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A Logistic Regression Model for Estimating Turbine Mortality at Hydroelectric Generating Stations

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Abstract.—We present a method that allows separation of fish mortality caused by handling and capture techniques from that caused by passage through a turbine. Fish that are naturally entrained into the turbine tube are captured with nets deployed in the turbine tailrace for varying lengths of time. The live or dead status of captured fish is modeled as a binomial response that is a function of the duration of net deployment. Within this model, the intercept is an estimate of the mortality of fish that have spent zero time in the net. For species that do not suffer high mortality from other components of the capture process (such as removal from the net), this intercept may be interpreted as an estimate of turbine mortality. If mortality from other components is high, the intercept cannot be interpreted as turbine mortality without correction for mortality from the other sources. We suggest a modification to the model that allows estimation of mortality from these components. We demonstrate the method with data for 12 species of fish captured at the Annapolis Tidal Generating Station, Nova Scotia, Canada. Acute turbine mortality estimates ranged from 0.0% for sea lamprey *Petromyzon marinus* to 23.4% for American shad *Alosa sapidissima*.

Accurate estimates of turbine mortality at hydroelectric generating stations are fundamental for assessments of the fish-related impacts of these facilities. Though turbine mortality studies at hydroelectric stations are numerous, the results of these studies are often conflicting (Mathur et al. 1994), making it difficult to generalize about species or make comparisons between locations. Separation of mortality resulting from turbine passage from that resulting from capture and handling is a fundamental problem that may have lead to the overestimation of turbine mortality in some studies (Mathur et al. 1994).

Turbine mortality studies can be loosely divided into two groups: those that use fish released into the turbine intake (e.g., Hogans 1987; Dubois and Gloss 1993; Mathur et al. 1994) and those that use naturally entrained fish captured in nets in the turbine tailrace (e.g., Stokesbury and Dadswell 1991; Navarro et al. 1996). Attempts to use naturally entrained fish to estimate turbine mortality may be confounded by capture and handling mortality because mortality increases with time in the net (Stokesbury and Dadswell 1991). Determination of an appropriate duration for the control experiments is problematic, because the length of time the entrained fish are in the net is unknown (Gibson 1996).

Here, we suggest that the problem of control experiment duration can be overcome with a slight modification to the methods used in many studies. By varying the duration of net deployment, the probability that a captured fish is alive can be modeled as a function of the duration of the net deployment. The intercept of the resulting model can be interpreted as an estimate of the survival of fish that have spent zero time in a net, i.e. turbine mortality. To demonstrate this method, we present preliminary estimates of turbine mortality for several species of fish passing through the STRAFLO turbine at the Annapolis Tidal Generation Station (Annapolis Royal, Nova Scotia, Canada) and suggest a modification to the model to include a correction for capture and handling mortality.

Study Area

The Annapolis Tidal Generating Station has operated since 1984 in the Annapolis River estuary. The station was constructed to test the feasibility

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of using the STRAFLO turbine in marine environments, in anticipation of larger-scale tidal generating development in the upper Bay of Fundy (Dadswell et al. 1986). The station only generates when water is running seaward, and operates within a head range of 1.4–6.8 m (Douma and Stewart 1981). The head varies with the stage of the tide. The 7.6-m-diameter turbine runs at 50 rotations per minute (rpm). Output at a head of 5.5 m is 17.8 MW, with a corresponding discharge of 408 m^3/s. Two fishways currently exist to facilitate fish passage at the station. For a 0.3-m head, discharge is 42.7 m^3/s for the fishway located near the sluice gates and 10.1 m^3/s for the fishway located near the turbine tube (Stokesbury and Dadswell 1989). At least 35 species of fish are present in the vicinity of the Annapolis Tidal Generating Station at some time during the year. The majority of fish moving seaward past the station are thought to travel through the turbine tube (Stokesbury 1987; Gibson 1996).

Previous fish mortality studies at this site suggest that 21.3 ± 19.8% (mean ± 95% confidence interval) of adult, postspawning American shad *Alosa sapidissima* do not survive passage through the turbine (Hogans 1987) and that age-0 clupeid turbine mortality is 46.3% (Stokesbury and Dadswell 1991). While the accuracy of these estimates has been debated (Dadswell and Rulifson 1994; Gibson 1996), no studies exist that conclusively support or disprove them. Survival of other species through the Annapolis Tidal Generating Station turbine has not been studied.

Methods

When naturally entrained fish are used to estimate turbine mortality, selection of an appropriate duration for control experiments is problematic because the length of time that a captured fish is held in the net is unknown and varies between fish within a net deployment and between net deployments. As the duration of the net deployment increases, variability and uncertainty also increase. In the limiting case, that of a deployment of zero duration, the duration of the net deployment equals the length of time that a fish is in the net. While a net cannot be deployed for a time period of zero, survival at this limit can be estimated by varying the duration of the net deployment and using the statistical methods described below.

Field methods.—Sampling for this study was integrated into an assessment of the effectiveness of an ultrasound fish diversion system (Gibson and Myers 2000) at the Annapolis Tidal Generating Station during the fall of 1999. As such, fish captured in the tailrace during the diversion system assessment were used as test specimens for modeling mortality.

