Assessment of a High-Frequency Sound Fish Diversion System at the Annapolis Tidal Generating Station with Notes about the Survival of Fish Moving Seaward at the Annapolis Causeway

Final Report
to
Nova Scotia Power Inc.

prepared by:
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EXECUTIVE SUMMARY

During the fall of 1999, a fish diversion system utilising high-frequency sound was installed at the Annapolis Tidal Generating Station in Nova Scotia to test its the effectiveness of reducing the passage of fish, primarily anadromous *Alosa*, through the turbine tube. The fish diversion system tested was a band-limited random noise signal projected into the turbine forebay by 4 transducers mounted across intake. The signal pulse was presented at a 33% duty cycle (0.5 seconds on, followed by 1 second off) with the majority of energy focused between 122 and 128kHz. The 160 dB threshold was reached at a distance of 10 to 12 m from the intake face.

The system was partially effective for *Alosa*, but not for the other 11 species for which data were analysed. With the system turned on, the rate of passage of American shad through the turbine tube was 35% less than when the system was turned off. The rate of blueback herring passage apparently only decreased (26%) if the largest catches are excluded from the analysis, suggesting that effectiveness is decreased when large schools of fish are present. The rate of passage through the fishway located nearest the turbine increased by factors of 3.6 times for American shad and 4.1 times for alewife when the diversion system was turned on. The diversion system apparently did not affect the rate of fish passage through the fishway located by the sluice gates. The effectiveness of the system could potentially be improved by angling the barrier towards the fishway located near the turbine.

On an opportunistic basis we explored methods of estimating turbine mortality at the TGS. As an alternative to direct measurement of handling mortality, a significant source of error in turbine mortality studies, we used a logistic model to relate the probability of death to the duration of the net deployment. The intercept in this model is an estimate of turbine mortality. The model appeared to provide believable estimates for robust species, but requires that an offset be fitted for fish that are more susceptible to damage from handling. Preliminary results suggest that turbine mortality is between 0.0 and 6.3 % for alewife, sea lamprey, Atlantic silversides, pipefish, winter flounder and windowpane. Some species, such as Atlantic silverside, show a preference for passing seaward through the new fishway.

Development of behavioural guidance systems can be costly and are without a guarantee of success. Ultimately, the effectiveness of a fish diversion system should be measured using the population-level response. For a species with a high compensatory capacity and low mortality associated with passage at the causeway (fishway usage coupled with turbine mortality), the effect of diversion at the population-level may be negligible. If a multi-species diversion system is to be developed, target species should be chosen based on a population-level risk assessment, to avoid the developmental cost for species for which the benefit may be minor.

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1.0 BACKGROUND

1.1 Overview

The Annapolis River Estuary, Nova Scotia, is home to Canada's only tidal hydroelectric generating station (TGS). This station, which has been in operation since 1984 (Stokesbury 1987), was constructed to test the feasibility of using the StraFlo^{TM} turbine in marine environments, in anticipation of larger scale tidal generating development in the upper Bay of Fundy. As such, the station has also served as a test unit for the environmental impacts of such a facility. At this time, two fishways exist to augment fish passage at the station. Studies of the effectiveness of these fishways suggest that the majority of fish moving seaward past the causeway travel through the turbine tube (Stokesbury 1987, Gibson 1996a). These findings, coupled with concerns about turbine mortality, have prompted Nova Scotia Power Inc. (NSPI) to explore technologies to reduce the incidence of turbine passage, thus increasing the effectiveness of the fishways.

During the last decade, behavioral guidance systems have come to the leading edge of fish passage research. Stimuli such as light, sound and electric shock are used to elicit an avoidance response from fish, repelling fish from a given area, such as intake pipes at cooling water plants or hydroelectric generating stations. Of these technologies, the use of high frequency sound, or ultrasound, to repel American shad and some other species is perhaps the most compelling (Popper and Carlson 1998). This technology is thought to be effective for some species (e.g. *Alosa*) and not others (e.g. salmonids), but has not been tested for many estuarine species. Additionally, while ultrasound is known to elicit avoidance responses in species such as alewife, it is not known whether these responses can be used to direct fish towards fish passage facilities.

This study was therefore undertaken to test the feasibility of using ultrasound to deter fish from passing through the turbine at the Annapolis TGS. Different species of fish detect and respond to different sound stimuli. The three species of anadromous *Alosa* that utilize the Annapolis River and Estuary as spawning and nursery habitat (American shad, blueback herring and alewife) were chosen as the target species for this study. The sound barrier was designed specifically to deter these species. Due to the uncertainty of the range of hearing of many species of fish, it was unknown whether any of the other species present in the Annapolis Estuary would respond to this signal. The diversity of the fish community in the vicinity of the TGS provided the opportunity to test whether the selected signal produced a measurable response in several other species, albeit without the anticipation of success for the majority of these.

The primary objectives of the study were therefore to determine whether an ultrasound barrier reduced the incidence of fish passage through the turbine (with the primary focus on *Alosa*) and, if fish were deterred from passing through the turbine, whether they would still be able to pass through the fishways. Peripheral to the main objectives, the opportunity to study fish diversion at the TGS, also provided the opportunity to explore alternative methods to study fish mortality at the TGS. While mortality of clupeids passing though the turbine has been studied at the TGS, a number of issues still require resolution before accurate and precise estimates of mortality are obtained. Mortality has not been studied for other species. A secondary objective was therefore to explore an alternative methodology for determining turbine mortality.

1.2 The Study Area

The Annapolis River Estuary is a macrotidal estuary located in south-western Nova Scotia (Figure 1). The upper reaches of this estuary are near Bridgetown and the estuary extends seaward to the Digby Gut, a distance of about 60 kilometers. In 1960, a tidal dam was built across the estuary near Annapolis Royal, which limited the tidal exchange upstream of the dam (Jessop 1976). This construction transformed the estuary upriver of the causeway from a vertically homogenous estuary with about a 10 m tidal range (similar to many around the Bay of Fundy) to a highly stratified salt wedge estuary with about 0.5 m tides (Daborn *et al.* 1979). This causeway protects about 1740 ha of reclaimed marshland from flooding (Daborn *et al.* 1979) and eliminated the need for the dykes beside the river channel that formerly performed this function. The estuary receives the combined fresh water discharges from the Annapolis River, Nictaux River, Fales River, and Black River at its upstream limit near Bridgetown; Bloody Creek, Round Hill Brook and numerous smaller streams flow into the estuary upstream of the causeway; and Bear River, Allain Creek and Moose River flow into the Basin on the seaward side of the dam. The total watershed area is 2408.4 km^2 (Gregory *et al.* 1993). The Annapolis River itself originates near Aylesford, N.S., and meanders southwestward through the Annapolis Valley for about 97 km before reaching Annapolis Royal, N.S. (Melvin *et al.* 1985).

The Annapolis Tidal Generating Station (TGS) was constructed at the Annapolis causeway between 1980 and 1984 and has been in operation since August, 1984 (Stokesbury 1987). The operation of this turbine increased the flow of tidal water upstream, and the tidal range on the upstream side of the causeway is currently between 0.5 m and 1.0 m. The normal operating head range for the turbine is 1.4 to 6.8 m (Douma and Stewart in Andrews and McKee 1991). Output at a head of 5.5 m is 17.8 MW, with a corresponding discharge of 408 m^3/s. Two fishways have been constructed to augment fish passage at the causeway. The old fishway is an open slot (4 m wide) located beside the sluice gates (Figure 2). The new fishway is 3 m wide and runs between the turbine forebay and the tailrace. Water depth in both fishways varies between 1.5 m and 3 m depending on the stage of the tide. Discharges through the fishways are 42.7 m^3/s and 10.1 m^3/s for the old and new fishways respectively, for a 0.3 m head (Stokesbury and Dadswell 1991).

1.3 Previous Fish-Related Research at the Annapolis TGS

To date, the fish-related impact assessments of these alterations have been focused on the impact of the generating station, i.e. turbine mortality (e.g. Stokesbury 1987, Hogan 1987), fish passage at the causeway (e.g. Ruggles and Stokesbury 1990, Gibson and Daborn 1993, Gibson 1996a), fish diversion studies (e.g. McKinley and Patrick 1988), and comparative stock assessments (e.g. Melvin *et al.* 1985, Dadswell and Themelis 1990a,b, Gibson and Daborn 1995, Gibson 1996b). Research prior to 1991 was reviewed by Andrews and McKee (1991), and more recent research by Gibson (1996b).

The American shad has been used as the model species for most of this research. This is appropriate due to its seasonally relatively high abundance (increasing the feasibility of research), its importance supporting recreational and commercial fisheries, and the general decline of the species throughout eastern North America (Rullifson 1994). During studies of the effectiveness of fish passage facilities (Stokesbury 1987, Gibson 1996a) and young-of-the-year (YOY) shad turbine mortality (Stokesbury 1987), data for other YOY clupeids (alewife, blueback herring and Atlantic herring) were used to supplement limited YOY American shad catch data, supplying some information about these species. Data for other species captured during these studies have not been analyzed in any great detail.

The research on American shad and the Annapolis TGS has been both interesting and informative. The need for population-level impact assessments was recognized prior to the construction of the turbine, leading to assessments of the American shad stock during 1981 and 1982, before the turbine came on-line. These assessments were intended to serve as a baseline to assess long-term population change as a result of the construction of the turbine. At that time, the American shad stock was characterized as a long-lived, slow growing population with an annual spawning run size of about 100,000 to 150,000 individuals (Melvin *et al.* 1985). To date, four post-operational stock assessments have been carried out and provide a basis for the comparison. Studies in 1989 (Dadswell and Themelis 1990a) and 1990 (Dadswell and Themelis 1990b) characterized the stock at the time the first generation of juveniles to have been impacted by the turbine were expected to return to spawn. Studies in 1995 (Gibson and Daborn 1995) and in 1996 (Gibson 1996b) characterized the stock at about the time that the second generation of impacted shad would be returning to the river to spawn. While differences in sampling methods have so far precluded rigorous quantitative comparisons between these assessments (Gibson 1996b), it is evident from these studies that many population characteristics have changed since the construction of the turbine. These changes include decreases in mean age, mean length, maximum age observed, maximum length observed, lifetime fecundity, and Von Bertalanffy's theoretical maximum length; and increases in age at first spawning, instantaneous mortality rates and Von Bertalanffy's growth coefficent. Changes in stock size are not yet quantified.

Turbine mortality estimates have been obtained for both adult American shad (Hogans and Melvin 1986; Hogans 1987) and YOY clupeids (Stokesbury 1987). Using radiotelemetry methods, Hogans and Melvin (1986) reported an adult, American shad turbine mortality estimate of 46.3%, based on a sample size of 20 test fish. This study was criticized due to the small sample size, high control mortality, and the use of both pre and postspawning shad as test specimens (Andrews and McKee 1991). Hogans (1987) reported the results of a similar study that indicated that 21.3 % ! 19.8% (95 % C.I.) of adult, post-spawning American shad do not survive passage through the turbine. This study, addressed some of the criticisms of the previous study, although the sample size remained small (26 test fish). While Andrews and McKee (1991) considered the later estimate to be more “reliable”, Dadswell and Rulifson (1994) suggested that differences in operating efficiency between 1985 and 1986 may have caused the between year differences in the turbine mortality estimates, in addition to experimental procedure.

During the 1986 study, samples of 4 to 6 fish were passed through the turbine on five occasions. Turbine mortality was calculated as a percentage for each of the five experiments and the overall estimate was calculated as the mean mortality in these five experiments, accompanied by a confidence interval calculated under the assumption of a normally distributed response. This approach has several shortcomings: all fish do not carry equal weight in the final estimate (a fish that is part of a small sample has more leverage than a fish that is part of a large sample), and that symetrical confidence intervals are inappropriate for this kind of data (mortality cannot be negative). Treating the live dead status of each fish as the outcome of a binomial experiment alleviates these problems, and slightly reduces the turbine mortality estimate to 19.5% (bootstrapped 95% confidence interval: 0 to 37.5%).

Studies of YOY clupeid turbine mortality during 1985 and 1986, published in several forms: Stokesbury (1985), Stokesbury (1986), Stokesbury (1987) and Stokesbury and Dadswell (1991), suggest that mortality of juvenile clupeids passing through the turbine was 46.3 % during these years. This estimate is probably biased high (Andrews and McKee 1991) due to mortality problems associated with the fish collection techniques. During a fishway utilization study, Gibson and Daborn (1993) were unable to distinguish between mortality caused by the plankton nets used to sample the fishways and turbine tailrace and mortality associated with passage through the turbine. Gibson (1996a) suggested that the duration of the control experiments to quantify sampling mortality (30 and 60 minutes) was too short given the length of time a fish would have been expected to have been in the net (3.1 hours; 95% C.I. = 2.45 to 3.83 hours). This would have lead to an over-estimation of turbine mortality during these studies.

The studies of fishway utilization by migrating *Alosa* have also been published in a number of forms. Stokesbury (1985), Stokesbury (1986), Stokesbury (1987) and Stokesbury and Dadswell (1989), report the results of the 1985 and 1986 studies during which less than 2 % of their juvenile *Alosa* catch at the causeway came from the fishways. These results imply that the fishways do not play an important role in the migration of juvenile *Alosa* to the sea. Gibson and Daborn (1993), Gibson and Daborn (1995) and Gibson (1996) report the results of field studies in 1993 and 1994. Using modified ichthyoplankton nets they found that 23.3 % and 19.4 % of the juvenile *Alosa* catch came from the fishway located nearest the turbine, implying that the fishway may play a more important role in the passage of *Alosa* than suggested by the earlier studies. Still, the majority of outmigrating YOY *Alosa* apparently did pass through the turbine. While the fishways may not be important passages for larger fish that prefer open water, they do appear to be important for species such as Atlantic silversides and YOY Atlantic herring that approach the station by following the shoreline (Gibson 1996). As part of a fish diversion study, McKinley and Patrick (1988) and McKinley and Kowalyk (1989) found greater concentrations of fish near the sluice gates than in the turbine forebay, leading Andrews and McKee (1991) to suggest that the fishway located near the sluice gates played an important role in downstream fish passage. None of the studies that monitored fish passage at this location found evidence to support this conclusion (Dadswell and Stokesbury 1989; Ruggles and Stokesbury 1990; Gibson 1996a).

Behavioral guidance systems have previously been tested at the Annapolis Tidal Generation Station. McKinley and Patrick (1988) experimented with the use of light and a mechanical fish pulser to deter adult and juvenile clupeids away from the turbine intake, and the use of a fish drone to attract fish towards the new fishway. Light was found to be ineffective due to limited light penetration, and the drone ineffective due to high background noise from the turbine and fishway. The mechanical fish pulsers appeared effective in deterring adult shad, and were tentatively (due to low numbers of fish) thought to be effective for juveniles. McKinley and Kowalyk (1989) reported the results of a study to test the effectiveness of a sound fish deterrent system that utilized 4 fish pulsers. They concluded the system was about 70% effective for adult American shad and gaspereau, and 48 - 66% effective for juvenile gaspereau. The pulsers were therefore used in the following years, but their reliability in salt water limited their utility as a long-term solution (Terry Toner, personal communication).

2.0 AN ASSESSMENT OF A HIGH-FREQUENCY SOUND FISH DIVERSION SYSTEM AT THE ANNAPOLIS TIDAL GENERATING STATION

2.1 Use of Ultrasound to Divert Fish

Ultrasound has been demonstrated to elicit avoidance responses in alewives (e.g. Dunning *et al.* 1992), blueback herring (e.g. Nestler *et al.* 1992) and American shad (e.g. Mann *et al.* 1997) in cages and in pens. These kinds of experiments, led to the development of behavioral guidance systems using ultrasound, that have been tested at several power plants with varying degrees of success. At the Richard B. Russell Dam, Nestler *et al.* (1992) concluded that ultrasound could be used to divert blueback herring from areas from which they could be entrained, but that responses to the signal varied between day and night. Blueback herring eventually became accustomed to the sound, leading to the suggestion that ultrasound would be most effective in situations where fish would be exposed for less than one hour. A pulsed, ultrasonic deterrent system (122 to 128 kHz at a pressure level of 190 dB re: 1*u*Pa, 1 m from the source) was tested at the James A. Fitzpatrick (JAF) Nuclear Power Plant cooling water intake. Ross *et al.* (1993) reported a reduction in impingement of up to 87% at this location with the diversion on, and found that environmental variables such as wind direction, time of day and water temperature affected the rate of impingement. Ross *et al.* (1996) conducted a follow-up study that utilized an improved deterrent system, and concluded that the system should be 87% effective in most years.

The Public Service Electric and Gas Company (PSE&G) has conducted extensive studies on the feasibility of using sound to deter fish from vicinity of the cooling water intake structure at the Salem Generating Station, the results of which are summarized by Popper (1999). In an initial set of cage experiments in 1994, nine species of fish were evaluated for responses to 21 one-half octave frequency bands, ranging from 100 Hz to 145 kHz, at three sound pressure levels, using two waveforms. During a second phase of the cage experiments one-tenth octave bands were used to determine if some species respond better to a very narrow range of frequencies. These tests indicated that American shad, blueback herring and alewife exhibited strong startle responses to ultrasonic signals and moved away from the signal. Results for other species were not as clear and most positive responses were to signals below 5000 Hz. Striped bass, white perch and spot showed weak, non-directional responses. Weakfish and bay anchovy showed startle responses, but usually without directional movements. Atlantic croaker showed startle responses and some directional movement. Studies utilizing an improved experimental design in 1998, showed a very limited response to sound signals by Atlantic croaker, bay anchovy and weakfish. The results of these experiments lead to the development of a deterrent system using two sounds: an ultrasonic "chirp" intended to deter *Alosa* species, and a sound consisting of a sequential presentation of two "chirps" (one at 476 Hz, the other at 2700 Hz) followed by a sound similar to that made by croakers. This system was tested *in situ* at the Salem plant cooling water intake. Limited reductions in impingement were found for bay anchovy, Atlantic silverside and possibly Atlantic croaker and weakfish. Reductions in impingement of alewife were not statistically significant, and for blueback herring were only significant in one of the three statistical tests that were used. Too few American shad were impinged to warrant analysis.

Using ultrasound to guide actively migrating fish to a by-pass at a hydroelectric dam is a different problem than repelling fish from the vicinity of a cooling water intake. Using the general specifications developed at the JAF, fish by-pass systems using high-frequency sound and by-passes located outside the sound field were tested at the Crescent and Vischer Ferry hydroelectric stations (Ross 1999). At the Vischer Ferry site, over 90% of blueback herring used the by- pass. At the Crescent site, the system was effective for YOY blueback herring, but wasn't for adults. Ross (1999) suggested that the effectiveness for adult blueback herring could be increased by moving the sound field. In this study, hydrology, time of day and condition of the fish were found to influence both the abundance of fish and the effectiveness of the ultrasonic barrier. In the absence of an easily accessible by-pass, both YOY and adult blueback herring passed through the sound field into the headrace canals. Even when a by-pass was provided, the fish apparently habituated to the sound and moved through the sound field (Ross 1999).

2.2 The Fish Diversion System

The fish diversion system tested during this assessment was a band-limited random noise signal projected into the turbine forebay by 4 transducers mounted across intake. The signal pulse was presented at a 33% duty cycle (0.5 seconds on, followed by 1 second off) with the majority of energy focused between 122 and 128kHz. The 160 dB threshold, above which avoidance responses have been observed to occur, was reached at a distance of 10 to 12 m from the intake face. A description of the equipment generating the signal, and the results of sound field measurements taken after the installation of the system, are included as Appendix I.

2.3 Methods

2.3.1 Experimental Design

The experimental design discussed below was selected with three questions in mind:

1. Does the fish diversion system effectively reduce fish passage through the turbine?

2. If fish passage through the turbine is reduced, do the fish move seaward through the fishways?

3. Given the close proximity of the new fishway to the turbine intake, there was some speculation that the fish diversion system might also reduce passage through the new fishway. In this event, would fish passage through the new fishway increase during the 1 to 1.5 hour period between the end of generation (when the diversion was turned off) and the start of flood sluicing?

We chose the number of fish of each species captured at each location during a generation cycle as the experimental unit for this assessment. While this decision limited the sample size, the alternative, to turn the diversion system off and on a number of times during a generation cycle, was rejected due to uncertainty about the independence of the samples and the time lag between fish moving downstream and being captured. Additionally, the rate of downstream passage is not randomly distributed throughout a generation cycle, leading to an increase in the complexity of the model.

Resources were available to monitor fish passage on 40 generation cycles (50% of the available generation cycles between September 7, 1999 (the first date available for sampling) and October 15, 1999 (the anticipated end of the project). In order to distribute the sampling effort more or less evenly throughout this period, we divided the series of generation cycles to be sampled into pairs, and randomly chose one cycle from each pair to be sampled. We then took the series of generation cycles to be sampled, divided it into pairs, and randomly chose to turn the diversion on during one cycle in each pair, and off during the other. As such, we initially planned to sample 20 cycles with the diversion system turned on, and 20 cycles with the diversion turned off. Eight additional generation cycles were added opportunistically throughout the project, and the diversion status randomly chosen for each of these cycles. Storm conditions and an equipment malfunction near the end of the project resulted in the diversion being turned off during more cycles (n=28) than it was on (n=20).

2.3.2 Modeling the Effectiveness of the Annapolis TGS Fish Diversion

Based on the previous studies of fish passage at the Annapolis causeway (Gibson 1996), we anticipated that environmental conditions (temperature, salinity, tidal range, time of day and seasonal migratory patterns) would markedly influence the number of fish captured on any given tide throughout the study. We therefore decided to approach this assessment in two ways: by modeling the catch at each location only as a function of the on/off status of the diversion system (Model 1), and by removing the effects of the environmental variables by including them as terms in the model prior to those for the status of the diversion (Model 2).

2.3.2.1 Notation:

P = routes of passage at the causeway, denoted by p, with 3 levels:
- n = the new fishway
- o = the old fishway
- t = the turbine tube (sampled in the tailrace)

I = the sampling location, denoted by i, with 4 levels:
- n = the new fishway
- o = the old fishway
- t_s = the south side of the tailrace
- t_n = the north side of the tailrace

t = the tide or generation cycle sampled

D_t = an indicator variable denoting the status of the diversion system on tide t, denoted d, with 2 levels:
- on = diversion system turned on
- off = diversion system turned off

E = the "effectiveness" of the diversion for a given species (defined below)

F_i = the "fishway factor" for a given species and fishway i (defined below)

$R_{p,d}$ = for a given species, the rate of passage through passage p, with diversion status d

$C_{i,d,t}$ = for a given species, the number of fish captured at site i, with diversion status d, on tide t

X_t = water temperature on tide t

S_t = salinity on tide t

R_t = tidal range on tide t

L_t = on tide t, proportion of the sampling period that occurred after sunset and before sunrise

2.3.2.2 Definition of "Effectiveness"

1.We defined the effectiveness of the sound barrier for diverting fish from passing through the turbine as the proportion of fish diverted:

For a given species, let E be the "effectiveness" of the diversion:

$$E = 1 - \frac{R_{\text{on}}}{R_{\text{off}}}$$

where R_{on} is the rate of fish passage through the turbine with the diversion turned on, and R_{off} is the rate of passage with the diversion turned off.

2. To assess whether fish that are diverted are able to find the fishways, we defined a "fishway factor" that relates the rate of fish passage through a fishway with the diversion turned on to the rate of fish passage with the diversion turned off:

For a given species, let $F_{s,p}$ be the "fishway factor" for fishway p:

$$F_p = \frac{R_{p,\text{on}}}{R_{p,\text{off}}}$$

where $R_{p,\text{on}}$ is the rate of fish passage through the fishway p with the diversion turned on, and $R_{p,\text{off}}$ is the rate of passage through fishway p with the diversion turned off.

3. Treating the duration of each net deployment at a given site as constant (the length of the generation period for tailrace sites and the length of the ebb tide for fishway sites), and assuming that fishing efficiency is constant at each site, the number of fish captured at each site during a sampling period is a measure of the rate of passage at that site during that period. The effectiveness of the sound barrier can therefore be estimated as:

$$E = 1 - \frac{\overline{C}_{\text{on}}}{\overline{C}_{\text{off}}}$$

where: $\overline{C}_{\text{on}}$ is the mean number of fish of a given species captured in the turbine tailrace with the diversion turned on, and $\overline{C}_{\text{off}}$ is the mean number of fish of that species captured in the turbine tailrace with the diversion turned off, and the fishway factor estimated as:

$$F_p = \frac{\overline{C}_{p,\text{on}}}{\overline{C}_{p,\text{off}}}$$

where $\overline{C}_{p,\text{on}}$ is the mean number of fish of that species captured in fishway p with the diversion turned on, and $\overline{C}_{p,\text{off}}$ is the mean number of fish of that species captured in fishway p with the diversion turned off.

From the above, it follows that both the effectiveness of the diversion and the fishway factor for a given species are transformations of the same quantities: the ratios of the mean number of fish captured with the diversion turned on to the mean number captured with the diversion turned off.

2.3.2.3 Model 1:

Assume that for a given species, the number of fish moving through passage p on tide t, $C_{p,t}$ is a extra-Poisson distributed random variable that is a function the passage (P), the on/off status of the diversion system on tide t (D_t) and a dispersion parameter (ϕ) that is the ratio of the variance of $C_{p,t}$ to the expectation of $C_{p,t}$:

Because the response variable is bounded ($C_{p,t}$ cannot be less than 0) and its variance is not constant (it is a function of the expectation of $C_{p,t}$), the use of a least squares approach for modeling these data would require a transformation of the response variable prior to fitting the model. The log transformation typically used for this kind of data has the disadvantage that a small quantity must be added to each value prior to transform because the log(0) is not defined. After such a transformation the statistical properties of the response are not easy to define. Additionally, a log tranformation will only correct the residual heteroscedasticity if the data are not over or underdispersed. An alternative approach, that of using log-linear models within a generalized linear modeling framework, overcomes these disadvantages (McCullagh and Nelder 1989). Because the catch data was not strictly Poisson distributed, we used quasi-likelihood with an extra-Poisson error structure that allowed estimation of the dispersion parameter simultaneously with the model coefficients. This approach does not change the parameter estimates, but results in more realistic estimates of their associated error than would be obtained under an assumption of a Poisson distributed error.

Catches at the two tailrace sites were different, although we expected the effectiveness to be the same at both these sites. We therefore removed between site differences in catchability by including site as a factor at the beginning of the model. The expectation of the catch $\mathrm{E}(C_{i,t,d})$ of each species at site *i*, on tide *t*, given diversion status *d* was therefore modeled as:

$$\mathrm{E}(C_{i,t,d}) = e^{\mu + \beta_i + \beta_{p,d} D_t}$$

where: μ is the grand mean of the natural logarithm of the catch of the given species,

β_i is the coefficient for site *i*,

and $\beta_{p,d}$ is the coefficient for the diversion status *d* at passage *p*.

As such, the quantity $\left(\beta_{\mathrm{t,off}} - \beta_{\mathrm{t,on}}\right)$ is the difference in the natural logarithms of the mean catch in the tailrace with the diversion off and the mean catch in the tailrace with the diversion on. From this quantity, the ratio of the number of fish captured with the diversion turned on to the number captured with the diversion turned off, and hence an estimate of the effectiveness of the diversion can be directly calculated. Similarly, an estimate of the fishway factor for the new fishway follows directly from the quantity $\left(\beta_{\mathrm{n,off}} - \beta_{\mathrm{n,on}}\right)$.

2.3.2.4 Model 2:

Model 2 is an extension of Model 1 to include a set of environmental variables that may influence the rate at which fish move downstream past the causeway. This approach is motivated by the fact that if some of the variability in the catch could be explained by the environmental variables, the standard error of the estimated effectiveness would be reduced. Environmental variables added to the model include the temperature, salinity, tidal range, and the proportion of the sampling period occurring after sunset and before sunrise proportion of darkness). These variables were assumed to affect all sampling sites equally. In the cases of temperature and salinity, we postulated that an optimal value for fish passage existed for each species, and that the rate of passage would decrease normally as the value deviated from the optimal. Once log transformed, a normal likelihood function can be re-parameterized to take a quadratic form, so the squares of the temperature and salinity terms were also included as variables in the model. Again, the site effects, and the effects of the environmental variables, are removed by including them at the beginning of the model.

For a given species, assume that the number of fish moving through passage *p* on tide *t*, $N_{p,t}$ is a extra-Poisson distributed random variable that is a function of the passage (*P*), the water temperature on tide *t* (X_t), the salinity on tide *t* (S_t), the tidal range on tide *t* (R_t), the proportion of darkness on tide *t* (L_t), the on/off status of the diversion system on tide *t* (D_t) and the dispersion parameter (ϕ).

For the reasons previously stated, the quasi-likelihood approach discussed under Model 1 was also used to model the data:

$$\mathrm{E}(C_{i,t}) = e^{\mu+\beta_i+\beta_r R_t+\beta_l L_t+\beta_{x1} X_t+\beta_{x2} X_t^{\,2}+\beta_{y1} S_t+\beta_{y2} S_t^{\,2}+\beta_{p,d}\, D_t}$$

where: μ is the intercept,

β_i is the coefficient for the catchability at site *i*,

β_r is the regression coefficient for the variable "tidal range",

β_l is the regression coefficient for the variable "proportion of darkness",

β_{x1} is the regression coefficient for the variable "temperature",

β_{x2} is the regression coefficient for the variable "temperature squared",

β_{y1} is the regression coefficient for the variable "salinity",

β_{y2} is the regression coefficient for the variable "salinity squared",

and $\beta_{p,d}$ is the coefficient for the status of the diversion *d* at passage *p*.

This model was fit for each species individually. After fitting the model, the effectiveness of the diversion, and the fishway factor follow directly as per model 1.

2.2.3.5 Other models

A variety of similar models were also fit to both the data and subsets of the data, that resulted in minor differences in the coefficient estimates. These did not appreciably change the interpretation of the effectiveness of the diversion system. The two models discussed above were therefore chosen for presentation in this report.

Other models fit to the data include:

1. Models 1 and 2 fitted to subsets of the data representing each sampling site individually.
2. Models 1 and 2 fitted to the data after removing periods at the beginning and end of the time series if the species under investigation was not present at those times.
3. The effectiveness of the diversion system was estimated after removing the effects of the environmental parameters modeled as second and third order polynomials, for both the full data set and for subgroups of the data selected by sampling site.
4. The effectiveness of the diversion system was estimated after removing seasonal trends by including "tide number" as second and third order polynomial terms in the model, again using both the full data set and subgroups based on sampling site.
5. Environmental parameters were included as both linear terms and second and third order polynomials in the model, after removing seasonal trends, to adjust for short-term fluctuations in catch that were not a part of the seasonal trend (e.g. "proportion of darkness").

6. For each species, the means of the ratio of the new fishway catch to the tailrace catch with the diversion on was compared to that with the diversion off using a Mann-Whitney *U* test.

For species present in adequate abundance, Models 1 and 2 were fit after trimming subsets of the largest catches from the data, to determine if the effectiveness of the diversion changed as abundance increased. Only in the case of blueback herring did the interpretation of the effectiveness change when the largest catches were removed. The model output for blueback herring with the three largest catches removed is included in the results under the label “blueback herring trimmed”.

To test whether the use of the diversion system increased the catch in the new fishway during the period between the end of generation and the start of flood sluicing, we fitted Model 1 and Model 2 to a subset of the data representing the catch during that time.

