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METHODS FOR EVALUATING THE POTENTIAL CUMULATIVE EFFECTS OF POWER PLANT INTAKES ON COASTAL BIOTA

PIER FINAL PROJECT REPORT

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Preface

The California Energy Commission's Public Interest Energy Research (PIER) Program supports public interest energy research and development that will help improve the quality of life in California by bringing environmentally safe, affordable, and reliable energy services and products to the marketplace.

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Methods for Evaluating the Potential Cumulative Effects of Power Plant Intakes on Coastal Biota is one of three final reports for the Contract Number 500-04-025 conducted by Stratus Consulting. The information from this project contributes to PIER's Energy-Related Environmental Research Program.

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Abstract

A significant portion of California's generating capacity is represented by power plants located along the state's coast and estuaries that use once-through cooling technology. This technology requires the use of millions of gallons of water that are drawn into the power plant, and then discharged. There is increasing interest in evaluating the potential cumulative effects to marine life from the use of this cooling technology, but there remains little guidance on how to put the concept into practice. Of special interest is how to evaluate the cumulative effects of power plant entrainment, where small organisms carried along with the water drawn into the power plant where they are subjected to thermal, physical or chemical stresses.

This document seeks to fill this gap by evaluating suitable methods for assessing the cumulative effects of entrainment from power plants, and by identifying the relative advantages and disadvantages of each method. Entrainment can be evaluated at different levels of biological organization (individual, population, community, and ecosystem) depending on available data and assessment goals.

The report concludes with a recommendation for a cumulative impact analysis that involves the integration of entrainment studies into a regional framework encompassing multiple stressors and predator-prey relationships among species. Such an approach would be an important advance in the analysis of cooling water intake structure impacts and would support the growing interest among state and federal agencies in ecosystem-scale approaches to natural resource management.

This study benefits California ratepayers by providing a framework for stakeholders to select the most appropriate methodology to assess the cumulative effects of once-through cooling for power plants that is consistent with available data, local conditions, and assessment goals.

Keywords: cumulative impact analysis; cooling water intake structures; Clean Water Act section 316(b); entrainment; impingement; once-through cooling; marine fishes

Executive Summary

Introduction

In California, there are a large number of power plants located along the state's coast and estuaries that use once-through cooling technology. An estuary is a semi-enclosed body of water with rivers or streams flowing into it and with a connection to the open sea. Once-through cooling technology is where millions of gallons of water, withdrawn by the power plant from a water body, is passed once by the plant's condenser to remove waste heat and then discharged to a water body. As water is diverted into the power plant, organisms are carried along in the cooling water flow. The larger of these organisms may be trapped against the water intake screens (known as impingement) while the smaller organisms, those that can pass through the intake screens, are carried into the power plant's cooling system (known as entrainment) and subjected to thermal, physical, or chemical stresses.

For California, entrainment is in general a greater concern than impingement, primarily because most water intakes have relatively effective technologies for reducing impingement. Therefore, this document focuses on entrainment; however, in general the discussion applies to both impingement and entrainment.

Given the large number of coastal power plants in California that use once-through cooling, there is considerable interest in the potential cumulative effects of these facilities. Because power plants operate for decades, there is concern that impingement and entrainment losses may be cumulatively significant over time, even if losses in a single year may appear relatively minor. Similarly, even if the losses at one power plant may seem inconsequential, the cumulative effects of many intakes in the same water body could be a concern. Additionally, a single power plant in a water body with many stressors may contribute to significant cumulative impacts on local biological populations because of the combined effects of all stressors, including both the discharge and intake impacts of power plants.

Purpose

The report reviews the available approaches for cumulative impact analysis as they may apply to the evaluation of impingement and, in particular entrainment at California's coastal power plants that use once-through cooling. Because of the site-specific nature of these and other environmental impacts, the report does not prescribe a single approach, but rather is intended as a basis for collaborative decision-making by local stakeholders and agency representatives.

Project Approach

This study evaluated a number of methods that are suitable to assessing the cumulative impacts of entrainment from cooling water withdrawal by power plants, and identified the relative advantages and disadvantages of each method. These approaches or methods have different analysis endpoints (individual, population, community, and ecosystem) that, depending on available data and assessment goals, determine the model to be used.

Project Results

The most direct analysis of potential cumulative impacts to the larval stage quantifies the total losses of individuals (by number or weight) at the intake for a single facility through time, or for multiple intakes located on the same water body that affects the same biological populations. This involves simple counting of organisms entrained based on in-plant sampling or sampling in the water body directly in front of the intake. This approach, however, has the least value in assessing how these losses affect the overall ecosystem.

Population modeling is the minimum level of analysis needed to develop an ecologically meaningful context for interpreting entrainment losses and is the primary assessment method used in California. These assessments rely heavily on the Empirical Transport Model. This model estimates the proportion of larvae lost to entrainment to the population of that larvae in the source water. The source water is the area where larvae are at risk of being entrained and is determined by biological factors, current and tidal flows. Recent determinations using the Empirical Transport Model have calculated the average mortality across a few selected species and used this value as the best estimate of mortality for all entrained organisms.

A limitation of the Empirical Transport Model is the difficulty in determining the size and extent of the larval source water area and larval movements within that area, particularly for populations affected by strong tides and ocean currents, such as those in coastal California. Under these conditions, there can be significant uncertainty in defining the source water area and the duration of larval exposure to entrainment. This is a major concern because these factors have a profound influence on mortality estimates. While it is true that this model identifies the fraction of the larval population lost to entrainment, additional information is needed to judge the ecological significance of these losses.

In recent California studies, an approach known as Habitat Production Foregone or Area Production Foregone has also been used to help interpret the proportional mortality from entrainment. This method is a simple way to estimate the fraction of the source water area that corresponds to the fraction of larvae entrained and is therefore used to estimate how much restoration is needed to reduce the larval losses from entrainment. While such an approach is attractive because of its simplicity, this approach does have a number of assumptions that may or not be accurate. Other, more complex population models can be used but are often limited due to a lack of information on a species' life history.

Food web or trophic modeling, which describes the transfer of energy (food) between species within an ecosystem, can provide the complete ecological context for interpreting the significance of impingement and entrainment losses. The main disadvantage is that these models require considerably more data than population models. Information required for each species in the food web includes standing stock biomass (population in the study area), feeding rates, mortality rates, and other factors. Defining the area incorporated into the model are similar to those The spatial definition of the food web may also present many of the problems noted for determining larval population boundaries for the Empirical Transport Model for open populations in coastal waters. Disadvantages in using this model may be offset by the value of understanding the ecological significance of entrainment losses and of having a food web model that can be used to address a variety of other policy and management questions in a single region.

Conclusions

Each of the methods described in this review have advantages and disadvantages for use in a cumulative impact analysis and involve different tradeoffs. On the one hand, the least expensive and least uncertain assessment endpoint, individual entrainment losses, has the least ecological relevance. On the other, the method that provides the most information on the cumulative effects of entrainment losses on the ecosystem, food web modeling, is the most data intensive, most costly, and most likely to require assumptions with more uncertainty than the other approaches discussed.

Recommendations

Given the variety of possible assessment goals and the different tradeoffs involved in selecting among the available methods, results of this review support the use of an iterative approach to the assessment of cumulative effects. An iterative approach is a process where a model would be developed, tested and refined as additional information is introduced. This process would begin with a screening level analysis of individual losses, and proceed to the assessment of higher level endpoints depending on thresholds of concern and assessment goals. Ideally, all stakeholders will agree on the decision criteria for determining the level of loss that will trigger additional study at any given site and region.

A cumulative impact analysis will be subject to many of the same uncertainties as the underlying entrainment estimates, including natural variability in larval densities and rates of growth and mortality, that make ecological impacts difficult to detect. The ability to measure cumulative effects, whether at the population or community level, will be substantially less than the ability to measure impingement and entrainment. For any given situation, therefore, it will be important to balance the additional degree of uncertainty associated with higher level assessment endpoints against the additional understanding that could be gained from more complex analyses. These include:

- Screen existing impingement and entrainment data.
- Define impacts in a regional context and delineate the assessment area.
- Integrate assessment with other regional planning and management and.
- Conduct monitoring.

Benefits to California

The State Water Resources Control Board is currently considering a statewide policy for assessing cooling water intake structure impacts. The potential value of cumulative impact analysis is recognized by these and other agencies as well as various stakeholder groups, but until now there has been little guidance on how to put the concept into practice. This document provides information about available analytical approaches to help stakeholders make decisions consistent with available data, local conditions, and assessment goals.

1.0 Introduction

1.1. Background and Overview

The U.S. Environmental Protection Agency (EPA) is developing regulations to minimize the adverse environmental impact of cooling water intake structures, pursuant to section 316(b) of the federal Clean Water Act (CWA) (33 U.S.C. §1326). As the intake structures of once-through cooling power plants withdraw water from the surrounding water body, small organisms are drawn into the cooling water flow and subsequently trapped against intake screens (impingement) or carried into the power plant's cooling system (entrainment). Section 316(b) requires that the "location, design, construction, and capacity" of cooling water intake structures reflect the best technology available (BTA) for minimizing these impacts.

The State of California Water Quality Control Board and associated regional boards have been delegated the authority to implement section 316(b) under the federal National Pollutant Discharge Elimination System (NPDES) program (State Water Resources Control Board and California Environmental Protection Agency 2008). In addition, the California Energy Commission (Energy Commission) requires a license before construction or operation of a new power plant over 50 megawatts (MW) and when an operator upgrades or repowers a facility over 50 MW.

