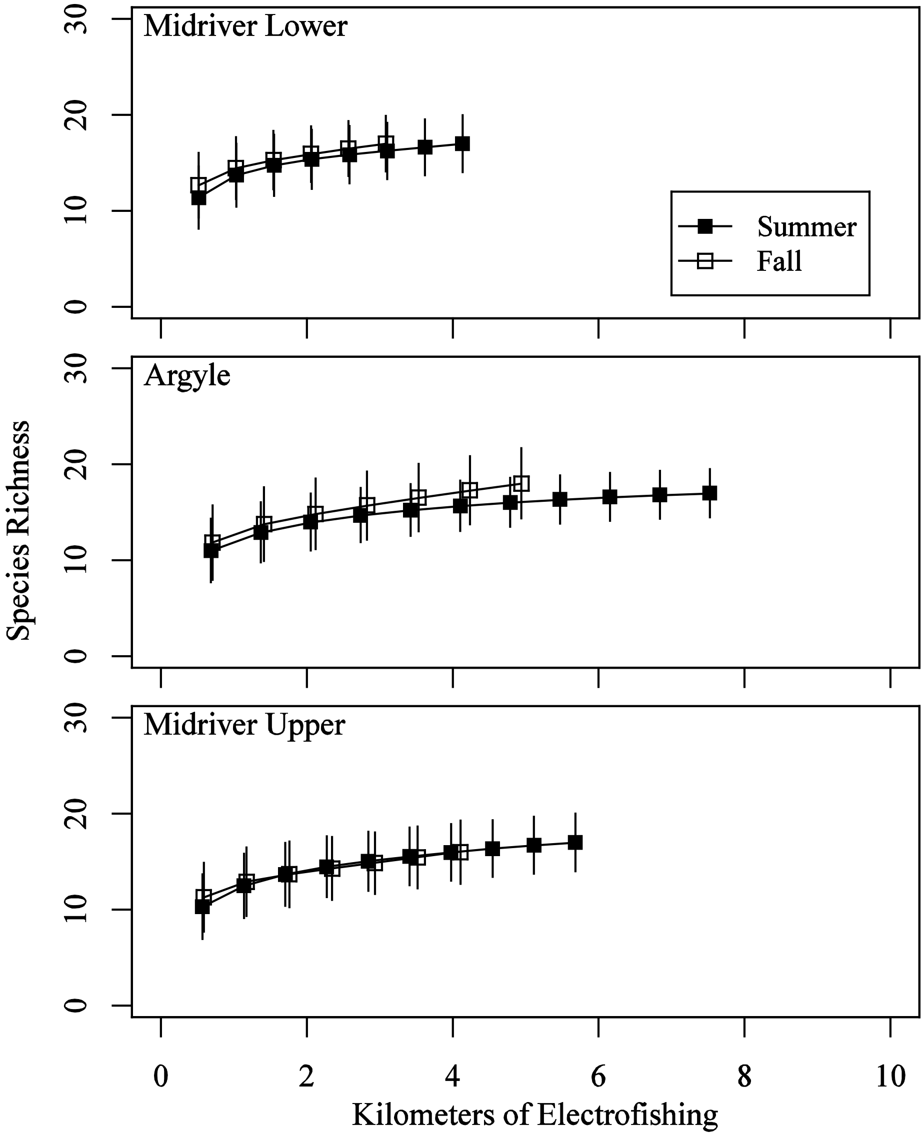

Figure A.3. Species-accumulation curves for summer and fall sampling within each stratum between Milford Dam and West Enfield Dam. Curves were constructed using 2010 and 2011 boat electrofishing data on the Penobscot River.

APPENDIX B. MODELING THE CAPTURE OF SPECIES ON THE PENOBSCOT RIVER

Nine nonlinear models (Table B.1), all of which have been used in other studies of species accumulation curves (Flather 1996; Tjorve 2003; Jimenez-Valverde et al. 2006), were fitted to seven seasonal species accumulation curves (R Development Core Team 2010). Model fit was evaluated using Akaike's Information Criteria (AIC), along with a visual inspection. The best fit model for each curve was then used to estimate how much sampling effort was required to capture 80% and 90% of our previously calculated S_{est} from that season.

The Morgan-Mercer-Flodin model was consistently the best-fit model, and was only replaced as the best-fit model in one of the species-accumulation curves by the exponential function. For most curves, the Morgan-Mercer-Flodin and Wiebull fit well and similarly, although the Morgan-Mercer Flodin always exhibited a lower AIC score. Some models such as the negative exponential function fit all curves poorly. Our sampling designs yield comparable results in terms of capturing species richness. Overall, to reach 90% of total estimated species richness, nearly double to triple the sampling distance is required relative to 80% of total estimated species richness (Table B.2); the exact distance to reach 80-90% of total estimated species richness is variable among seasons, with lower estimates of effort during seasons with low total estimated species richness. If future goals require 80% of total estimated species richness, we recommend sampling a minimum of 10-12 kilometers per sampling event on the Penobscot River, whereas if future goals require 90% of total estimated species richness, we recommend 20-25 kilometers of electrofishing per sampling event.

Table B.1. Functions for modeling species-accumulation curves on the Penobscot River during 2010 and 2011 for stratified-random and fixed-station sampling designs.

