

# Extrapolating Impingement and Entrainment Losses to Equivalent Adults and Production Foregone



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*Technical Report*

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# **Extrapolating Impingement and Entrainment Losses to Equivalent Adults and Production Foregone**

**1008471**

Final Report, July 2004

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# REPORT SUMMARY

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This report provides guidance on the use of two types of fish loss extrapolation models: equivalent adult (EA) and production foregone (PF) models. The report is a companion to EPRI report 1007821, which summarizes impingement survival information and EPRI report 1000757, which summarizes entrainment survival information. It complements EPRI reports TR-112013 and 1005176, which review fish population assessment methods in general.

## **Background**

The U. S. Environmental Protection Agency (USEPA) has recently issued new Clean Water Act §316(b) regulations that effect existing power producing facilities. Facilities affected are subject to performance requirements based on reducing fish and shellfish impingement mortality and, in some cases, entrainment at a Cooling Water Intake Structure (CWIS). The USEPA regulations emphasize minimization of these losses, subject to an exception that the costs of reducing those losses must not be significantly greater than the benefits of the reductions. One of the most common approaches for evaluating the benefits of reducing these losses of early life stages is to extrapolate the losses to equivalent reductions in numbers of adult fish or of biomass production available to predators. These equivalent adult or equivalent biomass estimates can be more easily valued than the raw loss estimates themselves.

## **Objectives**

To provide guidance on the use of two key types of fish loss extrapolation models: equivalent adult (EA) and production foregone (PF) models, including proper model selection and model parameterization and an explanation of the uncertainties in the modeling results.

## **Approach**

EPRI's previous reports such as TR-112013 and 1005176 provide theoretical descriptions of EA and PF models and citations to examples but do not provide implementation guidance. To provide this guidance, the project team identified and assembled relevant detailed information from the peer-reviewed technical literature, CWIS permit applications, and corporate project reports. Specifically, the team reviewed historical uses of EA and PF models, identified and documented relevant model equations, identified sources of information for developing input parameters, and applied the models to six representative marine and freshwater species. They also documented and discussed methods for conducting uncertainty analyses.

## Results

This report provides explicit guidance for the use of EA and PF models in the context of the new Clean Water Act §316(b) regulations. It includes:

- Review of historical uses of EA and PF estimates
- Explanation of equations, input parameters, and input data requirements
- Guidance concerning parameter estimation for species- and life-stage-specific parameters used by the models
- Identification of sources of information for developing input parameters
- Example calculations for marine and freshwater fish species, including sensitivity analyses and discussion of uncertainties and potential biases
- Recommended procedures for uncertainty analyses and reality checks.

EA and (less frequently) PF estimates have sometimes been used as endpoints in §316(b) adverse impact determinations. However, because of the role of EA and PF estimates in assessments of economic benefits of alternative intake fish protection technologies, the relationships of EA and PF to quantities used in benefits analyses are emphasized throughout this document. EPA's *Guidelines for Preparing Economic Analyses* (USEPA 2000) provide an exhaustive list of the types of ecological values that are amenable to quantitative benefits analyses, grouped into four categories: market values, non-market values, indirect ecosystem values, and non-use values. The individual types of ecological values most likely to be relevant to Section 316(b) alternative intake technology assessments are (1) commercial fishing (a market value); (2) recreational fishing (a non-market value); and (3) forage fish production (an indirect ecosystem value). EA models can be used to quantify reductions in commercial and recreational values; PF models can be used to quantify reductions in forage fish production values.

## EPRI Perspective

This report provides power plant operators and associated environmental services personnel and others involved in CWIS permitting with implementation guidance for quantifying fish losses associated with operation of a CWIS. Estimating these fish losses is a critical component of the cost-benefit analysis test in the new EPA Clean Water Act §316(b) regulations. The information contained in this report will cost-effectively guide users in developing the requisite information that, in most cases, they must submit as part of §316(b) permit applications.

## Keywords

Clean Water Act Section 316(b)  
Cooling Water Intake Structure (CWIS)  
Fish Population Modeling  
Impingement and Entrainment



## **EXECUTIVE SUMMARY**

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The U.S. Environmental Protection Agency (EPA)'s regulations for implementing Section 316(b) of the Clean Water Act (Federal Register Vol. 69, No. 131; July 9, 2004) emphasize minimization of entrainment and impingement losses of fish and shellfish, subject to an exception that the costs of reducing those losses must not be significantly greater than the benefits of the reductions. One of the most common approaches for evaluating the benefits of reducing these losses of early life stages is to extrapolate the losses to equivalent reductions in numbers of adult fish, or of biomass production available to predators. These equivalent adult or equivalent biomass estimates can be more easily valued than can the raw loss estimates themselves.

This report provides guidance on the use of two types of extrapolation models: equivalent adult (EA) models and production foregone (PF) models. The report includes:

- A review of historical uses of EA and PF estimates in 316(b) assessments, focusing on the Brunswick, Chalk Point, Diablo Canyon, Salem, and Mercer power stations and on EPA's 316(b) case study.
- An explanation of equations, input parameters, and input data requirements for four of the models used in the historical applications,
- Guidance concerning parameter estimation for species- and life-stage-specific parameters used by the models,
- Identification of sources of information for developing input parameters,
- Example calculations for six marine and freshwater fish species, including sensitivity analyses and discussion of uncertainties and potential biases, and
- Recommended procedures for uncertainty analyses and reality checks.

EA and PF models do not require site-specific data on the distribution or abundance of vulnerable populations, and they can be applied at any power plant for which entrainment or impingement losses can be estimated. However, these models rely on conservative assumptions, are highly sensitive to uncertainties in input parameter values, and cannot be independently validated using site-specific data. Although EA and PF models can be very useful for interpreting entrainment and impingement losses in an ecological or human use context, significant care is needed both in model selection and in model parameterization to ensure that the results are credible.

This report provides general guidance on the use of EA and PF models, but does not provide recommended parameter values for specific fish species or sites. Parameter estimates for species commonly entrained or impinged in marine, estuarine, and freshwater ecosystems will be documented in a follow-on report.

## **ACKNOWLEDGMENTS**

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# 1

## INTRODUCTION

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EPA's regulations for implementing Section 316(b) of the Clean Water Act emphasize minimization of entrainment and impingement losses of fish and shellfish, subject to an exception that the costs of reducing those losses must not be significantly greater than the benefits of the reductions. The benefits of reductions in entrainment and impingement losses of early life stages can best be evaluated by placing them in an ecological or human use context, for example, by extrapolating the losses to equivalent reductions in numbers of adult fish, or of biomass production available to predators. Such extrapolations can greatly facilitate the implementation of several of the compliance approaches included in EPA's final Existing Facilities Rule (Federal Register Vol. 69, No. 131; July 9, 2004)). For example, the cost-benefit evaluation required to support a site-specific performance standard requires that monetary values be placed on the commercial, recreational and ecological benefits of complying with the applicable national performance standard. Estimation of these values requires the establishment of quantitative connection between the numbers of fish (often eggs or larvae) lost due to entrainment and impingement and subsequent reductions in harvest, recreational opportunity, or ecological function. If restoration is chosen as a compliance alternative, then an applicant must compare the expected fish and shellfish production of the restored habitat to the losses of fish and shellfish due to impingement mortality and entrainment. Such comparisons can only be made if the production and the losses are expressed in common metrics such as pounds of production or numbers of fish of a given age or size. The new regulations also allow states to develop entrainment/impingement loss trading programs. Because different facilities located on the same water body will often entrain or impinge different species, or different life stages of the same species, a common biological metric for trading must be established when such programs are implemented.

EPRI's Catalog of Assessment Methods for Evaluating the Effects of Power Plant Operations on Aquatic Communities identifies two especially useful approaches for making the types of extrapolations required to implement the new EPA regulations: Equivalent Adult (EA) models and Production Foregone (PF) models (EPRI 1999, 2003). Both of these approaches extrapolate losses of individual organisms at intake structures (e.g., numbers of eggs, larvae, and juveniles) to numbers or production of older fish. The models require estimates of mortality and growth rates for vulnerable species, but do not require site-specific data on the distribution or abundance of vulnerable populations. Given appropriate guidance on model implementation, these models could be applied at any power plant for which entrainment or impingement losses can be estimated.

More than 99.9 percent of the young spawned by a typical female fish can be expected to die prior to adulthood. Similarly, of the fish entrained or impinged at a cooling water intake structure, only a fraction would have survived to reproduce or to be harvested by fishermen. EA models express entrainment and impingement losses in terms of the number of fish that would

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have survived to some given future age. Most of natural mortality to young fish is due to predation by other organisms, including other fish. When fish die due to entrainment or impingement, they are no longer available to be consumed by other organisms. The loss of biomass available to predators consists of two components: the weight of the fish at the time of entrainment or impingement, and the future growth that would have occurred prior to predation, had the organisms not been entrained or impinged. PF models quantify the loss in potential predator consumption by integrating the future probabilities of growth and mortality of fish entrained or impinged at any given stage or age.

EPRI's catalog (EPRI 1999) provides theoretical descriptions of EA and PF models, as well as citations to examples, but does not provide implementation guidance. This report provides such guidance, including:

- Review of historical uses of EA and PF estimates in 316(b) assessments,
- Explanation of equations, input parameters, and input data requirements,
- Guidance concerning parameter estimation for species- and life-stage-specific parameters used by the models,
- Identification of sources of information for developing input parameters,
- Example calculations for marine and freshwater fish species, including sensitivity analyses and discussion of uncertainties and potential biases, and
- Recommended procedures for uncertainty analyses and reality checks.

EA and (less frequently) PF estimates have sometimes been used as endpoints in Section 316(b) adverse impact determinations. However, because of the role of EA and PF estimates in assessments of economic benefits of alternative intake fish protection technologies, the relationships of EA and PF to quantities used in benefits analyses are emphasized throughout this document. EPA's *Guidelines for Preparing Economic Analyses* (USEPA 2000) provide an exhaustive list of the types of ecological values that are amenable to quantitative benefits analyses, grouped into four categories: market values, non-market values, indirect ecosystem values, and non-use values. The individual types of ecological values most likely to be relevant to Section 316(b) alternative intake technology assessments are: (1) commercial fishing (a market value); (2) recreational fishing (a non-market value), and (3) forage fish production (an indirect ecosystem value). EA models can be used to quantify reductions in commercial and recreational values; PF models can be used to quantify reductions in forage fish production values.

This report provides general guidance on the use of EA and PF models, but does not provide recommended parameter values for specific fish species or sites. Parameter estimates for species commonly entrained or impinged in marine, estuarine, and freshwater ecosystems will be documented in a follow-on report.

# 2

## CONCEPTS AND DEFINITIONS

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Application of EA and PF models requires understanding of some specific concepts and definitions.

*Entrainment* refers to the drawing of small organisms such as fish eggs and larvae that are too small to be strained out by trash screens through a cooling water system. In a once-through (open-cycle) cooling system, the water and organisms are returned to the source water body. Mortality to entrained organisms can occur due to a combination of mechanical and thermal stresses. Mortality studies have shown that there is substantial variation in sensitivity among species, with some species suffering mortality of <50% due to entrainment and others suffering 100% mortality of all life stages (EPRI 2000). In a closed-cycle cooling system, the water withdrawn from the source water body is repeatedly recycled and subject to chemical treatment so that all entrained organisms are killed. The size range of organisms susceptible to entrainment depends on the mesh size of the traveling screens. Under EPA's 316(b) rules, organisms that pass through 3/8 inch screens are considered entrained. Typically eggs, larvae, and early juvenile fish (~20 to 30 mm in length, depending on body shape) are susceptible.

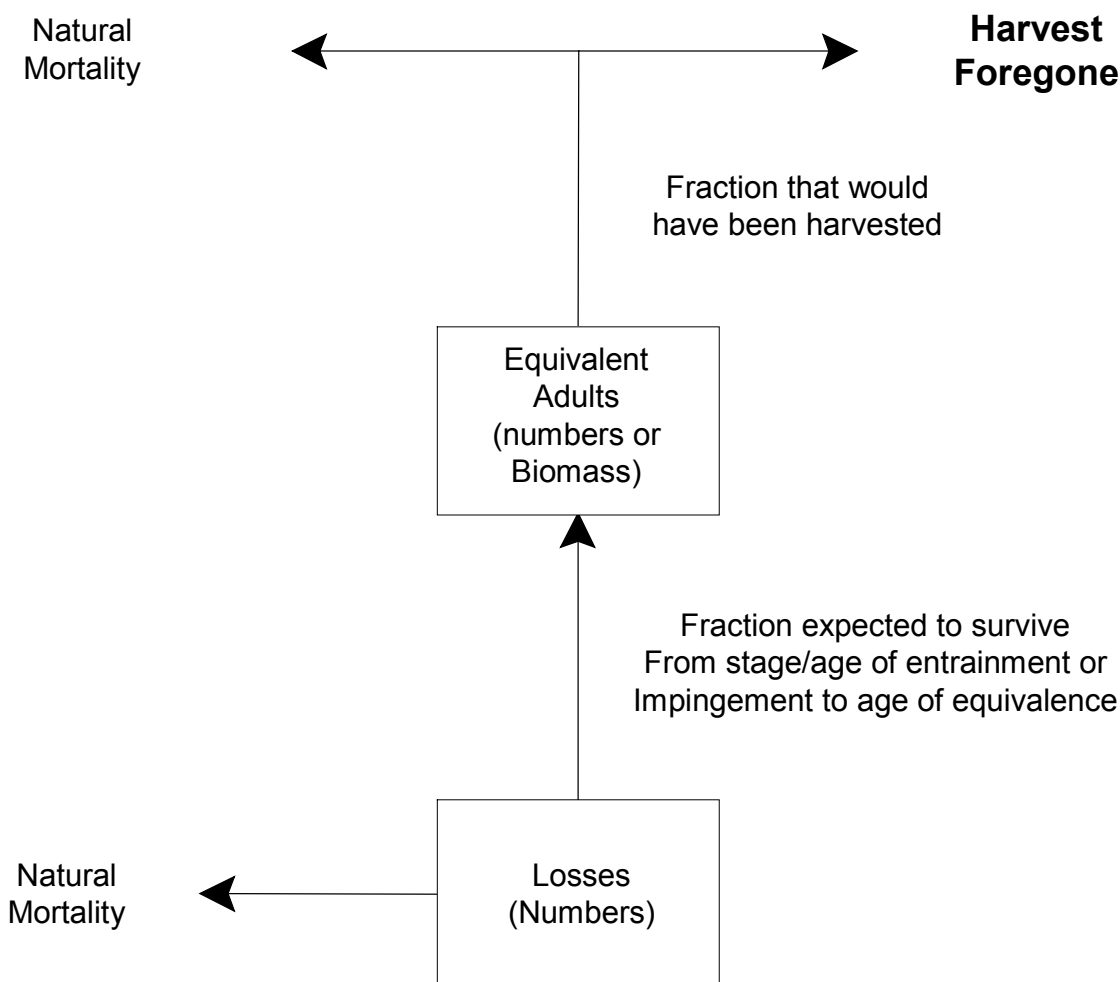
*Impingement* refers to the trapping of fish on trash screens at the intakes. Under EPA's 316(b) rules, organisms trapped on 3/8 inch mesh screens are considered impinged. Although fish of virtually any size may occasionally be impinged, the great majority of impinged fish are typically young-of-the-year fish (~20 to 100 mm in length). Mortality to impinged fish can occur because of physical damage (e.g., scale loss) or because of stress exhaustion (EPRI 2003). Sensitivity to impingement-related stresses varies greatly among species, and mortality rates are also strongly influenced by intake configurations and operating modes. Survival rates of 80% or more have been observed for some species at plants where fish return systems have been installed and rotating trash screens are operated in continuous mode (EPRI 2003).

*Losses* refer to the total numbers of organisms killed due to entrainment or impingement during some specific time interval, e.g., monthly, annually, or over the expected lifetime of a facility. Losses are typically tabulated by species, life stage, and (in some cases) length.

*Equivalent adult losses* are estimates of the number of entrained or impinged fish removed from a population that otherwise would have survived to some future age (termed the *age of equivalence*). The Equivalent Adult (EA) model was first described by Horst (1975) and Goodyear (1978). As depicted in Figure 2-1, the EA model adjusts the losses to account for natural mortality that would have occurred between the age at entrainment or impingement and the age of equivalence. Depending on the purpose for which the model is being used, the

*Concepts and Definitions*

equivalent adult estimates can be extrapolated further to estimates of reductions in commercial or recreational harvests.



**Figure 2-1**

**Relationship between entrainment/impingement losses and foregone fishery harvest, as quantified using equivalent adult (EA) models** Estimates of age- or stage-specific survival rates are used to estimate number of these fish that would have survived to the age of equivalence (assuming no biological compensation); age-specific weights are used to convert numbers to biomass (if needed). Yield-per-recruit models or assumed fishery exploitation rates are used to calculate the resulting reduction in harvest by fishermen.

*Age of equivalence* refers to the age to which losses are extrapolated using an EA model. The age of equivalence is specified by the assessor and can differ depending on the purpose of the assessment. Typical ages of equivalence include age 1, age at sexual maturity, and age at entry to a fishery.

*Equivalent yield reduction* refers to the reduction in harvest of economically valuable species due to entrainment and impingement losses. Yield reductions can be either direct, resulting from entrainment and impingement of the harvested species itself, or indirect, resulting from



entrainment and impingement of forage species. Calculation of equivalent yield reduction involves estimation of the fraction of fish of a harvestable size or age that are actually harvested. Historically, in many assessments it has been conservatively assumed that all equivalent adults would have been harvested. However, where available (e.g., for many commercially exploited marine species), yield-per-recruit models developed by fisheries management agencies can provide more realistic estimates.

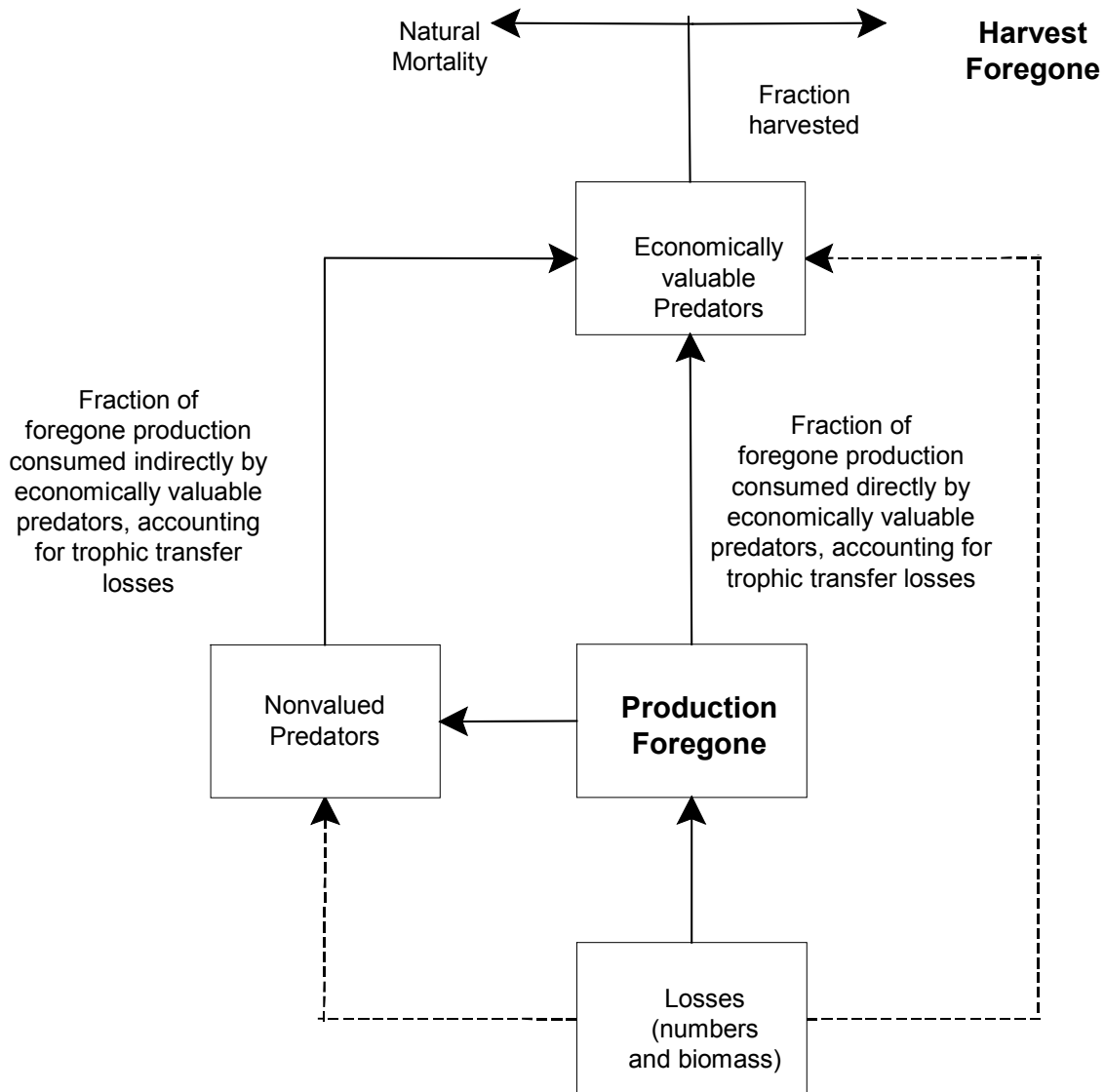
*Production Foregone* refers to the reduction in prey biomass available to predators because of the entrainment and impingement losses of prey, including the expected future growth of these prey prior to consumption by predators. As depicted in Figure 2-2, part of the foregone production would have been consumed by harvestable predator species, resulting in an indirect reduction in predator harvest. Historically, in many assessments it has been conservatively assumed that 100% of the foregone production would have been consumed by harvestable predators. ESSA (2000) argued that estimates of reductions in available prey biomass should include, in addition to foregone future production, the total biomass of entrained and impinged organisms. Although it is not clear that this source of biomass could significantly contribute to the total reduction in available prey or even that it represents an actual reduction (given that all entrained fish and often the impinged fish as well are returned to the source waterbody rather than being removed), this source is included in Figure 2-2 for the sake of completeness.

*Trophic transfer efficiency* refers to the percentage of prey biomass consumed that is transformed into predator biomass.

Relationships between entrainment/impingement losses, EA and PF estimates, and quantifiable benefits reductions are depicted in Figure 2-3. Relatively few studies to date have included all of the listed components; however, it is clear that a comprehensive approach to benefits analysis requires, in addition to input parameter values for the EA and PF models, a variety of assumptions (or data) concerning trophic transfer efficiencies and biomass flow pathways within source waterbodies.

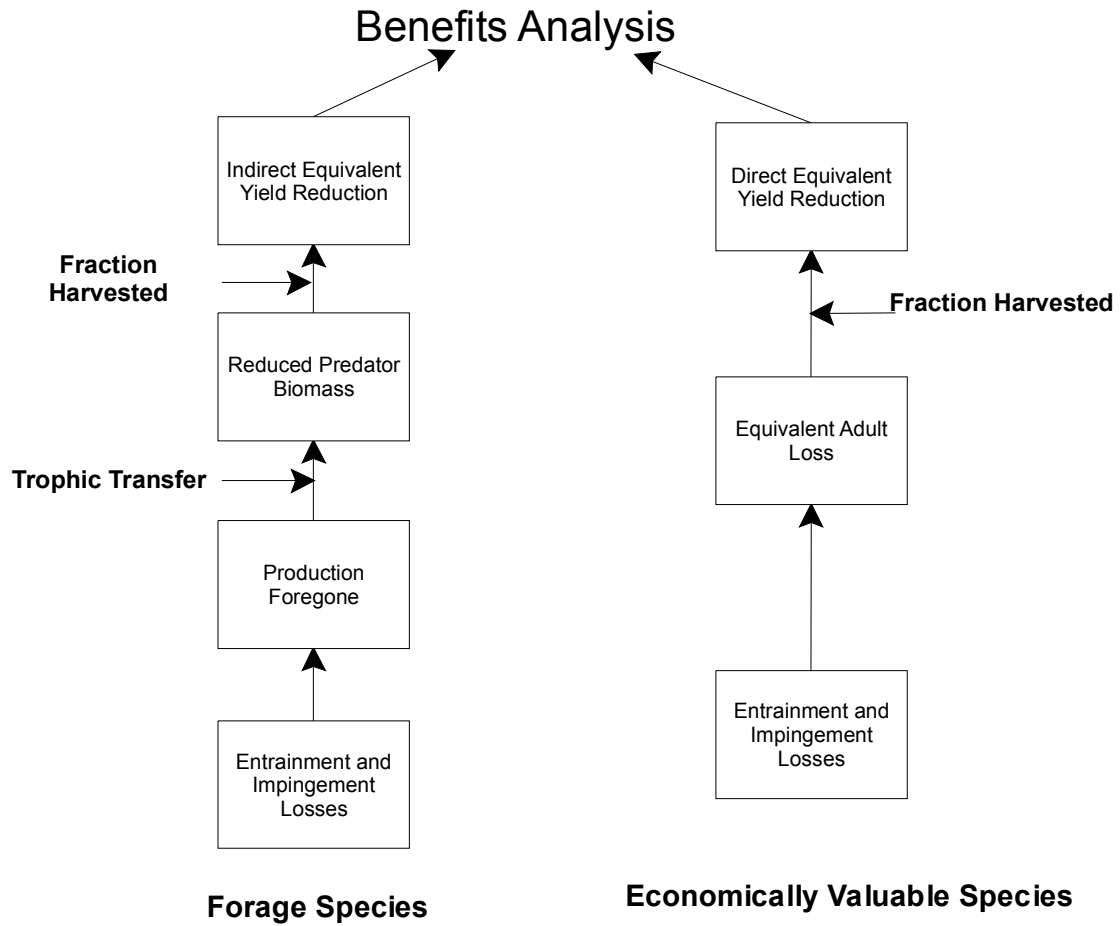
None of the EA or PF models that are currently used incorporate density-dependence. The models implicitly assume that the fish that *are not* entrained or impinged do not grow faster or survive at a higher rate because of reduced competition, and that the entrained and impinged fish are removed permanently from the source waterbody rather than (as is nearly always the case, at least for entrainment) being returned to the source water body where they can serve as prey. For this reason, EA and PF models are inherently biased toward overestimation of the actual reductions in abundance or yield caused by entrainment and impingement. Benefits analyses performed using these models similarly contain a conservative bias toward overestimation of the actual economic benefits of alternative intake technologies.