We captured naturally entrained fish with modified ichthyoplankton nets at two locations in the turbine tailrace (Figure 1). The 1.0-m-diameter nets consisted of three sections: a 2.0-m-long cylindrical section made of 1.0-cm-mesh nylon netting; a 3.0-m-long middle section made of 2.0-mm Nitex net, tapering from 1.0 m to 0.2 m in diameter 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 that were 0.75 m in length. Entrances to the collectors were funnel-shaped to keep fish from escaping from the net, and the tail ends of the collectors were designed so that they could be opened and closed with drawstrings. Nets were fished for the full generation period (about 5.5 h), but the catch was removed from the nets and placed in buckets at predetermined intervals ranging from 0.25 to 5 h throughout the generation period. The catch was identified and enumerated, and the live or dead status of each fish was recorded about 10 min after removing the fish from the net. The resulting data set was used to estimate acute mortality (the probability that a fish was dead at the time of entering the net) for each species captured in sufficient numbers.

Sampling was conducted during 48 randomly chosen generating cycles between 7 September and 22 October 1999, covering over 50% of the generating time between these dates. The time of day or night varied with the precession of the tides. The interval at which nets were emptied was held constant throughout a generating cycle, in order to evenly distribute the number of fish captured over the time intervals. Nets were deployed a total of 447 times during the study period.

Fish that were in very poor condition (e.g., not swimming and bleeding badly) were counted as dead. While this introduces an element of subjectivity, any resulting bias produces an increased, more precautionary estimate of mortality.

Statistical analysis.—We assumed that the fish that were dead (either at the time of capture or shortly thereafter) died as a result of turbine passage, capture in the net, or some interaction between the two variables. For each species, we treated the live or dead status of each fish (Y) as a binomial response (0 = alive, 1 = dead), and used the logistic model (Collett 1991),

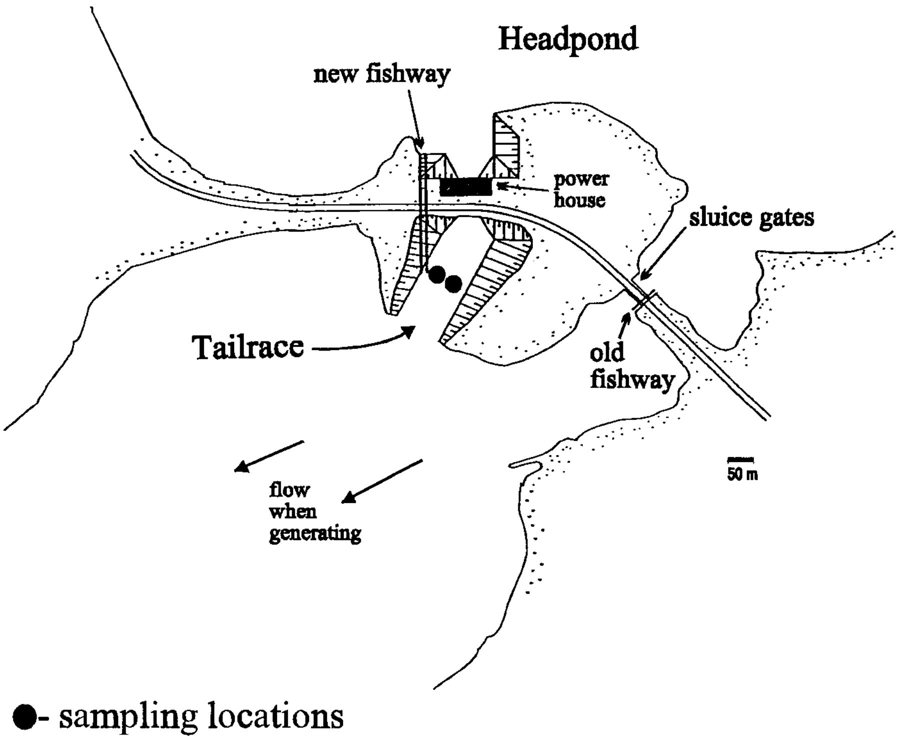

FIGURE 1.—Map of the Annapolis Tidal Generating Station, Annapolis Royal, Nova Scotia, Canada, showing the sampling locations used during this study (figure adapted from Ruggles and Stokesbury 1990).

$$E(Y) = \frac{\exp(\beta_0 + \beta_1 d)}{1 + \exp(\beta_0 + \beta_1 d)}, \tag{1}$$

to model $E(Y)$, the expected probability that a captured fish will be dead as a function of d, the duration of net deployment. Within this model, β_0 and β_1 are the linear regression coefficients on a logistic scale. An estimate of the probability that a fish spending zero time in the net will be dead ($M_{d=0}$) can be calculated as

$$M_{d=0} = \frac{\exp(\beta_0)}{1 + \exp(\beta_0)}. \tag{2}$$

This approach to estimating turbine mortality requires extrapolation outside the range of observed data, which is potentially hazardous with binary data if the model formulation is inappropriate (McCullagh and Nelder 1989). We examined the robustness of our estimates in relation to model selection in two ways: by adding a power parameter (λ) to control the degree of nonlinearity in the model and by examining two other transformations as outlined below.