Model	Function	Mean AIC
Morgan-Mercer-Flodin	$ax^c/(b+x^c)$	-29.14
Weibull	$a(1-\exp(-bx^c))$	-16.77
Rational Function	$(a+bx)/(1+cx)$	-10.45
Chapman-Richards	$a(1-\exp(-bx))^c$	-2.69
Exponential	$a+b\log x$	-0.42
Clench	$a(x/(b+x))$	21.95
Power	ax^b	31.22
Logistic	$a/(1+\exp(-bx+c))$	34.14
Negative Exponential	$(a/b)(1-\exp(-bx)$	56.88

Table B.2. The amount of electrofishing distance required to capture 80-90% of total estimated species richness. Electrofishing distance (km) was estimated at the intersection of the best-fit model with 80% and 90% of the first order jackknife estimates for total species richness (S_{est}). Estimates were derived for stratified-random (S-R) and fixed-station (F-S) sampling designs for each sampling event during 2010-2011 if effort was determined sufficient for modeling.

						Effort (km) to capture x% of S_{est} estimated by the best-fit model	
	Design	S_{obs}	S_{est}	$\%S_{est}$ Obs.	Best Fit Model	80%	90%
Summer 2010	S-R	20	22.9	87.3%	MMF	7	22
Summer 2010	F-S	21	24.6	85.4%	MMF	8	16
Fall 2010	S-R	21	26.6	78.9%	MMF	10.5	18
Fall 2010	F-S	21	25.5	82.4%	MMF	9.5	20.5
Summer 2011	S-R	27	31.8	84.9%	MMF	11.5	26.5
Summer 2011	F-S	24	29.7	80.8%	EXP	12	21.5
Fall 2011	S-R	21	22.0	95.6%	MMF	3.5	7

APPENDIX C. EVALUATION OF PREVIOUS DATA FROM KLEINSCHMIDT ASSOCIATES, INC.

During 2008 and 2009, Kleinschmidt Associates sampled along fixed-station transects on the Penobscot River and major tributaries (Kleinschmidt Associates 2009a;2009b). We sampled along each of these fixed-station transects during 2010 and 2011. In total, 28 species were captured during 2008-2009 (Kleinschmidt Associates 2009a;2009b), compared to 35 species by our sampling during 2010-2011. On the main-stem Penobscot River where much of the effort was focused, 24 species were captured by fixed-station sampling during 2009-2010, whereas 30 species were captured along the same transects during 2010-2011 (Table C.1). This represents a 25% increase in the number of species captured by our sampling relative to data from Kleinschmidt Associates; additionally, the average number of fish captured per kilometer was elevated by 193% during our sampling (Table C.1). The increase in the number of species and the total number of fish captured was accompanied by a 34% increase in mean sampling effort per transect (Table C.1).

Table C.1. Comparison of catch and effort for fixed-station transects on the main-stem Penobscot River by two different sampling agencies.

	Kleinschmidt 2008-2009	Kiraly et al. 2010-2011	Change
Number of Species	24	30	+25%
Catch per km (Mean±SE)	104 ± 13	305 ± 56	+193%
Seconds per transect (Mean±SE)	3254 ± 68	4373 ± 228	+34.4%

Although it is possible that some sampling during that time period produced data comparable to our 2010-2011 sampling (i.e. sampling from transects where relatively few fish are present), we believe that insufficient effort was expended during 2008-2009 along transects which often yield high ($n > 300$ fish/km) numbers of small (length < 100mm) fish. These transects required more seconds of electrofishing in order to effectively capture a representative sample; low sampling effort could affect the number of species captured along with proportional abundance of each species. Catch per unit can be standardized by seconds of electrofishing instead of by distance electrofished in order to make appropriate comparisons, but this does not alleviate the non- or under-representation of species exhibited by low sampling effort. Because our assemblage analyses involved the distribution and proportional abundance of all species captured, we did not use the 2008-2009 data from Kleinschmidt Associates in our analyses.

APPENDIX D. CPUE AND MPUE BY SAMPLING EVENT AND STRATUM

The tables below show catch and mass per effort for each stratum, calculated for all fish captured per kilometer of electrofishing from within each stratum. Mean and standard error values incorporate the variability among transects within each stratum, and were calculated with the ratio estimation formulae from Hansen et al. (2007):

$$\hat{R} = \frac{\sum_{i=1}^{n} y_i}{\sum_{i=1}^{n} x_i}$$

$$SE(\hat{R}) = \frac{1}{\sqrt{n}\,\bar{x}}\sqrt{\frac{\sum_{i=1}^{n}(y_i - \hat{R}x_i)^2}{n-1}}$$

Table D.1. Catch by numbers per kilometer of all fish captured per strata for each sampling event. Standard errors are shown in parentheses.

Stratum	Name	Summer 2010	Fall 2010	Summer 2011	Fall 2011
1	Hampden	35.7(24.61)	99.1(NA)	177.1(55.59)	152.9(88.71)
2	Bangor Lower	80.0(27.8)	69.5(55.67)	253.5(68.26)	135.7(31.83)
3	Bangor Upper	142.7(30.04)	156.9(59.47)	329.1(52.5)	555(204.26)
4	Veazie Impound	90.7(32.7)	336.8(61.05)	398.9(82.61)	392.5(56.77)
5	Orono	179.6(79.87)	281.9(158.21)	503.2(328.36)	402.4(220.64)
6	Milford	356.2(148.21)	1108.3(291.74)	144.5(40.19)	1372.3(300.7)
7	Midriver Lower	640.9(260.43)	2843.2(456.75)	1191.7(294.71)	1729.4(462.87)
8	Argyle	471.7(154.06)	1521.5(639.84)	544.5(169.45)	1035.5(391.31)
9	Midriver Upper	700.4(176.87)	664.1(111.81)	470.7(106.33)	985.6(219.71)

Table D.2. Catch by kilograms per kilometer of all fish captured per strata for each sampling event. Standard errors are shown in parentheses.