Concepts and Definitions



**Figure 2-2**

**Relationship between entrainment/impingement losses and foregone fishery harvest, as quantified using Production Foregone (PF) models. Estimates of age- or stage-specific growth and survival rates are used to estimate the biomass production of the losses over the lifetime of the fish (assuming no biological compensation). Assumptions are made concerning the fraction of the foregone production that would have been converted into biomass of economically valued species, by both direct and indirect consumption pathways. Yield-per-recruit models or assumed fishery exploitation rates are used to calculate the resulting reduction in harvest by fishermen. ESSA (2000) argued that the total biomass of entrained and impinged organisms should be counted as a production loss, although this biomass is usually returned to the source waterbody.**



**Figure 2-3**  
Conceptual relationship of Equivalent Adult estimates and Production Foregone estimates to economic benefits analysis.



# 3

## HISTORICAL APPLICATIONS OF EA AND PF MODELS

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EA models were originally proposed as a relatively simple tool for using readily available data to express numbers of entrained and impinged organisms in a form useful for decision-making. In early applications, the models were used as comparative indicators of adverse environmental impacts. For example, an EA model can be used to express entrainment and impingement losses in terms of numbers or pounds of harvestable-sized fish, which can then be compared to known commercial harvest rates. If the losses, expressed in terms of harvestable fish, are very much smaller than the harvests, it can be argued that the losses were insignificant. More recently, EA models have been used in formal economic benefits analyses performed as part of alternative fish protection technology assessments. In such applications, the EA models have sometimes been coupled to yield-per-recruit models, thereby expressing the losses as equivalent reductions in harvest by commercial and recreational fishermen.

Like EA estimates, production foregone estimates were originally viewed as comparative indicators of potential adverse impacts on aquatic ecosystems (Rago 1984). These estimates have since been used more often as inputs to benefits analyses, in which estimates of production foregone are extrapolated to estimates of harvestable predator biomass foregone.

The following summaries of historical applications of these models are intended to illustrate the variety of approaches that have been used to calculate equivalent adult losses and production foregone, and the range of species and water body types to which EA and PF models have been applied. Recent applications are emphasized because they are probably more relevant to future applications, which appear likely to emphasize economic benefits analysis rather than adverse impact determinations. The water bodies, numbers of species addressed, and models used in these assessments are listed in Table 3-1.

### 3.1 Brunswick

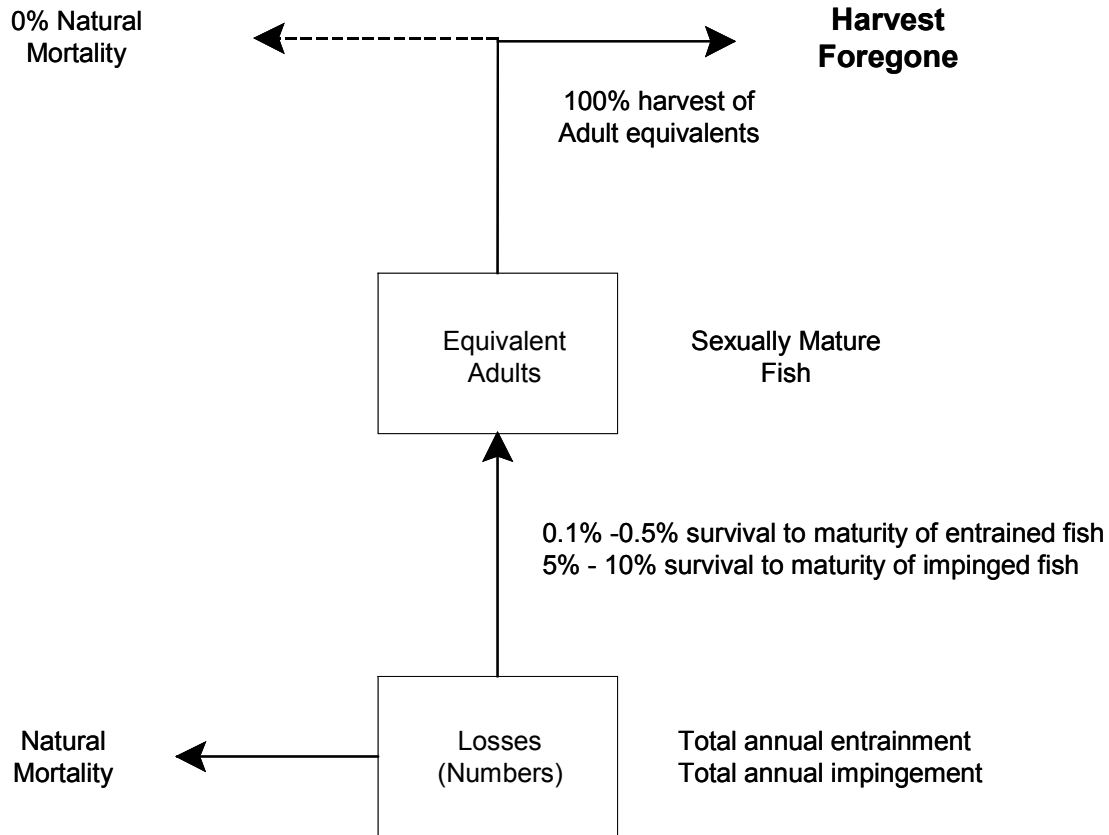
Lawler et al. (1981) documented application of an EA model to fish and shellfish species entrained and impinged at the Brunswick Steam Electric Plant located on the Cape Fear estuary, North Carolina. Although the species evaluated included spot, Atlantic croaker, flounder, mullet, seatrout, Atlantic menhaden and bay anchovy, detailed descriptions of methods and results were provided only for spot and croaker. The approach used by Lawler et al. (1981) is summarized in Figure 3-1. Only total annual entrainment and impingement losses were available; there was no breakdown of the losses by length or life stage. The age of equivalence was defined to be the age at sexual maturity, about 1-2 years for these two species. Rates of survival to the age of equivalence were estimated, based on a combination of site-specific

*Historical Applications of EA and PF Models*

modeling and expert judgment, to be 0.1% to 0.5% for entrainable life stages and 5%-10% for impingeable juveniles.

**Table 3-1**  
**Summary of Historical Studies**

Study	Water body	Number of Species addressed	Model	Model Use
Brunswick (Lawler et al. 1981)	Cape Fear Estuary	7	EA (sexual maturity)	Comparison to commercial catch (pounds and dollars) and bycatch (pounds)
Chalk Point (Otto 1989)	Patuxent River	1	EA (30 mm juvenile)	Comparison of resultant loss in predator biomass to commercial catch (expressed as dollars)
Diablo Canyon (Tenera 2000)	Pacific Ocean	8	EA (sexual maturity) FH (sexual maturity)	Comparison to commercial and recreational catch (expressed as pounds and as dollars)
Salem/Mercer (PSEG 1999, 2001)	Delaware Estuary	12	EA (age-1 equivalents) Production Foregone Yield foregone (direct and indirect)	Input to formal economic benefits analysis
EPA 316(b) case study (USEPA 2002)	Delaware Estuary Narragansett Bay Atlantic Ocean Tampa Bay Pacific Ocean Ohio River Lake Erie	50	EA (age-1 equivalents) Production foregone Yield foregone (direct and indirect)	Input to formal economic benefits analysis



**Figure 3-1**  
**Overview of EA approach used at Brunswick (Lawler et al. 1981).**

The significance of the EA estimates was evaluated by (1) comparison to commercial catch statistics for North Carolina, and (2) comparison to reported bycatch in the North Carolina shrimp fishery. For these comparisons, the EA losses were converted into pounds per year using the average weight per fish in the North Carolina commercial fishery. To provide an additional perspective, economic values were calculated by multiplying the total weight of equivalent adult fish by the average landed price per pound in the North Carolina fishery. The calculations assumed 100% harvest of adult equivalents.

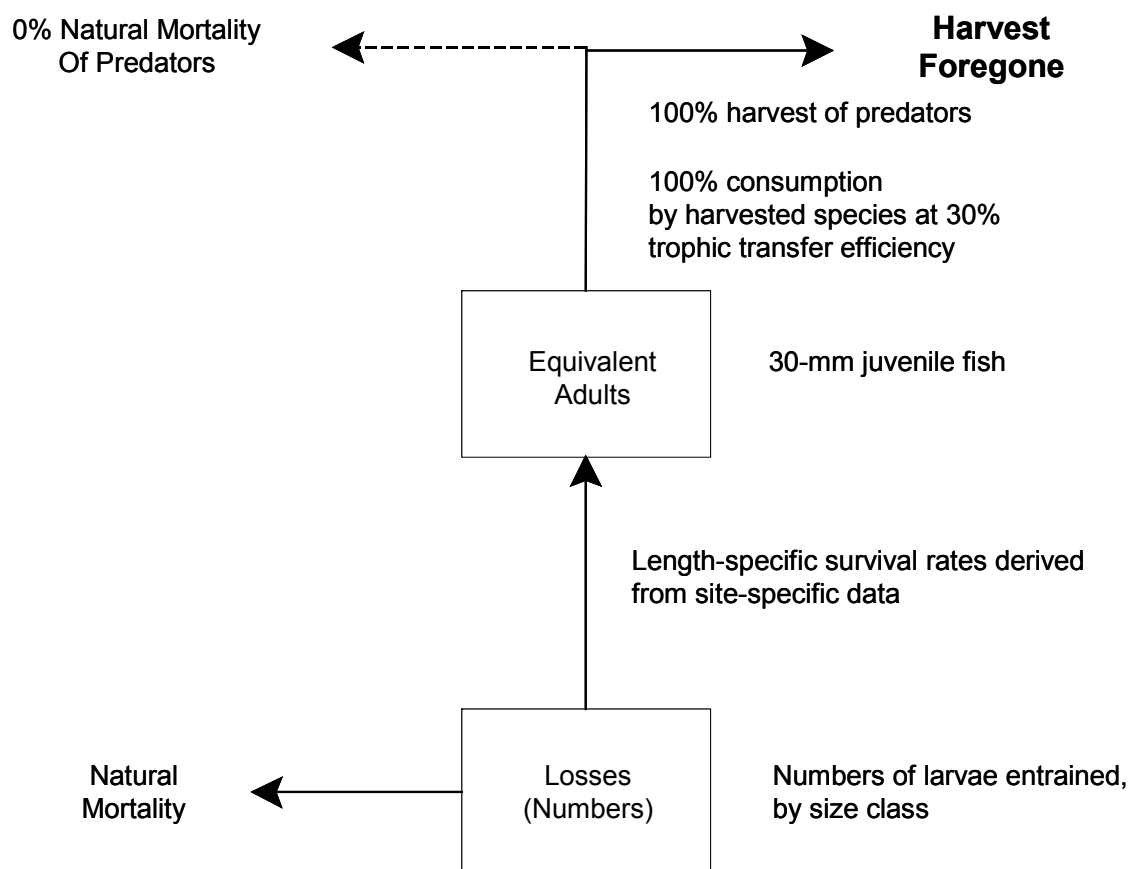
### 3.2 Chalk Point

Otto (1989) assessed impacts of entrainment of bay anchovy at the Chalk Point Station on the Patuxent River. The approach involved elements of both EA and PF models. As depicted in Figure 3-2, numbers of bay anchovy larvae, by size class, were extrapolated to estimates of equivalent early juveniles, expressed both as numbers of fish lost and as total dry weight biomass. Survival probabilities were calculated for each length class of entrained larvae, using a survivorship curve developed from site-specific data. The equivalent juvenile biomass was summed over all entrainment length classes and then converted into an estimate of lost predator biomass, assuming a biomass conversion efficiency of 30% and 100% harvest of all lost predator

*Historical Applications of EA and PF Models*

production. The estimates of lost predator production were then converted to dollars using an economic valuation model that considered both commercial and recreational fishing.

This approach, which does not consider foregone production at earlier or later stages, was justified on the grounds that bay anchovy eggs and larvae do not contribute significantly to potential predator production, and that 30-mm bay anchovy are the predominant size class preyed on by juvenile striped bass, weakfish, and other similar species.



**Figure 3-2**  
**Overview of EA approach used at Chalk Point (Otto 1989).**

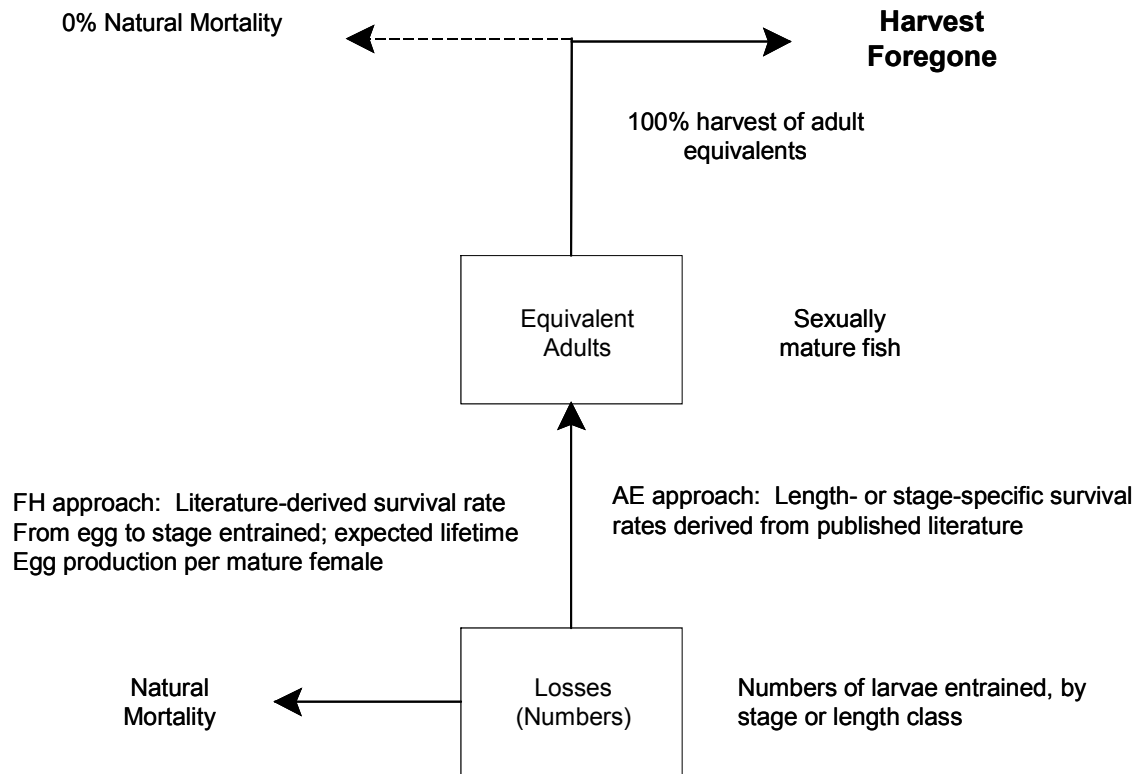
### 3.3 Diablo Canyon

Tenera (2000) assessed impacts of entrainment of various fish species at the Diablo Canyon Power Plant, located on the Pacific coast between Morro Bay and Avila Beach, California. As depicted in Figure 3-3, Tenera employed two different types of EA models. In the first model, numbers of entrained larvae, by stage, were extrapolated to numbers of reproductive adults using estimates of expected survival from the larval stage to reproductive maturity. This model is similar to the EA models used in the other cases examined in this report. In the second model, losses of entrained eggs and larvae were "hindcast" to the numbers of adult females required to



produce them, using estimates of the average total lifetime fecundity of a female fish and the expected survival rate from the egg to the larval stage. For purposes of model comparison, throughout this report the first approach is termed the “Forward Projection” (FP) approach; the second approach is termed the “Fecundity Hindcasting” (FH) approach.

Tenera (2000) provided equations for calculating the variance in both FP and FH estimates, and evaluated the sensitivity of the results to uncertainty in input parameters. The FP model was applied to Pacific sardine, northern anchovy, blue rockfish, blackeye goby, and sanddab; the FH model was applied to these same species plus brown rock crab, slender crab, and white croaker. In some cases the FP and FH models produced similar estimates of equivalent adult losses, but in other cases the results differed substantially. Neither method produced consistently lower or higher values. Tenera (2000) did not perform a formal economic benefits analysis. As one measure of the impacts of entrainment on northern anchovy and white croaker, Tenera extrapolated adult equivalent estimates for these species to dollars using average weights of fish and average landed prices in the commercial fishery.



**Figure 3-3**  
Overview of EA approach used at Diablo Canyon (Tenera 2000).

### 3.4 Salem and Mercer

PSEG (1999, 2001) used EA and PF models in entrainment and impingement impact assessments prepared for the Salem and Mercer Generating Stations on the Delaware Estuary. The Species addressed at Salem included weakfish, Atlantic croaker, spot, striped bass, white perch, American shad, alewife, and blueback herring. Species addressed at Mercer included striped bass, white perch, American shad, alewife, blueback herring, channel catfish, bluegill, and blue crab.

In these assessments, the models were used primarily as inputs to economic benefits assessments rather than as measures of impacts on populations. The EA model was applied in a two-step process (Figure 3-4a). In the first step, entrainment and impingement losses were converted to “age-1-equivalent” losses, i.e., to equivalent numbers of one-year-old fish or crabs. For finfish, stage-specific mortality rates for eggs, yolk-sac larvae, post yolk-sac larvae, and juveniles were derived from the scientific literature. For blue crab impinged at Mercer, age-1 equivalents were estimated using size-based rather than stage-based mortality rates. Impinged crabs were classified into 10-mm length classes; estimates of the number of crabs that would have survived to reach the average size of an age-1 blue crab were calculated from (1) an estimate of the average survival for each 10-mm length increment, and (2) the number of length-class transitions required to reach age-1 size.

In the second step, yield-per-recruit models were used to calculate the yield foregone by fishermen due to entrainment and impingement losses, for all ages subsequent to age 1. These models used estimates of age-specific natural mortality and fishing mortality for each species to calculate the expected lifetime harvest that would be obtained per age-1 equivalent fish. Mortality rates documented in National Marine Fisheries Service (NMFS) stock assessments were used when available. Where data permitted, yield reductions were subdivided into recreational vs. commercial fishing, and separate economic benefits assessments were performed for each fishery.

PF models were used for both the Salem and the Mercer assessments (Figure 3-4b), however, different models were used in each and neither assessment used the Rago (1984) model. For Salem, a cohort simulation model with a daily time step was used to calculate production foregone for bay anchovy (PSEG 1999, Appendix F, Attachment 4). Entrainment and impingement estimates, by life stage (for age 0 fish) and age (for age 1+ fish) were tabulated on a weekly basis. For all members of a given stage or age entrained or impinged during a given week, future growth and mortality were tracked over the expected lifespan of the species. The daily growth increment per individual was simulated using a stage-specific Von Bertalanffy growth function. The fraction of the organisms alive at the beginning of the day that would be expected to die during that day was calculated using stage-specific daily mortality rates. The production foregone for any cohort on any day was calculated as the number that would have been alive at the beginning of the day multiplied by the fraction dying during that day and by the expected weight at the end of that day. The number that would have survived to the end of the day was used as the input number for the next day’s calculation. This step was repeated for every day through the end of the life stage within which a fish was entrained or impinged, after

which the simulation continued using the growth function and daily mortality rate for the next stage. The total production foregone for any year was then calculated by summing PF values over all days and over all cohorts. The total annual PF values were then converted to foregone yield of predators by assuming 10% trophic transfer efficiency and 100% harvest of all foregone predator biomass.

The Mercer assessment used a model termed the “biomass lost model” (BLM) to develop PF estimates for spottail shiner. The BLM model uses the same stage or age-specific mortality rates used by the EA model, however, rather than focusing on the fraction of fish surviving to some age of equivalence, the BLM focuses on the fraction dying at each stage or age over the lifespan of the species. The production foregone at any given stage or age was estimated from the number of entrained or impinged fish that would have been alive at the beginning of the stage or age, the fraction expected to die during that stage or age interval, and the average weight per fish at that stage or age. As in the Salem assessment, the PF values were then converted to foregone yield of predators by assuming 10% trophic transfer efficiency and 100% harvest of all foregone predator biomass.

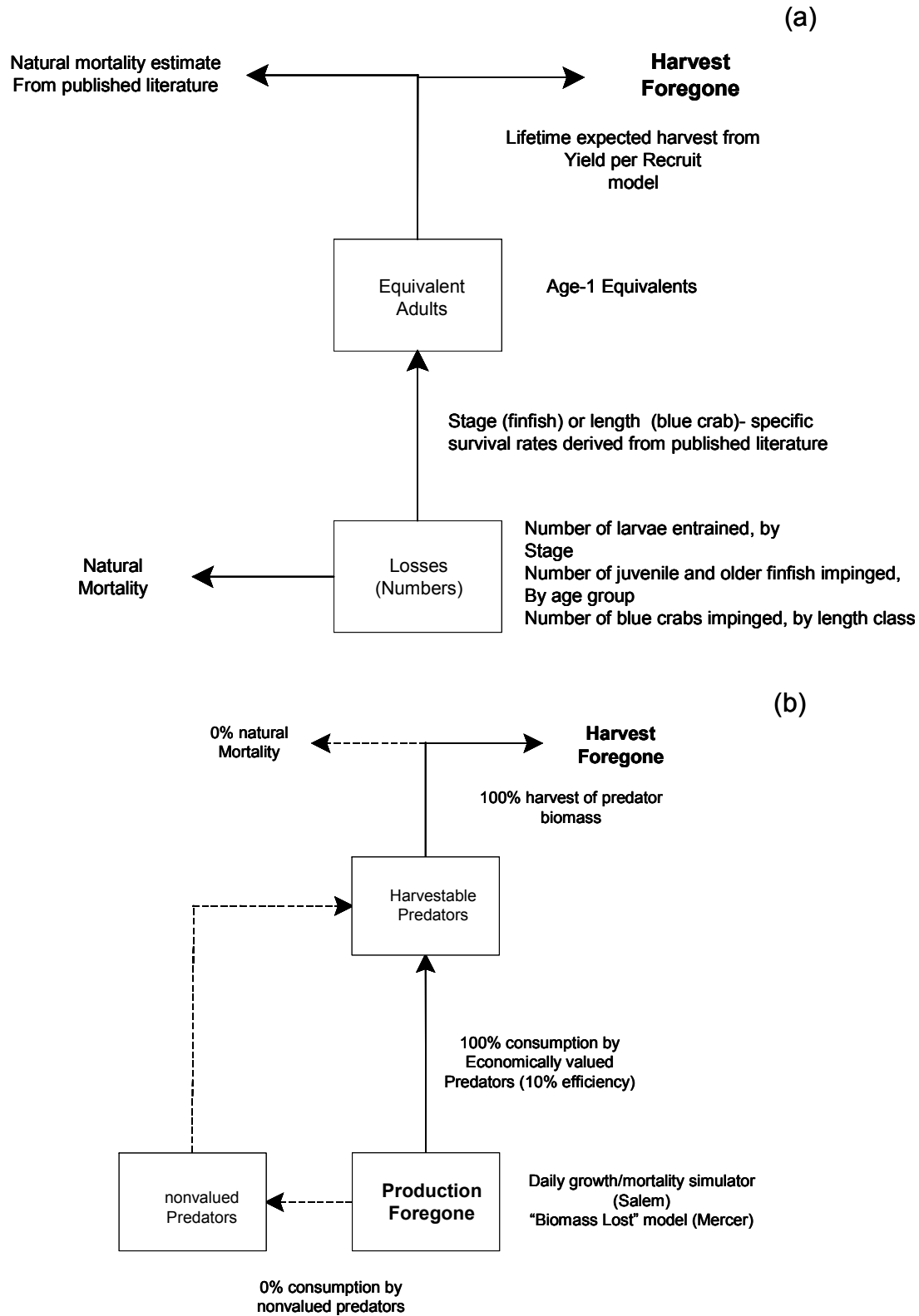
### 3.5 EPA Section 316(b) Case Study

EPA (USEPA 2002) used EA and PF models in a benefits analysis performed to support the proposed Section 316(b) Phase II Existing Facility Rule. EPA applied the models to data from numerous power plants located on the east coast (Salem, Brayton Point, Pilgrim, Seabrook), gulf coast (Big Bend), Ohio River (W.H. Sammis, Cardinal, Kammer, Philip Sporn, Kyger Creek, W. C. Beckjord, Miami Fort, Tanners Creek, Clifty Creek), Great Lakes ( J.R. Whiting, Monroe), and Pacific Coast (Pittsburg, Contra Costa).