Even though NPDES permits are issued for individual facilities, Section 13142.5(a)(4) of California's Porter-Cologne Act requires that cumulative impacts are considered in all NPDES decisions. The California Environmental Quality Act (CEQA) also requires that cumulative impacts be considered in making a determination of significance.

Because power plants operate for decades, impingement and entrainment losses may be cumulatively significant over time even if losses in a single year appear minor. Similarly, even if the losses at one power plant seem inconsequential, the cumulative effects of many intakes in the same water body may be substantial. In addition, a single power plant in a water body with many stressors may contribute to significant cumulative impacts on local biological populations because of the combined effects of all stressors, including both the discharge and intake impacts of power plants.

Given the large number of coastal power plants in California that use once-through cooling, there is considerable interest in the potential cumulative effects of these facilities. However, there is little guidance on how to put the concept into practice. The purpose of this report is to review the available approaches for cumulative impact analysis as they may apply to the evaluation of impingement and entrainment. Because of the site-specific nature of these and other environmental impacts, the report is not prescriptive, but rather is intended as a basis for collaborative decision-making by local stakeholders.

1.2. Project Objectives

The objective of this project was to assemble and review approaches for evaluating the potential cumulative effects of impingement and entrainment, and to consider how these approaches may apply to the evaluation of impingement and entrainment by California's coastal power plants. Section 2.0 of the report outlines the study approach, Section 3.0 presents project results, Sections 4.0 discusses the conclusions of the study and Section 5.0 provides recommendations and describes project benefits for California.

2.0 Approach

2.1. Introduction

Studies of the cumulative impacts of impingement and entrainment, both in California and elsewhere in the United States, were assembled and reviewed, and the results were organized according to the level of biological organization of the assessment endpoint (individual, population, and community/ecosystem).

In general, entrainment is a greater concern in California than impingement, primarily because most intakes have relatively effective technologies for reducing impingement (Steinbeck et al., 2007; Ferry-Graham et al. 2008). Therefore, this document focuses on entrainment; however, in general, the discussion applies to both impingement and entrainment.

2.2. Project Outcomes

The following sections discuss methods for conducting a cumulative impact analysis of intake impacts, along with the relative advantages and disadvantages of each method. Examples of analytical results are also provided.

2.3. Cumulative Impact Analysis of Individual Losses

The most direct analysis of potential cumulative impacts quantifies the total losses of individuals (by number or weight) at the intake for a single facility through time, or for multiple intakes located on the same water body that affects the same biological populations. This involves simple enumeration of organisms entrained based on in-plant sampling or sampling in the water body directly in front of the intake.

2.3.1. Hudson River Entrainment

Among the first studies to assemble a time series of data for species affected by multiple intakes were those that quantified the annual losses of early life stages of fishes entrained at power plants along the Hudson River. The Hudson River is estuarine and under tidal influence for 243 kilometers (km) from Manhattan to the federal lock and dam at Troy, New York. A number of power plants that use once-through cooling withdraw water from this tidal portion of the river (Barnthouse et al. 1988).

Table 1 presents the average annual numbers of organisms entrained (eggs, larvae, and juveniles) over the period 1981-1987 for five fish species at three power plants in the southern portion of the river (Indian Point, Roseton, and Bowline). These three plants withdraw 4.6 billion gallons per day (BGD) of Hudson River water for cooling purposes. Entrainment numbers are based on in-plant sampling and assume 100% through-plant mortality. The data in the table indicate that cumulative entrainment averages over 2 billion organisms per year at the three facilities (Central Hudson & Gas Electric Corporation 1999).

Table 1. Estimated average annual losses of selected fish species at Roseton, Indian Point, and Bowline Stations on the Hudson River, 1981-1987. Figures are absolute numbers of entrainable life stages (eggs, larvae, and juveniles).

	Roseton	Indian Point	Bowline	Totals
American Shad	3,128,571	13,380,000	346,667	16,855,238
Bay Anchovy	1,892,500	326,666,667	81,000,000	409,559,167
River Herring	345,714,286	466,666,667	13,814,286	826,195,238
Striped Bass	129,857,143	158,000,000	15,571,429	303,428,571
White Perch	211,428,571	243,333,333	13,257,143	468,019,048
Totals	692,021,071	1,208,046,667	123,989,524	2,024,057,262

Source: Appendix VI-1-D-2 in Central Hudson & Gas Electric Corporation (1999).

2.3.2. California Entrainment

In California, a larger number of intakes and greater intake flows result in cumulative entrainment that is substantially higher than in the Hudson River. This is not surprising given that the source water volumes are higher by several orders of magnitude. The estimates of absolute losses of fish and macroinvertebrate larvae in Tables 2 and 3 give an indication of the relative magnitude of annual entrainment in California on a per facility basis. The data are for species or species groups (taxa) making up 90% or more of a facility's entrainment based on recent studies conducted with the same sampling and analytical methods (described in Steinbeck et al. 2007). Data are organized according to the water body type of the intake source water.

Entrainment of Fish Larvae

Table 2 indicates that under actual flows, annual entrainment of larval fishes at the time of sampling has ranged from a low of 65.3 million larvae at the Harbor facility to a high of 3.65 billion at the Haynes station, both in Southern California. Under maximum flows, which represent the design capacity for the intake, entrainment ranged from a low of 153.3 million at the Harbor facility to a high of 4.53 billion at the Haynes facility.

Table 2: Estimated annual entrainment of fish larvae (#/yr) at once-through cooling power plants in California. Plants are grouped according to the water body type of the intake source water (bay/estuary, ocean, enclosed bay/harbor).

Facility	Design Flow (MGD)	Location	Most Vulnerable Species/Taxa	Annual Entrainment-Actual Flow (Number per year)	Annual Entrainment-Maximum Flow (Number per year)
Bay/Estuary					
Alamitos	1,273	Los Cerritos Channel, Alamitos Bay	combtooth blennies (*Hypsoblennius* spp.), silversides (Atherinopsidae), anchovies (Engraulidae) unidentified gobies	1,686,757,809	--
Haynes	1,014	Alamitos Bay, within Long Beach Marina	unidentified gobies, silversides, combtooth blennies, white croaker (*Genyonemus lineatus*), anchovies	3,649,208,392	4,527,644,084
Morro Bay	668	Morro Bay, near mouth	combtooth blennies, unidentified gobies, shadow goby (*Quietula y-cauda*), Pacific staghorn sculpin (*Leptocottus armatus*), northern lampfish (*Stenobrachius leucopsarus*), KGB rockfishes*, jacksmelt (*Atherinopsis californiensis*), white croaker, Pacific herring (*Clupea pallasi*), cabezon (*Scorpaenichthys marmoratus*)	--	508,296,011
Potrero	505	South San Francisco Bay	bay goby(*Lepidogobius lepidus*), unidentified gobies, yellowfin goby (*Acanthogobius flavimanus*), northern anchovy (*Engraulis mordax*), Pacific herring	--	284,066,254
South Bay	602	San Diego Bay	anchovies (*Anchoa* spp.), combtooth blennies, CIQ gobies**, longjaw mudsucker (*Gillichthys mirabilis*), silversides	--	2,420,527,779

Diablo Canyon	2,670	open coast rocky cove	KGB rockfish complex[1], blue rockfish (*Sebastes mystinus*), smoothhead sculpin (*Artedius lateralis*), snubnose sculpin (*Orthonopias triacis*), white croaker, monkeyface prickleback (*Cebidichthys violaceus*), clinid kelpfishes (*Gibbonsia* species), blackeye goby (*Rhinogobiops nicholsii*)	2,017,273,000	--
El Segundo	607	Santa Monica Bay	anchovies, silversides, sea basses (*Paralabrax* spp.), white croaker, queenfish (*Seriphus politus*), combtooth blennies, CIQ gobies[2], California halibut (*Paralichthys californicus)*, diamond turbot (*Pleuronichthys guttulatus*), sanddabs (*Citharichthys* spp.), English sole (*Parophrys vetulus*)	222,275,332	--
Huntington Beach	516	South Coast, South Palos Verdes Region	spotfin croaker, queenfish blennies and unidentified gobies, anchovies and croakers	--	354,877,227
Redondo Beach	1,146	King Harbor, Santa Monica Bay	anchovies, silversides, clingfishes, sea basses, queenfish, unid. croakers, garibaldi (*Hypsypops rubicundus*), combtooth blennies, Labrisomid blennies, kelp blennies (*Gibbonsia* spp.), unidentified gobies, blind goby (*Typhlogobius californiensis*), California halibut	291,196,721	--

Scattergood	496	Santa Monica Bay	anchovies, silversides, sea basses, white croakers, queenfish, senorita (*Oxyjulis californica*), combtooth blennies, CIQ gobies[2], Pacific barracuda, California halibut, diamond turbot, sanddabs, spotted turbot (*Pleuronichthys ritteri*), English sole	--	524,202,652
			Enclosed Bay/Harbor		
Harbor	108	Los Angeles Harbor	CIQ gobies[2], yellowfin goby, white croaker, bay goby (*Lepidigoobius lepidus*), combtooth blennies, anchovies	65,298,000	153,331,013
Moss Landing	1,226	Moss Landing Harbor/Elkhorn Slough	blennies, white croaker, unid. gobies, bay goby, blackeye goby, Pacific herring, Pacific staghorn sculpin	--	476,292,160

1. KGB rockfish complex = kelp rockfish (*Sebastes atrovirens*), gopher rockfish (*S. carnatus*), black-and-yellow rockfish (*S. chrysomelas*)
2. CIQ goby complex = *Clevelandia ios* (arrow goby), *Ilypnus gilberti* (cheekspot goby), and *Quietula y-cauda* (shadow goby)

Sources:
a) Alamitos: Tables 4.5-3, 4.5-4, 4.5-5, 4.5-8 in MBC and Tenera (2007a). Data are for all units.
b) Haynes: Tables 4.5-2 and 4.5-4 in MBC, Tenera, and URS (2007b).
c) Morro Bay: Tenera (2001).
d) Potrero: Table 4.2 in Tenera (2005a). Estimate is for Unit 3 only.
e) South Bay: Table 4-1 in Tenera (2005b).
f) Diablo Canyon: Table 23 in Steinbeck et al. (2007). Original study is Tenera (2000a).
g) El Segundo: Tables 4.5-2 and 4.5-4 in Tenera and MBC (2008). Data are for all units.
h) Huntington Beach: Tables 4-1 and 4-2 in MBC and Tenera (2005).
i) Redondo Beach: Tables 6.2-1 and 6.2-3 in MBC and Tenera (2007b). Data are for Units 5-8.
j) Scattergood: Tables 4.5-2 and 4.5-4 in MBC, Tenera, and URS (2007c.
k) Harbor: Tables 4.5-2 and 4.5-4 in MBC, Tenera, and URS (2007a).
l) Moss Landing: Tenera (2000b). Data are for Units 1-7.