Stratum	Name	Summer 2010	Fall 2010	Summer 2011	Fall 2011
1	Hampden	2.0(1.28)	1.3(NA)	1.7(0.34)	1.8(0.15)
2	Bangor Lower	7.5(2.29)	0.7(0.11)	6.5(1.96)	3.6(1.34)
3	Bangor Upper	14.3(2.69)	9.1(2.04)	15.6(7.48)	5.3(3.34)
4	Veazie Impound	5.3(2.62)	6.3(2.45)	8.5(1.69)	3.7(0.83)
5	Orono	21.7(6.07)	19.7(7.76)	41.7(9.02)	20.3(1.92)
6	Milford	4.2(1.08)	8.8(1.51)	7.2(1.1)	8.5(3.71)
7	Midriver Lower	10.4(1.75)	9.0(1.33)	13.2(3.5)	15.2(3.6)
8	Argyle	8.5(1.53)	13.5(3.72)	14.5(1.78)	14.5(1.16)
9	Midriver Upper	8.2(0.96)	6.0(1.57)	15(1.75)	12.9(1.76)

APPENDIX E. PROPORTIONAL ABUNDANCE AND MASS BY LIFE HISTORY

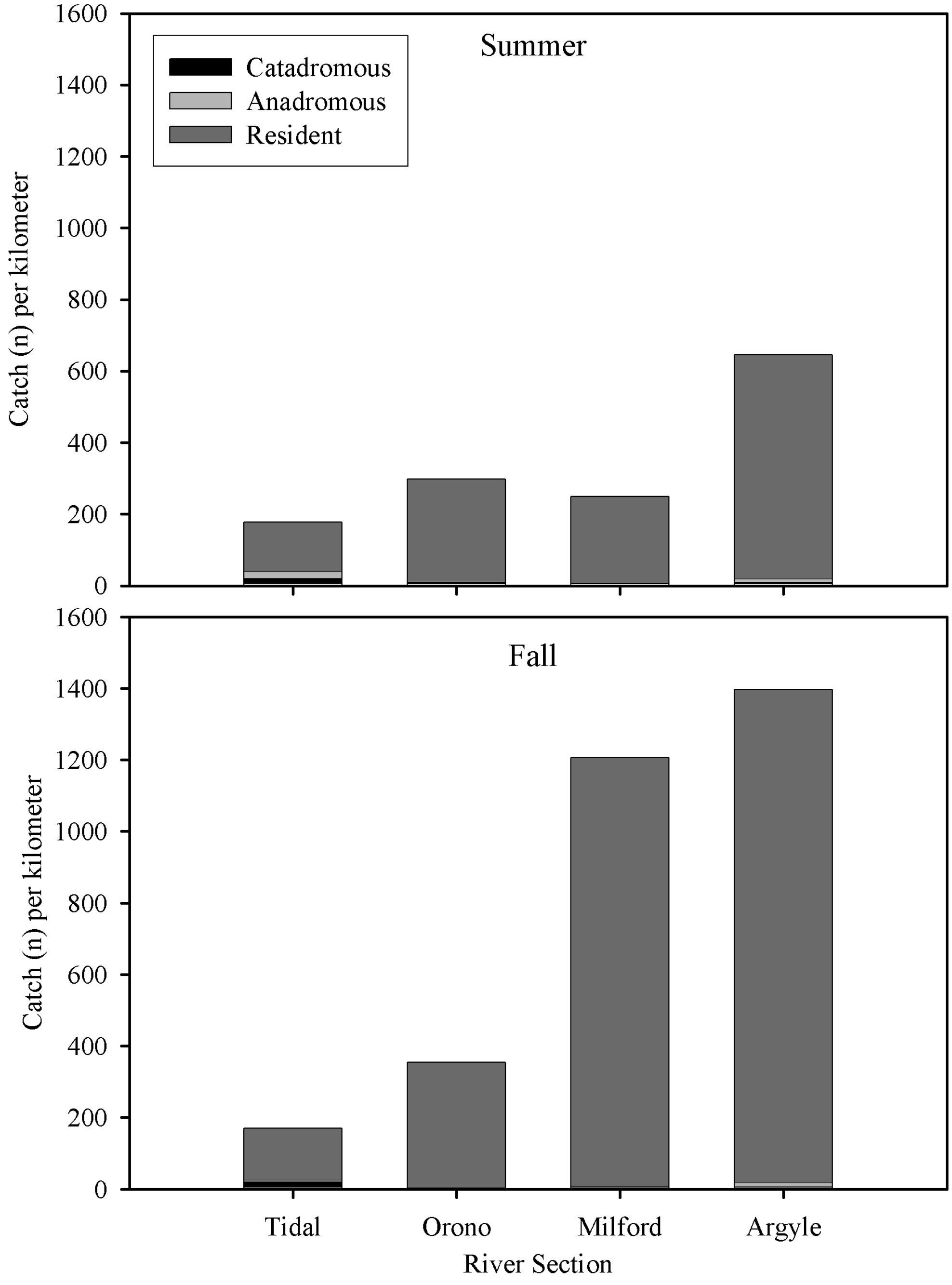

Figure E.1. Catch per kilometer for all fish captured within river sections on the Penobscot River, 2010-2011, by life history.