An assumption of the model (equation 1) is that mortality is linearly related to the duration of net deployment on the logistic scale. Equation (1) is the specific case, where λ equals 1.0, of the general model,

$$E(Y) = \frac{\exp(\beta_0 + \beta_1 d^\lambda)}{1 + \exp(\beta_0 + \beta_1 d^\lambda)}. \tag{3}$$

We examined the linearity assumption by using maximum likelihood to fit equation (3) to the acute mortality dataset for λ values ranging from 0.01 to 3.0 and by using profile log likelihoods as a guide for selecting an appropriate λ. The log likelihood for λ, β_0, and β_1 is given by

$$l(\lambda, \beta_0, \beta_1) = \sum_i \left\{ y_i \log\left(\frac{\exp(\beta_0 + \beta_1 d_i^\lambda)}{1 + \exp(\beta_0 + \beta_1 d_i^\lambda)}\right) + (1 - y_i)\log\left[1 - \left(\frac{\exp(\beta_0 + \beta_1 d_i^\lambda)}{1 + \exp(\beta_0 + \beta_1 d_i^\lambda)}\right)\right]\right\},$$

and the log profile likelihood for $\lambda[l_p(\lambda)]$ is

$$l_p(\lambda) = \max_{\beta_0, \beta_1} l(\lambda, \beta_0, \beta_1).$$

The maximum likelihood estimate of λ occurs where $l_p(\lambda)$ achieves its maximum value. The plausibility of other possible values of λ was evaluated by comparing their log likelihoods with the maximized log likelihood. A likelihood ratio-based 95% confidence interval for λ is calculated as

$$\lambda: 2[l_p(\lambda^{\mathrm{MLE}}) - l_p(\lambda)] \leq \chi_1^2(0.95)$$

where λ^{MLE} is the maximum likelihood estimate of λ (Kalbfleisch 1985). The profile log likelihoods suggested that $\lambda = 1$ was appropriate (see Results), and we therefore retained equation (1) as the model.

The logistic model (equation 1) has the advantage that it can be fitted as a generalized linear model (McCullagh and Nelder 1989) with three components: (1) a random component where Y is a binomial random variable with $E(Y) = u$ and $\mathrm{var}(Y) = u(1 - u)$; (2) a systematic component $g(u) = g[E(Y|d)] = \beta_0 + \beta_1 d$; and (3) a link function:

$$g[E(Y)] = \mathrm{logit}[E(Y)] = \log\left[\frac{E(Y)}{1 - E(Y)}\right].$$

This model also has the advantage that the model parameters can be interpreted directly as the log odds ratio, so that the mortality estimate can be easily calculated from the parameter estimates. However, to ensure its suitability as the link function and for extrapolation to the intercept, we also modeled the data with two other link functions (McCullagh and Nelder 1989), the probit or inverse normal function,

$$g[E(Y)] = \Phi^{-1}[E(Y)],$$

and the complementary log–log function,

$$g[E(Y)] = \log\{-\log[1 - E(Y)]\}.$$

Maximized log likelihoods (Collett 1991) were used as guides to choose between the models (higher values imply better fits). The logistic model provided a slightly better fit (see Results) and was therefore retained as the model.

The generalized linear modeling framework has the advantage that it allows easy calculation of confidence intervals based on the assumption of asymptotic normality. These intervals were used for the turbine mortality estimates. Confidence limits were calculated on the logistic scale as $\beta_i \pm z_{1-\alpha/2}s$, where s is the standard error of β_i and $z_{1-\alpha/2}$ is the critical value of a standard normal distribution for a given confidence level. Estimates of β_1 were considered statistically different from zero if their confidence intervals did not include zero. Confidence limits for β_0 were transformed with the inverse of the link function (equation 2) to obtain confidence intervals for the rate of turbine mortality ($M_{d=0}$). We compared these confidence intervals to likelihood-ratio-based intervals. When not corrected for overdispersion (see below), the intervals based on the normal approximation for β_0 were slightly larger than the likelihood-ratio-based intervals.

If the survival of a fish within a net is not independent of other fish in the net, there would be greater variability in survival than predicted by binomial sampling error. We therefore also modeled the data grouped by net deployment and looked for systematic deviations in the relationship between the expectation and variance of the mortality probability that would occur if the data were under- or overdispersed. We calculated a dispersion parameter (ϕ) as the sum of the squared Pearson residuals divided by the residual degrees of freedom (McCullagh and Nelder 1989). A value of ϕ much greater than 1.0 indicates that either the data are overdispersed or that the model fit is not good for some data points. For most species, ϕ was near 1.0 (see Results). Examination of residual plots (e.g., deviance residuals versus fitted values and normal quantile plots of the Pearson residuals) suggested the fits were reasonable. We therefore rescaled the standard error of β_0 by ϕ, and used the rescaled values for confidence interval calculations. This approach does not affect the turbine mortality estimate but provides more realistic variance estimates than would be obtained under the assumption that mortality was independent for all fish. Overall, of the three methods used to produce confidence intervals, this method produced the widest intervals. We report these intervals because they provide the most precautionary estimate of the precision of the turbine mortality estimates.