Entrainment of Macroinvertebrate Larvae

Invertebrates are difficult to sample and early life stages are difficult to distinguish at the species level. Therefore, invertebrate sampling at California's coastal power plants typically focuses on the larvae of only a few groups of larger invertebrates, primarily crabs (Steinbeck et al., 2007).

Table 3 presents absolute losses of macroinvertebrates targeted for sampling at California power plants, organized according to the water body type of the intake source water. In recent sampling, annual entrainment under operational flows ranged from a low of 4.3 million macroinvertebrate larvae at Alamitos to a high of 273.3 million at Scattergood, both in Southern California. Annual entrainment at maximum flows ranged from 3.7 million larvae at Moss Landing in Central California to 473.6 million at Huntington Beach in Southern California.

Table 3: Estimated annual entrainment of macroinvertebrate at once-through cooling power plants in California. Plants are grouped according to the water body type of the intake source water (bay/estuary, ocean, enclosed bay/harbor).

Facility	Design Flow (MGD)	Location	Most Vulnerable Species/Taxa	Annual Entrainment-Actual Flow (Number per year)	Annual Entrainment-Design Flow (Number per year)
Bay/Estuary					
Alamitos[a]	1,273	Los Cerritos Channel, Alamitos Bay	shore crabs (Grapsidae), kelp crabs, *Pugettia* spp.), pea crabs (*Pinnixa* spp.), yellow shore crabs (*Hemigrapsus oregonensis*), striped shore crabs (*Pachygrapsus crassipes*), porcelain crabs (Porcellanidae)	4,329,952	--
Haynes[b]	1,014	Alamitos Bay, within Long Beach Marina	shore crabs, kelp crabs, pea crabs	14,854,700	18,495,973
Morro Bay[c]	668	Morro Bay, near mouth	*Cancer* crabs	--	13,577,334
Ocean					
El Segundo[d]	607	Santa Monica Bay	*Cancer* spp. crabs	23,685,773	--
Huntington Beach[e]	516	South Coast, South Palos Verdes Region	*Cancer* spp. crabs, mole crabs	--	473,628,497

Redondo Beach[f]	1,146	King Harbor, Santa Monica Bay	California spiny lobster (*Panulirus interruptus*), pea crabs, spider crabs (Majidae), black-clawed crabs (*Lophopanopeus* spp.), kelp crabs	23,256,655	--
Scattergood[g]	496	Santa Monica Bay	kelp crabs, pea crabs, market squid (*Loligo opalescens*)	27,322,839	40,628,889
			Enclosed Bay/Harbor		
Harbor[h]	108	Los Angeles Harbor	kelp crabs, spider crabs, pea crabs	18,901,336	41,345,075
Moss Landing[i]	1,226	Moss Landing Harbor/Elkhorn Slough	crabs	--	3,754,510

Sources:
a) Tables 4.5-3, 4.5-4, 4.5-5, 4.5-8 in MBC and Tenera (2007a). Data are for all units.
b) Tables 4.5-2 and 4.5-4 in MBC, Tenera, and URS (2007b)
c) Tenera (2001).
d) Tables 4.5-2 and 4.5-4 in Tenera and MBC (2008). Data are for all units.
e) Tables 4-1 and 4-2 in MBC and Tenera (2005).
f) Tables 6.2-1 and 6.2-3 in MBC and Tenera (2007b). Data are for Units 5-8
g) Tables 4.5-2 and 4.5-4 in MBC, Tenera, and URS (2007c).
h) Tables 4.5-2 and 4.5-4 in MBC, Tenera, and URS (2007a).
i) Tenera (2000b). Data are for Units 1-7.

2.3.3. Advantages and Disadvantages of Individual Loss Estimates

Enumeration of individual losses is the most direct method of determining the magnitude of entrainment and the potential cumulative effects of entrainment through time. As such, it has the least uncertainty of all the methods discussed in this report. This advantage is offset, however, if the goal of the assessment is to determine the ecological significance of the losses. Because most early life stages of fishes and macroinvertebrates have naturally high mortality rates, most of these organisms would die before reaching age 1, even in the absence of entrainment. As a result, high losses of larvae don't necessarily mean that the adult population will be adversely affected.

2.4. Cumulative Impact Analysis of Larval Populations

Losses of individual larvae are sometimes referenced to the total population of larvae in the surrounding water body. This type of analysis begins by determining the entrainment mortality rate assuming no other sources of mortality. This mortality rate, known as the conditional mortality rate (CMR) due to entrainment, is equivalent in concept to the CMR due to fishing described by Ricker (1975). Expressed as a percentage, the entrainment CMR indicates the percent of larvae entrained from the vulnerable larval population.

In its simplest form, the entrainment CMR is estimated as the rate at which planktonic organisms are entrained per unit volume of intake flow. Expressed as a percentage, the entrainment CMR indicates the percentage of the population in the surrounding water body that is entrained. This concept has been applied in a number of hydrodynamic and empirically-based models of fractional losses, as described in the following sections.

2.4.1. Hydrodynamic Fractional Loss Model

To conduct a cumulative impact analysis of entrainment by intakes in Delaware Bay, Edinger and Kolluru (2000) used a hydrodynamic model to represent entrainment based on a simulated dye release into different volumes of the modeled water body. The dye indicated the mass of water that would be entrained from any given region. The model, known as the 3D Generalized Longitudinal Lateral and Vertical Hydrodynamic and Transport Model (GLLVHT) (Edinger et al. 1993), included a detailed hydrodynamic grid with 26 5-km longitudinal segments and five lateral segments, with 2 meters (m) thick layers. The width of the grid increased in the down estuary direction, and a freshwater inflow was established at the head of the estuary. The grid was overlain with 24 entrainment regions, eight along the estuary and three laterally. The lateral regions included a near shore over bank region, a channel region, and a far shore over bank region. The eight hypothetical intakes were located in the middle of each near shore region.

Model results indicated that cumulative entrainment could be significant in that the simulated daily entrainment rates were similar to or exceeded natural mortality rates, which Edinger and Kolluru (2000) estimated are in the range of 2-7% per day for most estuarine fish species. Results also showed how entrainment rates, and the relative contribution of individual intakes to the overall impact, could vary as a function of simulated variations in flows, currents, intake sizes, and intake locations.

2.4.2. Empirical Transport Models

The next level of sophistication in fractional loss models involves actual sampling of organisms in place of inferences based on hydrodynamic modeling.

Hudson River Entrainment

The Empirical Transport Model (ETM) was developed to take advantage of existing data on river-wide distributions of fish eggs and larvae in the Hudson River (Boreman et al. 1978, 1981). A similar model is also available for impingement (Barnthouse et al. 1979). According to the ETM, the fractional entrainment of a given life stage during any time interval is given by the ratio of the water withdrawn to the water volume occupied by that life stage in the surrounding water body. The model considers spatial and temporal differences in the distribution and abundance of each life stage in the water body where the intake is located, and assumes that each larval cohort is vulnerable to entrainment for a specified number of days, after which it grows or moves out of the area. To make it possible to estimate CMRs for more than one power plant or more than one source of mortality, the model divides the water body into regions. Daily entrainment mortality rates are calculated for each cohort, age, and life stage in each region during each model time step, and then these rates are combined into an overall annual conditional mortality rate due to entrainment for each species (Boreman et al. 1978, 1981; Central Hudson & Gas Electric Corporation 1999).

The conditional entrainment mortality rate (CEMR) model is a variant of the ETM. The major difference is that the CEMR is based on actual counts of entrained organisms at the intake instead of the intake flow volume. The method was developed to take advantage of in-plant entrainment monitoring that was conducted at three Hudson River plants (Bowline, Roseton, Indian Point) in 1981 and from 1983 through 1987. According to this model, the CEMR of a given life stage is the ratio of the average density of the life stage entrained at the intake to the density of the life stage in the water body. Because the CEMR model uses actual entrainment data, the New York Department of Environmental Conservation (NYDEC) recommends use of the CEMR model instead of the volumetric ETM whenever possible (Central Hudson & Gas Electric Corporation, 1999).