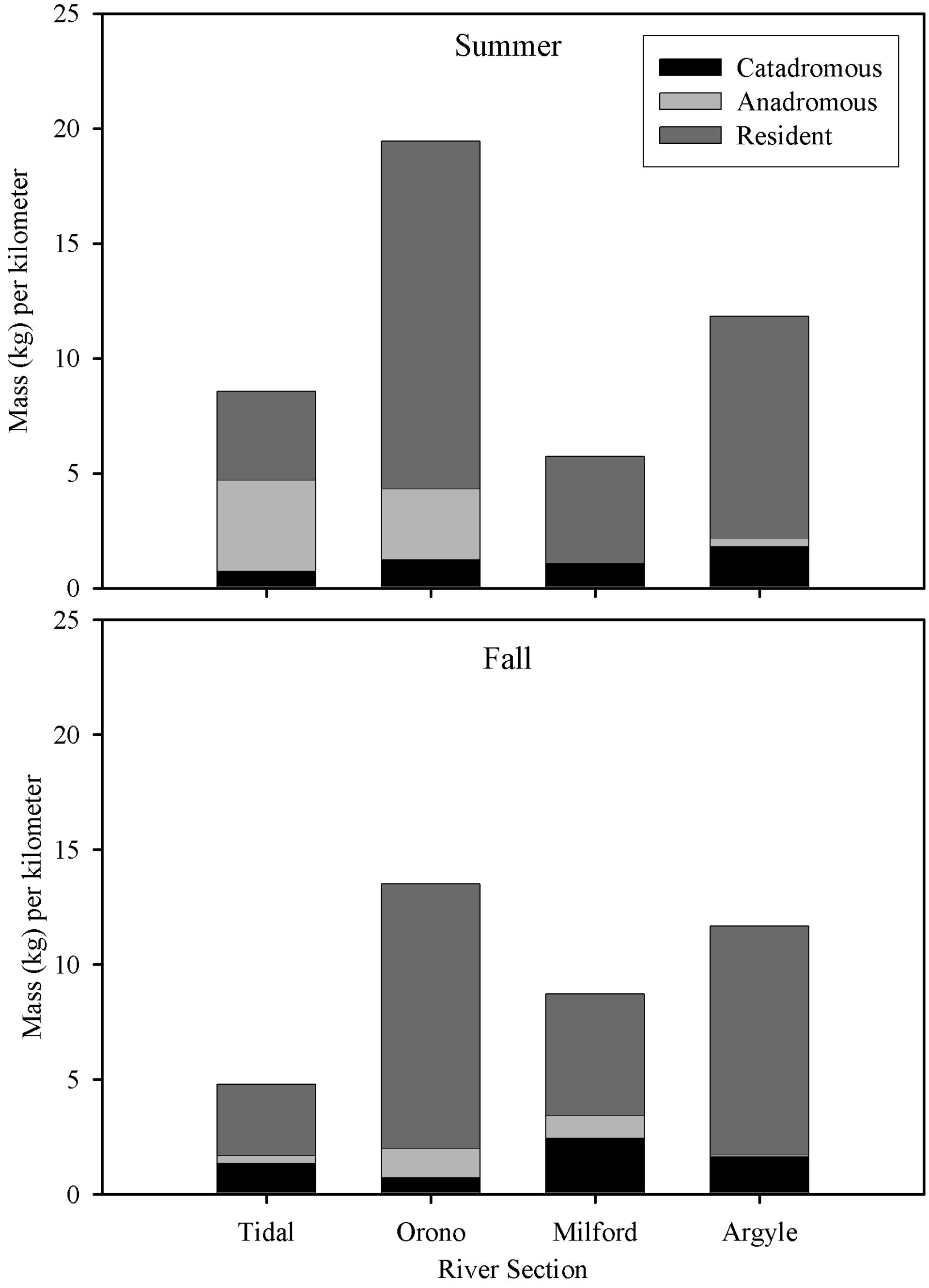

Figure E.2. Kilograms per kilometer for all fish captured within river sections on the Penobscot River, 2010-2011, by life history.

APPENDIX F. PERCENT CATCH TABLES FOR ALL SPECIES

Table F.1. Percent of catch per kilometer for all fish captured within each stratum on the Penobscot River, summer 2010. Strata locations are shown in Figure 1.1.