Results

In total, 2,784 fish belonging to 21 species were captured during our experiments (Table 1). Atlantic silverside was the most abundant species (1,160 captured). Twelve of the 21 species were caught in numbers sufficient to warrant detailed analysis.

The choice of the link function had only minor effects on the resulting mortality estimates. The estimates from the complementary log–log model were slightly higher than those of the logistic and probit models (Table 2). With the exception of At-

TABLE 1.—The total number of specimens and number of live specimens of each species captured in 447 net sets during 48 randomly-selected generation periods between 7 September and 22 October 1999 at the Annapolis Tidal Generating Station, Nova Scotia. Data for species in boldface were considered sufficient for analysis.

Species	Number of fish captured: Total	Alive
Sea lamprey *Petromyzon marinus*	20	20
American eel *Anguilla rostrata*	10	9
Age-0 blueback herring *Alosa aestivalis*	206	144
Age-0 alewife *Alosa pseudoharengus*	33	20
Age-0 American shad *Alosa sapidissima*	39	22
Age-0 Atlantic herring *Clupea harengus*	840	441
Rainbow smelt *Osmerus mordax*	5	3
Hake spp. *Urophycis* spp.	88	69
Atlantic silverside *Menidia menidia*	1,160	1,054
Blackspotted stickleback *Gasterosteus wheatlandi*	68	65
Northern pipefish *Syngnathus fuscus*	202	175
Cunner *Tautogolabrus adspersus*	3	3
Butterfish *Peprilus triacanthus*	32	22
Arctic sculpin *Myoxocephalus scorpioides*	1	1
Lumpfish *Cyclopterus lumpus*	1	1
Smooth flounder *Pleuronectes putnami*	1	1
Winter flounder *Pleuronectes americanus*	31	28
Windowpane *Scophthalmus aquosus*	28	24
Mummichog *Fundulus heteroclitus*	6	6
Atlantic mackerel *Scomber scombrus*	9	4
Fourbeard rockling *Enchelyopus cimbrius*	1	1
Total	2,784	2,092

lantic herring maximized log likelihoods were similar between the three models. Overall, the fit of the logistic model was slightly better than the probit and complementary log–log models, a difference that was not statistically significant except for Atlantic herring. The mortality estimate from the complementary log–log model for Atlantic herring was 5% higher, but the fit of the log–log model was significantly worse. We concluded that the logistic model was the most suitable choice for modeling these data.

When λ is included in the model (equation 3), the turbine mortality estimate $M_{d=0}$ is sensitive to the choice of λ (Figure 2). As λ approaches zero, $M_{d=0}$ also approaches zero. As λ increases, $M_{d=0}$ approaches the proportion of dead fish within the sample. The log likelihood was maximized at λ values approximately equal to 1.0 for most species (Figure 2), except for species with small sample sizes (American shad and alewife) and species rarely captured dead (sea lamprey, blackspotted stickleback, winter flounder, and windowpane). For these exceptions, the profile log likelihood is almost flat or ramped, indicating that there is little

TABLE 2.—Comparison of mortality estimates and log likelihoods for logit, probit, and complementary log–log link functions for fish species entrained at the Annapolis Tidal Generating Station.

Species	Acute mortality estimate: Logit	Probit	Log–log	Log likelihood: Logit	Probit	Log–log
Atlantic herring	15.7	15.9	21.6	−500.6	−501.4	−505.6
Alewife	7.7	6.6	11.6	−17.2	−17.1	−17.3
Blueback herring	8.0	7.4	11.1	−108.2	−108.4	−109.2
American shad	23.4	22.8	24.1	−25.5	−25.5	−25.5
Atlantic silverside	2.2	1.7	2.4	−312.7	−312.4	−313.1
Blackspotted stickleback	<0.1	<0.1	<0.1	−8.2	−8.1	−8.3
Sea lamprey	<0.1	<0.1	<0.1	0.0	0.0	0.0
Hake spp.	8.7	8.1	9.6	−41.6	−41.6	−41.6
Butterfish	8.7	7.7	10.2	−16.1	−16.0	−16.0
Northern pipefish	2.2	1.7	2.8	−64.3	−64.3	−65.0
Winter flounder	5.8	5.8	5.8	−9.4	−9.4	−9.4
Windowpane	8.8	8.1	9.1	−11.3	−11.3	−11.3
Total				−1,115.6	−1,116.0	−1,122.8