2.3.3 Field Methods

Fish passage was monitored by sampling with modified ichthyoplankton nets in the two fishways and in the tailrace below the turbine. Nets deployed in the tailrace (Figure 3) were 1.0 m in diameter and consisted of three sections: a 2.0 m long section, cylindrical in shape, made of 1 cm mesh nylon netting; a middle section made of 2 mm Nitex net, 3 m in length, tapering from 1 m to 17.8 cm during the final meter, and the collector. The collectors were 1.75 m long, and were constructed with Spandex cloth fitted over 0.5 m diameter, aluminum cylinders (0.75 m in length). Entrances to the collectors were funnel-shaped to keep fish from escaping from the net. The tail ends of the collectors were designed so that they could be opened and closed with drawstrings allowing them to be emptied. Nets deployed in the fishways were of similar design, but were 0.75 m in diameter.

The sampling protocol over a generation cycle typically went as follows:

1. The net was deployed in the new fishway just as the flow through the fishway turned seaward.
2. The net was deployed in the old fishway shortly thereafter (the time lapse was usually about 10 minutes).
3. The two tailrace nets were suspended from the boom line in the tailrace, about 50 m downstream of the turbine, just before the start of generation.
4. If required, the diversion system was turned on c. 5 minutes before the scheduled start of generation.
5. Fish were removed from the tailrace nets at pre-determined intervals throughout the generation period, based on the requirements of the mortality experiment.
6. About 0.5 hr. before the scheduled end of generation, fish in the net in the new fishway were removed, and the net re-deployed.
7. About 0.25 hr. before the scheduled end of generation, the tailrace nets were emptied for the final time.

8. If required, the diversion system was turned off immediately after generation stopped.
9. The old fishway net was pulled about 0.5 hr. after the end of generation.
10. The new fishway net was pulled for the final time just as water began to flow upriver through the fishway.

When a high catch was anticipated, the fishway nets were checked more frequently to reduce fish mortality caused by extended time in a crowded net. Fish were then identified, enumerated, and, if alive, released. The tailrace nets were typically fished for the duration of generation (c. 5.5 hr.), and the fishway nets for the duration of the ebb tide in the headpond (c. 8 hr.).

Water temperature and salinity were measured in the mouth of the new fishway using a Y.S.T SCT-1000 meter just prior to setting the nets on each tide. The proportion of darkness during each sampling period was calculated using the time of sunrise and sunset predicted using ASTRONOMY LAB 1.13. Tide range was predicted using harmonic constants for Digby, N.S., using the computer program TIDES 3.05.

The fishing efficiency of the tailrace nets was estimated by passing known numbers of dead fish (marked with stain) through the turbine and counting the number of these fish captured in the nets (Sorenson et al. 1998). Mortalities from previous net deployments were used for this purpose. This method would provide a reasonable estimate of the proportion of fish passing through the turbine that were captured in the nets under the assumption that the capture probability is the same for live and dead fish. This approach is reasonable if a large proportion of fish are killed by the turbine, or if live fish exiting the turbine are momentarily stunned or disoriented.

2.3 Results

2.3.1 Species, Quantity and Distribution of Captured Fish

During this study, nets were deployed and checked 447 times during 48 generation cycles between Sept. 7th and Oct. 21st, 1999. In total, just over 53,000 fish were captured, representing 27 taxa (Table 1). The Atlantic silverside was by far the most abundant (48,007 captured). Notable in this table, are the Meek’s halfbeak and the flying gurnard, which we believe to be first records for the Annapolis Estuary. This study was also the first time that four-beard rockling and bluefish were captured at the TGS, but these species are probably regular visitors to the Annapolis Estuary. Blackspotted sticklebacks have not been previously mentioned in studies at the TGS, but were probably abundant and mis-identified as fourspine sticklebacks.

Catches of 14 species were of a sufficient number to warrant analysis (greater than 50 fish). The relative abundance of these species varied throughout the study, and between sample locations. Blueback herring were captured most frequently in the tailrace (north side) and new fishway (Figure 4), and peaked at all four locations during the last week of

September. The majority of alewives (Figure 5) were captured in the new fishway, with peaks during the last week of September and mid-October. American shad were also most frequently captured in the new fishway (Figure 6), peaking coincidentally with alewife. No shad were captured in the old fishway. Fewer Atlantic herring were caught in the old fishway than at other locations (Figure 7), a species that was most abundant at the end of the study. Atlantic silversides were by far the most abundant fish at all sampling locations, the largest catch being 9000 fish in the new fishway in early September (Figure 8). Diel variability in the catch is evident in this graph during periods when the generation cycles occurred only during the day or at night. Blackspotted sticklebacks were present throughout the study period (Figure 9), while hake (Figure 10) were present only during the later part of the study. The majority of American eels were captured in the new fishway (Figure 11). Mummichogs were only abundant during a five day period between Sept. 26^{th} and Oct. 1^{st}, just after a heavy rainfall (Figure 12). Winter flounder, the majority of which were captured in the tailrace, were most abundant during the later part of the study (Figure 13). The other flatfish captured, the windowpane, was most abundant during the third week of September and the third week of October (Figure 14). Sea lamprey were not present in the study area until Oct. 16^{th} (Figure 15), after which their relative abundance was high. Pipefish (Figure 16) were abundant throughout the study period. The magnitudes of the catches of this species in the fishways deceased relative to those of the tailrace during the study, perhaps suggesting a behavioral change during this period. Butterfish were captured intermittently throughout the study period, the largest catches occurring at the beginning of October (Figure 17).

Dispersion parameters provide an indication of the randomness of the distribution of the fish: a value of 1 means the fish are randomly distributed, a value less than one means that fish are distributed more uniformly than random, and a value greater than one indicates that the fish distribution is clumped, i.e. the fish are present in shoals. The dispersion parameter for all clupeids was greater than 1 (Table 2). Winter flounder was the only species for which the dispersion parameter was less than 1 (Table 3). The effect of the dispersion parameter is to increase the standard error of the model parameter estimates as the dispersion parameter increases.

2.3.2 Environmental Data

Water temperature averaged 14.5 ^{o}C, decreasing from about 20^{o}C in early September to 10.1 ^{o}C at the end of the study (Figure 18). Salinity showed less of a seasonal trend, but higher variability on a daily basis (Figure 19). It averaged 27.3 ppt and ranged between 18.0 ppt and 33.5 ppt. The study spanned about 1.5 spring/neap tidal cycles, during which the tidal range varied between 4.7 m and 8.4 m (Figure 20). The proportion of darkness during the generation cycle (Figure 21) also varied somewhat systematically throughout the study, due to the daily precession of the time of high tide. Generation cycles occurring entirely during the day or night coincided with lower tidal ranges.

The influence of these variables on the rate of fish passage can be seen by examining the estimates of the Model 2 coefficients, presented in Tables 2 (clupieds) and 3 (nonclupieds). Of these parameters, the influence of daylight (proportion of darkness) was greatest, and was statistically significant at a 90% or higher confidence level for all species except sea lamprey, mummichog and butterfish. Of these, sea lamprey and mummichogs were only captured during brief periods that only provided slight contrasts in the "proportion of darkness" variable. Of the species for which "proportion of darkness" was significant, blackspotted stickleback was the only species for which the catch increased with increasing daylight. All other species tended to move past the causeway between sunset and sunrise. Catches of all clupeid species were positively correlated with tidal range. Statistically significant negative correlations were found between the catch and tidal range for pipefish, winter flounder and windowpane. Within the clupeids, salinity was only a useful predictor of the rate of fish passage for Atlantic herring. Within the non-clupeids, statistically significant negative correlations with salinity existed for the estuarine species: silversides, sticklebacks, pipefish and eels; and a statistically significant positive correlation for the marine species: butterfish. Temperature apparently influenced the rate of passage of alewives, Atlantic herring, Atlantic silversides, pipefish, winter flounder and windowpane.

2.3.3 Effectiveness of the Fish Diversion System

Estimates of the difference in the logarithm of the catch with the diversion on and the diversion off in the tailrace, estimated using Model 1, are shown in Figure 22. While none of these estimates are statistically significantly different from zero, together they suggest that the diversion may have some limited effectiveness for most species. When environment variables are included in the model (Model 2), the estimates of the difference are closer to zero and have smaller standard estimates (Figure 23). While still not statistically significant from zero, the estimates suggest an effectiveness of 35% for American shad, and 26% for blueback herring (after the 3 largest catches are removed from the data).

For the new fishway, estimates of the difference in the logarithm of the catch with the diversion on and the diversion off, estimated using Model 1, showed that the *Alosa* catch increased at this location when the diversion system was turned on (Figure 24). When environmental parameters were included in the model, the standard errors on these estimates decreased (Figure 24), resulting in factor estimates that are significantly different from zero for American shad and alewife (increases in the catch of 3.6 and 4.1 times when the diversion was turned on). As with the tailrace site, trimming the three largest blueback herring catches from the data changed the estimate of the factor, again suggesting a limited effectiveness for this species.

For the old fishway site, the standard errors of the diversion parameter estimates are large relative to the corresponding parameter estimates for all species except pipefish (Table 3). This is primarily due to the low catch of most species at this location (Table 1). As such, the data provide little indication about the effectiveness of the diversion at this site, other than it does not appear to be an important passage for most species either with or without the diversion system. In the case of pipefish, the catch at the old fishway decreased by a factor of about 3.5 times when the diversion system was turned off.

At the new fishway, the period after the end of generation and before the start of flood sluicing was sampled after 46 generation periods. The diversion was turned on during 22 of these generation periods, and off for the remainder. With the exception of Atlantic silversides, Atlantic herring and blackspotted sticklebacks, the catch of each species during this period was too small to warrant analysis (Table 4). Estimated using Model 2, the diversion coefficient for Atlantic herring was statistically different from zero at the 90% confidence level (Table 4), suggesting an increase in the catch by a factor of 3 times if the diversion was turned on during the preceding generation period. The diversion coefficient for this time period was not statistically significantly different from zero for Atlantic silversides or blackspotted sticklebacks.

2.3.4 Forks Lengths of Clupeids

Fork lengths were measured on a sample of clupeids captured during this study, the results of which are summarized in Table 5. All American shad, all but 2 alewives and one blueback herring are of a size suggesting they are young-of-the-year. Atlantic herring fell into two distinct size classes, the smaller averaging 53mm FL and the larger 109mm FL.

2.3.5 Fishing Efficiency of the Tailrace Nets

Fish were marked and passed through the turbine to estimate the fishing efficiency of the tailrace nets on three occasions, the results of which are shown in Table 8. Of 4170 test fish passed through the turbine, 10 were captured in the tailrace nets. Treating these data as the results of 3 binomial experiments yields an estimate of the fishing efficiency of 0.00298, or the 2 nets catch about one out of every 335 fish that pass through the turbine.

2.4 Discussion

The effectiveness of the diversion was evaluated at three sites, of which two, the new fishway and the tailrace, provided interpretable results. None of the previous studies at the TGS during which fish passage was monitored at the old fishway have suggested that this passage is important for the downstream passage of fish. This study supports these findings. No evidence was collected during this study that suggests that the ultrasound barrier, as tested, increases its importance.

Evaluating the effectiveness of the barrier at two locations provides a check against spurious results. If the diversion is effective and fish are able to find the new fishway, then an increase in the catch at the new fishway that is the result of the diversion system should be accompanied with a decrease in the catch in the tailrace. If the catch in the tailrace also increases when the diversion is on, then some other factor is probably influencing the outcome, such as an environmental parameter not included in the model, or chance.

Of the two models of the diversion effectiveness presented herein, Model 2 (environmental variables included) provides the more believable estimates. While the results from Model 1 for the tailrace suggest an effectiveness between 10% and 50% for many species (exceptions are Atlantic silverside, blackspotted stickleback, lamprey, winter flounder and mummichog), these estimates are accompanied by wide standard errors and are not significantly different from an effectiveness index of 0% at a confidence level of 95%. Additionally, in the cases of Atlantic herring, American eel, pipefish, windowpane, hake and mummichog, the decrease in the tailrace catch with the diversion turned on coincided with decreases in the catch in the new fishway. These contradictory results suggest that environmental variables may have played a more important role in determining the rate of fish passage than the on/off status of the diversion system for these species.

Overall, the Model 2 estimates have lower standard errors and are less contradictory than those from Model 1. Taken on the whole, these results suggest a limited effectiveness for members of the genus *Alosa*, but not for other species. Decreases in catch in the tailrace with the diversion on (35% for American shad and 26% for blueback herring estimated using the trimmed data set) were not significantly different from 0% at the 95% confidence level. Increases in the catch in the new fishway with the diversion on for American shad (3.6 times) and alewife (4.1 times) were significant. In the new fishway, the catch of blueback herring with the diversion on only appeared to increase (1.7 times) after the three largest catches were removed. These are the only species for which statistically significant diversion effects were found, or for which non-significant effects were found that for which the tailrace and the new fishway estimates were non-contradictory.

During this study we estimated that the two tailrace nets together would capture about 1 out of every 335 fish that pass through the turbine. Based on experiments of Gibson (1996a), a 0.75 m net fished in the new fishway would catch 6% of the fish utilizing that pathway. In conjunction with the catch at each location (Table 1), these results suggest that c.57% of Atlantic silversides use the new fishway to move seaward (treating the role of the old fishway as inconsequential). Estimates on the new fishway usage by *Alosa* range for 3.5% for blueback herring to 15.6% for alewife.

Based on these data,the increase in the *Alosa* catch in the new fishway with the diversion on is of about the same of magnitude as the decrease in the *Alosa* catch in the tailrace when the diversion is turned on. For example, if 5% of the fish use the new fishway and all diverted fish find the fishway, a diversion effectiveness of 25% should increase the catch in the fishway by 5.75 times. Estimates of the effectiveness and new fishway factor for *Alosa* obtained during this study fall near these values. This result suggests that for *Alosa* diverted fish either moved seaward through the new fishway, or eventually acclimated to the sound and moved seaward through the turbine.

For most, species the catch in the new fishway during the period between the end of generation and the start of the flood tide was too small to test whether diverted fish move past the causeway turning this period after the diversion is turned off. This observation itself suggests that the intention of deterring fish from passing through the turbine during generation with the expectation that they will move seaward through the new fishway after the end generation is unrealistic. Additionally, Gibson (1996a) found that migratory species such as *Alosa* tend to move seaward past the causeway during the first part of the generation period. In order for this strategy to be viable, *Alosa* would have to be deterred for up to 5.5h. Habituation to ultrasound may occur in alewife within 0.3 to 3h, depending on the signal (Dunning *et al.* 1992). Additionally, in the studies at the Vischer Ferry and Crescent hydroelectric generating stations, Ross (1999) reported that a readily accessible by-pass was necessary for the ultrasound barrier to be effective.

The results of this study also lead to the suggestion that the effectiveness of the diversion system may be partially dependent on the abundance of target species. This is seen by comparison of the results when the blueback herring data are modeled on the whole, and when the three largest observations are trimmed from the data. When larger shoals of fish are actively trying to move past the causeway, fish within the shoal may be unable to respond to the signal due to the close proximity of the surrounding fish. These fish could be pushed into the sound field to the point where they are entrained by the turbine. While our data are insufficient to fully investigate this hypothesis, they support similar observations at the Crescent hydroelectric generating station that effectiveness is reduced in the presence of large shoals (Ross 1999).

Alosa at the Annapolis TGS are thought to move into the turbine forebay on its south side (McKinley and Kowa1yk 1989). The new fishway is located on its north side. If an ultrasound barrier is to be part of a fish deterrent system at the TGS, its effectiveness may be increased by angling the barrier towards the new fishway. In this way, fish may be expected to approach the barrier on an angle, reducing the tendency of fish to be pushed through the barrier by surrounding fish. Additionally, fish avoiding the sound may be directed towards the new fishway. Both these hypotheses are based on the assumption that the avoidance response is in a direction orthogonal to the sound field.

As anticipated at the beginning of the study, ultrasound did not prove effective for non-*Alosa* species at the Annapolis TGS. These results reflect those at the Salem Generating Station (Popper 1999). During the *in situ* portion of that study, Atlantic silversides showed an unexpected decline in impingement in response to the ultrasound signal. While the response was thought to be spurious, it prompted the conclusion that further investigation was warranted. In this study, the statistically insignificant increase in the Atlantic silverside catch in the new fishway with the diversion turned on was accompanied by an increase in the catch in the tailrace, and therefore does not provide evidence that the ultrasound barrier was partially effective for this species. Rather, sampling was probably not randomly distributed across the set of variables that determine the rate of passage for this species.

Environmental conditions and the composition of the catch during this study are sufficiently different from other years to warrant mention. For example, in comparison with 1994 (Gibson 1996a), the data set used to develop the sampling protocol and statistical models, water temperature in 1999 was 3.2 °C warmer at the start of the study on Sept. 7th. Water temperature averaged 2 °C warmer throughout September. Salinity in 1999 was 32 mg/l in early September, in comparison with 29 mg/l in 1994, suggesting a lower flushing rate during late summer in 1999. The effects of these changes are reflected in the catch. Bluefish, fourbeard rockling and pollock are regularly found in the outer Bay of Fundy, but 1999 is the first year that these species were captured while monitoring at the Annapolis causeway. The abundance of species present in both years was also different. For example, for the marine species Atlantic herring and hake spp., the catch-per-unit-effort (CPUE) in the tailrace was 7.9 and 5.9 times that of 1994 for the same time period. Conversely, the CPUE of *Alosa* was much lower: 7.3% for American shad, 14.7% for alewife and 29.6% for blueback herring. The effect of these changes was to substantially reduce the statistical power of the experiment for *Alosa* (the target species in this study, and the species for which ultrasound was expected to act as a deterrent), while increasing the statistical power for other species (for which no effect was the anticipated outcome).

The presence of a Meek’s halfbeak (1st Nova Scotia record) and a flying gurnard (1st Bay of Fundy record), both strays from the southern USA, is suggestive of wider scale environmental differences between 1999 and other years that have been monitored.

Of the environmental variables included in the model, the proportion of darkness was the most important determinant of the rate of fish passage at the TGS (statistically significant for all species except butterfish and mummichog). All species except blackspotted stickleback tended to move past the causeway at night. The significance of the other environmental variables differed between species. While not quantified during this study, these variables probably influence a fish’s response to a stimulus, and hence the effectiveness of a diversion system. As discussed by Popper and Carlson (1998), the “motivational state” of fish has been shown to vary throughout the year. While the timing of this study was appropriate for YOY *Alosa*, the target species in this study, since they are only thought to be present at the TGS during the fall. If a behavioral guidance system is to be utilized at the Annapolis TGS, then monitoring during the late spring and summer will be also required to determine whether the system is also effective for adult *Alosa*.

From the results of this study, and others (e.g. Popper 1999, Ross 1999), it is apparent that fish deterrent systems that utilize ultrasound require site specific design and tuning to be effective. However, given the successes reported in the literature, the potential of ultrasound as a fish deterrent is still intriguing, particularly in combination with other behavioral stimuli. In the case of the Annapolis TGS, the system could potentially be improved in a number of ways. As previously discussed, angling the barrier to direct fish towards the new fishway could improve its effectiveness if fish passage though the new fishway is increased as a result. Fish passage through the new fishway during the brief period after the end of generation and before the onset of flood sluicing did not appear to increase with the use of the diversion system in this study. However, providing an alternate passage before the start of generation by keeping an extra sluice gate open prior to the onset of generation might reduce the number of fish upstream of the turbine at the start of generation, leading to an overall reduction of turbine passage, particularly if the effectiveness of the diversion system decreases with increased fish abundance, as suggested by the blueback herring results. This suggestion is made without consideration of other factors (higher headpond water levels, erosion, etc.) that could preclude its feasibility.

In summary, a fish deterrent system utilizing ultrasound has the potential to reduce the incidence of *Alosa* passage through the turbine at Annapolis, but further consideration should be given to the design of the system. It does not appear to be effective for other species captured during this study, and some other system, perhaps also based on sound, may be required for these species. Prior to installing such a system, considerable experimental testing will be required to select a stimulus that will be effective for these species. Ultimately, the effectiveness of a diversion system should be evaluated at the population level. For example, if turbine mortality is low for a species characterized by a short life span, high fecundity, and density-dependent population regulation, a reduction in the incidence of turbine passage will not appreciably change the abundance of fish in this population, as discussed in the next section.

3.0 ESTIMATION OF TURBINE MORTALITY USING LOGISTIC REGRESSION.

3.1 Introduction

Accurate estimates of turbine mortality at hydroelectric generating stations are fundamental for fish-related impact assessments for these facilities and for the development of appropriate mitigation of these impacts. While turbine mortality studies are numerous, the results of these studies are often conflicting (Mathur *et al.* 1994) making it difficult to generalize between species or locations. Inferences about the impacts of tidal power are even more difficult, as the fish communities are often larger, more complex and more valuable (Dadswell *et al.* 1985).

Turbine mortality studies can be loosely divided into two groups: those that use naturally entrained fish (e.g. Stokesbury and Dadswell 1991, Navarro *et al.* 1996) and those that use fish released into the turbine tube (e.g**.** Mathur *et al.* 1994, Hogans 1987). In the first case, fish are captured using nets in the turbine tailrace, and turbine mortality is estimated using either live/dead criteria (e.g. Ruggles *et al.* 1990) or by autopsying dead fish (e.g. Stokesbury and Dadswell 1991). Estimates are reported either corrected for handling mortality (Stokesbury and Dadswell 1991) or not (Navarro *et al.* 1996). Introduced fish may be either recaptured in nets in the turbine tailrace (Dubois and Gloss 1993), followed via radio tags (Hogans 1987), or recaptured using some other method (e.g. Heisey *et al*. 1992). Mortality estimates are often reported as acute (or immediate or short term: the proportion dead at or near the time of capture) or delayed (or long term: the proportion dead after holding the fish for some time period, typically 12, 24 or 48h).

Each of these methods has its relative strengths and disadvantages, based on both scientific and practical considerations. When fish are abundant, estimation of acute mortality using naturally entrained fish captured in nets in the tailrace is a relatively easy and cost effective method of obtaining data. However, acute mortality is difficult to interpret in the context of the overall effect of the turbine on a fish stock. Additionally, if control fish are used to correct for mortality caused by capture and handling, determining an appropriate duration for the control experiments is difficult because the length of time the entrained fish are in the net is unknown (Gibson 1996a). The use of autopsies to determine the cause of death is also problematic, because injuries caused by nets may appear similar to those caused by turbines (Gibson 1993).

Introducing test fish into the turbine tube provides better experimental control than the use of naturally entrained fish, but increases the handling of the fish, and thus the stress to the fish associated with handling. If effects of stress are cumulative and culminate in increased mortality, the individual effects of handling, turbine passage and recapture can be difficult to distinguish. For example, at a generating station on the Sissibou River, Nova Scotia, Ruggles and Palmeter (1989) reported a turbine mortality estimate of 14% for naturally entrained alewife, versus a turbine mortality estimate of 66.5% using alewives released into the the turbine tube and recaptured in a tailrace net. A significant improvement to the release method appears to be the HI-Z Turb'N tag-recapture technique (Heisey *et al.* 1992). Fish are recaptured via a tag that buoys the fish to the surface of the water, after which fish are scooped out of the water using a bucket. This method substantially reduces the mortality associated with recapture in nets, a factor that has lead to the overstatement of turbine mortality in many studies (Mathur *et al.* 1994).

In this report we suggest that the use of naturally entrained fish captured in nets in turbine tailrace can produce reasonable turbine mortality estimates with a slight change in the methods used in many studies. Mortality associated with the capture of fish with nets in the turbine tailrace increases with time in the net (Stokesbury and Dadswell 1991). Our approach is simply to vary the duration of the net deployment, and to model the survival of fish as a binomial response that is a function of the length of the net deployment. The Y-intercept is then an estimate of the survival of fish that haven't spent time in a net, i.e. turbine mortality. This approach has many advantages: handling of fish is reduced, it can be applied in situations where test fish are not readily available (e.g. flatfish in this study), and estimates can be obtained for several species simultaneously without an increase in the required effort. Additionally, the method can be easily integrated into sampling for some other purpose, as in this study. We present turbine mortality estimates for several species passing through the STRAFLO turbine at the Annapolis Tidal Generation Station, Annapolis Royal, Nova Scotia.

3.2 Methods

3.2.1 Field Methods

Sampling to test this turbine mortality estimation technique was integrated into the sampling to estimate the effectiveness of the fish diversion system. As such, fish that were captured in the tailrace during the assessment of the fish diversion system were used as test specimens for modeling mortality. The equipment and sampling protocol are as previously described (Section 2.3.3). During each monitored generation period, fish were removed from the tailrace nets at pre-determined intervals, ranging from every 0.25 h to 5 h). The catch was identified, enumerated and the live/dead status of each fish recorded, about 10 minutes after removing the fish from the net. The resulting data set was used to estimate acute mortality for each species for which sufficient numbers were captured.

To estimate delayed mortality, fish were transferred to a 1m x 1m x 2m holding box (plywood with a Vexar bottom and top) anchored in a cove near the TGS. The live/dead status of these fish was recorded at the start of the next generation cycle (between 7h and 13h after their capture). Fish that were in poor condition at the end of this period were counted as dead. While an element of subjectivity is introduced by this decision, the effect of any resulting bias would be to increase the mortality estimate, hopefully providing a more precautionary estimate as a result.

3.2.2 The Turbine Mortality Model

Assume that fish captured in the tailrace that are dead, either at the time of capture, or at some time shortly thereafter, died either as a result of turbine passage, capture in the net, or some interaction of these variables. Since the duration of the net deployment is varied, mortality caused by time in the net can be removed using a regression model. As such, treating the live/dead status of each fish as a binomial response (0 = alive, 1 = dead), turbine mortality for each species was estimated as the intercept (β_0) in the logistic model:

$$E(M) = \frac{1}{1 + \exp(\beta_0 + \beta_1 D)}$$

where: $E(M)$ is the expectation of the probability that a fish is dead, D is the duration of the net deployment, and β_0 and β_1 are the linear regression coefficients. As such, all mortality that is not a function of the length of the net deployment is attributed to the turbine. This model can be fitted to both acute mortality and delayed mortality data sets. This model has the form of a generalized linear model (McCullagh and Nelder 1989). Parameter estimates for this model can be obtained with any statistical package that includes generalized linear model routines (e.g. SAS or S-Plus).

Fish that die as a result of being captured die as a result of either entering the net, while in the net or when being removed from the net. The above model only encompasses mortality that is a function of time in the net, and therefore produces estimates that are biased high. In instances where the resulting estimate of turbine mortality is low, the effect of this bias may be inconsequential. Where the resulting turbine mortality estimate is high, estimates should not be believed without quantification of this bias.

3.3 Results

It was apparent at the start of the livebox experiments that the combined mortality from handling and the turbine was higher than anticipated. Attempts with a beach seine to collect test fish for a handling mortality experiment were unsuccessful, due to the time of year (early October). To determine whether fish captured in an ichthyoplankton net in the new fishway were suitable for this purpose, we removed fish from this at half hour intervals and placed them in the livebox. After 12h, survival of robust species such as pipefish and lamprey was 100% (Table 8). Survival of less robust species was lower (e.g. 35% for Atlantic herring).

Of the 27 species captured during the assessment of the diversion system, 12 were of sufficient abundance in the tailrace to warrant analysis (Table 9). Estimates of acute mortality for these species range from 0.0% for sea lamprey and blackspotted stickleback, to 23.4 % for American shad. The proportion of dead fish increased rapidly with increased net deployment duration for clupeids (Figure 26), but more slowly for more robust non-clupeids (Figures 27 and 28).

Believable estimates of turbine mortality obtained from the livebox trials were obtained for alewife (Figure 29), sea lamprey (Figure 31), Atlantic silversides (Figure 30), pipefish (Figure 30), winter flounder (Figure 30) and windowpane (Figure 30). Estimates for these species ranged between 0 and 6.3% (Table 10). Estimates for other species are considered unreliable due to low numbers of test fish (sticklebacks, American shad and butterfish), high mortality associated with capture (American shad, blueback herring, butterfish and hake) or both.

Eight species of fish captured in the tailrace were of too low abundance to warrant analysis species. Of these species, the proportion dead ranged between 0% for cunner, smooth flounder and longhorn sculpin, to 33% for rainbow smelt (Table 11).

3.4 Discussion

Intuitively, it appears easier to over-estimate turbine mortality than to under-estimate it, because many of the biases within a study lead to over-estimation. For example, if partial capture methods are used, net avoidance by live fish will lead to an over-estimation of turbine mortality. Failure to completely account for capture mortality will also bias turbine mortality estimates upwards. High capture mortality introduces a similar bias (Ruggles *et al.* 1990), perhaps due to the cumulative effects of handling, turbine passage and capture. Given the nature of these biases, low estimates are more believable at face value, whereas methods must be critically examined before higher estimates can be accepted.

Separation of collection mortality from that due to turbine passage is one of the fundamental difficulties researchers must overcome (Heisey *et al.* 1992). In this study, collection mortality is only partially quantified, since mortality as a fish enters the net, or when the fish is being removed from the net was not included in the model. For turbine mortality estimates that are small, the extent of this bias must also be small, and therefore the estimates are believable, even if biased high. For estimates that are higher the extent of this bias is unknown, and the resulting estimates should therefore not be believed without further substantiation. As such, all estimates presented in this report are probably higher than the true rate of turbine mortality. The approach presented in this report provides a framework for estimating handling mortality from other sources. For example, if control fish are placed in the nets for varying lengths of time, the offset between the control and experimental groups would provide an estimate of turbine mortality.

Separation of acute and delayed mortality provides some indication about the source of mortality from the turbine. Mortality from mechanical strike, shear and cavitation is probably evidenced in the acute mortality estimate. The low acute mortality estimates obtained in this study are in agreement with pre-operational predictions of turbine mortality at the TGS (Collins 1984). Mortality from other sources may be evidenced in the delayed mortality estimate. Pressure injuries such as pinholes in the gas bladder and burst blood vessels, or scale loss in species with deciduous scales, may not result in mortality until some time after turbine passage. While pressure change was not anticipated to be a significant source of mortality at the TGS due to its low operating head (1.4 to 6.8 m), Stokebury and Dadswell (1991) reported that 64.5% of injuries to juvenile clupeids passing through the turbine were caused by pressure change. Burst blood vessels in the eye accounted for 69% of pressure related injuries, although this injury is also caused by capture in nets (Gibson and Daborn 1993). Turbine mortality estimates from hydroelectric generating stations with greater heads suggest that this estimate of pressure damage may be high. For example, at the Safe Harbour Hydroelectric Station (c.18m head), Heisey *et al.* (1992) estimated turbine mortality for American shad from all sources to be <5% for mixed flow and Kaplan turbines. At the Fourth Lake Generating Station on the Sissibou River, N.S. (c.22.7m head), Ruggles et al. (1990) reported turbine mortality of naturally entrained juvenile alewife to be 14%. Because of these inconsistencies, until an experiment at is conducted at the TGS that adequately separates injuries caused by capture and turbine passage, this issue should be considered unresolved.

While considerable effort has been focused on the estimation of turbine mortality, the interpretation of mortality estimates has received less attention. A turbine mortality estimate should be interpreted as the probability of an individual fish surviving turbine passage. As such, it effects the life expectancy of a fish and its lifetime fecundity. When a fish is removed from a population, it ceases to act as a competitor, and may therefore increase the growth and survival of other members of the population in response to its death. The effect at the population level is the combined effects from all fish and will vary depending on factors such as the proportion of fish passing through the turbine, when turbine passage occurs relative to reproduction, when it occurs relative to compensatory mortality, and life history characteristics of the species. It follows that a species with a short life span, high fecundity and that reproduces prior to passing through the turbine could potentially withstand a comparatively high turbine mortality rate with little impact at the population level. Conversely, for a species that has a long life span, low fecundity, reproduces late in life and passes through the turbine several times prior to reproducing, a comparatively low turbine mortality rate could mean extinction. With increasing interest in the development of tidal hydroelectric generation, an increasing number of species are being exposed to turbines. It remains to develop a framework that allows not only estimation of turbine mortality for a given species based on design parameters of turbines, but to develop a framework that will allow prediction of the population response based on the general biology of the fish.