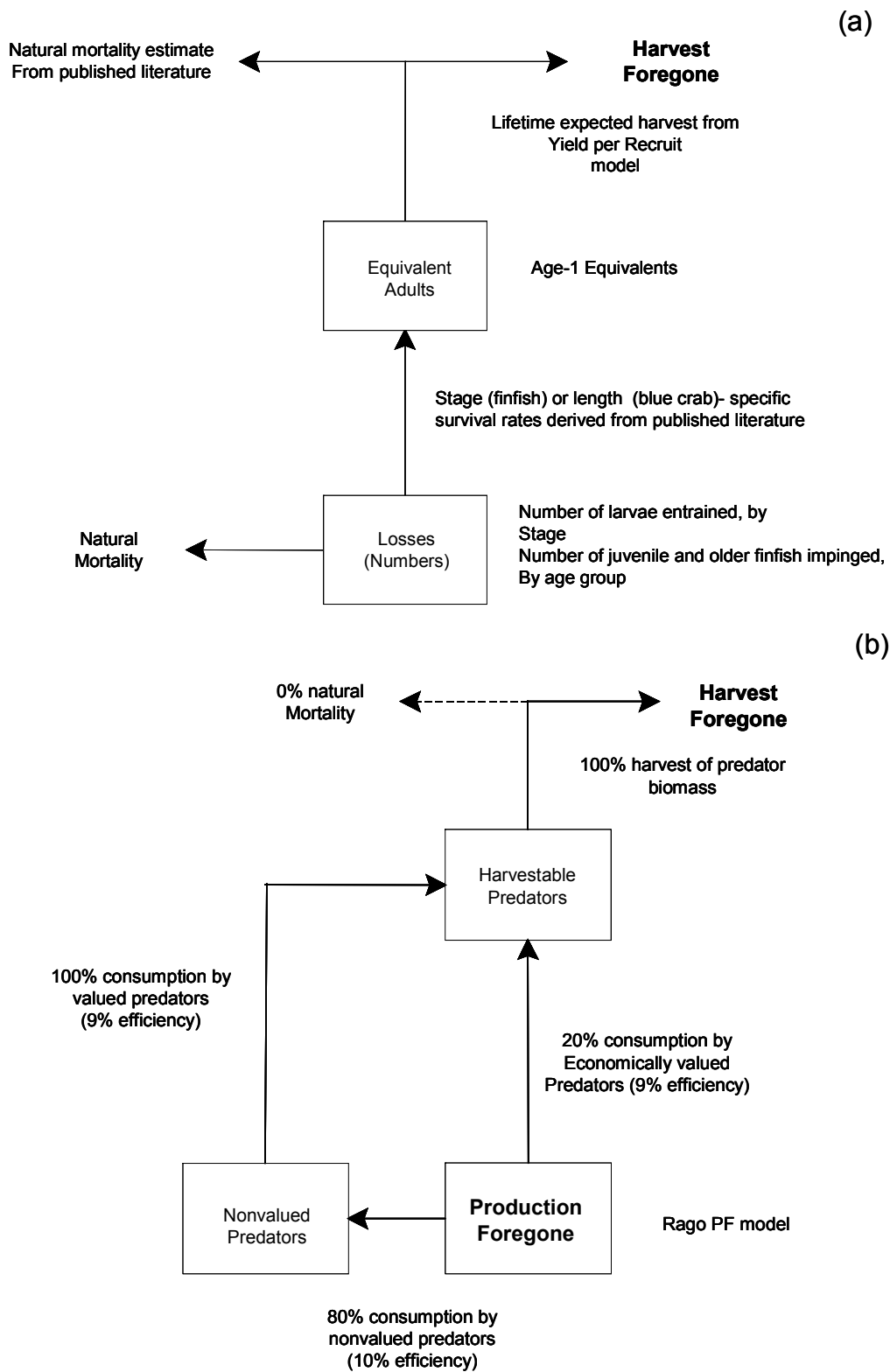
The EA methodology used by EPA was similar to the approach used at Salem and Mercer (Figure 3-5a). Entrainment and impingement losses of exploited species were converted to age-1 equivalents, and then extrapolated to yield reductions using yield-per-recruit models. Yield reductions were apportioned between commercial fisheries and recreational fisheries and valued separately.

EPA used the Rago (1984) model to calculate PF for forage species (Figure 3-5b). Rather than assuming 100% consumption of production by harvestable predators, EPA assumed that only 20% of the PF would be directly consumed by economically valuable predators.

The remaining production was assumed to be converted to predator production by a two-step indirect transfer pathway, being consumed first by nonvalued predators which are in turn consumed by economically valuable predators. The combination results in a net 2.5% transfer efficiency. EPA assumed 100% harvest of the foregone predator biomass.



**Figure 3-4**  
Overview of EA and PF approaches used at Salem (PSEG 1999) and Mercer (PSEG 2001).



**Figure 3-5**  
Overview of EA and PF approaches used in the EPA Section 316(b) case study (USEPA 2002).

### 3.6 Summary of Historical Studies

Although the methods used in the most recent applications of EA and PF models are somewhat more complex than in earlier applications, the cases discussed in this report share numerous common elements. All of the EA models are stage/age based, meaning that the key biological input parameters are stage-specific (for the various life stages within the first year of life) and age-specific (for one-year-old and older fish) mortality rates. In most of these studies, early life stage losses were projected forward to estimates of equivalent adult losses. The only exception is the “Fecundity Hindcasting” (FH) model used at Diablo Canyon, in which early life stage mortality rates and lifetime female egg production rates were used to hindcast estimates of the number of adult females required to produce the lost organisms. The PF models examined are more diverse, with three distinctly different models being used: daily mortality/growth simulation (Salem), a biomass loss model analogous to the classical EA model (Mercer), and the Rago PF model (EPA). The daily mortality/growth and Rago models use a growth function; the BLM uses only stage-specific average weights. It is not clear, however, whether these approaches are really different because the Rago model’s growth rate function would be parameterized based on stage-specific weights. Without direct comparisons between different models applied to the same data sets, it is impossible to determine whether any of the three PF approaches is inherently superior to the others with respect to conceptual validity, computational simplicity, data requirements, or sensitivity to parameter values. The Chalk Point assessment is somewhat anomalous, in that an EA model was used for the same purpose that PF models were used in the other case studies, i.e., to calculate a reduction in predator biomass resulting from entrainment of prey.

Despite the similarity of the modeling approaches, the case studies differ substantially in the way the model results are extrapolated to reductions in harvest of economically valuable species. The EPA PF approach assumes a net 2.5% transfer efficiency to harvestable predators, primarily because of its assumption that only 20% of the foregone production is directly consumed by economically valuable species. All of the other assessments assumed 100% consumption by economically valuable species, with transfer efficiencies ranging from 10% (Salem, Mercer, Brunswick) to 30% (Chalk Point). All of the case studies except Salem and EPA assumed that 100% of the estimated reduction in harvestable biomass would have been harvested. In these two studies, the EA calculations used yield-per-recruit models to account for natural mortality of harvestable fish.

With respect to the water bodies and species addressed, it is clear from Table 2-1 that EA and PF models have been applied far more frequently to marine and estuarine water bodies than to other water body types, however, examples exist for every major waterbody type in North America. The models have been applied to a wide variety of marine, estuarine, and freshwater species. These historical applications clearly provide an ample foundation for Section 316(b) assessments.

# 4

## EXPLANATION OF EQUATIONS AND PARAMETERS

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This section documents the equations, variables, and parameters used in most of the models described in Section 3. The terminology and symbols used in many of the original references have been modified to ensure consistency, but the equations themselves have not been altered. However, in several cases the models have been extended to account for factors or conditions not considered by the original authors. In particular, several of the models have been extended to account for entrainment or impingement of fish throughout a life stage rather than only at the beginning of that stage. These extensions were made to ensure consistency between the assumptions used in different models. Whether the extended version or the original version is appropriate for a given assessment is best determined through evaluation of available data concerning the age or length distribution of entrained or impinged fish.

### 4.1 Equivalent Adult Models

As noted in Section 3, two approaches to equivalent adult modeling were identified in the historical review. The first approach, termed here the forward projection (FP) approach, uses estimates of age or stage-specific survival fractions to scale entrainment and impingement losses to numbers of adults surviving to some future age, termed the “age of equivalence.” The second approach, termed here the fecundity hindcasting (FH) approach, uses estimates of fractional survival from the egg stage to the stage/age of entrainment or impingement to calculate the number of female fish required to produce the lost fish.

Although it might intuitively be expected that, given the same input data, these models would produce identical estimates of equivalent adult losses, in practice these models can produce very different estimates. As shown in Appendix B, the two models provide identical estimates only if the life history parameters used correspond to a perfectly balanced population that is neither increasing nor decreasing.

#### 4.1.1 Forward Projection Approach

In the FP approach, the losses at any given stage or age are simply multiplied by the fraction of fish at that stage or age that would be expected to survive to the age of equivalence:

$$EA = S_A N \quad (\text{Eq. 1})$$

*Explanation of Equations and Parameters*

Where:

$EA$  = equivalent adult loss

$N$  = number of fish lost due to entrainment or impingement

$S_A$  = fraction of fish expected to survive from the age at which they are impinged or entrained to the age of equivalence

One of the major benefits of this model is that it can be used to express losses imposed on different ages or life stages in common equivalent adult units:

$$EA = \sum_{i=1}^n S_{i,A} N_i \quad (\text{Eq. 2})$$

Where:

$N_i$  = number of fish lost at age  $i$

$S_{i,A}$  = fraction of fish expected to survive from age  $i$  to the age of equivalence

Survival rates of early life stages of fish are often expressed on a life-stage-specific basis (e.g., eggs, larvae, juveniles), so that the fraction surviving from any particular life stage to adulthood is expressed as the product of survival fractions for all of the life stages through which a fish must pass before reaching adulthood.

$$S_{i,A} = \prod_{j=i}^{j_{\max}} S_j \quad (\text{Eq. 3})$$

Where:

$j_{\max}$  = the stage immediately prior to the age of equivalence

As defined above, the life stages can have varying durations, ranging from a few days for eggs to several months for juveniles. Very long life stages, especially the juvenile stage, can be broken down into substages. For example, PSEG (1999) subdivided the juvenile life stage into two substages, termed J1 and J2. Survival rates for long life stages can also be broken down by week or by month, to accommodate impingement or entrainment data reported on a weekly or a monthly basis.

**Variables and parameters used in forward projection models**

$EA$  = equivalent adult losses

$N_i$  = losses of fish at stage or age  $i$

$S_{i,A}$  = fraction of fish expected to survive from age  $i$  to the age of equivalence

$S_j$  = survival fraction from stage  $j$  to stage  $j+1$

$j_{\max}$  = the stage immediately prior to adulthood



The probability that a fish entrained or impinged at any given life stage would have survived to adulthood is greater if the fish is near the end of that stage than if it is at the beginning of that stage, because it would have already survived most of the natural mortality that occurs during that stage. However, Equation 3 as written assumes that all fish lost at a given stage are lost at the beginning of that stage. For early life stages, in which mortality rates are very high, this assumption could lead to a substantial underestimation of the actual equivalent adult losses. The reason for this potential bias is that organisms entrained near the end of a given life stage have survived most of the mortality risk imposed on that stage and have a much higher probability of surviving to the next stage (and to subsequent stages) than do organisms entrained at the beginning of a stage. In some cases (e.g., for entrainable juveniles), data on the lengths of the entrained fish can be used to determine the age distribution of entrainment losses. If this is not possible, the most reasonable assumption is that all ages within a life stage are equally vulnerable. Assuming equal vulnerability throughout a life stage  $i$ , the survival fraction for stage  $i$  is adjusted as follows:

$$S_i^* = 2S_i e^{-\ln(1+S_i)} \quad (\text{Eq. 4})$$

The derivation of Equation 4 is provided in Appendix A to this report. The adjustment is applied only to the stage at which entrainment or impingement occurs. Equation 3 then becomes

$$S_{i,A} = S_i^* \prod_{j=i+1}^{j_{\max}} S_j \quad (\text{Eq. 5})$$

Caution is needed when applying the adjustment factor, because inappropriate application can lead to substantial overestimation of equivalent adult losses. Overestimation is likely to occur when susceptibility to entrainment or impingement declines within a stage (e.g., because of growth or settlement); the potential magnitude of the bias increases with the duration of the life stage in question. If a comparison between the length distribution of entrained or impinged fish and the length distribution of fish present in the source water body indicates that entrainment or impingement is largely limited to the smallest or youngest individuals in a particular life stage, the adjustment factor should not be applied.

The above equations can also be applied to situations in which fish are classified by length rather than by stage, provided that estimates of fractions surviving from one length class to the next can be obtained. Where entrainment losses are tabulated separately by sampling interval (e.g., day, week, or month), the model can be applied separately to each cohort of entrained or impinged fish. The equivalent adult losses for each cohort (e.g., all of the eggs, larvae, and juveniles entrained in a given week) are calculated separately, and then summed over all cohorts.

#### 4.1.2 Fecundity Hindcasting Approach

The FH approach is, in essence, an equivalent adult model that operates in reverse. Given the number of organisms entrained or impinged at a given stage, the model “hindcasts” the number of adult female fish that would have been required to produce the entrained or impinged fish:

Explanation of Equations and Parameters

$$EF = \frac{1}{E_L} \sum_{i=1}^n \frac{N_i}{S_{Ei}} \quad (\text{Eq. 6})$$

Where:

$EF$  = number of equivalent adult (sexually mature) females

$S_{Ei}$  = fraction of eggs expected to survive to age or stage  $i$

$E_L$  = expected lifetime fecundity of a female fish

The expected lifetime fecundity of a female fish is a function of the proportion of females expected to spawn at any given age, age-specific fecundity, and age-specific survival:

$$E_L = \sum_{i=A_{\min}}^{A_{\max}} S_{Ai} M_i E_i \quad (\text{Eq. 7a})$$

Where:

$S_{Ai}$  = fraction of females expected to survive from age  $A_{\min}$  to age  $i$

$M_i$  = fraction of age  $i$  females that are sexually mature

$E_i$  = average number of eggs spawned by a mature female at age  $i$

$A_{\min}$  = minimum age of a sexually mature female

$A_{\max}$  = oldest age in population

**Variables and parameters used in fecundity hindcasting models**

$EF$  = number of equivalent adult (sexually mature) females

$N_i$  = losses of fish at stage or age  $i$

$S_{Ei}$  = fraction of eggs expected to survive to age or stage  $i$

$S_{Ai}$  = fraction of females expected to survive from age  $A_{\min}$  to age  $i$

$A_{\min}$  = minimum age of a sexually mature female

$A_{\max}$  = oldest age in population

$M_i$  = Fraction of age  $i$  females that are sexually mature

$E_i$  = average number of eggs spawned by a mature female at age  $i$

$E_L$  = expected lifetime fecundity of a female fish

$E_{ave}$  = annual fecundity of adult fish averaged over all ages and sizes

Tenera (2000) did not calculate expected lifetime fecundity using Equation 7 because estimates of age-specific survival and fecundity of adult females were not available for any of the species of interest. Instead, Tenera (2000) used as an approximation the midpoint of the adult lifespan multiplied by literature-derived estimates of the average annual fecundity per adult female.

$$E_L \equiv \left( \frac{A_{\max} - A_{\min}}{2} \right) E_{ave} \quad (\text{Eq. 7b})$$

Where:

$E_{ave}$  = annual fecundity of adult fish averaged over all ages and sizes

As with the FP approach, life stages can be defined by length rather than stage, and daily, weekly, or monthly cohorts can be treated separately depending on the temporal resolution of the available loss data.

As documented by Tenera (2000), the FH approach assumes that all organisms entrained or impinged at any life stage are lost at the beginning of that life stage. For the same reasons discussed in section 4.1.1, if organisms are entrained or impinged throughout a life stage, this assumption results in an underestimation of the equivalent number of females required to produce the lost organisms. The adjusted survival rate, as defined in Equation 4, can be used to correct for this potential bias. However, since the age interval of interest is the interval *prior* to the median age-at-death, the fraction surviving *to* the median age-at-death rather than the fraction surviving from the median age-at-death to the end of the stage is used for the adjustment:

$$EF = \frac{1}{E_L} \sum_{i=1}^n \frac{N_i}{S_{Ei} (S_i / S_i^*)} \quad (\text{Eq. 8})$$

The derivation of the adjustment factor is provided in Appendix A to this report. As with the forward projection approach, inappropriate application of the adjustment factor for the fecundity hindcasting approach can lead to substantial overestimation of equivalent adult losses.

Overestimation is likely to occur when susceptibility to entrainment declines within a stage (e.g., because of growth or settlement); the potential magnitude of the bias increases with the duration of the life stage in question.

## 4.2 Production Foregone Models

This section discusses two of the three production foregone models identified in the historical review: The Rago (1984) model, which was used in the EPA case study (Section 3.5), and the biomass lost model (Section 3.4) which was used in the Mercer 316(b) Demonstration. The cohort simulation model used at Salem (Section 3.4) is not included, for two reasons. First, this model appears to provide no conceptual or practical advantages relative to the other two models. Second, the data needed to support a daily simulation approach to growth and mortality are unlikely to be available for most facilities.

### 4.2.1 Rago Approach

Using Rago's (1984) production foregone model, the production foregone due to fish entrained or impinged at any given life stage or age is calculated by integrating the instantaneous rates of growth and mortality over that stage or age:

$$P_i = \frac{G_i N_i \bar{W}_i (\exp(G_i - Z_i) - 1)}{G_i - Z_i} \quad (\text{Eq. 9})$$

Where:

- $P_i$  = production foregone for a specific age or stage  $i$
- $G_i$  = instantaneous growth rate for individuals of age or stage  $i$ ,
- $Z_i$  = instantaneous total mortality rate for individuals of age or stage  $i$ ,
- $\bar{W}_i$  = average weight of individuals of age or stage  $i$

Equation 9 applies only to the age or stage at which a particular fish or group of fishes is entrained or impinged. To account for all of the production foregone due to the loss of those fish, it is necessary to calculate the PF at later stages and ages. Equation 9 can be used in these calculations, however, estimates of PF at any future stage  $j$  must be adjusted to account for natural mortality occurring between stage  $i$  and stage  $j$ . The PF at any future age or stage  $j$ , for a fish entrained at age or stage  $i$ , is equal to the PF at stage  $j$  (from Equation 9) multiplied by the fraction of fish expected to survive from stage  $i$  to stage  $j$ :

$$P_{i,j} = P_j S_{i,j} \quad (\text{Eq. 10})$$

Where:

- $P_{i,j}$  = production foregone at life stage  $j$ , due to fish entrained or impinged at life stage  $i$
  - $S_{i,j}$  = fraction of fish expected to survive from age  $i$  to age  $j$
- By analogy with Equation 3, the term  $S_{i,j}$  in Equation 10 is given by:

$$S_{i,j} = \prod_{k=i}^{j-1} S_k \quad (\text{Eq. 11})$$

The total production foregone for fish entrained or impinged at age or stage  $i$ , for the entire expected life span (e.g., to some maximum age  $A_{\max}$ ) of a fish alive at the beginning of age or stage  $i$  (termed  $P_{i,T}$ ), is equal to the expected production at each future age or stage multiplied by the probability of surviving to that stage or age:

$$P_{i,T} = \sum_{j=i}^{A_{\max}} P_j S_{i,j} \quad (\text{Eq. 12})$$

When multiple ages and stages are entrained and impinged in a given year, the total production foregone due to all of these losses ( $P_T$ ) is given by:

$$P_T = \sum_{i=0}^{A_{\max}} P_{i,T} \quad (\text{Eq. 13})$$

As published and as used in historical assessments, the Rago model assumes that all organisms entrained or impinged at a stage are entrained or impinged at the beginning of that stage. However, to be fully consistent with the equivalent adult models documented in Section 4.1, the model should account for the fact that fish may in many cases be entrained or impinged throughout a given age or life stage rather than only at the beginning of that age or stage. As documented above, the model overestimates the production foregone during the stage in which entrainment or impingement occurs because the lost organisms were available as prey up until the day on which the loss occurs. However, the model underestimates production foregone during later stages because it underestimates the fraction of the lost organisms that would have survived to reach later stages.

#### Variables and parameters used in the Rago production foregone model

$N_i$  = number of fish lost due to entrainment or impingement at age or life stage  $i$   
 $P_i$  = production foregone for a specific age or stage  $i$   
 $P_{ij}$  = production foregone at life stage  $j$ , due to fish entrained or impinged at age or life stage  $i$   
 $P_{i,T}$  = total lifetime production foregone due to fish entrained or impinged at age or life stage  $i$   
 $P_T$  = total lifetime production foregone due fish entrained or impinged at all ages or life stages  
 $S_{ij}$  = fraction of fish expected to survive from age  $i$  to age  $j$   
 $G_i$  = instantaneous growth rate for individuals of age or stage  $i$ ,  
 $g_i$  = daily instantaneous growth rate for individuals of age or stage  $i$   
 $Z_i$  = instantaneous total mortality rate for individuals of age or stage  $i$ ,  
 $z_i$  = daily instantaneous total mortality rate for individuals of age or stage  $i$   
 $d_i$  = duration of life stage  $i$  (days)  
 $\hat{d}_i$  = median age-at-death for individuals in life stage  $i$   
 $\bar{W}_i$  = average weight of individuals at the start of age or stage  $i$

It is possible that in practice the positive and negative biases are approximately equal. However, the Rago model can be easily modified to account for both biases. The adjustment involves dividing the total growth ( $G_i$ ) and mortality ( $Z_i$ ) rates, which are measured over the entire duration of life stage  $i$ , by the duration of life stage  $i$  ( $d_i$ ) to convert them to daily growth ( $g_i$ ) and total mortality ( $z_i$ ) rates. Equation 9 can then be expressed as:

$$P_i = \frac{G_i N_i \bar{W}_i \{\exp[(g_i - z_i)d_i] - 1\}}{G_i - Z_i} \quad (\text{Eq. 14})$$

*Explanation of Equations and Parameters*

As shown in Appendix A, the median age-at-death for organisms that are alive at the beginning of life stage  $i$  but die during life stage  $i$ , as measured from the beginning of the stage ( $\hat{d}_i$ ), is a function of the daily instantaneous mortality rate ( $z_i$ ) and the stage duration ( $d_i$ ). Like the adjusted equivalent adult model, the adjusted production foregone model assumes that organisms are entrained or impinged at the median age-at-death within a stage rather than at the beginning of the stage. To make the adjustment, it is necessary only to substitute the time interval between the median age at death and the end of the stage for the stage duration in Equation 14:

$$P_i = \frac{G_i N_i \bar{W}_i \{\exp[(g_i - z_i)(d_i - \hat{d}_i)] - 1\}}{G_i - Z_i} \quad (\text{Eq. 15})$$

Equation 11 can also be readily modified to account for underestimation of the fraction of fish lost at stage  $i$  that would have survived to subsequent stages, using the same adjustment factor defined in Equation 5. By analogy with Equations 4 and 5, the fraction of fish expected to survive from age  $i$  to age  $j$  (Equation 11) can be written as:

$$S_{i,j} = S_i^* \prod_{k=i+1}^{j-1} S_k \quad (\text{Eq. 16})$$

Substitution of this equation for Equation 11 corrects the bias. Like adjusted equivalent adult models, adjusted production foregone models can overestimate future production foregone if the susceptibility of organisms to entrainment or impingement declines during a given age or life stage.

Another, usually unrecognized, problem with the Rago model is that it estimates only the *biomass production* lost, and not the total biomass lost due to entrainment and impingement. The weight of the entrained and impinged organisms at the time of death, which should logically be included in the total loss of biomass consumed by predators, is not included in the estimate. Adding the biomass of entrained and impinged organisms to the production foregone corrects for this bias.

#### **4.2.2 Biomass Lost Model**

The model employed at Mercer (PSEG 2001), calculates the present and future biomass lost due to entrainment and impingement of forage species by summing the average biomass of the fish expected to die at each life stage. This sum includes (1) biomass lost due to fish entrained or impinged at that life stage, and (2) biomass lost due to fish entrained or impinged at earlier stages that would otherwise have survived to reach that stage and then died during that stage:

$$BL = \sum_{j=1}^k \left\{ \sum_{i=1}^n [(N_i S_{i,j}) - (N_i S_{i,j+1})] W_j \right\} \quad (\text{Eq. 17})$$

Note that, while the weights used in the Rago model are the weights of fish *at the beginning* of each stage or age, the weights use in the BLM are *average* weights of organisms during each stage or age.