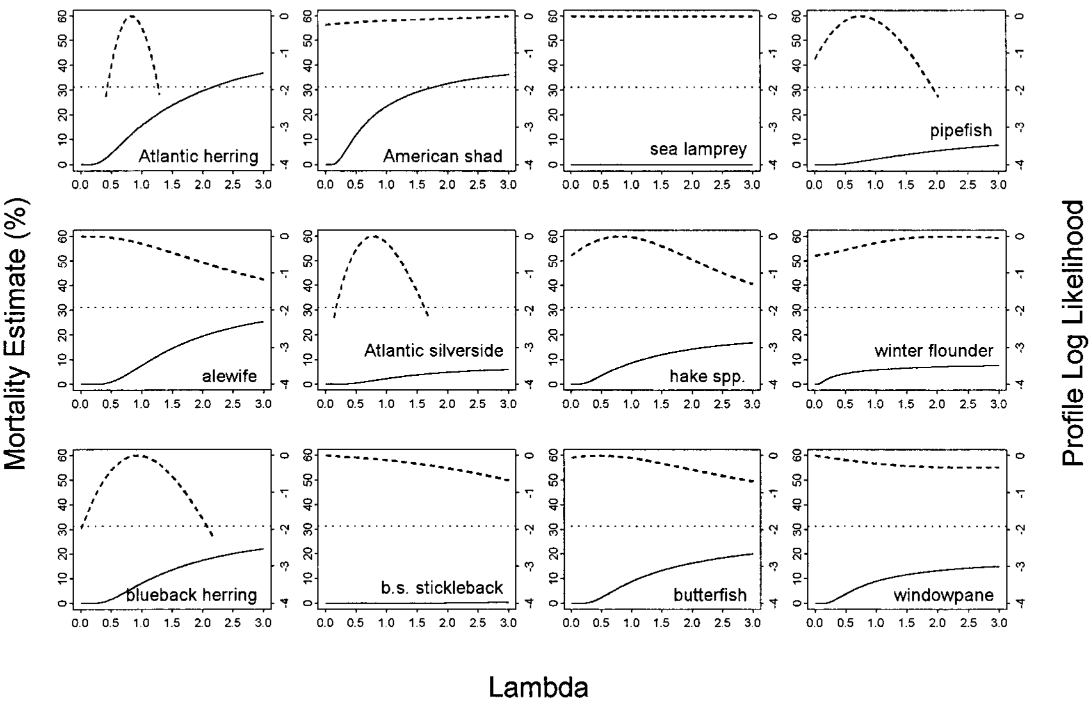

FIGURE 2.—The relationship (solid line) between the mortality estimate ($M_{d=0}$) and lambda (λ), a power parameter that determines the functional form of the relationship between mortality and net deployment duration, for 12 species of fish at the Annapolis Tidal Generation Station. The dashed line shows the profile log likelihood for λ, standardized by subtracting the maximum log likelihood from each estimate. The intersections between the dotted line and the dashed line show the 95% confidence interval for λ.

or no information about λ in these datasets. For species with enough information to estimate λ, the λ value was not statistically different from 1.0. We concluded that the specific case of λ equal to 1.0 was reasonable for most species, and we used this model for the remaining analysis.

The probability that a captured fish was dead increased rapidly with net deployment duration for clupeids, but more slowly for other species such as Atlantic silverside, northern pipefish, and black-spotted stickleback (Figure 3). The increase in probability was statistically significant at $\alpha = 0.05$ for all species except American shad, winter flounder, windowpane, and sea lamprey (Table 3). For the latter three species, very few dead fish were captured at any deployment duration (Figure 3). Figure 3 also shows that if turbine mortality is estimated from fixed-duration net deployments, the resulting estimates would be very sensitive to the length of time that the net was deployed. For example, alewife mortality was about 18% for a 1-h net deployment and about 62% for a 3-h net deployment. Regressing to a net deployment of zero duration provides an estimate of 7.7%. Estimates of the mortality rates ranged from 0.0% for sea lamprey to 23.4% for American shad (Table 3).

Discussion

In this paper, we have demonstrated that with a slight change in the methods used in many studies, estimates of turbine mortality obtained from naturally entrained fish can be improved by modeling the probability that a captured fish will be dead as a function of the duration of the net deployment. When net deployments of fixed duration are used to estimate turbine mortality, the resulting estimate is not only a function of the rate of turbine mortality, but also the duration of net deployment (Figure 3; see also Figure 2 in Stokesbury and Dadswell 1991). The duration of control experiments to correct for handling mortality cannot be chosen without knowledge of the length of time that fish are in the net, which varies both between deployments and within individual net deployments. By varying the duration of the net deployments, regression methods can be used to estimate the mortality for a net deployment of zero duration, pro-