4.0 CONCLUSIONS

Based on the results of this study, it appears that the ultrasound fish diversion system installed at the Annapolis TGS was partially effective for *Alosa*, and as expected, was not effective for other species. The effectiveness for *Alosa* should improve, if fish are directed towards the new fishway, perhaps by angling the barrier. Under the assumption that multi-species solutions should be the objective at plants such as the Annapolis TGS, stimuli effective for other species need to be found, if a behavioral guidance system is to be used at Annapolis.

In the opinion of this author, behavioral guidance technology for fish is not sufficiently advanced for installation of a system for other species present at the TGS, without substantial effort to choose an appropriate stimulus for each species. Development of these technologies can be costly and are without a guarantee of success. Ultimately, the effectiveness of a fish diversion system should be measured using the population-level response to the diversion system. For a species with a high compensatory capacity and low mortality associated with passage at the causeway (fishway usage coupled with turbine mortality), the effect of diversion at the population-level would probably be negligible. Based on the peripheral results presented herein, Atlantic silverside is an example of such a species. Therefore an assessment of the risk, at the population-level, associated with passage at the causeway (for which accurate estimates of turbine mortality and fishway effectiveness are essential) should be conducted for each species to determine the necessity of diversion prior to developing the technology to divert the fish. After the risk associated with passage is evaluated, the benefit of diversion for each species can then be determined. Based on this kind of analysis, species for which diversion is expected to have a population-level effect can be included for development of a diversion system, without incurring the cost of developing diversion technology for species that may not appreciably benefit from the diversion.

Accurate estimation of turbine mortality is not a trivial undertaking, as evidenced by the conflicting results in many studies. The modeling approaches for estimating turbine mortality developed during this study appear to provide reasonable estimates of acute and delayed turbine mortality for more robust species such as sea lamprey and Atlantic silversides. An offset for capture mortality needs to be fitted to provide reasonable estimates for more delicate species such as the clupeids and butterfish, but believable estimates should be obtained once this offset is included. Fish collected in the new fishway are not suitable test fish for this purpose, although the seining methods developed for marking YOY *Alosa* in the Annapolis Estuary (Gibson 1996a) would povide suitable specimens, if the field project starting in by early August. Once turbine mortality estimates are obtained, the risk to the population can be evaluated using life history characteristics of the species. This information could then be used to choose species that would benefit appreciably from diversion away from the turbine.

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6.0 TABLES

Table 1. Total number and species of all fish captured in the two fishways and the tailrace between Sept 7 and Oct 21, 1999.

Species	Tailrace South Side	Tailrace North Side	New Fishway	Old Fishway	Total
sea lamprey	2	18	36	102	158
American eel	4	7	101	5	117
Blueback herring	55	138	93	10	296
alewife	14	21	124	2	161
American shad	16	24	62	0	102
Atlantic herring	272	619	880	88	1859
rainbow smelt	4	2	7	2	15
Hake spp.	38	57	23	5	123
Atlantic silverside	579	784	35,927	10,717	48,007
Blackspotted stickleback	30	46	612	173	861
pipefish	82	130	432	403	1,047
cunner	0	3	2	1	6
wrymouth	0	0	0	1	1
butterfish	4	30	47	7	88
Longhorn sculpin	1	0	0	0	1
lumpfish	0	3	3	0	6
smooth flounder	0	1	1	0	2
winter flounder	17	20	3	2	42
windowpane	14	16	22	9	61
mummichog	4	2	97	66	169
Atlantic mackerel	6	4	1	1	12
Meek’s halfbeak	0	0	1	0	1
flying gurnard	0	0	1	0	1
four-beard rockling	0	1	0	0	1
pollock	0	0	3	0	3
bluefish	0	0	2	0	2
White perch	0	0	2	1	3
TOTAL	1,142	1,925	38,482	11,595	53,144

Table 2. Estimates of the Model 2 coefficients and standard errors for clupeids captured during this assessment. The “site” coefficients are scaled against the tailrace (south side) site. The “diversion” coefficients are the difference ($logN_{off}$ -$logN_{on}$). Asterisks indicate the level of statistical significance (t-test; null hypothesis: coeff. = 0): “*” indicates significance at a 90% confidence level, “**” indicates significance at a 95% confidence level, and “***” indicates significance at a 99% confidence level.

Species	Dispersion Parameter	Intercept	Site: New Fishway	Site: Old Fishway	Site: Tailrace (north side)	Proportion of Darkness	Tidal Range	Salinity	Salinity Squared	Temperature	Temperature Squared	Diversion: New Fishway	Diversion: Old Fishway	Diversion: Tailrace
American Shad	2.25	-0.23 (9.64)	3.58 (6.08)	-6.75 (18.22)	1.76 (6.08)	1.37** (0.59)	0.39* (0.19)	0.07 (0.49)	0.00 (0.03)	-0.59 (0.94)	0.02 (0.03)	-1.29*** (0.44)	n/a	0.43 (0.54)
blueback herring	3.90	-13.45* (7.01)	0.26 (0.29)	-1.14** (0.48)	0.78*** (0.23)	1.71*** (0.43)	0.24* (0.15)	0.39 (0.40)	-0.01 (0.01)	1.14 (0.91)	-0.03 (0.03)	0.03 (0.42)	-7.72 (16.93)	-0.38 (0.31)
bb. herring (trimmed)	3.27	-8.72 (7.14)	0.70** (0.29)	-0.75 (0.46)	0.64** (0.25)	1.34*** (0.45)	0.27* (0.16)	0.26 (0.40)	0.00 0.03	-1.57* (0.94)	-0.05* (0.03)	-0.58 (0.46)	-7.87 (16.78)	0.30 (0.36)
Alewife	2.26	-12.29 (7.65)	2.24*** (0.37)	-1.56* (0.83)	-0.13 (0.43)	1.85*** (0.51)	0.08 (0.17)	-0.17 (0.48)	0.00 (0.01	2.54** (1.08)	-0.10*** (0.04)	-1.42*** (0.33)	-6.27 (11.79)	-0.04 (0.58)
Atlantic herring	6.56	15.80*** (3.39)	0.71*** (0.16)	-1.18*** (0.28)	0.62*** (0.14)	2.65*** (0.29)	0.17* (0.10)	-0.63** (0.25)	0.01** (0.00)	-0.93** (0.36)	0.02* (0.01)	0.11 (0.19)	-0.77 (0.58)	-0.29 (0.20)

Table 3. Estimates of the Model 2 coefficients and standard errors for non-clupeid species. The “site” coefficients are scaled against the tailrace (south side) site. The “diversion” coefficients are the difference ($\log N_{off} - \log N_{on}$). Asterisks indicate the level of statistical significance (t-test; null hypothesis: coeff. = 0): “*” indicates significance at a 90% confidence level, “**” indicates significance at a 95% confidence level, and “***” indicates significance at a 99% confidence level.

Species	Dispersion Parameter	Intercept	Site: New Fishway	Site: Old Fishway	Site: Tailrace (north side)	Proportion of Darkness	Tidal Range	Salinity	Salinity Squared	Temperature	Temperature Squared	Diversion: New Fishway	Diversion: Old Fishway	Diversion: Tailrace
Atlantic silverside	538.78	-0.76	2.22***	0.71	1.33*	1.91***	0.16	-0.63**	0.01**	1.11*	-0.02	-0.40	0.16	-1.15
		(7.00)	(0.41)	(0.47)	(0.72)	(0.39)	(0.13)	(0.31)	(0.01)	(0.64)	(0.02)	(0.28)	(0.47)	(1.38)
pipefish	5.11	-2.66	0.56**	-0.11	-0.02	2.23***	-0.19**	-0.50*	0.01**	1.03**	-0.03**	0.48*	1.29***	-0.41
		(5.50)	(0.23)	(0.27)	(0.22)	(0.26)	(0.09)	(0.29)	(0.01)	(0.41)	(0.01)	(0.28)	(0.33)	(0.34)
b.-s. stickleback	6.99	10.52*	1.72***	0.47	-0.93**	-2.09***	0.21**	-0.56	0.01*	-0.34	0.01	-0.04	-0.04	-0.06
		(6.35)	(0.26)	(0.31)	(0.37)	(0.38)	(0.10)	(0.343)	(0.01)	(0.55)	(0.02)	(0.23)	(0.45)	(0.62)
American eel	1.57	1.68	3.78	-5.57	1.15	1.38***	0.01	-0.75*	0.01	0.79	-0.03	0.10	7.11	0.25
		(7.32)	(3.36)	(10.02)	(3.37)	(0.44)	(0.15)	(0.40)	(0.01)	(0.73)	(0.02)	(0.29)	(13.38)	(0.80)
winter flounder	0.39	-28.59***	0.51	-6.70	3.16	2.39***	-0.47***	-0.90	0.02	6.18***	-0.24***	0.61	7.78	-0.38
		(8.87)	(3.39)	(10.07)	(3.36)	(0.38)	(0.17)	(0.77)	(0.01)	(1.25)	(0.05)	(0.78)	(13.44)	(0.28)
sea lamprey	8.26	-193.3	0.53	1.59**	0.04	2.65	-6.70	13.99	-0.22	4.48	-0.28	7.40	7.36	6.98
		(208.8)	(0.79)	(0.69)	(0.80)	(5.85)	(8.40)	(11.77)	(0.20)	(21.58)	(0.88)	(7.06)	(7.02)	(7.13)
hake spp.	1.62	-7.60	-0.29	-1.61***	1.15***	1.99***	-0.29	0.55	-0.01	0.37	-0.03	0.38	0.04	-0.21
		(8.90)	(0.44)	(0.69)	(0.31)	(0.45)	(0.19)	(0.72)	(0.01)	(0.85)	(0.03)	(0.60)	(1.17)	(0.33)
butterfish	2.22	-37.19**	0.71	-0.66	1.04**	0.79	0.31	2.08*	-0.04*	0.71	-0.02	0.86	0.55	-0.10
		(16.99)	(0.55)	(0.85)	(0.45)	(0.57)	(0.21)	(1.21)	(0.02)	(1.16)	(0.04)	(0.60)	(1.25)	(0.57)
mummichog	32.22	-139.9	1.37	0.78	-1.43	0.18	1.47	6.41	-0.11	4.77	-0.14	1.38	1.99	0.05
		(108.8)	(1.78)	(1.95)	(3.25)	(2.47)	(1.04)	(7.81)	(0.14)	(7.91)	(0.26)	(1.62)	(1.97)	(4.96)
windowpane	1.19	-13.48	0.47	-1.06*	0.47	2.51***	-0.35*	0.14	0.00	1.51*	-0.05*	0.03	0.84	-0.34
		(9.83)	(0.39)	(0.61)	(0.34)	(0.49)	(0.18)	(0.60)	(0.01)	(0.89)	(0.03)	(0.49)	(0.87)	(0.44)

Table 4. Number of fish captured in the new fishway during the period between the end of generation and the start of flood sluicing.

Species	Total Number	Number with Diversion On	Number with Diversion Off
American shad	0	0	0
blueback herring	13	2	11
alewife	12	7	5
Atlantic herring	57	40	17
Atlantic silverside	1291	502	789
American eel	6	1	5
pipefish	24	4	20
winter flounder	0	0	0
windowpane	0	0	0
hake (spp.)	1	0	1
b.-s. stickleback	90	42	48
sea lamprey	8	8	0
butterfish	9	0	9
mummichog	9	8	1

Table 5. Estimates of the coefficients and standard errors for Model 2 fitted to the catch in the new fishway after the end of generation and before the start of flood sluicing. The "diversion" coefficients are difference ($\log N_{off}$-$\log N_{on}$). Asterisks indicate the level of statistical significance (t-test; null hypothesis: coeff. = 0): "*" indicates significance at a 90% confidence level, "**" indicates significance at a 95% confidence level, and "***" indicates significance at a 99% confidence level.

Species	Disper-sion Parameter	Intercept	Proportion of Darkness	Tidal Range	Salinity	Salinity Squared	Temper-ature	Temper-ature Squared	Diversion New Fishway
Atlantic herring	1.75	-0.13 (18.80)	3.66*** (1.25)	0.48 (0.36)	-1.37 (1.44)	0.03 (0.03)	2.11 (2.01)	-0.08 (0.07)	-1.12* (0.67)
Atlantic silverside	67.4	-.50 (21.86)	1.56 (0.944)	0.38 (0.36)	-0.16 (0.96)	-0.00 (0.01)	1.18 (2.68)	-0.03 (0.09)	0.12 (0.74)
b.s. stickleback	3.4	3.55 (16.04)	-0.81 (0.68)	0.06 (0.27)	-0.76 (0.59)	0.01 (0.01)	1.04 (1.83)	-0.03 (0.907)	0.22 (0.307)

Table 6. Fork lengths of clupeids captured during this assessment.

Species	n	mean (mm)	minimum (mm)	maximum (mm)	standard deviation
Am. shad	24	105.3	79.0	144.0	12.5
blueback herring	96	117.8	95.0	143.0	7.9
alewife	38	104.2	58.0	214.0	30.9
Atl. herring	222	83.2	36.0	213.0	30.8

Table 7. Summary of the fishing efficiency tests for the tailrace nets

Date	Number of test fish	Number Captured	
		north net	south net
Sept. 17th	1441	1	n/a
Sept. 21th	1255	1	0
Sept. 23rd	1474	7	3

Table 9. The live/dead status of fish of fish captured in the new fishway after net deployments of 0.5h. The "Acute" columns are the numbers alive at the time of capture. The "Delayed" columns are the numbers alive after being held in the livebox for 12h (including fish that were dead at the time of capture).

Species	Acute		Delayed	
	number captured	number alive	number captured	number alive
Atlantic herring	26	21	26	9
alewife	1	1	1	1
Atlantic silverside	27	26	27	18
sea lamprey	24	24	24	24
pipefish	4	4	4	4
pollock	3	3	3	3
b.-s. stickleback	2	2	2	2

Table 10. Estimates of acute mortality for 12 species of fish captured during this assessment.

Species	n	Mortality (%) mean	95% C.I. upper limit	95% C.I. lower limit
American shad	39	23.4	6.6	56.7
blueback herring	208	8.1	4.2	14.9
alewife	34	7.7	1.5	31.3
Atlantic herring	840	15.7	11.9	20.4
sea lamprey	20	0.0	n/a	n/a
b. s. sticklebacks	68	0.0	0.0	1.7
Atlantic silverside	1160	2.2	1.4	3.3
pipefish	202	2.2	0.8	5.8
butterfish	32	8.7	1.8	3.2
winter flounder	31	5.8	1.0	28.9
windowpane	28	8.8	0.7	56.8
hake (spp.)	88	8.7	3.4	20.2

Table 11. Estimates of turbine mortality (acute + delayed) for 12 species of fish captured during this assessment. Estimates marked with a strike-through are unbelievable for reasons discussed in the text.

Species	n	Mortality (%) mean	95% C.I. upper limit	95% C.I. lower limit
American shad	9	~~19.9~~	~~0.4~~	~~93.2~~
blueback herring	35	~~26.9~~	~~2.7~~	~~82.8~~
alewife	15	4.7	0.1	63.1
Alosa (combined)	59	~~20.6~~	~~0.1~~	~~87.0~~
Atlantic herring	382	~~73.2~~	~~60.4~~	~~82.9~~
sea lamprey	13	0.0	n/a	n/a
b. s. sticklebacks	27	~~19.9~~	~~1.7~~	~~78.0~~
Atlantic silverside	186	5.2	2.3	11.2
pipefish	63	2.8	0.3	19.2
butterfish	13	~~78.8~~	~~24.4~~	~~97.7~~
winter flounder	21	2.7	0.2	29.4
windowpane	14	6.3	0.2	6.8
hake (spp.)	67	~~43.7~~	~~24.8~~	~~64.7~~

Table 12. The live/dead status of fish for which too few fish were captured to warrant analysis. The "Acute" columns are the numbers alive at the time of capture. The "Delayed" columns are the numbers alive after being held in the livebox until the start of the next generation cycle (including fish that were dead at the time of capture).

	Acute		Delayed	
Species	number captured	number alive	number captured	number alive
Atlantic mackerel	9	6	2	2
lumpfish	1	0	1	1
mummichog	6	6	2	2
longhorn sculpin	1	1	1	1
American eel	10	9	2	2
rainbow smelt	5	3	0	0
cunner	3	3	0	0
smooth flounder	1	1	0	0

7.0 FIGURES

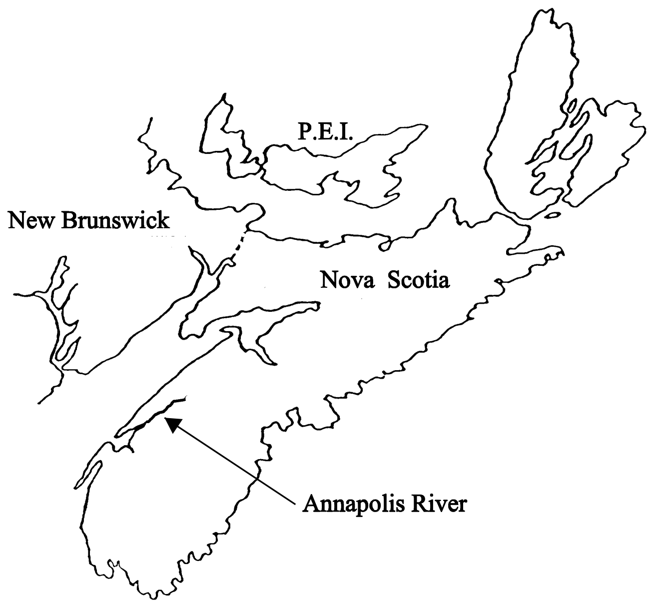


Figure 1. Location of the Annapolis River in Nova Scotia.

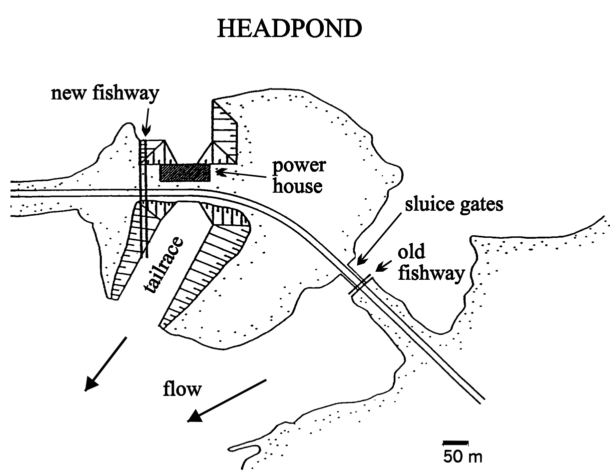


Figure 2. The Annapolis Tidal Generating Station in Annapolis Royal, Nova Scotia.

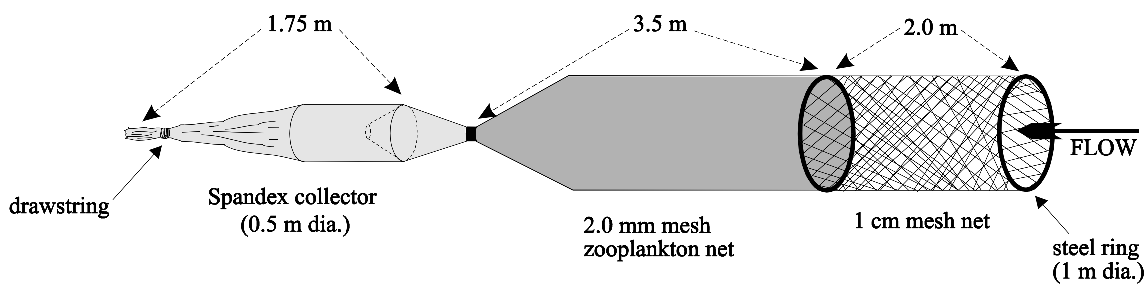


Figure 3. Nets used to monitor fish passage at the Annapolis Tidal Generating Station.

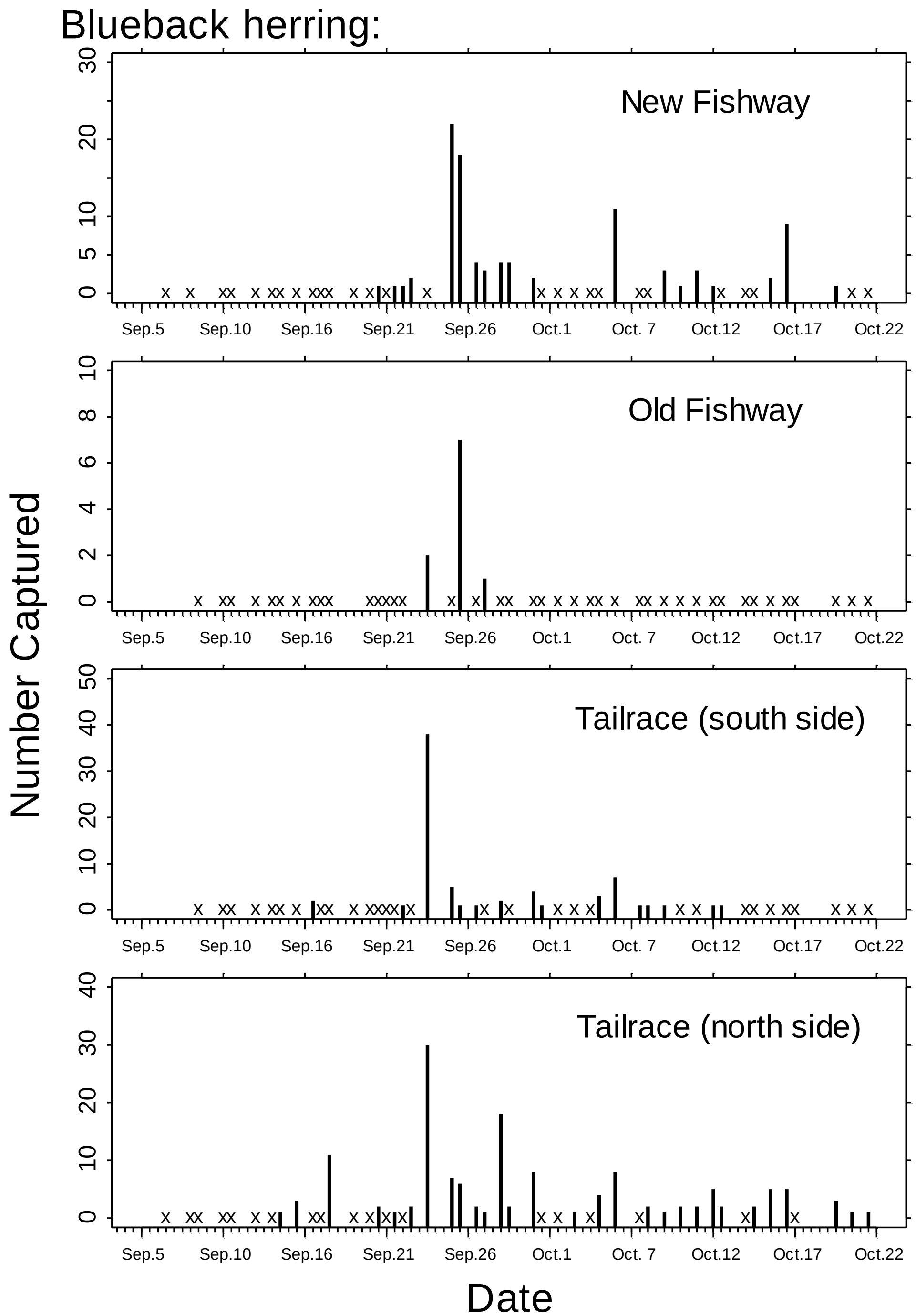


Figure 4. The number of blueback herring captured at each site during each generation cycle sampled throughout this study. Cycles that were sampled without catching blueback herring are marked with an “x”. Note scale differences between sites.

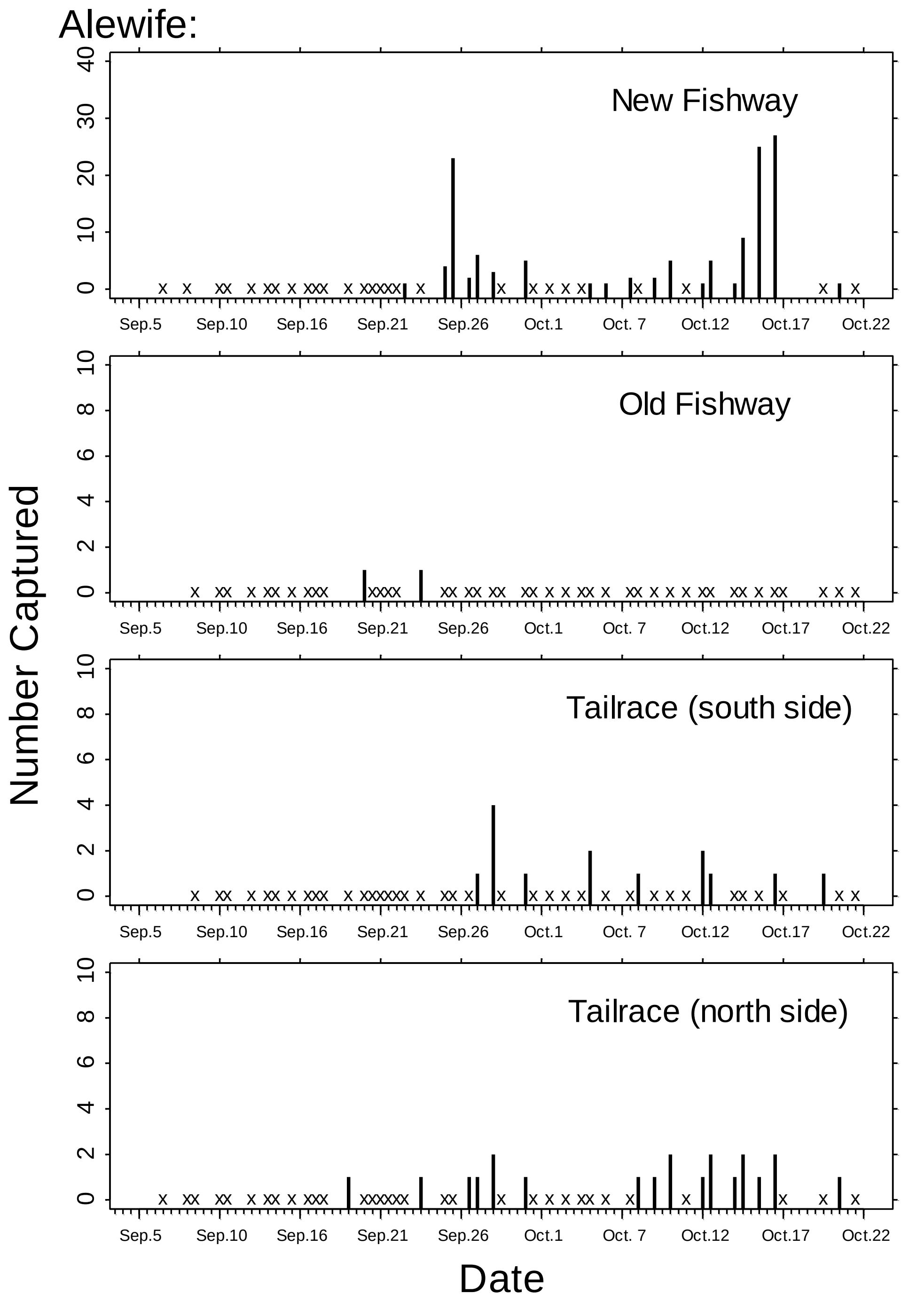


Figure 5. The number of alewife captured at each site during each generation cycle sampled throughout this study. Cycles that were sampled without catching alewife are marked with an "x". Note scale differences between sites.

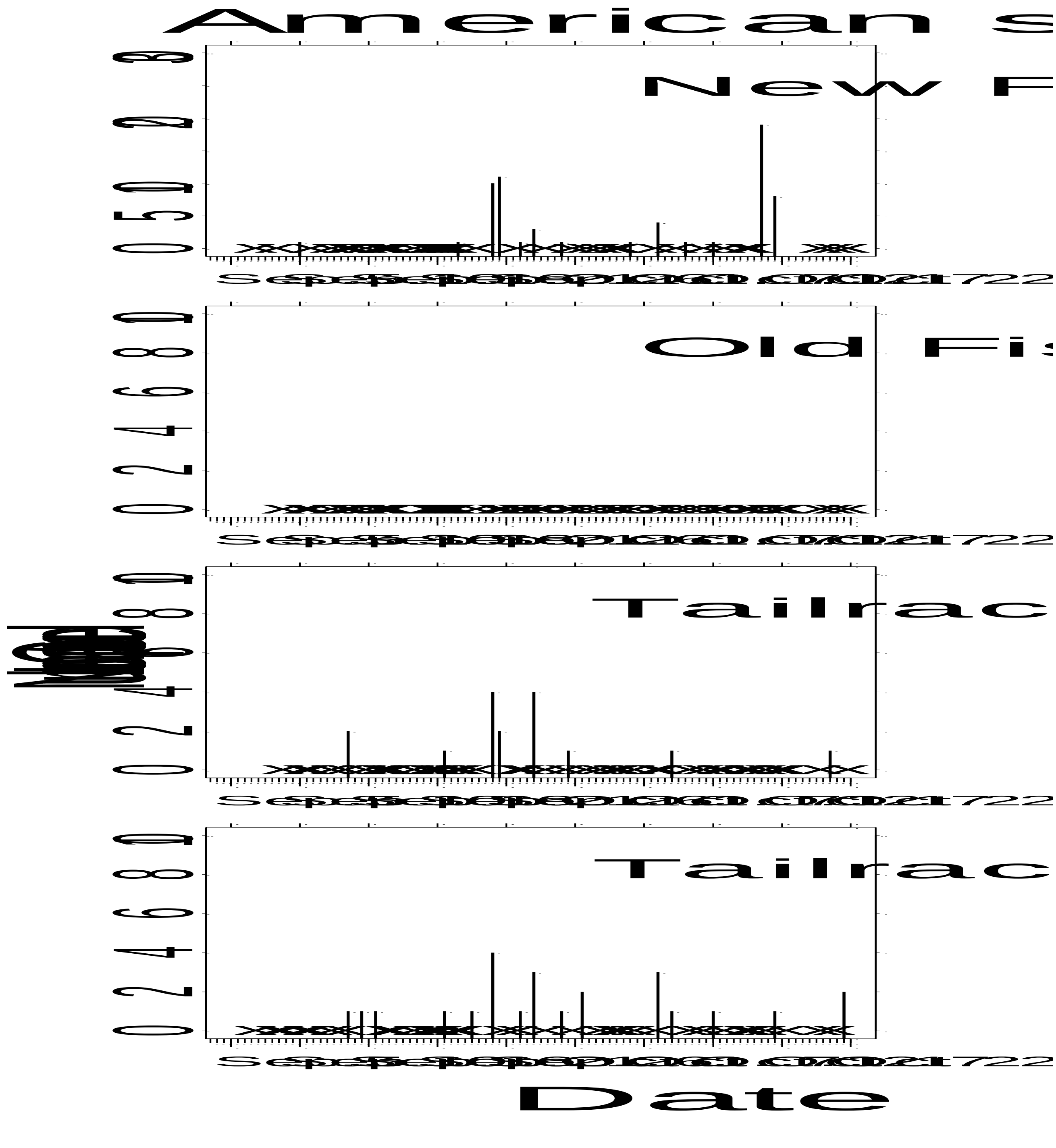


Figure 6. The number of American shad captured at each site during each generation cycle sampled throughout this study. Cycles that were sampled without catching shad are marked with an “x”. Note scale differences between sites.