### Variables and parameters used in the Biomass Lost Model

$N_i$  = losses of fish at stage or age  $i$

$S_{ij}$  = fraction of fish expected to survive from stage or age  $i$  to age  $j$

$S_{i,j+1}$  = fraction of fish expected to survive from stage or age  $i$  to stage or age  $j+1$

$S_{i,j}^*$  and  $S_{i,j+1}^*$  = corresponding adjusted survival fractions

$W_j$  = average weight of fish at stage or age  $j$

The BLM can be adjusted to account for entrainment and impingement of fish throughout a life stage rather than only at the beginning of a life stage. Equation 17 calculates biomass lost as the difference between the numbers present at the beginning of a life stage and the number present at the end of the life stage, multiplied by the average weight of fish during that life stage. Half of expected deaths during that stage should occur prior to the median age-at-death for that stage, and half afterwards. Only the production occurring after the median age-at-death should be counted as production foregone. The simplest way to adjust the model to account for this bias is to subtract one-half of the production calculated for the life stage during which entrainment or impingement occurs. The bias due to underestimation of the fraction surviving to the next stage can be accounted for using the adjusted survival rate as discussed in previous sections:

$$BL = \sum_{j=1}^k \left\{ \sum_{i=1}^n [(N_i S_{i,j}^*) - (N_i S_{i,j+1}^*)] W_j \right\} - \sum_{j=1}^k \frac{N_j (1 - S_{j,j+1}) W_j}{2} \quad (\text{Eq. 18})$$

Unlike the Rago model, the BLM accounts for the biomass of organisms at the time they are entrained or impinged. Like adjusted equivalent adult models, adjusted production foregone models can overestimate future production foregone if the susceptibility of organisms to entrainment or impingement declines during a given age or life stage.

### 4.3 Equivalent Yield Model

The Equivalent yield model calculates the loss to fishermen due to entrainment and impingement of economically valuable species using Baranov's Catch Equation (Ricker 1975), a model that is widely used in fisheries science and management:

$$Y = EA_R \sum_{i=R}^n S_{R,i} W_i \left[ \frac{F_i (1 - e^{-Z_i})}{Z_i} \right] \quad (\text{Eq. 19})$$

### Variables and Parameters used in the Equivalent Yield Model

$EA_R$  = Equivalent adult losses calculated at the age of entry to the fishery

$R$  = Age of entry to the fishery

$N_j$  = Number of fish impinged at age  $j$ , where  $j > R$

$S_{R,i}$  = fraction of fish expected to survive from the age of entry to age  $i$

$W_i$  = Average age of a fish at age  $i$

$N$  = total number of age classes present in the fishery

$F_i$  = annual instantaneous rate of fishing mortality at age  $i$

$Z_i$  = annual instantaneous rate of total mortality (natural mortality + fishing mortality) at age  $i$

The specific value chosen for the age of entry is irrelevant, provided that it is smaller than or equal to the youngest age at which fish are harvested. The reason for this is that, for ages younger than the actual age of entry, the rate of fishing mortality is equal to zero and there is no yield foregone. If, however, some fish are impinged at an age *older* than the age of entry, Equation 19 must be modified. The reason is that for these older fish there is no yield foregone prior to the age at which they were impinged. Presumably, they were among the fish that survived exposure to the fishery at these earlier ages.

$$Y = EA_R \sum_{i=R}^n S_{R,i} W_i \left[ \frac{F_i (1 - e^{-Z_i})}{Z_i} \right] + \sum_{j=R+1}^n N_j \sum_{k=j}^n S_{j,k} W_j \left[ \frac{F_k (1 - e^{-Z_k})}{Z_k} \right] \quad (\text{Eq. 20})$$

According to Equation 20, foregone yield due to impingement of fish older than the age of entry is accrued beginning at the age at which the fish were impinged.



# 5

## GUIDANCE ON PARAMETER ESTIMATION

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This section provides some general guidance on procedures for estimating the life history parameters used in EA and PF models. References to studies documenting the development of these parameters are also provided. It's not possible to be complete because of the variety of methods that have been used by various investigators to develop parameter estimates. Separate sections are provided for age 0 and for age 1 and older fish because the data and methods available for parameter estimation differ greatly between these two age groups. Specific recommended values for commonly entrained and impinged fish species in marine and freshwater environments will be documented in a future EPRI report.

### 5.1 Age 0 Fish

The critical early life stage parameters used in EA and PF models include stage-specific survival fractions, cumulative survival fractions over several life stages, instantaneous mortality rates, stage-specific weights, and instantaneous growth rates. Stage-specific survival fractions and instantaneous mortality rates are linked by the following mathematical relationships:

$$S_j = e^{-Z_j} = e^{-d_j z_j} \quad (\text{Eq. 21})$$

Where:

- $S_j$  = survival fraction from stage  $j$  to stage  $j+1$
- $Z_j$  = stage-based instantaneous mortality rate for stage  $j$
- $d_j$  = duration of stage  $j$ , in days
- $z_j$  = daily instantaneous mortality rate for stage  $j$

As discussed in Section 4.1.1 and defined in Equation 3, the cumulative survival fraction from any stage  $j$  to any future stage  $k$  is simply the product of the survival fractions for each of the intervening stages. For example, the fraction of eggs expected to survive to reach the post yolk-sac stage is simply the product of the fraction of eggs surviving to the yolk-sac stage and the fraction of yolk-sac larvae surviving to the post yolk-sac stage.

At least in principle, fractional survival rates for individual life stages can be calculated from estimates of the total numbers of each life stage present in the population during a given year:

$$S_j = \frac{N_{j+1}^P}{N_j^P} \quad (\text{Eq. 22})$$

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Where:

$N_{j+1}^P$  = number or density of life stage  $j+1$  individuals present in the population, and  
 $N_j^P$  = number or density of life stage  $j$  individuals present in the population

This method was used by Polgar (1977) to estimate mortality rates for striped bass eggs and larvae in the Potomac River.

More commonly, survival fractions are estimated indirectly, from daily instantaneous mortality rates and life stage durations, using Equation 21. In this method, abundances or densities are measured periodically (e.g., weekly) throughout the period during which a given life stage is present, and the daily instantaneous mortality rate ( $z_j$ ) is estimated from the rate of decline in abundance of the life stage:

$$\ln N_{j,t}^P = \ln N_0^P - z_j t \quad (\text{Eq. 23})$$

Where:

$N_{j,t}^P$  = number or density of lifestage  $j$  individuals present in the population  $t$  days following the start of the life stage, and  
 $N_0^P$  = number or density of life stage  $j$  individuals present at the beginning of the life stage

In this approach, estimates of  $z_j$  are obtained by fitting a linear regression model to a sequential series of estimates of the abundance or density of life stage  $j$ . This approach was used by Klauda et al. (1988) to estimate mortality rates for white perch post yolk-sac larvae and juveniles in the Hudson River, by Houde and associates for eggs, larvae, and juveniles of several fish species (Cowan and Houde 1989; Dorsey et al. 1996; Houde et al. 1989), and by PSEG (1984, 1999) for various bay anchovy and weakfish life stages in the Delaware Estuary. Life stage durations for eggs and yolk sac larvae can often be estimated from laboratory studies; durations of older stages must be inferred from the duration of the period during which organisms of a given stage are present.

Both of the above methods are subject to significant limitations. Spawning in most fish populations is spread out over an interval ranging from a few weeks to several months, so that the population of any life stage present at a given time is composed of individuals who have been in that stage for varying amounts of time. This means that the total number of organisms in that stage produced during a given year cannot be estimated from either a single estimate of abundance or by summing a series of estimates. The rate of decline in apparent abundance is a function of the relative rates of recruitment, mortality, and development of individuals belonging to that life stage. A second complication is that the spatial distribution of organisms changes because of hydrological factors and age. Changes in spatial distribution both within and between stages introduces biases, because mortality is confounded with movement. Finally, the susceptibility of organisms to typical sampling gear changes, and usually decreases with size or age, so that estimates of daily rate of decline overestimate mortality.

The most accurate method currently available involves using otolith increments to track abundance and mortality for individual day-cohorts of organisms. This is the method used by Houde (1989) to estimate daily mortality rates for bay anchovy larvae. The abundance of each cohort is tracked through time, and a mortality rate is estimated using Equation 22. When spawning is spread out over an extended interval, this method can be used to estimate variations in mortality rates between cohorts spawned on different dates. The otolith-based approach eliminates biases due to temporally distributed spawning, but is still affected by spatial movements and gear avoidance. Moreover, the approach can only be applied to species for which daily increments can be accurately identified and counted.

The approaches that can be used to estimate weights and growth rates for use with PF models differ depending on the model being used. For both of the models discussed in this guidance, weight is measured as wet (fresh) weight rather than dry weight. The Rago model requires estimates of the average weight of organisms at the beginning of each life stage. For life stages other than eggs, initial weights can be obtained from the size distributions of individuals within each life stage, determined from ichthyoplankton sampling. The weight parameter for each life stage is simply the average weight of the smallest organisms in that life stage. The Rago model assumes that growth within a given life stage is exponential. Hence, the growth rate parameters are estimated from the initial weights of the life stages:

$$G_i = \ln(\overline{W}_{i+1} / \overline{W}_i) \quad (\text{Eq. 24})$$

The BLM uses average weights rather than initial weights. There are several different ways in which an average weight could be estimated. In principle, the stage-specific average weight used in the model should be the average weight of the organisms consumed by predators. This quantity is a function of the stage-specific growth and death rates, and could be calculated if a suitable growth model is available. In practice, however, it may be sufficient to simply approximate the average weight-at-death from the median or mean weight of organisms collected in ichthyoplankton samples, or from the midpoint between the initial weights of successive life stages.

## 5.2 Age 1 and Older Fish

For the models discussed in this guidance, the key parameters for age 1 and older fish include age-specific survival fractions, fishing mortality rates, growth rates, average weights, and fecundities. Estimates of most of these parameters are often available from resource management agency reports or from the published scientific literature. However, certain precautions must be taken in using these estimates.

### 5.2.1 Age-specific Mortality Rates

All EA and PF models require estimates of age-specific survival fractions ( $S_i$ ). Equivalent yield models require, in addition, estimates of the annual rate of fishing mortality ( $F_i$ ). These parameters are related in the following way:

$$S_i = e^{-Z_i} = e^{-(F_i+M_i)} \quad (\text{Eq. 25})$$

Where:

$M_i$  = annual rate of natural mortality

Estimates of  $S_i$  and  $Z_i$  can be obtained from data on the age composition of the adult population, obtained from analysis of scales or otoliths. Published estimates are available for most harvested species and for many nonharvested species as well. Some care must be taken in using values obtained from the literature, because mortality rates in all fish populations are affected by a variety of ecological and climatic factors. Because total mortality includes fishing mortality, total mortality rates will be higher for harvested than for unharvested populations of the same species, even if other factors are similar. If literature-derived rather than population-specific estimates must be used, preference should be given to data from a similar population, with a similar age structure, in the same region, with a similar harvesting pattern.

As noted above, equivalent yield models require separate estimates of natural, fishing, and total mortality. Resource management agencies develop these estimates from long-term data on total landings and on the age-composition of the harvest. Consequently, values are available for relatively few species. In the absence of data, estimates of the fraction of total mortality that is attributable to fishing must be made on the basis of expert judgment.

### 5.2.2 Weights and Growth Rates

PF and equivalent yield models require estimates of age-specific average weights and growth rates. Since fish grow continuously through a large fraction of the year, the date(s) on which the data are collected can have a substantial influence on both the parameter estimates and the outputs from the models. As discussed above, the Rago model calculates growth rates from the average weight of fish *at the beginning* of each age interval. The BLM, in contrast, uses an average weight *for the entire age interval*. Equivalent yield models use the average weight of fish *harvested at any given age*.

These three types of weight estimates require different sampling regimes, and data that are appropriate for one type of estimate may be inappropriate for others. For implementing the Rago model, estimating weights requires identifying a specific annual start date for each species (e.g., January 1 or the first day of the first month in which spawning typically occurs), and then calculating the average weight of fish on that date. For implementing the BLM, estimating weights requires calculating the average weight of fish over the entire year. Both kinds of estimates can be obtained from field samples collected during appropriate time intervals, however, they also may be obtained from growth rate models (e.g., the von Bertalanffy model) that provide estimates of length or weight as a function of age.

For implementing the equivalent yield model, estimating weights simply involves calculating the average weight of fish harvested, regardless of season. Estimates of age-specific weight-at-

harvest are often available from resource management agencies. If not, then annual average weights such as those used in the biomass lost model are an appropriate surrogate.

### **5.2.3 Fecundity**

Fecundity estimates are required to implement the fecundity hindcasting model documented in Section 4.1.2. Because fecundity in fish is usually a function of weight, these estimates could be based on fecundity-weight relationships derived from data on egg production per unit weight measured over a wide range of fish sizes. Data of this quality are not available for most fish species. Fecundity is not a parameter that is used in the management of most fish populations, so agencies do not routinely collect data on fecundity. Fecundity-weight relationships are readily available for species that are used in hatchery stocking programs, however, for the great majority of species of interest in 316(b) studies, the published scientific literature is the only available source of information.

Published fecundity estimates are subject to several important limitations. In many fish species, individual females spawn in batches over intervals ranging from a few days to several months. Egg counts obtained from the ovaries of any given female will miss eggs spawned prior to the date of collection and probably also eggs that would have matured after the date of collection. Moreover, not all eggs are viable and not all spawned eggs will be fertilized. The result is that published estimates of fecundity may either overestimate or underestimate actual fecundity, and published values or fecundity-weight relationships should be evaluated carefully prior to use.

## **5.3 Sources of Information for Developing Life History Parameters**

The most comprehensive single source of life history data for fish is the on-line FishBase database ([www.fishbase.org](http://www.fishbase.org)), maintained by the WorldFish Center under the sponsorship of the Food and Agriculture Organization of the United Nations, the European Commission, and various international fisheries research organizations. FishBase contains information on the life history, distribution, and population dynamics of a wide variety of marine and freshwater species. The types and quantities of information provided vary among species, with more being available for marine than for freshwater species, and more for harvested than for unharvested species. However, the intent of the FishBase project is to collect and synthesize all available information concerning all fish species. Typical entries include length-frequency and length-weight distributions, growth rates, morphometrics, spawning seasons, photographs, distribution maps, a reference list, and notes on various other topics. Sources of all data are identified both by author and by geographic location. FishBase also includes generic models that can be used to predict natural mortality, lifespan, age at maturity, and generation time as functions of maximum length, growth rate, and temperature. These models are based empirical relationships developed using data for hundreds of species. The most significant limitation of FishBase is that it currently contains very limited information concerning early life stages. A companion “LarvalBase” database has been established and can be accessed from the FishBase home page, but this database is still in an early stage of development and contains only limited information.

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FishBase is an excellent resource, but not the only resource and for many species not the best resource. Entries in FishBase are compiled from a wide variety of published literature, without formal peer review, and it can be difficult to distinguish high-quality data from questionable data. Moreover, the data and especially the parameter estimates derived from the models reflect approximations applicable to species and groups of species and are not specific to any particular population. For managed species, population-specific information obtained from management agencies is clearly preferable to generic information obtained from FishBase. For example, the stock assessments performed by interstate fisheries management commissions such as the Atlantic States Marine Fisheries Commission are based on detailed analyses of population-specific data and undergo rigorous peer reviews organized by the National Marine Fisheries Service. Life history parameters published in the assessment reports should always be used when available. Data on specific populations of interest may also be available from state agencies.

There are no convenient sources of information on fecundity or on early life stage growth rates, mortality rates, or stage durations. Only limited information on these topics is available in FishBase, and no information at all on these parameters is included in typical stock assessments. Research on early life stages is, however, funded by both NMFS and the U.S. Fish and Wildlife Service; agencies maintain on-line publication lists that can be searched for information on particular species. Permit-related reports such as 316(b) Demonstrations prepared by utility companies can also be a useful source of information, because many of these include extensive reviews of early life history information for species vulnerable to entrainment or impingement. The Salem 316(b) Demonstration (PSEG 1999), for example, included a comprehensive review of early life stage data for 13 finfish species. Similar compilations were prepared to support the Diablo Canyon assessment. EPA made extensive use of data contained in permit applications for its Section 316(b) case study (USEPA 2002). Citations to many relevant reports can be found in the reference list for EPA's study.

Regardless of the source, users of this guidance should wherever possible obtain the original documents supporting any parameter value and evaluate them for applicability to the site and population for which the assessment is being performed. A wide variety of factors relating to the design of a study (e.g., gear used, criteria used to identify species and life stages, study duration, spatial boundaries) affect the utility of any parameter estimate; simply adopting a value without investigating these factors can lead to inaccurate model projections. Specific recommended values for commonly entrained and impinged species in freshwater and marine environments will be documented in a future EPRI report. This report is currently scheduled for publication in late 2004.

# 6

## EXAMPLE CALCULATIONS FOR MARINE AND FRESHWATER SPECIES

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This section illustrates application of EA and PF models to six representative fish species inhabiting freshwater, estuarine, and marine waterbody types: yellow perch, gizzard shad, alewife, bay anchovy, striped bass, and Pacific sardine. Section 6.1 provides brief descriptions of the life history of each species, and identifies the sources of data used to develop model parameters. Section 6.2 documents the application of the models to each species. Where feasible and relevant, both forward projection (FP) and fecundity hindcasting (FH) versions of EA models, and both the Rago and BLM versions of production foregone (PF) models were applied.

### 6.1 Life History Descriptions

Parameters needed to implement the models include age-specific weights and growth rates, stage-specific mortality rates and (for the FH approach) either age-specific or average fecundities. Although all of the species used to illustrate the application of the models are commonly entrained or impinged at power plants, the availability and quality of life history data available varies substantially among species. The most data are available for intensively managed marine species such as striped bass and Pacific sardine, and for popular research species such as bay anchovy. The least data are available for gizzard shad, but even for this species the data are sufficient to apply all of the models. To illustrate the diversity of life history types that can occur even within a single species, two life history descriptions were developed for alewife: an anadromous life history, representative of the native east coast populations, and a Great Lakes life history, more suitable for application to introduced freshwater populations of this species.

#### 6.1.1 Yellow Perch

Yellow perch (*Perca flavescens*) are abundant in freshwater throughout the northern and eastern United States, including the Great Lakes, and throughout central and eastern Canada. Spawning of yellow perch occurs in the spring in shallow water, usually on or near rooted vegetation (Scott and Crossman 1973). Eggs are deposited in transparent strands attached to submerged vegetation or occasionally to the bottom. Larvae and early juveniles are planktonic, and are susceptible to entrainment. Older juveniles, and even adults, are susceptible to impingement. Females reach maturity at approximately two years of age. Yellow perch is a popular recreational species, and a commercial fishery for yellow perch occurs in the Great Lakes.

*Example Calculations for Marine and Freshwater Species*

Yellow perch are relatively small (maximum length ~ 30 cm in the Great Lakes, smaller in other water body types), and serve as prey for larger predators such as largemouth bass, walleye, and Northern pike.

Because it is both a harvested species and a forage species, both EA and PF models were applied to yellow perch. Data suitable for applying all of the models documented in Sections 4 and 5 were available from the EPA case study (USEPA 2002, Part I, Monroe Power Plant). Values for stage and age-specific natural mortality, age-specific fishing mortality, and age-specific fecundity were adopted directly from EPA's input data spreadsheet for Monroe (DCN-4-2046). EPA's spreadsheet also provides values for mean stage and age-specific weights.

Implementation of the Rago PF model, however, requires estimates of stage-specific start and end weights and stage-specific growth rates. The required values were approximated using the mean weights provided in the case study. To perform the approximation, growth rates were assumed to be constant between successive estimates of mean weights ( $\hat{W}$ ), rather than between the beginning and the end of each stage. The mean weights were assumed to be approximately equal to the median weight-at-death for organisms dying during a given stage, and the growth rates were then calculated from:

$$G'_i = \ln(\hat{W}_{i+1} / \hat{W}_i) \quad (\text{Eq. 26})$$

Where:

$\hat{G}_i$  = growth rate from the median time-to-death for stage  $i$  to the median time-to-death for stage  $i + 1$

The start weight for each stage  $i+1$  was then calculated by applying the above growth rate to the interval between the median time-to-death for stage  $i$  and the beginning of stage  $i+1$ :

$$\bar{W}_{i+1} = \hat{W}_i e^{G'_i(1-\hat{D}_i)} \quad (\text{Eq. 27})$$

Where:

$\hat{D}_i$  = median time-to-death for stage  $i$ , expressed as a fraction of the total life stage duration (See Appendix A).

Values of all life history parameters used in the yellow perch model applications are listed in Table 6-1. Mortality rate parameters in Table 6-1 and in the corresponding tables for all other species are expressed as stage-based rates rather than as daily rates. For stages or ages that are not vulnerable to fishing, the total mortality rate ( $Z_i$ ) is equal to the natural mortality rate. For stages or ages that are 100% vulnerable to fishing, the total mortality rate is equal to the sum of the natural mortality rate and the fishing mortality rate. For age classes that are only partially vulnerable to fishing (e.g., age 3 yellow perch), the total mortality rate is a weighted sum of the mortality imposed on vulnerable (natural + fishing mortality) and invulnerable (natural mortality only) fish.



**Table 6-1**  
**Life History Parameters for Yellow Perch<sup>a</sup>**

Stage or age	Natural Mortality Rate	Fishing Mortality Rate	% Vulnerable	Total Mortality Rate	Fraction Surviving	% Mature	% Female	Fecundity	Start weight (lbs.)	Growth Rate	Median Weight-at Death (lbs.)
Eggs	2.75	0	0%	2.75	0.06	0%	50%	0	2.20E-07	2.86	2.20E-07
Larvae	3.56	0	0%	3.56	0.03	0%	50%	0	1.99E-06	8.71	3.84E-06
Juveniles	2.53	0	0%	2.53	0.08	0%	50%	0	4.56E-03	0.05	2.32E-02
1	0.36	0	0%	0.36	0.70	1%	50%	17,000	2.42E-02	0.57	2.45E-02
2	0.25	0	0%	0.25	0.78	47%	50%	23,000	3.35E-02	0.82	4.35E-02
3	0.84	0.36	50%	1.02	0.36	51%	50%	27,000	6.72E-02	0.29	9.87E-02
4	0.84	0.36	100%	1.20	0.30	86%	50%	27,000	1.18E-01	0.23	1.32E-01
5	0.84	0.36	100%	1.20	0.30	100%	50%	27,000	1.52E-01	0.25	1.66E-01
6	0.84	0.36	100%	1.20	0.30	100%	50%	34,000	1.93E-01	0.00	2.14E-01

<sup>a</sup>Mortality and growth rates expressed on a life stage basis

### 6.1.2 Gizzard Shad

Gizzard shad (*Dorosma cepedianum*) are abundant throughout the eastern United States and Canada, and occur in estuarine and marine environments as well as in freshwater. Gizzard shad spawn at the surface in open water; eggs, larvae, and early juveniles are all planktonic or pelagic and are susceptible to entrainment. Older fish are susceptible to impingement. Female fish reach maturity at 1-2 years of age. Age 0 gizzard shad are important prey species for largemouth bass and other predators, however, because of their rapid growth rate and large adult size (up to 50 cm), adult gizzard shad are relatively invulnerable to predation.

A commercial fishery for gizzard shad occurs in the Great Lakes, therefore, both EA and PF models were applied to this species. Data sufficient to implement all of the models documented in Sections 4 and 5 are available from the EPA case study (USEPA 2002, Part C, Ohio River watershed; Part I, Monroe Power Plant). Because it is both a harvested species and a forage species, both EA and PF models were applied to gizzard shad. Values for stage and age-specific natural mortality and for age-specific fishing mortality, were adopted directly from EPA's input data spreadsheet for Monroe (DCN-4-2046). Age-specific fecundity data are not available from the input spreadsheet, however, an average value is provided in Section I3-2 of the case study report. As in the case of yellow perch, EPA's spreadsheet provides values for mean stage and age-specific weights rather than start and end weights. Start and end weights for gizzard shad were approximated using the methodology described above for yellow perch.

The mortality rates for adult gizzard shad imply that only 8% of fish that reach age 2, the age at which sexual maturity occurs, would survive to reach age 3. Thus, even though the longevity of gizzard shad is reported by EPA to be up to 10 years, most of the annual reproduction of this species would be provided by two-year-old fish. Hence, multiplying the average fecundity by the midpoint of the lifespan of the species (as in Section 4.1.2, equation 7b) would greatly overestimate the expected lifetime fecundity of a 2-year-old female gizzard shad. Instead, the average fecundity provided by EPA was used directly as an estimate of average lifetime fecundity, on the assumption that a typical female gizzard shad spawns only once before dying.

Values of all life history parameters used in the gizzard shad model applications are listed in Table 6-2.

**Table 6-2**  
**Life History Parameters for Gizzard Shad<sup>a</sup>**

Stage or age	Natural Mortality Rate	Fishing Mortality Rate	% Vulnerable	Total Mortality Rate	Fraction Surviving	% Mature	% Female	Fecundity	Start weight (lbs.)	Growth Rate	Median Weight-at Death (lbs.)
Eggs	2.30	0.00	0%	2.30	0.10	0%	50%		2.20E-06	1.10E+00	2.20E-06
Larvae	6.33	0.00	0%	6.33	0.00	0%	50%		4.98E-06	7.39E+00	6.63E-06
Juveniles	0.51	0.00	0%	0.51	0.60	0%	50%		4.78E-03	2.58E+00	1.07E-02
1	1.45	1.45	50%	2.18	0.11	0%	50%		4.57E-02	1.22E+00	1.41E-01
2	1.27	1.27	100%	2.54	0.08	100%	50%	378,990	3.18E-01	2.94E-01	4.77E-01
3	0.97	0.97	100%	1.93	0.14	100%	50%		5.77E-01	3.24E-01	6.40E-01
4	0.87	0.87	100%	1.75	0.17	100%	50%		7.82E-01	2.79E-01	8.85E-01
5	0.30	0.30	100%	0.61	0.55	100%	50%		1.05E+00	2.75E-01	1.17E+00
6	0.30	0.30	100%	0.61	0.55	100%	50%		1.36E+00	0.00E+00	1.54E+00

<sup>a</sup>Mortality and growth rate parameters expressed on a life stage basis.