To produce a consistent time series of CEMRs for Hudson River fishes for the entire period 1974 through 1997, the ETM was modified to have an algebraic structure like the CEMR model and calibrated using the available in-plant entrainment data from the three plants during the period 1981 to 1987 (Central Hudson & Gas Electric Corporation, 1999). Table 4 presents the resulting annual river-wide CEMRs for the five target species in Table 1, along with the average annual CEMR for each species. On this basis, the average annual percentage of fish larvae entrained ranged from a low of 20.64 for bay anchovy to a high of 40.57 for striped bass.

Table 4. Annual riverwide entrainment of five Hudson River fish species, expressed as percents, assuming 100 percent through-plant mortality.

Year	American Shad	Striped Bass	White Perch	River Herring	Bay Anchovy
1974	3.62	32.13	28.99	30.78	19.03
1975	42.31	39.89	35.26	43.92	16.76
1976	41.56	35.60	25.75	35.88	10.47
1977	8.37	52.67	30.64	37.11	25.00
1978	22.50	39.17	23.78	32.33	24.71
1979	34.75	47.01	28.86	31.26	20.43
1980	43.92	47.49	24.46	28.64	30.77
1981	17.97	28.48	22.50	14.51	29.06
1982	13.70	45.32	19.13	11.72	11.32
1983	21.62	40.01	43.78	24.37	17.93
1984	24.57	55.37	29.67	29.38	14.55
1985	22.32	28.71	13.84	17.20	21.84
1986	12.40	63.13	39.38	11.69	11.84
1987	35.07	35.31	24.90	37.63	21.25
1988	45.00	52.34	31.32	27.93	29.11
1989	45.40	29.99	28.35	37.64	15.97
1990	54.40	42.77	26.44	34.33	33.12
1991	38.37	41.47	25.00	25.45	21.01
1992	59.87	40.90	25.42	45.47	17.54
1993	11.29	31.20	14.33	13.44	17.81
1994	25.43	36.20	18.64	20.18	15.66
1995	14.82	34.15	17.92	16.09	26.40
1996	7.45	45.04	18.09	8.80	23.44
1997	19.91	29.29	21.30	24.96	20.25
AVERAGE	27.78	40.57	25.74	26.70	20.64

Source: Table X-23 of Appendix VI-I-B of the Draft Environmental Impact Statement (Central Hudson & Gas Electric Corporation, 1999).

2.4.3. *Modified Empirical Transport Model*

A version of the ETM that is similar to the CEMR method has been widely used in California (Steinbeck et al. 2007). Based on concepts in MacCall et al. (1983), the modified ETM uses entrainment and water body sampling of larvae to estimate proportional entrainment (PE), the fraction of larvae of a given species (or taxon) that is entrained from the larval source population during a given sampling event on a given day, expressed as:

$$PE = \frac{N_E}{N_G}$$

where:

N_E = estimated number of larvae entrained during the day, calculated as (estimated daily concentration of larvae entrained) x (daily cooling water intake flow)

N_G = estimated number of larvae in the water body sampling grid that day.

The ETM is used to combine the PEs from all sampling events to estimate the annual probability of mortality on the larval source water population due to entrainment, known as the PM, which is given as:

$$PM = 1 - \sum_{i=1}^{n} f_i (1 - PS * PE_i)^d$$

where:

PE_i = proportional entrainment for survey i

PS = the ratio of the number of larvae vulnerable to entrainment and the number of larvae in the source population

f_i = annual proportion of total larvae hatched during survey i

d = days of larval exposure

The days that larvae of a given species are exposed to entrainment is determined from otoliths or based on the size frequency distribution of the individuals entrained, coupled with a length-at-age relationship, usually taken from the scientific literature (Steinbeck et al. 2007).

If the distribution of larvae in the source water area (SWA) is representative of the distribution in the extrapolated water volume, PS can be estimated as the sampled source water volume to the total source volume. Note that the term "source water area" as used in the context of ETM modeling has an entirely different meaning than in section 316(b) of the CWA, where it refers to the source water of the intake, i.e., the source of the cooling water.

If it is not possible to estimate PS (e.g., because larval distributions include nearshore ocean waters that are difficult to sample), the PS is estimated using estimates of the length and depth of the sampling area and the length of alongshore current displacement based on the number of days larvae are exposed to entrainment. If the entire source water is sampled, then PS = 1 (Steinbeck et al. 2007).

For example, assume the simplest case where PS = 1. The PS term falls out of the PM equation when the ratio of the number of larvae vulnerable to entrainment and the number of larvae in the source population is the same or 1. If the larvae of the given species are entrained between 4 and 12 days of age, and PE = 0.1, then the estimate of PM = 1 - $(1 - 0.1)^8$ = 0.5695, or about 57%.

Table 5 presents PMs of larval fishes at power plants located south of Point Conception, the biogeographic boundary between Northern and Southern California. The data in the table are for larvae only because of the difficulties identifying eggs to species and because some of the affected species have demersal, adhesive eggs not vulnerable to entrainment (e.g., gobies) or brood eggs internally (e.g., rockfishes). Steinbeck et al. (2007) note that because the egg stage is not included in these estimates, the period of exposure to entrainment is underestimated for species with planktonic eggs (e.g., anchovies, croakers, flatfishes), resulting in an underestimate of the PM.

Table 5 indicates that in Southern California, most PMs are under 1%. However, three facilities in Southern California with intake flows that can reach a billion gallons per day or more (Haynes, Alamitos, Redondo Beach) show significantly higher PMs for some species. The highest PMs are 25.15% for gobies, 13.89% for combtooth blennies, and 40.95% for silversides, all at the Haynes station. At Redondo Beach, PMs reach up to 11.78% for clingfishes. At Alamitos, the highest PM is 13.32% for gobies.

Table 5: Percentage of proportional mortality for entrainment of fish larvae at power plants south of Point Conception*.

Family	Species	Scattergood	El Segundo	Redondo Beach	LA Harbor	Huntington Beach	Alamitos	Haynes
Atherinopsidae	unidentified silversides	3.04	1.78	3.96			8.39	40.95
	California grunion							
	Jacksmelt							
Blenniidae	unidentified blennies					0.8		
	combtooth blennies	0.39	0.28	7.18	0.06		8.99	13.89
Clinidae	unidentified kelpfishes			4.60				
	giant kelpfish							
Engraulidae	unidentified anchovies	0.19	0.18	0.84	0.71			0.76
	northern anchovy					1.20		
Gobiidae	unidentified gobies	5.07	1.50	5.07	2.65		13.32	25.15
	bay goby				0.24			
	blind goby			9.46				
	cheekspot goby							
	CIQ gobies**					1		
	yellowfin goby				0.65			
Gobiesocidae	unidentified clingfishes			11.78				

Family	Species	Scattergood	El Segundo	Redondo Beach	LA Harbor	Huntington Beach	Alamitos	Haynes
Labridae	senorita	0.56						
Labrisomidae	unidentified labrosomid blennies			5.78				
Paralichthyidae	sanddabs	0.08	0.10					
	California halibut	0.26	0.17	0.16		0.3		
Pleuronectidae	diamond turbot	0.94	0.79			0.6		
	spotted turbot	0.10						
Pomacentridae	garibaldi			8.88				
Sciaenidae	unidentified croakers	0.64	0.42	0.40	0.19			
	black croaker					0.1		
Sciaenidae (cont'd)	corbina							
	spotfin croaker					0.3		
	white croaker	0.37	0.29	0.64	0.19	0.7		0.63
	queenfish	0.06	0.05	0.14		0.6		
Serranidae	sand bass	0.17	0.33	0.29				
Sphyraenidae	Pacific barracuda	0.36						
AVERAGE		**0.87**	**0.50**	**4.23**	**0.67**	**0.62**	**10.23**	**16.28**

* Proportional mortality estimates were corrected for the period of time that the eggs are exposed to entrainment by adding the estimated egg duration to the total duration used in the PM calculations. This assumes that PE is the same for eggs and larvae.
** CIQ gobies = *Clevelandia ios* (arrow goby), *Ilypnus gilberti* (cheekspot goby), and *Quietula y-cauda* (shadow goby)

Sources:
a) Scattergood – MBC, Tenera, and URS 2007c.
b) El Segundo – Tenera and MBC 2008.
c) Harbor – MBC, Tenera, and URS 2007a.
d) Huntington Beach - MBC and Tenera 2005.
e) Alamitos - MBC and Tenera 2007a.
f) Haynes – MBC, Tenera, and URS 2007b.
g) Redondo Beach – MBC and Tenera (2007b)

Table 6 presents PMs for power plants north of Point Conception. Although gobies and blennies dominate entrainment samples both north and south of Point Conception, PMs for these taxa are significantly higher at three power plants in Central California that have intake flows in excess of a billion gallons per day (Diablo Canyon, Morro Bay, and Moss Landing). For example, PMs for combtooth blennies reach 49.1% at the Morro Bay plant compared to the highest PM for this species in Southern California (13.8% at the Haynes facility). A number of the species showing relatively high PMs at the three Central California facilities are the same as those with high PMs at the high flow facilities in Southern California, including combtooth blennies and several goby species. Several additional species entrained in Central California that show relatively high PMs include white croaker and Pacific herring (both coastal pelagics) and smoothhead sculpin, monkeyface prickleback, Pacific staghorn sculpin (*Leptocottus armatus*), jacksmelt, and kelpfishes (nearshore species).

With these exceptions, the PMs indicate that entrainment as a fraction of larval populations is substantially less in California (Tables 4 and 5) compared to the Hudson River (Table 3). This is not unexpected given that the Hudson River is a relatively narrow, semi-enclosed water body with a unidirectional flow, and power plants are located close together, resulting in impacts that are concentrated. Even if they could be reliably estimated, the SWAs of open populations along the California coast will tend to be larger, resulting in lower fractional losses even in cases where absolute losses may be considerably higher than for Hudson River fishes.