	Penobscot River Stratum								
Species	1	2	3	4	5	6	7	8	9
ALE	0	13.43	21.09	0	0	0	0	0	0
ATS	0	0	0.66	0	0	0.09	0	0.06	0.16
BBH	0	0	0.16	1.58	1.79	0.27	1.03	1.15	0.05
BKF	0	0.93	0.33	0	0	0.27	0.21	0.06	0
BKT	0	0	0.16	0	0	0	0	0	0
BND	0	0	0	0	0	0	0	0	0
BNS	0	0	0	0	0	0	0	0	0
CHP	2.75	7.41	4.28	1.05	0.89	0.46	1.03	1.04	0.38
CRA	0	0	0	0	0	0	0	0	0
CRC	0	0	0	1.05	0	0	0.1	0.06	0.05
CSH	0	0.46	5.93	0.53	14.29	29.79	38.08	40.5	38.76
CSK	0	0	0	0	0	0	0.41	0.35	0
EEL	66.06	1.85	17.79	6.32	5.06	1.01	1.65	2.53	0.22
ESM	0	0	0	0	0	0	0	0	0
FF	0.92	7.41	21.58	8.95	31.85	35.47	30.34	35.14	50.41
FSD	0	0	0	0	0	0	0	0	0
GSH	0	1.39	1.32	4.74	0.3	2.2	0.1	0	0
HER	0	18.98	2.14	0	0	0	0	0	0
LAM	2.75	0	0.33	0	0.3	0	1.44	1.9	0
LMB	0	0	0	0	0	0	0	0	0
LNS	0	0	0	0	0	0	0	0	0.05
MUM	0	0	0	0	0	0	0	0	0
NSS	0	0	0	0	0	0	0	0	0.05
PS	9.17	2.31	0.49	18.95	2.98	0.55	0.1	0.06	0.16
RBD	0	0	0	0	0	0	0	0.06	0
RBS	11.01	5.56	5.6	12.11	16.67	2.02	2.99	6.45	1.37
SHD	0.92	0	0	0	0	0	0	0	0
SMB	5.5	20.37	14.66	16.84	18.45	4.31	0.93	1.27	1.48
SSC	0	0	0	0	0	0	0	0	0
STB	0	0.93	0	0	0	0	0	0	0
SGN	0	0	0	0	0	0	0	0	0
TSS	0	0	0	0	0	0	0	0	0
WP	0	0	0	0	0	0	0	0	0
WS	0.92	18.98	2.64	27.37	7.44	23.46	21.16	8.81	6.54
YP	0	0	0.82	0.53	0	0.09	0.41	0.58	0.27

Table F.2. Percent of catch per kilometer for all fish captured within each stratum on the Penobscot River, fall 2010. Strata locations are shown in Figure 1.1.

	Penobscot River Stratum								
Species	1	2	3	4	5	6	7	8	9
ALE	3.03	4.82	0	0	0	0	0	0	0
ATS	0	0	0	0	0	0.03	0	0	0
BBH	0	0	0	0	0	0.03	0.21	1.01	0.42
BKF	33.33	86.75	0.79	0.74	0	1.54	0.3	0.02	0
BKT	0	0	0	0	0	0	0	0	0
BND	1.01	0	0	0	0	0	0	0.02	0
BNS	0	0	0	0	0	0	0	0	0
CHP	1.01	1.2	0.4	0	0	0.22	0.23	0.42	0.12
CRA	0	0	0	0	0	0	0	0	0
CRC	0	0	0	0	0	0	0.02	0.04	0
CSH	3.03	1.2	5.74	3.53	0	42.01	31.34	26.92	3.13
CSK	0	0	0	0	0	0.03	0	0.08	1.8
EEL	0	0	28.12	1.77	2.33	0.58	0.09	0.73	0.78
ESM	0	1.2	0	0	0	0	0	0	0
FF	16.16	1.2	9.7	13.55	11.67	29.38	44.78	41.59	30.85
FSD	0	0	0	0	0	0	0	0	0
GSH	1.01	0	0.4	5.89	0	1.63	0.02	0.44	0
HER	0	0	0	0	0	0	0	0	0
LAM	0	0	0.2	0	0	0.03	0.56	0.95	1.74
LMB	1.01	0	0.4	0.15	0	0	0	0	0.12
LNS	0	0	0	0	0.18	0	0	0	0
MUM	34.34	0	0	0	0	0	0	0	0
NSS	0	0	0	0	0	0	0	0	0.06
PS	0	0	1.78	3.24	1.44	1.38	0.07	0.2	1.8
RBD	0	0	0	0	0	0	0	0	0
RBS	0	0	14.65	38.73	24.78	9.74	7.53	14.62	13.23
SHD	0	0	0	0	0	0	0	0	0
SMB	6.06	3.61	34.06	31.81	56.55	11.25	3.83	7.72	40.17
SSC	0	0	0	0	0	0	0	0	0
STB	0	0	0	0	0	0	0	0	0
SGN	0	0	0	0	0	0	0	0	0
TSS	0	0	0	0	0	0	0	0	0
WP	0	0	0	0	0	0.03	0	0	0.12
WS	0	0	3.76	0.44	3.05	1.89	10.91	4.54	5.29
YP	0	0	0	0.15	0	0.22	0.09	0.69	0.36

Table F.3. Percent of catch per kilometer for all fish captured within each stratum on the Penobscot River, summer 2011. Strata locations are shown in Figure 1.1.