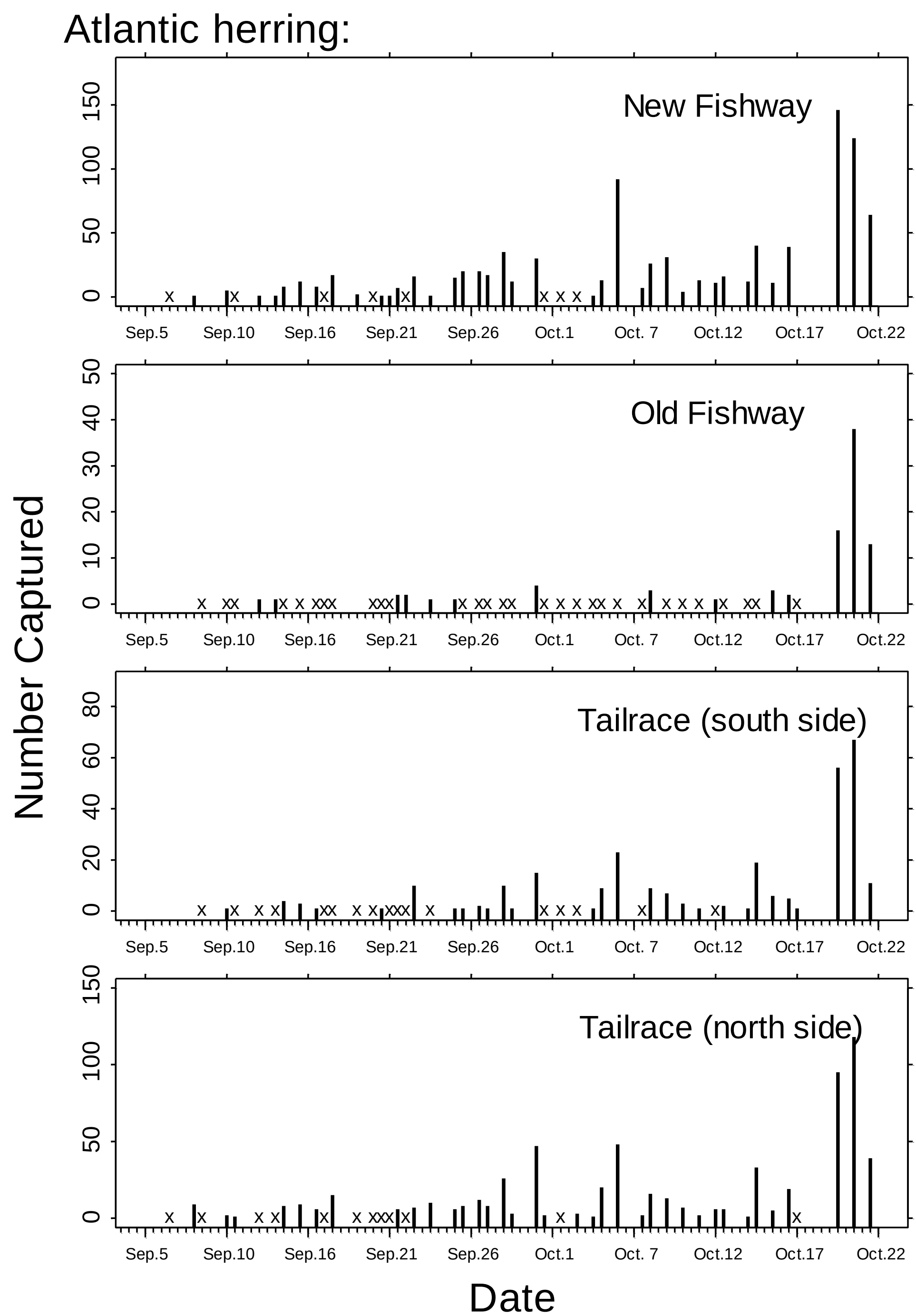


Figure 7. The number of Atlantic herring captured at each site during each generation cycle sampled throughout this study. Cycles that were sampled without catching Atlantic herring are marked with an “x”. Note scale differences between sites.

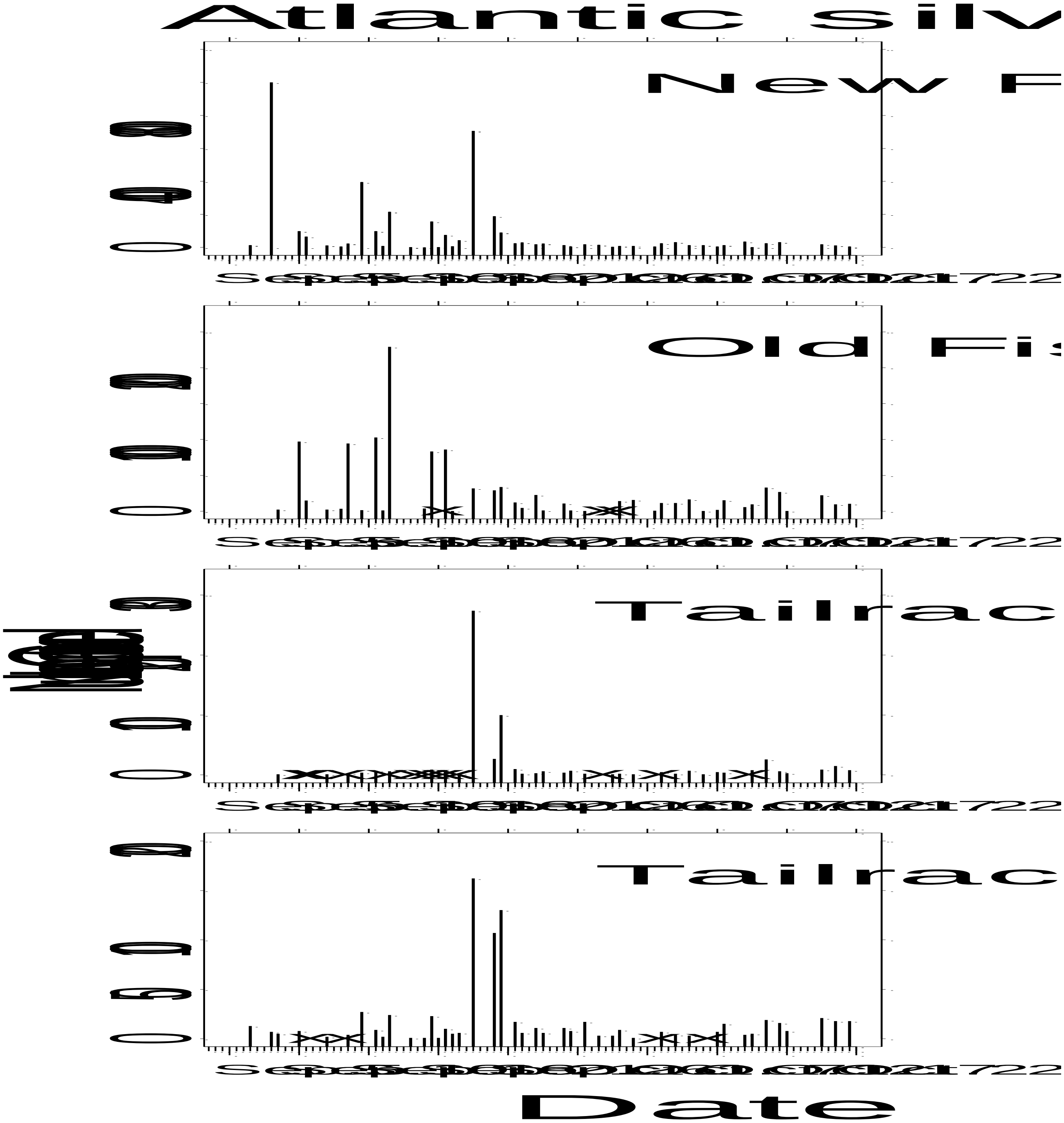


Figure 8. The number of Atlantic silversides captured at each site during each generation cycle sampled throughout this study. Cycles that were sampled without catching silversides are marked with an “x”. Note scale differences between sites.

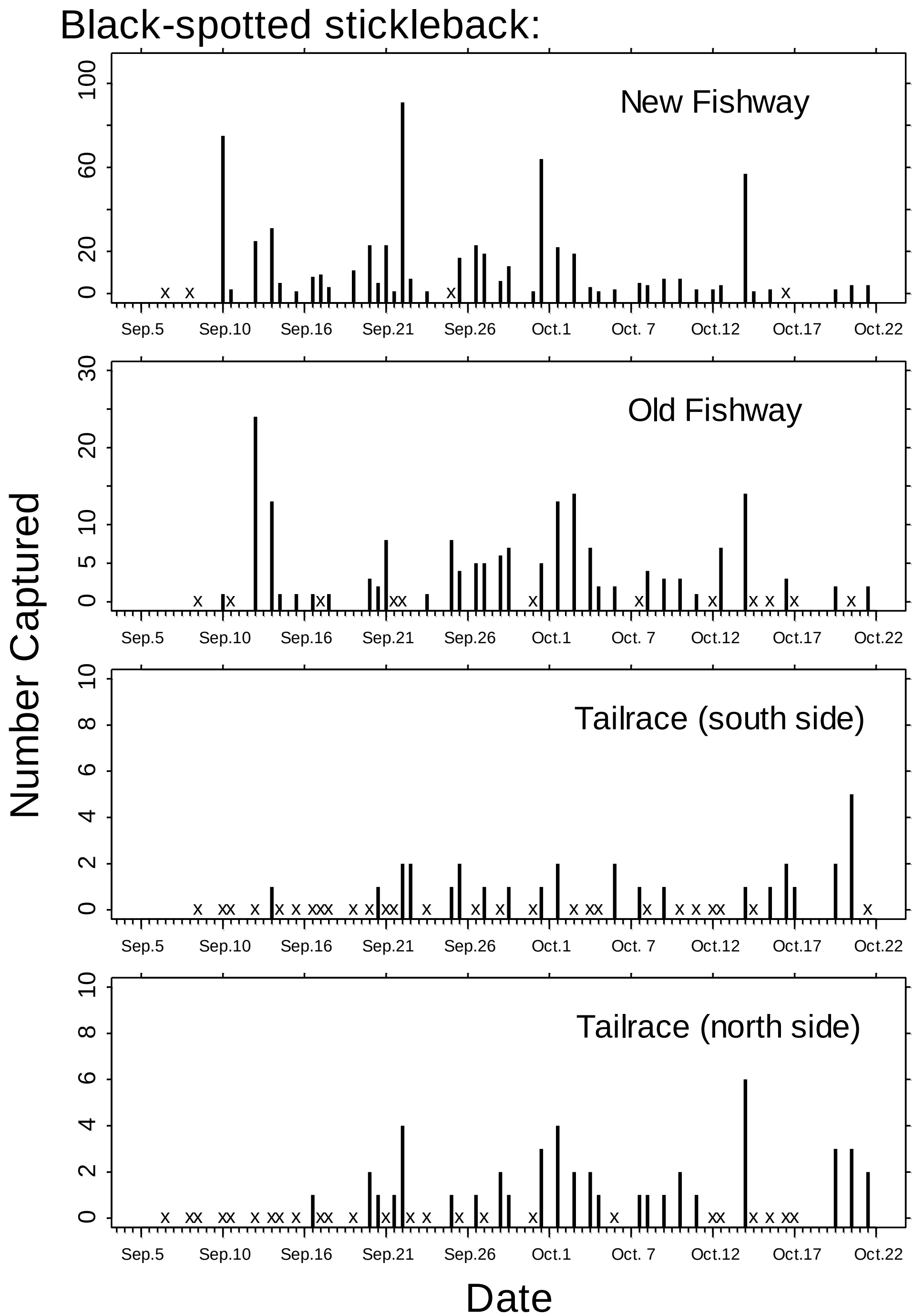


Figure 9. The number of blackspotted sticklebacks captured at each site during each generation cycle sampled throughout this study. Cycles that were sampled without catching sticklebacks marked with an "x". Note scale differences between sites.

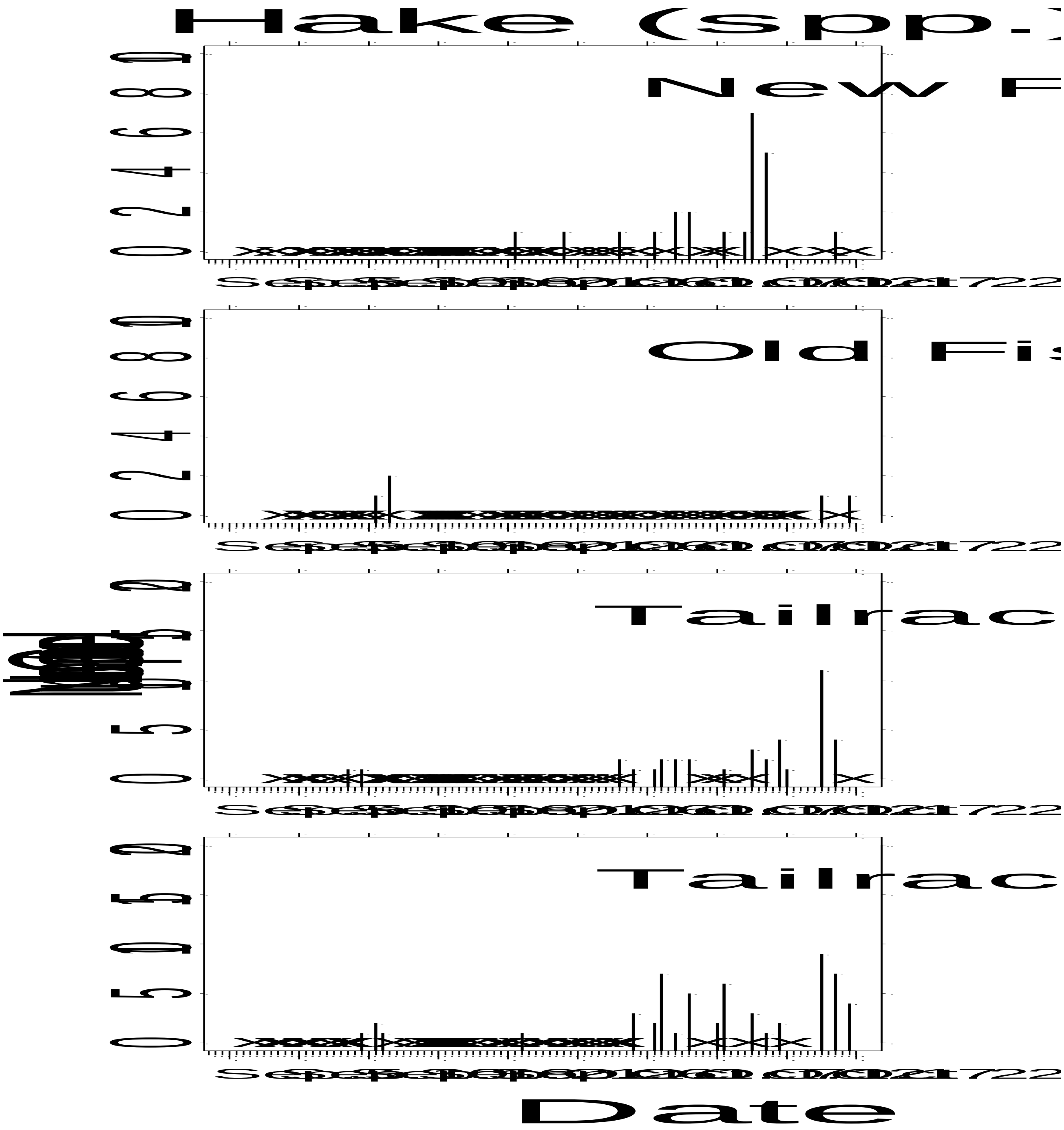


Figure 10. The number of hake captured at each site during each generation cycle sampled throughout this study. Cycles that were sampled without catching hake are marked with an “x”. Note scale differences between sites.

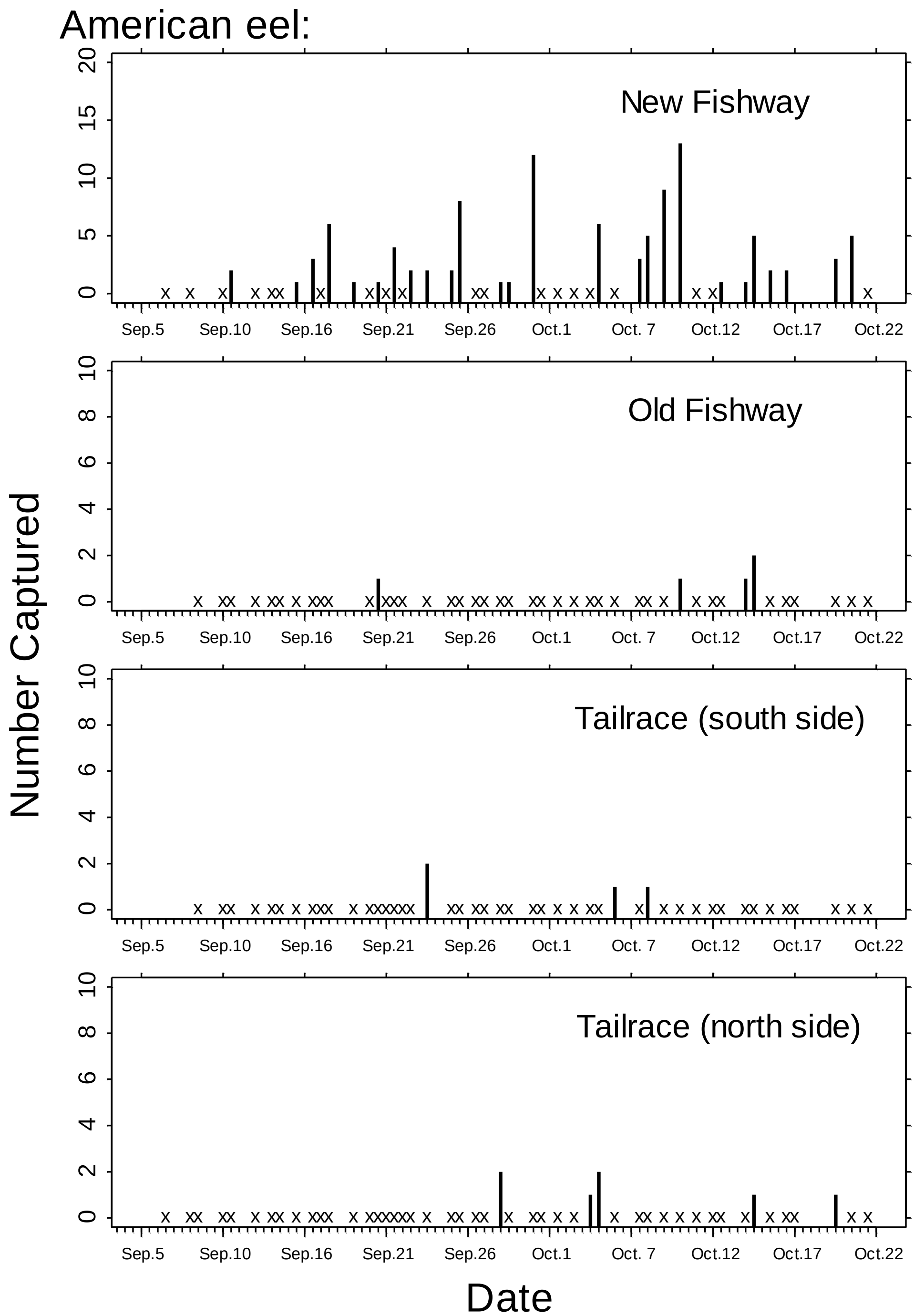


Figure 11. The number of American eels captured at each site during each generation cycle sampled throughout this study. Cycles that were sampled without catching eels are marked with an "x". Note scale differences between sites.

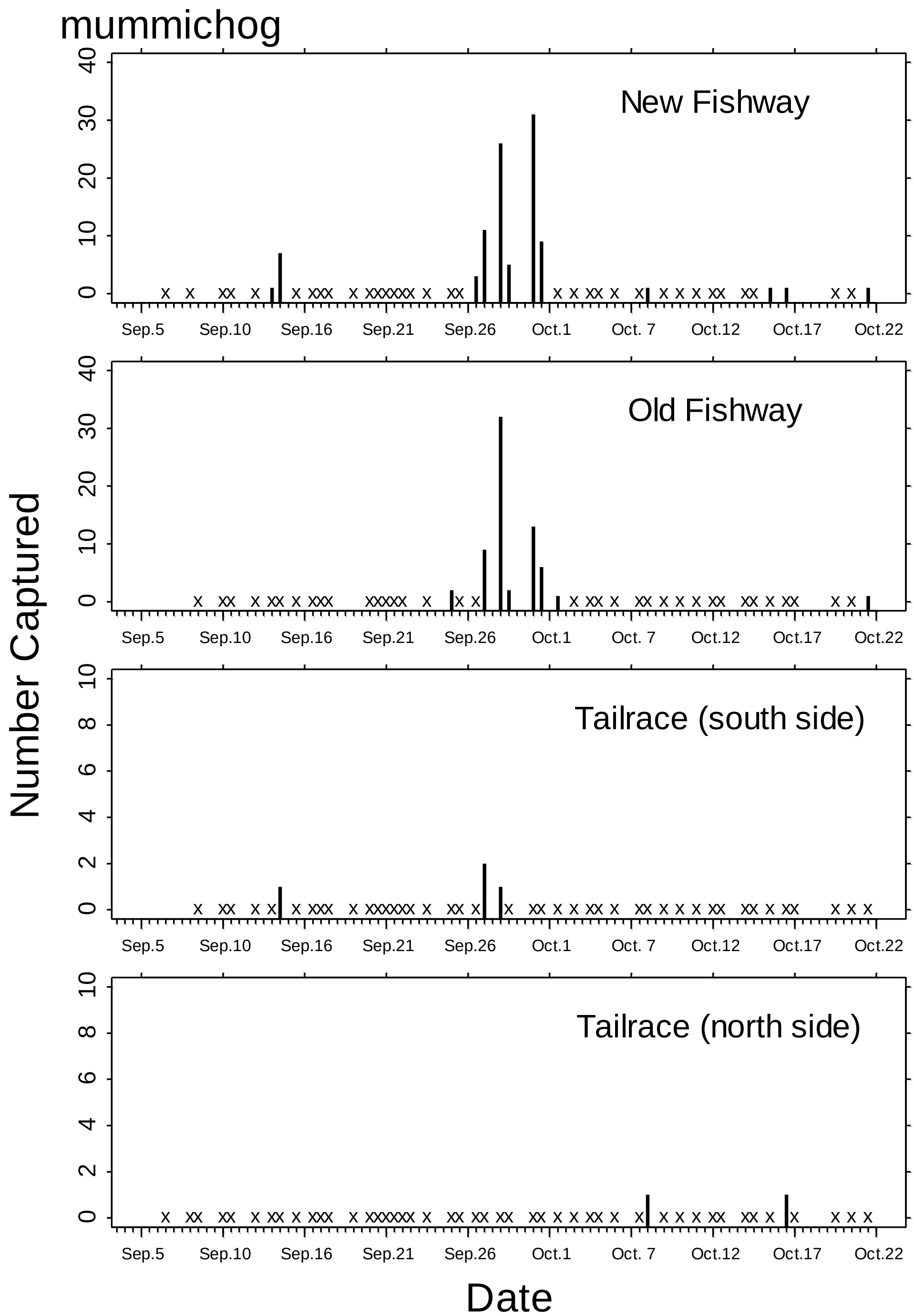


Figure 12. The number of mummichogs captured at each site during each generation cycle sampled throughout this study. Cycles that were sampled without catching mummichogs are marked with an “x”. Note scale differences between sites.

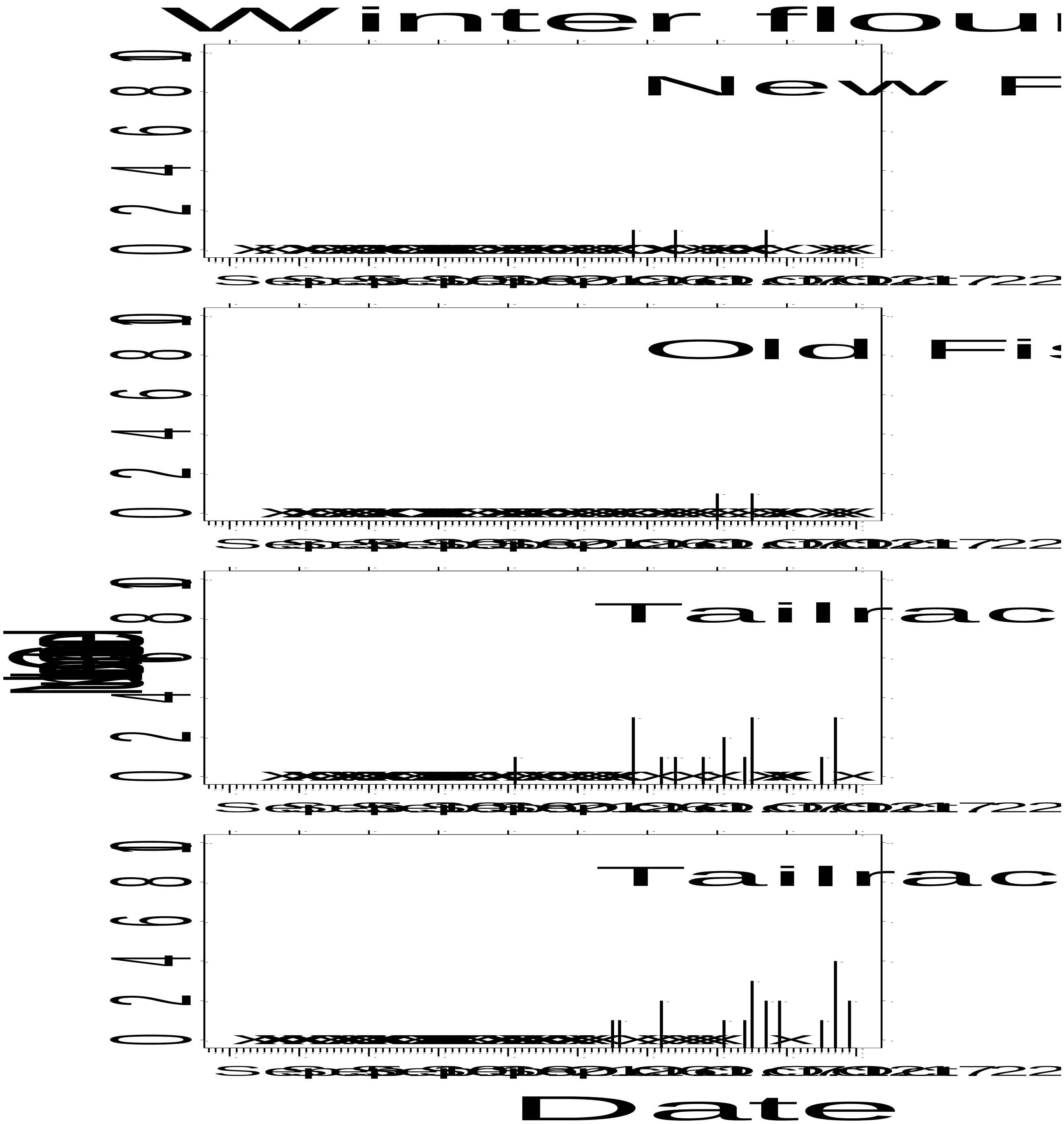


Figure 13. The number of winter flounder captured at each site during each generation cycle sampled throughout this study. Cycles that were sampled without catching winter flounder are marked with an “x”. Note scale differences between sites.

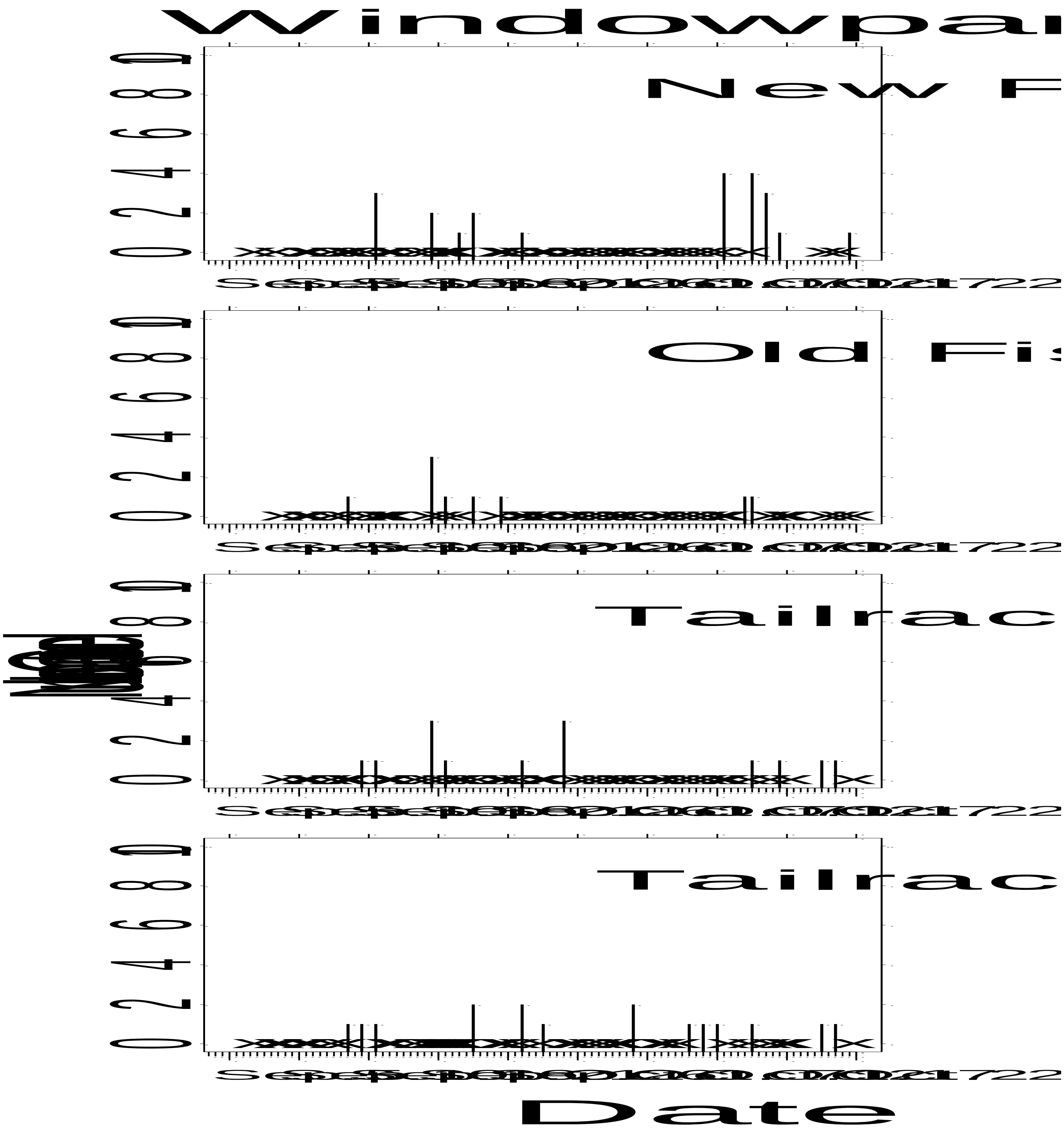


Figure 14. The number of windowpane captured at each site during each generation cycle sampled throughout this study. Cycles that were sampled without catching windowpane are marked with an “x”.