Table 6. Proportional mortality, expressed as Percentage, for entrainment of fish larvae at power plants north of Point Conceptiona.

Family	Species Common Name	Diablo Canyon	Morro Bay	Moss Landing	Potrero
Atherinopsidae	jacksmelt		21.9		
Blenniidae	combtooth blennies		49.1	18.2	
Clinidae	unidentified kelpfishes	18.9			
Clupeiformes	Pacific herring		1.2	13.4	0.35
Cottidae	cabezon	1.1	3.7		
	monkeyface prickelback	13.8			
	Pacific staghorn sculpin		5.1	11.8	
	smoothead sculpin	11.4			
	snubnose sculpin	14.9			
Engraulidae	northern anchovy				0.29
Gobiidae	unidentified gobies		11.5	10.7	0.48
	bay goby			21.0	0.25
	blackeyed goby	11.5		7.5	
	longjaw mudsucker			8.9	
	shadow goby		1.4		
	yellowfin goby				0.17
Hexagrammidae	painted greenling	6.3			
Myctophidae	northern lampfish		2.4		
Paralichthyidae	California halibut	0.5			0.08
	sanddabs	1.0			
Sciaenidae	white croaker	0.7	2.1	12.9	0.49
	queenfish				
Scorpaenidae	blue rockfish	0.4			
	KGB rockfish complex[b]	3.9	2.4		
AVERAGE		**10.55**	**10.08**	**13.1**	**0.30**

a. The egg stage was not accounted for in PM estimates for Diablo, Morro Bay, and Moss Landing, where many of the fishes do not have planktonic egg stages.
b. KGB rockfish complex = KGB rockfish complex = kelp rockfish (*Sebastes atrovirens*), gopher rockfish (*S. carnatus*), black-and-yellow rockfish (*S. chrysomelas*)

Sources: Diablo Canyon- Table 23 in Steinbeck et al. (2007) (for sampling period 3) based on Tenera (2000a).Morro Bay-Tenera (2001). Moss Landing-Tenera (2000b). Potrero-Tenera (2005a).

2.4.4. ETM Model of Cumulative Impacts in the Southern California Bight

A recent study evaluated the potential cumulative effects of the intakes of 12 power plants in the Southern California Bight (Bight) using the modified ETM (MBC and Tenera 2005). Intakes evaluated included those at Mandalay, Ormond Beach, Scattergood, Segundo, Redondo Beach, Harbor, Long Beach, Alamitos, Haynes, Huntington Beach, San Onofre, and Encina. The analysis was considered a "first order" effort and as such did not include details such as seasonal abundance changes that could be important.

The Bight is an area about 78,000 km^2 between Point Conception and Baja California, Mexico. Because of a lack of entrainment data at the time of the analysis for several of the facilities, PEs for all plants and species were estimated as the ratio of the cooling water volume to the volume of the Bight (similar to the original ETM in New York). Estimates were calculated using a range of durations representative of the larval durations of fishes collected in recent Southern California studies. Each plant's permitted flow was used for the intake flow (MBC and Tenera 2005).

This formulation assumes that larval densities are uniform throughout the source water volume and that the volume of the Bight adequately represents the size of the source populations. The model was calculated using maximum design flows. The use of maximum flows is conservative and would tend to overestimate cumulative effects since most plants run at levels below 100% (MBC and Tenera 2005).

PE estimates were converted to survival estimates as:

$$S = e^{-PE\,t}$$

where:

S = survival

t = days of larval exposure

To account for the potential overlapping effects of the many power plants, cumulative effects were estimated as the product of all of the survival estimates (MBC and Tenera 2005).

Results were very sensitive to the estimated size of the SWA. This is a direct result of the model. According to the model formulation, as the source water area increases, the mortality rate decreases; as exposure to entrainment increases, the morality rate increases. When the estimated volume of the Bight was defined as the area shoreward of 75-m depth limit and an average larval period of 40 days was assumed for all species, the estimated cumulative morality rate due to entrainment was 1.4 percent for the 12 facilities. The estimate was over three times higher if a 35-m depth limit was used to define the source water volume (i.e., if the SWA was smaller) (MBC and Tenera 2005).

MBC and Tenera (2005) calculated cumulative mortality for a range of source water volumes and days of larval exposure to entrainment. Results indicated that the cumulative mortality rate due to entrainment increased exponentially as source water volumes decreased. Mortality varied as a function of the duration of the larval period and the spread of power plant locations.

As a result, the estimated PM increased as exposure increased but decreased the greater the spread because of the associated increase in the estimated source water volume.

Given these sensitivities, the result of this cumulative impact analysis should be viewed with caution. Because these different factors will affect mortality rates due to entrainment in different ways, they could lead to either an under- or overestimate of the mortality rate due to entrainment depending on what particular assumptions are made. For example, the assumption of an average larval duration could lead to an under- or overestimate of larval losses depending on the actual distribution of the periods of exposure of affected species (MBC and Tenera 2005).

Because of the many data uncertainties involved in an analysis of this kind, and the strong degree to which model assumptions about the source water volume and period of larval exposure influence model results, a more complete analysis would develop species- and facility specific estimates, in addition to the impact of all intakes combined, to determine if any facilities showed rates of concern that might not be apparent when rates are combined.

There are a number of reasons why individual results could differ from the average. Actual entrainment will be higher than the average for species with a smaller SWA than the Bight. In addition, some species' populations may show greater sensitivity to entrainment mortality because of current population status. Although effects averaged over many species may appear minor, there may be individual populations of concern. For example, among those species known to be entrained, California grunion (*Leuresthes tenuis*) and some rockfish species are a concern in Southern California because of declining populations (Love et al. 1998). Long-term data show declining trends in the populations of six croaker species, including white croaker, yellowfin croaker (*Umbrina roncador*), black croaker, California corbina, white sea bass (*Atractoscion noblis*), and spotfin croaker (Herbinson et al. 2001). Therefore, results need to be evaluated on a species by species basis, with greater attention focused on species that may be a risk due to population status or other factors.

2.4.5. Advantages and Disadvantages of Fractional Loss Models

This section has focused on the ETM because it is the primary assessment method used in California. An obvious advantage of calculating an entrainment CMR is that it isolates mortality due to entrainment from other sources of mortality. This makes it possible to evaluate an empirically based estimate of conditional mortality due to once-through cooling relative to natural mortality rates.

The modified ETM has some advantages over the original model of Boreman et al. (1978, 1981) because, like the CEMR model, it uses actual intake entrainment data. In addition, the modified model requires less life history data than the original ETM. Whereas the original model incorporates many time-, space-, and age-specific estimates of mortality, as well as information regarding spawning periodicity and larval duration, the only life history information required to estimate PM using the modified ETM is an estimate of larval ages and the time the larvae are susceptible to entrainment (Steinbeck et al. 2007). This advantage is outweighed to some extent because the resulting estimate is based on less biological detail.

Analysts note that the PEs from many surveys may approximate the ratio of the intake volume to the source water volume for some species and will also occur if a species is uniformly distributed in the sample source water. Under these conditions the PM is largely a function of the duration of larval exposure to entrainment. Therefore, if a power plant is operating at full power, the ratio of the cooling water flow to the source water flow will be a constant, and the PE ratio will represent the average long-term entrainment at the plant (Steinbeck et al. 2007).

However, Steinbeck et al. (2007) also observe that, "This advantage of the ETM could be affected if actual cooling water flows varied considerably seasonally and among years." This is an important consideration given that the generating units of most of California's once-through cooling power plants operate as peaking facilities and therefore operate only when energy is in greatest demand (mostly during summer) or when the baseloaded nuclear units are taken out of service for maintenance (State Water Resources Control Board and California Environmental Protection Agency, 2008).

Another limitation of the ETM is the difficulty determining the size and extent of the larval SWA and larval movements within that area, particularly for populations affected by strong tides and ocean currents, such as those in coastal California. Although the variance associated with estimates of PE is used to account for variance in the PM calculation, this does not account for error associated with estimates of PS, the duration of the larval period, and volumes associated with the source water, entrainment sampling, and outflow (MBC and Tenera 2007a).

In a water body like the Hudson River, sampling can be relatively effective at determining the size of the SWA and the boundaries of the population of organisms likely to be entrained. However, in coastal California, where the complex influences of tides and currents, and changes in the direction and rate of flows can have dramatic effects on flow circulation and larval movements, sampling error is a much greater concern. Under these conditions, there can be significant uncertainty in defining the SWA and the duration of larval exposure to entrainment. This is a major concern because these factors have a profound influence on PM estimates (Steinbeck et al. 2007).

Analysts have noted that in California's estuary and ocean waters larval distributions are often "patchy" rather than uniform throughout the water column (Largier 2003). At small spatial scales larvae may tend to be aggregated due to larval swimming behavior and the characteristics of ocean mixing at smaller scales. In addition, oceanographic conditions may vary in ways that affect the movement of patches. For example, tides may move patches back and forth in front of an intake multiple times, resulting in higher rates of entrainment if these movements are not captured by the sampling program. Similarly, when current direction reverses during spring upwelling, patches are carried back in front of intakes passed previously (Largier 2003; Young and Foster 2005).

All of these factors will influence the definition of the SWA and the error associated with any given PE and PM estimate. The error is likely to be substantial unless hydrodynamic processes are adequately captured by the sampling design and the methods used to estimate the SWA.

However, even with the best methods, it will remain particularly challenging and costly to determine the SWA for open populations.