### **6.1.3 Alewife**

Alewife occur both in the Great Lakes and marine/estuarine environments throughout northeastern North America, however, the life history of alewife in the Great Lakes, where the species was introduced in the 1950s, is distinctly different from the life history of native marine/estuarine populations. Native alewife populations are anadromous. Spawning occurs during spring in the freshwater zones of coastal rivers and ponds with outlets accessible to migrating fish. Spawning occurs near the surface in shallow water; eggs, larvae, and early juveniles are susceptible to entrainment. Older juveniles are susceptible to impingement, but only until they emigrate to the ocean (October-November of the year in which they are spawned). Alewife return to their natal estuaries to spawn upon reaching sexual maturity, at 4-5 years of age. Alewife larvae and juveniles are an important prey source for estuarine predators such as weakfish, Atlantic croaker, bluefish, and striped bass; adult alewife support commercial fisheries from North Carolina to New England.

In the Great Lakes, alewives mature earlier and at a much smaller size than do anadromous alewives. Great Lakes alewives mature at an age of 2-4 years and a weight of approximately 20 g. Adults spawn in shallow water, often near the mouths of rivers. As with anadromous alewife, eggs, larvae, and early juveniles are susceptible to entrainment. Unlike anadromous alewife, all ages of Great Lakes alewife are potentially susceptible to impingement. Alewife in the Great Lakes are an important forage species for coho salmon, lake trout, and other predators; there is no fishery for this species in the Great Lakes.

Data for applying both EA and PF models to anadromous alewife were available from reports prepared for PSEG's 1999 Salem permit application (PSEG 1999). Great Lakes-specific information was obtained from Bronte et al. (1991) and from the FISHBASE on-line database (<http://www.fishbase.org>). For Delaware Estuary alewife, values for stage and age-specific natural mortality and for age-specific fishing mortality were obtained directly from Appendix L, Tab 18 of the Salem permit application. Unlike the other species considered in this report, alewife suffer significant mortality related to post-spawning stress. Consequently, mortality in this species generally increases with age. The adult mortality estimates developed for Salem include a non-spawning component and a post-spawning component that is applied only to the sexually mature fraction of the population.

The mortality rates for Delaware Estuary eggs, larvae, and juveniles were assumed to apply to Great Lakes alewife as well. Age and stage-specific weights of Great Lakes alewife were calculated from Lake Ontario length data (Bronte et al. 1991) and from a length-weight relationship obtained from FishBase. The values for age 1 and older Lake Ontario values were assumed to represent average values for fish that grow continuously between early spring and late fall. The midpoints of the intervals between age-specific average lengths were used to define the start lengths and end lengths for each age. For example, the start-length of 2-year-old fish was defined to be the mid-point of the interval between the age-1 average length and the age-2 average length; the end-length of 2-year-old fish was defined to be the mid-point of the interval between the average age-2 length and the average age-3 length. Alewife eggs have a

diameter of about 5 mm, and early life stages of fish do not gain weight until they begin feeding during the post yolk-sac stage. Therefore, the start and end lengths of eggs and post yolk-sac larvae were set at 5 mm; the start length of the post yolk-sac stage was set at 5 mm and the end length at 20 mm, the average length of alewife larvae at the time they transform to the juvenile stage.

Weights corresponding to each start and end length were calculated using the following equation, from FishBase:

$$W = 0.0076L^{3.01} \quad (\text{Eq. 28})$$

Where:

$L$  = length (cm), and

$W$  = weight (g)

An age-specific maturity schedule for Lake Superior alewife was available from Bronte et al. (1991). Age-specific fecundity estimates were obtained from the age-specific average lengths using a length-fecundity relationship documented in Attachment C-7 of the 1999 Salem permit application (PSEG 1999). Bronte et al. (1991) found that the annual mortality of adult female alewife in Lake Superior was approximately 62% per year. This value is equivalent to an annual instantaneous mortality rate of 0.68. As with anadromous alewife, much of this mortality is likely to be post-spawning mortality. The natural mortality rate unrelated to spawning was assumed to be the same as for Delaware Estuary alewife (0.3); the remainder of the mortality (0.38) was assumed to be post-spawning mortality and was applied only to the sexually mature fraction of the population.

Values of all life history parameters used in the Delaware Estuary and Great Lakes model applications are listed, respectively, in Tables 6-3 and 6-4.

## Example Calculations for Marine and Freshwater Species

**Table 6-3**  
**Life History Parameters for Alewife – Delaware Estuary<sup>a</sup>**

Stage or age	Natural Mortality Rate	Post-Spawning Mortality Rate <sup>b</sup>	Fishing Mortality Rate	% Vulnerable	Total Mortality Rate	Fraction Surviving	% Mature	% Female	Fecundity	Start weight (g)	Growth Rate	Median Weight-at Death (g)
Eggs	0.56	0.00	0.00	0%	0.56	0.57	0%	50%	0	9.43E-04	0	9.43E-04
YSL	1.83	0.00	0.00	0%	1.83	0.16	0%	50%	0	9.43E-04	0	9.43E-04
PYSL	1.74	0.00	0.00	0%	1.74	0.18	0%	50%	0	9.43E-04	4.17	3.38E-03
Juveniles	6.26	0.00	0.00	0%	6.26	0.00	0%	50%	0	6.12E-02	4.89	1.05E-01
1	0.30	0.00	0.10	0%	0.30	0.74	0%	50%	0	8.15E+00	1.31	1.49E+01
2	0.30	0.00	0.10	0%	0.30	0.74	0%	50%	0	3.01E+01	1.01	4.80E+01
3	0.30	0.00	0.10	0%	0.30	0.74	0%	50%	0	8.23E+01	0.54	1.06E+02
4	0.30	1.20	0.10	45%	0.77	0.46	50%	50%	131,161	1.41E+02	0.32	1.64E+02
5	0.30	1.20	0.10	90%	1.59	0.20	100%	50%	165,279	1.94E+02	0.20	2.13E+02
6	0.30	1.20	0.10	100%	1.60	0.20	100%	50%	190,997	2.37E+02	0.13	2.51E+02
7	0.30	1.20	0.10	100%	1.60	0.20	100%	50%	209,484	2.69E+02	0.08	2.79E+02
8	0.30	1.20	0.10	100%	1.60	0.20	100%	50%	222,390	2.92E+02	0	2.92E+02

<sup>a</sup>Mortality and growth rate parameters expressed on a life stage basis.

<sup>b</sup>Additional mortality imposed on sexually mature fish

**Table 6-4**  
**Life History Parameters for Alewife – Great Lakes<sup>a</sup>**

Stage or age	Natural Mortality Rate	Post-Spawning Mortality Rate <sup>b</sup>	Fishing Mortality Rate	Total Mortality Rate	Fraction Surviving	% Mature	% Female	Fecundity	Start weight (g)	Growth Rate	Median Weight-at Death (g)
Eggs	0.56	0.00	0	0.56	0.57	0%	50%	0	9.43E-04	0	9.43E-04
YSL	1.83	0.00	0	1.83	0.16	0%	50%	0	9.43E-04	0	9.43E-04
PYSL	1.74	0.00	0	1.74	0.18	0%	50%	0	9.43E-04	4.17	3.38E-03
juveniles	6.26	0.00	0	6.26	0.00	0%	50%	0	6.12E-02	4.81	1.04E-01
1	0.30	0.00	0	0.30	0.74	0%	50%	0	7.49E+00	0.80	1.08E+01
2	0.30	0.67	0	0.54	0.58	44%	50%	38,904	1.66E+01	0.20	1.83E+01
3	0.30	0.67	0	0.72	0.48	71%	50%	40,637	2.03E+01	0.11	2.14E+01
4	0.30	0.67	0	0.86	0.42	88%	50%	50,130	2.28E+01	0.12	2.40E+01
5	0.30	0.67	0	0.97	0.38	100%	50%	53,258	2.56E+01	0.03	2.59E+01
6	0.30	0.67	0	0.97	0.38	100%	50%	53,258	2.63E+01	0	2.63E+01
7	0.30	0.67	0	0.97	0.38	100%	50%	53,258	2.63E+01	0	2.63E+01
8	0.30	0.67	0	0.97	0.38	100%	50%	53,258	2.63E+01	0	2.63E+01

<sup>a</sup>Mortality and growth rate parameters expressed on a life stage basis.

<sup>b</sup>Additional mortality imposed on sexually mature fish

#### **6.1.4 Bay Anchovy**

Bay anchovy (*Anchoa mitchilli*) is among the most abundant estuarine fish species along the Atlantic and gulf coasts of the United States and Mexico. Bay anchovy spawn over an extended period from late spring through the end of summer. A single female fish can spawn up to 50 times over this period. Spawning occurs at night in open water, over a wide range of salinities. Eggs, larvae, and juveniles are planktonic and are susceptible to entrainment. Older fish are susceptible to impingement. All ages of bay anchovy are important as forage for predator species. There is no fishery for this species.

Data suitable for applying all models were available from documents prepared to support PSEG's 1999 permit application for Salem (PSEG 1999). Mortality rates for all stages and ages of bay anchovy are provided in Appendix L, Tab 18 of the application. Stage-specific start/end lengths and length-weight relationships suitable for applying the Rago PF model are also available from Appendix L, Tab 11 of the application.

Values of all life history parameters used in the bay anchovy model applications are listed in Table 6-5.

#### **6.1.5 Striped Bass**

Striped bass (*Morone saxatilis*) is among the most important recreational and commercial fish species inhabiting Atlantic and gulf coastal waters. Along the Atlantic coast, striped bass historically spawned in estuaries from northern Florida to Nova Scotia. Major spawning populations are currently limited to the Roanoke River, Chesapeake Bay, Delaware River, and Hudson River. Striped bass were introduced into San Francisco Bay in the 19<sup>th</sup> century, and the San Francisco Bay/Delta system also supports a large striped bass population. Striped bass spawn during spring in tidal freshwater. Spawning occurs in open water, and all early life stages are planktonic. Eggs, larvae, and early juveniles, therefore, are susceptible to entrainment. Striped bass are vulnerable to impingement from the late juvenile stage until approximately age 2. Striped bass are fast-growing predators that become sexually mature at age 5-7, and can grow to a weight of 25 kg or more.

Because striped bass is not a significant forage species, PF models were not applied. Data suitable for applying the two EA models were available from documents prepared to support PSEG's 1999 permit application for Salem. Mortality rates for all stages and ages of striped bass are provided in Appendix L, Tab 18 of the application.

Values of all life history parameters used in the striped bass model applications are listed in Table 6-6.



**Table 6-5**  
**Life History Parameters for Bay Anchovy<sup>a</sup>.**

Stage or age	Natural Mortality Rate	Fishing Mortality Rate	Total Mortality Rate	Fraction Surviving	% Mature	% Female	Fecundity	Start weight (g)	Growth Rate	Median Weight-at Death (g)
Egg	1.04	0	1.04	0.352	0%	50%	0	7.83E-07	2.37	1.91E-06
Prolarvae	1.57	0	1.57	0.209	0%	50%	0	8.42E-06	1.61	1.41E-05
Postlarvae 1	2.11	0	2.11	0.122	0%	50%	0	4.21E-05	3.90	1.23E-04
Postlarvae 2	4.02	0	4.02	0.018	0%	50%	0	2.09E-03	2.37	3.11E-03
Juvenile 1 (20-30)	0.08	0	0.08	0.921	0%	50%	0	2.25E-02	1.39	4.43E-02
Juvenile 2 (30-40)	0.09	0	0.09	0.917	0%	50%	0	9.01E-02	0.99	0.15
Juvenile 3 (40-50)	0.13	0	0.13	0.879	0%	50%	0	2.41E-01	0.76	0.35
Juvenile 4 (50-60)	0.99	0	0.99	0.370	0%	50%	0	5.18E-01	0.57	0.64
1	1.62	0	1.62	0.197	100%	50%	38,206	9.14E-01	0.58	1.10
2	1.62	0	1.62	0.197	100%	50%	38,206	1.63E+00	0.13	1.69
3	1.62	0	1.62	0.197	100%	50%	38,206	1.85E+00	0.00	1.85

<sup>a</sup>Mortality and growth rate parameters expressed on a life stage basis

## Example Calculations for Marine and Freshwater Species

**Table 6-6**  
**Baseline Life History Parameters for Striped Bass<sup>a</sup>**

Stage or age	Natural Mortality Rate	Fishing Mortality Rate	% Vulnerable	Total Mortality Rate	Fraction Surviving	% Mature	% Female	Fecundity	Start weight (g)	Growth Rate	Median Weight at Death (g)
Eggs	1.38	0	0%	1.38	0.25	0%	50%	0	1.02E-04	0.00E+00	1.02E-04
YSL	2.21	0	0%	2.21	0.11	0%	50%	0	1.02E-04	3.97E+00	2.93E-04
PYSL	5.08	0	0%	5.08	0.01	0%	50%	0	5.38E-03	2.56E+00	7.62E-03
Juv 1	2.27	0	0%	2.27	0.10	0%	50%	0	6.99E-02	4.94E+00	2.55E-01
Juv 2	1.00	0	0%	1.00	0.37	0%	50%	0	9.81E+00	2.14E+00	2.22E+01
1	1.10	0.31	0%	1.10	0.33	0%	50%	0	8.38E+01	1.51E+00	1.46E+02
2	0.15	0.31	6%	0.17	0.85	0%	50%	0	3.78E+02	1.02E+00	6.16E+02
3	0.15	0.31	20%	0.20	0.81	0%	50%	0	1.05E+03	4.30E-01	1.29E+03
4	0.15	0.31	63%	0.33	0.72	4%	50%	400,962	1.61E+03	3.70E-01	1.92E+03
5	0.15	0.31	94%	0.44	0.65	13%	50%	546,162	2.32E+03	3.14E-01	2.70E+03
6	0.15	0.31	100%	0.46	0.63	45%	50%	742,201	3.18E+03	2.92E-01	3.66E+03
7	0.15	0.31	100%	0.46	0.63	89%	50%	1,098,521	4.26E+03	2.39E-01	4.78E+03
8	0.15	0.31	100%	0.46	0.63	94%	50%	1,296,779	5.41E+03	1.20E-01	5.73E+03
9	0.15	0.31	100%	0.46	0.63	100%	50%	1,470,715	6.10E+03	1.43E-01	6.54E+03
10	0.15	0.31	100%	0.46	0.63	100%	50%	1,742,843	7.04E+03	1.72E-01	7.65E+03
11	0.15	0.31	100%	0.46	0.63	100%	50%	1,817,953	8.36E+03	2.00E-01	9.20E+03
12	0.15	0.31	100%	0.46	0.63	100%	50%	2,106,938	1.02E+04	9.27E-02	1.07E+04
13	0.15	0.31	100%	0.46	0.63	100%	50%	2,570,847	1.12E+04	6.85E-02	1.16E+04
14	0.15	0.31	100%	0.46	0.63	100%	50%	3,356,147	1.20E+04	3.22E-01	1.40E+04
15 +	0.15	0.31	100%	0.46	0.63	100%	50%	4,087,464	1.65E+04	0.00E+00	1.65E+04

<sup>a</sup>Mortality and growth rate parameters expressed on a life stage basis

### 6.1.6 Pacific Sardine

The Pacific sardine (*Sardinops sagax*) is an offshore species that is abundant throughout the northeastern Pacific Ocean. Along the California coast, spawning occurs from January through June. Spawning occurs in open water. Eggs and larvae are pelagic and are susceptible to entrainment. Although juveniles of this species are also pelagic, they occur offshore and have limited contact with intake structures. Pacific sardines grow to a maximum length of approximately 30 cm and a maximum weight of about 200 g. Sexual maturity occurs at approximately age 2.

Pacific sardine is an important forage species and also supports a substantial commercial fishery. Therefore, both EA and PF models were applied to this species. Data needed to implement the models was obtained from the 316(b) Demonstration for the Diablo Canyon Power Plant (Tenera 2000) and from FishBase. Estimates of percent maturity, age-specific fecundity and survivorship, early life stage mortality, and fishing mortality were available from the 316(b) Demonstration (Tables 5.2.2-1, 5.2.2-2, and 5.2.2-3). Weights of Pacific sardine eggs and early juveniles were obtained from FishBase. These values were used to calculate start weights for the various larval and juvenile substages identified in the 316(b) demonstration, assuming a constant growth rate within each stage. Age-weight relationships for age 1 and older fish were also available from FishBase.

Values of all life history parameters used in the Pacific sardine model applications are listed in Table 6-7.

## 6.2 Species-Specific Model Applications

For each species, model projections were performed using the life history parameters documented above, together with hypothetical values of entrainment and impingement losses. The loss values used are arbitrary and are not intended to reflect actual losses at any specific station, however, the relative magnitude of the losses at different ages are intended to be typical of the pattern that would be observed at most stations. In addition to these “baseline” model projections, additional projections for each species were performed to evaluate the potential influence of uncertainties concerning stage-specific mortality and growth rates. Additional model projections were performed using five alternative parameter sets: 20% increase in stage-specific natural mortality for eggs, larvae, and juveniles; 20% decrease in stage-specific natural mortality for the same life stages; 20% increase in mean weight-at-death for eggs, larvae, and juveniles; 20% decrease in mean weight-at-death for the same life stages, and adjustment of egg, larval, and juvenile mortality rates to produce a perfectly balanced life table with a zero net growth rate.

*Example Calculations for Marine and Freshwater Species*

**Table 6-7**  
**Baseline Life History Parameters For Pacific Sardine<sup>a</sup>**

Stage or age	Natural Mortality Rate	Fishing Mortality Rate	% Vulnerable	Total Mortality Rate	Fraction Surviving	% Mature	% Female	Fecundity	Start weight (g)	Growth Rate	Median Weight-at Death (g)
Eggs	2.16	0	0%	2.16	0.12	0%	50%	0	6.00E-05	0.00E+00	6.00E-05
YSL	2.01	0	0%	2.01	0.13	0%	50%	0	6.00E-05	0.00E+00	6.00E-05
Early larvae	2.66	0	0%	2.66	0.07	0%	50%	0	6.00E-05	1.66E+00	8.86E-05
Late larvae	3.37	0	0%	3.37	0.03	0%	50%	0	3.15E-04	5.27E+00	8.82E-04
Early juv	1.40	0	0%	1.40	0.25	0%	50%	0	6.14E-02	4.56E-01	7.16E-02
Juv 1	0.99	0	0%	0.99	0.37	0%	50%	0	9.67E-02	9.11E-01	1.37E-01
Juv 2	0.44	0	100%	0.44	0.64	0%	50%	0	2.41E-01	2.00E+00	5.88E-01
Juv 3	0.36	0	100%	0.36	0.70	0%	50%	0	1.79E+00	2.33E+00	5.17E+00
1	0.40	0.2	100%	0.60	0.55	0%	50%	0	1.84E+01	1.08E+00	2.99E+01
2	0.40	0.4	100%	0.80	0.45	100%	50%	146,754	5.40E+01	5.37E-01	6.87E+01
3	0.40	0.5	100%	0.90	0.41	100%	50%	388,188	9.23E+01	3.04E-01	1.06E+02
4	0.40	0.5	100%	0.90	0.41	100%	50%	599,640	1.25E+02	1.82E-01	1.36E+02
5	0.40	0.5	100%	0.90	0.41	100%	50%	849,490	1.50E+02	1.13E-01	1.58E+02
6	0.40	0.5	100%	0.90	0.41	100%	50%	1,167,457	1.68E+02	7.13E-02	1.74E+02
7	0.40	0.5	100%	0.90	0.41	100%	50%	1,487,528	1.80E+02	4.55E-02	1.84E+02
8	0.40	0.5	100%	0.90	0.41	100%	50%	1,617,450	1.89E+02	2.92E-02	1.91E+02
9	0.40	0.5	100%	0.90	0.41	100%	50%	1,887,025	1.94E+02	1.88E-02	1.96E+02
10+	0.40	0.5	100%	0.90	0.41	100%	50%	2,156,600	1.98E+02	1.22E-02	1.99E+02

<sup>a</sup>Mortality and growth rate parameters expressed on a life stage basis

### 6.2.1 Yellow Perch

The four models documented in Section 4 of this report were applied to yellow perch, using the life history parameters documented in Table 6-1 and assuming a hypothetical entrainment and impingement loss scenario. Because yellow perch are reported to become sexually mature at age 1, this age was chosen as the age of equivalence for applying the two EA models. Because yellow perch eggs are not planktonic, eggs were assumed to be invulnerable to entrainment. Older life stages were assumed to be vulnerable to entrainment or impingement. Table 6-8 presents the results of the model applications, including breakdowns by life stage and totals over all life stages.

The results in Table 6-8 illustrate a number of properties of the models, and also highlight some significant differences between models. As expected, the contribution of larvae to both prey biomass loss and equivalent adult estimates is small regardless of which model is used, in spite of the fact that larvae comprise more than 80% of the total losses. The reason for this is the very high natural mortality rate of larvae. As can be seen from Table 6-1, only 3% of larvae would be expected to survive to the juvenile stage, even if there were no entrainment losses. Juvenile and older fish, which together account for less than 20% of the losses, account for 90% or more of the prey biomass and equivalent adult losses.

Differences between the models were surprisingly large, given that the same life history parameters and loss scenario were applied to every model. As discussed in Section 4, the BLM and the Rago model outputs are only comparable if the biomass-at-death of entrained and impinged organisms is added to the production foregone estimate from the Rago model. Even including biomass-at-death, however, the predicted prey biomass loss for the Rago model is about 16% lower than the predicted loss for the BLM model. For most life stages, the biomass of the entrained and impinged organisms at death is larger than the production foregone as estimated using the Rago model.

Differences between the two equivalent adult models are even more surprising. The age-1 equivalent loss estimate from the FP model is more than 35% higher than the estimate from the FH model. As noted in Section 4 and demonstrated in Appendix B, these two models provide identical projections only for populations in which reproduction and mortality are perfectly balanced, so that there is no net rate of population growth or decline. Clearly, the yellow perch life table used in this example is not balanced. In fact, the values in Table 6-1 imply a population growth rate of approximately a factor of 1.6 per generation.

Example Calculations for Marine and Freshwater Species

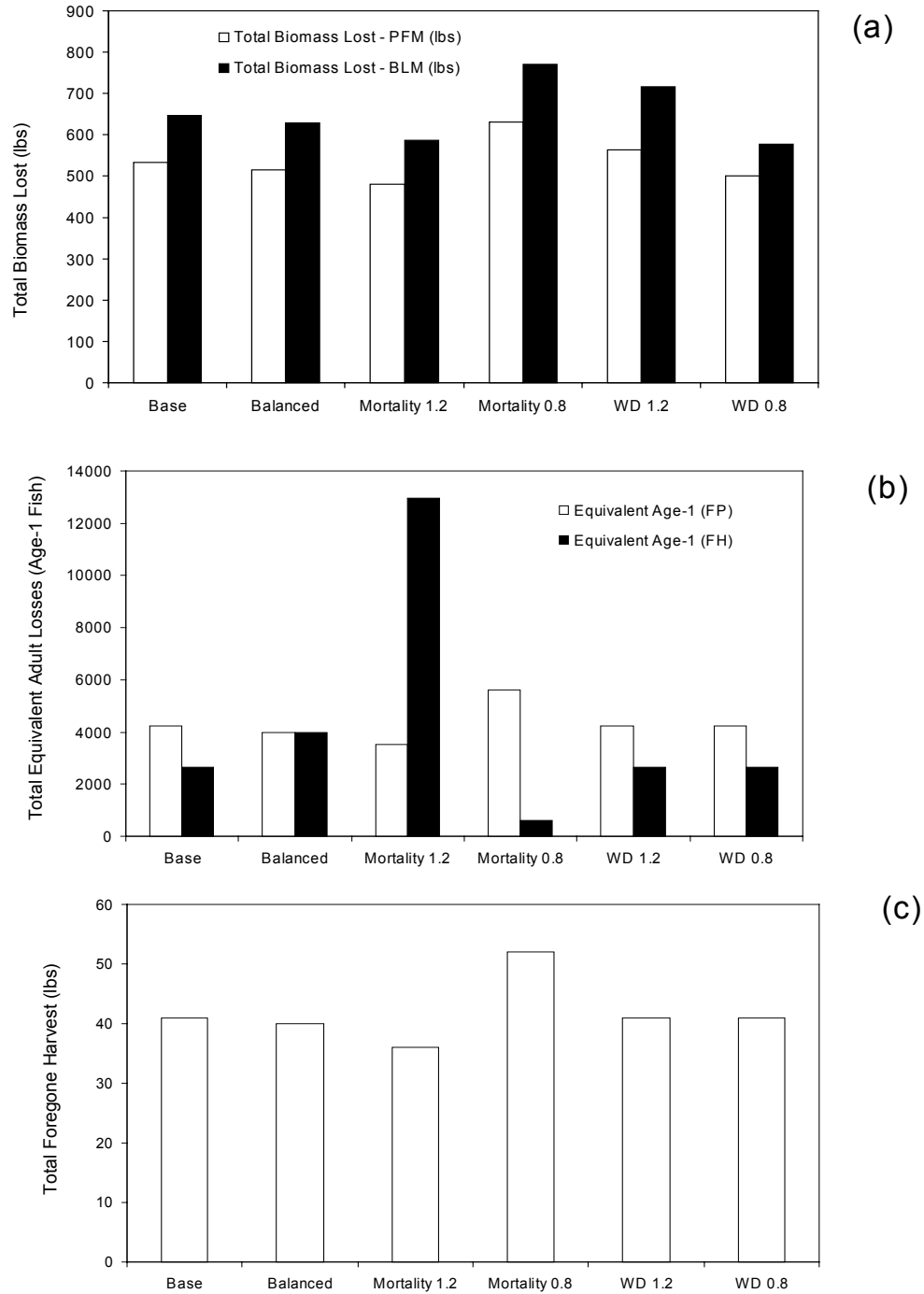
**Table 6-8**  
**Baseline Model Projections for Yellow Perch**

Stage or age	Number Lost	Production Foregone (lbs)	Biomass at Death (lbs)	Total Biomass Lost (PFM)	Total Biomass Lost (BLM)	Equiv. Adults (age 1) - FP	Equiv. Adults (Age 1) - FH	Harvest Foregone (lbs)
Eggs	0	0	0	0	0	0	0	0
Larvae	50,000	37	0.19	37	76	222	139	2
Juveniles	10,000	102	232	334	403	1482	932	11
1	1,000	67	25	91	87	1179	741	14
2	500	24	22	46	54	806	507	10
3	200	6	20	25	28	541	341	5
Total		235	298	534	648	4229	2,660	41

Figure 6-1 shows the results of the sensitivity analysis. Figure 6-1(a) shows that, for all parameter sets, the total biomass lost estimates, summed over all life stages, are higher for the BLM than for the Rago model. The relative difference between the models is, however, approximately the same for all parameter sets. Reductions in early life stage mortality and increases in weight-at-death increase the estimates of total biomass lost according to both models; conversely, increases in mortality and decreases in weight-at-death decrease the estimates of total biomass lost. The model projections for the balanced life cycle are essentially identical to the projections for the baseline parameter set.