	Penobscot River Stratum								
Species	1	2	3	4	5	6	7	8	9
ALE	0.6	0.81	1.65	0	0	0	0	0	0
ATS	0	0	0.36	0	0.55	0	0	0.05	0
BBH	0.3	0.49	0.79	0.12	0.74	0.66	0.8	1.15	0.9
BKF	13.83	1.46	1.22	1.1	0	0.22	0.42	0	0
BKT	0	0	0	0	0	0	0	0	0
BND	0.45	0.16	0	0	0	0	0	0	0
BNS	0.15	0	0.14	0	0	0	0	0	0
CHP	0	0	0.36	0.97	0.18	1.32	3.07	2.25	2
CRA	0.75	1.29	0.29	0	0	0	0	0	0
CRC	0.3	0	0	0	0	0	0	0	0
CSH	0.9	0.16	5.32	2.31	6.72	17.58	46.79	34.78	42.7
CSK	0	0	0	0	0.09	0.22	0	0.43	1.52
EEL	6.32	12.62	7.41	1.46	3.22	6.59	0.51	2.39	2.34
ESM	0.9	0	0.22	0	0	0	0.06	0	0
FF	6.77	2.59	9.35	15.71	15.92	41.54	30.06	36.6	35.26
FSD	0	0	0	0	0	0	0.03	0	0
GSH	22.41	4.53	14.89	12.18	23.28	0.66	1.41	0.62	0.07
HER	5.56	12.46	1.08	0	0	0	0	0	0
LAM	1.35	4.05	0.29	1.22	1.1	0.44	0.58	3.39	1.17
LMB	0	0	0.07	0	0	0	0	0	0
LNS	0	0	0	0	0	0	0	0	0
MUM	2.56	0	0	0	0	0	0	0	0
NSS	0	0	0	0	0	0	0	0	0.07
PS	19.4	9.22	32.09	22.17	22.54	0.44	0.38	0.67	0.9
RBD	0	0	0	0	0	0	0	0.05	0
RBS	3.31	9.71	4.46	4.51	4.88	7.91	8.44	9.7	4.2
SHD	0	0	0.14	0	0	0	0	0	0
SMB	11.13	24.43	13.53	38.25	16.65	20.88	0.86	4.25	4.34
SSC	0	0	0	0	0	0	0	0	0
STB	0	0	0	0	0	0	0	0	0
SGN	0	0	0	0	0	0	0	0	0
TSS	0.6	0	0	0	0	0	0	0	0
WP	0.45	0.81	0.14	0	0	0	0	0	0
WS	0.9	0.16	1.65	0	4.14	1.32	5.98	3.2	3.1
YP	1.05	15.05	4.53	0	0	0.22	0.61	0.48	1.45

Table F.4. Percent of catch per kilometer for all fish captured within each stratum on the Penobscot River, fall 2011. Strata locations are shown in Figure 1.1.

	Penobscot River Stratum								
Species	1	2	3	4	5	6	7	8	9
ALE	0	7.6	0.38	0	0	0	0	0	0
ATS	0	0	0	0	0.29	0.04	0	0	0
BBH	0	0	0	0	0.39	0	0.55	0.23	0.19
BKF	48.77	2.05	0.19	0	0.39	0.44	0.29	0	0
BKT	0	0	0	0	0	0	0	0	0
BND	0	0	0	0	0	0	0	0	0
BNS	0	0	0	0	0	0	0	0	0
CHP	0	0	0	0.16	0.39	0.62	5.13	1.37	0.95
CRA	0	0	0.19	0	0	0	0	0	0
CRC	0	0	0	0	0	0	0	0	0
CSH	1.23	1.17	6.67	1.45	27.29	73.4	60.94	60.64	45.63
CSK	0	0	0	0	0	0	0	0	0.63
EEL	5.83	11.99	6.1	0.81	0.58	0.31	0.22	0.4	0.95
ESM	0	0	0.19	0	2.41	0	0.07	0.11	0
FF	6.75	10.23	18.67	8.24	32.69	11.55	22.54	23.63	23.57
FSD	0	0	0	0	0	0	0	0	0
GSH	0	0	3.05	0	0.77	2.71	3.96	1.6	0.32
HER	0	0	1.71	0	0	0	0	0	0
LAM	0.31	0.58	0	0	0.1	0.04	0.33	0.51	0.82
LMB	0.31	0	0.57	0	0	0	0.04	0	0
LNS	0	0	0	0	0	0	0	0	0
MUM	0.31	0	0	0	0	0	0	0	0
NSS	0	0	0	0	0	0	0	0	0
PS	14.72	5.56	5.9	6.79	3.18	0.09	0.4	1.26	1.27
RBD	0	0	0	0	0	0	0	0	0
RBS	7.06	9.06	9.71	14.86	1.64	1.69	1.36	1.49	3.93
SHD	0	0	0	0	0	0	0	0	0
SMB	9.51	51.17	44.76	67.69	25.46	7.42	1.8	6.18	18.95
SSC	0	0	0	0	0	0	0	0	0
STB	0	0	0	0	0	0	0	0	0
SGN	0	0.29	0	0	0	0	0	0	0
TSS	0	0	0	0	0	0	0	0	0
WP	5.21	0	0	0	0	0.09	0.15	0.06	0
WS	0	0.29	1.52	0	3.66	0.89	0.62	1.43	2.34
YP	0	0	0.38	0	0.77	0.71	1.61	1.09	0.44

APPENDIX G. PERCENT MASS TABLES FOR ALL SPECIES

Table G.1. Percent of mass per kilometer for all fish captured within each stratum on the Penobscot River, summer 2010. Strata locations are shown in Figure 1.1.