The difficulties in defining SWAs for larvae subject to ocean intakes has been well documented, and the degree of uncertainty involved raises questions about the appropriateness of applying the ETM to these situations. The larger the SWA, the smaller the PM, all else equal, pointing to the risk inherent in incorrectly defining large SWAs for larvae in the open ocean. In these cases the PM may not be a sufficiently sensitive measure of entrainment impact. A lack of apparent impact at the scale of the Southern California Bight should not be taken as evidence that local impacts are unimportant.

2.4.6. Interpretation of PM

It is often stated that ETM results indicate the ecological significance of losses of individuals from entrainment (e.g., Steinbeck et al. 2007; State Water Resources Control Board and California EPA 2008; Ferry-Graham et al. 2008). While it is true that the PM provides an indication of the fraction of the larval population lost to entrainment, additional information is needed to judge the ecological significance of these losses. Losses of larvae affect the current generation only, and without additional information it is not possible to determine if these losses will result a long-term change in the size of the adult population (e.g., Barnthouse 2000; EPRI 2007). While this is important from an ecological point of view, it is also important to recognize that it may not be important from other perspectives, including regulatory requirements, policy goals, and the environmental values of local stakeholders.

Even for the larval population itself, the PM can be difficult to interpret. For example, a high PM from a large larval population may not be a significant concern, while even a low value could indicate a significant impact for a depleted population. One alternative for addressing this issue, at least for fishery species, is to compare the PM to the harvest level set for the species by fisheries managers in order to maintain a sustainable yield (MacCall et al. 1983). For example, the Potrero ETM analysis estimated PMs ranging from 0.3 to 0.5 %, which the analysts noted are negligible relative to the 30-40% harvest levels established to ensure the sustainability of many local fish stocks (Tenera 2005a). The available data indicate that the maximum sustainable fishing rate is somewhat below the natural mortality rate (Quinn and Deriso 1990).

Another alternative is to compare the daily larval entrainment rate (PE) to the natural larval mortality rate. If the entrainment rate is small relative to the natural mortality rate, or small relative to the uncertainty in the natural mortality rate, then entrainment is unlikely to be a concern for the population. However, if the entrainment rate approaches the natural mortality rate, the population may be at risk. If the entrainment rate exceeds the natural mortality rate, the population may not be sustainable.

In recent California studies, an approach known as Habitat Production Foregone (HPF) or Area Production Foregone (APF) has also been used to help interpret PMs. The method is a simple way to estimate the fraction of the SWA corresponding to the fraction of larvae entrained, given as the product of the PM and the SWA. For example, if the PM is 10% and the SWA is 1,000 acres, then the HPF is:

10% x 1,000 acres = 100 acres

i.e., 10% of the larval population entrained can be assumed to come from an area of about 100 acres. An estimate of acres has proven more meaningful for stakeholders than PMs directly (Steinbeck et al. 2007).

While such an approach is attractive because of its simplicity, there are a number of reasons why HPF estimates may not lead to successful restoration. First, the HPF approach does not account for the type of habitat associated with larval recruitment, which may vary for different species, or the current condition of the habitats to be restored. The method simply assumes that if the HPF estimate of acres is applied to the mix of habitats in the vicinity of the power plant, larvae will be produced.

In addition, the HPF uses an estimate of the larval standing stock, based on ETM sampling or other fish surveys, as a "proxy" for the annual production of larvae. This is only reasonable if it can be demonstrated that:

- The area sampled and the density of larvae are the same or similar to the habitat where the larvae are spawned (e.g., coastal wetlands).[1]
- Those larvae sampled include all the larvae produced that year.
- The sampled standing stock is not turned over.

There are many reasons why these conditions may not be met. The sampled larval standing stock may be less than annual production if there is immigration, multiple spawning bouts not covered by the sampling regime, sampling inefficiency, or failure to adequately sample a patchy habitat. Abundance may be greater than production if there is emigration.

Finally, the HPF does not account for the time period over which fish losses occur and the years of increased fish production required to offset those losses (e.g., Thun 2007). These and other issues related to restoration scaling are discussed in detail in the substantial peer-reviewed literature on restoration scaling (e.g., NOAA 1997; Peterson and Kneib 2003; Allen et al. 2005; Thun 2007; Barnthouse and Stahl 2002).

2.5. Cumulative Impact Analysis of Adult Population

2.5.1. Leslie Matrix Population Modeling

There is a long history of using matrix population models to simulate the dynamics of biological populations, including fish populations subject to impingement and entrainment (e.g., Perry et al. 2002). The rows and columns of a matrix population model are the rates of survival and fecundity for each life stage in the population. The change in population abundance during time t to t+1 is calculated by multiplying a population vector giving the abundance in each life stage during the current year, $\mathbf{N}(t)$, by the population matrix, $\mathbf{L}$ (Caswell 1989):

$$\mathbf{N}(t+1) = \mathbf{L}\,\mathbf{N}(t)$$

1 Note that not all fish species will have the same spawning areas or larval habitat requirements.

Newbold and Iovanna (2007) developed a matrix model for evaluating impingement and entrainment that takes into account life stage-specific rates of survival and reproduction as they may be influenced by changes in harvest rates, impingement and entrainment rates, and density-dependent effects. The values for the survival rates of individuals in each life stage and the average weight of individuals in each life stage are obtained from the scientific literature. Reproduction is assumed to occur as a pulse at the end of the year. The value for R, the average number of eggs per pound is set so that the dominant eigenvalue of the matrix equals the intrinsic growth rate of the population, *r*, at low population size.

Once the basic matrix is constructed for a given fish population, additional factors are incorporated to consider life stage-specific rates of impingement and entrainment (IE) mortality, fishing mortality, and density dependence.

The model decomposes natural mortality into mortality due to IE and mortality due to all other sources, so that IE mortality can be considered separately. Thus, total mortality is defined as:

$$Z = M + IE + F$$

where:

Z = total mortality

M = natural mortality from all sources other than I&E

IE = morality due to I&E

F = fishing mortality

By separating impingement and entrainment from other sources of mortality, the effects on population size of IE mortality can be evaluated explicitly. A key advantage of this formulation is that if population size, N, and rates of impingement and entrainment mortality, IE, are unknown, estimates of N and IE can be developed by reference to known values for harvest levels (H), empirical impingement and entrainment loss levels (L), and the intrinsic growth rate of the population (r):

$$H_t = \sum_{k=1}^{K} \left[w_k N_k \left(1 - e^{-Z_k}\right) \frac{F_k}{Z_k} \right]$$

where:

H_t = total harvested biomass in year *t* (from historical landings data)

w_k = the average weight in pounds of individuals in life stage *k*

F_k = instantaneous fishing mortality rate for life stage *k* (from fish stock assessments)

Z_k = total instantaneous mortality rate for life stage *k* (from the literature)

Nk = abundance of life stage k

Once N_k is calculated, life stage specific rates of I&E mortality, IE_k , can be estimated based on the equation:

$$L_{k,t} = N_k \left(1 - e^{-Z_k}\right) \frac{IE_k}{Z_k}$$

where:

Lkt = empirical IE losses for life stage k at time t (from facility monitoring records)

IEk = instantaneous IE mortality rate for life stage k and Nk and Zk are defined as above. Thus, when Lkt , Nk , and Zk are known, the equation can be solved for IEk .

The model also considers the potential effect on population dynamics of density dependence operating in a single life stage, with or without IE or fishing mortality. The per capita rate of survival of individuals in the stage is modeled as a linear function of abundance, as in the logistic model of population growth:

$$N_{t+1,k+1} = \frac{-Z_k}{b - \left(\frac{Z_k}{N_{t,k}} + b\right) e^{Z_k}}$$

where:

b is the carrying capacity of the species (set so that the equilibrium harvest levels predicted by the model under baseline conditions match average historic harvest levels) and the other terms are defined as before. Thus, for one life stage, mortality changes according to N proportional to the factor *b*.

Once the model is calibrated to baseline conditions as outlined above, different impingement and entrainment scenarios can be simulated by changing the IE_k' s. For example, IE′ can be defined as equal to e*k*IE*k* , where e*k* is the effectiveness of a mitigation measure for stage *k*. The complete transition matrix projects the population through time with the new IE_k ’s , taking into account life stage-specific rates of fishing mortality and density-dependent survival in one life stage, until a new equilibrium stock size is reached. These results can then be used to estimate the change in population size resulting from changes in impingement and entrainment under alternative technology scenarios.

Key assumptions of the model are:

- Rates of growth and reproduction are constant.
- Demographic rates are known.
- Population is stable.
- Intrinsic growth rate is known.
- Annual yield of the population is known.
- Density dependence occurs in only one stage.

- Compensation is a linear function of abundance.

An important advantage of the model is that historical landings data and estimates of *r* for particular stocks can be used to calibrate the model. In this way, a lack of information on the average number of eggs laid per pound (R), carrying capacity (*b*), and age-specific rates of I&E mortality (IE_k) can be overcome.

Newbold and Iovanna (2007) used their model to evaluate the population-level effects of eliminating impingement and entrainment. Effects were less than 2.5% for 12 of the 15 fish species evaluated. Effects for the other three species ranged from 22.3% for the Atlantic Coast stock of striped bass to as high as 79.4% for Atlantic croaker, suggesting that effects may be significant for some species (Newbold and Iovanna 2007).

A disadvantage of this calibration approach is that uncertainty in landings data and estimates of *r* will be transferred to variables that are estimated using these data. For example, by calibrating R to *r*, any uncertainties about *r* are propagated throughout the population projection. Likewise *b* is calibrated to landings data, and because mortality changes according to N proportional to the factor *b*, the magnitude of *b*, and uncertainty about the value of *b*, can strongly affect results.