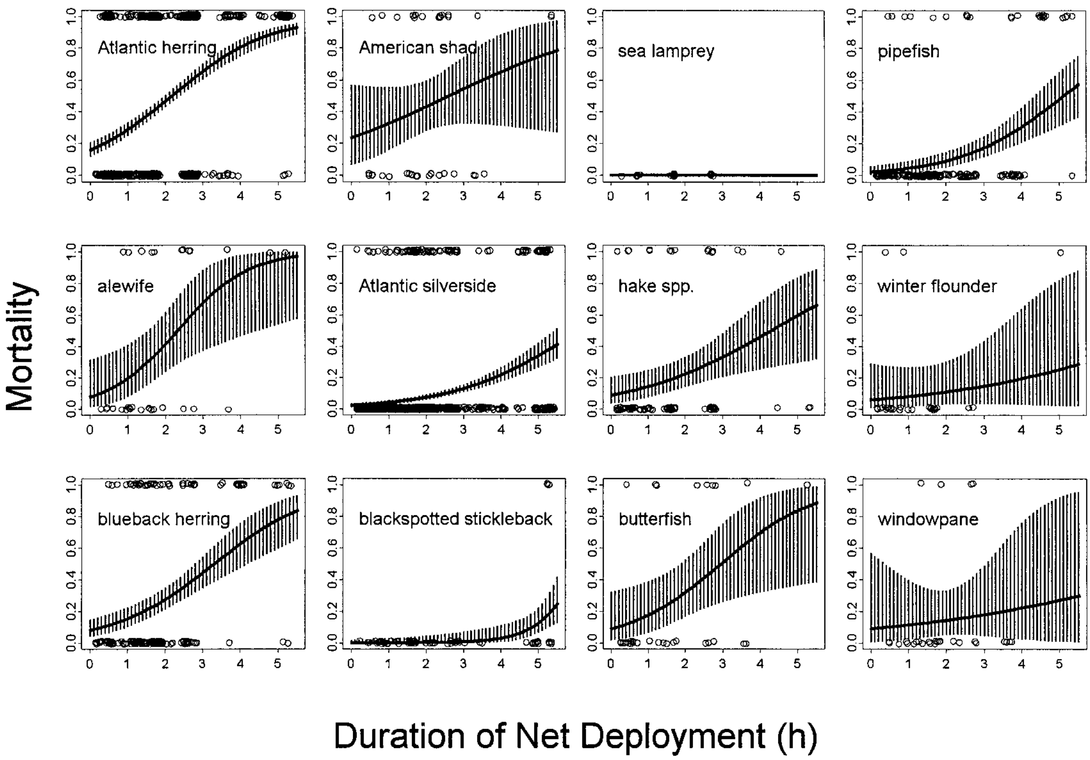

FIGURE 3.—Mortality (solid line) as a function of the duration of net deployment for 12 species of fish at the Annapolis Tidal Generating Station. Each point represents a fish that was either alive (0) or dead (1) at the time of capture. Points are slightly staggered to facilitate display. The *y*-intercept is an estimate of acute turbine mortality in the absence of any time in the net. Error bars are 95% confidence intervals.

viding a method to estimate the mortality attributable to turbine passage or passage via other structures.

A number of methods have been developed to estimate turbine mortality. These methods can be divided into two categories: those that use naturally entrained fish (e.g., Stokesbury and Dadswell 1991; Navarro et al. 1996) and those that use fish

TABLE 3.—Logistic regression coefficients and estimates of turbine mortality for 12 species of fish at the Annapolis Tidal Generating Station. Numbers in parentheses are standard errors. An asterisk indicates estimates for which the increase in mortality with increasing net deployment duration (β_1) is significantly different from zero at a 95% confidence level. Confidence intervals (C.I.) for mortality are calculated using normal approximations on the link scale and are corrected for overdispersion.

	Regression coefficients			Mortality (%)	
Species	β_0	β_1	ϕ	Mean	95% C.I.
Age-0 American shad	−1.18 (0.73)	0.45 (0.32)	1.17	23.4	6.1–58.8
Age-0 blueblack herring	−2.43 (0.35)	0.74 (0.14)*	1.59	8.1	3.5–17.2
Age-0 alewife	−2.48 (0.88)	1.07 (0.42)*	0.96	7.7	1.5–31.4
Age-0 Atlantic herring	−1.67 (0.16)	0.77 (0.07)*	1.80	15.7	10.8–22.1
Sea lamprey				0.0	
Blackspotted stickleback	−9.99 (5.73)	1.61 (1.11)*	0.40	<0.1	<0.1–5.6
Atlantic silverside	−3.78 (0.21)	0.62 (0.07)*	2.23	2.2	1.1–4.1
Northern pipefish	−3.77 (0.50)	0.73 (0.14)*	1.23	2.2	0.7–6.4
Butterfish	−2.34 (0.81)	0.79 (0.34)*	1.15	8.7	1.7–34.5
Winter flounder	−2.78 (0.91)	0.34 (0.36)	1.24	5.8	0.8–31.2
Windowpane	−2.32 (1.29)	0.26 (0.56)	1.14	8.8	<0.1–59.4
Hake spp.	−2.34 (0.49)	0.55 (0.19)*	1.10	8.7	0.3–20.9

released into the turbine intake (e.g., Hogans 1987; Mathur et al. 1994). In the first case, fish are captured with nets in the turbine tailrace, and turbine mortality is estimated either by live or dead criteria (e.g., Navarro et al. 1996) or by conducting autopsies on dead fish (e.g., Stokesbury and Dadswell 1991; Sorenson et al. 1998). In the second case, fish released into the turbine tube may be recaptured in nets in the turbine tailrace (e.g., Ruggles et al. 1990; Dubois and Gloss 1993), followed via radio tags (e.g., Hogans 1987), detected downstream via passive integrated transponder tags (e.g., Skalski et al. 1998) or recaptured by some other method (e.g., Heisey et al. 1992). Estimates are reported either corrected for handling and capture mortality (e.g., Stokesbury and Dadswell 1991; Mathur et al. 1994) or without such correction (Navarro et al. 1996). The resulting mortality estimates from both methods are often reported as acute (immediate or short term: the proportion dead at or soon after capture) or delayed (long term: the proportion dead after holding for extended periods, typically 12, 24, or 48 h).