	Penobscot River Stratum								
Species	1	2	3	4	5	6	7	8	9
ALE	0	17.51	32.33	0	0	0	0	0	0
ATS	0	0	11.52	0	0	0.19	0	0.15	0.49
BBH	0	0	0.36	7.67	2.56	4.58	19.13	12.27	1.08
BKF	0	0.01	0	0	0	0.02	0.04	0	0
BKT	0	0	0.11	0	0	0	0	0	0
BND	0	0	0	0	0	0	0	0	0
BNS	0	0	0	0	0	0	0	0	0
CHP	7.62	12.21	5.83	2.53	2.54	1.32	10.7	7.22	2.01
CRA	0	0	0	0	0	0	0	0	0
CRC	0	0	0	0.02	0	0	0.01	0.01	0.02
CSH	0	0	0.11	0.01	0.15	2.82	4.82	3.49	6.4
CSK	0	0	0	0	0	0	1.64	0.55	0
EEL	48.44	3.55	6.8	6.43	5.89	19.02	17.26	26.5	4.47
ESM	0	0	0	0	0	0	0	0	0
FF	0.77	0.17	1.32	0.59	3.82	8.9	6.58	5.14	16.19
FSD	0	0	0	0	0	0	0	0	0
GSH	0	0.05	0.01	2.74	0	0.55	0	0	0
HER	0	21.56	2.77	0	0	0	0	0	0
LAM	9.78	0	2.09	0	1.45	0	0.34	0.48	0
LMB	0	0	0	0	0	0	0	0	0
LNS	0	0	0	0	0	0	0	0	2.36
MUM	0	0	0	0	0	0	0	0	0
NSS	0	0	0	0	0	0	0	0	0
PS	3.23	0.97	0.33	8.79	0.33	1.38	0.05	0.04	0.44
RBD	0	0	0	0	0	0	0	0	0
RBS	9.21	5.08	6.6	21.89	14.16	10.68	18.32	13.44	7.82
SHD	13.31	0	0	0	0	0	0	0	0
SMB	5.7	17.76	24.55	32.76	32.1	43.08	16.19	9.93	28.22
SSC	0	0	0	0	0	0	0	0	0
STB	0	20.55	0	0	0	0	0	0	0
SGN	0	0	0	0	0	0	0	0	0
TSS	0	0	0	0	0	0	0	0	0
WP	0	0	0	0	0	0	0	0	0
WS	1.95	0.58	4.82	16.28	37.01	7.21	3.26	18.81	30.18
YP	0	0	0.45	0.3	0	0.26	1.66	1.96	0.31

Table G.2. Percent of mass per kilometer for all fish captured within each stratum on the Penobscot River, fall 2010. Strata locations are shown in Figure 1.1.

	Penobscot River Stratum								
Species	1	2	3	4	5	6	7	8	9
ALE	0.74	1.4	0	0	0	0	0	0	0
ATS	0	0	0	0	0	0.14	0	0	0
BBH	0	0	0	0	0	0.02	3.12	11.5	1.68
BKF	5.17	18.57	0.04	0.05	0	0.12	0.11	0	0
BKT	0	0	0	0	0	0	0	0	0
BND	0.11	0	0	0	0	0	0	0	0
BNS	0	0	0	0	0	0	0	0	0
CHP	72.98	28.05	1.49	0	0	2.66	13.04	12.3	2.03
CRA	0	0	0	0	0	0	0	0	0
CRC	0	0	0	0	0	0	0.01	0.01	0
CSH	0.41	0.06	0.17	0.28	0	5.02	9.52	3.33	0.61
CSK	0	0	0	0	0	0.18	0	0.28	7.44
EEL	0	0	23.68	15.08	5.26	30.58	10.25	17.43	11.9
ESM	0	0.35	0	0	0	0	0	0	0
FF	2.79	0.23	0.6	2.01	1.73	6.11	18.11	6.98	7.87
FSD	0	0	0	0	0	0	0	0	0
GSH	0.02	0	0.01	1.07	0	0.34	0.19	0.09	0
HER	0	0	0	0	0	0	0	0	0
LAM	0	0	0.01	0	0	0.02	0.89	0.51	1.11
LMB	1.16	0	0.22	0.11	0	0	0	0	0.06
LNS	0	0	0	0	2.5	0	0	0	0
MUM	7.76	0	0	0	0	0	0	0	0
NSS	0	0	0	0	0	0	0	0	0
PS	0	0	1.25	6.91	0.51	0.71	0.13	0.55	0.73
RBD	0	0	0	0	0	0	0	0	0
RBS	0	0	8.52	13.83	7.86	7.02	11.23	9.05	7.56
SHD	0	0	0	0	0	0	0	0	0
SMB	8.84	51.33	39.45	58.35	39.23	28.75	8.94	30.14	50.6
SSC	0	0	0	0	0	0	0	0	0
STB	0	0	0	0	0	0	0	0	0
SGN	0	0	0	0	0	0	0	0	0
TSS	0	0	0	0	0	0	0	0	0
WP	0	0	0	0	0	0.16	0	0	0.04
WS	0	0	24.58	1.94	42.92	16.23	23.58	5.98	8.06
YP	0	0	0	0.35	0	1.94	0.87	1.86	0.3

Table G.3. Percent of mass per kilometer for all fish captured within each stratum on the Penobscot River, summer 2011. Strata locations are shown in Figure 1.1.