Another important limitation of the model is that historical records of yield and estimates of *r* are only available for exploited species, whereas the majority of impingement and entrainment losses are forage species. In addition, density dependence is a necessary feature of the model (i.e., it cannot be turned on and off). Therefore, it isn't possible to compare results with and without an assumption of density dependence. In addition, density dependence in only a single life stage may not reflect density dependence in real fish populations, in which density dependence could affect multiple life stages.[2]

2.5.2. Age-Structured Stock Assessment

Heimbuch et al. (2007) conducted a cumulative impact analysis of the coastwide effects of impingement and entrainment on Atlantic menhaden (*Brevoortia tyrannus*), which is managed as a single stock from Florida to Maine. The analysis used a standard age-structured stock assessment model with the addition of power plant mortality as a source of mortality during the first year of life (age 0).

Like the Newbold and Iovanna (2007) model, a key feature of this model is that it decomposes total mortality so that impingement and entrainment can be considered as a separate source of mortality. The sum of power plant mortality and natural mortality at age-0 is set to equal the estimated total instantaneous mortality rate for the species of interest. In the menhaden example, the total instantaneous mortality rate was taken from the Atlantic Coast menhaden stock assessment and allocated among four individual age-0 life stages (Heimbuch et al., 2007).

For each annual cohort, the model requires the following input data (Heimbuch et al., 2007):

- Annual age-specific estimates of the number of fish in the population.

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2 To include density dependence it was necessary to make this simplifying assumption so that the model would remain tractable.

- Age-specific estimates of the number of eggs produced per mature female.
- Age-specific estimates of the proportion of females that are mature.
- The duration (in days) of each life stage within the first year of life.
- Estimates of age-specific dry weights for all days during the first year of life.
- Estimates of the number of fish in each age 0 life stage that die from impingement and entrainment.

Most of these data were available from the menhaden stock assessment and the scientific literature. However, there were insufficient data in available 316(b) studies to characterize inter-annual variability in impingement and entrainment. As an alternative, the cumulative impacts analysis was conducted for a range of hypothetical impingement and entrainment levels, including total coastwide annual losses (i.e., the sum over all life stages impinged or entrained for an annual cohort) of 10 million, 100 million, 1 billion, and 10 billion fish (Heimbuch et al., 2007).

For the hypothetical impingement and entrainment loss scenarios, the average estimated reduction in annual recruitment to age 1 ranged from 42,425 fish for the scenario of 10 million impingement and entrainment losses per year to 42,638,000 fish for the scenario of 10 billion losses per year. By comparison, the average number of age 1 recruited over the period evaluated was 4.3 billion fish per year. Expressed in terms of fishery yield, the corresponding reductions in yield ranged from 2 metric tons for the scenario of 10 million fish impinged and entrained to 2,221 metric tons for the scenario of 10 billion fish lost. The average annual yield of menhaden is 273,600 metric tons (Heimbuch et al. 2007).

This model is a reasonable approach for estimating cumulative effects on a coastwide fish stock, but its applicability is limited to cases where there are several years of impingement and entrainment data for each facility included in the analysis or when there is a reasonable basis for determining a range of impingement and entrainment loss scenarios to evaluate, as in Heimbuch et al. (2007). As with matrix models, use of the method will be limited by the availability of the required life history data, most of which are generally available only from stock assessments of fishery species.

2.5.3. Metapopulation Modeling

A metapopulation model may provide another alternative for the assessment of a coastwide population. A metapopulation consists of a number of local populations connected by migrating individuals. Marine and fisheries scientists are increasingly using metapopulation concepts to better understand coastwide fish stocks (e.g., Schtickzelle and Quinn, 2007). Unlike the original metapopulation concept, which emphasized extinction-recolonization dynamics and patch occupancy, these approaches focus on the coupling of spatial scales and the degree of demographic connectivity among subpopulations (Kritzer and Sale, 2004). Such models may be particularly appropriate for species with coastwide distributions and many local subpopulations. A metapopulation model has not been developed for any fish stock subject to impingement and entrainment, but software packages are available that may facilitate the development of such a model (e.g., the RAMAS software for metapopulation modeling; see http://www.ramas.com/ramas.htm#metapop).

2.5.4. Cumulative Impact Analysis of Fish Community

There have been only a few attempts to develop models at the scale of the fish community or local food web to evaluate the potential cumulative effects of impingement and entrainment. Food web (trophic) models examine the transfer of energy (biomass) from primary and secondary producers to higher level consumers, including fish-eating fish and birds. This section provides examples of these types of models.

2.5.5. Patuxent Estuarine Trophic Model (PETS)

The Patuxent Estuarine Trophic Model by Summers (1989) was designed to evaluate the implications of forage fish losses for predator fish in the Patuxent River in Maryland. The model is driven by annual cycles of phytoplankton, zooplankton, and benthic biomass, and includes biomasses of Patuxent River populations of forage fish (naked goby, bay anchovy, silversides) and predatory fish (striped bass, bluefish, and weakfish). The recruitment rate of forage fish into the juvenile stage is modified by entrainment of early life stages.

Two feeding scenarios and three entrainment rates were evaluated. Predators consumed forage fish either 1) in proportion to the densities of the forage fish, or 2) by preference for a particular forage species. Three entrainment rates were evaluated for the two feeding scenarios: 70%, 30%, and 0% reductions in juvenile recruitment of forage species as a result of entrainment (Summers 1989).

The model indicated that the feeding preferences of predator species could be strongly influenced by entrainment of forage fishes depending on the magnitude of entrainment. Results indicated that striped bass, bluefish, and weakfish could experience up to a 25% loss in within-year population production if they preyed preferentially on bay anchovy and silversides, and entrainment of these species was 70% or more of juvenile recruitment. If the diets of the predator species were not dominated by bay anchovy and silversides, the model predicted that the loss would be 5% or less (Summers 1989).

2.5.6. Network Analysis

Ecological network analysis is a method for evaluating direct and indirect trophic interactions among predators and prey. Compartments in the network represent single species or groups of species with similar prey and predators, and compartments are linked by their feeding relationships (e.g., Heymans and Baird 2000).

Ecopath is a Windows-based software package for conducting network analysis that has gained wide acceptance within the ecological community, particularly for fisheries applications. Ecopath facilitates the creation of a static, mass-balanced system of biomass budget equations that connects predators and prey. The software is freely available (http://www.ecopath.org) and is currently the most widely used approach for ecosystem modeling of aquatic systems. Spatial and temporal dynamics can be added to the Ecopath model using Ecosim and Ecospace (Pauly et al. 2000; Christensen and Walters 2004).

Ecopath creates a static "snapshot" of the species in a local food web and their interactions at a given point in time. The food web is represented by trophically linked biomass 'pools' consisting of a single species or a group of ecologically similar species. The model involves two basic equations, one for production and the other for consumption:

Production = catch + predation + net migration + biomass accumulation + other mortality.

Consumption = production + respiration + unassimilated food.

Ecopath data requirements include species-specific biomass estimates, total mortality estimates, consumption estimates, diet compositions, and fishery catches. Such information is generally available from stock assessments of fishery species or local ecological studies and the scientific literature for other species.

There are Ecopath/Ecosim models for some water bodies with cooling water intake structures, and these models could presumably be used to evaluate impingement and entrainment implications for local food webs. For example, the NOAA Chesapeake Bay Office (CBO) has developed a model for Chesapeake Bay that includes 45 functional groups of organisms, representing all trophic levels in the bay. Input data include species' biomass, mortality rates, catch rates, and fishing effort from local fish surveys and ecological studies, supplemented by parameter estimates in the literature (http://noaa.chesapeakebay.net/ecosystemmodel.aspx). Impingement and entrainment could be incorporated into the existing model as an additional source of mortality. This has been done by Lee et al. (2004), who constructed an Ecopath trophic model of a bay in Taiwan to evaluate the potential effects of impingement and entrainment on the bay's aquatic community.

2.6. Summary of Advantages and Disadvantages of Methods Reviewed

Each of the methods described in this review has advantages and disadvantages for use in a cumulative impact analysis and involve different tradeoffs. On the one hand, the least expensive and least uncertain assessment endpoint, individual entrainment losses, has the least ecological relevance.[3] On the other, the method that provides the most ecologically meaningful approach to cumulative impact analysis, food web modeling, is the most data intensive, most costly, and most likely to require assumptions with more uncertainty than the other approaches discussed.

Population modeling is the minimum level of analysis needed to develop an ecologically meaningful context for interpreting entrainment losses. ETM approaches indicate the fraction of the larval population entrained, but as analysts involved with ETM modeling on the Hudson River have noted, the high natural mortality rates of early life stages mean that information on larval losses alone has limited meaning (e.g., Barnthouse 2000). The utility of ETM modeling is also constrained by the ability to define the distributions of the species interest, which will be

3 However, individual loss estimates may be all that is needed to evaluate impacts on rare species or species for which there is already a substantial base of data.

difficult if not impossible for larvae in open coastal systems without a very high degree of uncertainty (e.g., Barnthouse 2000).

Population models can evaluate potential cumulative effects of impingement and entrainment on a single population over time or on a population vulnerable to multiple intakes. The Newbold and Iovanna (2007) and Heimbuch et al. (2007) models provide a way to evaluate the combined effects of fishing mortality and impingement and entrainment on a single population. A matrix approach may also be useful depending on the life history data available for the species of interest or the ability of analysts to make defensible assumptions about life history parameters.