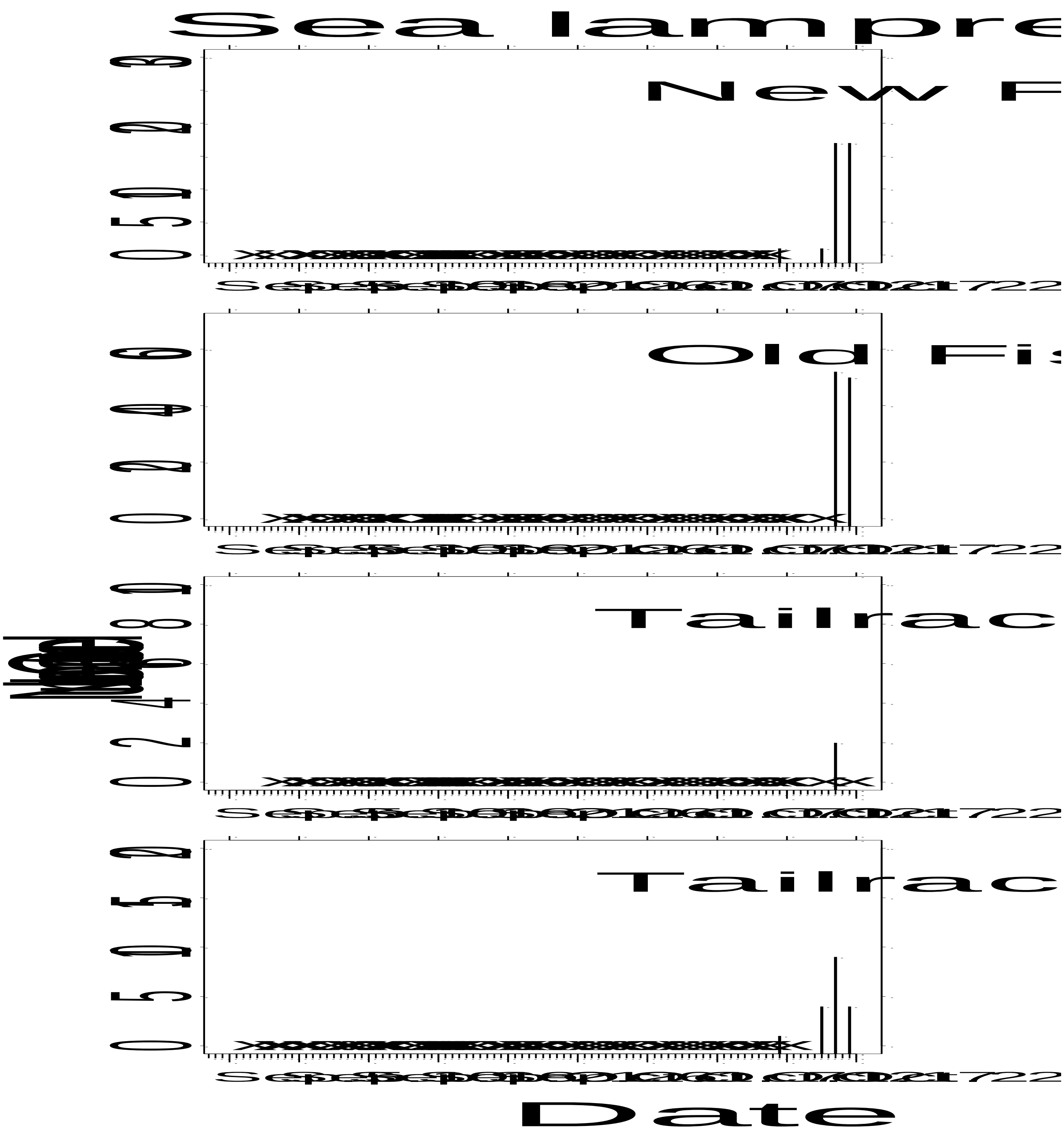


Figure 15. The number of sea lamprey captured at each site during each generation cycle sampled throughout this study. Cycles that were sampled without catching lamprey are marked with an “x”. Note scale differences between sites.

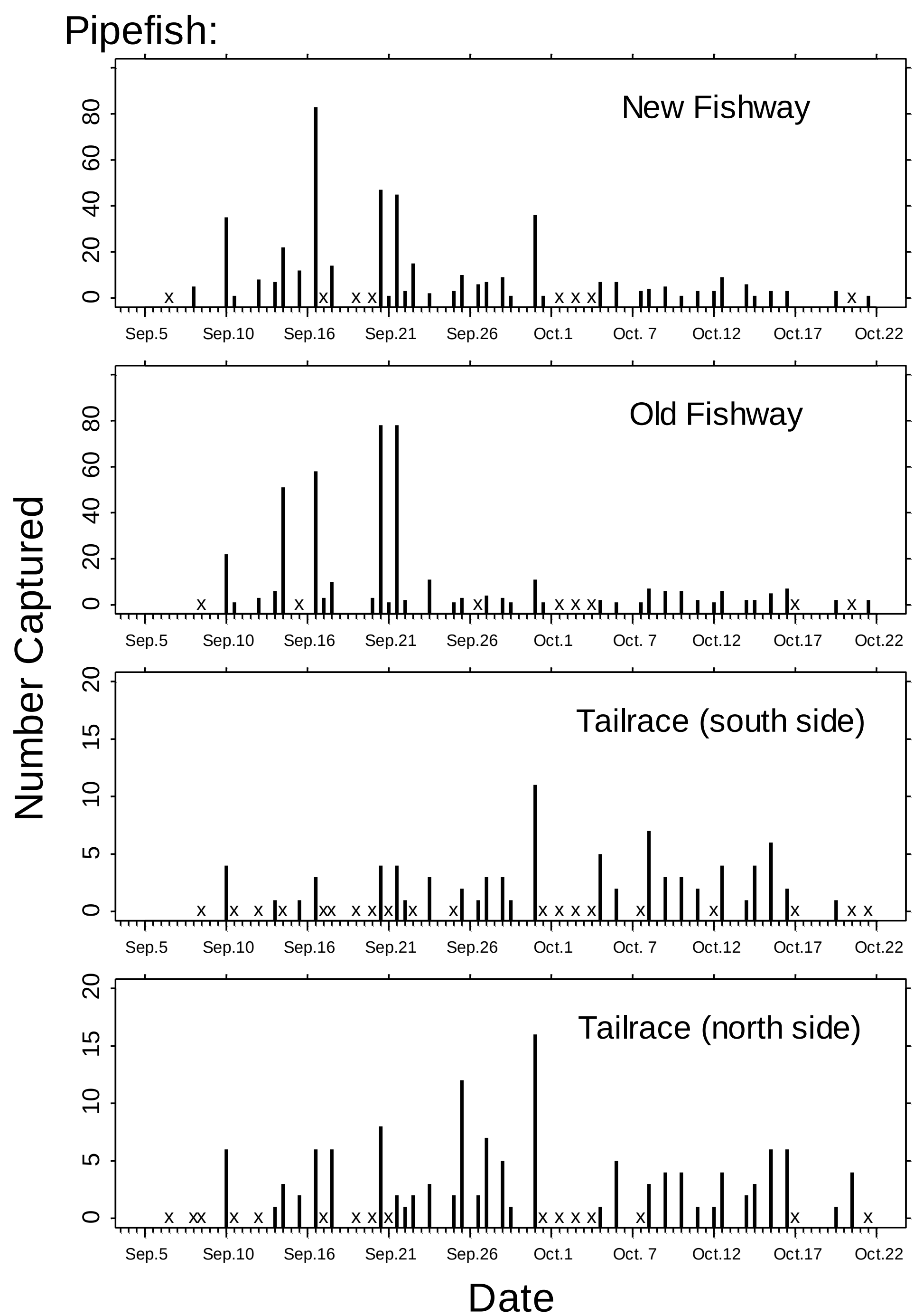


Figure 16. The number of pipefish captured at each site during each generation cycle sampled throughout this study. Cycles that were sampled without catching pipefish are marked with an “x”. Note scale differences between sites.

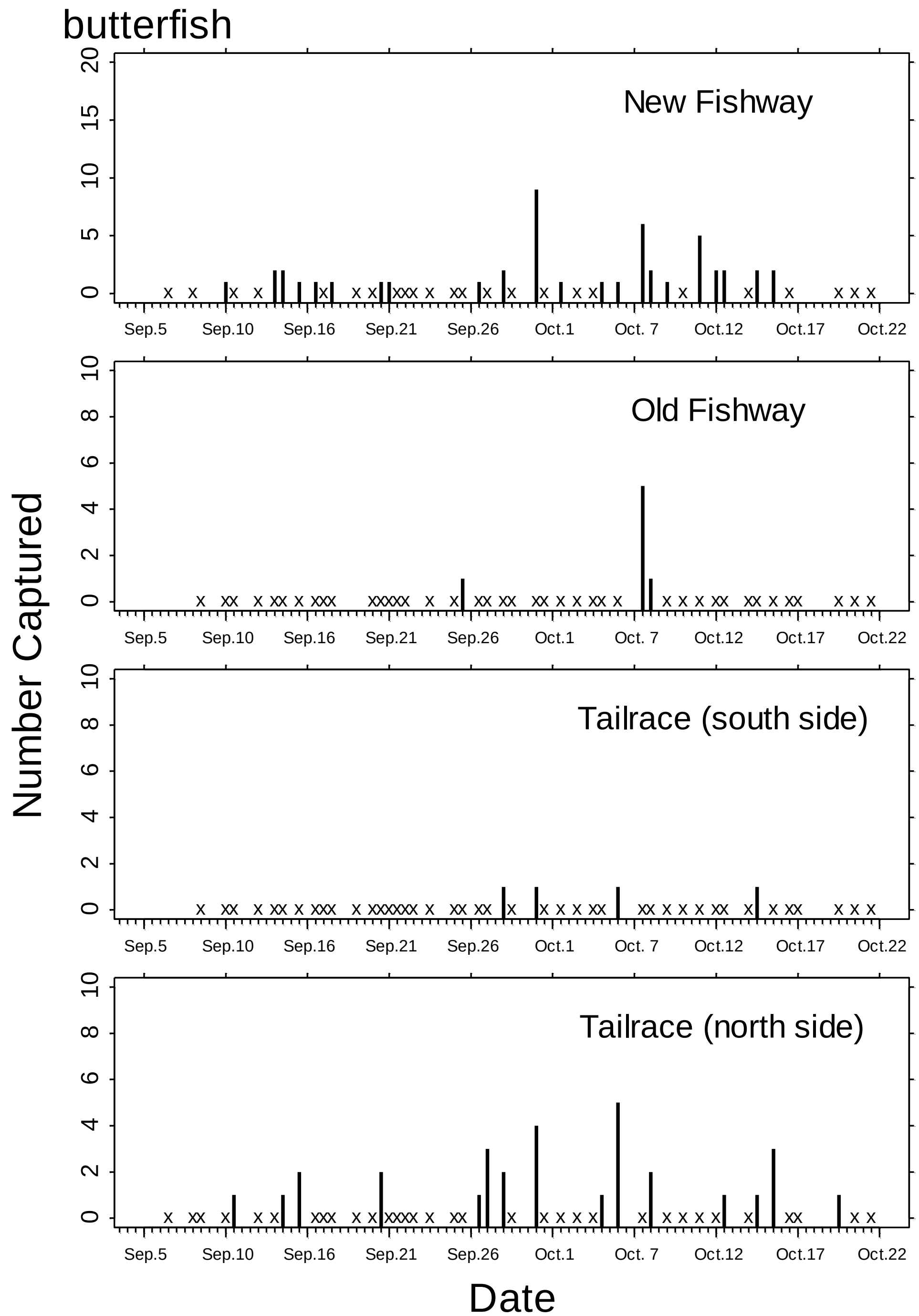


Figure 17. The number of butterfish captured at each site during each generation cycle sampled throughout this study. Cycles that were sampled without catching butterfish are marked with an “x”. Note scale differences between sites.

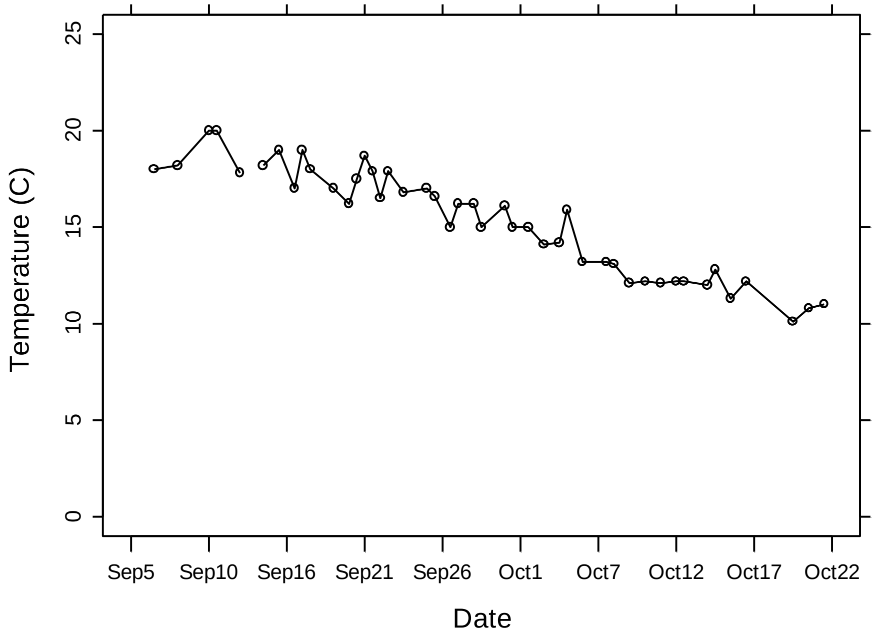


Figure 18. Water temperature measured at the mouth of the new fishway at the start of each generation cycle that was sampled during this study.

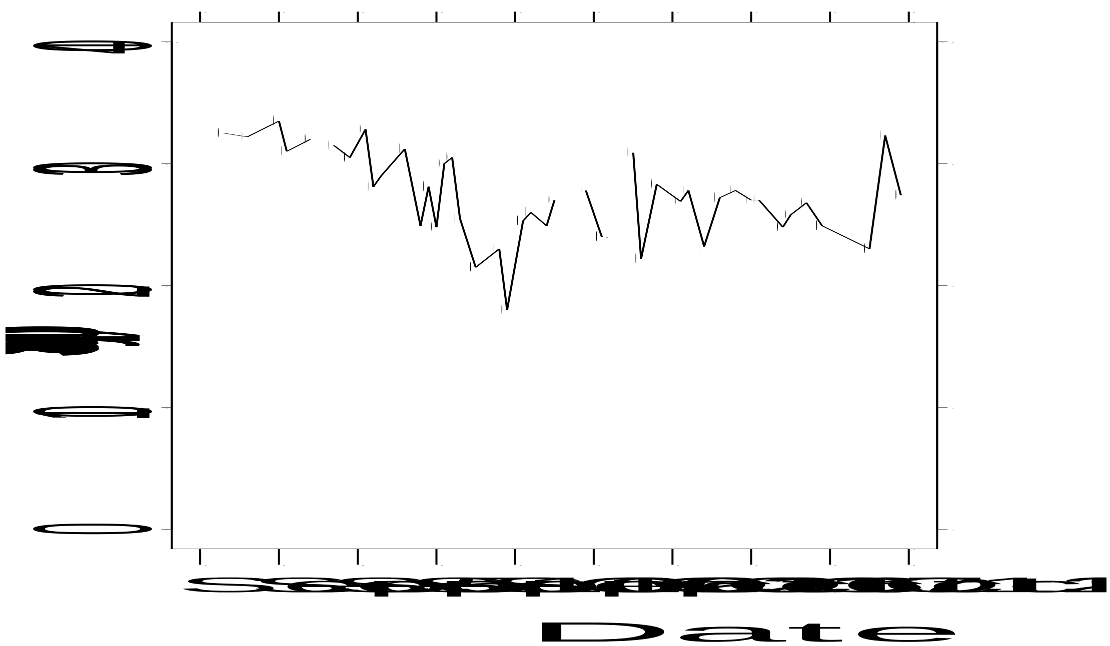


Figure 19. Salinity measured at the mouth of the new fishway at the start of each generation cycle that was sampled during this study.

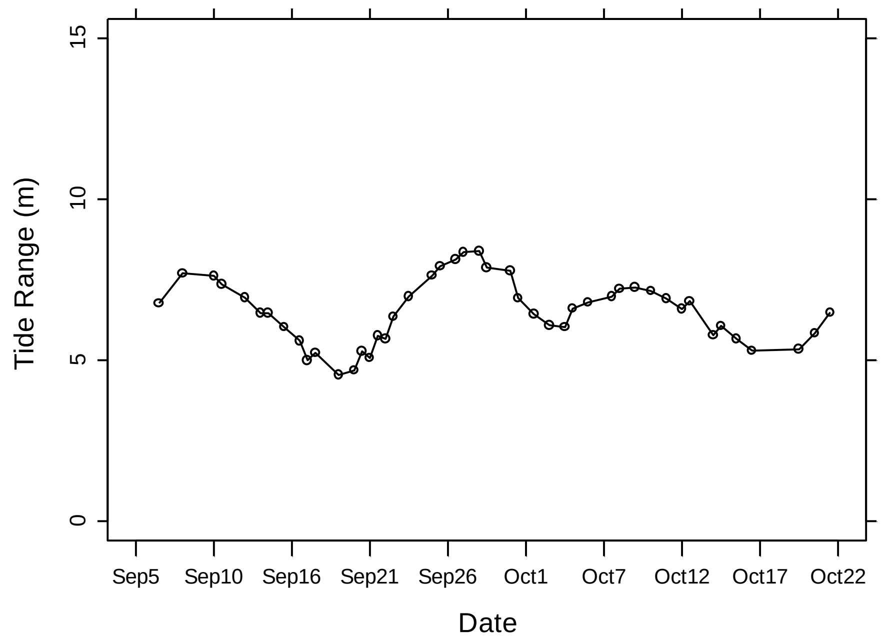


Figure 20. Tide range predicted for each generation cycle that was sampled during this study.

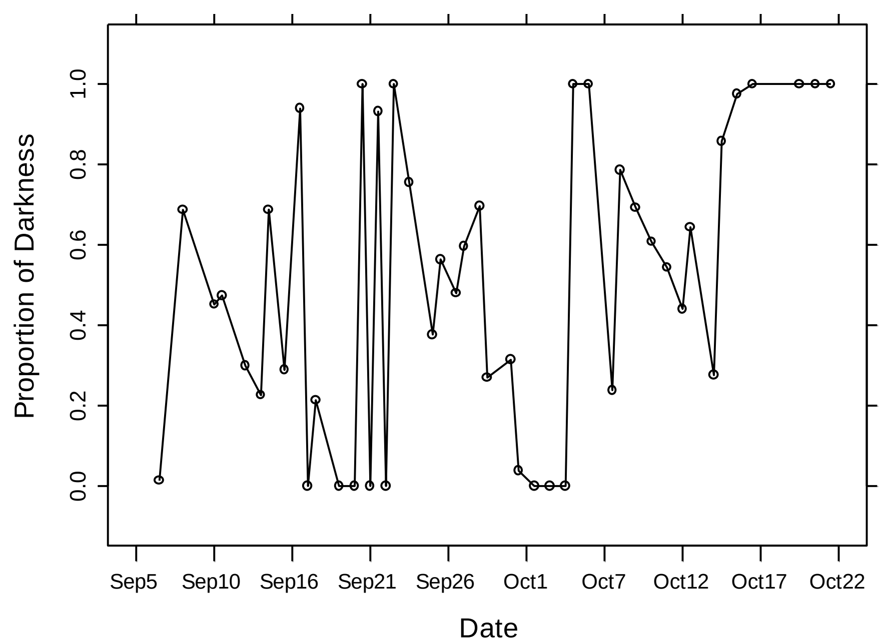


Figure 21. Proportion of each generation cycle sampled during this study that fell between sunset and sunrise.

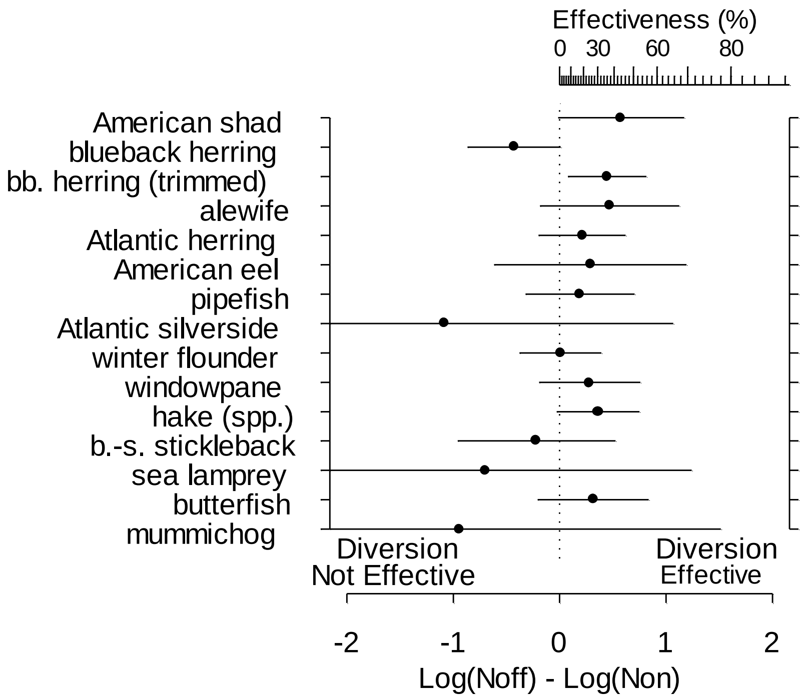


Figure 22. Estimates of the effectiveness of the fish diversion system, obtained from Model 1 (environmental variables excluded). Error bars are standard errors.

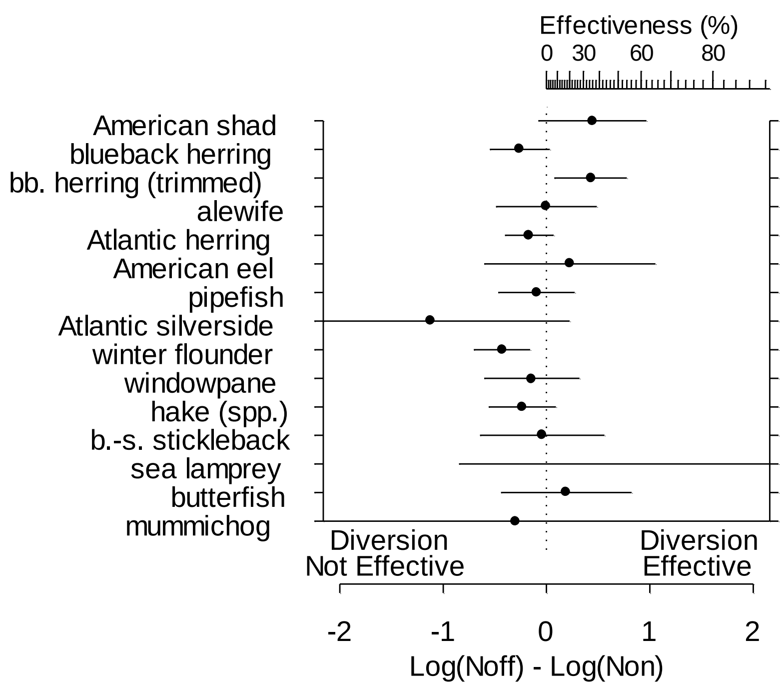


Figure 23. Estimates of the effectiveness of the fish diversion system, obtained from Model 2 (environmental variables included). Error bars are standard errors.

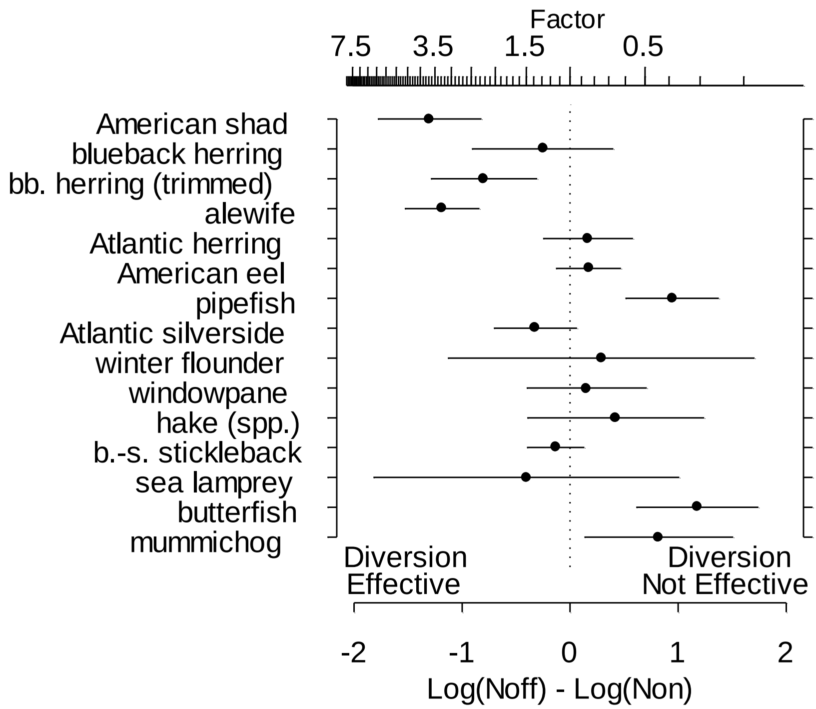


Figure 24. Estimates of the "fishway factor" obtained from Model 1 (environmental variables excluded). Error bars are standard errors.

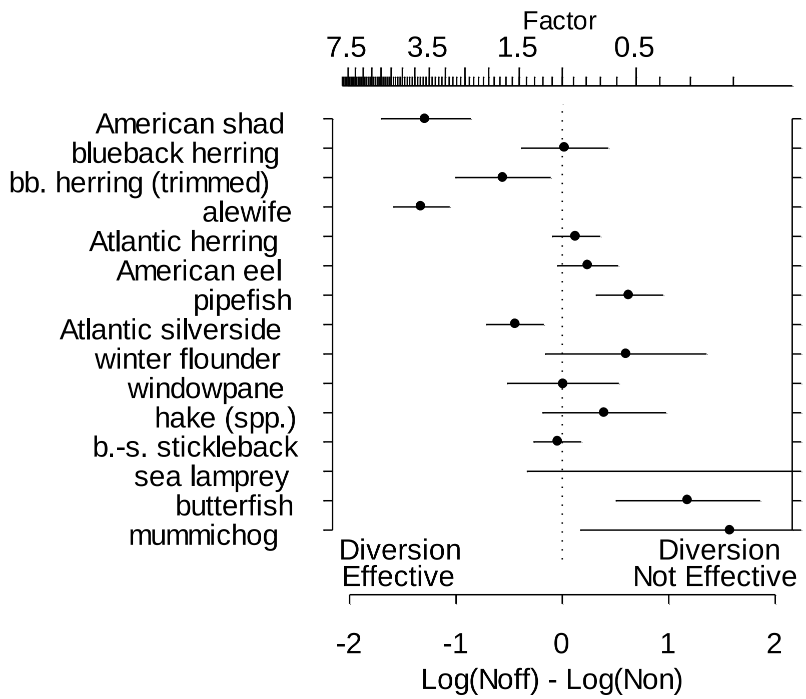


Figure 25. Estimates of the "fishway factor" obtained from Model 2 (environmental variables included). Error bars are standard errors.

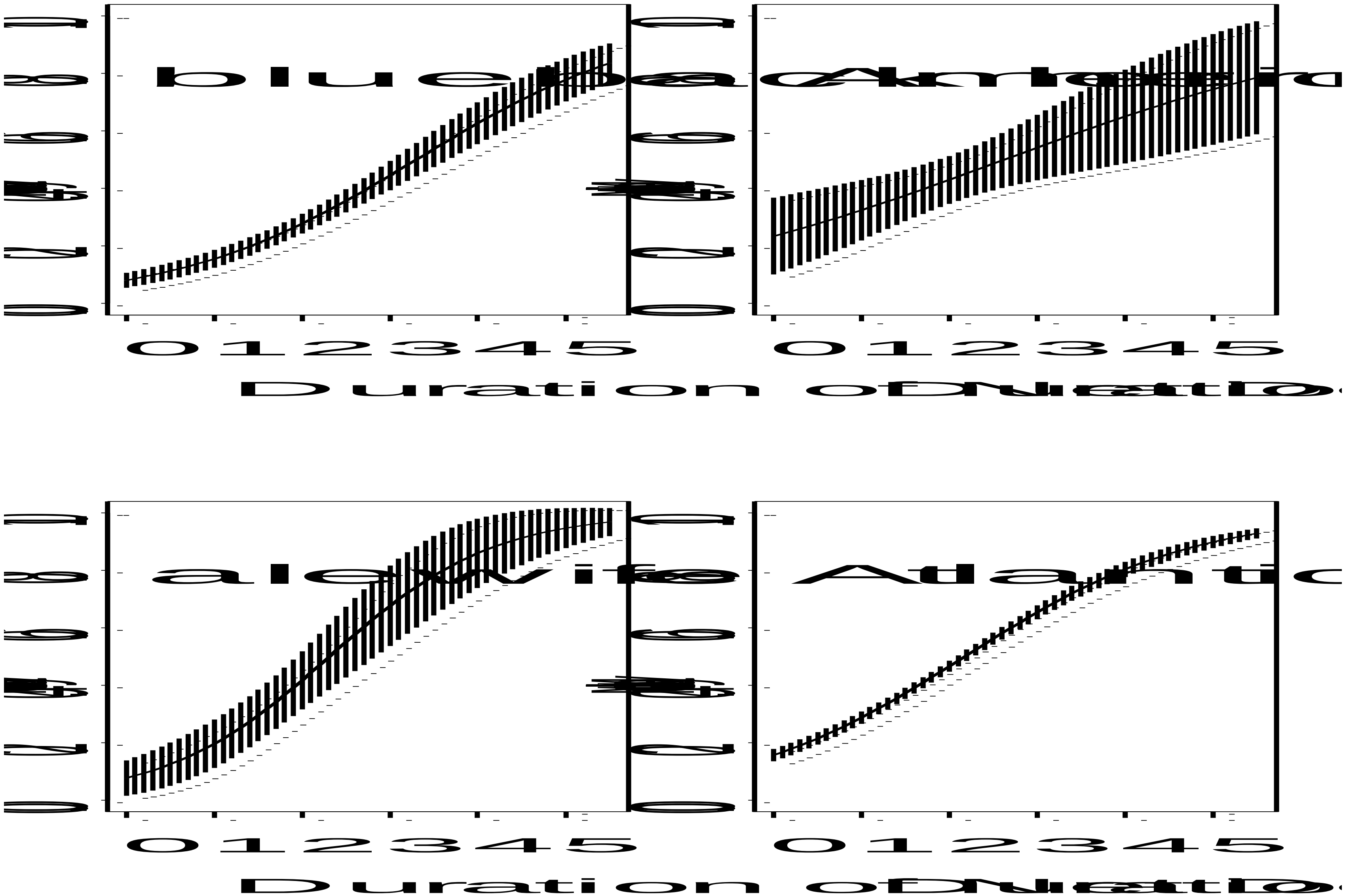


Figure 26. Acute mortality as a function of the duration of net deployment for clupeids. The y-intercept is an estimate of acute mortality in the absence of any time in the net, interpreted as acute turbine mortality. Error bars are standard errors.