Figure 6-1(b) shows that the two equivalent adult models respond in very different ways to changes in early life stage mortality parameters. Increased mortality greatly inflates the equivalent age-1 estimates from the FH model relative to the baseline parameter set; decreased mortality greatly reduces these estimates. The projected age-1 equivalent losses according to this model are more than 20 times as high for the increased mortality parameter set as compared to the decreased mortality parameter set. The FP model shows the opposite pattern of responses, but is much less sensitive to variations in mortality. The projected age-1 equivalent losses for this model are approximately twice as high for the low mortality parameter set as compared to the high mortality parameter set. The two models, as expected, produce identical age-1 equivalent estimates for the balanced life cycle parameter set. For both models, the projections for the high weight-at-death and low weight-at-death parameter sets are identical to the baseline because weights and growth rates do not affect either model.

Figure 6-1(c) shows the estimates of foregone harvest for each parameter set. Because the foregone harvest model is an extension of the FP version of the equivalent adult model, the projections from this model have the same relative sensitivity to parameter uncertainties as the FP model.



**Figure 6-1**  
**Results of model sensitivity analysis for yellow perch: (a) production foregone models, (b) equivalent adult models, (c) foregone harvest. Definitions of cases: Base: original parameter set; Balanced: base case early life stage (ELS) mortality rates adjusted to achieve a stable population; Mortality 1.2: base case ELS mortality rates multiplied by 1.2; Mortality 0.8: base case ELS mortality rates multiplied by 0.8; WD 1.2: base case ELS weight-at-death estimates multiplied by 1.2; WD 0.8: base case ELS weight-at-death estimates multiplied by 0.8.**

## 6.2.2 Gizzard Shad

The four models documented in Section 4 were applied to gizzard shad, using the life history parameters documented in Table 6-2 and assuming a hypothetical entrainment and impingement loss scenario. Because gizzard shad are reported to become sexually mature at age 2, this age was chosen as the age of equivalence for applying the two EA models. Because gizzard shad eggs are planktonic, all early life stages of this species were assumed to be vulnerable to entrainment. Older life stages were assumed to be vulnerable to entrainment or impingement. Table 6-9 presents the results of the baseline model applications, including breakdowns by life stage and totals over all life stages.

As in the case of yellow perch, the contributions of eggs and larvae to both prey biomass and equivalent adult losses are small compared to the contributions of juvenile and older fish, even though eggs and larvae comprise nearly 95% of the total entrainment and impingement losses. Also similar to yellow perch, the BLM model predicts somewhat higher reductions in total prey biomass than the Rago model, although in the case of gizzard shad the difference between the two model projections is less than 10%. The age-2 equivalent loss estimate from the FP model is approximately 40% higher than the estimate from the FH model.

**Table 6-9**  
**Baseline Model Projections for Gizzard Shad**

Stage or age	Number Lost	Production Foregone	Biomass at Death (lbs)	Total Biomass Lost (lbs) - PFM	Total Biomass Lost (lbs) - BLM	Equiv. Adults (age 2) - FP	Equiv. Adults (Age 2) - FH	Harvest Foregone (lbs)
Eggs	100,000	7	0.2	7	4	2	0.5	0.2
Larvae	100,000	65	1	66	41	24	5	2
Juveniles	10,000	1180	107	1287	1390	852	185	89
1	1,000	98	141	239	263	204	44	15
2	500	33	239	272	348	927	201	65
3	200	25	128	153	186	349	960	332
Total		1,409	615	2,024	2,231	2,359	1,395	503

Figure 6-2 shows the results of the sensitivity analysis. Because age-specific fecundity estimates were not available for this species, no “balanced life cycle” projections could be performed. Figure 6-2(a) shows that, similar to yellow perch, the total biomass lost estimates for gizzard shad, summed over all life stages, are in all cases higher for the BLM than for the Rago model. The relative difference between the models is, however, approximately the same for all parameter sets. Reductions in early life stage mortality and increases in weight-at-death increase the estimates of total biomass lost according to both models; conversely, increases in mortality and decreases in weight-at-death decrease the estimates of total biomass lost.



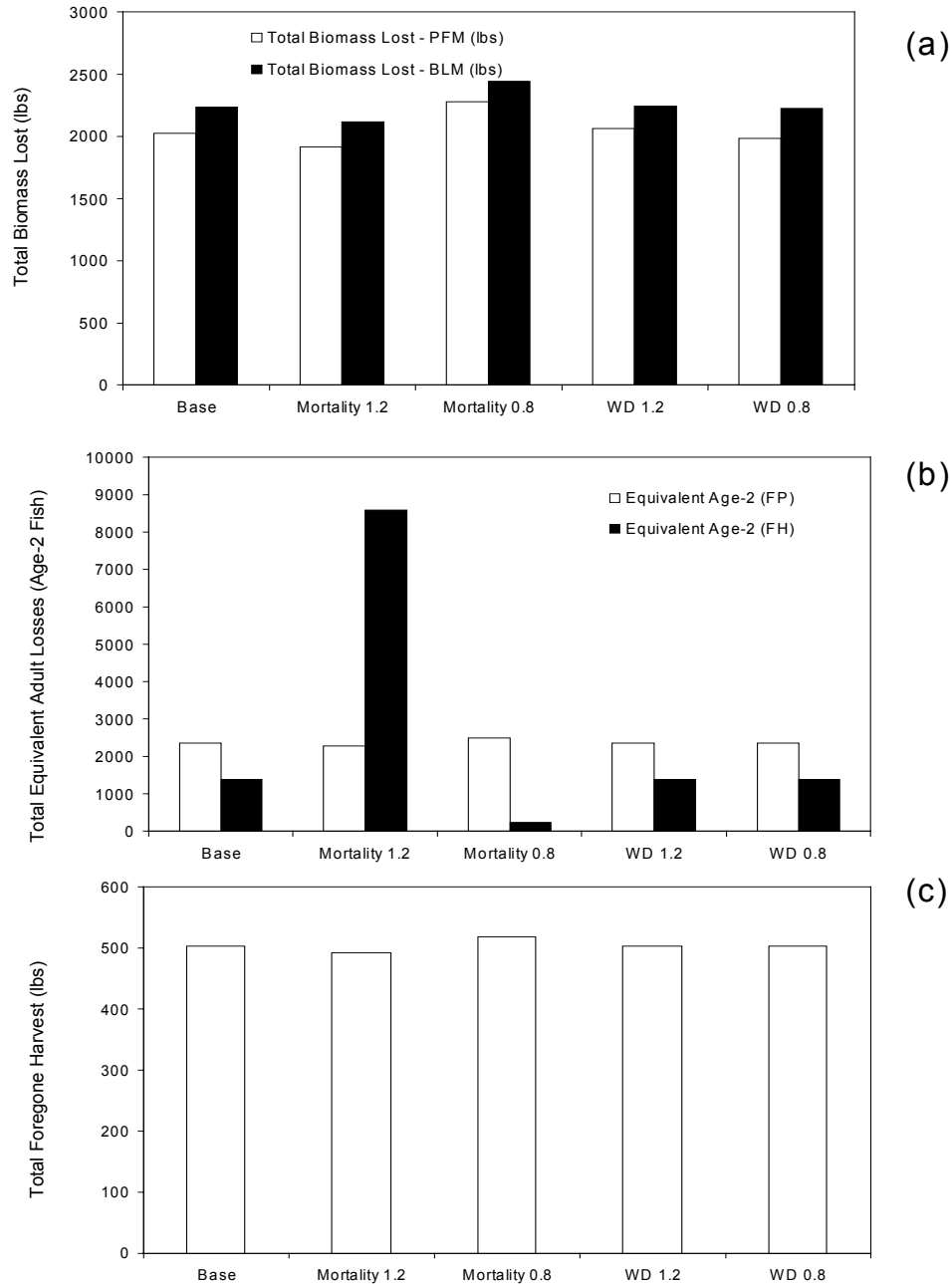
Figure 6-2(b) shows that, as in the case of yellow perch, the FH model is far more sensitive to changes in early life stage mortality parameters than is the FP model. The projected age-2 equivalent losses for the high mortality parameter set are nearly 40 times higher than the projected losses for the low mortality parameter set. The high and low mortality projections for the FP model, in contrast, are nearly equal. Because the age-2 equivalent loss estimates for the FP model projections are essentially identical for all parameter sets, the estimates of harvest foregone (Figure 6-1(c)) are also essentially identical.

### 6.2.3 Alewife

All four models documented in Section 4 were applied to Delaware Estuary and Great Lakes alewife, using the life history parameters documented, respectively, in Tables 6-3 and 6-4. Because anadromous alewife emigrate from their natal estuaries during the fall of their first year of life, only eggs, larvae, and juveniles were assumed to be vulnerable to entrainment and impingement. Alewife eggs are planktonic, therefore, all early life stages were assumed to be vulnerable to entrainment. Because spawning alewife are sometimes attracted to cooling water discharge canals, it was assumed that eggs could be entrained in relatively large numbers. Because Great Lakes alewife do not emigrate, the baseline projections for Great Lakes alewife assumed that age 1 and age 2 fish are impinged. Because anadromous alewives become sexually mature beginning at age 4, this age was chosen as the age of equivalence for applying the two EA models. Because alewife in the Great Lakes are reported to become mature at age 2, age 2 was chosen as the age of equivalence for Great Lakes alewife.

Tables 6-10 and 6-11 present the results of the baseline model applications, including breakdowns by life stage and totals over all life stages. As with the other species modeled, the contributions of eggs and larvae to prey biomass and equivalent adult losses are small compared to the contributions of juvenile and older fish. However, unlike the other modeled species, for Delaware Estuary alewife the Rago model predicts higher reductions in total prey biomass than does the BLM. This difference in model behavior is explained by the fact that all of the losses in the Delaware Estuary scenario consist of eggs, larvae, and juveniles. The biomasses of individual organisms at these stages, which determine the BLM projections, are relatively small. Conversely, the growth rates of these life stages, which determine the Rago model projections, are relatively high. In the case of Great Lakes alewife, most of the prey biomass lost is due to losses of age 1 and age 2 fish, which have relatively high individual biomasses and low growth rates. For this population, the prey biomass loss estimates are higher for the BLM than for the Rago model. The two equivalent age-4 estimates for the Delaware Estuary alewife population are nearly identical, reflecting the fact that the alewife life history parameters used in the 1999 Salem permit application were adjusted to achieve an approximately balanced life cycle (PSEG 1999, Tab 18). The baseline equivalent age-2 estimates for the Great Lakes alewife life history differ by nearly 50%, with the FH model producing the larger of the two estimates.

Example Calculations for Marine and Freshwater Species



**Figure 6-2**

**Results of model sensitivity analysis for gizzard shad: (a) production foregone models, (b) equivalent adult models, (c) foregone harvest. Definitions of cases: Base: original parameter set; Balanced: base case early life stage (ELS) mortality rates adjusted to achieve a stable population; Mortality 1.2: base case ELS mortality rates multiplied by 1.2; Mortality 0.8: base case ELS mortality rates multiplied by 0.8; WD 1.2: base case ELS weight-at-death estimates multiplied by 1.2; WD 0.8: base case ELS weight-at-death estimates multiplied by 0.8.**

**Table 6-10**  
**Baseline Model Projections for Delaware Estuary Alewife**

Stage or age	Number Lost	Production Foregone (kg)	Biomass at Death (kg)	Total Biomass Lost (kg) - PFM	Total Biomass Lost (kg) - BLM	Equiv. Adults (age 4) - FP	Equiv. Adults (Age 4) - FH	Harvest Foregone (kg)
Eggs	1,000,000	9	1	10	8	16	15	0.2
YSL	500,000	11	0.5	11	8	19	18	0.2
PYSL	250,000	28	1	29	24	58	56	0.6
Juveniles	100,000	53	11	63	56	155	150	1
Total		101	13	114	96	248	240	2

**Table 6-11**  
**Baseline Model Projections for Great Lakes Alewife**

Stage or age	Number Lost	Production Foregone	Biomass at Death (kg)	Total Biomass Lost (kg)- PFM	Total Biomass Lost (kg) - BLM	Equiv. Adults (age 2) -FP	Equiv. Adults (Age 2) - FH
Eggs	100,000	1	0.1	1	0.4	3	5
YSL	50,000	1	0.05	1	0.4	3	5
PYSL	25,000	1	0.1	2	1	11	16
Juveniles	10,000	2	1	3	2	28	44
1	1,000	7	11	18	20	851	1,318
2	500	2	9	11	11	632	979
Total		14	21	35	35	1,528	2,367

Figure 6-3 show the results of the sensitivity analysis for Delaware Estuary alewife. Figure 6-3(a) shows that the prey biomass loss estimates for this species are highly sensitive to variations in early life stage mortality rates. However, both models appear to be equally sensitive, with projected values for the low mortality case being about 7 times higher than the values for the high mortality case. Figure 6-3(b) shows that, unlike the other populations modeled, the FP model is more sensitive to variations in early life stage mortality than is the FH model. The equivalent adult estimate for the FP model is 17 times as high for the low mortality parameter set as for the high mortality parameter set. Because the harvest foregone estimates in Figure 6-3(c) are directly proportional to the FP-based equivalent adult estimates, the high and low estimates of harvest foregone also differ by a factor of 17.

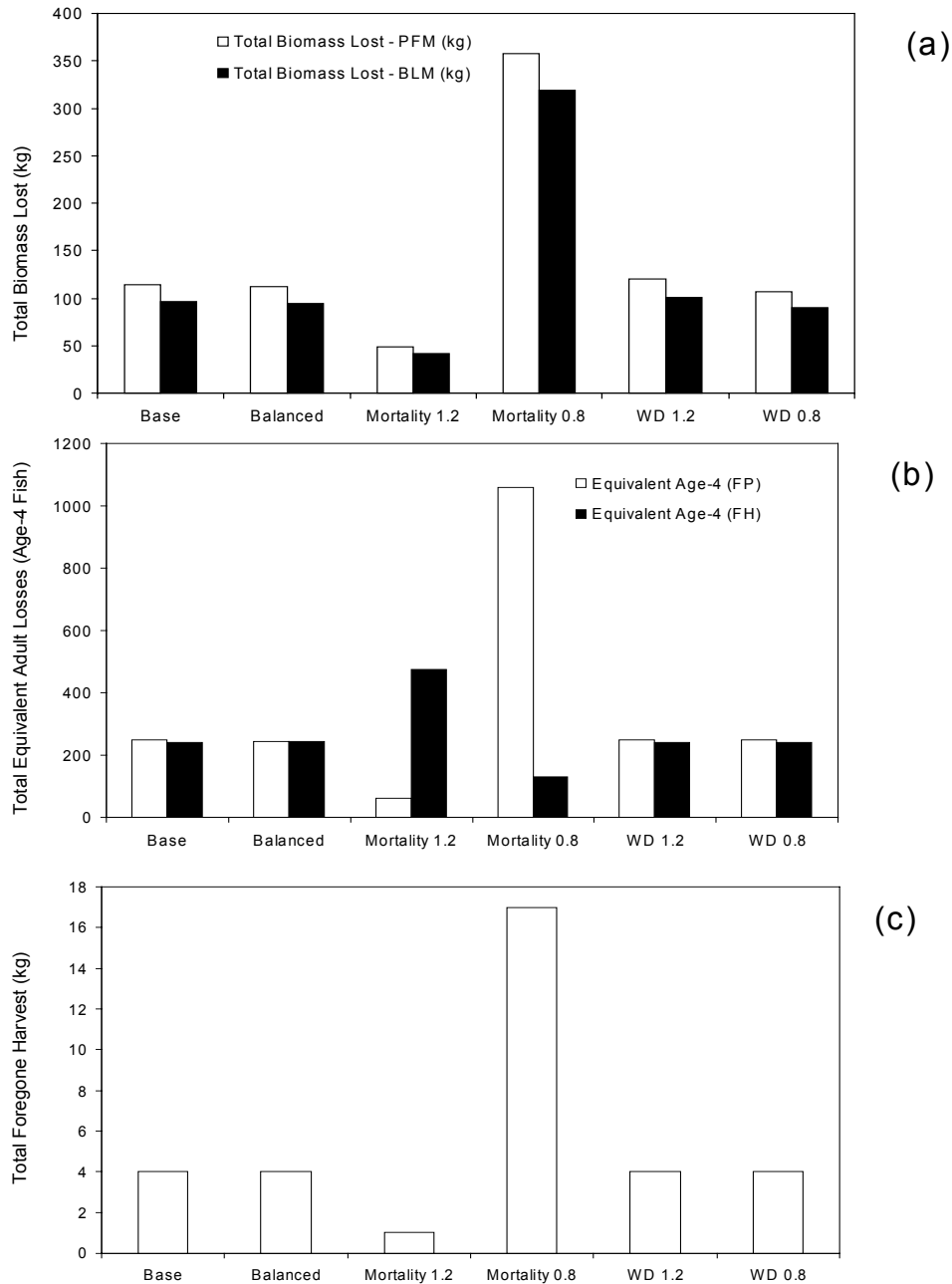
*Example Calculations for Marine and Freshwater Species*

As shown in Figure 6-4, the sensitivity analysis for Great Lakes alewife produced markedly different results. For Great Lakes alewife, variations in early life stage mortality and growth have very little influence on the prey biomass loss projections for either of the PF models. Similar to gizzard shad and yellow perch, the FH model is far more sensitive to variations in early life stage mortality than is the FP model.

Explanations for the differences in sensitivity between the models developed for these two populations are discussed in Section 6.3.

**6.2.4 Bay Anchovy**

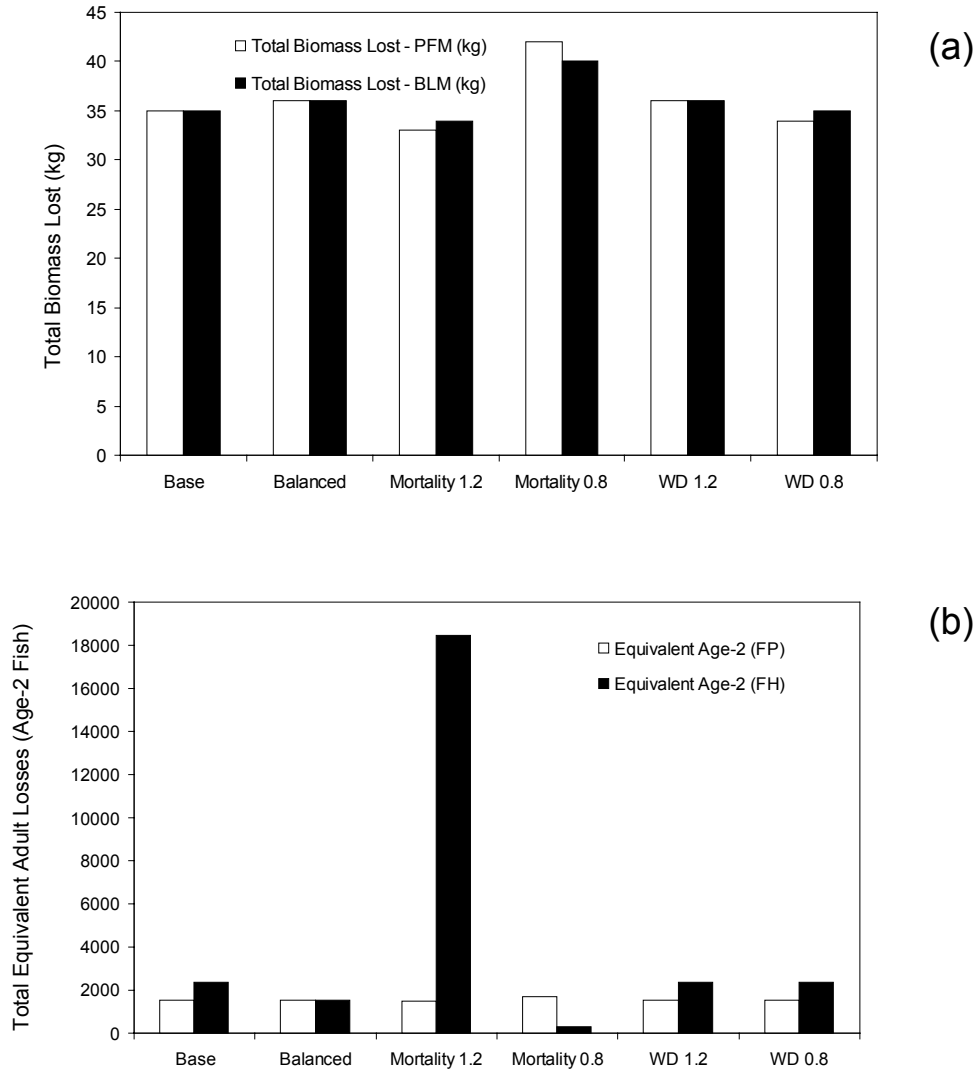
The four models documented in Section 4 were applied to bay anchovy, using the life history parameters documented in Table 6-5 and assuming a hypothetical entrainment and impingement loss scenario. All stages and ages were assumed to be vulnerable to entrainment or impingement. Because bay anchovy become sexually mature at age 1, this age was selected as the age of equivalence for EA model calculations. Table 6-12 presents the results of the baseline model applications, including breakdowns by life stage and totals over all life stages.



**Figure 6-3**

**Results of model sensitivity analysis for Delaware Estuary alewife: (a) production foregone models, (b) equivalent adult models, (c) foregone harvest. Definitions of cases: Base: original parameter set; Balanced: base case early life stage (ELS) mortality rates adjusted to achieve a stable population; Mortality 1.2: base case ELS mortality rates multiplied by 1.2; Mortality 0.8: base case ELS mortality rates multiplied by 0.8; WD 1.2: base case ELS weight-at-death estimates multiplied by 1.2; WD 0.8: base case ELS weight-at-death estimates multiplied by 0.8.**

Example Calculations for Marine and Freshwater Species



**Figure 6-4**

**Results of model sensitivity analysis for Great Lakes alewife: (a) production foregone models, (b) equivalent adult models. Definitions of cases: Base: original parameter set; Balanced: base case early life stage (ELS) mortality rates adjusted to achieve a stable population; Mortality 1.2: base case ELS mortality rates multiplied by 1.2; Mortality 0.8: base case ELS mortality rates multiplied by 0.8; WD 1.2: base case ELS weight-at-death estimates multiplied by 1.2; WD 0.8: base case ELS weight-at-death estimates multiplied by 0.8.**

**Table 6-12**  
**Baseline Model Projections for Bay Anchovy**

Stage or age	Number Lost	Production Foregone	Biomass at Death (kg)	Total Biomass Lost (kg) - PFM	Total Biomass Lost (kg) - BLM	Equiv. Adults (age 1) - FP	Equiv. Adults (Age 1) - FH
Egg	10,000,000	3	0.02	3	2	654	627
Prolarvae	1,000,000	1	0.01	1	1	208	199
Postlarvae 1	100,000	0.3	0.01	0.4	0.3	107	103
Postlarvae 2	100,000	3	0.3	3	3	970	929
Juv 1	50,000	34	2	36	36	14,301	13,704
Juv 2	20,000	12	3	15	15	6,222	5,963
Juv 3	10,000	5	3	8	8	3,461	3,316
Juv 4	5,000	2	3	5	5	2,700	2,588
1	4,000	1	4	6	6	6,683	6,404
2	1,000	0.1	2	2	2	8,478	8,125
3	1,000	0	2	2	2	43,025	41,229
Total		60	20	80	82	86,809	83,185

The results for bay anchovy are similar to results for most of the other species modeled. The contributions of eggs and larvae to both prey biomass loss and equivalent adult estimates are small compared to the contributions of juvenile and older fish, even though eggs and larvae comprise more than 99% of the total entrainment and impingement losses. The BLM predicts somewhat higher reductions in total prey biomass than the Rago model, although the difference between the two model projections is relatively small. The age-1 equivalent loss estimate from the FP and FH models are very close, reflecting the fact that early life stage mortality rates for the Salem permit application were adjusted to achieve an approximately balanced life cycle.