The primary advantage of trophic modeling is that it provides the complete ecological context for interpreting the significance of impingement and entrainment losses, the local food web. The main disadvantage is that food web models require considerably more data than analyses at lower levels of biological organization, including standing stock biomass, feeding rates, and mortality rates for the each species in the food web. The spatial definition of the food web may also present many of the problems noted for determining larval population boundaries for the ETM for open populations in coastal waters. The model itself can be time consuming and costly to develop, although the modeling software is free. In some situations these disadvantages may be offset by the value of understanding the ecological significance of forage fish losses and of having a food web model that can be used to address a variety of policy and management questions in a single region. Once developed, a trophic model can be used to rank risks from many different stressors, in addition to impingement and entrainment, and can be continually refined as more data become available.

3.0 Conclusions

Although the assessment of cumulative impacts is not currently required to meet state or federal 316(b) requirements, there are a number of reasons why cumulative impact analysis could be an important addition to California's 316(b) studies. Impingement and entrainment from once-through cooling are ongoing stressors, involving many intakes and a large portion of the California Coast. Single-facility, single-year assessments may fail to detect impacts at larger spatial and temporal scales. In all cases the affected fish populations are subject to ongoing impingement and entrainment over the operating life of a power plant, in addition to many other stressors such as water quality impairment. As a result, even low levels of annual impingement and entrainment could be a concern when viewed in a broader spatial and temporal context.

This perspective is consistent with the growing interest among state and federal agencies to adopt an ecosystem-scale approach to natural resource management. The integration of impingement and entrainment into a regional framework would be an important advance in the analysis of cooling water intake structure impacts.

The many recent 316(b) studies in California provide considerable data to support more sophisticated assessments. Species-specific estimates of entrainment mortality rates and larval distributions near power plant intakes provide a relatively clear picture of those species and locations that experience the highest loss levels and indicate which species could be the highest priority for cumulative impact analysis depending on permitting needs and the preferences of local stakeholders. Silversides (Atherinopsidae), gobies (Gobiidae), blennies (Blenniidae), sculpins (Cottidae), clingfishes (Gobiesocidae), clinid kelpfishes (Clinidae), and croakers (Sciaenidae) are among those species that consistently dominate entrainment samples.

Studies have also indicated some key characteristics of intakes and facility locations that are associated with the greatest degree of risk. Entrainment rates are significantly higher for plants with cooling water withdrawals above 1 billion gallons per day (BGD) (e.g., Diablo Canyon, San Onofre) and in enclosed areas where both adults and larvae occur and where early life stages are concentrated near intakes (e.g., Moss Landing and Haynes).

Given the data and insights from these studies, it is reasonable for agencies, facility operators, and other stakeholders to consider whether there are situations where a cumulative impact analysis is both feasible and likely to add significantly to current understanding. While local studies will continue to be important because of the site-specific factors that influence vulnerability to impingement and entrainment and the magnitude of impacts, regional-scale management will become increasingly useful in the context of the multiple, sometimes conflicting uses, of coastal resources by diverse stakeholders.

4.0 Recommendations

Given the variety of possible assessment goals and the different tradeoffs involved in selecting among the available methods, results of this review support the use of an iterative approach to the assessment of cumulative effects. An iterative approach would begin with a screening level analysis of individual losses and proceed to the assessment of higher level endpoints depending on thresholds of concern and assessment goals. Ideally, all stakeholders will agree on the "decision criteria" for determining the level of loss that will trigger additional study at any given site and region.

4.1. Iterative Assessment Process

A key consideration in selecting an appropriate method for cumulative impact analysis is the appropriate biological scale of the assessment endpoint given the nature of the impact and the goals of the analysis. Methods that focus on individual, population, and community/ecosystem endpoints provide different kinds of information with varying degrees of ecological relevance (Strange et al. 2002b). These distinctions are important in selecting an approach for any given situation. The following represent key steps in an iterative approach to evaluating cumulative effects.

4.1.1. Screen Existing Impingement and Entrainment Data

- Screen existing impingement and entrainment data to determine the relative magnitude of individual losses among affected species and fractional losses to larval populations.
- Develop thresholds of significance for individual and fractional losses to determine which species require additional analysis based on ecological considerations, assessment goals, and stakeholder interests.
- Consider the spatial and temporal extent of the losses of affected populations to help determine populations at greatest risk.
- Identify special status species or other species of concern to local stakeholders.
- Identify other stressors acting on at-risk populations.
- Identify any additional data collection needs to support cumulative impact studies.

4.1.2. Define Impacts in a Regional Context and Delineate the Assessment Area

- Define geographically and ecologically meaningful regions for an analysis of vulnerable populations.
- Identify other relevant regional criteria (e.g., local jurisdictions, state and federal habitat or species designations).
- Identify regional data (e.g., other stressors, locations of Essential Fish Habitat, water circulation patterns).

4.1.3. Integrate Assessment With Other Regional Planning and Management

- Identify other assessments in regions prioritized for analysis.
- Identify other stressors of concern in study regions.

- Develop collaborative process with all regional stakeholders.

4.1.4. Conduct Monitoring

- Identify "target" species for ongoing monitoring.
- Monitor impingement and entrainment to detect any changes in current rates.
- Track abundances and distributions of target species.
- Adjust management as needed.

4.2. Policy Considerations

There are many ways to define the " significance" of a given magnitude of impingement or entrainment. This review has focused on ecological significance, but other criteria, such as public preferences, economic values, policy goals, and regulatory requirements, may also play a role in impact assessments. In some cases, the best decision, from an ecological, regulatory, and cost perspective, may be to implement a technology without any additional assessment (Van Winkle and Kaladvny 2002). In fact, regulatory requirements may not depend upon a finding of ecological significance. Section 316(b) requires implementation of the "best technology available" to minimize adverse environmental impact, and EPA's view that any impingement and entrainment is an adverse impact has been upheld in federal court (*RiverKeeper*, Inc. v. EPA, 475 F.3d 83, 90, 2nd Cir. 2007).

4.3. Implications of Scientific Uncertainty

A cumulative impact analysis will be subject to many of the same uncertainties as the underlying entrainment estimates, including natural variability in larval densities and rates of growth and mortality, which make ecological impacts difficult to detect (Schmitt and Osenberg, 1996). It is important to recognize that the ability to measure cumulative effects, whether at the population or community level, will be substantially less than the ability to measure impingement and entrainment. For any given situation, therefore, it will be important to balance the additional degree of uncertainty associated with higher-level assessment endpoints against the additional understanding that could be gained from considering potential cumulative effects.

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6.0 Glossary

APF	area of production foregone
BGD	billions of gallons per day
CCA	California Coastal Act
CCC	California Coastal Commission
CEQA	California Environmental Quality Act
CWA	Clean Water Act
EPA	U.S. Environmental Protection Agency
ETM	Empirical Transport Model
HPF	habitat production foregone
IE	impingement and entrainment
MGD	million gallons per day
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
PM	proportional mortality
SWA	source water area, which refers to the source water area of the larval population in the context of the ETM; note that "source water" has a different meaning in the technical context of section 316(b) of the CWA, where it refers to the source water of the intake, i.e., the source of the cooling water.

6.1. Species Scientific Names

Arrow goby (*Clevelandia ios*)
Atlantic menhaden (*Brevoortia tyrannus*)
Bay goby (*Lepidogobius lepidus*)
Black-clawed crabs (*Lophopanopeus* spp.)
Blackeye goby (*Rhinogobiops nicholsii*)
Blind goby (*Typhlogobius californiensis*)
Blue rockfish (*Sebastes mystinus*)
Cabezon (*Scorpaenichthys marmoratus*)
California halibut (*Paralichthys californicus*)
California spiny lobster (*Panulirus interruptus*)
Cheekspot goby (*Ilypnus gilberti*)
CIQ goby complex –
    Arrow goby (*Clevelandia ios*),
    Cheekspot goby (*Ilypnus gilberti*),
    Shadow goby (*Quietula y-cauda*)
Clinid kelpfishes (*Gibbonsia* spp.)
Combtooth blennies (*Hypsoblennius* spp.)
Diamond turbot (*Pleuronichthys guttulatus*)
Garibaldi (*Hypsypops rubicundus*)
Jacksmelt (*Atherinopsis californiensis*)
KGB rockfishes –
    Kelp rockfish (*Sebastes atrovirens*), Gopher rockfish (*S. carnatus*), Black-and-yellow rockfish (*S. chrysomelas*)
Kelp blennies (*Gibbonsia* spp.)
Kelp crabs, *Pugettia* spp.)
Longjaw mudsucker (*Gillichthys mirabilis*)
Market squid (*Loligo opalescens*)
Monkeyface prickleback (*Cebidichthys violaceus*)
Northern anchovy (*Engraulis mordax*)
Northern lampfish (*Stenobrachius leucopsarus*)
Pacific herring (*Clupea pallasii*)
Pacific staghorn sculpin (*Leptocottus armatus*)
Pea crabs (*Pinnixa* spp.)
Queenfish (*Seriphus politus*)
Rock crab (*Cancer* spp.)
Rockfishes (*Sebastes* spp.)
Senorita (*Oxyjulis californica*)
Shadow goby (*Quietula y-cauda*)
Smoothhead sculpin (*Artedius lateralis*)
Snubnose sculpin (*Orthonopias triacis*)
Spotfin croaker (*Roncador stearnsii*)
Spotted turbot (*Pleuronichthys ritteri*)
Striped shore crabs (*Pachygrapsus crassipes*)
Yellow shore crabs (*Hemigrapsus oregonensis*)
Yellowfin goby (*Acanthogobius flavimanus*)
White croaker (*Genyonemus lineatus*)