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 with nets in the tailrace is a relatively easy and cost-effective method of obtaining data. Handling of fish is reduced, and the method can be applied in situations where test fish are not readily available (e.g., flatfish in this study). Estimates can be obtained for several species simultaneously without an increase in the required effort (e.g., Navarro et al. 1996; this study). The method can be easily integrated into sampling for other purposes (e.g., Stokesbury and Dadswell 1991; Navarro et al. 1996; this study). However, if controls are used to correct for capture and handling mortality, determination of an appropriate duration for the control experiments is difficult because the length of time the entrained fish are in the net is unknown (Gibson 1996). The regression methods presented herein provide a possible solution to this problem by estimating turbine mortality for a net deployment of zero duration.

Releasing fish into the turbine tube provides better experimental control than the use of naturally entrained fish but increases handling stress. Where the effects of stress are cumulative and culminate in increased mortality, the individual effects of handling, turbine passage, and recapture can be difficult to distinguish (Ruggles et al. 1990). Improved handling and recapture methods, such as the HI-Z Turb’N tag–recapture technique (Heisey et al. 1992), have substantially reduced handling mortality, leading to improved mortality estimates over studies where handling mortality was high (Mathur et al. 1994). The experimental and statistical methods described herein can be used to detect and reduce biases in turbine mortality estimates, although as discussed below, they do not completely alleviate problems when handling mortality is high.

Two aspects of turbine mortality not addressed by our study are delayed mortality and mortality resulting from capture and handling other than time in the net. If delayed mortality is high, acute mortality may not be a sufficient measure of the overall effect of a turbine on a fish stock (Kostecki et al. 1987; Dubois and Gloss 1993). The importance of delayed mortality has varied among studies and locations. Dubois and Gloss (1993) reported differences of up to 45% between acute and 24-h mortality for striped bass *Morone saxatilis* passed through Ossberger crossflow turbines, although high control mortality indicated that delayed mortality may have been overestimated. Mathur et al. (1996) reported differences in acute and 48-h turbine survival estimates of less than 1% for chinook salmon *Oncorhynchus tshawytscha* smolts at a dam on the Columbia River. Our mortality modeling approach can be used to estimate delayed mortality if fish are held in pens for a period of time prior to evaluating their live or dead status. The regression methods can be applied directly to the resulting dataset. However, holding facilities and handling protocols must not be a significant source of mortality if delayed mortality is to be reliably estimated (Ruggles et al. 1990; Dubois and Gloss 1993; Mathur et al. 1994).

Mortality resulting from capture and handling other than time in the net can lead to an overestimate of turbine mortality. Mortality resulting from the capture process can be separated into three components: the proportion of fish that die while entering the net (e.g., fish that are impinged or abraded against the net), the proportion that die as a result of time in the net (e.g., due to crowding or suffocation), and the proportion that die while being removed from the net (e.g., as a result of handling). Our regression model (equation 1) estimates $M_{d=0}$, where the mortality that is a result of time in the net is zero. If $M_{d=0}$ is interpreted as turbine mortality without estimation of the other components, the resulting turbine mortality estimate will be biased high. In instances where the resulting estimate and confidence interval of $M_{d=0}$

are small, the other components must also be small ($<M_{d=0}$) and hence the effect of their bias on the estimate must be small. If $M_{d=0}$ is low enough that neither mitigation nor remediation are required, mortality can reasonably be attributed to the turbine without quantification of the bias because correction for this bias would not alter management decisions. When the resulting mortality estimate is high relative to the compensatory capacity of the stock, the estimate should not be interpreted as turbine mortality without quantification of mortality from other components of the capture process.

Our approach can be extended to incorporate mortality from the other components of the capture process. Controls for handling mortality typically involve placing marked fish that have not passed through the turbine into the net for some period of time (e.g., Stokesbury and Dadswell 1991). To incorporate the handling control into the regression model, the duration of control experiments should also be varied to allow extrapolation to zero time spent in the net. The results of the control experiments can then be modeled simultaneously with those from the naturally entrained fish. A logistic model incorporating the results of the control experiments is

$$E(Y) = \frac{\exp(\beta_0 + \beta_1 d)}{1 + \exp(\beta_0 + \beta_1 d)}t + \frac{\exp(\beta_0' + \beta_1' d')}{1 + \exp(\beta_0' + \beta_1' d')}t', \qquad (4)$$

where β_0, β_1, and d are the terms for the naturally entrained fish, β_0', β_1', and d' are the terms for the control fish, and t and t' are factors with two levels (0 or 1.0) that state whether or not a fish passed through the turbine. Turbine mortality can be calculated from β_0 and β_0' by combining equation (2) and the model of Burnham et al. (1987) based on competing risk theory:

$$M_{d=0} = 1 - \left[\frac{\exp(\beta_0)\exp(1 - \beta_0')}{\exp(\beta_0')\exp(1 - \beta_0)}\right]. \qquad (5)$$