	Penobscot River Stratum								
Species	1	2	3	4	5	6	7	8	9
ALE	0.44	2.73	4.38	0	0	0	0	0	0
ATS	0	0	46.57	0	26.37	0	0	9.83	0
BBH	8.16	1.72	2.75	0.04	1.62	4.53	10.95	8.84	5.23
BKF	3.88	0.15	0.06	0.07	0	0.01	0.06	0	0
BKT	0	0	0	0	0	0	0	0	0
BND	0.02	0	0	0	0	0	0	0	0
BNS	0.01	0	0	0	0	0	0	0	0
CHP	0	0	1.95	2.48	0.12	0.02	6.62	3.3	6.37
CRA	0.72	0.4	0.05	0	0	0	0	0	0
CRC	0.05	0	0	0	0	0	0	0	0
CSH	0.08	0.01	0.25	0.23	0.22	0.58	7.88	2.84	3.46
CSK	0	0	0	0	0.43	0.43	0	1.06	2.98
EEL	16.46	17.34	5.82	3.45	7.18	18.58	6.81	19.65	13.42
ESM	0.16	0	0.01	0	0	0	0.01	0	0
FF	2.05	0.94	0.98	2.99	1.9	3.56	10.05	6.23	9.12
FSD	0	0	0	0	0	0	0.01	0	0
GSH	2.17	0.36	0.61	0.76	0.36	0.03	0.17	0.03	0
HER	8.24	5.05	0.34	0	0	0	0	0	0
LAM	0.5	0.66	0.02	0.14	0.98	0.05	0.22	0.54	0.22
LMB	0	0	0.01	0	0	0	0	0	0
LNS	0	0	0	0	0	0	0	0	0
MUM	0.65	0	0	0	0	0	0	0	0
NSS	0	0	0	0	0	0	0	0	0
PS	3.94	0.92	2.58	3.97	0.68	0.06	0.7	0.09	0.16
RBD	0	0	0	0	0	0	0	0	0
RBS	14.9	20.46	3.18	1	2.86	16.08	22.48	8.16	6.92
SHD	0	0	2.62	0	0	0	0	0	0
SMB	35.16	45.12	22.69	84.84	24.96	49.08	9.11	11.2	26.53
SSC	0	0	0	0	0	0	0	0	0
STB	0	0	0	0	0	0	0	0	0
SGN	0	0	0	0	0	0	0	0	0
TSS	0.09	0	0	0	0	0	0	0	0
WP	0.26	0.42	0.06	0	0	0	0	0	0
WS	1.48	0.02	4.28	0	32.32	6.47	23.08	27.53	24.24
YP	0.56	3.69	0.8	0	0	0.52	1.86	0.7	1.35

Table G.4. Percent of mass per kilometer for all fish captured within each stratum on the Penobscot River, fall 2011. Strata locations are shown in Figure 1.1.

	Penobscot River Stratum								
Species	1	2	3	4	5	6	7	8	9
ALE	0	0.48	0.32	0	0	0	0	0	0
ATS	0	0	0	0	19.4	29.96	0	0	0
BBH	0	0	0	0	1.64	0	9.02	2.52	1.11
BKF	14.61	0.13	0.08	0	0.02	0.08	0.09	0	0
BKT	0	0	0	0	0	0	0	0	0
BND	0	0	0	0	0	0	0	0	0
BNS	0	0	0	0	0	0	0	0	0
CHP	0	0	0	0.14	1.13	9.48	18.82	8.67	3.58
CRA	0	0	0.78	0	0	0	0	0	0
CRC	0	0	0	0	0	0	0	0	0
CSH	0.05	0.03	0.78	0.19	1.25	7.74	8.53	15.41	5.6
CSK	0	0	0	0	0	0	0	0	3.24
EEL	40.06	41.73	22.05	5.25	3.34	24.15	6.3	15.94	18.2
ESM	0	0	0.03	0	0.08	0	0.02	0.02	0
FF	10.51	1.25	5.22	2.29	4.87	5.15	13.13	11.15	5.37
FSD	0	0	0	0	0	0	0	0	0
GSH	0	0	0.83	0	0.07	2.39	0.75	0.38	0.21
HER	0	0	1.75	0	0	0	0	0	0
LAM	0.14	0.11	0	0	0.01	0.03	0.16	0.21	0.39
LMB	0.1	0	1.1	0	0	0	0.03	0	0
LNS	0	0	0	0	0	0	0	0	0
MUM	0.02	0	0	0	0	0	0	0	0
NSS	0	0	0	0	0	0	0	0	0
PS	17.93	2.47	7.59	5.14	1.13	0.48	1.24	0.71	1.1
RBD	0	0	0	0	0	0	0	0	0
RBS	6.44	2.77	7.43	12.94	1.28	1.18	2.37	2.74	5.15
SHD	0	0	0	0	0	0	0	0	0
SMB	9.4	23.58	46.76	74.06	38.92	14.19	17.83	20.78	52.86
SSC	0	0	0	0	0	0	0	0	0
STB	0	0	0	0	0	0	0	0	0
SGN	0	27.34	0	0	0	0	0	0	0
TSS	0	0	0	0	0	0	0	0	0
WP	0.74	0	0	0	0	0.4	0.75	0.08	0
WS	0	0.11	4.78	0	26.68	4.25	13.25	19.09	3.1
YP	0	0	0.51	0	0.17	0.51	7.71	2.31	0.07

BIOGRAPHY OF THE AUTHOR

Ian Kiraly was born in Oneonta, New York on August 2nd, 1985. He was raised on a small dairy farm near Franklin, New York, where he graduated from Franklin Central School in 2003. Ian attended Cornell University and graduated in 2007 with a Bachelor of Science degree in Natural Resources, concentrating in Applied Ecology. He conducted fisheries research for Cornell University and New York State's Department of Environmental Conservation before acceptance as a research assistant within the University of Maine's graduate program. Ian is a candidate for a Master of Science degree in Wildlife Ecology from the University of Maine in August, 2012.