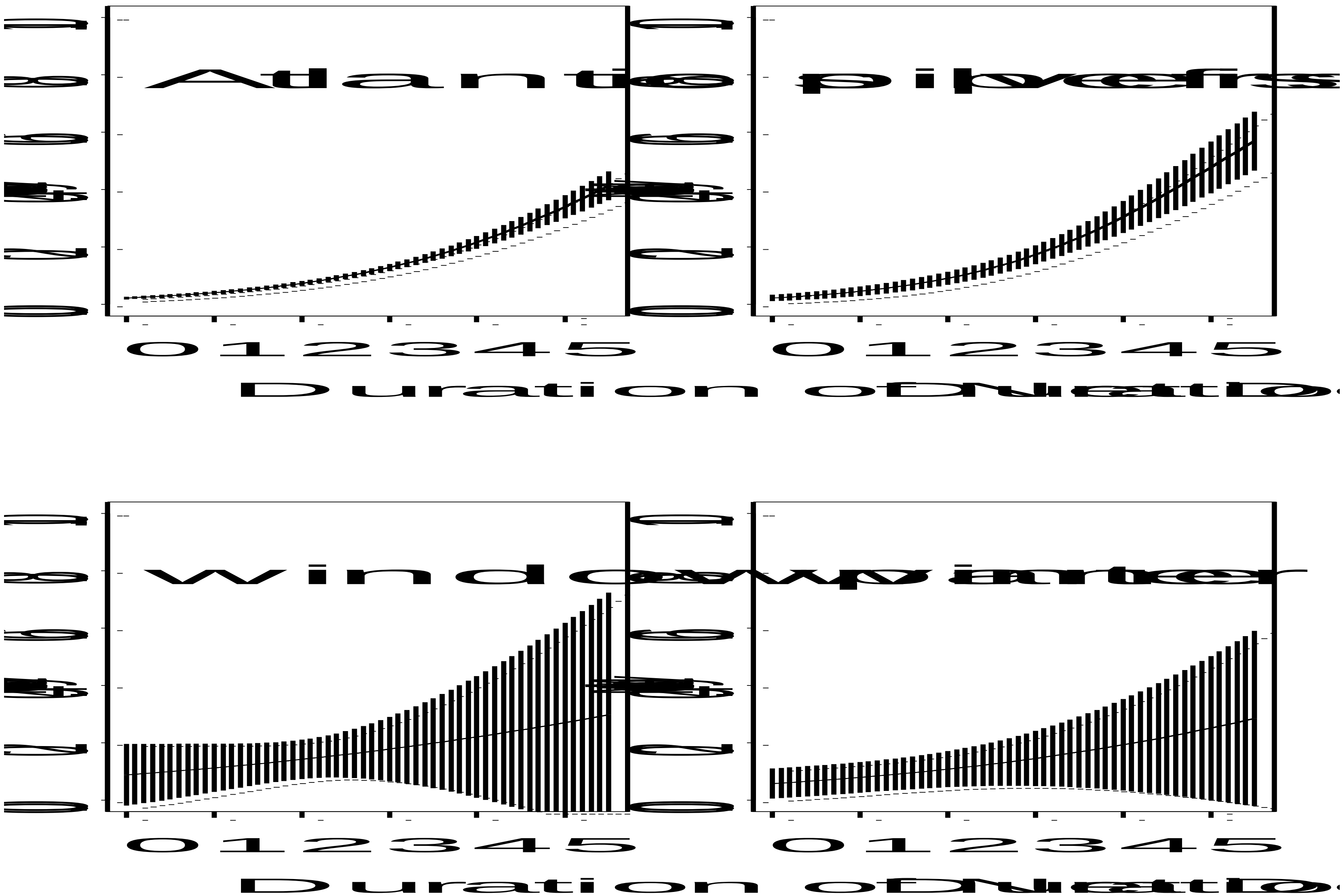


Figure 27. Acute mortality as a function of the duration of net deployment for Atlantic silversides, pipefish and flatfish. The y-intercept is an estimate of acute mortality in the absence of any time in the net, interpreted as acute turbine mortality. Error bars are standard errors.

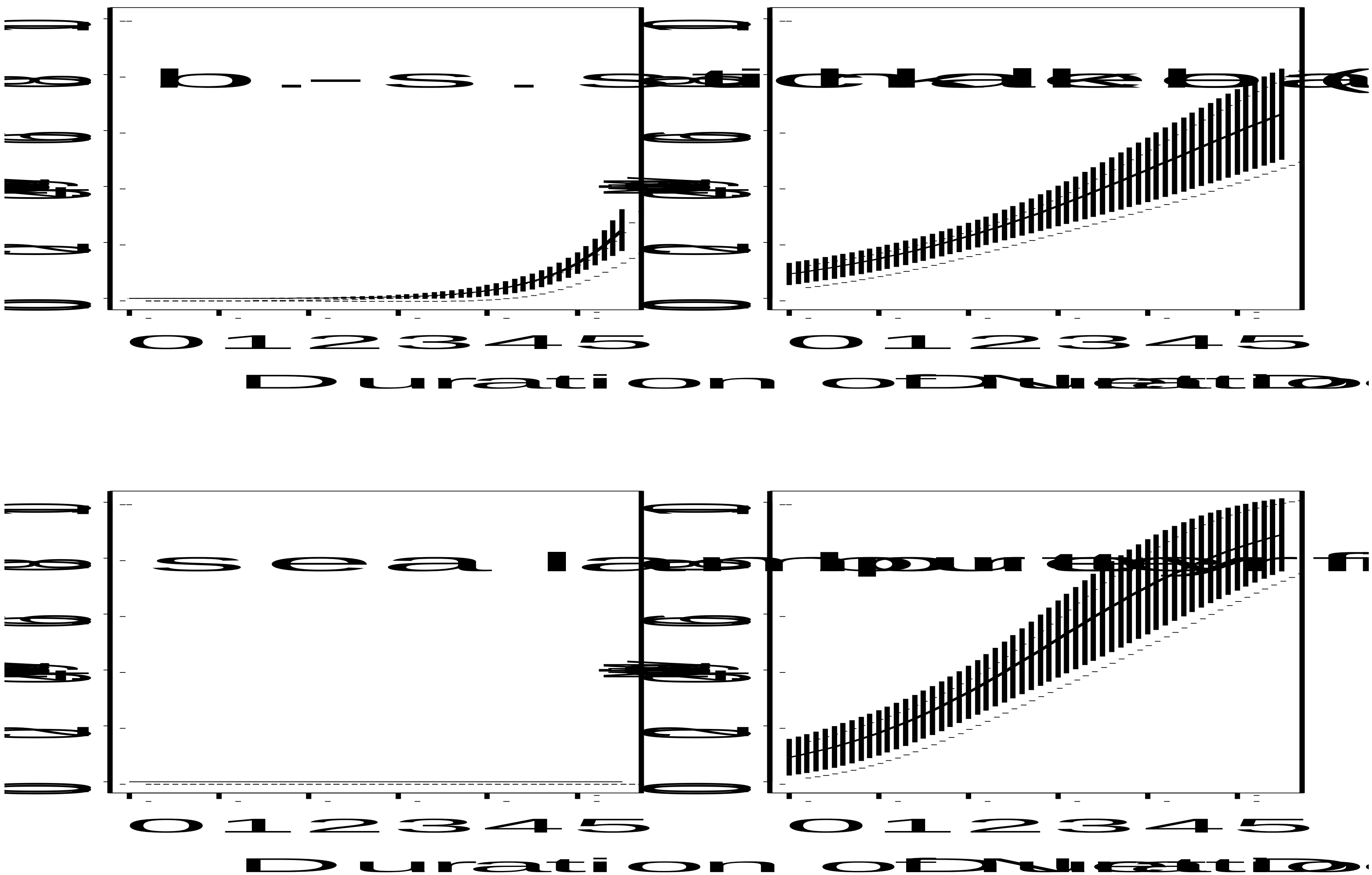


Figure 28. Acute mortality as a function of the duration of net deployment for sticklebacks, hake, lamprey and butterfish. The y-intercept is an estimate of acute mortality in the absence of any time in the net, interpreted as acute turbine mortality. Error bars are standard errors.

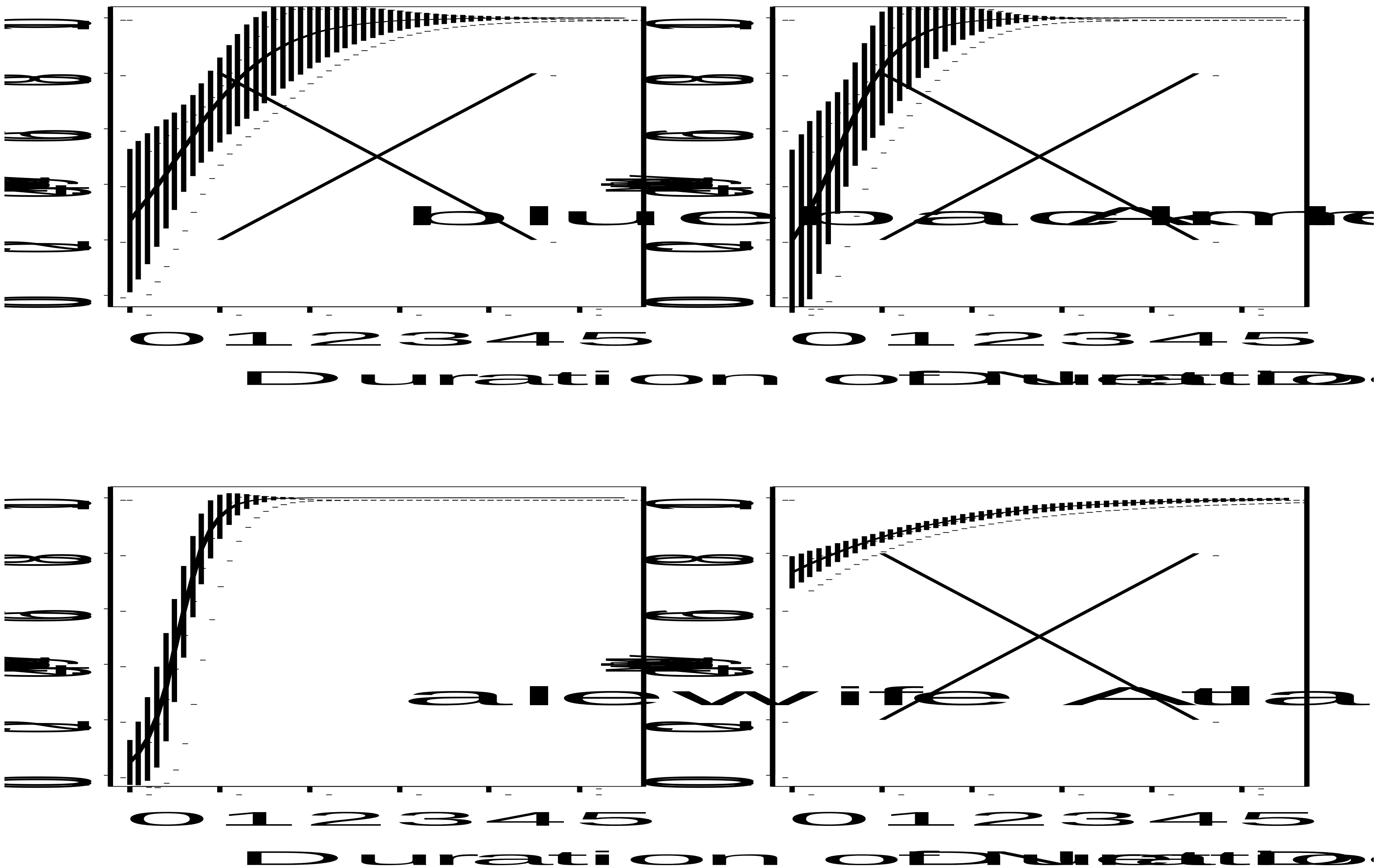


Figure 29. Mortality as a function of the duration of net deployment for clupeids, assessed after c. 13h. The y-intercept is an estimate of mortality in the absence of any time in the net, interpreted as turbine mortality. Error bars are standard errors. All estimates are biased high due to the absence of controls. Estimates from figures marked with a “x” are unbelievable (see text).

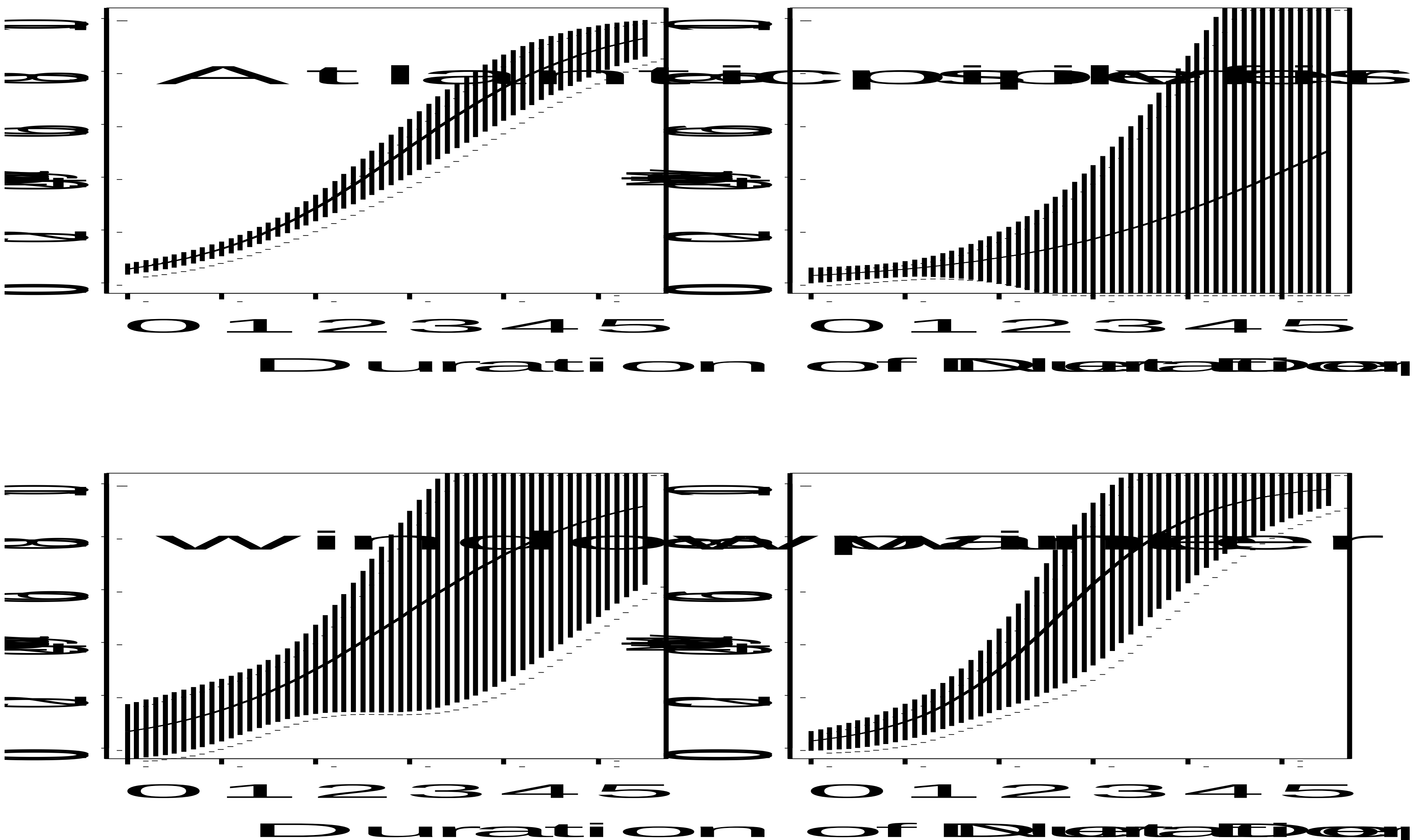


Figure 30. Mortality as a function of the duration of net deployment for Atlantic silversides, pipefish and flatfish, assessed after c. 13h. The y-intercept is an estimate of mortality in the absence of any time in the net, interpreted as turbine mortality. Error bars are standard errors. All estimates are biased high due to the absence of controls. Estimates from figures marked with a “x” are unbelievable (see text).

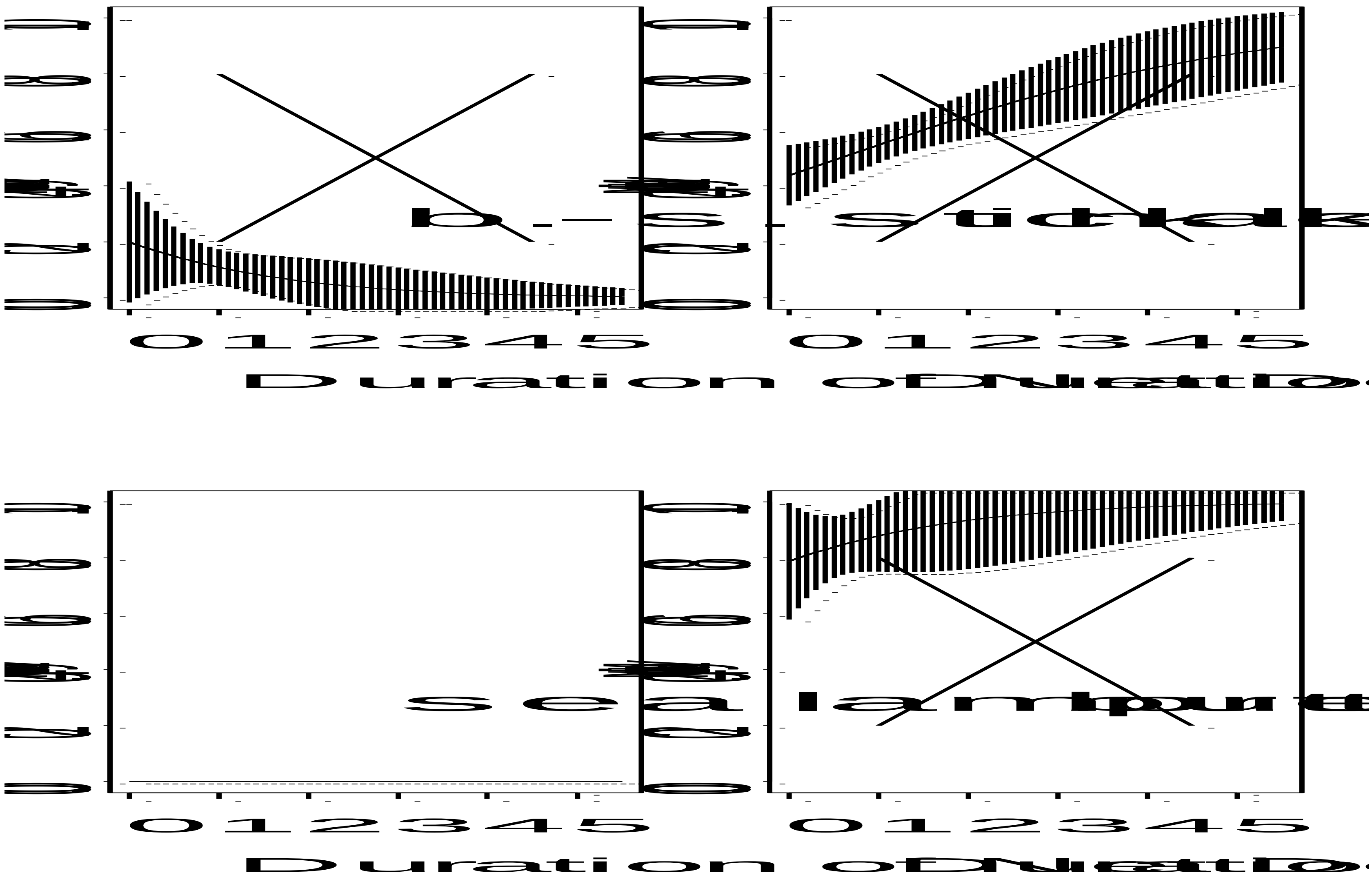


Figure 31. Mortality as a function of the duration of net deployment for sticklebacks, hake, lamprey and butterfish, assessed after c. 13h. The y-intercept is an estimate of mortality in the absence of any time in the net, interpreted as turbine mortality. Error bars are standard errors. All estimates are biased high due to the absence of controls. Estimates from figures marked with a “x” are unbelievable (see text).

APPENDIX 1 SOUND SYSTEM INSTALLATION AND SOUND FIELD MEASUREMENTS

Scientific Solutions, Inc.

Memo

To: Terry Toner, Nova Scotia Power, Inc.

From: Michael R. Birmann

CC: Fred Winchell, Alden Research Laboratory, Inc.

Date: November 23, 1999

Re: Sound System Installation and Sound Field Measurements

References:

1. Memorandum, from Fred Winchell to Terry Toner, 8/13/99 Re: Annapolis Sound Signal.

2. Memorandum, from Michael R. Birmann to Terry Toner, 8/24/99 Re: Transducer Deployment & Predicted Sound Fields

1.0 INTRODUCTION

On September 7th and 8th, 1999, a high frequency sound system was installed at the Annapolis Tidal Station intake. The purpose of the sound system is to project a band-limited (122 - 128 kHz) random noise signal into the intake canal of the generating station in support of a study to determine if such signals are effective at reducing turbine passage of juvenile fish of the *Alosa* species. Following installation of the system, acoustic output of the transducer array was verified by hydrophone measurements in the headpond.

2.0 SOUND SYSTEM

A block diagram of the sound system is shown in Figure 1. The system consists of 4 International Transducer Corporation Model 3406 transducers driven by an Instruments, Inc. Model L6 power amplifier. Input signals to the power amplifier were generated by a PC based system, using a Keithley Metrabyte Arbitrary Waveform Generator (AWFG) board. The 4 transducers are driven in parallel. The sound signal used is as described in [1,2] with the exception that the signal pulse was presented at a 33% duty cycle - a 0.5 second pulse followed by a 1.0 second blank interval. The location and orientation of the transducers was as presented in [2]. Figure 2 shows a measurement of the voltage signal applied to the transducers (measured at the output voltage monitor tap on the power amplifier). Figure 3 shows the spectrum of this measurement. A summation over frequency results in a total in-band applied voltage level of 42.7 dBV (rms). Using a transmit voltage response (TVR) of 138 dB re 1 uPa-m/Volt, the expected transducer source level is therefore 181 dB re 1 uPa @ 1 m. The predicted SPL contours presented in [2] were based on an anticipated transducer source level of 183 dB, so these must be decremented by 2 dB to represent the final sound system installation.

3.0 SOUND FIELD MEASUREMENTS

Once the sound system was operational, a series of hydrophone measurements were made across the intake canal to verify the acoustic output of the transducer array. A PC based data acquisition system running off a car battery and an inverter was deployed from the NSP workboat. A block diagram of the data acquisition system is given in Figure 4. The hydrophone received signal was sampled at 500 kHz. A 2.0 second record was displayed on the monitor screen, allowing one of the 0.5 second transmit intervals to be identified, the hydrophone record cropped to include the transmit interval, and the record stored to disk file. A program feature allowed the overall sound pressure level (SPL) of the received hydrophone signal to be calculated and recorded. Hydrophone spectra were computed at a later date.

The hydrophone measurements were performed on the morning of September 8, 1999, at slack water just before the start of the generating cycle. The flow was just beginning to turn at the fishway during the measurements. The water elevation during the measurements was approximately 101 meters. Figure 5 diagrams the measurement positions for the sampling of the sound field. A traverse was done across the intake canal, parallel to the intake face at a range of approximately 15 meters. The first traverse was performed with the hydrophone at a depth of 10 feet (hydrophone elevation approximately 98 m). Subsequently, a traverse was done on the centerline of the intake canal, starting at approximately 15 m out, then moving in to about 7.5 m, and finally in to about 3.75 meters. At each of these locations, measurements were recorded for hydrophone depths of 5, 10, and 15 feet. A final hydrophone measurement was made at a range of 2 meters, straight out from a transducer mounting pole at a depth of 10 feet. A matrix of measurement locations and measured sound pressure levels are given in Table 1.

The measurements show a variation in SPL between 155 dB and 160 dB for the lateral traverse at 10 ft depth. In comparison to the expected SPL contours based on the modeled sound fields in [2], these levels appear to be about 2 dB low. An examination of the axial traverse reveals a similar discrepancy. Based on these measurements, the 160 dB "threshold", as described in [1], is produced at a range of about 10 - 12 meters out from the intake face.

Measurement	Location *	Hydrophone Depth (ft)	Overall SPL (dB re 1 uPa (rms)
T1P1	P1	10	156.3
T1P2	P2	10	159.5
T1P3	P3	10	157.9
T1P4	P4	10	155.8
T1P5	P5	10	157.5
T1P6	P6	10	157.0
T1P7	P7	10	155.2
T1P8	P8	10	155.0
T1P9	P9	10	158.0
T1P10	P10	10	156.0
T1P2D2	P2	15	155.3
T1P2D3	P2	5	158.1
T1P12D3	P12	5	160.0
T1P12D1	P12	10	160.7
T1P12D2	P12	15	161.3
T1P13D1	P13	10	163.1
T1P13D2	P13	15	164.7
T1P13D3	P13	5	155.1
T1P14D1	P14	10	167.4