The standard error of $M_{d=0}$ can be calculated from the standard errors of β_0 and β_0' based on the rules for calculating the standard errors of functions of random variables (Kendall et al. 1983). This model structure has the advantage that it can be fitted as a generalized linear model and thus can be easily modified to test assumptions about the functional form of the model by changing the link function. However, the calculation of confidence intervals for $M_{d=0}$ requires the assumption of normality for $M_{d=0}$ and, as such, can produce intervals that include values less than 0.0% or greater than 100%. An alternative approach, outside the generalized linear models framework, is to estimate $M_{d=0}$ directly within the model based on maximum likelihood. (The functional form of the model is obtained by solving equation (5) for β_0 and substituting the result into equation 4). In this method, confidence intervals that do not require the normality assumption can be derived from the profile likelihood.

The regression methods described here require extrapolation beyond the range of the data. The success of extrapolation depends heavily on the correctness of the assumed model (McCullagh and Nelder 1989). With our data, we found that the logistic model provided a slightly better fit than the probit or complementary log–log models and that our estimates were not very sensitive to the choice of link function. The estimates were sensitive to the degree of nonlinearity (λ) in the model, although the linear, logistic form was most appropriate. The best model could potentially vary between locations, and we suggest that researchers employing these methods perform similar diagnostics to ensure that the selected model is appropriate.

Our estimates of the dispersion parameter ϕ ranged between 0.40 for blackspotted sticklebacks (most were captured alive) and 2.23 for Atlantic silversides and were greater than 1.0 for all but two species. These results suggest that survival of individual fish within a net deployment was not independent of other fish captured at the same time. This situation could arise, for example, if mortality rates varied with operating efficiency (which varies with the stage of the tide) or if crowding within the net affected survival. The dispersion parameter is used to rescale the standard errors of parameter estimates (on the logistic scale) and can therefore significantly change estimates of statistical significance and confidence intervals. Where possible, the binomial assumption should be tested to ensure that the precision of the resulting estimates is not overstated.

As discussed, the use of naturally entrained fish to estimate turbine mortality entails a loss of experimental control relative to the use of experimentally released fish. When regression methods are used, designing an experiment to achieve a specified confidence interval width requires consideration of a number of factors, including prior knowledge (or guesses) of capture rates, regression slope and intercept, and the ϕ value. Small perturbations in the parameter estimates can markedly affect experimental design efficiency (Chaloner and Larntz 1989). Confidence intervals for turbine mortality should be smallest when the duration of sampling effort is very short or very long (Sebastiani and Settimi 1997), although this design does not allow for testing of the functional form of the model (i.e., determine whether the linear logistic model is appropriate). Selection of an appropriate design requires balancing of the relative costs and benefits of these requirements. In our study, the relationship between net deployment duration and mortality varied substantially among species. If a pilot study showed that mortality does not increase with net deployment duration, these results could be used to select appropriate deployment durations for fixed-duration experiments, substantially simplifying the experimental design.

Turbine mortality of clupeids has been previously studied at the Annapolis Tidal Generating Station. Stokesbury and Dadswell (1991) estimated that 46.3% of age-0 clupeids do not survive turbine passage. Our estimates of acute turbine mortality for American shad (23.4%), Atlantic herring (15.7%), alewife (7.7%), and blueback herring (8.1%) are all lower than 46.3%, differences that are statistically significant at a 95% confidence level, with the exception of American shad. Two possible explanations exist for discrepancies between the two studies. The duration of the control experiments used by Stokesbury and Dadswell (1991) was short relative to the duration of the net deployments and the expected length of time that fish would have been in the nets (Gibson 1996). Turbine mortality might therefore have been overestimated in their study. Alternatively, the majority of turbine-induced injuries reported by Stokesbury and Dadswell (1991) were pressure-related injuries, such as eye hemorrhage or gas bladder damage (Dadswell and Rulifson 1994). If mortality resulting from pressure-induced injuries is not acute, it would not have been detected in our study. The results of the two studies are therefore not directly comparable without further information about the timing of pressure-induced mortality. Estimates of acute turbine mortality for other species in our study ranged from 0.0% for sea lamprey to 8.8% for windowpane, the first turbine mortality estimates for these species.

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, the survival probability affects the life expectancy and lifetime fecundity of a fish. Removal of fish from a population may elicit a compensatory response such as increased growth and survival of other members of the population. The effect at the population level encompasses the combined effects from all fish and will vary depending on factors such as the proportion of fish passing through the turbine, the timing of turbine passage relative to reproduction or to compensatory processes, and the life history characteristics of the species. A species with a short life span and high fecundity and that reproduces prior to passing through the turbine, could withstand a comparatively high turbine mortality rate with little impact at the population level. Conversely, for a species with a long life span and low fecundity that 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 exposed to turbines. A framework should be developed that allows not only estimation of turbine mortality for a given species based on turbine design, but also prediction of the population response based on the general biology of the fish. Turbine mortality studies have traditionally focused on commercially important groups such as salmonids and clupeids. Development of such a framework requires estimates for a wide range of species and turbine designs.

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