Figure 6-5 shows the results of the sensitivity analysis. Figure 6-5(a) shows that, similar to most other species, the total biomass lost estimates for bay anchovy, summed over all life stages, are in all cases higher for the BLM than for the Rago model. The relative difference between the models is, however, approximately the same for all parameter sets. For both models, the highest and lowest estimates of total prey biomass differ by only a factor of 1.4.

*Example Calculations for Marine and Freshwater Species*

Figure 6-5(b) shows that, as in most other species, the FH model is far more sensitive to changes in early life stage mortality parameters than is the FP model. Using the FH model, the projected age-1 equivalent losses for the high mortality parameter set case are more than 40 times higher than the projected losses for the low mortality parameter set. The high and low mortality projections for the FP model, in contrast, are nearly equal.

### 6.2.5 Striped Bass

The two EA models were applied to striped bass, using the life history parameters documented in Table 6-6 and assuming a hypothetical entrainment and impingement loss scenario. Although the median age at maturity of female striped bass is about 6 years, a small percentage of striped bass become mature at age 4. Therefore, age 4 was chosen as the age of equivalence for applying the two EA models. Because striped bass as young as 2 years old are harvested, age 1 was chosen as the age of equivalence for calculating harvest foregone. Because striped bass eggs are planktonic, all early life stages of this species were assumed to be vulnerable to entrainment. Older life stages were assumed to be vulnerable to entrainment or impingement.

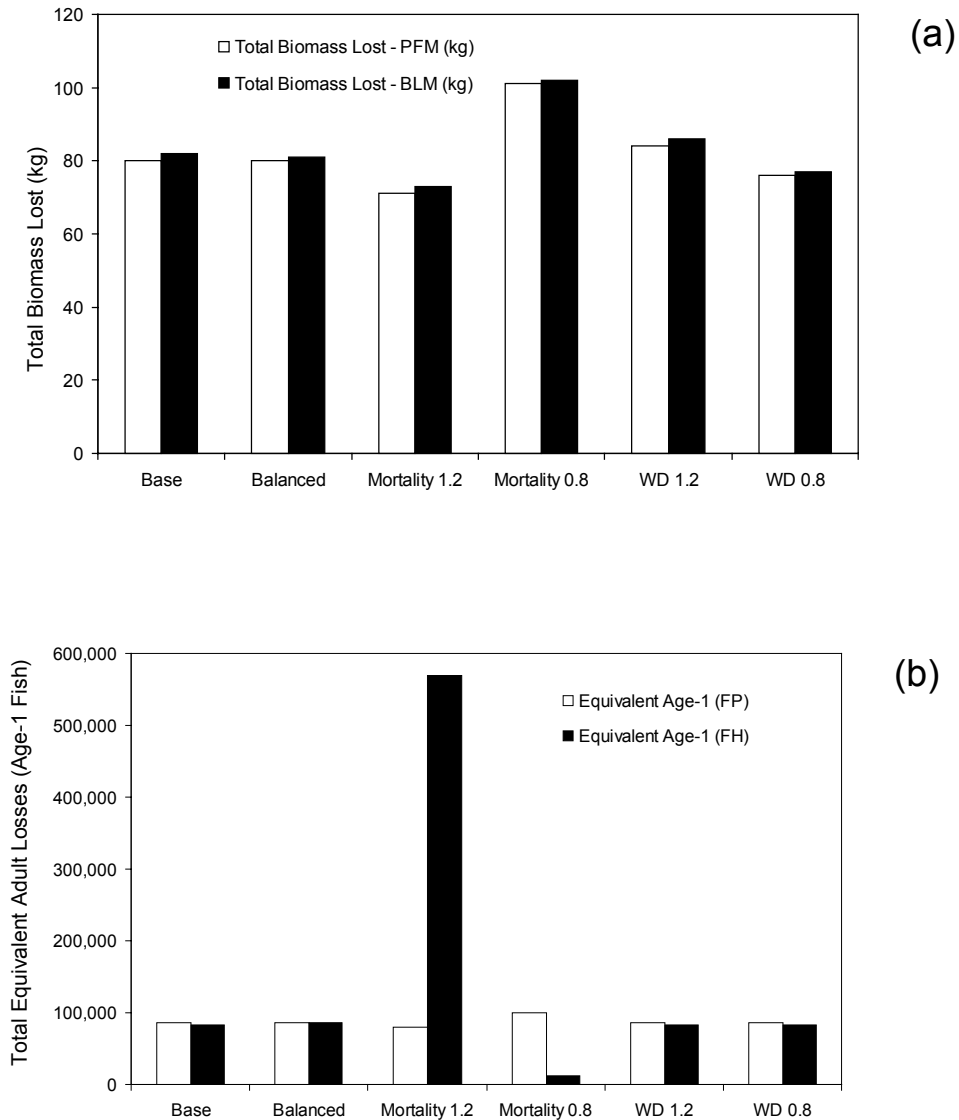
Table 6-13 presents the results of the baseline model applications, including breakdowns by life stage and totals over all life stages. Juvenile 1 and older fish account for nearly all of the equivalent adult losses and foregone harvest. As with bay anchovy and Delaware Estuary alewife, the life history parameters for striped bass that were used in the 1999 Salem permit application were adjusted to approximate a balanced population. Thus, the FP and FH projections for the baseline case are nearly identical.

Figure 6-6 shows the results of the sensitivity analysis. Figure 6-6(a) shows that, as in most other species, the FH model is far more sensitive to changes in early life stage mortality parameters than is the FP model. The projected age-4 equivalent losses for the high mortality parameter set are approximately 75 times higher than the projected losses for the low mortality case parameter set. The high and low mortality projections for the FP model, in contrast, differ by only about 50%. Because the harvest foregone estimates were derived using an extension of the FP model, the high and low values of harvest foregone also differ by about 50%.

**Table 6-13**  
**Baseline Model Projections for Striped Bass**

Stage or age	Number Lost	Equiv. Adults (age 4) -FP	Equiv. Adults (Age 4) - FH	Harvest Foregone
Eggs	100,000	0	0	0.03
YSL	50,000	1	1	0.06
PYSL	50,000	5	5	0.57
juv1	25,000	397	392	46
juv2	10,000	1243	1227	179
1	1,000	346	342	49
Total		1,992	1,967	274

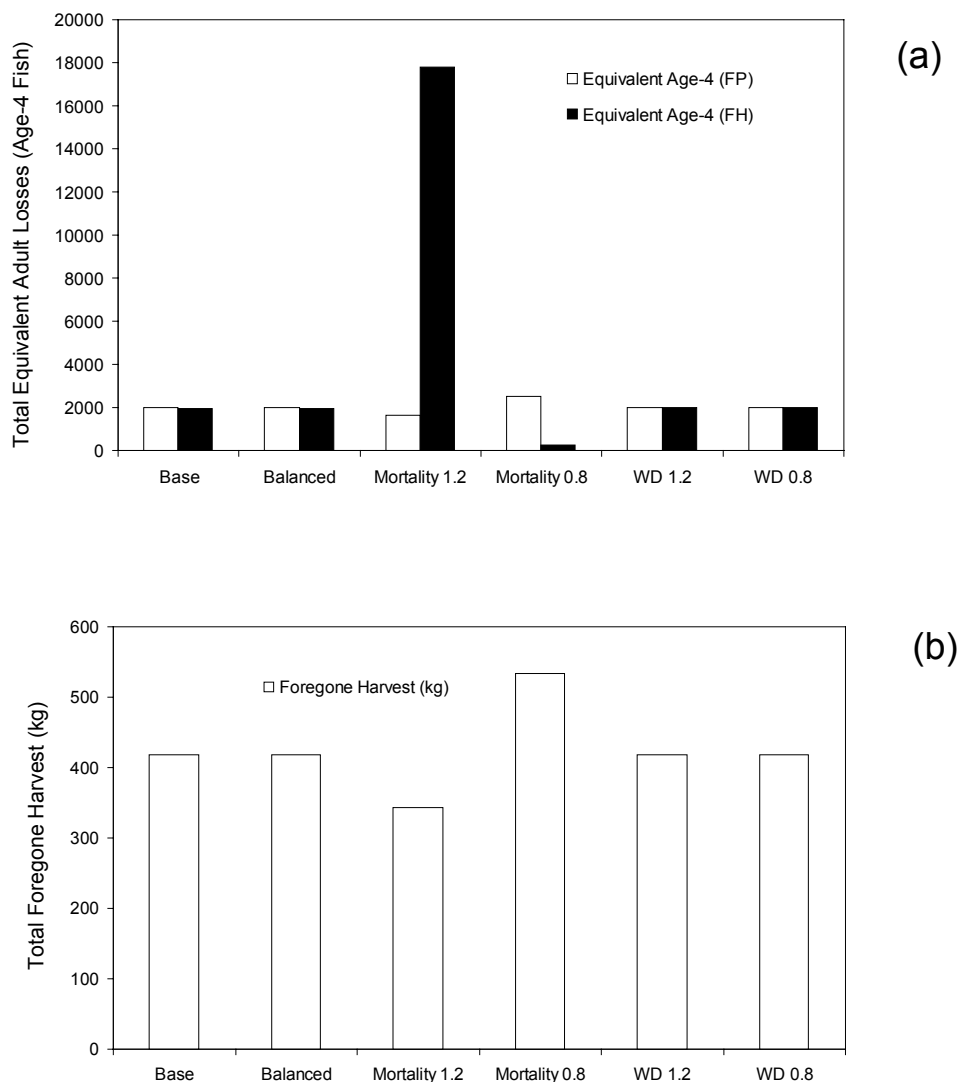




**Figure 6-5**

**Results of model sensitivity analysis for bay anchovy: (a) production foregone models, (b) equivalent adult models. Definitions of cases: Base: original parameter set; Balanced: base case early life stage (ELS) mortality rates adjusted to achieve a stable population; Mortality 1.2: base case ELS mortality rates multiplied by 1.2; Mortality 0.8: base case ELS mortality rates multiplied by 0.8; WD 1.2: base case ELS weight-at-death estimates multiplied by 1.2; WD 0.8: base case ELS weight-at-death estimates multiplied by 0.8.**

*Example Calculations for Marine and Freshwater Species*



**Figure 6-6**

**Results of model sensitivity analysis for striped bass: (a) equivalent adult models, (b) foregone harvest. Definitions of cases: Base: original parameter set; Balanced: base case early life stage (ELS) mortality rates adjusted to achieve a stable population; Mortality 1.2: base case ELS mortality rates multiplied by 1.2; Mortality 0.8: base case ELS mortality rates multiplied by 0.8; WD 1.2: base case ELS weight-at-death estimates multiplied by 1.2; WD 0.8: base case ELS weight-at-death estimates multiplied by 0.8.**

### 6.2.6 Pacific Sardine

All four of the models documented in Section 4 were applied to Pacific sardine, using the life history parameters documented in Table 6-7 and assuming a hypothetical entrainment and impingement loss scenario. Because Pacific sardine were assumed in the Diablo Canyon 316(b) Demonstration to be sexually mature at age 2, this age was chosen as the age of equivalence for applying the two EA models. Because Pacific sardine eggs are planktonic, all

early life stages of this species were assumed to be vulnerable to entrainment. Since impingement data for Diablo Canyon show that very few Pacific sardine are impinged, it was assumed that entrainment is the only significant source of station-related mortality for this species.

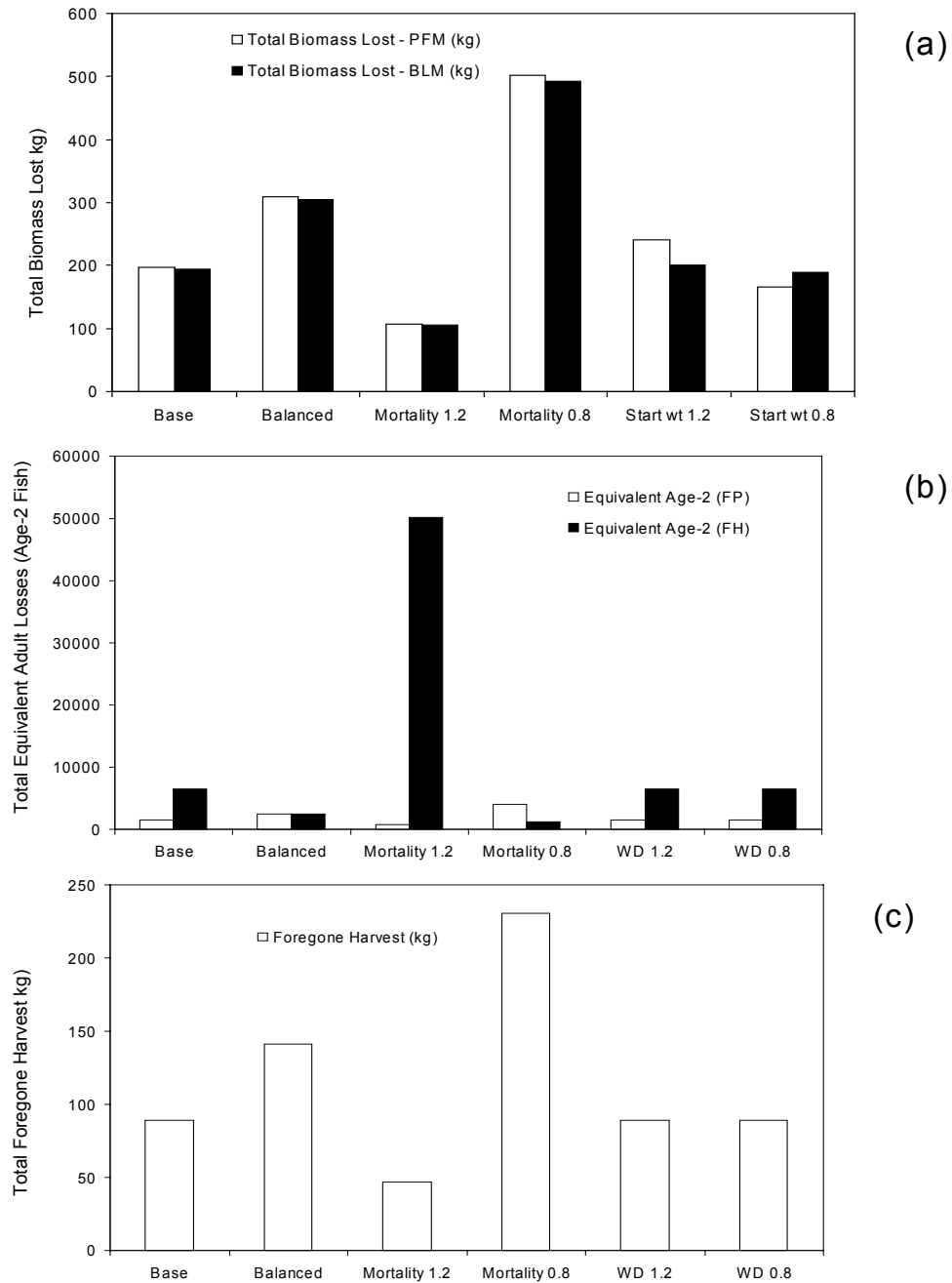
Table 6-14 presents the results of the baseline model applications, including breakdowns by life stage and totals over all life stages. Late larvae and juveniles account for approximately 90% of the total prey biomass loss according to both the Rago model and the BLM, even though these life stages account for less than 1% of the total entrainment losses. For this species, projections from the two PF models are nearly identical. The two EA models produce very different results, with the estimates derived using the FH model being four times higher than the estimates from the EA model.

Figure 6-7 shows the results of the sensitivity analysis. Figure 6-7(a) shows that, although the two PF models produce similar results for all parameter sets, both models are highly sensitive to variations in early life stage mortality rates. For both models, the estimates obtained for the low mortality parameter set are approximately 5 times higher than the estimates obtained using the high mortality parameter set. Figure 6-7(b) shows that, although both of the EA models are sensitive to variations in early life stage mortality rates, the FH model is much more sensitive than the PF model. Using the FH model, the projected age-2 equivalent losses for the high mortality parameter set are approximately 40 times higher than the projected losses for the low mortality parameter set. Using the FP model, the projected losses for the low mortality parameter set are approximately 5 times higher than for the high mortality parameter set. The harvest foregone estimates (Figure 6-7(c)) show the same result.

**Table 6-14**  
**Baseline Model Projections for Pacific Sardine**

Stage or age	Number Lost	Production Foregone (kg)	Biomass at Death (kg)	Total Biomass Lost (kg) – PFM	Total Biomass Lost (kg) - BLM	Equiv. Adults (age 2) - FP	Equiv. Adults (Age 2) - FH	Harvest Foregone (kg)
Eggs	10,000,000	2	1	3	3	15	64	0.5
YSL	5,000,000	9	0.30	9	9	64	272	2
Early larvae	500,000	7	0.04	7	7	51	215	2
Late larvae	250,000	48	0.22	48	48	376	1590	11
Early juv.	10,000	46	1	46	46	365	1540	13
Juv 1	5,000	84	1	84	83	671	2834	27
Total		195	3	198	194	1,543	6,515	55

Example Calculations for Marine and Freshwater Species



**Figure 6-7**

**Results of model sensitivity analysis for Pacific sardine: (a) production foregone models, (b) equivalent adult models, (c) foregone harvest. Definitions of cases: Base: original parameter set; Balanced: base case early life stage (ELS) mortality rates adjusted to achieve a stable population; Mortality 1.2: base case ELS mortality rates multiplied by 1.2; Mortality 0.8: base case ELS mortality rates multiplied by 0.8; WD 1.2: base case ELS weight-at-death estimates multiplied by 1.2; WD 0.8: base case ELS weight-at-death estimates multiplied by 0.8.**

### 6.3 Discussion of Model Applications

The applications discussed above illustrate the variety of data types and sources that can be used to develop parameters for EA and PF models, and also the variety of species to which the models can be applied. The life history parameters for Delaware alewife, striped bass, and bay anchovy were developed from a combination of site-specific monitoring studies performed at the Salem station, published scientific literature, and stock assessment studies performed by resource management agencies. The original data sources used to develop these values are documented in the 1999 Salem permit application (PSEG 1999, Appendix F, Tab 18). Most of the parameter values for yellow perch and gizzard shad, which were used in the EPA case study (USEPA 2002), were derived in a similar way. No published studies contained data suitable for applying PF models to Pacific sardine, however, estimates could be developed from stage-specific weights and length-weight relationships obtained from FishBase. Although far more data were available for marine species as compared to freshwater species, and for managed species as compared to unmanaged species, data sufficient to apply EA and PF models were available for all of the species evaluated.

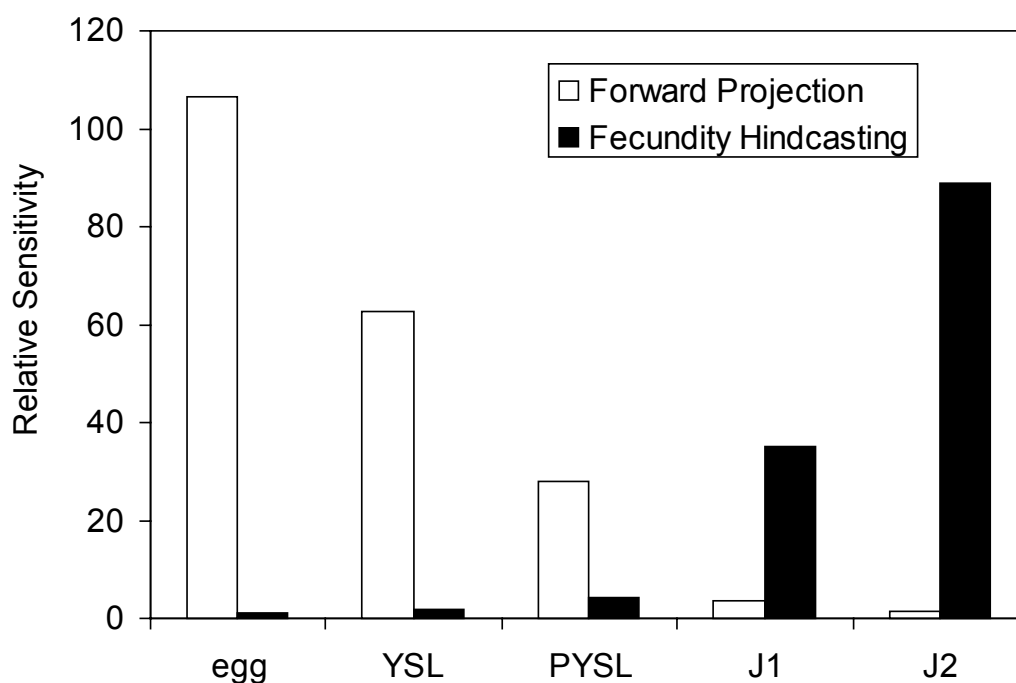
The applications also demonstrate that different models, applied using exactly the same input data, often give different results. Projections from the two PF models differed by no more than about 10% for any species, and for most species the differences were less than 5%. For all species, the two models responded in a similar manner to variations in early life stage mortality and growth parameters. Projections from the two EA models, in contrast, differed in some cases by a factor of 10 or more, and these two models responded in opposite directions to variations in early life stage parameters.

The large differences between projections from the two EA models may seem surprising, because they are derived from the same fundamental model of population dynamics. The only difference between them is that in the FP model the losses are projected forward to the age of first reproduction, and in the FH model the losses are projected backward to that same age. However, as shown in Appendix B, the two models produce identical results only in the case of a perfectly balanced population in which each newly mature female fish contributes exactly one female fish to the next generation. Increasing the rate of early life stage mortality decreases the probability that an entrained or impinged fish would have survived to reach the age of equivalence, and therefore decreases the number of equivalent adults projected using the FP model. The same increase in mortality has the opposite effect on the FH model. Increasing mortality increases the number of eggs required to produce the entrained or impinged fish, and therefore increases the number of equivalent adults projected using the FH model.

The two models also differ with respect to sensitivity to changes in mortality rates for individual life stages. Figure 6-8 plots, using striped bass base case as an example, the relative sensitivity of each life stage (eggs, yolk sac larvae, post yolk sac larvae, juvenile 1, and juvenile 2) to variations in mortality. This figure compares, for each life stage, the ratio of the highest to the lowest projected values for each model. For the FP model, this ratio is more than 100 for eggs but approximately 1 for juvenile 2 fish. For the FH model, the ratio is close to 100 for juvenile 2 fish but approximately 1 for eggs. The implication of Figure 6-8 is that the FP model is far more

*Example Calculations for Marine and Freshwater Species*

sensitive to variations in mortality rates for the earliest life stages and least sensitive to variations in mortality for the older life stages; the converse is true for the FH model. For both models, the aggregate projected number of equivalent adults, considering all entrained and impinged life stages, is a function both of the mortality rates used and of the relative numbers of fish entrained or impinged at each life stage. The pattern of entrainment and impingement across life stages explains the differences between the model projections for the Delaware and Great Lakes alewife examples. For both species, the loss scenario assumed entrainment of relatively large numbers of eggs and yolk-sac larvae. For these life stages, the FH model is much less sensitive to variations in mortality rates than is the FP model. Consequently, for the Delaware alewife example, the FP model is much more sensitive to variations in early life stage mortality than was the FH model. In addition, the Great Lakes example assumes that age 1 and older alewife would be susceptible to impingement, whereas the Delaware example assumes that age 1 and older fish are at sea and are not susceptible to impingement. For age 1 and older fish, the FP model is much less sensitive than is the FH model. Because of the inclusion of substantial numbers of older fish in the Great Lakes alewife loss scenario, the FH model is more sensitive than the FP model.



**Figure 6-8**

**Relative sensitivity of the forward projection (FP) and fecundity hindcasting (FH) models to uncertainty in natural mortality rates for different life stages of striped bass (YSL = yolk sac larvae; PYSL = post-yolk sac larvae; J1 = juvenile 1; J2 = juvenile 2). Plotted values are the ratios of the highest to the lowest model projections over a range of early life stage mortality rates.**

# 7

## CONCLUSIONS AND RECOMMENDATIONS

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It is apparent from the example calculations documented in Section 6 that, although many techniques are available for developing input parameter estimates needed to implement EA and PF models, significant care is needed both in model selection and in model parameterization to ensure that the results are credible and provide useful information for quantifying benefits associated with reductions in entrainment and impingement losses. The fact that different models sometimes produce different results does not, however, mean that any one model is inherently superior to any of the others, or that the models in general are wrong or useless.

The two PF models, i.e., the Rago and BLM models, produced similar results in nearly all cases, and were similarly sensitive to variations in early life stage mortality rates, weight-at-death estimates, and growth rates. Neither model has a clear conceptual advantage over the other, and the input data requirements are similar. It appears that the choice of which model to use would depend on the availability of data on growth rates and weight-at-age. One lesson is, however, apparent: The “classical” version of the Rago model underestimates the total biomass foregone to higher trophic levels because it includes only the future growth foregone due to entrainment and impingement. The total biomass foregone should include both the future growth and the biomass of the entrained and impinged fish at the time of death. For some species and life stages, biomass-at-death is an appreciable fraction of the total biomass foregone. If biomass-at-death is added to the production foregone calculated from the Rago model, the two models are essentially equivalent.