* Refer to Figure 5 for position chart

Table 1 - Measured Sound Pressure Levels

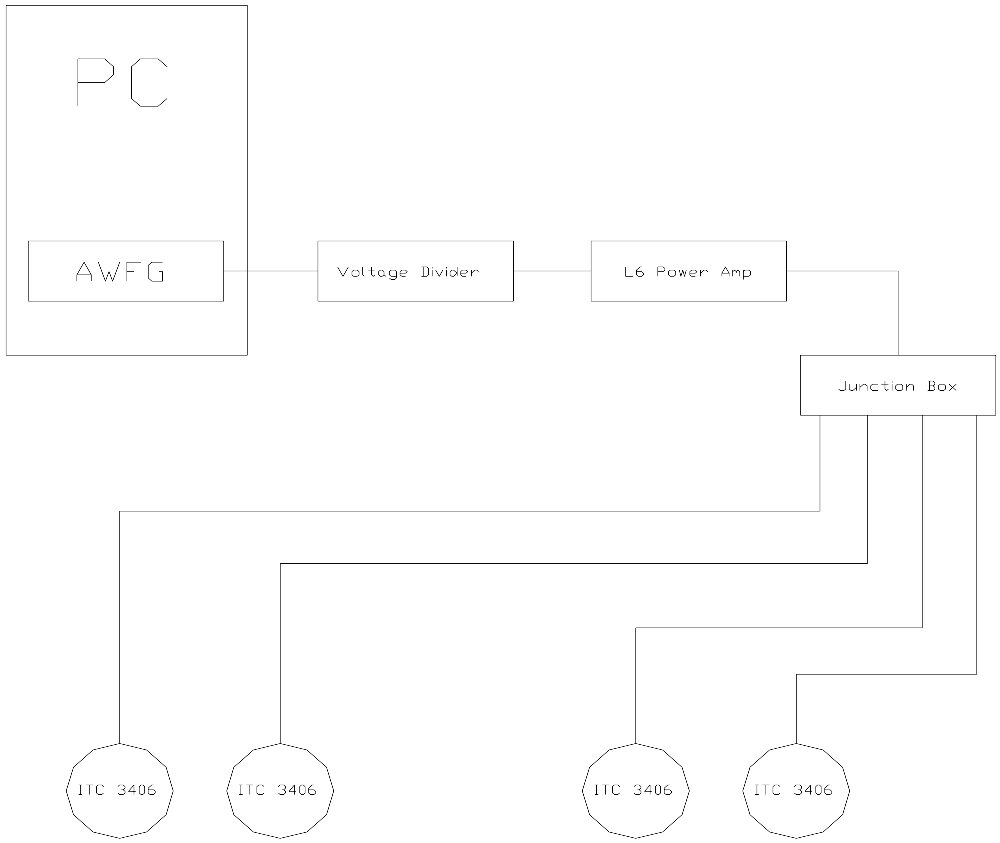


Figure 1 - Sound System Block Diagram

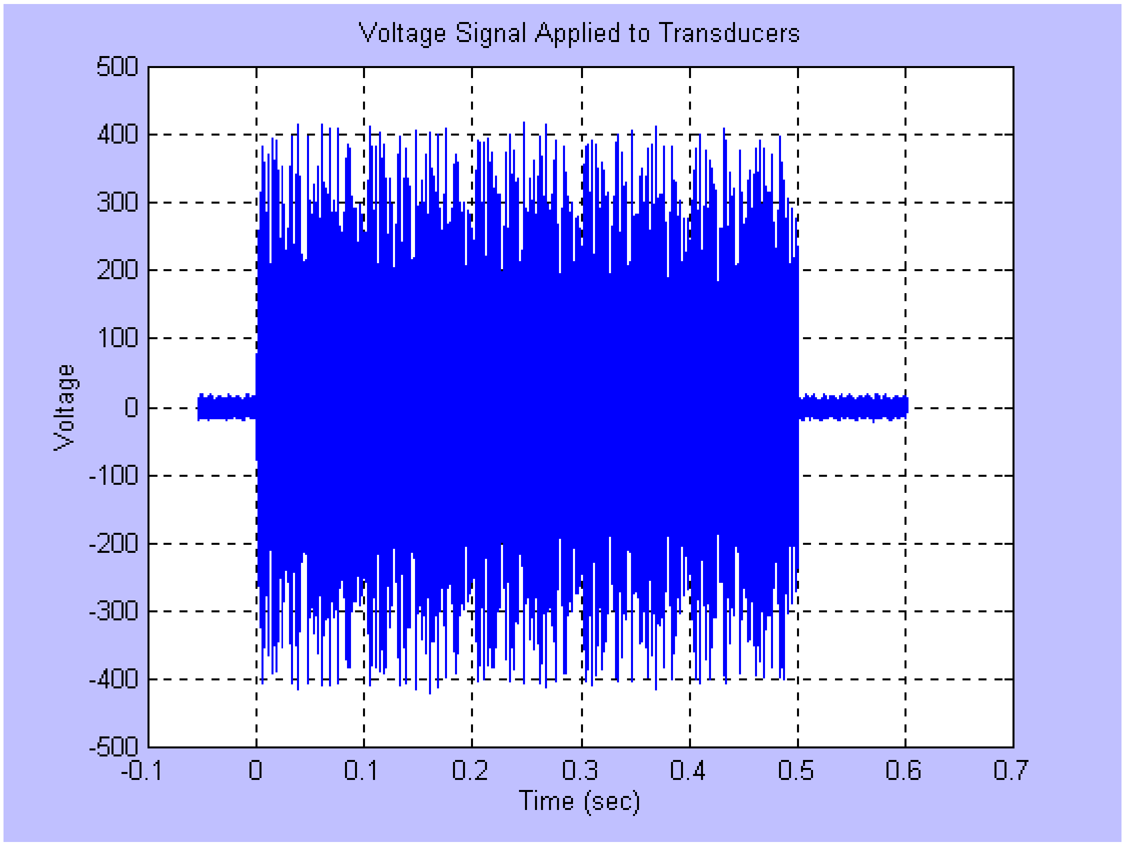


Figure 2 - Voltage Applied to Transducers

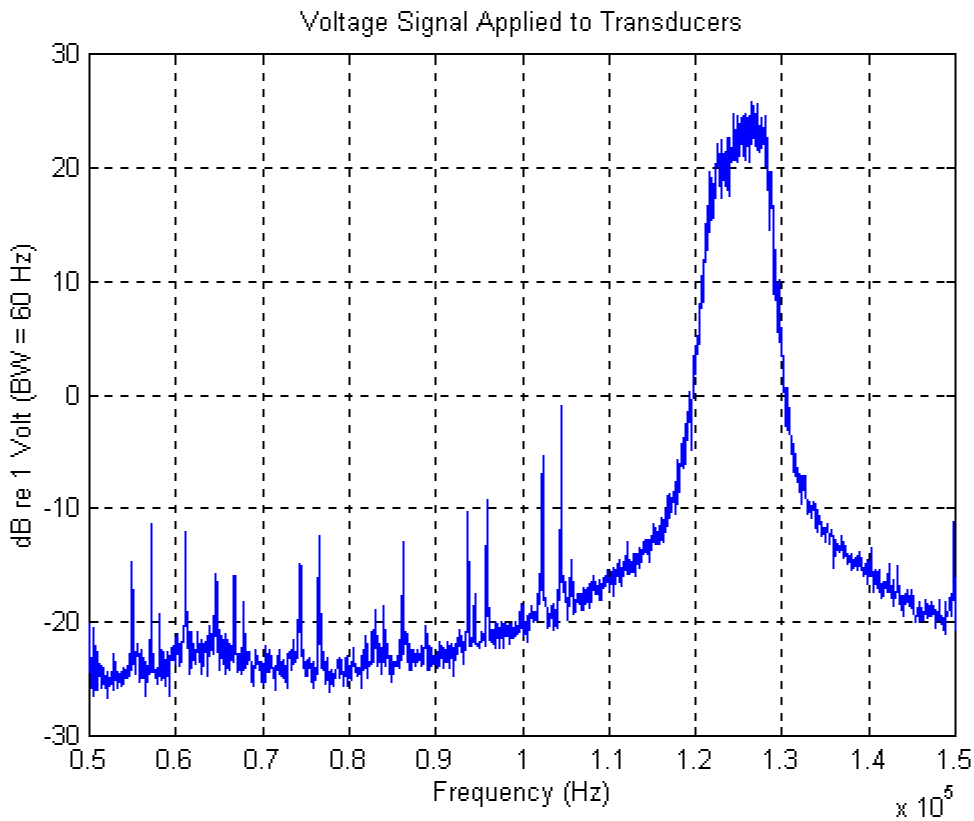


Figure 3 - Voltage Spectrum of Signal Applied to Transducers

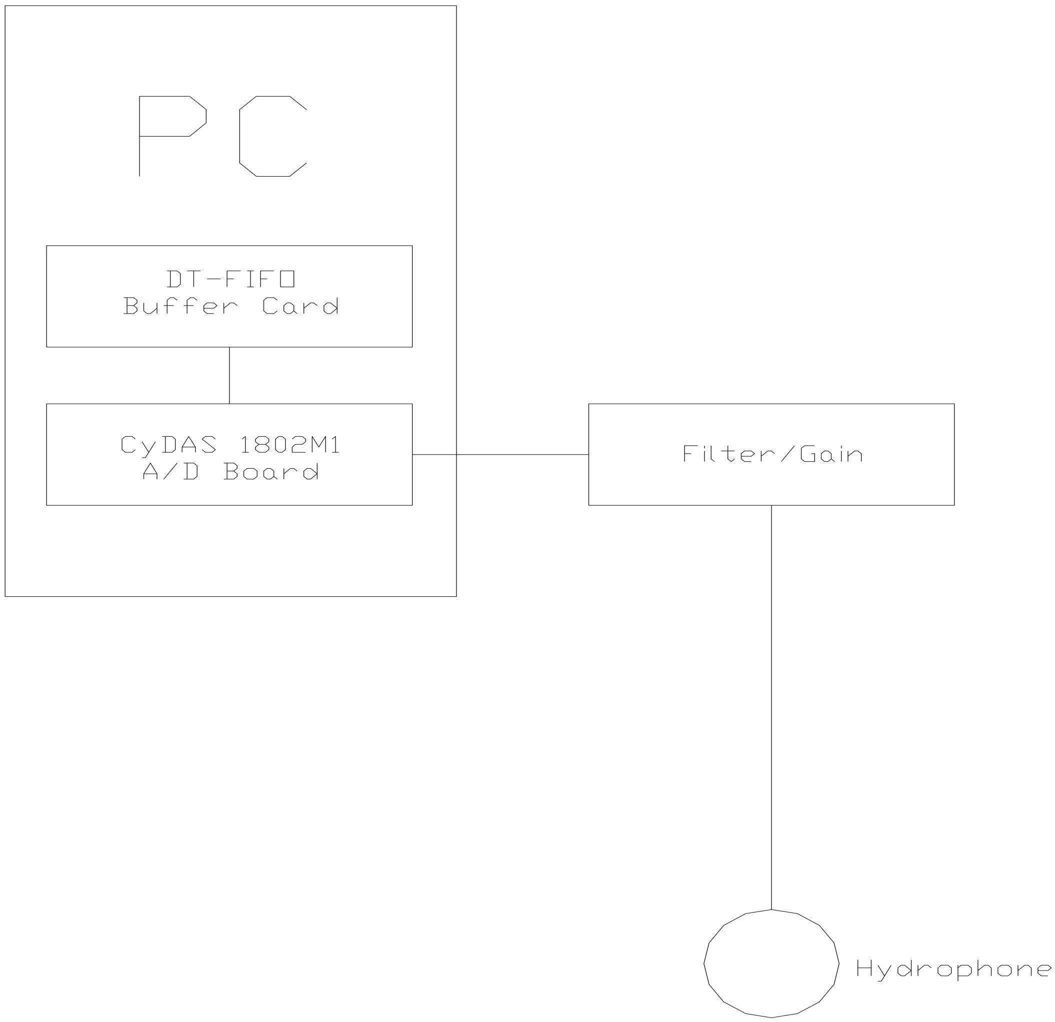


Figure 4 - Hydrophone Data Acquisition System

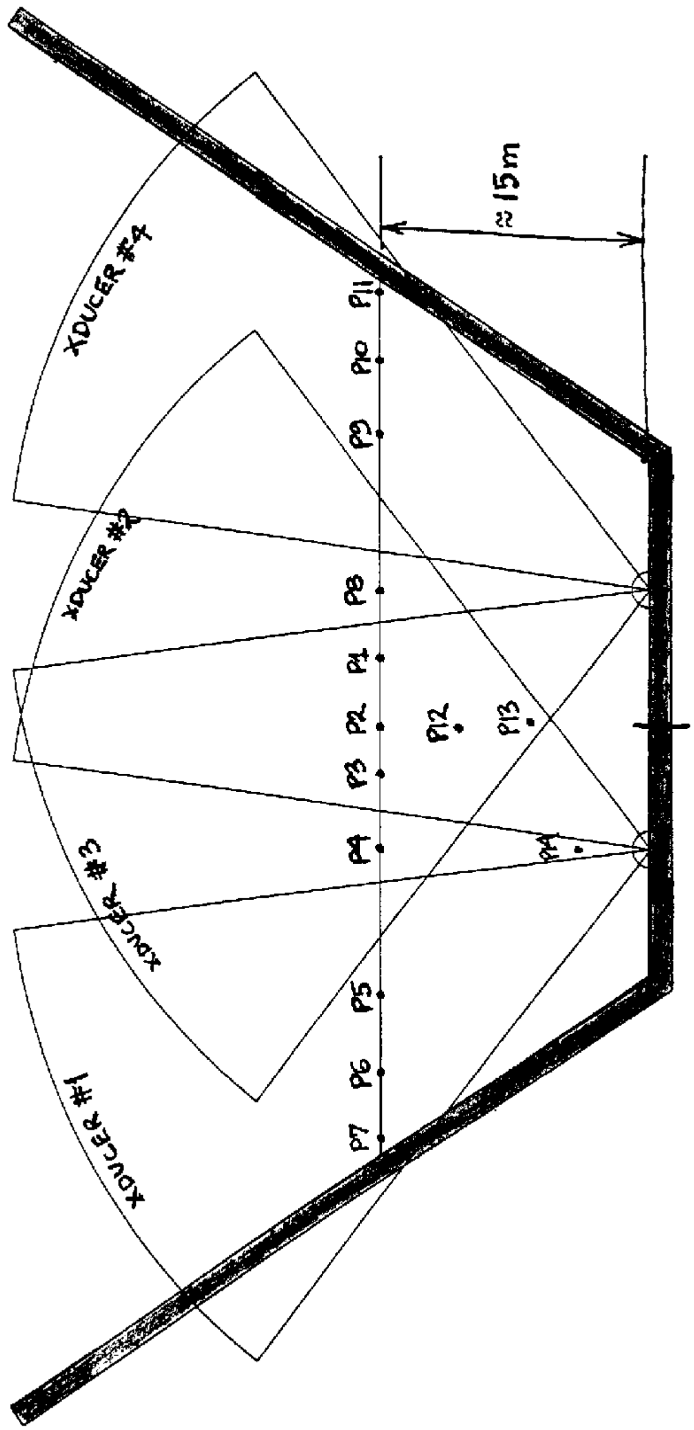
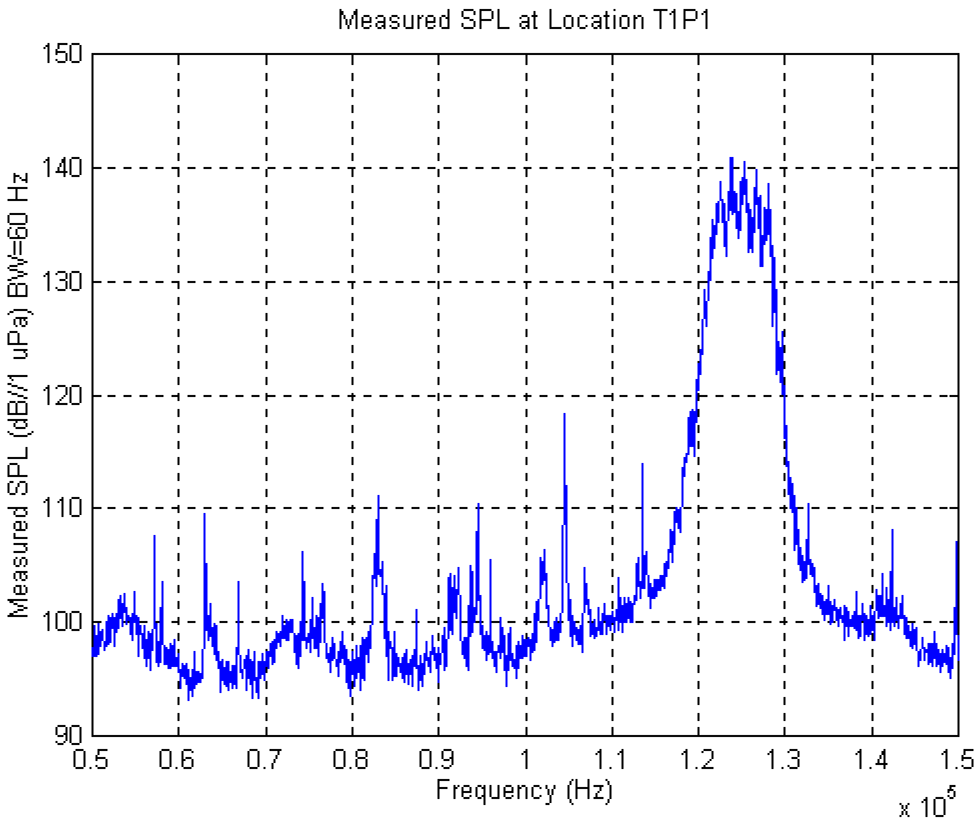
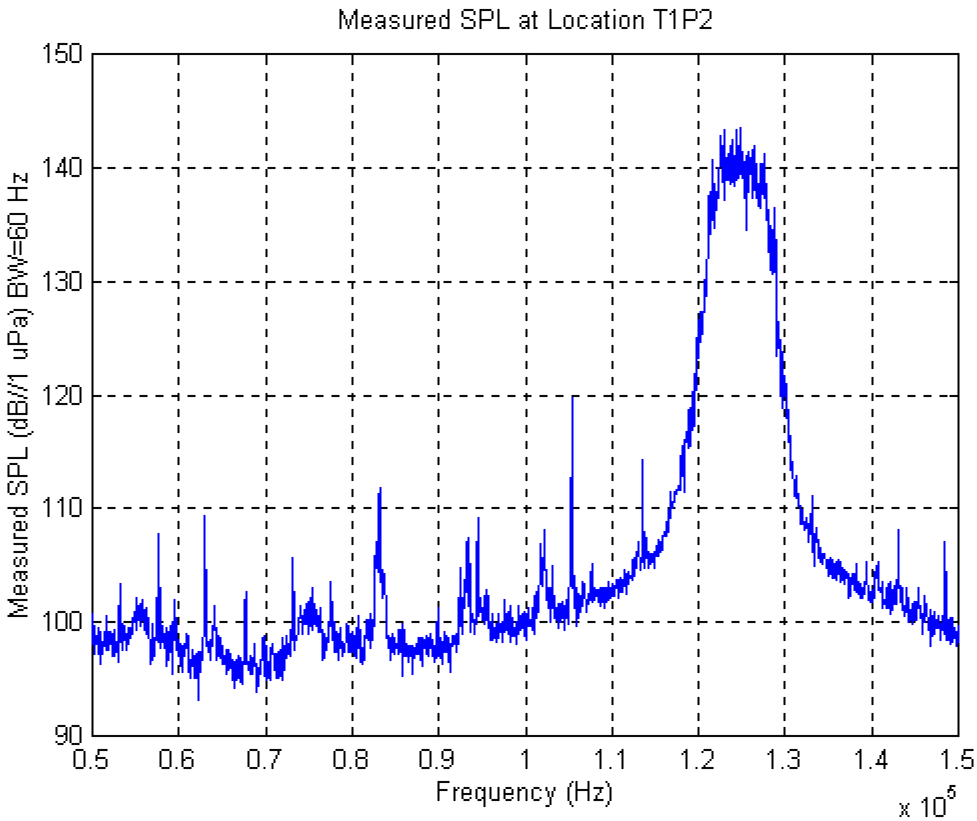


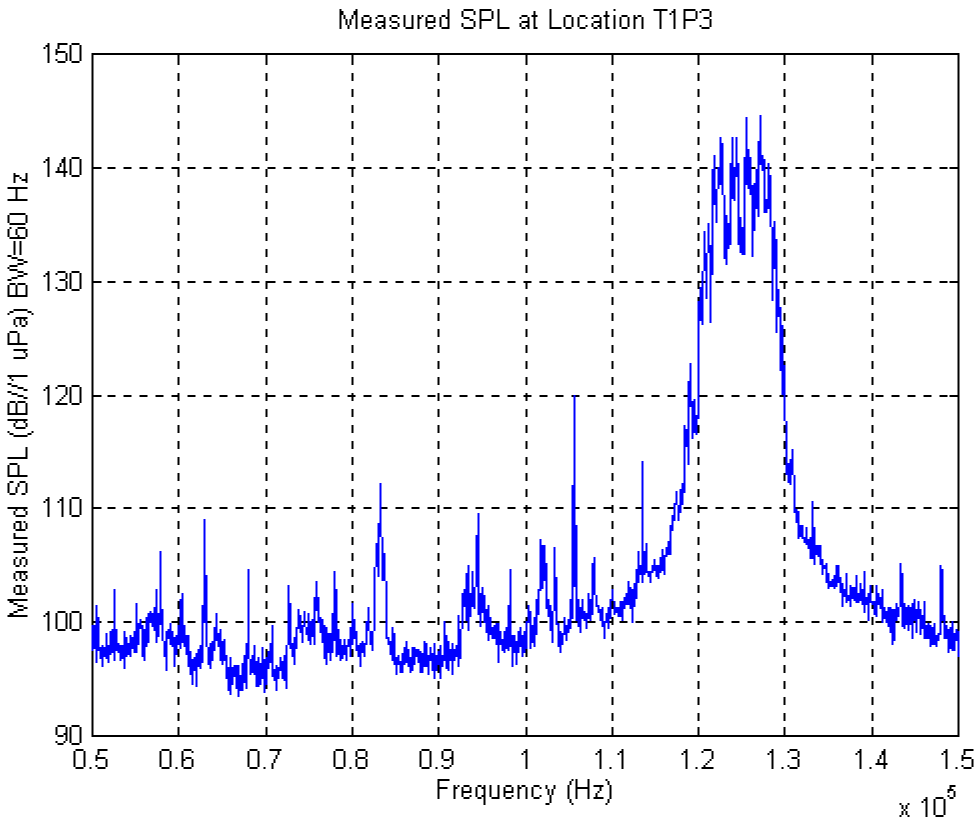
FIGURE 5 - SPL MEASUREMENT LOCATIONS

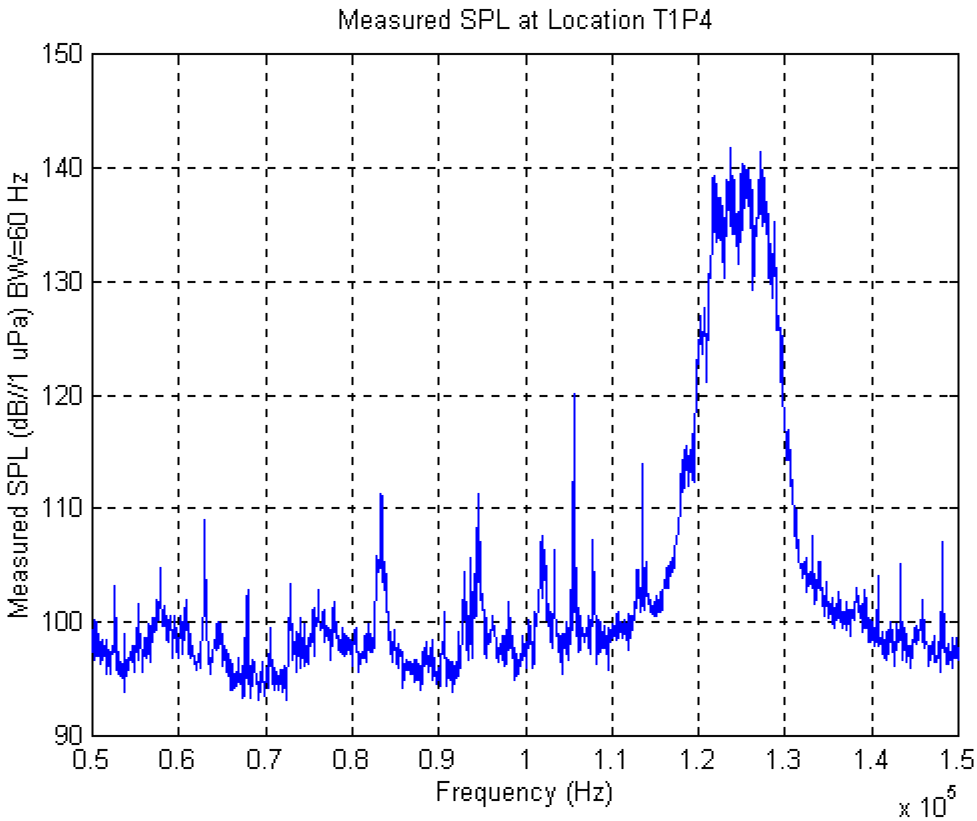
APPENDIX - SPL SPECTRUM PLOTS

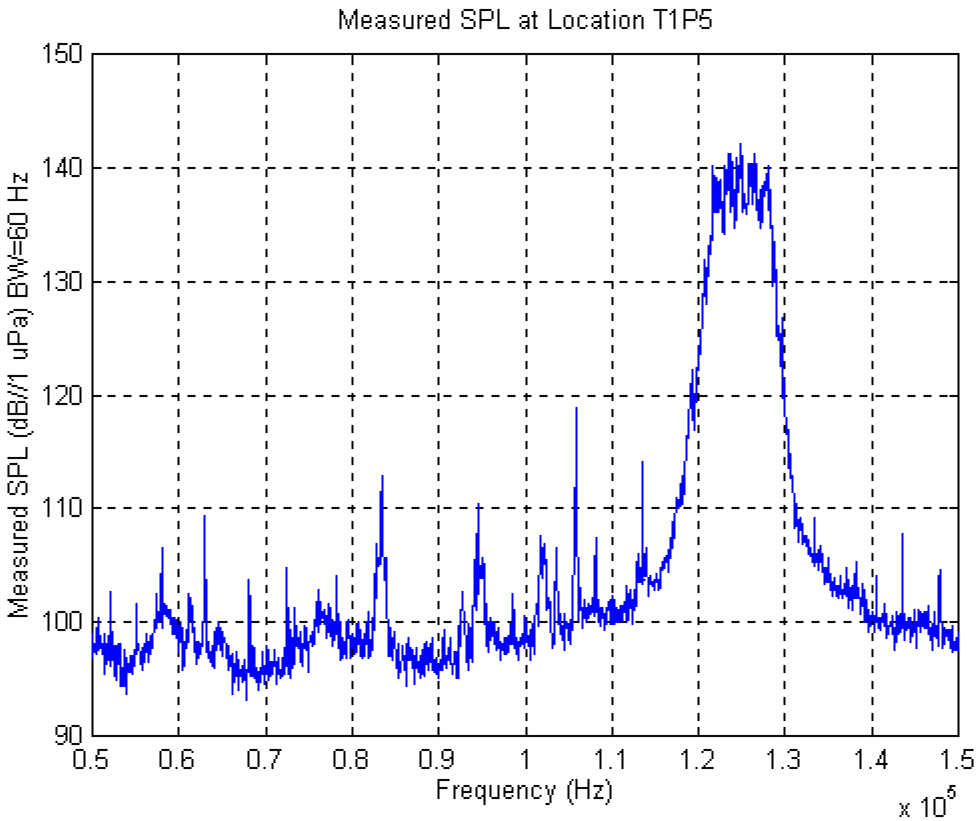


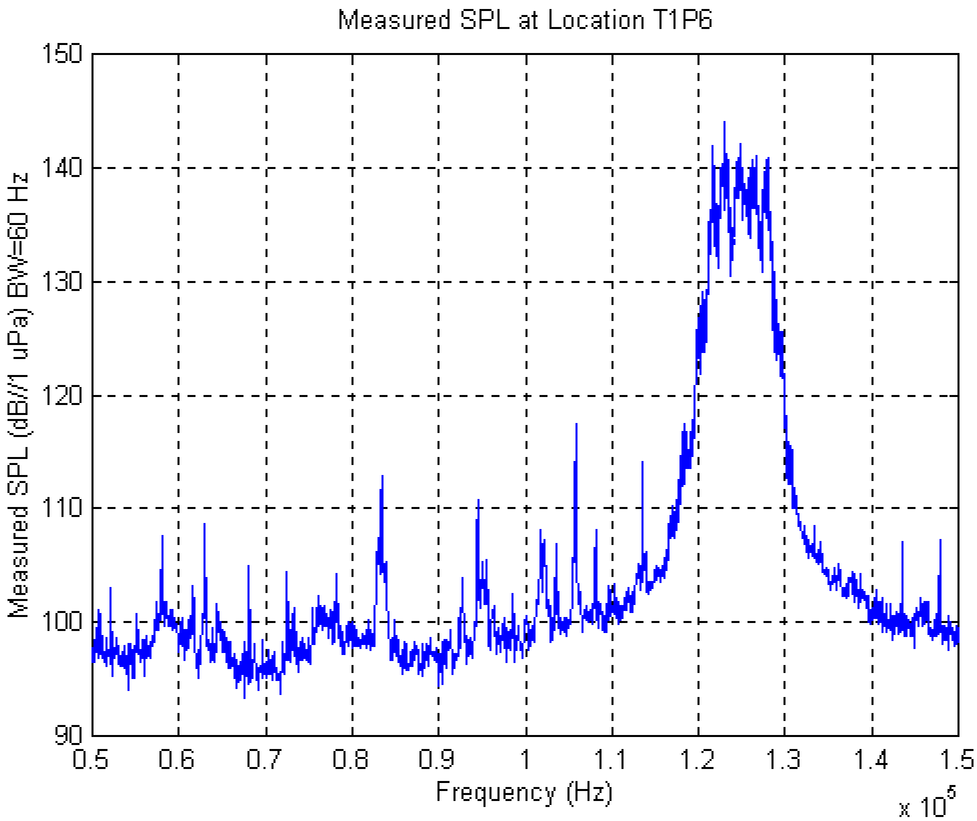
Measured SPL at Location T1P1
Measured SPL (dB//1 uPa) BW=60 Hz
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5

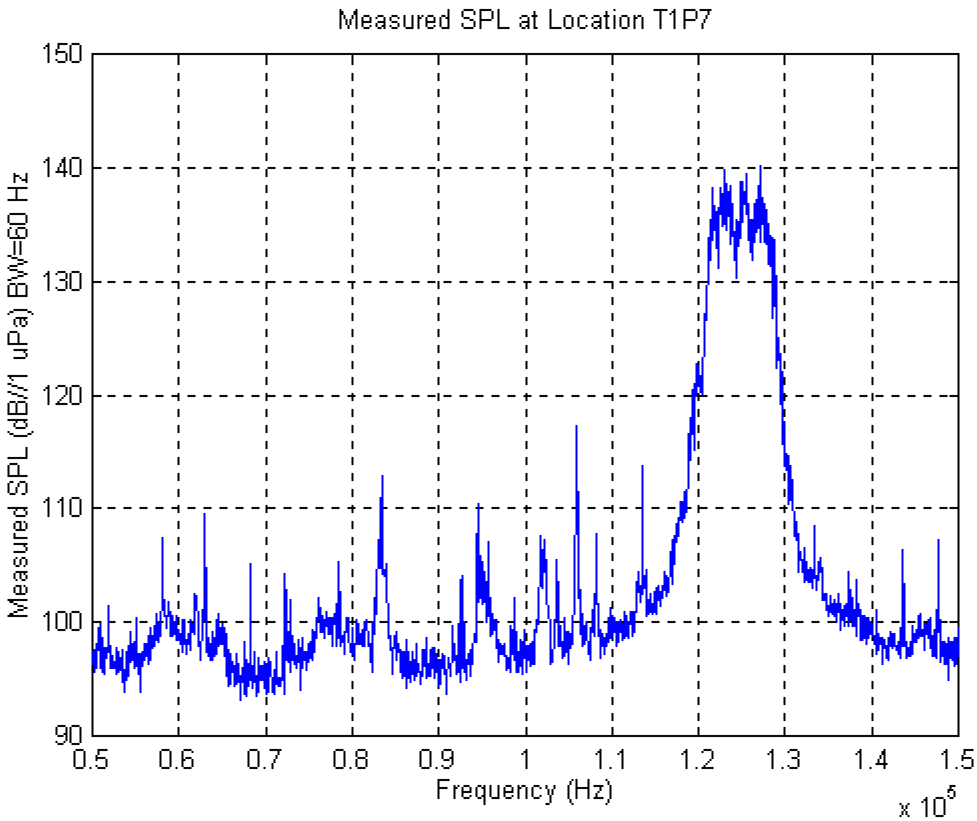

Measured SPL at Location T1P2
Measured SPL (dB//1 uPa) BW=60 Hz
Frequency (Hz)
x 10^5
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5


Measured SPL at Location T1P3
Measured SPL (dB//1 uPa) BW=60 Hz
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5

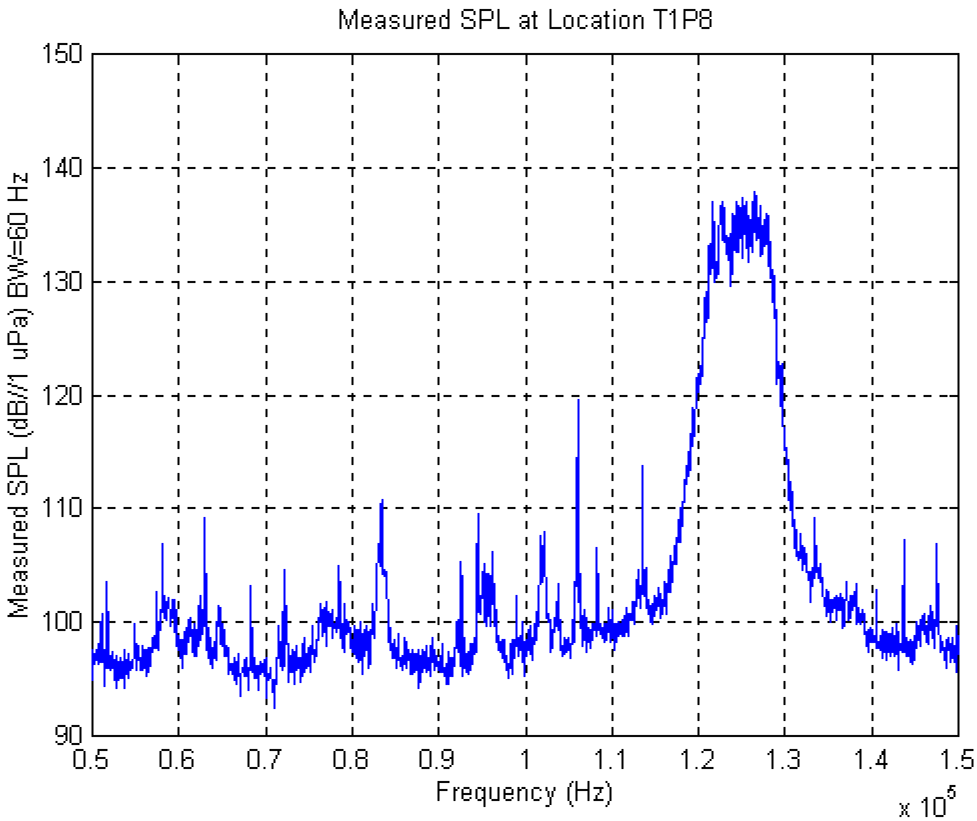

Measured SPL at Location T1P4
Measured SPL (dB//1 uPa) BW=60 Hz
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5

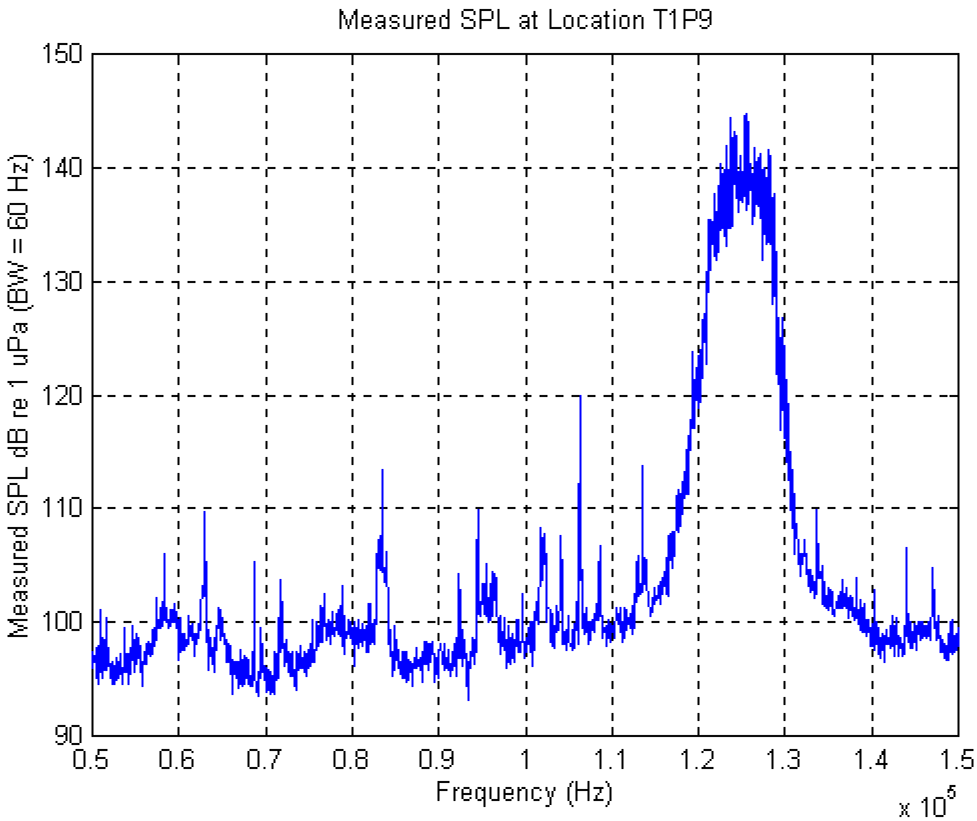

Measured SPL at Location T1P5
Measured SPL (dB//1 uPa) BW=60 Hz
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5

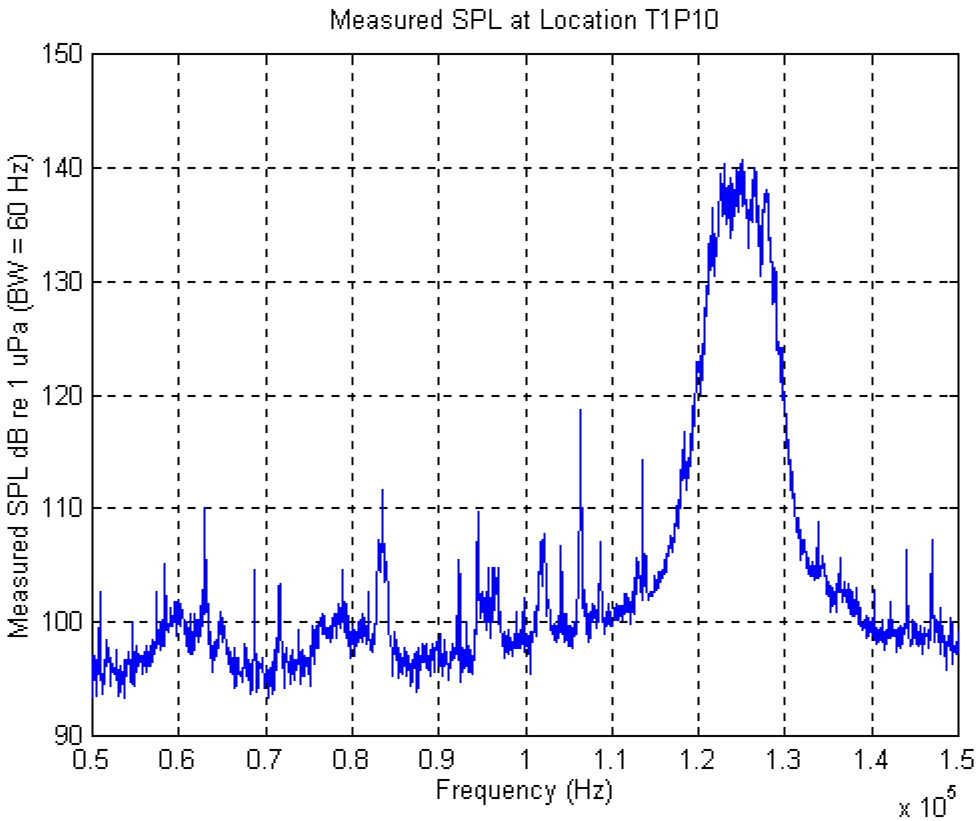

Measured SPL at Location T1P6
Measured SPL (dB//1 uPa) BW=60 Hz
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5

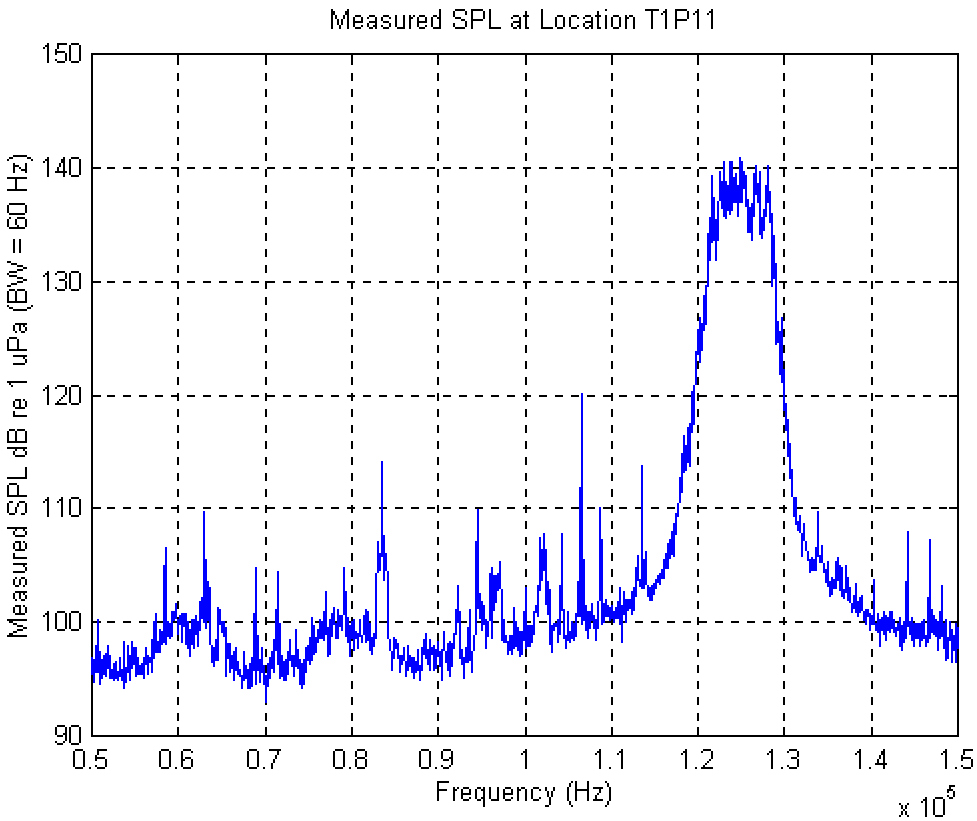


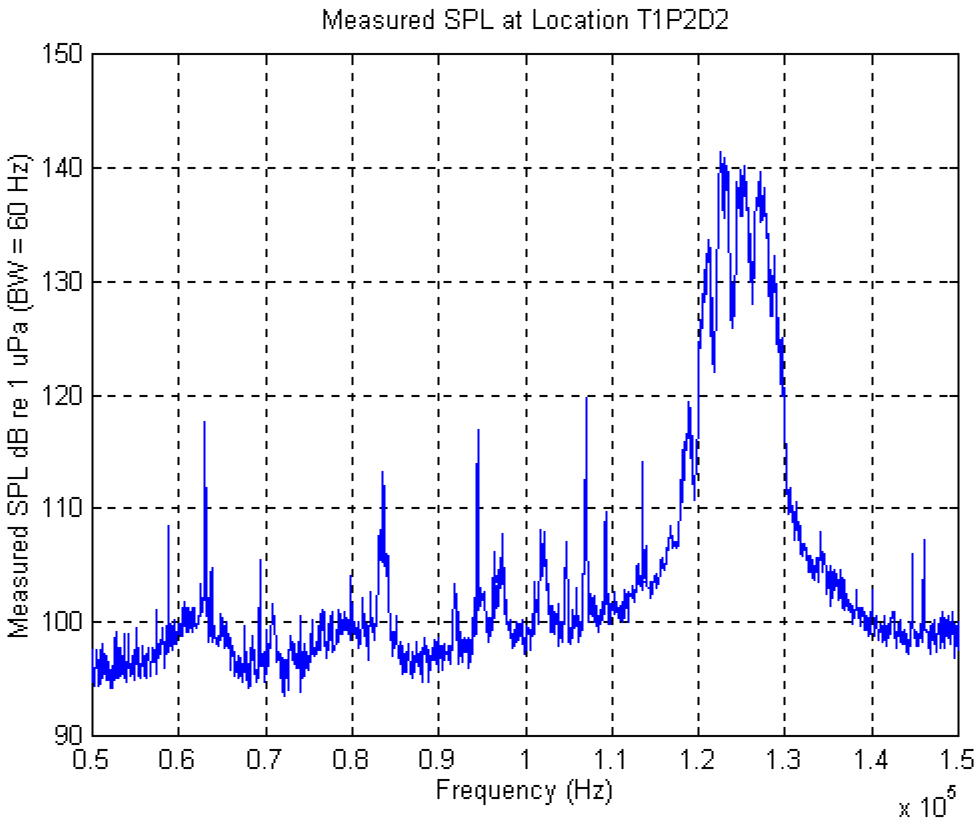
Measured SPL at Location T1P7
Measured SPL (dB//1 uPa) BW=60 Hz
Frequency (Hz)
x 10^5
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5

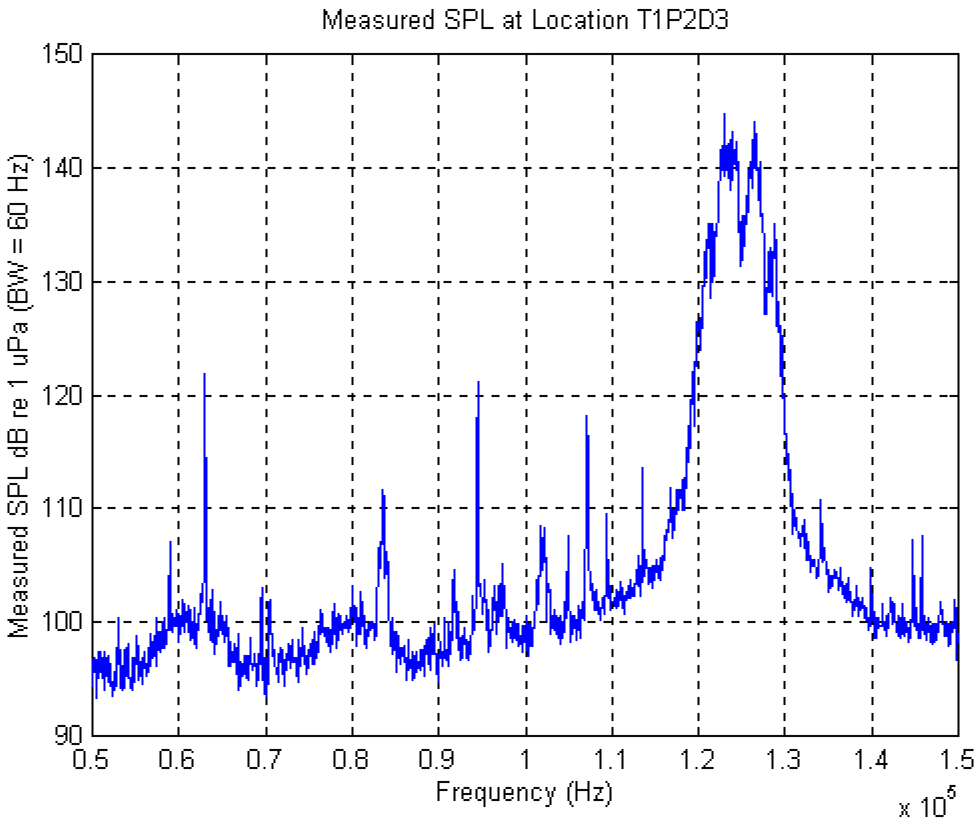

Measured SPL at Location T1P8
Measured SPL (dB//1 uPa) BW=60 Hz
Frequency (Hz)
x 10^5
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5


Measured SPL at Location T1P9
Measured SPL dB re 1 uPa (BW = 60 Hz)
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5

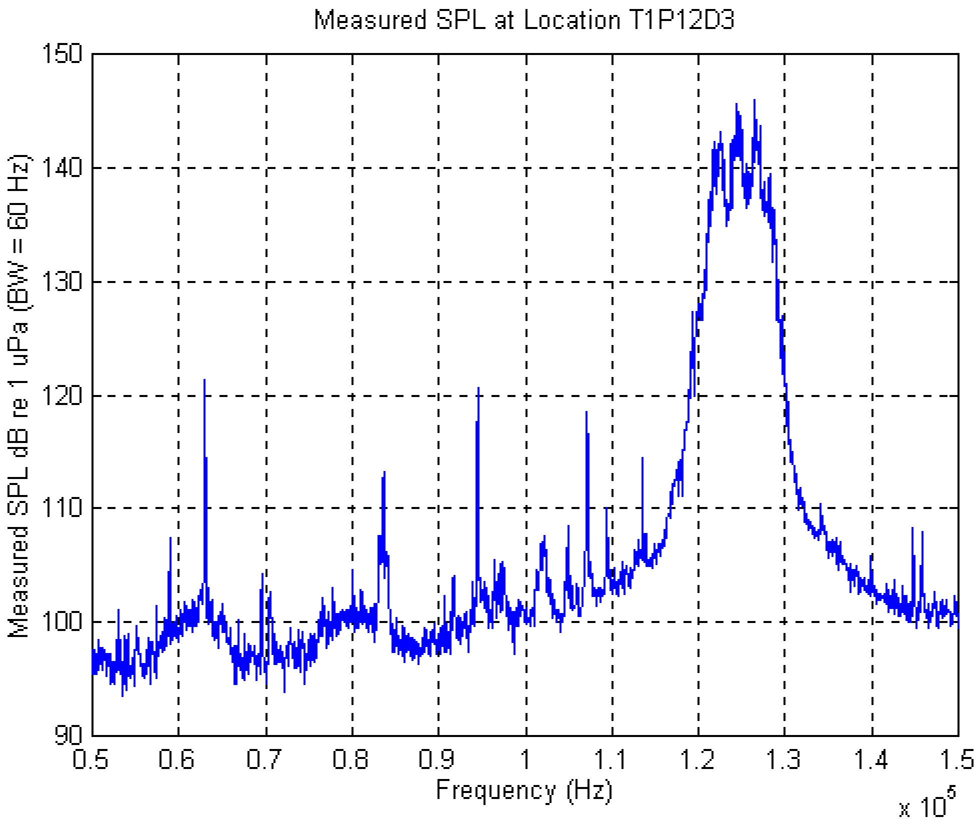

Measured SPL at Location T1P10
Measured SPL dB re 1 uPa (BW = 60 Hz)
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5


Measured SPL at Location T1P11
Measured SPL dB re 1 uPa (BW = 60 Hz)
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5

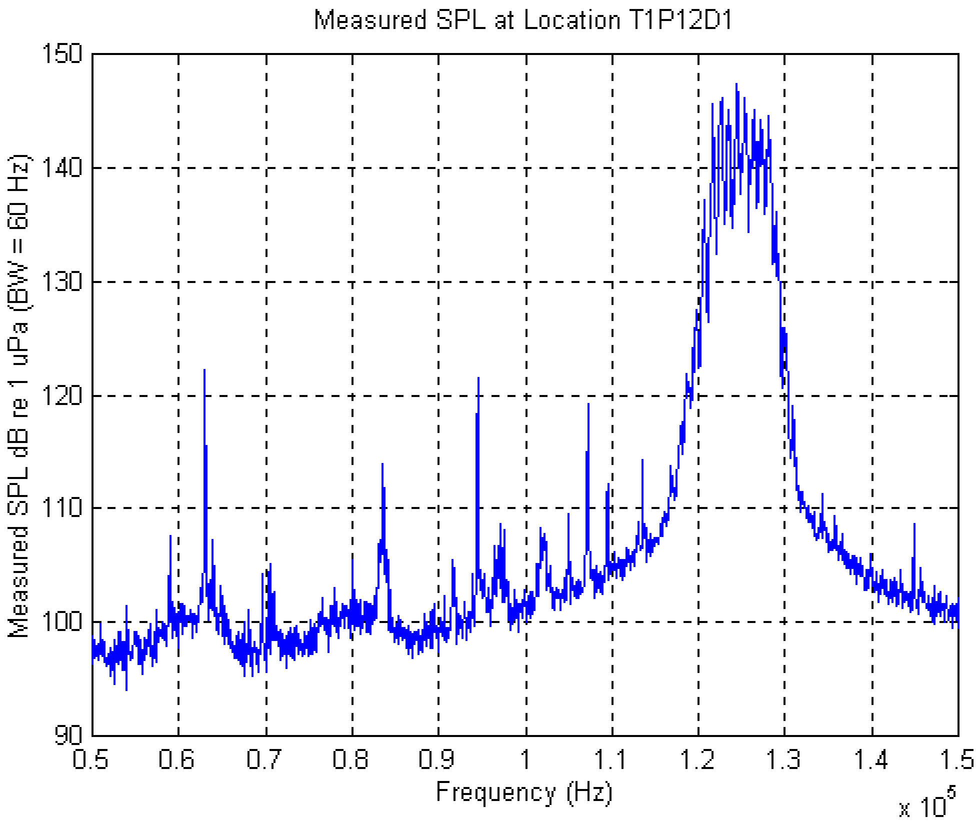

Measured SPL at Location T1P2D2
Measured SPL dB re 1 uPa (BW = 60 Hz)
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5



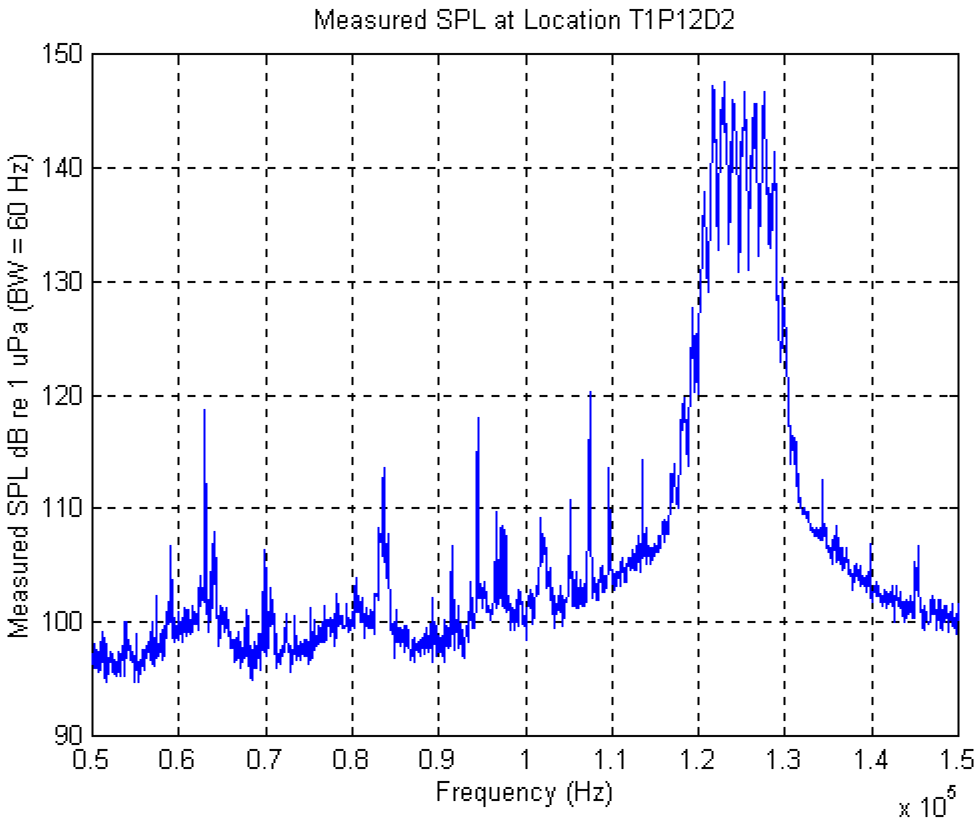
Measured SPL at Location T1P2D3
Measured SPL dB re 1 uPa (BW = 60 Hz)
Frequency (Hz)
x 10^5
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5



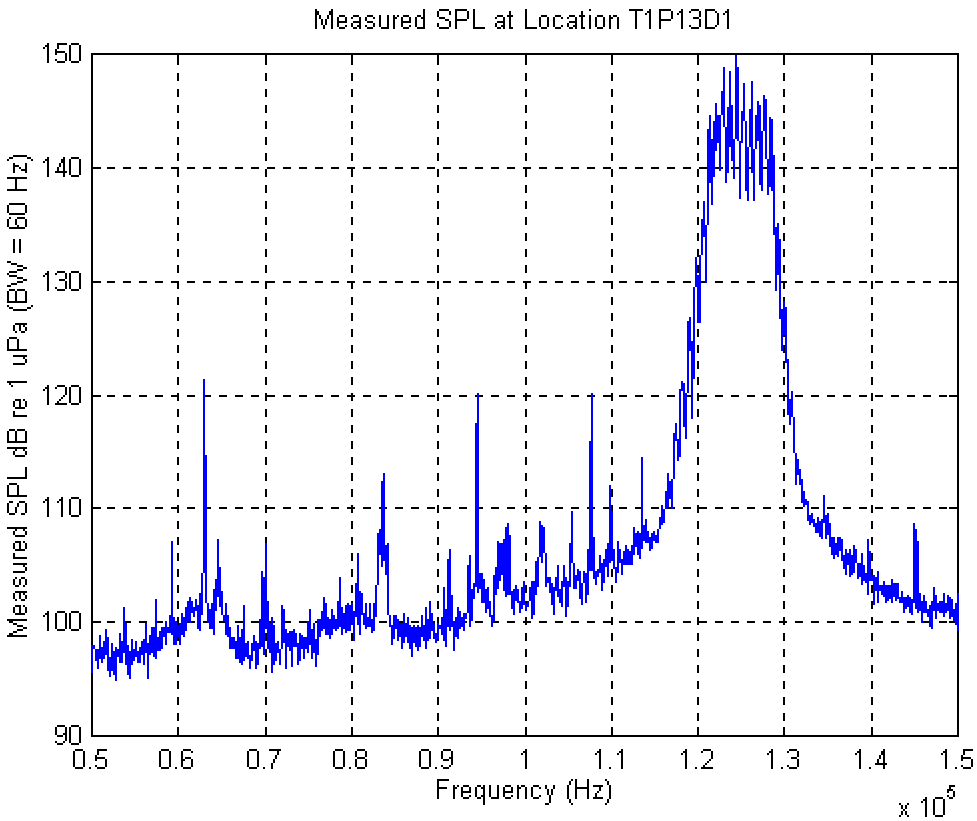
Measured SPL at Location T1P12D3
Measured SPL dB re 1 uPa (BW = 60 Hz)
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5



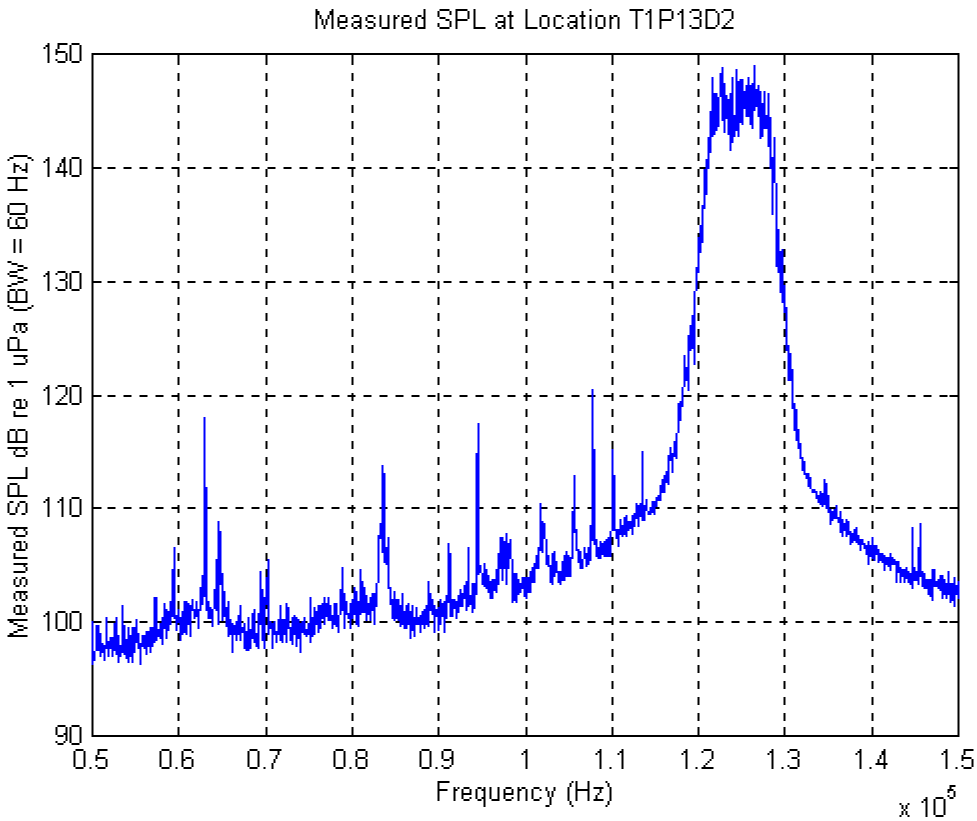
Measured SPL at Location T1P12D1
Measured SPL dB re 1 uPa (BW = 60 Hz)
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5



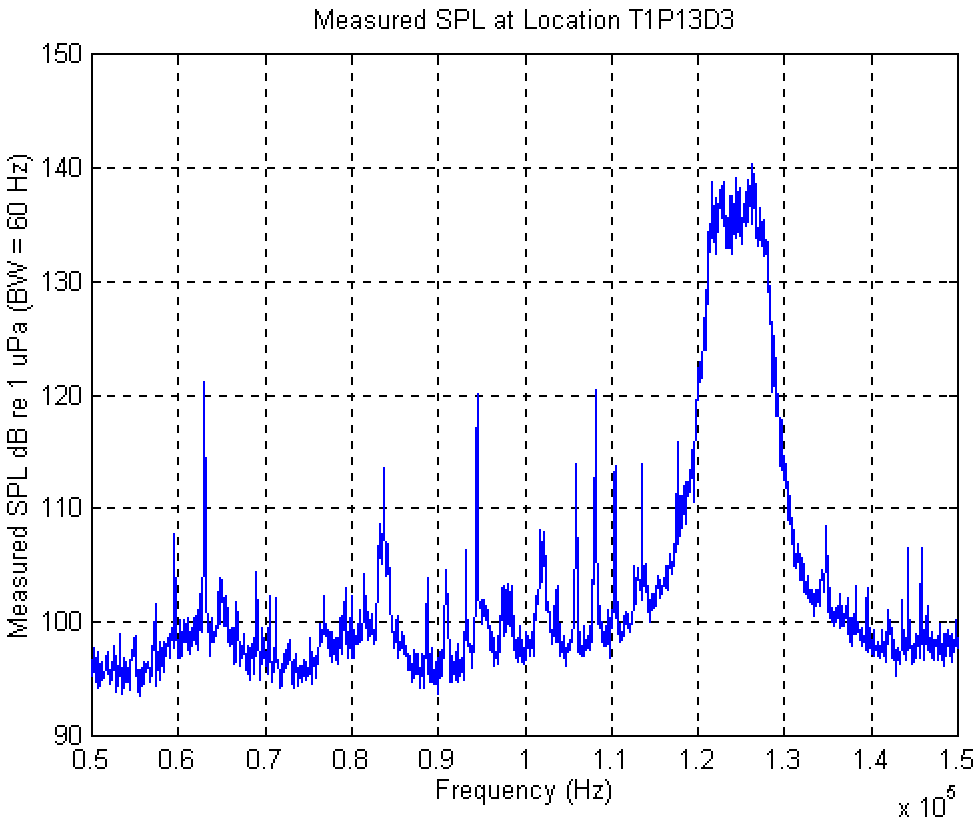
Measured SPL at Location T1P12D2
Measured SPL dB re 1 uPa (BW = 60 Hz)
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5



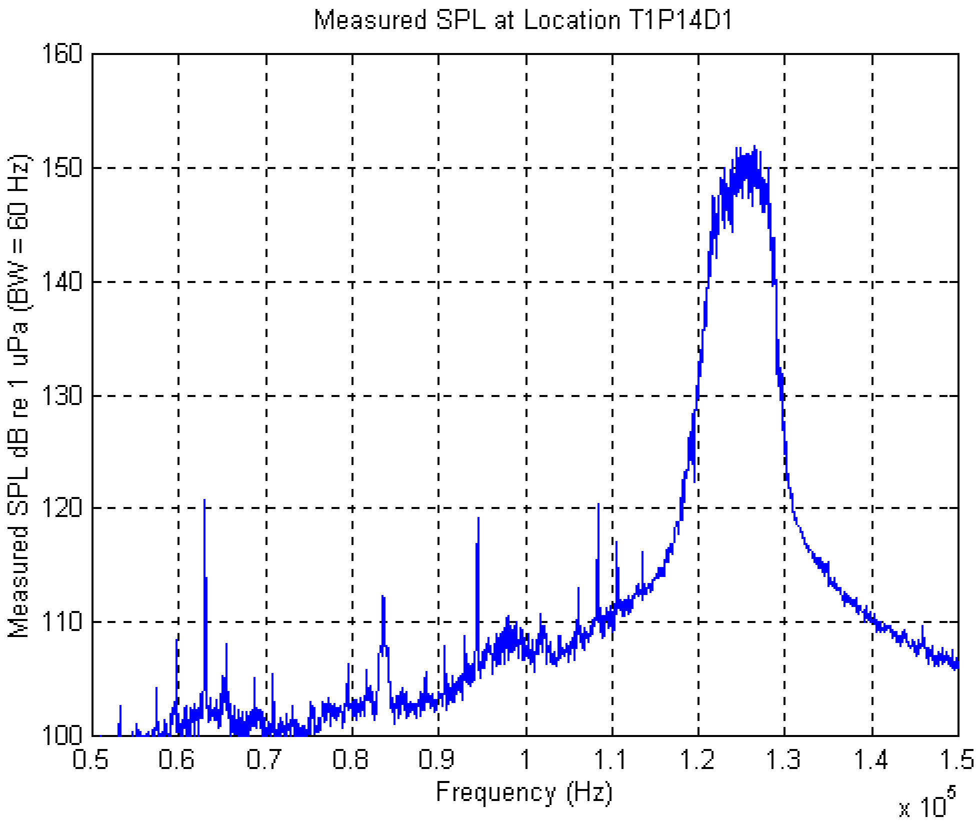
Measured SPL at Location T1P13D1
Measured SPL dB re 1 uPa (BW = 60 Hz)
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5



Measured SPL at Location T1P13D2
Measured SPL dB re 1 uPa (BW = 60 Hz)
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5



Measured SPL at Location T1P13D3
Measured SPL dB re 1 uPa (BW = 60 Hz)
150
140
130
120
110
100
90
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5



Measured SPL at Location T1P14D1
Measured SPL dB re 1 uPa (BW = 60 Hz)
160
150
140
130
120
110
100
0.5
0.6
0.7
0.8
0.9
1
1.1
1.2
1.3
1.4
1.5
Frequency (Hz)
x 10^5

APPENDIX 2 –SCIENTIFIC NAMES OF FISH MENTIONED IN THIS REPORT

Common Name	Scientific Name*
sea lamprey	*Petromyzon marinus*
American eel	*Anguilla rostrata*
blueback herring	*Alosa aestivalis*
alewife	*Alosa pseudoharengus*
American shad	*Alosa sapidissima*
Atlantic herring	*Clupea harengus harengus*
rainbow smelt	*Osmerus mordax*
hake spp.	*Urophycis sp.*
fourbeard rockling	*Enchelyopus cimbrius*
pollock	*Pollachius virens*
Atlantic silverside	*Menidia menidia*
blackspotted stickleback	*Gasterosteus wheatlandi*
pipefish	*Syngnathus fuscus*
white perch	*Morone americana*
striped bass	*Morone saxatilis*
bluefish	*Pomatomus saltatrix*
cunner	*Tautogolabrus adspersus*
wrymouth	*Cryptacanthodes maculatus*
Atlantic mackerel	*Scomber scombrus*
butterfish	*Peprilus triacanthus*
Meek's halfbeak**	*Hyporhamphus meeki*
flying gurnard	*Dactylopterus volitans*
longhorn sculpin	*Myoxocephalus scorpioides*
lumpfish	*Cyclopterus lumpus*
windowpane	*Scophthalmus aquosus*
smooth flounder	*Liopsetta putnami*
winter flounder	*Pseudopleuronectes americanus*

*from: Scott, W. B., and M. G. Scott. 1988 .Atlantic Fishes of Canada. Can. Bull. Fish. Aquat. Sci. 219: 731 p

** from: Banford, H.M. and B.B. Collette. 1993. *Hyporhamphus meeki*, a new species of halfbeak (Teleostei: Hemiramphidae) from the Atlantic and Gulf coasts of the United States. Proc. Biol. Soc. Wash. 106(2):369-384.