The two EA models, i.e., the forward projection and fecundity hindcasting models, in contrast, often give very different results and respond in opposite ways to variations in early life stage mortality rates. The FH model is especially sensitive to variations in mortality rates for juvenile and older fish; the FP model is most sensitive to variations in mortality rates for eggs and larvae. This difference in sensitivity suggests that the FH model would be more suitable for use in cases in which losses are limited to early life stages, but that the FP model would be preferable if significant numbers of juvenile and older fish are entrained or impinged.

Regardless of which model is chosen, it is apparent that all of the projected adult equivalent losses are highly sensitive to the values chosen for natural mortality rates. Published estimates of these values can vary widely, even for the same species and life stage. Great care should be taken in selecting from among the available values, and some tests are needed to determine the realism of the values.

One critical problem in using all of the models discussed in the report is that there is no independent means of verifying the accuracy of the results using empirical data. By definition,

*Conclusions and Recommendations*

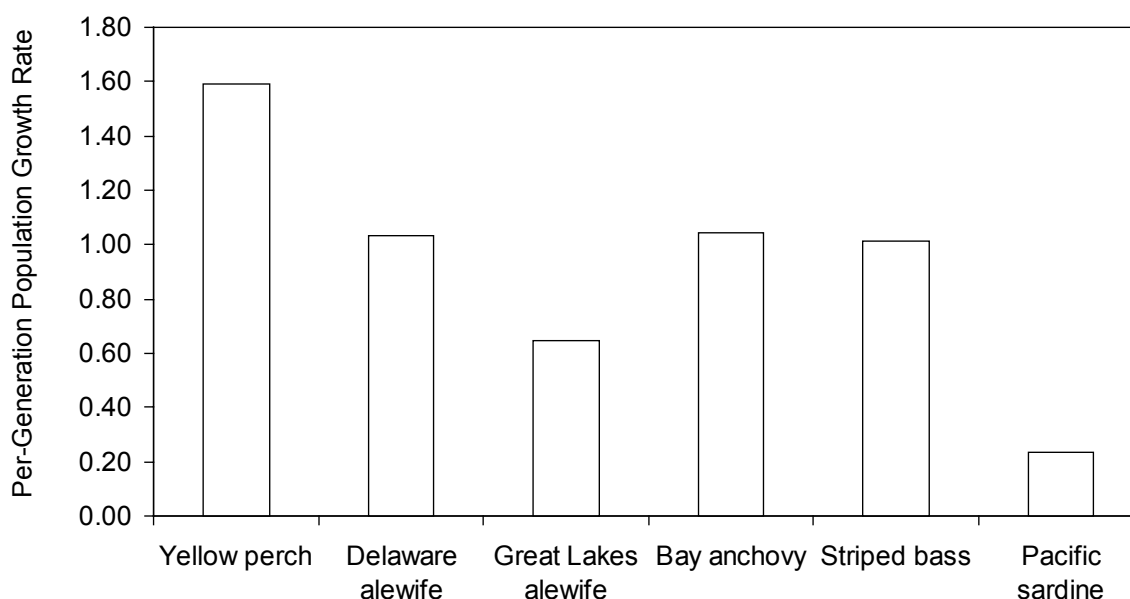
both equivalent adult losses and production foregone are theoretical quantities: estimates of the number of fish or quantity of biomass that could have been produced, given the assumptions used to make the calculations. There is no way to directly measure these quantities, and no way to independently validate the model projections. If the assumptions are wrong, or the parameters inaccurate, the model projections will also be inaccurate.

Even though EA and PF models cannot be independently validated, there are still methods available for identifying clearly unrealistic projections and for bounding the range of reasonable estimates. If, for example, a complete life table can be developed, the rate of population growth implied by the stage-specific mortality and fecundity parameters can be examined and compared to known or plausible rates of population change. If the life table implies that a population should be growing or shrinking at a high rate, whereas historical data show that the population has been relatively stable, then the life table parameters are clearly inaccurate. Figure 7-1 compares the per-generation population growth rates implied by the baseline parameter sets for yellow perch, Delaware alewife, Great Lakes alewife, bay anchovy, striped bass, and Pacific sardine. Values greater than 1 imply a growing population; values less than 1 imply a shrinking population. For all species except Pacific sardine, the baseline life tables imply a per-generation growth rate between 0.5 and 2, implying a rate of decline or growth of less than a factor of two per generation. For Pacific sardine, however, the life table implies that the population should be declining by about 80% per generation. Since this population has, in fact, been growing in the recent past this value is clearly implausible. The base case life table for Pacific sardine is very likely to be inaccurate and model projections derived from it are likely to be unreliable. In the Salem permit application (PSEG 1999), the life tables of all species were adjusted to achieve an approximate growth rate of zero per generation, on the grounds that in the long run all populations are approximately stable. This procedure, termed “life-cycle balancing” in the Salem permit application, can be applied to any species for which a life table can be developed. It provides a consistent approach to limiting the magnitude of the biases in model projections that can occur when parameter sets are developed by combining data from different studies, conducted at different times using different methods, on different life stages and populations. The assumption of stability is simple, and, at least in the case of EA models, makes the results of model applications less sensitive to model choice. However, it is still only an assumption. Alternatively, projections could be made using a range of parameter values corresponding to a plausible range of population growth rates.

Another approach to bounding the range of realistic model projections is to employ alternative models. This approach is most useful if the alternatives involve distinctly different assumptions or parameter sets. The EA and PF models documented in this report are closely related and were parameterized using the same data, so that it’s questionable whether they are distinct enough to qualify as alternative models. Nonetheless, the differences between the projections derived from the two EA and PF models provide additional insights into the range of uncertainty inherent in the estimates of prey biomass and equivalent adult losses. Application of multiple models is especially useful in the case of species such as gizzard shad. For this species, a complete life table could not be developed, so it was not possible to use implied population growth rates as a reality check on model projections. As shown in Table 6-9, the baseline equivalent adult projections from the FP and FH models differ by nearly a factor of two, even though the life



history parameter values are identical. This difference provides a minimum estimate of the uncertainty in estimating equivalent adult losses for this species, using the available data.



**Figure 7-1**

**Per-generation population growth rates implied by the baseline parameter sets for each species. Values greater than 1 indicate growing populations; values smaller than 1 indicate declining populations. The values for all species except Pacific sardine fall within a plausible range; the value for Pacific sardine conflicts with recent population trends and indicates that the baseline parameter set for this species is unrealistic.**

In addition to screening the model results for unrealistic projections and providing qualitative bounds on the uncertainties in those projections, it is clearly feasible to perform Monte Carlo uncertainty analyses on all of the models documented in this report. All of them can be readily implemented using off-the-shelf spreadsheet software and add-ins such as Crystal Ball® and @Risk®. Using the Monte Carlo approach, estimates (expressed as statistical distributions) of the uncertainties associated with individual parameter values are combined to provide estimates of the overall uncertainty (also expressed as a statistical distribution) associated with model projections. As an alternative to the Monte Carlo approach, Saila et al. (1997) described the use of “fuzzy arithmetic” to quantify uncertainty in equivalent adult losses of winter flounder, pollock, and red hake entrained and impinged at the Seabrook Station. Although EPA guidance encourages the use of quantitative uncertainty analyses in human health and ecological risk assessments performed at Superfund sites (USEPA 2001), it is not clear whether this approach would improve benefits assessments performed to support 316(b) determinations.

Regardless of what procedures are used to perform reality checks and uncertainty evaluations for these models, it is important to remember that all of them are ecologically simplistic and ignore important ecological processes that affect the growth and survival of fish. The most important of these processes is density-dependence (Rose et al. 2000). In reality, when some fish are lost due

### *Conclusions and Recommendations*

to entrainment or impingement, the survivors are likely to grow faster, have a higher probability of survival to adulthood, and provide a greater biomass-at-harvest than if there had been no entrainment or impingement losses. Moreover, rather than being permanently removed from ecosystems, the entrained and impinged fish are usually returned to the source water body where they can support future primary and secondary production. Methods for estimating the magnitude of the biases resulting from neglect of these compensatory processes are not currently available.

Recognizing the above limitations, the following sections provide specific recommendations concerning the use of the models discussed in this report.

## **7.1 Equivalent Adult Models**

The FP and FH models, although based on the same fundamental model of population dynamics, produce widely divergent results in many cases and respond in very different ways to variations in stage-specific natural mortality rates. The FH model is an appropriate choice if station losses are limited to entrainment of eggs and larvae, and if estimates of foregone harvest are not needed. If station losses include entrainment and impingement of substantial numbers of juvenile and older fish, or if estimates of foregone harvest are needed, then the FP model is the appropriate choice.

Both the FH and FP models are highly sensitive to uncertainties in estimates of natural mortality rates. Since site-specific estimates of natural mortality rates are rarely available (and are subject to numerous potential biases even when they are available), the values used should be supported by a thorough review of the available published estimates. If possible, a complete life table for each species should be developed and used to develop bounds on the range of plausible values for mortality rates. The assumption of long-term stability used in PSEG's (1999) life-cycle balancing procedure is a defensible default assumption for adjusting life history parameters. However, nonzero rates of population growth or decline could be used if justified by population-specific data. Available data on the size distribution and seasonal patterns of entrainment and impingement should be examined to determine whether the susceptible species are entrained and impinged throughout the distribution of each stage. If site-specific data show that losses occur primarily at the beginning of the stage, then the survival rate adjustment factor (Eq. 4, Section 4.1.1) should not be used.

In applying the FH model, the lifetime fecundity approximation (Eq. 7b, Section 4.1.2) should not be used because this approximation is likely to substantially overestimate the actual lifetime egg production of a typical female fish. Exact calculation from a life table (Eq. 7a, Section 4.1.2) is the preferred approach. If the age-specific mortality and fecundity parameters needed to develop a complete life table are not available, then available data on the ages or sizes of mature fish should be examined to estimate the number of years a typical female fish might be expected to survive. This value will almost always be much smaller than the maximum longevity of the species.

Only the FP model can be used to project equivalent adult losses to estimates of foregone harvest. In the case of managed species for which yield-per-recruit models are available, these models should be used to estimate the foregone harvest. In the absence of yield-per-recruit models, a conservative value can still be derived from the observation that for many fish species the maximum sustainable rate of fishing mortality is equal to or lower than the rate of natural mortality. If, over the lifetime of any group of fish, fishing mortality and natural mortality occur at equal rates, then one half of the fish will be harvested and one half will die of natural causes. As shown by Quinn and Deriso (1999), maximum sustainable rates of fishing mortality calculated using production models and spawner-recruit models are almost always somewhat lower than the corresponding natural mortality rates. Hence, a value of 50% appears to be a reasonable and conservative default estimate of the fraction of equivalent adults that could have been harvested. If estimates of the average weight of harvested fish are available, then an estimate of the foregone harvest can be obtained by multiplying the number of equivalent adults lost to the fishery by the average harvested weight.

## 7.2 Production Foregone Models

The Rago model and the BLM are similar in concept and, as shown in this report, produce similar results. The choice of which model to use should be made based on the types of data available, and in particular on the available data concerning stage-specific weights and growth rates. If the available data are limited to mean weights of organisms at each stage, then the BLM will be simpler to implement than the Rago model, which requires estimates of starting weights for each stage. If estimates of the size distributions of individuals in each stage are available from site-specific studies, then the Rago model may be preferable because the starting weights can be estimated directly from data. If the Rago model is used, however, this model will underestimate the total potential loss of prey biomass unless the weight of the entrained and impinged organisms is added to the production foregone estimate obtained from the model.

The choice of values for natural mortality rates is clearly much more important than the choice of which model to use. The applications documented in Section 6 of this report show that the Rago model and the BLM are equally sensitive to uncertainties in natural mortality rates. As in the case of equivalent adult models, the values used should be supported by a thorough review of the available published estimates. If possible, a complete life table for each species should be developed and used to develop bounds on the range of plausible values for mortality rates. Available data on the size distribution and seasonal patterns of entrainment and impingement should be examined to determine whether the susceptible species are entrained and impinged throughout the distribution of each stage. If site-specific data show that losses occur primarily at the beginning of the stage, then the survival rate adjustment factor (Eq. 4, Section 4.1.1) should not be used.

The assumptions used to convert estimates of prey biomass loss to estimates of foregone predator production and foregone harvest are also important concerns in the application of PF models. Two types of assumptions are involved: values for biomass conversion efficiency, and the fraction of prey biomass consumed by harvested vs. unharvested predators.

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Most previous applications of PF models have assumed a 10% conversion efficiency for all trophic levels. This value is supported by a wide variety of ecological studies performed over many years, including a comprehensive review by Pauly et al. (1995). In its Notice of Data Availability, EPA (USEPA 2003) used a value of 20%, on the grounds that this value is used in NOAA's Type A Natural Resource Damage Assessment model for the Great Lakes. The 10% value appears to have stronger support in the scientific literature than does the value used by EPA, and is recommended here as the most appropriate value in the absence of species-specific data. Trophic transfer efficiency is, however, an active area of ecological research and future studies may well support either higher or lower values.

The fraction of prey biomass consumed by different species or types of predators is generally unknown. Most previous applications of PF models have arbitrarily assumed that all prey biomass is consumed directly by harvestable predators. This is clearly a conservative assumption, because even if all mortality of prey species is assumed to be caused by predation, a substantial fraction of that predation may be from non-harvested species. EPA's assumption in the 316(b) case study (USEPA 2002) that only 20% of prey biomass is directly consumed by predators is also arbitrary. It would certainly be possible, given information on the relative abundance of different predator species in a water body of interest, to estimate the relative abundance of harvested vs. non-harvested predators that would consume the foregone prey biomass. In the absence of such studies, it would be more reasonable to use a range of values (e.g., 20% to 100%).

Many previous applications of PF models have also assumed that 100% of the predator biomass that could have been produced by consumption of the foregone prey biomass is actually harvested. The exploitation rate (100%) implied by this assumption conflicts with accepted principles of sustainable fishery management and is clearly overly conservative. In the case of managed species for which yield-per-recruit models are available, these models should be used to estimate the fraction of foregone predator biomass that could have been harvested. In the absence of yield-per-recruit models, a reasonable value can be derived using the approach recommended above for equivalent adult models. If, over the lifetime of any group of fish, fishing mortality and natural mortality occur at equal rates, then one half of the fish will be harvested and one half will die of natural causes. Because recent studies have shown that for the great majority of species the maximum sustainable rate of fishing mortality is somewhat lower than the natural mortality rate (Quinn and Deriso 1999), a value of 50% is a reasonable and conservative default estimate of the fraction of foregone predator biomass that could have been harvested.

### **7.3 Species to be Addressed in Companion Report**

One of the major findings of this report is that projections from all of the models are highly sensitive to uncertainties in life history parameter values such as natural mortality rates and growth rates. EPRI is currently funding a project to develop and document life history parameter estimates that can be used to implement EA and PF models at power plants throughout the United States, located at marine, estuarine, and freshwater sites. This project will develop

estimates of parameters needed to implement EA and PF for up to 50 fish and shellfish species that are commonly entrained and impinged at U.S. power plants. Regions from which species have been selected include the Atlantic, Gulf, and Pacific coasts (including both offshore and estuarine species), freshwater lakes and rivers, and the Great Lakes. The initial list of species is provided in Table 7-1. Within the two major categories (marine/estuarine and freshwater), the species are divided into two tiers. Tier 1 species consist of the species most frequently cited by EPRI members as being entrained or impinged at member-owned facilities. Tier 2 species consist of other frequently entrained or impinged species. All of the tier 1 species will be addressed in the companion report. Additional species from tier 2 will be addressed, depending on the availability of data and resources.

The parameters will be developed using a consistent parameter estimation approach based on methods used in existing model applications and on newly emerging methods from the scientific literature (e.g., general relationships between body weight and mortality that have been developed by biological oceanographers). Use of a consistent method for estimating parameters will help to ensure consistency between the parameter values developed for different species. The parameter sets developed for each species will be evaluated for internal consistency and biological plausibility using procedures recommended in section 6 of this report AE/PF guidance report. Where feasible and appropriate, ranges of values for key parameters will be developed.

The expected completion date for this companion report is December 31, 2004.

## Conclusions and Recommendations

**Table 7-1**  
**Species to be addressed in EPRI EA/PF Parameter Development Report**

Marine/Estuarine	Freshwater
<u>Tier 1: species frequently cited by EPRI members</u> Atlantic bumper Atlantic croaker Atlantic menhaden Atlantic silverside Bay anchovy Blueback herring Hogchoker Red drum Spot Striped bass Weakfish White croaker White perch Winter flounder Blue crab Brown shrimp White shrimp	<u>Tier 1: species frequently cited by EPRI members</u> Alewife Blackcrappie Channel catfish Emerald shiner Freshwater drum Gizzard shad Rainbow smelt White bass Threadfin shad Yellow perch
<u>Tier 2: other frequently reported species</u> Alewife American eel American shad Atlantic tomcod Bluefish Crevalle jack Gizzard shad Mummichog Northern pipefish Shiner surfperch Threespine stickleback Walleye surfperch White surfperch	<u>Tier 2: other frequently reported species</u> Bluegill Brown bullhead Carp Flathead catfish Golden shiner Largemouth bass Logperch Pumpkinseed Smallmouth bass Spottail shiner Trout-perch Warmouth White crappie

## 8

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# A

## DERIVATION OF THE ADJUSTED SURVIVAL FRACTION

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In the Salem and Mercer assessments and in the EPA Section 316(b) case study (USEPA 2002), equivalent adult losses were adjusted to account for the fact that early life stages of fish may be entrained at any point within a given life stage. This adjustment is needed because organisms entrained near the end of a given stage have a higher probability of reaching adulthood than do organisms entrained at the beginning of the stage. Neglecting the adjustment could lead to substantial underestimation of the numbers of equivalent adults entrained. This appendix documents the derivation of the adjustment factor.

Using the notation employed in Section 4 of this report, the number of organisms alive at the beginning of any given life stage ( $N_i$ ) that would be expected to survive to the beginning of the following life stage ( $N_{i+1}$ ) is given by:

$$N_{i+1} = N_i S_i \quad (\text{Eq. A-1})$$

Where:

$S_i$  = survival fraction from stage  $i$  to stage  $i+1$

If the probability of death for each organism is the same for every day during life stage  $i$ , then an exponential survival equation can be used to express the survival fraction  $S_i$  in terms of a daily instantaneous mortality rate and a life stage duration:

$$S_i = e^{-z_i d_i} \quad (\text{Eq. A-2})$$

Where:

$z_i$  = daily instantaneous mortality rate for age  $i$

$d_i$  = duration of life stage  $i$  (days)

The adjustment involves calculating the median age of death of organisms expected to die during life stage  $i$ . This median age-at-death age is defined as the time required for half of the organisms expected to die during life stage  $i$  to disappear from the population ( $\hat{d}_i$ ). Given  $N_i$  organisms entering life stage  $i$ , then  $N_i e^{-z_i \hat{d}_i}$  organisms would be expected to survive to enter life stage  $i+1$ , and  $N_i (1 - e^{-z_i \hat{d}_i})$  would be expected to die. Half of the deaths should occur on or

*Derivation of the Adjusted Survival Fraction*

prior to the median age-at-death,  $\hat{d}_i$ . The number of organisms still alive at the end of day  $\hat{d}_i$  is given by  $N_i e^{-z_i \hat{d}_i}$ , and therefore the value of  $\hat{d}_i$  can be calculated from:

$$N_i e^{-z_i \hat{d}_i} = N_i - 0.5 N_i (1 - e^{-z_i d_i}) \quad (\text{Eq. A-3})$$

From which it follows that:

$$\hat{d}_i = \frac{\ln 2 - \ln(1 + e^{-z_i d_i})}{z_i} \quad (\text{Eq. A-4})$$

Assuming that all organisms are equally vulnerable both to entrainment and to natural mortality on every day during life stage  $i$ , then the distribution of deaths due to entrainment should be the same as the distribution of deaths due to natural mortality. Therefore, one-half of entrainment deaths would be expected to occur prior to or on the median age-at death (i.e.,  $\hat{d}_i$ ) and one-half between the median age at death and the end of life stage  $i$ . The fraction of organisms entrained during life stage  $i$  can, under this assumption, be calculated by assuming that all organisms were entrained on day  $\hat{d}_i$ . The fraction surviving from  $\hat{d}_i$  to the beginning of stage  $i+1$  is the adjusted survival fraction ( $S_i^*$ ) used in Section 4 of this report. On day  $\hat{d}_i$  there are only  $(d_i - \hat{d}_i)$  days remaining in life stage  $i$  and, therefore, the adjusted survival fraction is given by:

$$S_i^* = e^{-(d_i - \hat{d}_i)z_i} = e^{-d_i z_i} e^{\hat{d}_i z_i} = S_i e^{\left[ \frac{\ln 2 - \ln(1 + e^{-z_i d_i})}{z_i} \right] z_i} = 2 S_i e^{-\ln(1 + S_i)} \quad (\text{Eq. A-5})$$

An analogous adjustment is needed in the fecundity hindcasting approach, however, because in fecundity hindcasting the quantity being estimated is the survival rate from spawning to the age at which entrainment and impingement occur, the time interval of interest is the period from the beginning of that stage to the median age-at-death. Because the product of the fraction surviving from the beginning of the stage to the median age-at-death and the fraction surviving from the median age-at-death to the end of the stage (i.e.,  $S_i^*$  as defined in equation A-5) must be equal to the fraction surviving over the entire life stage (i.e.,  $S_i$ ), the fraction surviving to the median age-at-death must be equal to  $S_i/S_i^*$ .

# B

## RELATIONSHIP BETWEEN EQUIVALENT ADULT ESTIMATES PROJECTED USING THE FORWARD PROJECTION (FP) AND FECUNDITY HINDCASTING (FH) APPROACHES

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The forward projection (FP) and fecundity hindcasting (FH) approaches to converting entrainment and impingement losses to numbers of equivalent adults are derived from the same fundamental model of population dynamics. It might be expected that, given any set of age-specific mortality and fecundity parameters, the two models should produce identical results. This expectation is incorrect. In fact, the two approaches produce identical results only in the case of a perfectly balanced population in which each mature female contributes exactly one mature female to the next generation of fish.

Using the FP approach, for any group of fish entrained or impinged at age or stage  $i$ , the number that would have survived to reach adulthood (defined as the earliest at which the first female fish become sexually mature) is given by:

$$EA_i = S_{iA} N_i \quad (\text{Eq. B-1})$$

Where:

$EA_i$  = equivalent adult loss due to fish entrained or impinged at age  $i$

$N_i$  = number of fish lost due to entrainment or impingement at age  $i$

$S_{iA}$  = fraction of fish expected to survive from age  $i$  to adulthood

Using the FH approach, the number of adult female fish required to produce the same number of entrained or impinged fish is given by:

$$EF_i = \frac{1}{E_L} \left( \frac{N_i}{S_{Ei}} \right) = \frac{N_i}{E_L S_{Ei}} \quad (\text{Eq. B-2})$$

Where:

$E_L$  = expected lifetime fecundity of a female fish

$S_{Ei}$  = fraction of eggs expected to survive to age  $i$

The ratio of the number of equivalent adults projected using the FP model to the number of equivalent females projected using the FH model is given by:

*Relationship between Equivalent Adult Estimates Projected using the Forward Projection (FP) and Fecundity Hindcasting (FH) Approaches*

$$\frac{EA_i}{EF_i} = \frac{S_{iA}N_i}{\left(\frac{N_i}{E_L S_{Ei}}\right)} = S_{Ei} S_{iA} E_L = S_{EA} E_L \quad (\text{Eq. B-3})$$

Where:

$S_{EA}$  = fraction of eggs expected to survive to adulthood

For the FP and FH models to produce identical results, the quantity on the right-hand side of Equation B-3, which is the lifetime fecundity of an adult female multiplied by the fraction of eggs that survive to become adults, would have to be equal to the ratio of equivalent adults from the FP model to the number of equivalent adult females from the FH model. Assuming a 50-50 sex ratio, this means that the results are equivalent if and only if:

$$S_{EA} E_L = \frac{1}{2} \quad (\text{Eq. B-4})$$

Since, under the 50-50 sex ration assumption, only half of the eggs produced by each female are female, Equation B-4 is equivalent to:

$$S_{EA} E'_L = 1 \quad (\text{Eq. B-5})$$

Where:

$E'_L$  = expected lifetime female egg production of a female fish

Hence, the FP and FH models produce identical results only if exactly one of the female eggs produced by each mature female fish survives to reach adulthood. This condition can occur only in a stable population in which each female contributes exactly one female to the next generation of fish. In a growing population, i.e., a population in which each adult female contributes more than one female to the next generation, the quantity ( $S_{EA} E'_L$ ) will be greater than 1.0, implying that the FP model will predict a greater number of equivalent adults per entrained or impinged fish than will the FH model. This is intuitively reasonable because higher survival implies that a greater fraction of the lost fish would have survived than would be the case in a stable population and, conversely, fewer adult females would be required to produce the observed losses. Conversely, in a declining population, lower survival implies that a smaller fraction of the lost would have survived to adulthood, and a greater number of adult females would be required to produce them.

This relationship between the FP and FH models explains the observation in Section 6 of this report that projections from these two models vary inversely with changes in the values of early life stage mortality rates.



*Program:*

Section 316(a) and 316(b)  
Fish Protection Issues

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